INDEX

TO THE ADMINISTRATIVE RECORD FOR THE DEPARTMENT OF THE INTERIOR LETTER PROVIDING COMMENTS, RECOMMENDATIONS, AND ATTACHED DECISION DOCUMENT WITH PRELIMINARY PRESCRIPTION FOR FISHWAYS

This is the Index to the Administrative Record for the United State Department of the Interior (Department) letter providing comments, recommendations, and Decision Document with Preliminary Mandatory Conditions. This Administrative Record supports the Department's Preliminary Prescription for Fishways made pursuant to Section 18 of the Federal Power Act, and submitted to the Federal Energy Regulatory Commission (Commission) with the Department's Decision Document for the Green Lake Project (FERC No. 7189), located on Green Lake and Reeds Brook in Hancock County, Maine.

All public records, scientific studies, documents, references, or other information cited, referenced, considered or relied upon in support of the Department's Preliminary Prescription for Fishways and indexed below are contained in the Commission's formal docket for the Green Lake Hydroelectric Project, and may be accessed through the Commission's e-Library records information system at http://www.ferc.gov/docs-filing/elibrary.asp.

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of the

Atlantic States Marine Fisheries Commission

Interstate Fishery Management Plan for American Eel

April 2000

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Interstate Fishery Management Plan for American Eel (*Anguilla rostrata)*

April 2000

Interstate Fishery Management Plan for American Eel (*Anguilla rostrata)*

Prepared by the
American Eel Plan Development Team

And

Approved by the
Atlantic States Marine Fisheries Commission
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This document was prepared in cooperation with the Atlantic States Marine Fisheries Commission's American Eel Management Board, American Eel Technical Committee, American Eel Plan Development Team, and the American Eel Advisory Panel.

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INTERSTATE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL

EXECUTIVE SUMMARY

American eel *(Anguilla rostrata)* occupy a significant and unique niche in the Atlantic coastal reaches and its tributaries. Historically, American eel were very abundant in the East Coast streams, comprising more than 25 percent of the total fish biomass (Smith and Saunders 1955; Ogden 1970). The abundance of this species declined from the historic levels but remained relatively stable until the 1970s. More recently, fishermen, resource managers, and scientists postulated a further decline in abundance from harvest and limited assessment data. This resulted in the establishment of working groups by the Atlantic States Marine Fisheries Commission (ASMFC) to develop a Fishery Management Plan (FMP) for the American eel in order to protect and restore the species. This FMP is a working document that describes the goals and objectives for the species, its current status, ecological challenges affecting the species, and management options and actions needed to reach and maintain the goals. The Plan also identifies issues that need additional research support. A summary of life history, recent abundance indices, and habitat issues is included in the FMP.

GOAL

The goal of this FMP is to conserve and protect the American eel resource to ensure its continued role in the ecosystems while providing the opportunity for its commercial, recreational, scientific, and educational use. Specifically, the goal aims to:

1. Protect and enhance the abundance of American eel in inland and territorial waters of the Atlantic States and jurisdictions and contribute to the viability of the American eel spawning population; and

2. Provide for sustainable commercial, subsistence, and recreational fisheries by preventing overharvest of any eel life stage.

Primary Objectives

- Improve knowledge of eel utilization at all life stages through mandatory reporting of harvest and effort by commercial fishers and dealers, and enhanced recreational fisheries monitoring.

- Increase understanding of factors affecting eel population dynamics and life history through increased research and monitoring.

- Protect and enhance American eel abundance in all watersheds where eel now occur.

- Where practical, restore American eel to those waters where they had historical abundance but may now be absent by providing access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel.

- Investigate the abundance level of eel at the various life stages, necessary to provide adequate forage for natural predators and support ecosystem health and food chain structure.

The American eel occupies and is exploited in fresh, brackish and coastal waters along the Atlantic from the southern tip of Greenland to northeastern South America. The species has a catadromous life cycle, reproducing only in the Sargasso Sea and spending the majority of its life in freshwater. After hatching and ocean drift, initially in the pre-larval stage and then in the leptocephalus phase, metamorphosis occurs. In most areas, glass eel enter the nearshore area, although there have been reports of leptocephalus found in freshwater in Florida (J. Crumpton, Florida Game and Freshwater Fish Commission, Eustis, pers. com.). Glass eel, elvers, yellow and silver eel are found in the marine environment during part of their life cycle. Elvers, yellow and silver eel also make extensive use of freshwater systems. Therefore, a comprehensive eel management plan and comprehensive set of regulations must consider the various unique life stages and the diverse habitats used, in addition to society's interest and use of this resource.

Harvest pressure and habitat losses are listed as the primary causes of any possible historic and recent decline in abundance (Castonguay et al. 1994a and 1994b). Several factors contribute to the risk that heavy harvest may adversely affect American eel populations: (1) American eel mature slowly, requiring 7 to 30+ years to attain sexual maturity; (2) glass eel aggregate seasonally to migrate; (3) yellow eel harvest is a cumulative stress, over multiple years, on the same year class; and (4) all eel mortality is pre-spawning mortality. Habitat losses have been a chronic problem since the arrival of Europeans. Blockage of stream access, pollution, and nearshore habitat destruction limit habitat availability for eel. Castonguay et al. (1994b) indicated that oceanic changes may now also contribute to decline in eel abundance. Busch et al. (1998) estimated that diadromous fish, dependent on access to Atlantic coastal watersheds, may be hindered from reaching up to 84% of upstream habitats.

Planning and regulatory activities require information, specifically, the abundance and status of the species and its habitat. Management is made difficult by the paucity of long-term data sets describing eel abundance at any life stage. Although eel have been continuously harvested, consistent data on harvest are often not available and when available, are not good indicators of abundance because harvest is dependent on demand for eel. Where available, most of the data are of short duration and data collections were not standardized between management agencies. Few other long-term data sets are available from fish ladders, impingement sampling, research collections, and monitoring programs. In addition, changes in year-class strength are not readily recognizable because most samples of fish include fish of similar sizes but from an unknown number of year classes.

A compilation of all available information on eel fisheries and biology suggests that the data are fragmented and/or incomplete. Therefore, the FMP identifies standardized commercial and recreational regulation and surveys and monitoring programs by each state. If harvest rates are determined to have a substantial, negative impact on the American eel population, harvest restrictions will be recommended.

Each state is responsible for implementing management measures and the identification and protection of habitat within its jurisdiction to ensure the sustainability of the American eel population that resides within state boundaries. Since the American eel is one panmictic population, significant management action will have range-wide implications. The FMP suggests new funding and improved coordination, in order to effectively standardize regulations, collection of abundance data at various life stages, and evaluation of habitat and restoration.

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DEFINITIONS

LIST OF ACRONYMS AND ABBREVIATIONS

ASMFC	Atlantic States Marine Fisheries Commission
Board	American Eel Management Board
GLFC	Great Lakes Fishery Commission
FDA	U.S. Food and Drug Administration
NMFS	National Marine Fisheries Service
OMNR	Ontario Ministry of Natural Resources
Plan	American Eel Fishery Management Plan
SAC	American Eel Stock Assessment Committee
Technical Committee	American Eel Technical Committee
USFWS	U.S. Fish and Wildlife Service
MRFSS	Marine Recreational Fishery Statistics Survey

AMERICAN EEL LIFE STAGES

Pre-leptocephalus	Short-lived larval stage from hatching to the free-swimming leptocephalus stage.
Leptocephalus	A long-lived larval stage which is flattened from side to side and shaped somewhat like a willow leaf. This stage drifts and swims in the upper 300 m (1,000 ft.) of the ocean for several months, growing slowly to a length of 5-6.4 cm (2-2.5 in.).
YOY or Young of Year	Young–of-the-year fish less than or equal to 8.5 cm in length, representing a single year class
Glass eel	For the purposes of this Fishery Management Plan, glass eel are metamorphosed leptocephali that are miniature, transparent eel that range in size from 5-10 cm (2-4 in.). Metamorphosis occurs at sea, perhaps near the edge of the continental shelf. Glass eel enter estuaries and ascend rivers during winter and spring, earlier in the southern portion of the range, later in the northern portion. Glass eel ascend estuaries by drifting on flooding tides and holding position near bottom on ebb tides and also by active swimming along shore in the estuaries and above tidal influence.
Elvers	For the purposes of the Fishery Management Plan, "elver" refers to the stage after glass eel. Elvers are pigmented juvenile eel, typically less than 10 cm (4 in.) in length. This life stage may encompass several age classes.

Yellow eel	Immature eel that are dark on the back and often yellowish on the ventral surface and are of variable size that varies by latitude and/or salinity, and also by sex when that is established. They have typically spent more than one year in a stream or estuary and are greater in length than 10 cm (4 in.).
Silver or migratory eel	Following a variable period of growth as a yellow eel, which may increase with latitude, another metamorphosis occurs to form the silver eel or migratory stage. Metamorphosis may include ventral color change to silver, increase in eye diameter, non-feeding behavior and usually a thickening of skin, although this stage can be highly variable. These mature eel move downstream and seaward to spawn in the Sargasso Sea that next winter or early spring (assumed but not documented).

OTHER DEFINITIONS

Catadromous	Spawning and larval development and migration occurring in the open ocean, feeding and growth occurring in estuaries and fresh waters, and adults returning to the ocean.
Dip net	An active capture gear consisting of a rigid frame filled with netting, firmly attached to a rigid handle and manually operated by a single person.
EEZ	Exclusive Economic Zone for the U.S. coastal ocean, extending from 3 to 200 nautical miles offshore
Escape panel or Excluder	Area of mesh in capture gear that allows pre-determined smaller sizes to escape or that prevents larger sizes from entering.
Fyke Net (elver or glass)	A funnel-shaped net designed to intercept moving marine organisms and retain them in a confined space. The net is of various length from cod end to wing tips and is fitted with various size netting. For glass eel the net measures 0.3 cm (1/8 in.) mesh square measure or less.
Hoop Net	A stationary cylindrical net fitted with mesh that is placed at the bottom of a body of water. The gear includes wings or leads attached to the mouth of the net.
Panmictic	Single breeding population exhibiting random mating. Offspring from any parents capable of inhabiting any suitable habitat in any portion of the range.
Pot	A cylindrical or rectangular trap with funnels that is baited. The gear is typically made of mesh.
Sheldon Eel Trap	A box trap with netted wings used to intercept and capture glass eel or elvers.

Spear	The historically most widely known and used method for capturing eel during the early eel fisheries, often consisting of a spatula-shaped center piece with three teeth on each side, each tooth having a single barb. A 3-9 m (10-30 ft.) long wooden pole is attached to this instrument for probing the soft muddy bottom through a hole in the ice or from a boat.
Trap	Passive gear similar to but smaller than weirs. May have one or two wings facing upstream to take descending silver eel. Wings, if present, do not block entire stream and unit is considered portable.
Weir	A trapping device consisting of two wings extending from opposite shores of the stream running obliquely downstream and converging to form a funnel, to which is attached a box trap. As silver eel descend streams, the wings guide them into the box trap. This passive capture gear is semi-permanent, constructed of wood or other solid material, and usually blocks most or the entire channel.

FOREWORD

Charge to Develop a Fishery Management Plan

The Atlantic States Marine Fisheries Commission (ASMFC), at its October 1995 Annual Meeting, voted to initiate the development and implementation of a Interstate Fishery Management Plan (FMP) for American Eel. Due to commercial harvest association with horseshoe crabs, the initial charge was for a joint plan. However, this charge was modified more recently based on biological and ecological differences between the species so that the management of these two species will be addressed in separate plans. The Atlantic coastal states concluded that a coordinated, interstate plan would best address conservation and fishery issues for the American eel. ASMFC is a compact of the fifteen Atlantic Coast states, created to promote the better utilization of the fisheries (marine, shell and anadromous) along the Atlantic seaboard by the development of a joint program for the promotion and protection of such fisheries.

Development of a Public Information Document

A Public Information Document (PID) was prepared to obtain input from the public and interested commercial and recreational users on alternatives and recommendations for state management programs in the development of the American Eel Fishery Management Plan (FMP). The PID briefly discussed the American eel life history and the problems associated with the species' management, status of stocks, current ocean and riverine fisheries, and monitoring and information needs. Public hearings on the PID were held during the spring of 1997.

Purpose of this Fishery Management Plan

The American Eel Fishery Management Plan is a working document that describes the goals and objectives for the species, its current status, recent and historical trends, the ecological challenges affecting the species, management options and actions needed to reach and maintain the goals, and issues that need additional research support. A summary of life history information, recent abundance indices, and habitat issues is included. Species management plans need to be dynamic and are designed to be updated as new data are obtained. This Fishery Management Plan will undergo periodic review to ensure that it reflects any changes in species status, the latest in research and resulting changes in Goals, Objectives and Strategies based on these findings, and changes in human attitudes and needs.

Upon completion and approval of the FMP, ASMFC states are obliged to implement its requirements. In the event that a state does not completely implement an ASMFC fishery management plan, the Atlantic Coastal Fisheries Cooperative Management Act (ACFCMA) provides that the U.S. Secretary of Commerce may impose a moratorium in that state's particular fishery. All ASMFC fishery management plans must include specific measurable standards to improve the status of the stocks and determine compliance with the standards.

A species plan aids in directing management and research efforts. It focuses attention on areas of management strength as well as those that need more development. It provides information to the public on the current knowledge concerning the species, including descriptions of ecological stresses that may limit the abundance and distribution of the species. Overall, a species management plan provides for the regulation of human activities that impact a species so that the population remains sustainable and viable. At the same time, it should allow for recreational and commercial harvest while also supporting the natural diversity of the ecological system(s) it inhabits.

1.0 INTRODUCTION

1.1 BACKGROUND INFORMATION

1.1.1 Statement of the Problem

American eel has a catadromous life cycle, reproducing in the ocean and spending the majority of its life in brackish or freshwater. Any management program must, therefore, involve both marine and inland stakeholders in the management process. Spawning occurs in the Sargasso Sea, producing the larval stage (pre-leptocephalus and leptocephalus) which drifts and swims towards the continental shelf and subsequently metamorphoses into glass eel. Glass eel, elvers, yellow eel, and silver eel are found in the marine environment during part of their life cycle. Elvers, yellow eel, silver eel, and possibly glass eel also make extensive use of freshwater systems. Therefore, a comprehensive eel management plan and comprehensive set of regulations must consider the various unique life stages and the diverse habitats used, in addition to society's interest in and use of this resource.

There is both substantive data and anecdotal information that suggest segments of the American eel population have declined in recent years. The cumulative effects of multiple life stage harvest impact the American eel population. Several factors contribute to the risk that heavy harvest may adversely affect American eel populations: (1) American eel mature slowly, requiring 7 to 30+ years to attain sexual maturity (K. Oliveira, Univ. of Maine pers. comm); (2) glass eel aggregate seasonally to migrate (Haro and Krueger 1988); (3) yellow eel harvest is a cumulative stress, over multiple years, on the same year class (Richkus and Whalen 1999); (4) all eel mortality is pre-spawning mortality (McCleave 1996); (5) changes in year-class abundance are not readily recognizable because harvest abundance data include fish of similar sizes but from a number of year classes (Ritter et. al. 1997). Other factors that may contribute to a possible population decline are structures impeding upstream and downstream passage, increased predation, habitat degradation, poor water quality, and variable oceanic conditions.

American eel have been and continue to be an important resource for biodiversity and human use. The eel and elver fishery in the United States has had a long history (Crawford 1996). The eel has a wide distribution and commercial value throughout its range. The American eel is also a species whose total range includes most of the east coasts of North America, Central America, and northern South America. Significant management action, therefore, has range-wide implications. In addition, the American eel is very important to many Native American tribes, not only as a subsistence food resource, but also for its cultural and spiritual values. The Atlantic States Marine Fisheries Commission's (ASMFC) American Eel Fishery Management Plan for the Atlantic Coast of the US is intended to aid in restoring a healthy and viable American eel population while providing surplus resources for a sustainable eel fishing industry.

1.1.2 Benefits of Implementation

Members of the public have expressed concern over the proper management of American eel to ensure ecological stability. An unregulated American eel fishery and loss of habitats may result in a population collapse with resulting losses to society and to other fish and wildlife resources. Progressive coast-wide management of the American eel population would ensure the long-term viability of the population for continued harvest and would provide necessary quantities of juveniles and adults for use by other fish and wildlife resources. Conservation of the species will provide for biodiversity in natural and existing community food webs (predator-prey interactions).

1.1.2.1 Biological and Environmental

A certain amount of American eel juvenile and adult biomass must be maintained to meet the needs of those species for which eel is an important food source. Despite the range of habitats occupied by the American eel, the importance of eel as prey for other fishes, aquatic mammals, and fish-eating birds has not been well documented. However, American eel juveniles and adults are a seasonal food item of various finfish and data are available that eel are preyed on by fish-eating birds and mammals such as mink (Sinha and Jones 1967; Seymour 1974). The degree of dependence upon the various life stages of American eel by these species is unknown.

1.1.2.2 Socioeconomic

The American eel population has long been important to recreational and commercial fisheries. The fisheries are seasonal, but economically important, providing direct and indirect employment such as gear manufacturing, food processing, and shipping. Landings for American eel fluctuate widely. Much of the commercial fishery is undocumented, but may be of significant economic value (Figure 1). Although relatively few people are engaged full-time in eel fishing, part-time and casual fishermen gain an essential supplementary income. In addition, many coastal multi-species fisheries could not be sustained in the absence of eel (F. Perry 1993/pers. comm.; ASMFC Pub. Hear. Dover, DE 1997).

The significance of American eel to Native American tribes' subsistence and culture is also well established. Tribal communities have documented use of American eel in addition to other fish and game for subsistence. In some cases, seasonal tribal eel harvests have historically provided food fish for up to a year (Speck 1940). In addition, the American eel represents cultural and spiritual values to many Native American tribes by contributing to their sustenance, a focal point of Native American philosophy and lifestyle that goes well beyond the mere value of a resource as food. For example, the passing down from generation to generation of skilled knowledge on basket trap and weir designs and use is a cultural value related to the American eel resource, thus contributing to Native American sustenance (Speck 1940).

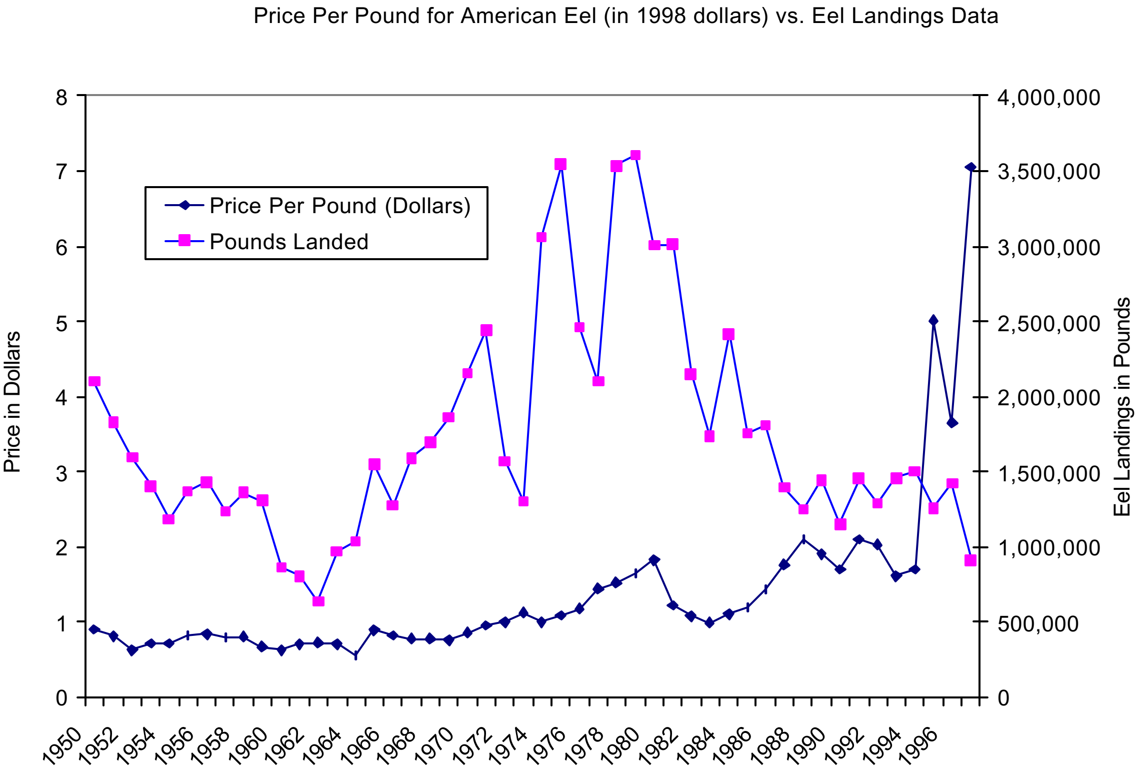


Figure 1. Price per pound for American Eel and the number of American eel pounds landed from 1950 to 1998 (NMFS, Fishery Statistics and Economics Division, 11-15-99, pers. comm.). Note that the last three years reflect the inclusion of reported glass eel landings and associated dollar values.

1.2 Description of the Resource

1.2.1 Species Life History

American eel are a unique and versatile fish species, which are highly migratory with multiple habitat requirements and feeding habits. Eel utilize a large geographic range from the entire east coast of the North and South American continents, into inland areas of the Mississippi and the Great Lakes drainages, and north into Canadian province tributaries. The species is supported throughout its range by a single source, as one spawning population in the Sargasso Sea provides all juvenile eel to be dispersed throughout its entire range each year (Figure 2). Eel have multiple habitat requirements, utilizing open oceans, large coastal tributaries, small freshwater streams, lakes and ponds. They are opportunistic feeders, requiring and utilizing multiple levels of the food chain including phytoplankton, insects, crustaceans, a multitude of fish species, and even larger prey. Individuals live for many years in freshwater and estuarine environments, before returning to the ocean as adults to reproduce once and die.

Despite the fact that in many respects American eel are an adaptable species, a multitude of known pressures on all life stages have a cumulative deleterious effect on the species as a whole. Specifically, the glass, elver, yellow and silver eel life stages are harvested commercially, which reduces their abundance at multiple life stages. This includes the adult reproductive stage since all eel mortality is pre-spawning mortality. The geographic range and habitat availability of American eel has been reduced by obstructions in migratory routes. Freshwater habitat degradation and consequential reduced food productivity levels negatively impact the freshwater life stages. It is possible that contaminants are having a negative impact on the reproductive success of American eel that grew to adulthood in contaminated habitat areas, since eel are known to have a high contaminant bioaccumulation rate (Richkus and Whalen 1999). Oceanographic changes influencing larval drift and migration could impact the overall year-class success (McCleve 1998; Castonguay 1994b), and the fact that the species consists of a single spawning population could make it particularly vulnerable to drastic oceanic variations.

It is, therefore, critical to understand the intricacies of the distinctly different life stages of the American eel. Despite this need, there is little information on any given life stage since there are few species to which the American eel life cycle could be compared, and all of the life stages are distinctly different from each other, with their own difficulties in researching. Specifically, little is known of what occurs in the last phase of the silver eel (mature) life stage; from the time the adult emigrates from freshwater, spawns and dies. The location of the spawning grounds in the Sargasso Sea has been generally identified by the appearance of larvae (leptocephali) in the plankton, but the exact location is unknown. There is also little information on the oceanic egg, leptocephali, and glass eel life stages prior to their arrival in coastal areas.

1.2.2 The Life Cycle

American eel are a catadromous fish species, spending most of their life in freshwater or estuarine environments and migrating back to the ocean to reproduce. The life cycle begins when the eggs hatch and leptocephali are carried by the Gulf Stream from the spawning grounds in the Sargasso Sea, a large portion of the western Atlantic Ocean east of the Bahamas and south of Bermuda. They are consequently dispersed by the prevailing currents along coastal areas, and the glass eel and elvers enter freshwater tributaries. Some elvers travel upstream to spend the majority of their life growing as yellow eel in rivers, streams, ponds and lakes. Mature adults migrate back downstream to return to the Sargasso Sea, where they reproduce in winter and early spring, and then die (Eales 1968; Jessop 1984).

Genetic evidence shows this species to be a panmictic population (Williams 1984) and recruitment levels throughout its range relate to the total number of eel combined from the entire range that survive to successfully reproduce. Potential changes in oceanographic conditions may have an impact on juvenile recruitment to coastal tributaries (Catonguay 1994a&b). American eel in the northern portion of their range mature at greater ages and sizes than in the southern portion, resulting in northern females being the most fecund and having a relatively long life span (Helfman 1987). More recent studies have indicated that the determination of sex may be density dependent (K. Oliviera, U. of Maine pers. comm.).

A potential threat to the overall health of the population is the non-indigenous eel swimbladder nematode (*Anguillicola crassus*). It is a parasite native to marine and freshwater areas of eastern Asia, from Japan and China to Vietnam. Its native host is the Japanese eel (*Anguilla japonica*).

The nematode has been documented to have significant negative impacts on the European eel (*Anguilla anguilla*), and on American eel in Texas and South Carolina.

1.2.3 The Life Stages

1.2.3.1 Egg

American eel spawn in the winter and early spring in the Sargasso Sea, a large portion of the western Atlantic Ocean east of the Bahamas and south of Bermuda and the eggs likely hatch in the same vicinity. Egg diameter is about 1.1 mm, however there is no information on the required environmental conditions or incubation period for the eggs. Artificially spawned Japanese eel (*Anguilla japonica*) eggs were hatched in 38-45 hours at 23 ^{o}C (Facey and Van Den Avyle 1987). American eel fecundity has been reported as a length - weight relationship that can range between 0.5 and 4.0 million eggs per female; large females (1000 mm in length), potentially produce as many as 8.5 million eggs (Facey and Van Den Avyle 1987). The relationship between eel size and fecundity can also be expressed as: $\log F = -4.29514 + 3.74418 \log TL$, $\log F = 3.2290 + 1.1157 \log W$, where F = number of eggs per female, TL = total length (mm), and W = total weight (g) (Wenner and Musick 1974). A fecundity of 0.4-2.6 million eggs was reported in females from Chesapeake Bay ranging from 50-72 cm in length (Wenner and Musick 1974). In the only other study of American Eel fecundity, 63 female eels in Maine were reported to have a fecundity of 1.4 – 21.9 million eggs for eels ranging from 45-113 cm in length (McCleve and Oliveira 1998). It is assumed that the spawning and nursery habitat that is found in the Sargasso Sea is an essential component in the hatching success.

American eel are benthic, long-lived and lipid rich. Therefore, American eel can accumulate high concentrations of contaminants, potentially causing an increased incidence of disease and reproductive impairment as is found in other fish species (Couillard et. al. 1997). An analysis of the contaminants in migrating silver eel in the St. Lawrence River showed that the highest concentrations of chemicals were found in the gonads. Concentrations of PCB and DDT were found to be 17% and 28% higher in the gonads than in the carcasses. The chemical levels in the eggs could exceed the thresholds of toxicity for larvae. Also, since the migrating females are not feeding, the chemical levels in the eggs could be even higher at hatching, increasing the likelihood of toxicity to the larvae (Hodson et.al. 1994).

Pressures/Impacts

- Contaminants may be having a negative impact on the reproductive success of American eel that grew to adulthood in contaminated habitat areas.
- Spawning habitat degradation caused by the harvest of seaweed/algae (*Sargassum sp.)* in the Sargasso Sea, the only known spawning grounds of American eel.

1.2.3.2 Leptocephalus

After hatching and a brief pre-larval stage, the American eel enter a larval leptocephalus stage. The larvae are shaped like a willow leaf, flattened from side to side. Leptocephali drift and swim in the upper 300 m of the water column for several months, growing slowly to a length of 5-6 cm (Kleckner and McCleave 1985). The spatial and temporal distribution of larvae is a result of oceanic circulation patterns and the swimming behavior of the larvae (Figure 2). At sea, perhaps at the edge of the continental shelf, the shape of the larvae dramatically metamorphoses into miniature transparent eel, termed glass eel.

Potential changes in oceanographic conditions may have an impact on juvenile recruitment to coastal tributaries. Catonguay (1994a) suggests two hypotheses for investigation: 1) a weak, slow Gulf Stream would cause larvae to miss the optimum period for metamorphosis and to be lost to the population when they reach the position of the stream where lateral transport would have ordinarily placed them and, 2) recent cooling events and oceanographic changes in the northwest Atlantic may have perturbed the physical processes that carry glass eel to the continent. Castonguay (1994b) also explores the indirect evidence of a weakening Gulf Stream and ways in which it may interfere with larval transport of American eel, as well as changes in the strength or location of thermal oceanfronts.

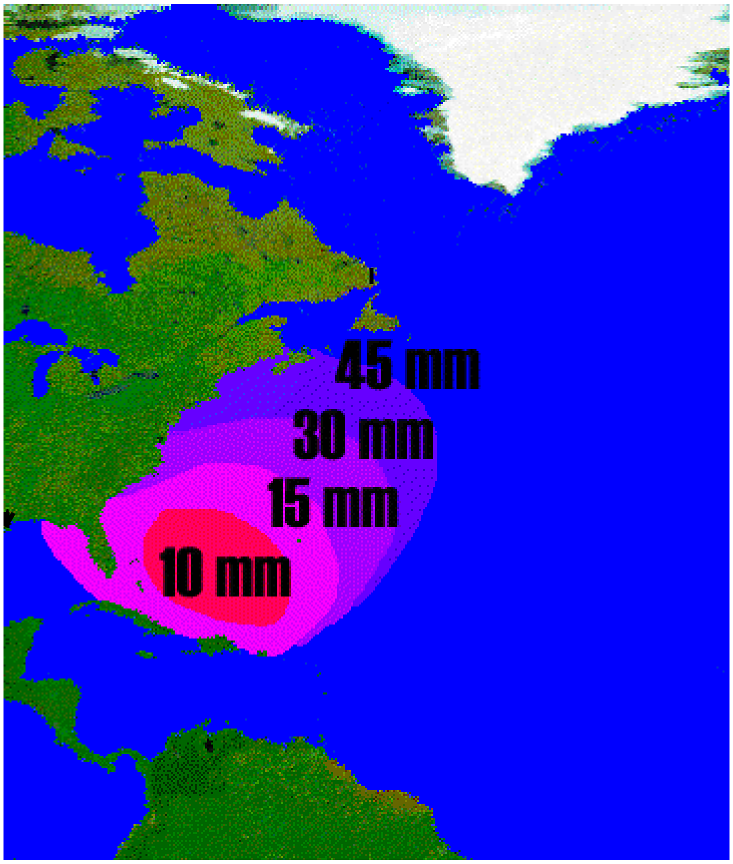
Pressures/Impacts

- Potential / exploratory harvest of leptocephali.
- Changes in oceanographic conditions, a weakening Gulf Stream and recent cooling events in the northwest Atlantic may potentially have an impact on juvenile recruitment to coastal tributaries.

1.2.3.3 Glass eel

The glass eel life stage occurs when the leptocephali metamorphose at sea to resemble miniature, transparent eel. They are transparent with elongated, rounded bodies and range in length from 4.8 to 6.5 cm (Hardy 1978). They actively migrate toward land and freshwater and ascend rivers during the winter and spring. It has been demonstrated, in European glass eel, that this change in behavior was caused by the detection of the odor of freshwater, as well as temperature gradients (Facey and Van Den Avyle 1987). This migration occurs earlier in the southern portion of the range and later in the northern portion (Helfman et al. 1984; McCleave and Kleckner 1982). Glass eel ascend estuaries by drifting on flooding tides and holding position near bottom on ebb tides and also by actively swimming along shore in the estuaries and

Figure 2. American eel leptocehpali spatial and temporal distribution by size. Source: Uwe Kils, Rutgers U.



above tidal influence (Barbin et al 1994). Glass eel in estuaries and those ascending into freshwater eventually become pigmented elvers.

Pressures/Impacts

- Since artificial reproduction is not yet feasible, the intensive aquaculture industry in eastern Asia (150,000 t production) is dependent upon and supported by wild-caught glass eel and elvers (Moriarty and Dekker 1997).
- Glass eel commercial fisheries are scattered throughout the American eel's range. A limited import trade in glass eel from Europe to the United States exists for the food industry. Glass eel harvest in recent years has given rise to serious concern as to the future viability of the eel industry.
- Lack of up and downstream passage for migrating glass eel.

1.2.3.4 Elver

The elver life stage occurs when the glass eel ascend into brackish or fresh water and become pigmented, generally at 10.0 cm or less in length. At this early stage, they are active at night and burrow during the day. They move into the water column on flood tides and return to the bottom during ebb tides (McCleave and Kleckner 1982). Elvers have been shown to be attracted to the odor of brook water and decaying leaf detritus and microorganisms (Facey and Van Den Avyle 1987). Upstream migration of elvers can occur over a broad period of time from May (during peak migration) through October (Richkus and Whalen 1999). The migration occurs earlier in the southern portion of its range and later in the northern portion (Helfman et al. 1984; McCleave and Kleckner 1982).

Elvers are brown in color and are usually fully pigmented at 6.5 mm to 9.0 cm in length (Hardy 1978), although pigmented American eel have been observed less than 6.5cm in Florida (J. Crumpton, Florida Game and Fresh Water Fish Commission, Eustis pers. comm.). They eventually begin swimming upstream possibly due to changes in water chemistry and river current velocities (Facey and Van Den Avyle 1987). They grow slowly, reaching about 12.7 cm after the first year in freshwater (Bigelow and Schroeder 1953). Growth rates are highly variable, leading to considerable variation in length within age groups and poor predictability of size at age (Facey and Van Den Avyle 1987).

Pressures/Impacts

- Since artificial reproduction using mature eel is not yet feasible, the intensive aquaculture industry in eastern Asia (150,000 t production) is dependent upon and supported by wild-caught glass eel and elvers (Moriarty and Dekker 1997).
- Elver commercial fisheries are scattered throughout the eel's range in both the marine and freshwater habitat areas. Elver harvest in recent years has given rise to serious concern as to the future viability of the eel industry. The elver fishery in the United States has had a long history with wide distribution and commercial value throughout its range.
- Lack of adequate up and downstream passage for migrating elvers.

1.2.3.5 Yellow Eel

The yellow eel resembles the adult form and occurs after the elver stage. Yellow eel are usually yellow or green in color and range in size up to about 28.0 cm for males and 46.0 cm for females (Hardy 1978). They inhabit bays, estuaries, rivers, streams, lakes, and ponds where they feed primarily on invertebrates and smaller fishes (Ogden 1970). Usually by Age II, the eel have entered into the yellow phase. Depending on where they cease their upstream migration, some yellow eel reach the extreme upper portions of the rivers while others stay behind in the brackish areas (Hardy 1978, Fahay 1978). The timing and duration of yellow eel upstream migration is watershed specific and can occur over a broad period of time from March through October, peaking in May through July. Yellow eel can continue migrating until they reach sexual maturity (Richkus and Whalen 1999). In the upper St. Lawrence River, yellow eel migration is monitored between June and October, and 72.2% of the upstream migration occurs between July 18 and August 17 (Casselman et al. 1997). The growth rates of yellow eel are variable, depending on latitudinal trends (slower growth occurs in the north than in the south) and habitat productivity (slower growth occurs in freshwater than in estuaries) (Richkus and Whalen 1999).

Timing of sexual maturity in the yellow eel has been correlated with specific size ranges. Most sexually mature males are over 28.0 cm and, in the northern populations, they are older than Age 3 (Hardy 1978, Fahay 1978). Most sexually mature females are over 46.0 cm and they are older than Age 4 in the northern populations (Hardy 1978, Fahay 1978). Length-age relationships vary considerably within the northern portion of their range. The following year-class size information has been reported for Rhode Island: Age 4 total length (TL) 27-46 cm; Age 5 - TL 28-51 cm; Age 6 -TL 28-51 cm; Age 7 - TL 29-58 cm; Age 8 - TL 33-64 cm; Age 9 - TL 38-62 cm; Age 10 -37-65 cm; Age 11 - TL 46-65 cm (Bieder 1971).

There are several environmental variables that can influence sexual determination in American eel, the resulting ratios of females and males, and age at sexual maturity. In the northern portion of their range eel mature at greater ages and sizes than in the southern portion, resulting in northern females being the most fecund and having a relatively long life span (Helfman 1987). For example, numerous studies have found the St. Lawrence River-Lake Ontario eel to be exclusively female (Dutil 1987; Vladykov 1966). J. Casselman (OMNR pers. com.) also found them to be relatively older and larger. McCleave (1996) found that females are more abundant in the northern part of their range, males are more abundant in the southern part of their range, and that females grow larger and mature later than males. However, Foster and Brady (1982) found only females in Maryland where sex could be determined (N=1,000); Helfman et al. (1984) found in a Georgia river that 64% of estuarine eel were female and 94% of freshwater eel were female; Hansen and Eversole (1984) noted that females outnumbered males 23 to 1 in South Carolina. Some data suggest that there is a further isolation of the sexes by salinity. Females were found to be more prevalent in freshwater systems while males more frequently inhabit estuaries (Facey and LaBar 1981). Recent work indicates that sex determination might be influenced by density (K. Olivera, U. of Maine pers. com.). If this is the case, sex ratios may be changing towards more females throughout their range due to lower numbers of eel.

Maturation occurs in 8 to 24 years in the Chesapeake Bay Region, but may occur earlier in southern regions and later in northern regions. In the southern regions, females older than eight years old or longer than about 70 cm were rare and males older than five years old or longer than 40 cm were also rare. In contrast, maturing females in the Newfoundland study averaged 13 years of age and more than 70 cm long (Bouillon and Haedrich 1985). Female eel from Lake Champlain averaged 16 years old and nearly 70 cm long (Facey and LaBar 1981). Eel greater than age 20 were found in Lake Champlain. Males were not present, or were not captured, in the two northern studies. There is evidence that males are rarer at higher latitudes and in inland waters (Helfman et al. 1987). The size and distributional differences between the sexes led Helfman et al. (1987) to hypothesize that male and female American eel experience different natural selection pressures which result in different life history traits. They suggested that males tend to be found in the more productive habitats, closer to the spawning area, favoring rapid growth and maturity at a small size. This is a time-constrained life history strategy. Females are distributed over all suitable habitats dispersed widely through the geographic range, and slower growth to greater size and age is favored. Increased size results in increased fecundity. This is an energy-constrained life history. The evolutionary scenario hypothesized by Helfman et al. (1987) requires further research, but may be a critical concept in managing the species in different parts of the geographic range and in different habitats.

Pressures/ Impacts

- Yellow eel spend a lengthy period of time before reaching sexual maturity, are harvested throughout that period, and are susceptible to overharvest.
- Lack of adequate up and downstream passage for migrating juveniles.

1.2.3.6 Silver Eel

The silver eel life stage, which is the migrating and sexually mature eel, begins after a lengthy period as a yellow eel. Between the time of beginning the downstream migration and leaving the estuary for the open ocean, the yellow eel metamorphose into the adult silver eel phase, which is better suited for ocean migration (Wenner 1973, Facey and Van Den Avyle 1987). Silver eel may begin their seaward spawning migration in late summer through fall from New England tributaries (Facey and Van Den Avyle 1987). The yellow eel undergoes several physiological changes in becoming a silver eel, including: (1) a color change from yellow/green to metallic, bronze-black sheen; (2) body fattening; (3) skin thickening; (4) enlargement of the eye and change in visual pigment; (5) increased length of capillaries in the rete of the swim bladder; and (6) digestive tract degeneration (Facey and Van Den Avyle 1987). These changes have not been observed often or at all in specific state waters and are capable of variying with latitude and temperature (J. Crumpton, Florida Game and Fresh Water Fish Commission, Eustis pers. comm.). Migrating silver eel have been observed to cover 38 km in 40 hours, showing considerable vertical movements in the water column with no behavioral changes associated with diel or tidal cycles (Stasko and Rommel 1977). Little is known about the oceanic spawning migration and the means by which the spawning grounds are located are poorly understood (Miles 1968). It has been suggested that American eel use the geoelectrical fields generated by ocean currents for orientation (Rommel and Stasko 1973). The depth at which American eel migrate in the ocean has been hypothesized to vary with light intensity and turbidity (Edel 1976). Migration has been suggested to occur within the upper few hundred meters of the water column (Kleckner et al. 1983; McCleave and Kleckner 1985). However, Robins et al. (1979) photographed two *Anguilla* eel, believed to be pre-spawn American eel, at depths of about 2,000 m (on the floor of the Atlantic Ocean) in the Bahamas.

No information exists on the spawning requirements, behavior, or the exact location of spawning within the Sargasso Sea. Adult eel are believed to spawn in the winter and early spring in the

Sargasso Sea, which is a large portion of the western Atlantic Ocean east of the Bahamas and south of Bermuda. Genetic studies indicate that American eel are a single panmictic breeding population (Williams and Koehn 1984). At this time only a few published studies of fecundity of the American eel exists where the relationship between eel size and fecundity was expressed as: $\log F = -4.29514 + 3.74418 \log TL$, $\log F = 3.2290 + 1.1157 \log W$, where F = number of eggs per female, TL = total length (mm), and W = total weight (g). A fecundity of 0.4-2.6 million eggs was reported in females from Chesapeake Bay ranging from 50-72 cm in length (Wenner and Musick 1974) while Barbin and McCleave (1997) reported a range of 1.8 to 19.9 million eggs.

Pressures/Impacts

- Commercial fisheries throughout the silver eel range in freshwater and estuarine habitat areas.
- Mortality caused by hydropower turbines during the downstream migration of adults.
- Harvest of the seaweed/algae (*Sargassum sp.*) in the Sargasso Sea and potential capture of silver eel prior to reproduction.

1.2.4 Food Habits

American eel depend on a wide range of food at different life stages and in different habitats. At various times and locations they feed on every level of the food chain.

Eel are carnivores and consume a variety of foods including demersal fishes and benthic invertebrates such as insects, crayfish, snails, and worms (Ogden 1970; Scott and Crossman 1973; Facey and LaBar 1981). Benthic organisms such as crayfish, various gastropods, and demersal fish are significantly more common in shallow littoral and stream habitats than in deep, cold water habitats. Godfrey (1957) concluded that about 10% of the eel examined had consumed whole fish, while 90% contained mostly insects. Facey and LaBar (1981) suggest that eel rely heavily on benthic organisms as evidenced by 43% of eel stomachs containing insects. Fish were found in 26% of the stomachs. Overall, smaller eel (43-57 cm) rely more on insects than larger eel (57 cm). In eight New Jersey streams, food size was also found to increase with eel size. Smaller eel fed on mayflies, megalopterans, and caddisflies (Smith 1985). Fish comprised at least 25% of the diet for approximately 20% of eel in New Jersey streams; bottom dwelling and sluggish species were most prevalent (Ogden 1970). Facey and LaBar (1981) indicated that the higher percent of fish in the diet of eel in Lake Champlain might have been due to the larger size of the eel in their samples (approximately 61-cm).

American eel leptocephali feeding habits have not been reported. However, the dentition and gape of the mouth suggest that they are capable of feeding on individual zooplankton and phytoplankton. Elvers collected from Cooper River, South Carolina, ate mostly larval and adult chironomids, cladocerans, amphipods, and fish parts (McCord 1977). More types of food were eaten by intermediate-sized yellow eel than by elvers or maturing yellow eel (Wenner and Musick 1975). Fish occur in the diet of intermediate-sized yellow eel during the winter and spring, while insects and mollusks were eaten from spring through fall (Wenner and Musick 1975). Yellow eel shorter than 40 cm in New Jersey streams mainly ate aquatic insects, whereas larger eel fed mostly on fish and crustaceans (Ogden 1970). Yellow eel in the lower Chesapeake Bay fed on crustaceans including blue crab (*Callinectes sapidus*), bivalves such as soft-shelled clams (*Mya arenaria*) and polychaetes (Wenner and Musick 1975). Eel have been considered to be significant predators on young salmonids, but this is not well supported in the literature (Facey and Van Den Avyle 1987; Godfrey 1957). Bigelow and Schroeder (1953) describe the American eel as feeding on whatever prey/food items happen to be found in its habitat. Given their poor eyesight and nocturnal feeding habits, yellow eel probably rely on their keen sense of smell to locate food (Fahay 1978). A diel foraging study in the Pettaquamscutt River estuary of Rhode Island showed that the foraging activity of estuarine eel was primarily nocturnal in late summer through autumn. The study also identified a peak of activity at nightfall, with most of their captures in traps occurring one hour after sunset (Sorensen et al. 1986). Yellow eel swallow some types of prey whole, but also can tear pieces from large dead fish, crabs and other items (Facey and Van Den Avyle 1987). Eel have been reported to accomplish this tearing off by biting and spinning rapidly (Helfman and Clark 1986).

1.2.5 Stock Assessment Summary

Historical Overview

The American eel has been an important food for native Americans since the pre-colonial era (Crawford 1996). Because eel are also present in European waters, this resource was well known to, and used by, the earliest European settlers to the North American continent. The first systematic records of eel harvests in Maine were collected in 1887 and harvests have been recorded more or less continuously since 1989. Atkins (1887) reported on the early Maine eel fisheries as follows: "Eel are taken with spears, in traps and pots set for the most part in tidal waters, and in weirs built across the streams that they descend in the autumn." Throughout the first half of the 20th century, the eel fishery was small (Crawford 1996).

European eel species and Asian eel species fisheries had declined by the late 1960s and their markets were in need of an external source (Crawford 1996). American eel that were exported from southern New England filled that need. The American commercial fishery has traditionally supplied American eel for the regional and the European food market, domestic trotline bait, and small bait eel for domestic sport fisheries. Glass eel and elvers are cultured to marketable size in Asia. When the Asian domestic stocks are inadequate, a strong market develops for American glass eel and elvers. The Asian market for American glass eel and elvers was strong from 1972-1977, declined dramatically in 1978, and began to strengthen in the 1990's.

Current Status

The current status of the American eel stock is poorly understood. This is due to limited and non-uniform stock assessment efforts and protocols across the range of this species. Reliable indices of abundance of this species are scarce. Limited data from indirect measurements (harvest by various gear types and locations) and localized direct stock assessment information are currently collected.

Although eel have been continuously harvested, consistent data on harvest are often not available. Harvest data is often a poor indicator of abundance, because harvest is dependent on demand and may consist of annually changing mixes of year classes. Most of the data collections were of short duration and were not standardized between management agencies. Harvest data from the Atlantic coastal states (Maine to Florida), indicate that the harvest has declined after a peak in the mid-1970s (Figure 3). Annual eel catch ranged from 885,267 lbs. to

3,608,357 lbs. between 1970 and 1998, but the catches averaged 2,540,599 lbs. between 1970 and 1984, and 1,356,434 lbs. between 1985 and 1998. The lowest harvest (between 1970 and 1998) was 885,267 lbs., which occurred in 1998. Because fishing effort data is unavailable, however, finding a correlation between population numbers and landings data is problematic.

In addition to commercial harvest, there are a few long-term data sets from fish ladders, impingement sampling, research collections, and monitoring programs. In 1974, Ontario Hydro and OMNR constructed the largest eel ladder in the world at the Moses-Saunders Hydroelectric Dam (Eckersley 1981; OMNR 1986). Eel count data from the ladder indicate there has been a significant and dramatic decrease in the number of eel ascending the ladder since the mid-1980's (Figure 4) (Casselman et al. 1997). However, this decline in eel counts may be an artifact of lock/water flow usage at the Beauharnois Dam which is downstream from the Moses-Saunders facility. Long-term data from the Conowingo Dam fish lift in Maryland, on the Susquehanna River, show a decline in elver counts from 1974 through 1996 (Figure 5).

Richkus and Whalen (1999) performed a trend analysis on eel migration data from 1984 to 1995, including data from the Moses-Saunders eel ladder (Table 1). Their results indicate significant negative trends for yellow and/or silver eel abundance in Ontario, Quebec, New York, and Virginia, although silver eel declines in the St. Lawrence River basin may be due to escapement reductions from upper St. Lawrence dams and water flow control rather than fisheries. The authors found no trends for glass eel or elvers, but those data sets were generally not complete and may not have covered the years where the largest declines were observed in other data sets (Richkus and Whalen 1999).

Richkus and Whalen's (1999) results support observations and concerns made by the state and federal fishery resource agencies, conservation organizations, and fisheries interests that the eel resource has been declining in abundance. As stated in the Goals of this Plan (Section 2.1) the purpose of this management effort is to reverse any local or regional declines in abundance and institute consistent fishery-independent and dependent monitoring programs throughout the management unit.

Recent Changes in Harvest

Domestic and overseas markets utilize American eel from most life stages. Most harvest data show a decreased recruitment and catch of glass, yellow and silver eel. Data on European eel also show a considerable decline in abundance since the late 1970's (Moriarty and Dekker 1997).

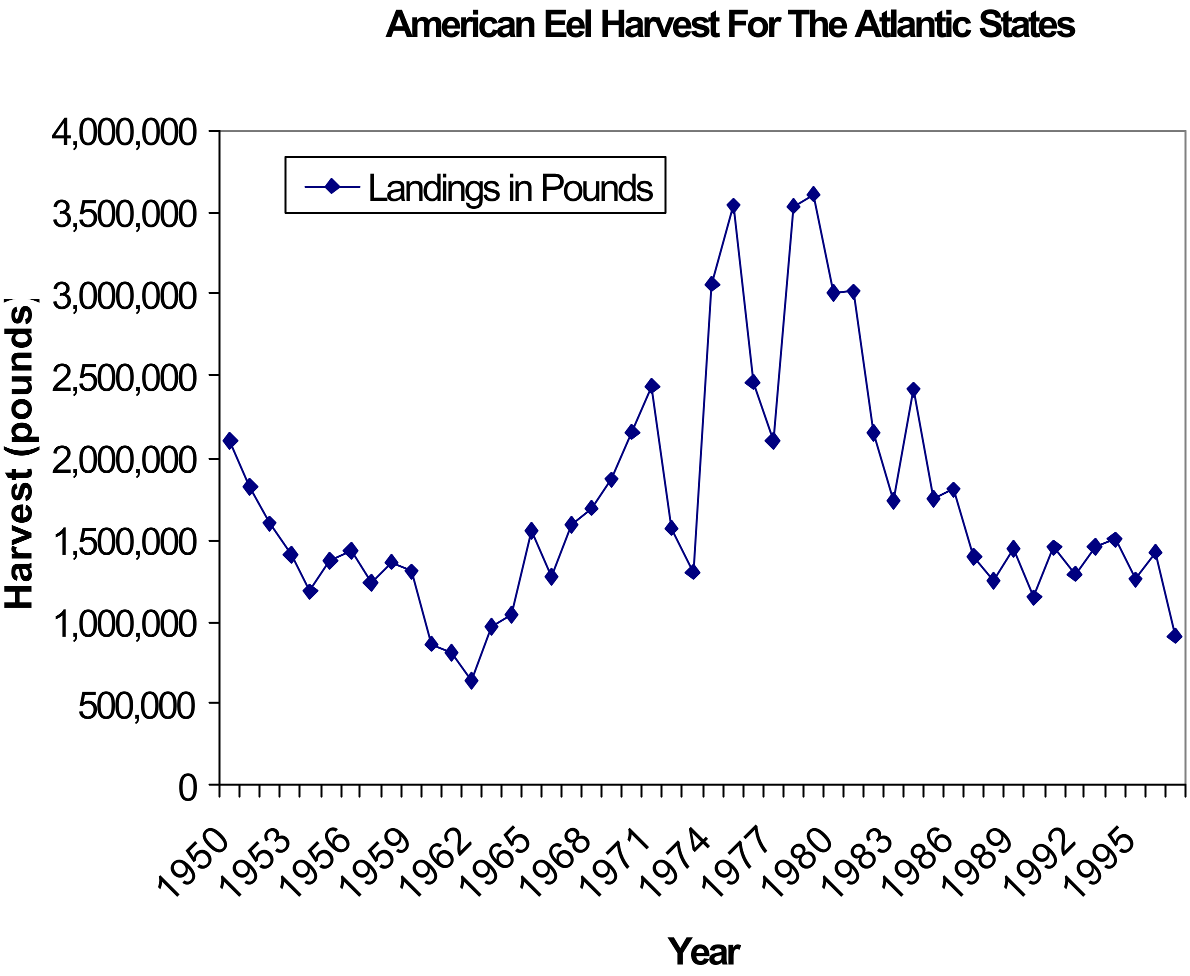


Figure 3. Annual harvest as reported by the Atlantic States from 1950 to 1998 (NMFS, Fishery Statistics and Economics Division, 11-15-99, pers. comm.).

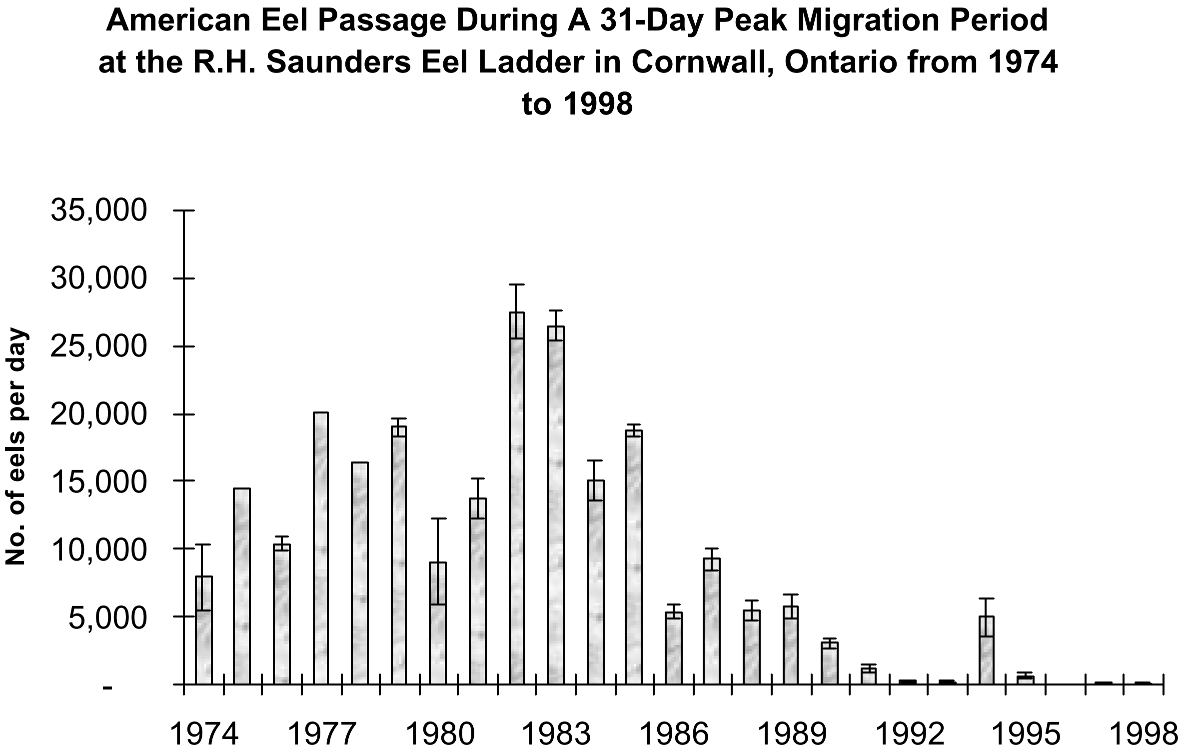


Figure 4. **Mean number of eel ascending the eel ladder per day at the Moses-Saunders Hydroelectric Dam at Cornwall, Ontario, during a 31-d peak migration period from 1974-98. Vertical bars indicate the 95% confidence intervals (from Casselman et al. 1997, Mathers et al 1998).**

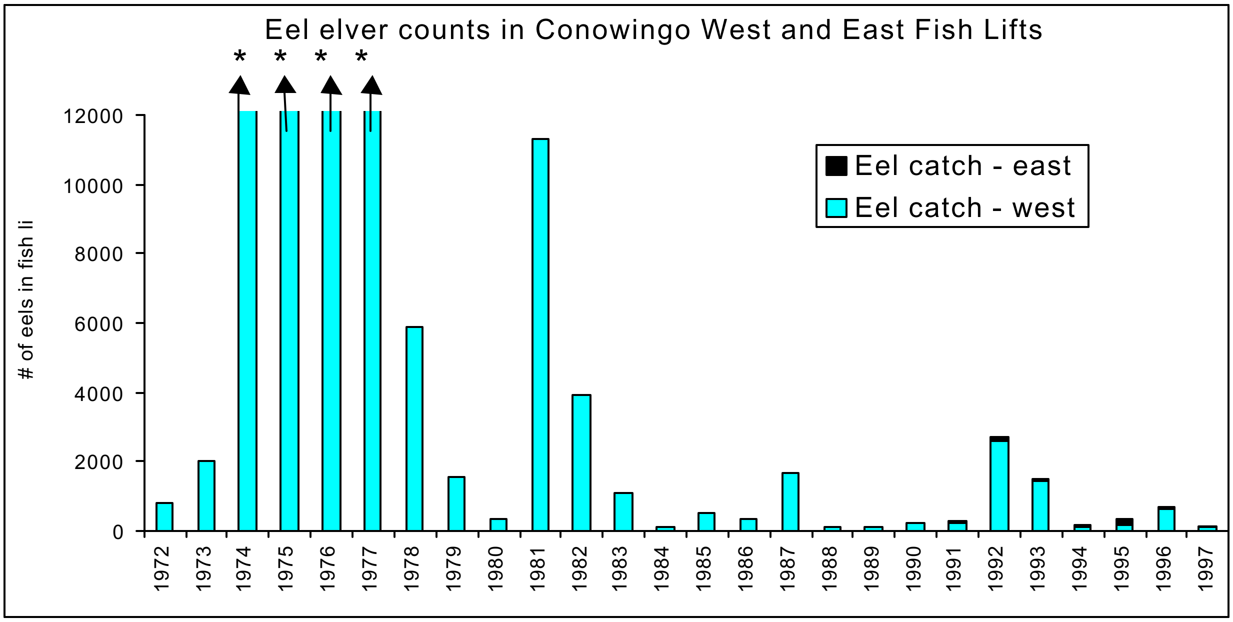


Figure 5. Data from Conowingo Dam fish lift, Susquehanna River, 1972-1997.

***Counts of eel in fish lifts for 1974, 1975 and 1976 were 126,543, 64,375, and 60,409 respectively (J. Weeder, MD DNR person. comm.). * Counts of fish per operating hour for 1974, 1975, 1976, 1977, and 1981 were 183.87, 209.69, 161.09, 35.35, and 41.20 respectively (J. Weeder, MD DNR person. comm.).**

Table 1. Summary of data sources used in Mann-Kendall trend analysis of eel abundance time series. Significance was determined at $\alpha = 0.05$; NS = not significant. Table is arranged approximately north to south. (Richkus and Whalen 1999)

State/ Province	Location	Available Years	Collection Method	Eel Life Stage	Mann-Kendall Trend Analysis (1984-95)
Nova Scotia	East River, Sheet Harbor	1990-97	Irish elver trap	Elver	NS
Ontario	St. Lawrence River	1974-95	Fish Ladder	Yellow eel	Negative $P < 0.001$
Ontario	Lake Ontario	1984-96	Commercial electrofishing	Yellow eel	Negative $P < 0.01$
Quebec	St. Lawrence River (lower)	1979-95	Weir trapping	Silver / Yellow eel	Negative $P < 0.01$
New Hampshire	Statewide	1988, 1990-97	Commercial eel pot	Yellow eel	NS
New York	Hudson River	1985-1995	Beach Seine Survey	Yellow eel	Negative $P < 0.1$
New York	Hudson River	1985-1995	Fall shoal survey	Yellow eel	NS
New York	Hudson River, Roseton	1973-96	Impingement sampling	Silver / Yellow eel	NS
New York	Hudson River, Danskammer	1974-96	Impingement sampling	Silver / Yellow eel	Negative $P < 0.001$
New Jersey	Little Sheepshead Creek	1989-94	Bridge netting	Glass eel	NS
PRFC	Potomac River	1988-97	Commercial eel pot	Yellow eel	NS
Virginia	North Anna River	1981-97	Electrofishing/ electroseining	Yellow eel	Negative $P < 0.01$
Virginia	VIMS trawl survey; rivers and estuaries	1954-96	Trawl sampling	< 180 mm (elvers/glass eels)	NS
Virginia	VIMS trawl survey; rivers and estuaries	1954-96	Trawl sampling	181 – 350 mm	NS
Virginia	VIMS trawl survey; rivers and estuaries	1954-96	Trawl sampling	< 350 mm (silver eel)	Negative $P < 0.05$
Virginia	VIMS trawl survey; rivers and estuaries	1954-96	Trawl sampling	All ages combined	NS

From the above data, it is apparent that overall eel harvest has declined. In addition, eel abundance in upstream migration has declined in the St. Lawrence (Casselman et al. 1997) and Susquehanna River Systems. Richkus and Whalen (1999) concluded that the trend analysis shows broad-based evidence for a stock-wide abundance decline of American eel from 1984 to 1995.

1.3 DESCRIPTION OF THE FISHERY

1.3.1 Commercial Fishery

Jessop (1997) provides a brief but highly concise summary of the status of the American eel fishery along the Atlantic seaboard of the United States. It is presented below with a few updates concerning Pennsylvania and New Jersey.

Glass eel/Elver Fishery

Interest in fishing for American elvers and glass eels, primarily for export to Asia for aquaculture, developed in **Florida**, **North** and **South Carolina**, **Virginia, Massachusetts** and **Maine** during the early 1970s (Fahay 1978; Keefe 1982; Mullis 1982). Elver/glass eel fisheries failed to develop in Florida, ceased in 1977 in North Carolina and probably also in South Carolina, and were prohibited in 1977 by a 15 cm minimum size limit in Virginia and a 10 cm minimum size limit in Massachusetts (CBP 1991). The **Potomac River Fisheries Commission** imposed a 6-inch minimum size effective January 1, 1992, applying to both commercial and recreational fisheries, therefore eliminating any glass eel/elver fishery. Reported catches in Maine were 10 t in 1977 and 7.6 t in 1978 (Dow 1982) but catch statistics are unavailable for the other states. The Maine elver/glass eel fishery collapsed after 1978 due to market conditions, but continued at a low level until growing substantially in 1994. Reported catches of 3.3 t occurred in 1994, 7.5 t in 1995, and 4.6 t in 1996 (CAEMM 1996; L. Flagg, Maine Department of Marine Resources, pers.comm.). With the exception of 1977 and 1978, elver/glass eel catches in Maine cannot be separated from yellow/silver eel catches prior to 1994 when specific records of elver/glass eel catches were initiated. During the late 1980s or early 1990s, elver/glass eel fisheries were developed or reestablished in **Connecticut**, **Rhode Island**, **New York**, **New Jersey**, **Delaware** and **South Carolina** but no catch data are available. Elver/glass eel fisheries do not occur in any **Gulf of Mexico** states.

The recent surge of interest in fishing for elvers/glass eels and the sometimes-chaotic nature of the fishery has evidently caught state fishery managers unprepared. Few states, which presently permit elver/glass eel fisheries (Maine, Connecticut, South Carolina, and Florida), have comprehensive regulations for those fisheries. Although 11of 15 Atlantic coastal states presently ban elver/glass eel fisheries, several states prohibited the elver/glass eel fishery only recently in response to a perception of uncontrolled development. Permits to fish elvers/glass eels may specify various conditions, such as quota, area to fish (all are restricted to tidal waters), gear types, season, etc.

Maine leads other elver/glass eel fishery states in modernizing its elver/eel fishery regulations. It has recently proposed and/or implemented regulatory changes to increase elver/large eel license fees to $200.00 in an effort to dedicate license fee revenues to eel research and provide enforcement for the fishery. Maine has imposed a March 15-June 15 fishing season and two day weekly closed time for elvers/glass eels (defined as eel less than 15 cm long). It will also limit the number, type, and methods of operation of gear units available to each fisher in an attempt to control fishing effort, limit elver/glass eel fishing to the intertidal area and the shoreward one-third of a stream (both shores), and prohibit both elver/glass eel fishing within 46 m of any dam and bycatch of other species (CAEMM 1996).

The **Connecticut** regulations were minimal until 1996, e.g., no small mesh fyke nets, but pots and dipping are permitted; catch reporting requirements permit minimal interpretation of catches. In 1996, Connecticut defined the glass eel as less than 10 cm in length, instituted a March 1-May 31 glass eel fishing season with a weekly closed period from 6:00 pm Saturday to 6:00 am Sunday, prohibited obstruction of more than 50% of the stream width and placement of traps within 7.6 m of each other, limited traps to a maximum of 10 within the state and 3 in any stream (dipnets are the preferred fishing gear) and required monthly catch reporting by logbook. The elver/glass eel fishery in **New Jersey** was unregulated prior to 1997 when it was restricted to dip-nets only and a fishery season was implemented (February 15-April 20) with a Sunday closure. The elver/glass eel fishery has been closed since 1998. (ASMFC 1997; J. F. McClain, New Jersey Division of Fish, Game and Wildlife, pers. comm.).

At various periods between 1957 and 1980, elvers (range 23,000 to 6,000,000 elvers) were annually stocked in the Susquehanna River, **Pennsylvania** upriver of hydroelectric dams, but no commercial eel fishery is permitted, and personal use harvesters are restricted to 50 eel per person per day.

Virginia issued, in 1996, two permits to fish a total of about 800 kg of elvers/glass eels for local aquaculture; no additional permits are planned for several years. When the cultured elvers have been reared to sale size, 10% must be returned to the state for release in the wild. **South Carolina** has an active elver/glass eel fishery. A limited fishery exists for elvers in Florida, and one experimental permit has been issued for harvesting glass eel.

The number of elver/glass eel fishers is generally unregulated in those states where an elver/glass eel fishery occurs (excluding **Virginia** where two permits exist, and **Florida** where the glass eel/elver fishery has been under limited regulation and three special device permits exist). In **Maine**, the number of commercial finfish permits (which may be used to fish eel as well as other species) almost tripled between 1985 and 1995 and more than doubled between 1994 and 1995 to over 3,300 permits, of which over 1,500 are believed to be elver/glass eel fishers (CAEMM 1996). As of the 1999 fishing season, Maine representatives claim that permits have been reduced by two-thirds of the 1994-1995 reports (J. Goldthwait Person. Comm.). **Connecticut** has had a moratorium on new commercial fishing licenses since 1995 but existing licensees can fish elver/glass eel if they choose. In **New Jersey**, over 2,100 licenses were issued for the 1997 elver/glass eel dip-net fishery (J. McClain, New Jersey Division of Fish, Game and Wildlife, pers. comm.). **South Carolina** had no mechanism for determining participation in the elver/glass eel fishery in coastal waters until 1996 when a permit was instituted (B. McCord, South Carolina Department of Natural Resources, pers. comm.). In 1997, about 65 permits for elver/glass eel hoop nets (a type of fyke net) and 11 permits for dip nets were issued. Each permit may authorize one or more units of gear.

Some states (**Connecticut**, **South Carolina**, and **Florida**) have no minimum length limit for eel retention (Table 2). **Maine** has a 6 in minimum size limit except during the elver season, which runs from March 15 through June 15. **New Hampshire** has a 10 cm minimum size limit as does **Massachusetts**, except for aquaculture (Amaral 1982). A 15 cm minimum size limit was imposed in **Virginia** in 1977 (CBP 1991) and has existed in **Georgia** since at least the early 1980s (J. Music, Georgia Department of Natural Resources, pers. comm.). **New York**, **Rhode Island**, **Delaware**, **Maryland, PRFC** and **North Carolina** have only recently (1992-1995) imposed a minimum length limit of 15 cm so as to protect elvers/glass eels for local aquaculture development or, more urgently, to prevent uncontrolled development of an elver/glass eel fishery. These states await the recommendations on elver/glass eel fishery development expected in the ASMFC fishery management plan for eel. In 1994, Maryland permitted a daily harvest per person of up to 25 eel of less than 15 cm for use as bait, primarily by anglers.

Maine has a defined elver/glass eel fishing season (March 15 to June 15). No states with an elver/glass eel fishery, other than Maine, have begun collection of catch statistics although this is expected to change when the ASMFC fishery management plan for eel is implemented. Poaching of elvers/glass eel is believed a serious problem in many states but enforcement of the often minimal regulations is poor due to the nature of the fishery (very mobile, nighttime operation) and low administrative priority.

Table 2. Commercial eel fishing regulations summary [1].

State/ Province	Minimum Length	Pot Mesh Size [2]	Freshwater Fyke	Weirs	License	Comments
Newfoundland	8"		2"/3" Stretch	No	No Data	
Prince Edward Island	18.4"		No	No	No Data	
Nova Scotia	8"		No	No	$ 10	
New Brunswick	8"		No	No	$ 10	
Maine	None (3/15-6/15) 6" (6/16-3/14)	½" x ½"	Yes	Yes	$ 33 + Gear Fee For Glass Eel For Residents $ 334 + Gear Fee For Glass Eel For Nonresidents $ 100 Weir/Pot	$75 For Dip Net. $100 Each For First Two Fyke Nets, $200 Each Next Three, Limit Five, For Glass Eel
New Hampshire	6"	Yes	No	No	$ 26 Resident $ 200+ Nonresident	Coastal Netting License Required For Nets & Pots
Massachusetts	4"		No	No	$ 65 Resident-Saltw. $130 Nonresident-Saltw.	Freshwater $25 plus state sports license $27.50
Vermont						
Rhode Island	6"		No	No	$ 200	
Connecticut	None	No	No	No	$ 50 Resident $ 100 Nonresident	Dip Net Glass Eel. 3/1-5/31 glass eel season with weekly closed periods. License Moratorium.
New York – Marine	6"	1" X ½"			$ 250 Resident $ 1250 Nonresident	License Moratorium
New York – Inland	None	No Opening Not > 2" Dia	Yes	Yes	$ 20 Resident $ 60 Nonresident	
Pennsylvania	6"					Ban on commercial eel fishing
New Jersey	6"	4/16" Bar	No	No	$ 10 Bait Net Resident $ 100 Bait Net Nonresident $ 100 Min. Fyke/Pot Residents $ 1000 Min. Fyke/Pot Nres.	Glass eel/elver fishery closed
Delaware	6"	No	No	No	$ 115 Resident $ 1150 Nonresident	
Maryland	6"	½" x ½" or escape panel	No	No	$300 Resident, tidal $350+ Nonresident, based on home state $100 unlimited finfish harvester	Limited entry
PRFC	6"	½" x ½"	No	No	$75 Per Boat	
District of Columbia						No Commercial Fishing
Virginia	6"	½" x ½" with 4" x 4" escape panels	No	No	$ 150 + Gear Fee	2 Year Wait
West Virginia	None				Resident + Conservation Stamp	Except 5/15-6/30. Gigging, Snagging, Snaring Are Prohibited
North Carolina	6"	1" x ½"	No	No	$ 10 Resident $ 50 Non-resident	20 Eel Limit Per Person Per Day
South Carolina	None	½" x ½"	Yes	No	$ 50 Resident $ 1000 Nonresident	Dip nets licensed, gear permit also required in addition to licenses
Georgia	6"	1½" x ½"	No	No	$ 12 Resident $ 118 Nonresident	
Florida	None	1" x ½"	No	No	$ 25 Resident $ 100 Nonresident	

[1] Regs subject to change: contact state for current requirements. [2] Escape panels of varying sizes by state required.

Yellow/Silver Eel

The United States fishery for American eel extends from Maine to the Gulf of Mexico. Different geographic regions (north, middle, and south Atlantic, Gulf of Mexico) exhibit differing trends and magnitudes in their eel fisheries, which reflect differences in their fisheries and stock abundances (Fahay 1978). The 1955-1973 fishery was most productive in the middle Atlantic region (New Jersey to Virginia), followed by the north Atlantic region (Maine to New York), south Atlantic region (North Carolina to Florida), and Gulf region where the catch was negligible (Fahay 1978). The regional catch summary statistics reported by Fahay (1978) are slightly lower than statistics recently available from the National Marine Fisheries Service (NMFS 1997), but the regional rankings are unchanged. For the years 1955-1973, regional mean catches were 146 t (range 75-251 t) for the north Atlantic region, 429 t (range 152-930 t) from the middle Atlantic region, and 80 t (range 19-192 t) from the south Atlantic region. For the years 1974-1995, regional mean catches increased to 160 t (range 7-556 t) in the north Atlantic region, to 567 t (range 106-1,349 t) in the middle Atlantic region, and to 236 t (range 6-792 t) in the south Atlantic region. The higher regional mean catch in the 1974-1995 period is accompanied by higher annual variability, reflecting the declining catch in all regions (and most states) from peaks in the mid-1970s and early 1980s to the low values of recent years.

For the Atlantic coast (Maine-Florida), annual eel catch ranged from 384 t to 1,645 t between 1970 and 1995, with values between 1.17 and 5.49 million U.S. dollars (ASMFC 1997). Eel catches averaged 1,179 t between 1970 and 1982 and 635 t between 1983 and 1995, indicating an overall decline in US catch.

Annual trends in reported eel catches by individual states (NMFS Fishery Statistics and Economcis Department pers. comm.) comprise three basic groups: declining catch, e.g., Rhode Island, New York; increasing catch, e.g., New Jersey, Delaware, Maryland; and catches that have returned to values typical of those reported prior to the peak catches of the 1970s and early 1980s, e.g., Maine, Massachusetts, Florida (Figure 6). Reported catches in some states declined sharply in 1996 but catch data may be incomplete.

Maine eel catches peaked in the late 1970s at 50-90 t annually and have since fluctuated between 4 t and 30 t, a level only slightly lower than reported between the early 1950s and early 1970s. In **Rhode Island**, eel catches varied moderately from about 9 t to 30 t between 1962 and 1984, then increased to between 19 t and 56 t between 1985 and 1988 before collapsing to about 1 t during 1989 and 1990 (Gray 1991). The catches reported by Gray (1991) during the mid-1980s are not evident in Figure 6 yet both data sets originate from the National Marine Fisheries Service. The variability in Rhode Island eel catch during the 1980s has been attributed to market forces rather than resource status.

Annual reported eel catches in **Connecticut** have usually been less than 10 t since about 1980 but some fishers blame the recent low catches on overharvesting of elvers/glass eels (NMFS 1997; S. Gephard, Connecticut Department of Environmental Protection, pers. comm.). Eel fisheries in inland (primarily Lake Ontario and Hudson River) waters of **New York** state have been closed due to organochloride contamination since 1976, with the exception of a “limited” fishery for export that closed in 1982 (Blake 1982; Lary and Busch 1997). Historically, catches of eel in New York were several times higher in coastal waters (1960-1978 mean catch of 68 t) than in inland waters (1960-1979 mean catch of 18 t). The export fishery evidently generated high catches in inland (mean 36 t) and coastal waters during the years 1980-1982 (Lary and Busch 1997).

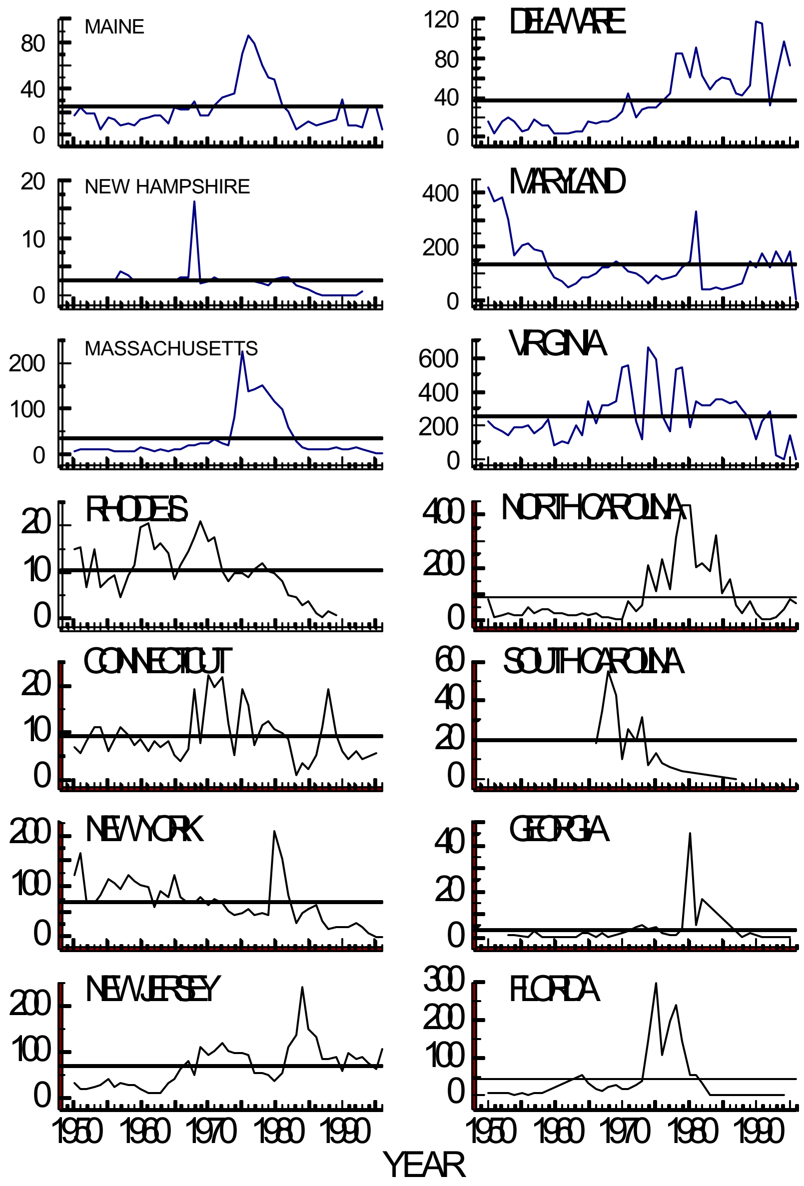


Figure 6. Annual reported (NMFS) catches of American eel, by state, for the Atlantic coast, 1956-96. The vertical line in each graph is the mean catch. Source: Jessop (1997).

In **New Jersey**, eel catches in the primarily coastal pot fishery ranged from 61-98 t between 1989 and 1993, down from the mid-1980s peak of 134 t but near the long-term mean. **Pennsylvania** issued 1 or 2 weir/chute licenses for use on the Delaware River. In 1997, the sole operator reported a harvest of less than one ton. No operations were conducted in 1998. The new regulations (no commercial sale, 50 fish daily limit, etc.) are expected to result in little, if any, interest in eel weir/chute operations.

In Chesapeake Bay (Maryland and Virginia) recent catches are near the long term (1945-1994) mean of about 450 t (CBP 1995). In **Maryland**, reported eel catches steadily declined from the peak of about 590 t in 1946 to about 45 t in 1963 and have since fluctuated between 45 t and 100 t (CBP 1995). The declining catch since the late 1980s evident in Figure 3 differs from the relatively stable catch reported elsewhere (CBP 1995). Reported catches in **Virginia** fluctuated between about 80 t and 190 t between 1946 and 1966, then increased irregularly to a peak of 659 t in 1974, before declining to 149 t in 1993 and rising to 360 t in 1994 (CBP 1995). Between 1984 and 1994, reported catches averaged 91 t (range 11-134 t) in Maryland (annual catch per fisherman increased from 0.9 to 2.0 t; CBP 1995) and 376 t (range 270-510 t) in Virginia (CBP 1995; Speir 1996). Reported catches in Maryland have thus shown no particular trend since about 1960 while Virginia catches remain near the long-term mean despite the decline from the 1974 peak.

Catches in **North Carolina** and **Georgia** have declined from the peaks in the early 1980s to levels not seen since the 1960s and early 1970s. Before 1970, annual eel catches in North Carolina were usually less than 45 t, then peaked at 436 t in 1980 (Keefe 1982) before declining to 6-26 t in the 1990s. The mean annual catch of the minor fishery in Georgia declined from 8.5 t between 1972 and 1982 to 1.6 t between 1983 and 1995 (J. Califf, Georgia Department of Natural Resources, Brunswick, pers. comm.). Although catches of over 100 t were reported from **Florida** during the mid and late 1970s, the only significant eel fishery in Florida today is the pot fishery of the Saint Johns River. Recently, catches in this fishery have declined, due in part to reduced fishing effort (NMFS 1997; J. Crumpton, Florida Game and Freshwater Fishery Commission, Eustis pers.comm.). Reports indicate that the maximum number of fishers involved in the Florida eel fishery has never exceeded 50 participants (J. Crumpton, Florida Game and Freshwater Fishery Commission, Eustis pers.comm.). Currently, there are 25 – 30 fishers involved in the fishery and participation has been stable since the mid 1980s.

Drawing conclusions from these trends is difficult because the available catch statistics are generally regarded as underestimates, perhaps varying in completeness over time, and fishing effort data are either unavailable or of questionable utility (Foster 1981; CBP 1991; Crawford 1996; NMFS 1997). The current status of the eel stock in most, if not all, states is unknown due to the absence of catch and/or effort statistics and an absence or scarcity of biological study of any kind. A widespread concern about the status of local eel stocks, except perhaps in Pennsylvania, Georgia, Florida, and the Gulf of Mexico where stocks and fisheries are not usually as large as in other areas, reflects more the absence of knowledge about the stocks rather than a well-founded knowledge of decline.

The economically important yellow/silver eel fishery in **Maine** occurs in both inland and tidal waters. The fishery is comparatively well documented and has recently received a comprehensive review and modernization of regulations (CAEMM 1996). Most large eel fisheries south of Maine seem to be primarily coastal pot fisheries with little management and few regulations, other than a license requirement and perhaps minimum size limit or gear and mesh size restrictions (Table 2). Eel fisheries are conducted during the period of natural availability, and few, if any, states have defined fishing seasons. **New Hampshire** has little coastline and no available data on eel fishing. Coastal town authorities (little if any eel fishing occurs inland) manage the coastal eel fisheries of Massachusetts; state regulations control permitted gear types (Amaral 1982). The tidal water, mainly pot fishery conducted between May and November in **Rhode Island** requires a commercial multispecies marine fishing license but no catch statistics are collected by state agencies for the eel fishery (Gray 1991). **Connecticut** has a relatively small, basically unmanaged, pot fishery for yellow eel in the tidal portions of, primarily, the Connecticut and Housatonic rivers (S. Gephard, Connecticut Dept. of Environmental Conservation, pers.comm.).

Table 3. Commercial landings and value of American eel in the State of Maine.

Year	Landings, pounds	Value	Average Price per Pound
1994	64,135	$85,473	$1.52
1993	14,521	28,022	1.93
1992	30,672	55,823	1.82
1991	18,217	27,331	1.50
1990	66,164	86,320	1.30
1989	27,900	29,247	1.05
1988	----[1]		
1987	13,288	13,700	1.03
1986	16,703	13,219	0.79
1985	24,100	18,288	0.76
1984	8,764	6,610	0.75
1983	11,900	8,925	0.75
1982	45,051	36,637	0.81
1981	55,125	45,308	0.82
1980	105,463	111,061	1.05
1979	111,206	89,214	0.80
1978	133,388	161,892	1.21
1977	175,711	262,596	1.49
1976	191,025	93,665	0.49
1975	154,836	82,380	0.53
1974	79,524	32,318	0.41
1973	79,890	29,555	0.39
1972	70,210	24,578	0.35
1971	54,300	15,204	0.28

[1] No data on landings collected in 1988

Licensed eel fishing in **New York** occurred, primarily in Lake Ontario, the Hudson River (prior to the 1976 closure), and the upper Delaware River (Blake 1982). Only eel less than 36 cm may be fished in the Hudson River proper and other inland waters and must be used for bait because of organochloride contamination. Coastal fisheries are unlicensed and fishing effort is not monitored in either inland or coastal waters. New York enacted, in 1995, a 15-cm minimum size limit and 1.25 x 2.5-cm minimum mesh size for trap nets in marine waters. **New Jersey** fishery regulations require a fishing license for fyke nets and pots, a minimum 4.8-mm bar mesh in pots and a 15-cm minimum size limit. Eel fisheries in **Delaware** were recently licensed and had a 15 cm minimum length limit set in 1995 but are otherwise unregulated and thus have no available catch data.

Maryland and **Virginia** primarily operate pot fisheries for eel in Chesapeake Bay, for which a management plan was developed in 1991 (CBP 1991, 1995; Speir 1996). Prior to the 1991 management plan, **Pennsylvania**, **Maryland**, and **Virginia** had no harvest quotas (**Pennsylvania** has a 50 eel per person per day creel limit), bycatch restrictions or closed season nor do they exist under the management plan. Prior to the 1991 management plan, Virginia had a 1.25 x 1.25-cm minimum mesh size and requirement for two 1.25 x 2.5-cm escape panels for eel pots. Maryland has implemented a similar minimum mesh size under the management plan. Large eel are exported whereas small eel are used for bait in the crab trotline fishery. Such use is of declining importance. Catch reports were not required in Virginia prior to 1973 and the Maryland eel fishery was unlicensed prior to 1981. Furthermore, Maryland did not require reporting of eel catches until 1990 (Foster 1981; CBP 1995; Speir 1996). The National Marine Fisheries Service made estimates of commercial eel landings based on interviews with fishhouse managers for both states from 1929 onward (CBP 1995).

North Carolina has a small, primarily coastal pot fishery, with no catch records maintained for inland waters, although they may be included in the total catch. **South Carolina** recently instituted a permitting system to document total eel gear and commercial harvest (B. McCord, South Carolina Department of Natural Resources, pers.comm.). Traps, pots, fyke nets, and dip nets are permitted in coastal waters. Fishing for eel in coastal waters is often conducted under the guise of fishing for crabs.

Eel fishing in **Georgia** was restricted to coastal waters prior to 1980 when inland fishing was permitted (Helfman 1982). Catch, but not effort, data is available because no specific license is required to fish eel. The **Florida** pot fishery has a 1.25 x 2.5-cm minimum mesh size and no minimum catch size limit, although frequency data indicates that the minimum size harvested by Florida pots is approximately 12 inches (J. Crumpton, Florida Game and Freshwater Fish Commission, Eustis pers. comm.).

1.3.2. Glass Eel Fishery

Maine landings of glass eel have been recorded separately from landings of adult eel since 1994. The elver/glass eel landings and value for 1994 and 1995 (DMR and DIFW 1996) were:

Year	Pounds landed	Value ($)	Average Price ($) per Pound
1994	7,347	367,350	50
1995	16,599	3,821,842	230

1.3.3 Bait Fishery

The information available from NMFS concerning eel harvested for bait indicate a decrease in pounds harvested. However, during this period average eel weight ranged from 0.25 to more than 1 pound. While the data needs to also be adjusted for numbers, it is arguable that this trend would remain apparent in light of such adjustments. In addition, recreational bait harvest is not recorded.

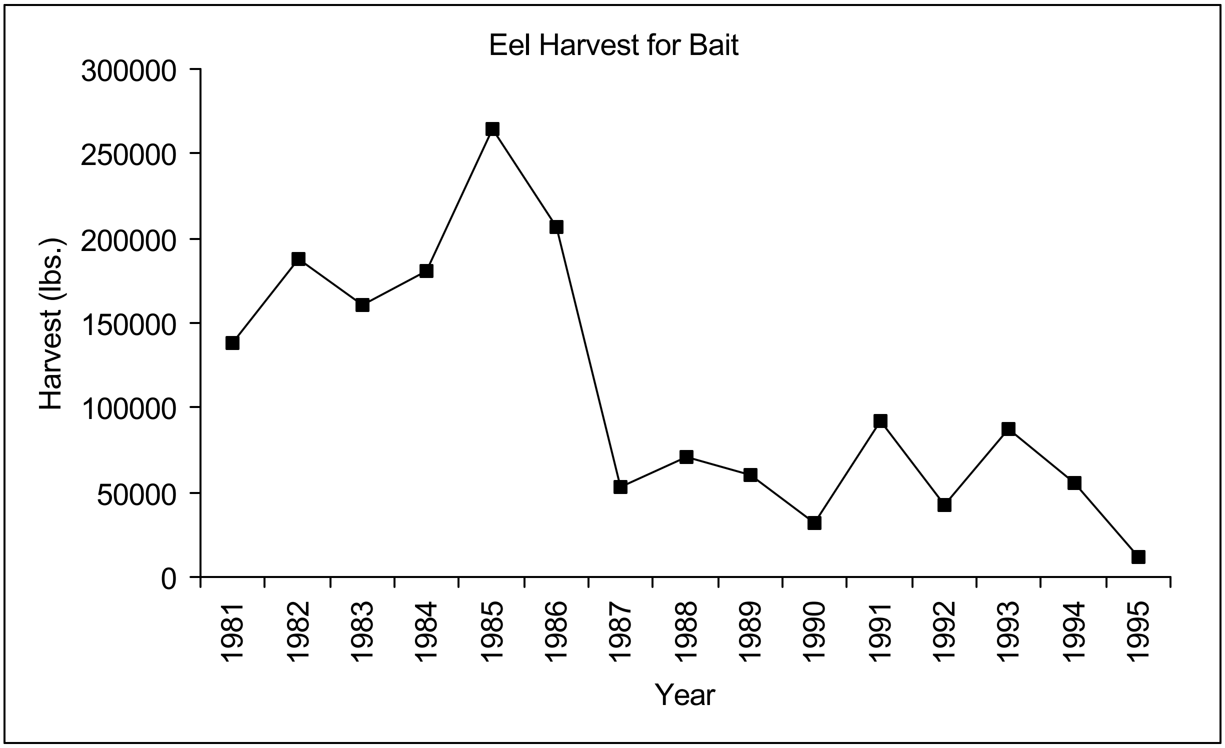


Figure 7. Harvest of American eel for bait (1981-1995) (NMFS)

1.3.4 Overall Commercial Fishery

1.3.4.1 Landings vs. Live Exports

Landings of American eel reported to the National Marine Fisheries Service (NMFS) were variable from 1970 to 1979 with lows in 1973 and 1977 of 592,091 kgs and 955,182 kgs and highs in 1975 and 1979 of 1,610,409 kgs and 1,648,607 kgs respectively. The trend shifted predominantly downward from 1979 to 1995 with a 76.7% decrease in kgs landed from 1979's high to a record low of 384,830 kgs in 1994.

Landings of American eel reported to NMFS were often far below the weight of eel exports reported by the U.S. Census Bureau. In 1993, a harvest of 400 tons of eel was reported to the NMFS but data from the Census Bureau indicate that 1,043 tons of live eel were exported, or 261% more than the reported harvest. Reported harvest decreased 45% during the three-year period from 1993 through 1995. By 1995, the difference in harvest reported to the NMFS or to the Census bureau had dropped to 3.6%.

1.3.4.2 Number and Value of Exports

The number of reported shipments of live American eel from 1992 to 1995 rose 153%, from a low of 240 to a high of 367. The number of reported shipments dropped again in 1996 to 308. The total value of American eel shipments rose dramatically during the time period. Values held relatively steady in 1992 and 1993 at around $4,600,000, but began to rise in 1994 to $6,967,019 and then increased in 1995 to $10,688,579. This represented a 230% increase from the 1992 low. The values dropped again in 1996 to $8,748,560, but remained 188% above 1992.

The mean value per shipment of American eel increased from 1993 to 1996, but showed a differential rate of increase dependent upon whether the shipment was destined for a European or Asian port. The mean value of a European bound shipment in 1993 was $14,184. The mean value for a similar shipment increased 65.0% in 1996 to a four year high of $23,438. The mean value of an Asian bound shipment in 1993 was $24,297. The mean value rose 59.9% in 1994 to $38,862 and continued to increase to a four year high of $42,707 in 1996, for a total 75.8% increase over the 1993 value. The difference in mean shipment value between shipments bound for European and Asian destinations increased from 1992 to 1996 with Asian shipments valued 72.3% more than European shipments in 1992 to an 82.2% higher value for Asian destinations in 1996.

During 1996 the number of live American eel export shipments showed a bimodal distribution with peaks in April and October of 63 and 34 shipments respectively. This contrasted with three shipments of 3 in January, 11 in August, and 8 in December. The total weight of those shipments showed a similar pattern ranging from 2,059 kgs in January to 122,321 kgs in April, dropping to 11,658 kgs in August, rising again to 78,102 kgs in October, and finally ending the year with 12,959 kgs in December. The value of live American eel shipments in 1996 likewise followed a bimodal pattern with peaks of $2,438,580 in April and $659,343 in October. The distribution of shipping patterns changed in both port of exportation and port of destination from 1993 to 1996. In both years New York handled the largest number of shipments with 135 in 1993 and 165 in 1996. Boston with 92 shipments and Washington, DC with 51 were second and third largest in 1993. However, by 1996 Maine border ports with shipments trucked to Canada tied for second largest number with Washington, DC at 46. Traffic at Boston dropped to only 13 shipments for 1996. In 1993, 280 shipments or 90.9% of total exportations of live American eel were destined for European ports while 28 or 9.1% were destined for Asian ports. Although the same number of shipments were exported in 1996 as in 1993 (308), the pattern of destination ports shifted so that 36.6% of shipments went to Asian ports, 47.4% to European ports, and 16.2% went to North American destinations.

Data for weight (kg) of American eel landed in the US for 1970 to 1995 were obtained from National Marine Fisheries Service (NMFS) (personal communication from the NMFS, Fisheries Statistics and Economics Division). Yearly export figures for live American eel shipments for 1993 through 1995 were obtained from U.S. Exports of Merchandise issued by the US Census Bureau (USCB). Monthly figures for live American eel exports for 1996 were obtained from individual monthly CD-ROMS for the U.S. Exports of Merchandise.

Information on American eel landings is collected by NMFS from the states of Connecticut, Delaware, Massachusetts, Maryland, Maine, North Carolina, New Jersey, New York, and Virginia. These data document landings from the majority of states with commercial American eel fisheries, but must be considered only a partial summary, as several other range states are not included.

Information in U.S. Exports of Merchandise is provided by shippers at the time of exportation via submission of a Shipper's Export Declaration (SED) to U.S. Customs Service (USCS). USCS forwards that information to the US Census Bureau for compilation and dissemination to the public. Shippers are required to furnish SED's for all export shipments valued in excess of $2,500, but this valuation level may mean that some small American eel shipments are not reported at the time of exportation.

There is a further caveat in the use of these data sets as neither differentiates among American eel life stages. In all likelihood, NMFS data consist almost exclusively of adult American eel as it is based on reported landings. USCB data is probably based on adult American eel shipments, but may include some portion of immature American eel, primarily the glass eel or elver stages.

1.3.5 Recreational fisheries

Few recreational anglers directly target eel. Eel, for the most part, are caught incidentally by hook and line fishermen when fishing for other species. The NMFS Marine Recreational Fisheries Statistics Survey (MRFSS), which has surveyed recreational catch in ocean and coastal county waters since 1981, shows a declining trend in the catch of eel during the latter part of the 1990's. From the Atlantic coast area surveyed, the estimated total annual catch of eel ranged from 212,690 eel per year in 1982 to 36,741 eel per year in 1997. About one half of the eel caught were released alive by the anglers. Eel are often purchased by recreational fishermen for use as bait for larger gamefish such as striped bass, and some recreational fishermen may catch eels and then utilize them as bait

1.3.6 Subsistence fisheries

Little is known as to the current extent (i.e., quantity) of subsistence fisheries for American eel. American eel are a valuable subsistence food source for some European and Asian ethnic groups, and , as noted earlier, represent an important food, cultural, and spiritual resource to many Native American tribes.

1.4 Habitat Considerations

1.4.1 Habitat Important to the Stocks

1.4.1.1 Description of Habitat

A habitat area of particular concern is defined, as those waters, substrate, and conditions required for population survival. Such habitat may be limiting for spawning, breeding, feeding, or growth to maturity.

Information inferred from commercial harvest records and various stock assessment efforts indicate that American eel are found in most types of habitats including the offshore, mid-water and bottom areas of lakes, estuaries and large streams. American eel are found to be most prevalent in the nearshore, shallow embayments and tributaries (Adams and Hankinson 1928, Facey and LaBar 1981, GLFC 1996, Helfman et al. 1983, NYSDEC 1997a & b).

American eel are classified as a warmwater species (Adams and Hankinson 1928) that are most abundant in relatively warm streams and shallow lakes or embayments (Ogden 1970), while relatively scarce in deep, steep gradient cold-water lakes (Smith and Saunders 1955). Based on distribution and diet preferences, American eel appear to be very adaptable creatures with the ability to exploit many habitat and food types. Some juvenile American eel, for example, seek out riverine habitat until reaching maturity at which time they return to the ocean. These habitats provide the conditions needed by the organisms (insects, crustaceans, fishes) that eel forage upon.

American eel are bottom dwellers while in estuaries, rivers, and lakes. The presence of soft, undisturbed bottom sediments may be important to migrating elvers for shelter (Facey and Van Den Avyle 1987). American eel have been reported in mud burrows with their heads protuding (Fahay 1978). Few other freshwater fishes display similar habitat use, and as a result, interspecific competition for living space may be limited (Facey and Van Den Avyle 1987). Estimates of the home range of eel extend to 3.4 ha in small streams, tidal rivers, and tidal creeks (Gunning and Shoop 1962, Bianchini et al. 1982, Bozeman et al. 1985) and 2.4 to 65.4 ha in a large lake (LaBar and Facey 1983).

Current research shows extensive use and home-range development of shallow lakes (< 17meters) by American Eel (Daniels 1999). Many riverine systems utilized by American eel in North America contain lakes and large bodies of water, but only the St. Lawrence basin includes the large inland Lake Ontario.This system is, therefore, the exception and raises doubt that the lake proper is the desired "end point " for the freshwater, inland migration of eel. While Lake Ontario may support a percentage of the stock at any one time, it is likely that the eel inherently continue to seek out riverine habitat in Lake Ontario tributaries, as they do in other East Coast streams. Lake Ontario is very limited in shallow habitats (due to its depth and narrow littoral area) and American eel must seek out their preferred forage in the habitat where it is abundant, such as in embayments and rivers where benthic invertebrate densities are found to be highest (Lary and Busch 1997).

Spawning Habitat

American eel are highly migratory, with spawning and larval development and migration occurring in the open ocean, feeding and growth occurring in estuaries and fresh waters, and migration of adults occurring in the ocean again to complete the life cycle [catadromous life cycle]. American eel spawn in the Sargasso Sea although it has never been directly observed in the field (Facey and Van Den Avyle 1987).

The Sargasso Sea is an oval area in the middle of the Atlantic Ocean, between the West Indies and the Azores, of nearly 5.2 million km^2 (2 million $miles^2$). Although the boundaries are not easily delineated, the area is identified as the "eye" of a large, slow, clockwise moving gyre of very clear, deep blue colored, warm surface waters, with elevated salinity. The Gulf Stream provides the western boundary, which along with other ocean gyres, such as the North Equatorial Current, encircles the Sargasso. According to Ginsberg (1996), Portuguese sailors named the area for its seaweed since the seaweed's bulbous floats are similar to grapes (sargaco is the Portuguese word for grape). Sargassum seaweed floats in patches and grows through budding. The warm waters of the Sargasso Sea are low in nutrients, which is attributed to its isolation from the deeper, nutrient rich, cold waters (average depth greater than 3 miles). Plankton production is about one-third the oceanic average, however, tiny crabs, shrimp, octopus and other marine animals are abundant among Sargassum.

Although specific spawning areas used by American eel and their habitat parameters have not been identified, Miller (1995) reported two major distribution patterns for leptocephali. The highest abundance of leptocephali were identified in areas located near fronts in the west of the Subtropical Convergence Zone (STCZ). The smallest leptocephali were reported by Miller (1995) to have been collected near the Bahama Banks in the Florida Current and at stations close to the southerly fronts in the western STCZ. Miller (1995) attributes the concentration of leptocephali to "entrainment by anticyclonic circulation northeast of the northern Bahamas."

American eel from throughout their range are believed to synchronize their arrival at the spawning grounds. Morphological and physiological evidence suggests that they may spawn in the upper few hundred meters of the water column (Kleckner et al. 1983, McCleave and Kleckner 1985). Spawning has been inferred to take place from February to April within a broad area in the vicinity of the Sargasso Sea between 52° to 72 ° W longitude and 19 ° and 29 ° N latitude (McCleave et al. 1987). Kleckner et al. (1983)

suggested that thermal fronts separating the northern and southern water masses of the Sargasso Sea form the northern limit of American eel spawning and that some feature of the surface water mass in the southern Sargasso Sea serves as a cue for adult American eel to cease migration and begin spawning activity. After spawning, the spent eel are assumed to die (Facey and Van Den Avyle 1987).

American eel are dioecious, oviparous, and rely on external fertilization. Fertilized eggs reached the gastrula stage before dying 15 h later at 20 ° C (Sorensen and Winn 1984). Artificially spawned Japanese eel (*Anguilla japonica*) are known to hatch in 38-45 hours at 23 ° C (Yamamoto and Yamauchi 1974). Spawning occurs in winter and early spring (Wippelhauser et al. 1985, Kleckner and McCleave 1985, McCleave et al. 1987) probably in association with, or delimited by, density fronts meandering east-west in the Sargasso Sea (Kleckner and McCleave 1988). Eggs hatch in about two days in the warm water (Yamamoto and Yamauchi 1974), releasing the leptocephali. Knowledge of the spawning area is based on the distribution of the smallest leptocephali, as adults have never been observed in the Sargasso Sea.

Leptocephali are transported from the spawning grounds to the eastern seaboard of North America by the Antilles Current, the Florida Current, and the Gulf Stream (Facey and Van Den Avyle 1987). The leptocephali drift and swim in the upper 300 m of the ocean for several months, growing slowly to a length of 5-6 cm (Kleckner and McCleave 1985). Most planktonic leptocephali undergo metamorphosis into glass eel at 5.5-6.5 cm in length at 8 to 12 months of age (Facey and Van Den Avyle 1987), that actively migrate from the offshore waters to the coastal embayments and rivers. American eel apparently take advantage of inflowing tides to move into tidal areas (Wippelhauser and McCleave 1987).

Nursery and Juvenile Habitat

Glass eel enter estuaries and ascend the tidal portion of rivers during winter and spring, earlier in the southern portion of the range, later in the northern portion (Helfman et al. 1984a, McCleave and Kleckner 1982) by drifting on flood tides and holding position near bottom on ebb tides, a migratory tactic known as selective tidal stream transport (McCleave and Kleckner 1982, Wippelhauser and McCleave 1987). Glass eel also ascend by active swimming along shore in the estuaries (Sheldon and McCleave 1985), and above tidal influence (Barbin and Krueger 1994).

Upstream migrating glass eel metamorphose into elvers. Glass eel and elvers burrow or rest in deep water during the day (Deelder 1958). Upstream migrations may be triggered by changes in water chemistry caused by the intrusion of estuarine water during high spring tides (Sorensen and Bianchini 1986).

Limited work on preferred freshwater habitats indicates both lentic and lotic habitats are used and growth appears to be more related to density and availability of food than to water body (Oliveira and Krueger 1999). Bigelow and Schroeder (1953) reported that some elvers are able to surmount obstacles such as falls, dams, and damp rocks during their upstream migrations.

Observation of elver migrations in coastal Rhode Island streams indicates that the main concentration of elvers required about one month to move a distance of 200 m above the tidal zone in a stream with an average gradient of 4 m/km (Haro and Krueger 1991). Elvers orient to river currents for their upstream migration (Tesch 1977) and are strongly attracted to the odor of decaying leaf detritus (Sorensen 1986). Further migration may occur gradually for months or even years (Haro and Krueger 1991).

Elvers exhibit drab pigmentation, dark on the back and often yellowish on the ventral surface, leading to the name yellow eel for this stage. Yellow eel inhabit a variety of habitats and feed opportunistically on various bottom-and near bottom-dwelling animals, mostly invertebrates and slower fishes (Ogden 1970, Wenner and Musick 1975, Facey and LaBar 1981, Lookabaugh and Angermeier 1992, Denoncourt and Stauffer 1993).

Telemetry studies showed that yellow eel in a tidal creek were generally inactive during the day and active at night (Helfman et al. 1983). Growth rates of yellow eel are quite variable, reflecting both latitude (slower growth in the north) and productivity of the habitat, perhaps sex, and probably some difficulty in interpreting putative annual rings in otoliths. Even within a habitat, growth rates of individuals are variable. In Lake Champlain, Vermont, weight of eel was well predicted by length (variation in length accounting for 93% of the variation in weight), but age was poorly predicted by length (accounting for only 27% of the variation in age) (Facey and LaBar 1981). Illustrating the latitudinal trend in length, eel five years of age post-metamorphosis from Georgia averaged about 40 cm long (Helfman et al. 1984b), from South Carolina about 50 cm (Harrell and Loyacano 1980; Hanson and Eversole 1984), while in New Jersey they were about 25 cm (Ogden 1970), and in Newfoundland only about 28 cm (Bouillon and Haedrich 1985). However, the trend is complicated by the habitat variability. In an estuarine habitat in Georgia, five-year-old eel averaged 38 cm, while in two freshwater habitats they averaged 33 cm and 40 cm (Helfman et al. 1984).

Adult Habitat

Yellow eel metamorphose into silver eel and migrate seaward to their spawning grounds. The American eel that are in freshwater drop downstream, traveling mostly at night (Bigelow and Schroeder 1953). During outmigration, adults may inhabit a broad range of depths throughout the water column. Turbine entrainment mitigation efforts at hydroelectric projects may be complicated since bypass systems must be accessed throughout the full depth of the turbine forebay (Richkus and Whalen 1999).

Adult oceanic habitat requirements are not known. However, American eel have been taken at depths greater than 6000 meters.

1.4.1.2 Identification of Habitat and Habitat Areas of Particular Concern

1.4.1.2.1 Ocean

Importance: Spawning - Reproduction for the panmictic population occurs in the Sargasso Sea, therefore, the area used for reproduction might be identified as a habitat area of particular concern. Until recently, no threats to the functional health of this area had been reported.

Concern: Sargassum seaweed is currently harvested in U.S. waters by trawling primarily by one company. The harvesting of sargassum began in 1976, but has only occurred in the Sargasso Sea since 1987. Since 1976, approximately 44,800 dry pounds of sargassum have been harvested, 33,500 pounds of which were from the Sargasso Sea (SAFMC 1998). It is unknown whether this harvest is having direct or indirect influences on American eel mortality. Harvesting sargasssum is being eliminated in the south Atlantic EEZ and State waters by January 1, 2001 through a management plan adopted by the South Atlantic Fisheries Management Council (SAFMC 1998). The extent of eel bycatch in these operations is unknown. The drift of leptocephalus larvae from the Sargasso Sea towards the Atlantic coast may be impacted by changes in the ocean currents. Such changes have been predicted to be due to global warming. The potential impact on the drift of larvae is unknown at this time. Currents, primary production, and potential influence of toxins transferred from the adults to the eggs influence the success of hatch, larval migration, feeding and growth.

1.4.1.2.2 Continental shelf

Importance: Larval migration, feeding, growth; juvenile metamorphosis, migration, feeding and growth.

Concern: Glass eel survival (growth, distribution and abundance) is probably impacted by a variety of activities. Channel dredging, shoreline filling, and overboard spoil disposal are common throughout the Atlantic coast, but currently the effects are unknown. Additionally, these activities may damage American eel benthic habitat. However, the significance of this impact also remains unknown. Changes in salinity in embayments, as a result of dredging projects, could alter American eel distribution.

1.4.1.2.3 Estuaries/Rivers

Importance: Juvenile, sub-adult and adult migration corridors and feeding and growth areas for juvenile and sub-adult.

Concern: Elver and yellow eel abundance is probably also impacted by physical changes in the coastal and tributary habitats. Lost wetlands or access to wetlands and lost access to the upper reaches of tributaries have significantly decreased the availability of these important habitats with wetland loss estimated at 54% (Tiner 1984), and Atlantic coastal tributary access loss or restriction estimated at 84% (Busch et. al 1998).

Habitat factors are probably impacting the abundance and survival of yellow and silver eel. The nearshore, embayments, and tributaries provide important feeding and growth habitat. The availability of these habitats influences the density of the fish and may influence the determination of sex. Therefore, since females may be more common in lower density settings (Krueger and Oliveira 1999, Roncrati et al. 1997, Holmgren and Mosegaard 1996, Vladykov 1966, Liew 1982, Columbo and Rossi 1978), it is crucial that the quantity and quality of these habitats be protected and restored (including upstream access). The blockage or restriction to upstream migration caused by dams reduces or restricts the amount of available habitat to support eel distribution and growth. Fish that succeeded to reach upstream areas may also face significant stresses during downstream migration. If eel have to pass through turbines, mortality rates range from 10 to 60 percent (J. McCleave, U. of Maine, Person. Com.) and the amount of injury is not well documented.

An estimate of nearshore habitat area was obtained from NOAA's Average-Annual, Three-Zone Salinity Metadata and for coastal stream length from Busch et al. (1998) as summarized in Table 4. Although the nearshore zones have been changed due to anthropogenic activities such as dredging, filling, discharges of waste and contaminants and the introduction of exotic species, nearshore habitat trend data are not available for this area. Preliminary data describing trends in lost stream habitat (access length) are presented in Section 1.4.1.2.3.3.

Table 4. Estimated current nearshore habitats (area) and length of access to historic river habitats (potential if currently restricted). Some geographic overlap occurs between the areal (nearshore) and linear (coastal rivers) habitat descriptions (Busch et al 1998).

	Habitat	North Atlantic	Mid Atlantic	South Atlantic
Near-shore	Seawater Zone (>25ppt)	5,096 km^2	8,382 km^2	2,713 km^2
	Mixing Zone (0.5 – 25ppt)	229 km^2	10,969 km^2	8,300 km^2
	Tidal Fresh Zone (<0.5ppt)	54 km^2	947 km^2	1,159 km^2
Length of Coastal Rivers	Historic (unrestricted)	111,482 km	199,312 km	246,007 km

The nearshore area totals are the summation of areas designated by NOAA by drawing boundary lines across open water from shorelines. NOAA's Coastal Assessment Framework (CAF) provided the geographies for the shorelines. Busch et al. (1998) used computer databases and a Geographic Information System to assess the quantity of historic (unrestricted) stream habitat available to American eel.

1.4.1.2.3.1 Access to Tributaries

Large numbers of elvers and yellow American eel migrate inland from coastal waters each year, but obstructions such as dams impede migrants in reaching appropriate upstream habitat. Because of their small size and limited swimming speed, elvers and young eel depend on tides to aid upstream migration. Altering stream flows may limit upstream recruitment. Although elvers will attempt to scale wetted substrates such as dam faces, for many of the migrants dams probably limit migration (Tesch 1977). Cost effective passageways designed specifically for elvers and eel have been developed and tested in Europe, Canada, and New Zealand. Knowledge of where migrants accumulate at a barrier and of migrant size (length) is necessary for construction of passageways.

Downstream passage at hydropower dams may represent a major source of mortality to pre-spawning adults (Ritter et al 1997), but has received relatively little attention. Mortality rates for European eel are reported to range from 5-30% depending on turbine type and river flow (Hadderingh 1994). The design of downstream passageways and the use of non-generating periods to reduce eel mortality is hindered by lack of knowledge of the downstream migration. For example, the environmental cues that trigger migration, the depth of migration, and the effects of light and water currents on eel behavior during migration, are all unknowns.

1.4.1.2.3.2. Fish passage

Fish passage is getting attention through the licensing or relicensing of dams for hydropower production and navigation. Upstream fish passage is usually a requirement but construction activities are mostly in the planning process. However, more than 90% of dams on the eastern seaboard are not hydroelectric facilities, and therefore have not been subject to continual relicensing and fish passage analysis.

Downstream passage of silver eel is a problem in streams with hydropower production facilities. Although the industry has been researching effective deterrence to passage mortality, turbine caused damage or mortality continues to be a problem.

1.4.1.2.3.3 Quantity-Stream Habitat

Busch et al (1998) used an ecosystem health assessment approach, developed for the Lake Ontario watershed (Busch and Lary 1996), to determine that Atlantic coastal streams from Maine to Florida have 15,115 dams that can hinder or prevent upstream and downstream fish movement. This results in a restriction or loss of access for fish to 84 percent of the stream habitat within this historic range. This is a potential reduction from 556,801 kilometers to 90,755 kilometers of stream habitat available for migratory and diadromous species such as American eel. The analyses were based upon the regional boundaries established by the USEPA database (Figure 8) and excluded obstruction caused by most natural barriers.

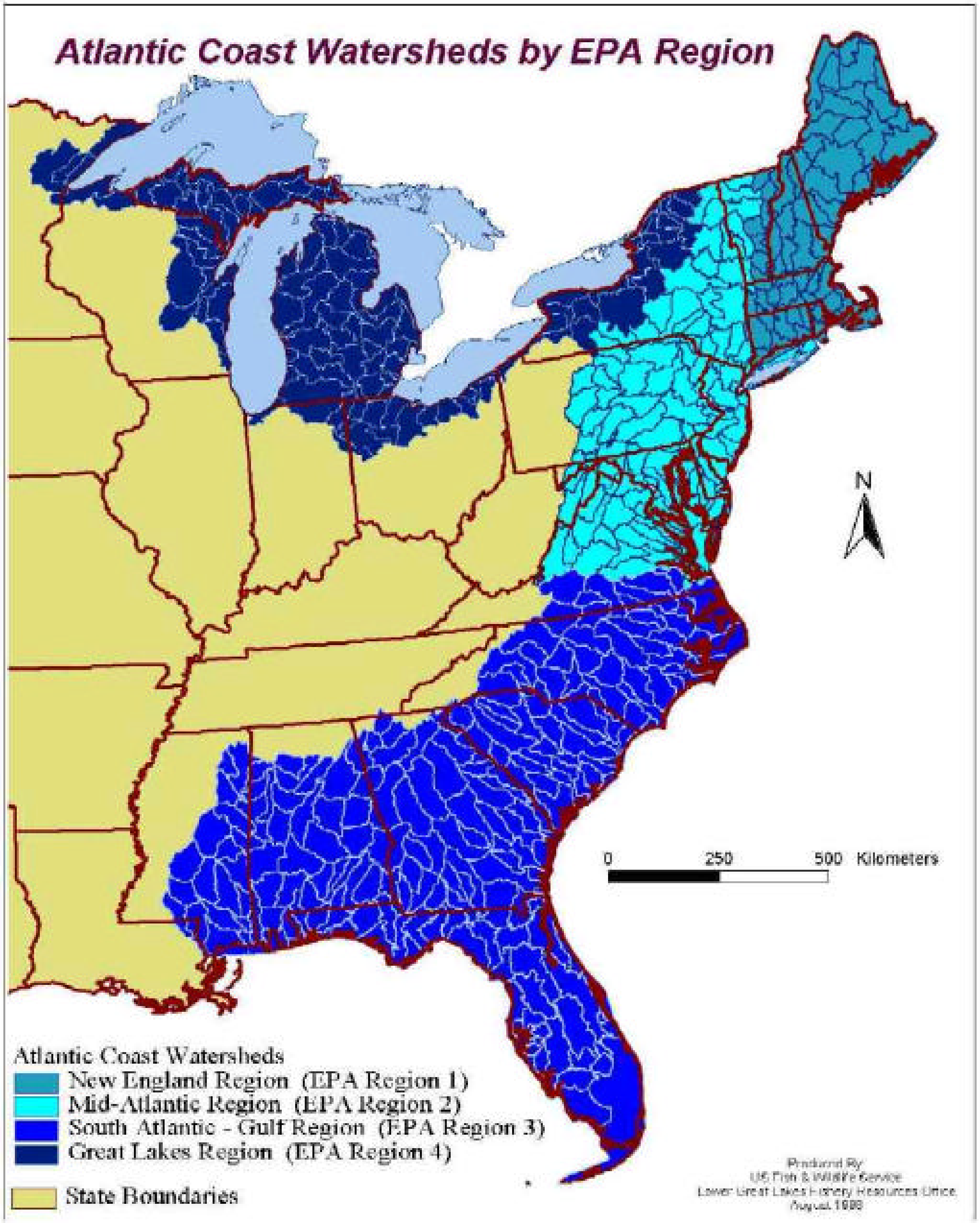


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Figure 8. The regional boundaries from the USEPA database as used by Busch et al. (1998)

By region, the potential habitat loss was greatest (91%) in the North Atlantic region (Maine to Connecticut) where stream access is estimated to have been reduced from 111,482 kilometers to 10,349 unobstructed kilometers of stream length (Table 5). Stream habitat in the Mid Atlantic region (New York through Virginia) is estimated to have been reduced from 199,312 km to 24,534 km of unobstructed stream length (88% loss) (Table 6). The stream habitat in the South Atlantic region (North Carolina to Florida) is estimated to have decreased from 246,007 km to 55,872 km of unobstructed stream access, a 77% loss (Table 7).

Table 5. Eel habitat, North Atlantic region (Maine to Connecticut)

Huc4 Number and Watershed Name	Historical length (km)	Current Length (km)	Number of dams	Dams <10 ft.	Dams 10-24 ft.	Dams 25+ ft.	Hydro-Electric	Nav.
101 St. John River Basin	11,335	148	37	3	19	15	10	0
102 Penobscot River Basin	15,245	207	75	9	49	17	53	0
103 Kennebec River Basin	9,186	208	97	11	66	20	54	0
104 Androscoggin River Basin	4,467	195	95	15	57	23	54	0
105 Maine Coastal – St. Croix	10,884	5,166	98	22	69	7	34	0
106 Saco, ME, NH, MA	9,414	1,685	212	28	155	29	74	0
107 Merrimack River Basin	11,006	10	533	87	348	98	93	0
108 Connecticut River Basin	20,874	99	941	93	538	310	119	0
109 MA-RI Coastal Area	7,886	1,589	708	133	487	88	13	4
110 Connecticut Coastal	10,335	1,188	713	42	467	203	49	0
111 St. Francois Rriver Basin	850	1	13	5	5	3	8	0
Totals	111,482	10,348	3,522	448	2,260	813	561	4

Table 6. Eel habitat, Mid Atlantic region (New York through Virginia)

Huc4 Number and Watersheds Name	**Historical length (km)**	**Current Length (km)**	**Number of dams**	**Dams <10 ft.**	**Dams 10-24 ft.**	**Dams 25+ ft.**	**Hydro-Electric**	**Nav.**
201 Richelieu Basin including Lake Champlain drainage	9,126	1	235	24	125	83	68	1
202 Upper Hudson	22,389	1	660	91	373	194	64	17
203 Lower Hudson – Long Island	7,781	1,431	519	64	324	127	8	0
204 Delaware Coastal Area	26,934	5,148	1068	179	656	231	21	0
205 Susquehanna River Basin	52,331	251	684	75	324	285	19	2
206 Upper Chesapeake	14,884	8,862	157	13	93	51	3	0
207 Potomac River Basin	28,140	3,281	443	7	141	295	12	0
208 Lower Chesapeake	37,727	5,559	884	22	527	337	22	0
Totals	199,314	24,533	4650	475	2563	1603	217	20

Table 7. Eel habitat, South Atlantic region (North Carolina to Florida)

Huc4 Number and Watershed Name	**Historical length (km)**	**Current Length (km)**	**No. of Dams**	**Dams <10 ft.**	**Dams 10-24 ft.**	**Dams 25+ ft.**	**Hydro Electric**	**Nav.**
301 Chowan-Roanoke Coastal Dr.	36,775	3,632	371	3	257	230	15	0
302 Neuse-Pamlico Coastal Dr.	23,324	12,452	445	6	268	149	1	0
303 Cape Fear Coastal Dr.	20,471	5,990	626	5	385	226	9	3
304 Pee Dee Coastal Dr.	35,880	6,139	1034	58	637	333	10	0
305 Edisto-Santee Coastal Dr.	41,504	7,003	1942	52	1073	810	66	0
306 Ogeechee-Savannah Coastal Dr.	34,604	4,508	1028	33	546	447	30	1
307 Altamaha-St. Marys Coastal Dr.	37,172	4,673	1353	31	763	559	10	0
308 St. Johns Coastal Dr.	82,334	6,582	40		18	19	0	4
309 Southern Florida Coastal Dr.	8,044	4,893	105	6	46	45	0	0
Totals	246,008	55,872	6944	194	3993	2818	141	8

In the assessment of the Atlantic Coast watersheds, the St. Lawrence River - Lake Ontario watershed was included. However, data were incomplete because only the United States' side of the Lake Ontario basin was assessed. Construction of the Moses Saunders Dam (1954-58) impeded upstream and downstream migration on the St. Lawrence River, restricting access by migratory fish from the Atlantic Ocean to Lake Ontario and the Finger Lakes system. In 1974, an eel ladder was constructed, which probably reduced the effects of the lack of upstream passage at the Moses Saunders Dam. The number of American eel ascending the ladder has decreased dramatically in recent years (see Figure 4).

While a number of American eel have utilized the Saunders eel ladder, an assessment of the percent passed to the total number of eel in the system has not been conducted. It is unknown whether the number currently passed is sufficient to sustain the Saint Lawrence River/Lake Ontario stock.

In the U.S. portion of the watershed, 455 dams result in 24,693 km of stream habitat lost or restricted from a total of 30,085 km (82% loss) to migratory fish originating in or having Lake Ontario as their destination (Table 8). Since dams on the St. Lawrence River hinder fish movement through the St. Lawrence River to and from the Atlantic Ocean, the total kilometers of stream access lost or restricted in the Lake Ontario and St. Lawrence River watershed is actually much larger.

Table 8. Eel habitat, Great Lakes region1 (New York and Ontario to Quebec)

Huc4 Number and Watershed Name	Historical length (km)	Current Length (km)	Number of dams	Dams <10 ft.	Dams 10-24 ft.	Dams 25+ ft.	Hydro-Electric	Nav.
412 Eastern Lake Erie Drainage	113	66	4	0	1	3	3	0
413 Southwestern Lake Ontario Drainage	8,076	1,827	67	7	45	15	9	1
414 Southeastern Lake Ontario Drainage	16,156	2,877	159	33	74	52	19	15
415 Lake Ontario-St. Lawrence Drainage	5,740	622	225	24	118	83	150	2
Totals	30,085	5,392	455	64	238	153	181	18

The dam database used by Busch et al. (1998) included information on dam heights (Tables 5-8). It identified 3,512 dams in the North Atlantic Region of which 448 are less than 10 ft. high, 2,260 are between 10 and 24 ft. high, and 813 are higher than 25 ft. Of all the dams, 561 are used for hydropower production. The Mid-Atlantic Region has 4,650 dams of which 475 are less than 10 ft. high, 2,563 are between 10 and 24 ft. high, and 1,603 are higher than 25 ft. And, 217 dams are used for hydropower production. In the South Atlantic Region, the 6,944 dams identified included 194 that are less than 10 ft. high, 3,993 between 10 and 24 ft., and 2,818 higher than 25 ft. Of the dams in this region, 141 are used for hydropower production. Dams in the US Lake Ontario basin include 64 that are less than 10 ft. high, 238 that are 10-24 ft. high, and 153 that are 25 ft. or higher. Hydropower production was the use identified for 181 dams.

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1 No Canadian data were available, therefore, data presented are only from the U.S. side of Lake Ontario.

Various factors influence successful upstream or downstream migration of American eel past dams. Busch et al. (1998) evaluated fish migration restrictions due to dams by examining limited data on the presence or absence of American eel above and below dams. The preliminary results indicate that although height and use (purpose) for the facility appear to be important factors, other criteria need to be evaluated including slope, construction material, water flow, location of the dam in the watershed, and operational procedures.

Dams that require special licenses such as for hydropower production or navigation provide opportunities for fish passage if required by the resource management agencies. However, only 1,100 were identified for hydropower production and 50 for navigation out of the total number of 15,570 identified dams. Therefore, only 7% of these dams are covered by regulatory programs that could provide fish passage. The other specific uses for dams identified in the database include water-level control, water supply, and recreation.

Downstream passage to the American eel's historic habitat is just as important as successful upstream access. Therefore, turbine-induced mortality during downstream migration needs to be resolved since it impacts prespawning adult silver eel. Investigations have found turbine-induced mortality of eel to range from 5 to 60%, depending on the flow through the turbines and on the length of the fish (Hadderingh 1990; McCleave Person. Comm.). Experiments using lights to deflect American eel from water intakes into bypass areas have been successful at some hydroelectric power stations (Hadderingh 1990). The reduced numbers of American eel which currently utilize Lake Ontario tributaries, such as the Oswego River, presumably move upstream via the locks and require downstream passage in order to reach Lake Ontario. Haro (1996) also provides information on various methods of mitigating turbine entrainment and mortality by diverting eel around turbine intakes to bypass entrances during downstream migration. Experiments carried out using behavioral mitigation techniques such as strobe lighting have shown some success in diverting eel from turbine intakes. Other behavioral methods such as water and air jet curtains and weak electric fields have not shown similar success (Richkus and Whalen 1999). Research on mechanical mitigation devices such as angled bar racks, louvers, and screens has provided mostly inconclusive although insightful results that might warrant further research (Richkus and Whalen 1999).

1.4.1.2.3.4 Quality

Temperature: American eel are capable of tolerating a wide range of physiochemical conditions. Elvers have been found in waters as low as -0.8 ^{o} C (Jeffries 1960). Yellow eel held at less than 5 ^{o} C for over 5 weeks stopped feeding and reduced their oxygen consumption (Walsh et al. 1983). Yellow eel are known to hibernate in the mud during the winter (Fahay 1978). Preferred summer temperatures have been reported at 17.4 ± 2 ^{o} C for yellow eel (Karlsson et al. 1984). American eel are apparently capable of surviving short-term thermal shocks. American eel have been reported to survive passage through a nuclear power plant, during which they were exposed to elevated temperatures for 1 to 1.5 h (Marcy 1973).

Salinity: Little work has been done on the salinity requirements of American eel. The leptocephali have been reported to be in near-ionic equilibrium with sea water (Hulet et al. 1972). Elvers are known to delay their upstream migration at the freshwater brackish interface which is believed to permit some physiological adjustments to the new freshwater regime (Sorensen and Bianchini 1986). Yellow eel occupy niches in freshwater and brackish regimes. Silver eel migrate from freshwater to the open ocean. From the above, postlarval American eel appear to be euryhaline.

1.4.1.3 HABITAT ISSUES

Habitat includes the physical, chemical and biological setting and requirements needed to support all life functions of American eel.

Spawning Areas

Spawning takes place in the Sargasso Sea. The specific location(s) and the specific habitat characteristics in this 5.2 million km^2 (2 million $miles^2$) area have not been reported. Loss of spawning habitat would result in significant impacts on American eel. Threats to American eel populations and spawning habitat include sea level rise / land subsidence, and contaminants. Global warming and the subsequent rise in sea level could adversely affect American eel spawning activities. Sea level is predicted to rise above current levels by approximately 50 centimeters to 1 meter by the year 2100 (Oerlemans 1989, Titus et al 1991). The effects on this sea level rise on the currents and oceanic conditions that conduct larval migration are completely unknown. Land subsistence along the Atlantic Coast adds to the effect of sea level rise, resulting in an increase of 25-30 centimeters greater than the global average (Hull and Titus 1986). Such an increase could fundamentally alter current eel habitat. In addition, American eel accumulate significant amounts of contaminants in reproductive tissue. Thus, the potential to impair reproduction, if contaminants are not carefully monitored in important eel habitats.

Feeding and Growth Areas

Data from commercial harvest records for elvers/glass eel, yellow eel and stock assessments indicate that eel are found in most types of habitat including the offshore, mid-water and bottom areas of estuaries, embayments, rivers, streams, and lakes. However, eel are found to be most prevalent in the nearshore, shallow embayments and tributaries (Adams and Hankinson 1928; Facey and LaBar 1981; Helfman et al. 1983; GLFC 1996; NYSDEC 1997a & b).

American eel are classified as a warmwater species (Adams and Hankinson 1928) that are most abundant in relatively warm streams and shallow lakes or embayments (Ogden 1970), while relatively scarce in deep, steep gradient cold-water lakes (Smith and Saunders 1955). Limited work on preferred freshwater habitats indicates both lentic and lotic habitats are used and growth appears to be related to density and availability of food (Krueger and Oliveira 1999). Stream use appears to be important to elvers (Bigelow and Schroeder 1953) and yellow eel.

Issues and Concerns

Various habitat stresses and losses impact American eel abundance, health, distribution, and growth rates (Lary and Busch 1997; Richkus and Whalen 1999). These impacts have not been adequately described. Furthermore, since habitat management is also the responsibility of agencies other than the primary participants in the ASMFC, habitat issues need to be addressed through interagency coordination and other avenues (i.e., legislation, policy, enforcement, etc.).

Channel dredging and overboard spoil disposal are common throughout the Atlantic coast, but currently have unknown effects on American eel. Changes in salinity as a result of dredging projects could alter American eel distribution. Additionally, dredging associated with whelk and other fisheries may damage American eel benthic habitat; however, the significance of this impact also remains unknown.

Although pollution has the potential to adversely impact all the life stages of American eel, there are no data to suggest unusual sensitivity by American eel to urban or agricultural contaminants (e.g., pesticides and herbicides). However, due to their longevity and habitat use, high levels of contaminants have been reported in eel (Hodson et al. 1994). Additional information needs to be obtained to determine the impacts of contaminants on American eel. Also a new, specific area of concern deals with coastal wetlands and the potential impact caused by spraying insecticides for mosquito control at the time glass eel enter these areas. Potential impacts from contaminants include mortality, changes in behavior, and decreases in fecundity.

2.0 GOALS AND OBJECTIVES

2.1 SPECIFICATION OF MANAGEMENT UNIT

The specific "management unit" for this Fishery Management Plan is defined as that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

Significant numbers of eel use areas/habitats that are outside the jurisdictional boundaries of the state agencies participating in the ASMFC. These include watersheds in the Canadian Atlantic Provinces, upstream freshwaters reaches that are managed by inland fish and wildlife agencies of ASMFC member states and regional institutions such as the Gulf States Marine Fisheries Commission, and those waters within Native American Reservations where Tribal Governments have jurisdiction. U.S. eel management needs to proactively include and coordinate the interests and approaches of the ASMFC with applicable jurisdictions/agencies in order to implement holistic management, including protection and enhancement of this species.

Since all eel reproduction occurs in the Sargasso Sea (Figure 2), the health and availability of this area to support reproduction is of significant importance. Activities impacting the health of the Sargasso Sea and reproductive success of eel, although outside direct management of the ASMFC, need to be addressed through other applicable authorities. The Secretary of Commerce and the National Marine Fisheries Service may take complementary management action in the Exclusive Economic Zone, as per the recommendations in Section 4.2.2.

The Goals of the Fishery Management Plan for American Eel are to:

1. Protect and enhance the abundance of American eel in inland and territorial waters of the Atlantic States and jurisdictions and contribute to the viability of the American eel spawning population; and

2. Provide for sustainable commercial, subsistence, and recreational fisheries by preventing overharvest of any eel life stage.

Primary Objectives

1. Improve knowledge of eel harvest at all life stages through mandatory reporting of harvest and effort by commercial fishers and dealers, and enhanced recreational fisheries monitoring;
2. Increase understanding of factors affecting eel population dynamics and life history through increased research and monitoring;
3. Protect and enhance American eel abundance in all watersheds where eel now occur;
4. Where practical, restore American eel to those waters where they had historical abundance but may now be absent by providing access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel; and
5. Investigate the abundance level of eel at the various life stages, necessary to provide adequate forage for natural predators and support ecosystem health and food chain structure.

Long-Term Objectives

A Encourage protection of eel spawning, nursery and growth habitats with and/or through the agencies having jurisdiction over these areas;
B Protect and enhance inland and coastal water quality to protect the health of the eel population and to reduce bioaccumulation of toxic substances; and
C Coordinate harvest and abundance monitoring with resource management agencies outside the East Coast of the U.S.

3.0 MONITORING PROGRAM SPECIFICATIONS/ELEMENTS

The American Eel FMP encourages all state fishery management agencies to pursue full implementation of the Atlantic Coastal Cooperative Statistics Program (ACCSP), which will meet the monitoring and reporting requirements of this FMP. The American Eel FMP recommends a transition or phased-in approach be adopted to allow for full implementation of the ACCSP. Until such time as ACCSP is implemented, the American Eel FMP encourages state fishery management agencies to initiate implementation of specific ACCSP modules, and/or pursue pilot and evaluation studies to assist in development of reporting programs to meet the ACCSP standards (please refer to the ACCSP Program Design document for specific reporting requirements and standards; Contact - Joe Moran, ASMFC). The ACCSP partners are the 15 Atlantic coastal states (Maine – Florida), the District of Columbia, the Potomac River Fisheries Commission, the National Marine Fisheries Service, the U.S. Fish and Wildlife Service, the three fishery management Councils, and the Atlantic States Marine Fisheries Commission. Participation by program partners in the ACCSP does not relieve states from their responsibilities in collating and submitting harvest/monitoring reports to the Commission as may be required under this FMP.

Management of American eel will be based on scientific advice provided by the scientific community, as well as input from public hearings and from the Advisory Panel. Management will strive for a long-term viable population, supporting fisheries and inter-dependent wildlife populations. Effective management will require monitoring population abundance at various life stages, monitoring fishing mortality (harvest and incidental), preventing habitat degradation, restoring fish habitat, as well as identifying and supporting research. The measures outlined below are designed to facilitate the management process. As new data become available and new assessment data provide new perspective, management elements will adapt in order to most effectively reach the goals and objectives.

3.1 ASSESSING ANNUAL RECRUITMENT

Little is known about annual recruitment of American eel. Although maximum fecundity can be estimated, natural larval mortality is estimated to be substantial. The number of larvae that survive to reach the coastal areas each year and transform to glass eel is unknown. Also, the annual variation in recruitment to elvers or yellow eel is unknown, as is the number that survive to sexual maturity. Because American eel are slow maturing and long-lived, current juvenile indexing techniques have limited applicability in describing the annual abundance and variations in the abundance of respective cohorts. This is due to the variability in age/length relationships, and therefore similar size classes of eel will include a number of year classes. Resolution of the aging issue requires further investigation and validation of techniques used for age determination, as is mentioned in Section 6 “Information and Research Needs.” Additional information regarding larval and juvenile survival is essential to assessing annual recruitment. Monitoring abundance of American eel for each of the defined life stages will be necessary for the establishment of multiple recruitment indices.

3.1.1 Annual Young-of-Year Abundance Survey

The glass eel and elver (young-of-year) life stages provide the most unique opportunity to assess the annual recruitment of each year’s cohort since young-of-year result from the previous winter’s spawning activity, and hence are all the same age. Known age is an attractive feature of the young-of-year life stage, which has shown to be problematic with all older life stages. Therefore, a fishery independent young-of-year abundance survey is proposed in accordance with the options provided below.

Measurement of young-of-year abundance is considerably cost effective since the gear required is inexpensive to purchase or manufacture, requires no additional expense for bait, and may be operated by relatively few persons. Also, since the young-of-year life stage and period of recruitment onto the Atlantic coast is short in duration, each annual assessment of young-of-year abundance would not amount to a long commitment of staff time.

Data from a young-of-year abundance survey could provide a barometer with which to gauge the efficacy of management action, given due consideration to the factors which affect spawning, larval survival, transport, metamorphosis, and subsequent recruitment of young-of-year onto the Atlantic coast. Young-of-year abundance indices may also provide a basis of inference for the future abundance of each year's cohort, similar to abundance indices validated for other fish species.

Accordingly, states/jurisdictions will conduct annual fishery-independent surveys for young-of-year American eel. Each participating jurisdiction shall deploy appropriate gear to capture young of the year at a minimum of two locations over a six-week period. A variety of gear types are available for use, and states should use the gear most suitable to the habitat and geography within their jurisdiction. The cost of most gear ranges from $200 to $400 per unit.

The timing and placement of the young-of-year sampling gear will coincide with those periods of peak onshore migration of young-of-year. The locations selected will be those previously shown to catch young-of-year American eel and should provide as wide a geographic distribution as possible. Initially, stock assessment biologists may need to alter the timing and placement of the sampling gear in order to determine peak migration period and locations for the annual survey. Thereafter, standard stations and procedures will remain fixed.

At a minimum, the gear will be set so that they are operational during periods of rising or flood tides occurring at nighttime hours. During these conditions, gear will be checked as often as possible and emptied of their catch. The catch will be sorted and all specimens identified to their lowest taxonomic order, measured, weighed and enumerated as appropriate. Species which appear to be predators of young-of-year will be denoted. The entire catch of young-of-year will be weighed and counted, and each individual measured for total length. The number of young-of-year per unit weight (gram) will be determined for each catch examined. Standard statistical techniques (sub-sampling) will be used in instances where the catch of young-of-year is too large (i.e., several hundred individuals or more) to warrant a complete census.

In addition to the catch and by-catch of young-of-year, various environmental and climatological data will be recorded for each catch. These will include date, water and air temperatures, salinity, tide stage, and soak time. Notation of wind speed, direction and precipitation will be recorded. Also, a subjective judgement of the condition of the gear at the time of sampling will be made on an ordinal scale of one to four, with one equal to good, two equal to fair, three equal to poor, and four equal to void or unsuitable for indexing. The judgement will relate to the condition the gear was found in relation to the condition it was left in the previous day. Young-of-the year captured at or near obstructions should be released upstream of these obstructions whenever possible.

All states/jurisdictions, except those exempted by the Management Board, are required to conduct an annual young-of-year abundance survey, beginning in the year 2000, as described above. The Technical Committee shall advise the Management Board on exemptions as necessary. Those states that are initially exempted will be required to conduct the annual young-of-year survey by the year 2001. States shall submit proposals for instituting their surveys as per Section 5.1.2.

3.1.2 Annual Report of Harvest or Catch Per Unit of Effort

A catch per unit effort (CPUE) reporting requirement will be initiated by every state, if not already required, in order to develop abundance indices for each life stage (see Section 3.4.1 for mandatory reporting requirements).

3.2 ASSESSING SPAWNING STOCK BIOMASS

The annual spawning stock biomass for American eel populations along the Atlantic Coast is unknown. NMFS landings data provide limited estimates of silver eel harvest: 423 tons to 1,813 tons were harvested between 1970 to 1995 from the Atlantic coast. The New England and Mid-Atlantic regions of the Atlantic Coast produce the majority of the American eel commercial harvest. However these data are of limited use due to inadequate sampling of inland harvest areas and dealer locations. Also, since the harvest data from a number of inland and marine agencies may include a number of species and an unknown ratio of mature (silver) and maturing (yellow) eel, the current fishery dependent data are inadequate to describe the annual abundance and variations in abundance between years. In short, any estimate of abundance or population trends based on existing harvest data is questionable because of inconsistent reporting requirements across jurisdictions. Furthermore, fishery independent abundance data are generally lacking.

3.2.1 Fishery-independent monitoring of adults/sub-adults.

The silver or migratory stage of American eel provides an opportunity to monitor the abundance of the spawning stock. Although these fish will be of various sizes and ages, they are on their way to reproduce and will jointly contribute to the abundance of the next cohort. Therefore, the fishery independent reporting of emigrant counts, should be maintained, standardized, and expanded. In addition, certain ongoing/recent state surveys for eel abundance and distribution may be useful for fishery-independent monitoring of silver and yellow eel populations.

3.3 ASSESSING MORTALITY

American eel mortality has three components: natural, fishing, and incidental. Natural mortality includes factors such as predation and disease; fishing mortality includes harvest and bycatch; incidental mortality includes anthropogenic impacts from fish passage (for example through hydroelectric turbines), chemical spills or hazardous chemical exposures.

A sustainable mortality rate will allow for a certain level of harvest and incidental losses while still maintaining a viable spawning stock biomass. This rate has not been calculated for eel because of the difficulty in obtaining abundance data (population and harvest) by age throughout the species' range. Combined mortality at all life stages in salt and fresh water is largely responsible for controlling the population size of American eel across its range.

3.3.1 Natural Mortality

Although not documented, natural mortality is presumed to be very high at the leptocephalus stage, glass eel and elver stages due to the high fecundity of the species. This notion is based on the high fecundity (Wenner and Musick 1974; Barbin and McCleave, 1997) of this species. Natural mortality for yellow and silver eel also lack documentation.

3.3.2 Fishing Mortality

Fishing mortality has two components: directed fishing mortality (e.g., intentional harvest) and non-directed mortality (e.g., by-catch). Although reported commercial landings data show a continuing decrease in harvest since the late 1970's, changes in fishing effort or mortality rates are not available. This situation will be addressed through the implementation of the harvest reporting requirements outlined in Section 3.4.1, and the ability to use consistent harvest data in future stock assessments.

The amount of American eel bycatch in commercial and recreational fisheries remains unknown. Additional information will be required to determine the impact of bycatch. It is likely that bycatch of American eel are commonly discarded in the recreational fishery and unreported in total harvest. Bycatch for American eel should be quantified within a bycatch-monitoring module of the Atlantic Coastal Cooperative Statistics Program (ACCSP).

3.3.3 Incidental Mortality

As defined in this FMP, incidental mortality is also caused by anthropogenic activities other than harvest. Activities include damming (e.g., impingement, entrainment, and turbine caused injury) navigation locks (e.g., impingement, entrainment), industrial/municipal water intakes (e.g., impingement, entrainment), and those caused by chemicals (drastic salinity changes, spills, point source releases, and non-point source releases such as the application of insecticides in glass eel nursery areas). Accumulated contaminants may impact individuals directly as well as egg viability and larval survival. Compression of range through habitat restrictions may increase the significance of predation mortality.

More research is needed on the extent and impact of incidental mortality in order to improve future stock assessments. See Section 6.3 for related research recommendations.

3.4 SUMMARY OF MONITORING PROGRAMS

Numerous state and federal agencies, universities, and private organizations are involved in data collection programs to directly determine American eel population status. While existing monitoring programs may be useful in identifying general trends within specific areas if consistent data have been collected, each is complicated by factors that may bias the data, such as sampling error, inappropriate equipment, or incomplete sampling effort. Most existing fishery dependent and independent monitoring programs lack a comprehensive data collection goal.

The goal of a comprehensive American eel monitoring program is to produce the data needed to obtain an accurate assessment of the American eel population for making management decisions. States must improve the reporting of eel harvest data by gear, season, and harvest effort and life stage, as well as fishery-independent data.

In order to collect information to support accurate management decisions, a comprehensive monitoring plan must be developed. Such monitoring efforts should be standardized and be conducted in each of the cooperating states within the ASMFC. Fishery-dependent reporting requirements will include pounds landed, harvest method, gear, season, effort, and life stage (see

Section 3.4.1). In addition, the NMFS Marine Recreational Fisheries Statistics Survey (MRFSS) and state surveys should be utilized to collect catch, harvest, and biological information regarding recreational and subsistence fisheries for American eel. States/jurisdictions are encouraged to fund expansion of the survey inland, where significant recreational fisheries for catadromous and anadromous fish are reported to occur. Lack of such information could have serious consequences in the assessment of the American eel stock. Wherever practical, state harvest reporting requirements will coincide with the current and future mandates of the ACCSP. Reporting elements not covered by the ACCSP should be covered by annual reports submitted in conjunction with this FMP.

3.4.1 Annual State Report on Regulations, Harvest, Bycatch and Fishery-Independent Surveys for American Eel.

Each state/jurisdiction shall be required to submit an annual report (in accordance with Section 5.1.2) detailing that state's regulations, catch, harvest, bycatch, fishery dependent and independent surveys, and characterization of other losses for American eel. The report will address each of the topics listed below.

1. Commercial fishery
    a. Synopsis of regulations in place
    b. Estimates of directed harvest, by month, by region as defined by the states
        1. Pounds landed by life stage and gear type (defined in advance by ASMFC)
        2. Biological data taken from representative sub-samples to include sex ratio and age structure (for yellow/silver eels), length and weight if available
        3. Estimated percent of harvest going to food versus bait
    c. Estimates of export by season (provided by dealers)
    d. Harvest data provided as CPUE (by life stage and gear type)
    e. Permitted catch for personal use, if available
2. Recreational fishery
    a. Synopsis of regulations in place
    b. Estimate of recreational harvest by season (if available)
        1. Biological data taken from representative sub-samples to include sex ratio, age structure, length and weight (if available)
3. Fishery-independent monitoring
    a. Results of the Annual Young-of-Year Abundance Survey (unless exempt)
    b. Description of other fishery-independent surveys performed (methods, location, etc.) and results (if required in FMP)
    c. Projects planned for next five years
4. Characterization of Other Losses

To the extent possible states/jurisdictions should attempt to characterize the losses of American eel, in number and weight by life stage or age, due to factors other than commercial and recreational fisheries. Such losses may include, but are not limited to the following:

a. Impingement/entrainment mortalities of eel at power generation facilities, water intakes, and navigation locks
b. Bycatch mortalities in commercial and recreational fisheries
c. Confiscated poundage from illegal or undocumented fisheries (i.e., poaching)

d. Scientific losses (i.e., samples collected for contaminants analysis, other studies)
e. Mass mortalities of eel due to disease, spills or other causes

Commercial Catch and Effort Data Collection Programs

The ACCSP commercial data collection program will be a mandatory, trip-based system with all fishermen and dealers required to report a minimum set of standard data elements (refer to the ACCSP Program Design document for details). Submission of commercial fishermen and dealer reports will be required after the 10th of each month.

Any marine fishery products landed in any state must be reported by a dealer or a marine resource harvester acting as a dealer in that state. Any marine resource harvester or aquaculturist who sells, consigns, transfers, or barters marine fishery products to anyone other than a dealer would themselves be acting as a dealer and would therefore be responsible for reporting as a dealer.

Recreational Catch and Effort Data Collection Programs

The ACCSP recreational data collection program for private/rental and shore modes of fishing will be conducted through a combination telephone and intercept survey. Recreational effort data will be collected through a telephone survey with random sampling of households until such time as a more comprehensive universal sampling frame is established. Recreational catch data will be collected through an access-site intercept survey. A minimum set of standard data elements will be collected in both the telephone and intercept surveys (refer to the ACCCSP Program Design document for details). The ACCSP will implement research and evaluation studies to expand sampling and improve the estimates of recreational catch and effort.

For-Hire Catch Effort Data Collection Programs

The ACCSP is conducting an evaluation study to determine the best method(s) of data collection for for-hire fisheries. A minimum set of standard data elements will be collected in all for-hire catch/effort surveys (refer to the ACCSP Program Design document for details).

Discard, Release, and Protected Species Interactions Monitoring Program

The ACCSP will require a combination of quantitative and qualitative methods for monitoring discard, release, and protected species interactions in commercial, recreational, and for-hire fisheries. Commercial fisheries will be monitored through an at-sea observer program and several qualitative programs, including strandings, entanglements, trend analysis of logbook reported data, and port sampling. Recreational fisheries will be monitored through add-ons to existing intercept surveys and additional questions added to the telephone survey. For-hire fisheries will be monitored through an at-sea observer program and several qualitative programs (refer to the ACCSP Program Design document for details).

3.4.2 Biological Information

The ACCSP will require the collection of baseline biological data on commercial, for-hire, and recreational fisheries. Biological data for commercial fisheries will be collected through port sampling programs and at-sea observers. Biological data for recreational fisheries will be collected in conjunction with the access-intercept survey. Biological data for for-hire fisheries will be collected through existing surveys and at-sea observer programs. A minimum set of standard data elements will be collected in all biological sampling programs (refer to the ACCSP Program Design document for details). Priorities and target sampling levels will be determined by the ACCSP Biological Review Panel, in coordination with the Discard/Release Prioritization Committee.

3.4.3 Social and Economic Information

Commercial Fisheries

The ACCSP will require the collection of baseline social and economic data on all commercial fisheries (refer to the ACCSP Program Design document for details). A minimum set of standard data elements will be collected by all social and economic surveys (refer to the ACCSP Program Design document for details).

Recreational Fisheries

The ACCSP will require the collection of baseline social and economic data on all recreational fisheries through add-ons to existing recreational catch/effort surveys (refer to the ACCSP Program Design document for details). A minimum set of standard data elements will be collected in all for-hire catch/effort surveys (refer to the ACCSP Program Design document for details).

3.4.4 At-Sea Observer Program

The ACCSP at-sea observer program is a mandatory program. As a condition of state and/or federal permitting, vessels should be required to carry at-sea observers when requested. A minimum set of standard data elements will be collected through the ACCSP at-sea observer program (refer to the ACCSP Program Design document for details). Specific fisheries priorities will be determined by the Discard/Release Prioritization Committee.

3.4.5 Vessel Registration System

The ACCSP has recommended the development of a standardized national fishing vessel registration system (VRS) through upgrades and expansions of the current Vessel Identification System (VIS). The VIS is an integration of the Coast Guard documentation and individual state registration systems. A minimum set of standard data elements will be collected through the VIS (refer to the ACCSP Program Design document for details).

4.0 MANAGEMENT PROGRAM IMPLEMENTATION

Management of American eel will be based on scientific advice provided by the Technical Committee, as well as input from public hearings and the Advisory Panel. In general, management will strive for a long-term sustainable population, with a surplus to support recreational, subsistence and commercial fisheries.

Each state must implement the required management measures and should protect American eel habitat within its jurisdiction to ensure the viability of the population segment residing within its boundaries. States must work with Native American tribal nations and other management jurisdictions within their boundaries in the management of American eel resources.

4.1 RECREATIONAL FISHERIES MANAGEMENT MEASURES

Currently there are observed but undocumented recreational fisheries for American eel. The harvest rate is unknown, as is the discard mortality rate of the bycatch of American eel from recreational fisheries for other species.

In order to minimize the chance of excessive recreational harvest, as well as circumvention of commercial eel regulations, the ASMFC member states/jurisdictions shall establish uniform possession limits for recreational fisheries of a six inch minimum size and a possession limit. Recreational anglers may possess no more than 50 eels per person, including crew members involved in party/charter (for-hire) employment, for bait purposes during fishing. Recreational fishermen will not be allowed to sell eel without a State license permitting such activity.

4.2 COMMERCIAL FISHERIES MANAGEMENT MEASURES

States shall institute licensing and reporting mechanisms to ensure that annual effort (including total units of gear deployed) and landings information by life stage (glass eel/elver, yellow eel, and silver eel) are provided by harvesters and/or dealers. In addition, the ACCSP will require a comprehensive permit/license system for all commercial dealers and fishermen.

4.2.1 Management Measures

States/jurisdictions shall maintain existing or more conservative American eel commercial fishery regulations, including gear specifications contained in Table 2, for all life stages. States with minimum size limits for commercial eel fisheries shall retain those minimum size limits, unless otherwise approved by the American Eel Management Board. The provisions listed within this paragraph are considered a compliance requirement and are effective immediately upon adoption of the FMP by the ASMFC.

Management measures include all mandatory monitoring and annual reporting requirements as described in Sections 3.4.1 and 5.1.2. Specifically**,** harvest, effort, and biological information shall be provided as per Section 3.4 for each life stage exploited in each jurisdiction. Wherever practical, monitoring requirements in Section 3.4.1 are consistent with current and future mandates of the Atlantic Coastal Cooperative Statistics Program (ACCSP). Monitoring elements not covered by ACCSP must still be covered by state agencies and reported as per Section 3.4.1. States may also propose alternative management programs as per Section 4.4.

4.3 HABITAT CONSERVATION AND RESTORATION

Protection of habitat such as nursery area is critical to the continued survival of American eel. Each state should identify, categorize, and prioritize important and historic American eel habitat within areas of its jurisdiction. Periodic monitoring should be designed and implemented to ensure the long-term viability of essential American eel habitat.

Barriers restrict or prevent migration into current and historical habitat, thereby, reducing total production. Successful upstream and downstream fish passage past barriers is essential to ensuring maximum spawning stock biomass of emigrating silver eels from the U.S. Atlantic coast (Lary and Busch, 1997).

In areas where residential and commercial development is adjacent to American eel habitat, state marine fisheries agencies should coordinate efforts with their inland fisheries/wildlife agencies and others (for example, state agencies with responsibility for soil and water conservation and water quality) to implement remedial actions to restore habitat. State marine fisheries agencies should also coordinate with their state water quality agencies responsible for developing and implementing river basin and wetland restoration plans, to ensure that American eel habitat is identified and considered in these plans, and that these plans are implemented. Also, state marine fisheries agencies should coordinate their concerns with the Army Corps of Engineers since they have authority to investigate, study, modify, and construct projects for habitat restoration, under Section 1135(b) of the Water Resources Development Act of 1986, and also under Section 206 of this same Act.

State marine fisheries agencies should coordinate with their state inland fisheries/wildlife agencies to identify migration times, through site-specific data collection and monitoring. This information should be used to provide comment to permitting agencies regarding seasonal restrictions on activities that may disturb or retard eel migration and feeding behaviors. Construction activities should be avoided in critical migration periods. However, the specific seasonal restriction dates for any particular area should be based on site-specific data and appropriate monitoring. States should consider obtaining land adjacent to critical migration corridors and staging areas to ensure their long-term protection. Protection of American eel habitat or areas of particular concern should be pursued through acquisition, deed restrictions, or conservation easements. State fisheries agencies should also work with their state soil and water conservation agencies and/or agricultural agencies to provide information on these habitats, to be used in their decisions regarding the state’s riparian buffer program.

4.3.1 Preservation of Existing Habitat

Sargasso Sea

State marine fisheries agencies should be proactive in identifying opportunities to protect the health of the Sargasso Sea area through partnerships with NOAA and NMFS, including the implementation of the SAFMC’s Fishery Management Plan for Pelagic *Sargassum* Habitat of the South Atlantic Region (SAFMC 1998).

4.3.2 Habitat Restoration, Improvement, and Enhancement

Reestablishment of Eel into Historic Habitats

ASMFC participating states/jurisdictions marine fisheries agencies are encouraged to collaborate with their sister inland management agencies, as well as with other Federal and State agencies, and Native American governments to mitigate to the extent possible the effects of various hazards to the upstream and downstream migration of American eel. Such mitigation should include, but not be limited to support of fish passage research, requirements for the construction of fish (eel) passage facilities upon construction of dams, power generating facilities and relicensing of same, and outright removal of identified hazards to eel passage.

Upstream passage

State marine fisheries agencies should cooperate with their inland fisheries/wildlife agencies and the USFWS to improve access to upstream reaches of streams currently restricted by dams with no ladders, helping to increase access to more habitat for feeding and growth. Although it is often assumed that navigation locks will provide unhindered upstream access for eel, this is not a proven, effective passageway due to the great fluctuations in water flow during lock operation (Lary and Busch 1997). Trap and truck methods have also been suggested as a process for eel passage. This has not been adequately evaluated as to effectiveness or the impact on the species, such as changes in the natural selection process. However, trap and transport of glass eels and elvers could be a cost effective, short-term method of upstream passage if it involved volunteers or harvesters who returned a portion of their glass eel/elver catch upstream of impassable blockages.

Downstream passage

State and federal agencies should investigate changes in turbine design to improve downstream fish passage and continue efforts to direct eel away from turbine passage to other higher survival passage opportunities. Investigations should also include feasibility of dam shut-downs during off-peak/night time hours to encourage passive escapement of migrating adult eels.

Monitor enhancement efforts

State and federal agencies should monitor and report on the amount of habitat opened through upstream passage projects and any associated changes in emigrating eel abundance. Passability of blockages for different size classes of eels should also be evaluated.

4.3.3 Avoidance of Incompatible Activities

Each state should establish windows of compatibility for activities known or suspected to adversely affect American eel life stages and their habitats (e.g. dredging, filling, aquatic construction) as well as notify the appropriate construction or regulatory agencies in writing.

Projects involving water withdrawal from important habitats (e.g. feeding grounds) should be scrutinized to ensure that adverse impacts resulting from impingement, entrainment, and/or modification of flow, temperature and salinity regimes due to water removal will not adversely impact American eel in any life stage.

Each state which contains growth areas within its jurisdiction should develop water use and flow regime guidelines which are protective of American eel habitat and which will ensure to the extent possible the long-term health and sustainability of the stock. States should endeavor to ensure that proposed water diversions/withdrawals from rivers tributary to important habitats will not reduce or eliminate conditions favorable to American eel which make use of these areas.

4.3.3.1 Contaminants

American eel accumulate high concentrations of contaminants, potentially causing increased incidence of reproductive impairments. In the St. Lawrence River migrating silver eel, vertebral malformations and basophilic foci (lesions) in the liver were found to be most common in contaminated eel, while nematodes were present in American eel that were less contaminated (Couillard, et. el. 1997). Another study found that the highest concentrations of chemicals were found in the gonads (Hodson et.al. 1994).

Documentation of American eel being used as an indicator species for contaminant levels could not be found. Little work has been done on the effects of pollutants and the tolerance limits of American eel (Facey and Van Den Avyle 1987). Toxicity studies of aquacultural chemicals effects on the various life stages of the American eel suggest increased tolerance with size and age (Hinton and Eversole 1978, 1979, 1980). However, an accidental release of toxins into the Rhine River in 1986 killed hundreds of thousands of European eel (Facey and Van Den Avyle 1987). American eel tend to bioaccumulate heavy metals endemic to their freshwater habitat (Moreau and Barbeau 1982). Apparently, they also bioaccumulate other toxins as well. In 1976, New York's Departments of Health and Environmental Conservation banned the sale and possession of American eel taken from Lake Ontario and the Hudson River because of excessive polychlorobiphenyls (PCB) levels (greater than the legal limit of 2 ppm). Hudson River American eel were reported to have from 50 to 75 ppm and the Lake Ontario eel had 2.5 to 4.5 ppm of PCB’s (Blake 1982). American eel are apparently sensitive to hypoxia and have been reported to select waters with high oxygen tensions (Hill 1969, Sheldon 1974). Tesch (1977) wrote, " the eel survives better in air than in poorly oxygenated or polluted water.” American eel are especially susceptible to the accumulation of toxic compounds because of their long residence in aquatic habitats and their accumulation of lipids prior to migration. The impact of these toxic compounds on the American eel themselves has not been studied. However, these compounds can pass through the food chain and accumulate in human and wildlife consumers of American eel where they can increase the risk of cancer or interfere with normal reproduction. Furthermore, while clearly posing some risk to all consumers, the bioaccumulation of contaminants is a particularly critical issue to subsistence users of American eel, such as Native American tribes. This is because such user groups likely consume fish at far higher rates than either recreational fishers or individuals that purchase and consume American eels from commercial sources. Clearly, maintaining good water quality is important for maintaining the health of both humans and wildlife. Federal and state fishery management agencies should take steps to limit the introduction of compounds which pose a threat to human or American eel health.

American eel from the Kennebec River (Richmond) and the Penobscot River (Bangor) have been tested for dioxin (Mower 1996), and American eel from the west branch of the Piscataqua River (Falmouth) have been tested for heavy metals, PCBs, and organochloride pesticides (Sowles et al. 1996). Dioxin levels for Kennebec River and Penobscot River eel exceeded the maximum allowable concentrations recommended by the Department of Human Service's Bureau of Health. Eel from the Piscataqua River exceeded the Bureau's recommended Fish Consumption Advisory Threshold for mercury; had the highest levels of chromium, zinc, and chlordane of all the fish collected from the site; exceeded the EPA's Risk Based Consumption Limit (RBCL) and screening value (SV) for PCBs and coPCBs; and exceeded the RBCL for DDT. The RBCL is the highest concentration that allows for unlimited consumption for the most conservative exposure scenario (e.g. children versus adults), and the SV is a recommended safe concentration based on effects to the general population of adults.

Toxicological studies have indicated the American eel in certain areas bioaccumulate polychlorinated biphenols (PCBs) in levels above the food health standard (2.0 ppm) (Sowles et al, 1997). American eel have a high fat content and a bioaccumulation of many toxins occurs in the fat of the fish. Studies have also shown bioaccumulation of mercury and other heavy metals, dioxin and chlordane at levels warranting attention in some jurisdictions. Some states have issued health advisories regarding consumption of American eel. The impact of these chemicals on the health and reproductive capacity of American eel themselves is unknown.

4.3.4 Fisheries Practices

The use of any fishing gear or practice, which is documented by management agencies to have an unacceptable impact on American eel (e.g. habitat damage, or bycatch mortality), should be prohibited within the effected important habitats.

4.4 ALTERNATIVE STATE MANAGEMENT REGIMES

With approval of the American Eel Management Board, a state may vary its regulatory specifications listed in Section 4, so long as that state can show to the Board's satisfaction that the goals and objectives of this FMP will still be met.

4.4.1 Procedures

Procedures to modify state regulations include the following:

(a) A state may submit a proposal for a change to its regulatory program or any mandatory compliance measure under the Plan to the ASMFC. Changes shall be submitted to the ASMFC staff, who will distribute the proposal to the Management Board, the Plan Review Team, the Technical Committee, the Stock Assessment Committee, and the Advisory Panel.

(b) States must submit a proposal at least two weeks prior to the Technical Committee's spring or fall meeting.

(c) The Plan Review Team is responsible for gathering the comments of the Technical Committee, the Stock Assessment Committee, and the Advisory Panel, and presenting these comments to the Management Board for action.

(d) The Management Board will approve the state proposal for an alternative management program if it determines that the alternative management program is consistent with the goals and objectives of this Plan.

4.4.2 *De minimis* Status

The ASMFC Interstate Fisheries Management Fisheries Program Charter defines *de minimis* as "a situation in which, under existing condition of the stock and scope of the fishery, conservation, and enforcement actions taken by an individual state would be expected to contribute insignificantly to a coast-wide conservation program required by a Fishery Management Plan or amendment."

Under this FMP, *de minimis* status would exempt a state from having to adopt the commercial and recreational fishery regulations for a particular life stage listed in Section 4 and any fishery-dependent monitoring elements for that life-stage listed in Section 3.4.1. States may apply for *de minimis* status for each life stage if (given the availability of data), for the preceding two years, their average commercial landings (by weight) of that life stage constitute less than one percent of coast wide commercial landings for that life stage for the same two-year period. States may petition the Board at any time for *de minimis* status, if their fishery falls below the threshold level. Once *de minimis* status is granted, designated States must submit annual reports to the Board justifying the continuance of *de minimis* status.

4.5. ADAPTIVE MANAGEMENT

Under adaptive management, the American Eel Management Board may vary the requirements specified in Sections 3 or 4 of this FMP. Such changes will be effective on January 1 (or on the first fishing day of the year), but may be put in place on an alternative date when deemed necessary by the Management Board.

Procedures to implement adaptive management are as follows:

(a) The Plan Review Team (PRT) will continually monitor the status of the fishery and the resource, and report to the Management Board on or about October 1. The PRT will consult with the Technical Committee, the Stock Assessment Committee, and the Advisory Panel, in making their review and report. The report will contain recommendations concerning proposed adaptive revisions to the management program.

(b) The Management Board will review the PRT report, and may consult independently with the Technical Committee, the Stock Assessment Committee, or the Advisory Panel. The Management Board may direct the PRT to prepare an addendum to effect changes it deems necessary. The addendum shall contain a schedule for the states to implement its provisions.

(c) The PRT will prepare a draft addendum as directed by the Management Board, and shall distribute it to all states for review and comment. The Management Board shall, in coordination with each relevant state, utilizing that state's established public review process, ensure that the public has an opportunity to review and comment upon proposed adaptive management changes. The PRT will also request comment from federal agencies and the public at large. After a 30-day review period, the PRT will summarize the comments and prepare a final version of the addendum for the Management Board.

(d) The Management Board shall review the final version of the addendum prepared by the PRT, and also shall consider the public comments received and the recommendations of the Technical Committee, the Stock Assessment Committee, and the Advisory Panel; it shall then decide whether to adopt or revise the addendum.

(e) Upon adoption of an addendum, states shall prepare plans to carry out the addendum and submit them to the Management Board for approval, according to the schedule contained in the addendum.

4.6 EMERGENCY PROCEDURES

Emergency procedures may be used by the American eel Management Board to require any emergency action that is not covered by or is an exception or change to any provision in this fishery management plan. Procedures for implementation are addressed in the ASMFC Interstate Fisheries Management Program Charter, Section 6 (c) (10) (ASMFC 1998).

4.7 MANAGEMENT INSTITUTIONS

4.7.1. Atlantic States Marine Fisheries Commission and ISFMP Policy Board

The Atlantic States Marine Fisheries Commission (Commission) and the Interstate Fisheries Management Program (ISFMP) Policy Board are responsible for the oversight and management of the Commission's fisheries management activities. The Commission must approve all fishery management plans and amendments thereto, and must make final determinations concerning state compliance or noncompliance. The ISFMP Policy Board reviews recommendations of the various Management Boards and, if it concurs, forwards them to the Commission for action.

4.7.2 American Eel Management Board

The American Eel Management Board is responsible for the development of a fishery management plan or amendment, and has voting representatives from Maine, New Hampshire, Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Pennsylvania, Delaware, Maryland, Potomac River Fisheries Commission, District of Columbia, Virginia, North Carolina, South Carolina, Georgia, Florida, the USFWS, and the NMFS. The Board shall provide the ISFMP Policy Board with review and recommendations based on the fishery management plan. The Board may, after the necessary plan or amendment has been approved by the Commission, continue to monitor the implementation and enforcement of the fishery management plan or amendment, advise the ISFMP Policy Board of its effectiveness, or take other actions specified in the fishery management plan that are necessary to ensure its full and effective implementation.

The Board may directly consult with the chairs of the Technical Committee, Plan Review Team, Citizens' Advisory Panel and a representative from the ASMFC Law Enforcement Committee.

4.7.3 Plan Review Team

The Plan Review Team (PRT) is a small group whose responsibility is to provide staff support necessary to carry out and document the decisions of the Management Board. The PRT is directly responsible to the Management Board for providing information and documentation necessary to carry out the Board's decisions.

4.7.4 Technical Committee

The Technical Committee will consist of one representative from each jurisdiction and federal agency with an interest in the American eel fishery. Its role is to act as a liaison to the individual state agencies, providing information to the management process and review and recommendations concerning the management program. The Technical Committee will report to the Management Board, normally through the PRT.

4.7.5 Stock Assessment Subcommittee

The Stock Assessment Subcommittee (SASC) will consist of those scientists with expertise in stock assessment methods. Its role is to assess American eel populations and provide scientific advice concerning the implications of proposed management alternatives, or to respond to other scientific questions of the Management Board. The Stock Assessment Subcommittee membership will be proposed by the Technical Committee, and approved by the Management Board. The Stock Assessment Subcommittee will report to both the Plan Review Team and the Technical Committee.

4.7.6 Advisory Panel

The American Eel Advisory Panel is established according to the ASMFC Advisory Committee Charter. Members of the Advisory Panel are citizens who represent a cross-section of commercial and recreational fishing interests and others concerned about American eel conservation and management. The Advisory Panel provides the Management Board with advice directly concerning the Commission's American eel management program.

4.7.7 Departments of Commerce and Interior

The Commission has accorded NMFS (Department of Commerce) and the USFWS (Department of the Interior) voting status on the ISFMP Policy Board and the American Eel Management Board. These federal agencies may participate on the Plan Review Team, the Technical Committee, and the Stock Assessment Committee.

4.8 RECOMMENDATIONS TO THE SECRETARIES

Secretary of Commerce

The ASMFC recommends that the Secretary of Commerce address and initiate controls over harvest and use of American eel in federal waters (3-200 nautical miles offshore) that are not landed in states. Specifically, the ASMFC recommends that the Secretary of Commerce ban harvests of American eel at any life stage in the EEZ, but permits the possession of up to 50 eel per person as bait

Secretary of Interior

The U.S. Fish and Wildlife Service should provide an annual report, using the Service's new nationwide fish impediment database, documenting the progress made in alleviating barriers to passage for species managed by the Commission, including American eel.

In addition to existing channels for documenting exports, it is also recommended that the Secretary of the Interior proceed with listing American eel glass eel and elvers in Appendix III of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). An Appendix III listing in no manner prohibits the harvest of American eel at any life stage. The Appendix III listing would improve law enforcement and shipment monitoring of glass eel and elvers in the lucrative but largely undocumented international trade. The listing provides for monitoring and inspection at the port of departure and also at the port of arrival of the importing country through the use of a permit system. A CITES Appendix III export permit indicates that a legal harvest has taken place in accordance with the permit issuing authority.

This listing has been recommended, in part, because of discrepancies in law enforcement reports that monitored only a portion of all live eel exports. In this limited number of inspected shipments, U.S. Customs Service records showed export weights that far exceeded National Marine Fisheries Service estimates of the east coast's entire American eel harvest. These data may indicate a need for better export tracking mechanisms through CITES permitting, but do not diminish the continuing need for state and local law enforcement in the field.

5.0 COMPLIANCE

5.1 MANDATORY COMPLIANCE ELEMENTS FOR STATES

Upon completion and approval of a management plan, Commission participating jurisdictions (ME, NH, MA, RI, CT, NY, NJ, PA, DE, MD, PRFC, DC, VA, NC, SC, GA, FL) are obliged to implement its requirements, unless exempted by *de minimis* status. If a state does not comply with the conservation measures of the Commission's fishery management plan, the law allows the U.S. Secretary of Commerce to impose a moratorium on that state's particular fishery. All Commission fishery management plans must include specific measurable standards to improve the status of the stocks and to determine if the states comply with the standards.

5.1.1 Mandatory Elements of State Programs

The following lists the mandatory program elements required for all participating states/jurisdictions to remain in compliance with this Fisheries Management Plan. Details of these compliance requirements are discussed in the identified sections.

1. Annual Young-of-Year Abundance Survey (Section 3.1.1)

2. Annual State Report on Regulations, Harvest, Effort, Bycatch, and Fishery Independent Surveys for American Eel (Section 3.4.1).

3. Recreational Fisheries Management Measures (Section 4.1).

4. Commercial Fisheries Management Measures (Section 4.2).

A state will be found out of compliance if:

(a) The American Eel Management Board has not approved the regulatory and management programs for American eel.

(b) It fails to meet any implementation schedule established in this FMP or any addendum prepared under adaptive management (see Section 4.5).

(c) It fails to conduct an annual young-of-year abundance survey, unless otherwise exempted by the Management Board, beginning in the year 2000. If initially exempted states fail to conduct the young-of-year survey by the year 2001 (See Section 3.1.1).

(d) It fails to implement a change to its program when determined necessary by the American Eel Management Board.

(e) It fails to adequately enforce any aspect of its regulatory and management programs.

5.1.1.1 Regulatory Requirement

All state programs must include a regime of restrictions on recreational and commercial fisheries consistent with the requirements of Sections 4.1 and 4.2; except that a state may propose an alternative management program under Section 4.4. If approved by the American Eel Management Board, the state's proposal may be implemented as an alternative regulatory requirement for compliance under the law.

5.1.1.2 Monitoring Requirements

All state programs must include the mandatory monitoring requirements contained in Section 3.4.1 of the Plan. States must submit proposals to the Commission for any proposed changes to the required monitoring programs if the change may affect the quality of the data or the ability of the program to fulfill the needs of the fishery management plan. State proposals for modifications to required monitoring programs will be submitted to the Technical Committee at least two weeks prior to its spring or fall meetings. Proposals must be on a calendar year basis.

The Technical Committee will make recommendations to the American Eel Management Board concerning whether the proposals are consistent with the Plan.

If a state realizes it will be unable to fulfill its fishery monitoring requirements, it should immediately notify the Commission in writing. The Commission must be notified by the planned commencement date of the monitoring program.

The Commission will work with the state to develop a plan to secure funding or to plan an alternative program that will satisfy the needs outlined in this FMP (the Plan).

Each year, the ASMFC's Law Enforcement Committee (LEC) shall discuss new or chronic problems in enforcing eel regulations or prosecuting violators of these regulations. The LEC shall also make recommendations to improve enforcement and understanding of the regulations.

5.1.2 State Reporting and Compliance Schedule

Each state must submit an annual report concerning its American eel fisheries and management program on or before September 1 each year. The report shall cover:

(a) The previous calendar year's fishery and management program, including activity and results of monitoring (as identified in Section 3.4.1. of the Plan), regulations that were in effect, and harvest, including estimates of non-harvest losses and effort.

(b) The planned management program for the current calendar year (summarizing regulations that will be in effect and monitoring programs to be performed) highlighting any changes from the previous year.

States must implement this Plan according to the following schedule:

May 1, 2000: States must submit state programs to implement the Plan for approval by the Management Board. Programs, including monitoring programs, must be implemented upon approval by the Management Board.

January 1, 2001: States with approved management programs must begin implementing the Plan (or earlier if desired).

5.2 PROCEDURES FOR DETERMINING COMPLIANCE

A. The PRT will continually review the status of state implementation of the Plan, and advise the American Eel Management Board whenever a question arises concerning state compliance. The PRT will review state reports submitted under Section 5.1.2 and prepare a report for the American Eel Management Board, summarizing the status of the resource and fishery and the status of state compliance on a state-by-state basis.

B. Upon receipt of a report from the PRT, or at any time by request from a member of the American Eel Management Board, the Management Board will review the status of an individual state's compliance. If the Management Board finds that a state's regulatory and management program fails to meet the requirements of this section, it may recommend that the state is out of compliance. The recommendation must include a specific list of the state's deficiencies in implementing and enforcing the Plan and the actions that the state must take in order to come back into compliance.

C. If the American Eel Management Board recommends that a state is out of compliance, as referred to in the preceding paragraph, it shall report that recommendation to the ISFMP Policy Board for further review according to the *ASMFC Charter for the Interstate Fisheries Management Program*.

D. A state that is out of compliance or subject to a recommendation by the American Eel Management Board under the preceding subsection may request at any time that the Management Board reevaluate its program. The state shall provide a written statement concerning its actions to justify a reevaluation. The Management Board shall promptly conduct such reevaluation (e.g., within 30 days), and if it agrees with the state, the Management Board shall recommend to the ISFMP Policy Board that the determination of noncompliance be withdrawn. The ISFMP Policy Board and the Commission shall address the Management Board's recommendation according to the ASMFC Charter for the Interstate Fisheries Management Program.

6.0 INFORMATION AND RESEARCH NEEDS

6.1 MANAGEMENT AND REGULATORY

Issues that have been identified as needed to support the management of American eel (order does not indicate importance). Information needed for regulations to manage harvest, include but not limited to:

- License fees, life stage, size, geographic area, and gear type.

- Design and implement an annual, fishery-independent, glass eel abundance survey.

- Assess American eel landing records for all life stages to determine their completeness and adequacy for evaluating the eel fishery; monitor population trends; commercial and recreational harvest; and, effects of gear type on harvest rates. If necessary, determine what data are needed to improve landing records.

- Evaluate the impact of American eel aquaculture on fish health, eel culture/hatcheries, and import and/or export concerns.

- Management of the species and its harvest by non-member jurisdictions (e.g., Vermont, West Virginia, Great Lakes States, Gulf Coast States and Canada).

- Quantify and qualify the economic considerations of exporting various American eel life stages.

- Quantify and qualify the economic considerations of the American eel bait fishery.

6.2 STOCK ASSESSMENT AND POPULATION DYNAMICS

To collect information to assist in future management decisions, a comprehensive monitoring plan must be developed throughout the Atlantic Coast as described in Section 3.4. In addition to the comprehensive monitoring plan, additional stock assessment and population dynamics information should be collected to assist in future management decisions including the following:

- Conduct additional stock assessments and determine harvest mortality rates. Use these data to develop a more reliable sustainable harvest rate.

- Further evaluate life history (table) information including sex ratio and population age structure.

- Formulate a coast wide sampling program for American eel using standardized and statistically robust methodologies.

- Contaminant effects on the fishery and effects of bioaccumulation with respect to harvest and sale prohibitions.

- Size-age-sex distributions within selected drainage containing different habitat types.

- Predator-prey relations: a) food habits of American eel in various habitats and b) predation on eel.

- Movements of American eel within a drainage during the yellow eel stage: a) degree of movement of eel between fresh waters and estuaries and b) degree of movements within fresh waters.

6.3 RESEARCH

Numerous additional data needs have been identified to improve the understanding of the life history of this species and the anthropogenic stresses that may influence its health and abundance.

- Stock assessment and determination of fishing mortality rates (F) to develop a sustainable harvest rate.

- Economic studies are necessary to determine the value of the fishery and the impact of regulatory management.

- Investigate: mechanism of sex determination; growth rates for males and females throughout their range; habitat preferences of males and females; predator-prey relationships; behavior and movement of American eel during their freshwater residency; oceanic behavior, movement and spawning location of mature adult American eel; and all information on the leptocephalus stage of the American eel.

- Evaluate contaminant effects on American eel and the effects of bioaccumulation with respect to impacts by age on survival and growth and effect on maturation and reproductive success.

- Investigate mode of nutrition of American eel leptocephali in the ocean.

- Determine growth rates of male and female American eel in different habitats.

- Determine if geographic sub-populations exist, which may have implications for management.

- Investigate larval and juvenile survival and mortality to assist in the assessment of annual recruitment. Such research could be aided by continuing and initiating new tagging programs within individual states.

- Determine food habits of glass eel while at sea.

- Investigate location and triggering mechanism for metamorphosis from leptocephalus to glass eel.

- Investigate mechanisms of exit from the Sargasso Sea and of transport across the continental shelf.

- Evaluate the impact, both upstream and downstream, of barriers on American eel with respect to population and distribution affects. Determine areas of extirpation and historical distribution.

- Investigate, develop, and improve technologies for American eel passage upstream and downstream.

- Evaluate the ecosystem importance of American eels as prey, predators, and mechanisms of transporting freshwater biomass to marine systems.

- Determine fecundity-length and fecundity-weight relations for female American eel from various parts of its geographic range.

- Determine mortality rates at different life history stages (leptocephalus, glass eel, yellow eel, and silver eel) and mortality rates with size within the yellow eel stage.

- Investigate mechanism of sex determination in American eel.

- Determine age at entry of glass eel into estuaries and fresh waters.

- Investigate migratory routes and guidance mechanisms for silver eel in the ocean.

- Investigate mechanisms of recognition of the spawning area by silver eel.

- Investigate mate location in the Sargasso Sea.
- Conduct studies on spawning behavior.
- Determine gonadal development in maturation.
- Conduct workshop on aging techniques.
- Sustainable fishing mortality rates (F) for American eel have not been examined. Researchers and fishery managers have not determined the best means to ensure the stability of the American eel populations
- Identification and understanding of American eel habitat needs for all life stages
- Model the effect of increased habitat availability and reductions in mortality at various freshwater lifestages on escapement.
- Research the impacts of elver fishing on the abundance and distribution of later lifestages within a watershed and what, if any, impacts there are on sexual determination and upstream migration.
- Research techniques (physical and behavioral) for providing upstream and downstream passage around dams
- Research the feasibility and ecological/genetic impacts of trap and truck programs for elvers
- Quantify and assess male eel habitat and male eel abundance
- Quantify and estimate the impact of the bait fishery for juvenile/bootstrap eels.

7.0 PROGRAM MANAGEMENT OPTIONS

7.1 IMPROVE IMPLEMENTATION EFFECTIVENESS

This FMP outlines a number of management actions addressing American eel (Section 3.1-3.3) and its habitats (Section 3.5). Since American eel are one population, management effectiveness would increase through focused coordination and standardization of most monitoring, assessment, and restoration activities throughout its range. This centralized approach could provide leverage for funding (internal and external), prioritization of research, and a central repository of information and data.

7.1.1 New Funding Options

New, dedicated funds would improve and expedite implementation of this FMP. Recommendation by the American Eel Management Board to the ASMFC members requesting their active support is needed. The following options have been suggested:

A. Advisory Panel member recommendation for a federal “migratory fish stamp,” similar to the migratory bird stamps, with the funds dedicated to habitat restoration and enhancement.

B. A current effort underway by members of the hydropower industry to obtain funds from Congress to target multi-year American eel research and management enhancement.

C. Improve coordination and partnerships with other agencies with complementary missions, such as USEPA and the USACOE, to assess the ecological health of coastal watersheds and to restore them.

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9.0 APPENDIX

ATTACHMENT 1: NMFS Commerical Landing for American Eel (pounds) for the Atlantic and gulf Coasts of the U.S. by jurisdiction (Personal Communication from the NMFS, Fisheries Statistics and Economics Division, 11-15-99).

State/Province	1950 - 1996 Total pounds	Percent of total	1987 - 1996 Total pounds	Percent of total
Virginia	**21228939**	**26.01**	**4087675**	**29.93**
Maryland	**12021898**	**14.73**	**1975350**	**14.46**
P.R.F.C.	**9666343**	**11.84**	**2457555**	**18.00**
North Carolina	**8102355**	**9.93**	**859843**	**6.30**
New Jersey	**7313446**	**8.96**	**1855346**	**13.59**
New York	**7088810**	**8.68**	**344310**	**2.52**
Delaware	**3798000**	**4.65**	**1398200**	**10.24**
Massachusetts	**3580451**	**4.39**	**195251**	**1.43**
Florida East Coast	**3080542**	**3.77**	**6685**	**0.05**
Maine	**2318655**	**2.84**	**259971**	**1.90**
Connecticut	**949058**	**1.16**	**165758**	**1.21**
Rhode Island	**912300**	**1.12**	**6900**	**0.05**
South Carolina	**600200**	**0.74**	**0**	**0.00**
Florida Inland Lakes	**355400**	**0.44**	**0**	**0.00**
Georgia	**250436**	**0.31**	**9048**	**0.07**
New Hampshire	**206237**	**0.25**	**2379**	**0.02**
Florida West Coast	**108508**	**0.13**	**32188**	**0.24**
Louisiana	**50614**	**0.06**	**0**	**0.00**
Texas	**98**	**0.00**	**98**	**0.00**
Total	**81632290**	**100**	**13656557**	**100**

Atlantic States Marine Fisheries Commission

ADDENDUM IV TO THE INTERSTATE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL

ASMFC Vision:
Sustainably Managing Atlantic Coastal Fisheries

Approved October 2014

EXECUTIVE SUMMARY

The Atlantic States Marine Fisheries Commission's American Eel Management Board (Board) initiated the development of Addendum III in August 2012 in response to the 2012 Benchmark American Eel Stock Assessment, which found the American eel population in U.S. waters is depleted. The assessment found the stock is at or near historically low levels due to a combination of historical overfishing, habitat loss and alteration, productivity and food web alterations, predation, turbine mortality, changing climatic and oceanic conditions, toxins and contaminants, and disease. In August 2013, the Board approved some of the measures from Addendum III (predominately the commercial yellow eel and recreational fishery management measures) and split out the remainder of the management measures for further development in Addendum IV. As the second phase of management response to the stock assessment, this Addendum addresses further addresses the commercial glass, yellow, and silver eel fisheries. Specifically, this Addendum modifies the previous management program as follows:

Commercial Glass Eel Fishery Management Program (Section 3.1.1)

- Maine's quota for the 2015-2017 commercial glass eel fishing seasons will be set at 9,688 pounds annually and will be re-evaluated prior to the start of the 2018 fishing season.
- Any state or jurisdiction can request an allowances for commercial harvest of glass eels based on stock enhancement programs implemented after January 1, 2011, subject to TC review and Board approval.
- For any state or jurisdiction managed with a commercial glass eel quota, if an overages occurs in a fishing year, then that state or jurisdiction will be required to deduct their entire overage from the quota the following year, pound for pound.
- Any state or jurisdiction with a commercial glass eel fishery is required to implement daily trip level reporting with daily electronic accounting to the state for both harvesters and dealers in order to ensure accurate reporting of commercial glass eel harvest.
- Any states or jurisdiction with a commercial glass eel fishery must implement a fishery independent life cycle survey covering glass, yellow, and silver eels within at least one river system.

Commercial Yellow Eel Fishery Management Program (Section 3.1.2)

The commercial yellow eel fishery will be regulated through a coastwide catch cap set at 907,671 pounds. Under this cap, there are two management triggers. Upon reaching either of these triggers, the Board is required to alter the management program as specified below in order to ensure the objectives of the management program are achieved.

Management Triggers

1. The coastwide catch cap is exceeded by more than 10% in a given year (998,438 pounds).
2. The coastwide catch cap is exceeded for two consecutive years, regardless of percent over.

Management Response
If either trigger is tripped, then there would be automatic implementation of a state-by-state commercial yellow eel quota. The annual coastwide quota is set at 907,669 pounds, with allocations as specified in Table 1.

Commercial Silver Eel Fishery Management Measures (Section 3.1.3)
The Delaware River silver eel weir fishery is restricted to nine annual permits. These permits will initially be limited to those permitted participants that fished and reported landings from 2010 to 2013. Permits may be transferred.

Sustainable Fishery Management Plans for American Eel (Section 3.1.4)
Fishing Mortality Based Plan – Under an approved fishing mortality plan, states and jurisdictions may petition the Board for alternative management based on the current level of mortality that is occurring on their population.

Transfer Plan – If states or jurisdictions implement quota management for at least one fishery, then a state may develop a Transfer Plan to request a transfer of quota from one fishery to another (e.g. from yellow to glass) based on the life history characteristic inherent to that area (e.g. state, river, or drainage).

Aquaculture Plan - Under an approved Aquaculture Plan, states and jurisdictions may harvest a maximum of 200 pounds of glass eel annually from within their waters for use in domestic aquaculture facilities provided they can objectively show that the harvest will occur from a watershed that minimally contributes to the spawning stock of American eel.

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1. INTRODUCTION

The Atlantic States Marine Fisheries Commission (Commission) has coordinated interstate management of American eel (*Anguilla rostrata*) from 0-3 miles offshore since 2000. American eel is currently managed under the Interstate Fishery Management Plan (FMP) and Addenda I-III to the FMP. Management authority in the exclusive economic zone (EEZ) from 3-200 miles from shore lies with NOAA Fisheries. The management unit is defined as the portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

2. BACKGROUND

2.1. STATEMENT OF THE PROBLEM

The Commission's American Eel Management Board (Board) initiated the development of Draft Addendum III in August 2012 in response to the 2012 American Eel Benchmark Stock Assessment, which found the American eel population in U.S. waters is depleted. The assessment found the stock is at or near historically low levels due to a combination of historical overfishing, habitat loss and alteration, productivity and food web alterations, predation, turbine mortality, changing climatic and oceanic conditions, toxins and contaminants, and disease. Draft Addendum III for Public Comment included a range of options for the commercial glass, yellow, and silver eel fisheries, as well as the recreational fishery. In August 2013, the Board approved some of the measures from Draft Addendum III for Public Comment (predominately the commercial yellow eel and recreational fishery management measures) and split out the remainder of the management measures (commercial glass and silver eel fisheries) for further development in Addendum IV. As the second phase of management in response to the 2012 stock assessment, the goal of Addendum IV is to continue to reduce overall mortality and increase overall conservation of American eel stocks. This Addendum addresses the commercial glass, yellow, and silver eel fisheries.

2.2. LIFE HISTORY

American eel (*Anguilla rostrata*) inhabit fresh, brackish, and coastal waters along the Atlantic, from the southern tip of Greenland to Brazil. American eel eggs are spawned and hatch in the Sargasso Sea. After hatching, leptocephali—the larval stage—are transported at random to the coasts of North America and the upper portions of South America by ocean currents. Leptocephali are then transformed into glass eels via metamorphosis. In most areas, glass eel enter nearshore waters and begin to migrate up-river, although there have been reports of leptocephali found in freshwater in Florida. Glass eels settle in fresh, brackish, and marine waters; where they undergo pigmentation, subsequently maturing into yellow eels. Yellow eel can metamorphose into a silver eel (termed *silvering*) beginning at age three and up to twenty-four years old, with the mean age of silvering increasing with increasing latitude. Environmental factors (e.g., food availability and temperature) may play a role in the triggering of silvering. Males and females differ in the size at which they begin to silver. Males begin silvering at a size typically greater than 14 inches and females begin at a size greater than 16-20 inches (Goodwin and Angermeier 2003). However, this is thought to vary by latitudinal dispersal. Actual metamorphosis is a gradual process and eels typically reach the silver eel stage during their migration back to the Sargasso Sea, where they spawn and die.

Eels make extensive use of freshwater systems, but they may migrate to and from or remain in brackish and marine waters. Therefore, a comprehensive eel management plan and set of regulations must consider the various unique life stages and the diverse habitats of American eel, in addition to society's interest and use of this resource.

2.3. STATUS OF MANAGEMENT

American eel occupy a significant and unique niche in the Atlantic coastal reaches and tributaries. Historically, American eels were very abundant in East Coast streams, comprising more than 25 percent of the total fish biomass. Eel abundance had declined from historic levels but remained relatively stable until the 1970s. Fishermen, resource managers, and scientists postulated a further decline in abundance based on harvest information and limited assessment data during the 1980s and 1990s. This resulted in the development of the Commission's Interstate Fishery Management Plan (FMP) for American Eel, which was approved in 1999. The FMP required that all states maintain as conservative or more conservative management measures at the time of implementation for their commercial fisheries and implement a 50 fish per day bag limit for the recreational fishery. The FMP also required mandatory reporting of harvest and effort by commercial fishers and/or dealers and specific fisheries independent surveys to be conducted annually by the states.

Since then the FMP was modified three times. Addendum I (approved in February 2006) established a mandatory catch and effort monitoring program for American eel. Addendum II (approved in October 2008) made recommendations for improving upstream and downstream passage for American eels. Most recently, Addendum III (approved in August 2013) made changes to the commercial fishery, specifically implementing restrictions on pigmented eels, increasing the yellow eel size limit from 6 to 9 inches, and reducing the recreational creel limit from 50 fish to 25 fish per day.

2.3.1. INTERNATIONAL MANAGEMENT

Despite data uncertainties with European eels and American eels in Canada, both the European Union and the Department of Fisheries and Oceans Canada have taken recent management actions to promote the rebuilding of local stocks.

2.3.1.1. EUROPEAN MANAGEMENT

While American and European eels (*Anguilla anguilla)* are two separate species, the spawning grounds and early life history habitats are believed to overlap. Therefore oceanographic changes could influence both stocks. Currently, the European eel stock is considered severely depleted (ICES, 2013). Major fisheries occur in the Netherlands, France, Sweden, and the United Kingdom, with total 2012 commercial harvest in the EU estimated at 5.2 million pounds and recreational harvest estimated at 1.1 million pounds (Figure 1; ICES, 2013). In 2007, the European Union (EU) passed legislation which required EU countries to develop and implement measures to allow 40% of adult eels to escape from inland waters to the sea for spawning purposes. In addition, beginning in 2008, EU countries that catch glass eel (defined as juvenile eels less than 4.7 inches long) were required to use 35% of their catch for restocking within the EU and increase this to at least 60% by 2013.

To demonstrate how they intend to meet the target, EU countries were required to develop national eel management plans at river-basin level. To date, the European Commission has adopted all plans submitted by 19 EU countries, plus a joint plan for the Minho River (Spain/Portugal). Management measures implemented though these plans vary from country to country, but are similar to most management measures considered or implemented in the U.S. The management measures include:

- Seasonal closures
- Size limits (11 – 21.6 inches)
- Recreational bag limit (2 - 5 fish/angler/day)
- Gear restrictions (banning fyke nets, increasing mesh size)
- Reducing effort (e.g. by at least 50%)
- Prohibiting glass, silver or all commercial fishing
- Commercial quotas
- Implementing catch and release recreational fisheries only
- Reducing illegal harvest and poaching
- Increasing fish passage
- Restocking suitable inland waters with glass eels

In 2013 the International Council on the Exploration of the Seas (ICES) completed an evaluation on the implementation of the national management plans (ICES, 2013a). ICES concluded that, given the short time since implementation, restrictions on commercial and recreational fisheries for silver eel has contributed the most to increases in silver eel escapement. The effectiveness of restocking remains uncertain (ICES, 2013a). ICES advises that data collection, analysis, and reporting should be standardized and coordinated to facilitate the production of stock-wide indicators to assess the status of the stock and to evaluate the effect of management regulations.

In response to the evaluation, European Parliament passed a resolution in September 2013 requesting the European Commission present new legislation to further conserve European eel populations. The new law must close the loopholes allowing the continued overfishing and illegal trade; evaluate current restocking measures and their contribution to eel recovery; require more timely reporting on the impact of eel stock management measures; and require member states that do not comply with the reporting and evaluation requirements to reduce their eel fishing effort by 50%. The European Commission's new legislative proposal, which is expected to be presented in early 2015, must aim to achieve the recovery of the stock "with high probability".

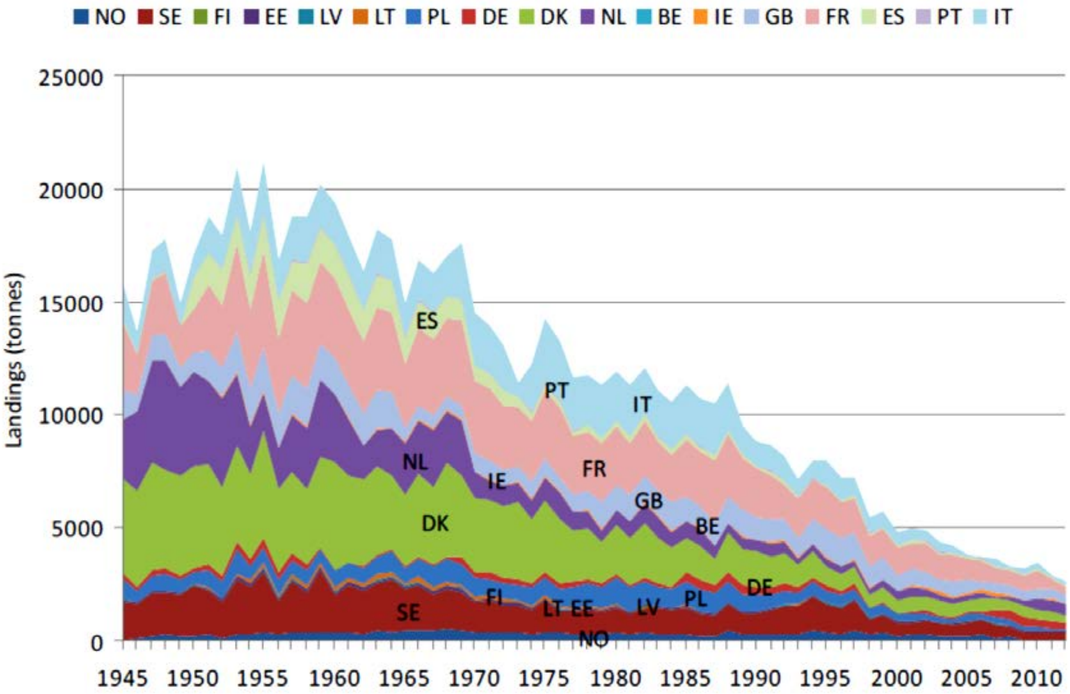

Figure 1. Total landings of European eel (all life stages) from 2013 Country Reports (Note: not all countries reported). NO = Norway, SE = Sweden, FI – Finland, EE = Estonia, LV = Latvia, LT = Lithuania, PL = Poland, DE = Germany, DK = Denmark, NL = Netherlands, BE = Belgium, IE = Ireland, GB = Great Britain, FR = France, ES = Spain, PT = Portugal, IT = Italy. *From ICES, 2013a.*

In November 2013, ICES completed an update on European stock status to provide management advice for the 2014 fishing year (ICES, 2013b). The update found that annual recruitment of glass eel to European waters has increased over the last two years, from less than 1% to 1.5% of the reference level in the "North Sea" series, and from 5% to 10% in the "Elsewhere" series[1], which may or may not be the result of the regulatory changes (Figure 2). However, despite recent increases, production of offspring is very low and there is a risk that the adult stock size is too small to produce sufficient amount of offspring to maintain the stock (ICES, 2013b). The biomass of escaping silver eel is estimated to be well below the target (ICES, 2013b). ICES continues to recommend that all anthropogenic mortality affecting production and escapement of silver eels should be reduced to as close as possible to zero, until there is clear evidence of sustained increase in both recruitment and the adult stock. The stock remains critical and urgent action is needed (ICES, 2013b).

2.3.1.2. CANADIAN MANAGEMENT

American eel are widespread in eastern Canada, but there are dramatic declines throughout its range, including Lake Ontario and the upper St. Lawrence. Although trends in abundance are highly variable, strong declines are apparent in several indices. The American eel was

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[1] The North Sea series are from Norway, Sweden, Germany, Denmark, Netherlands, and Belgium. The Elsewhere series are from UK, Ireland, France, Spain, Portugal, and Italy.

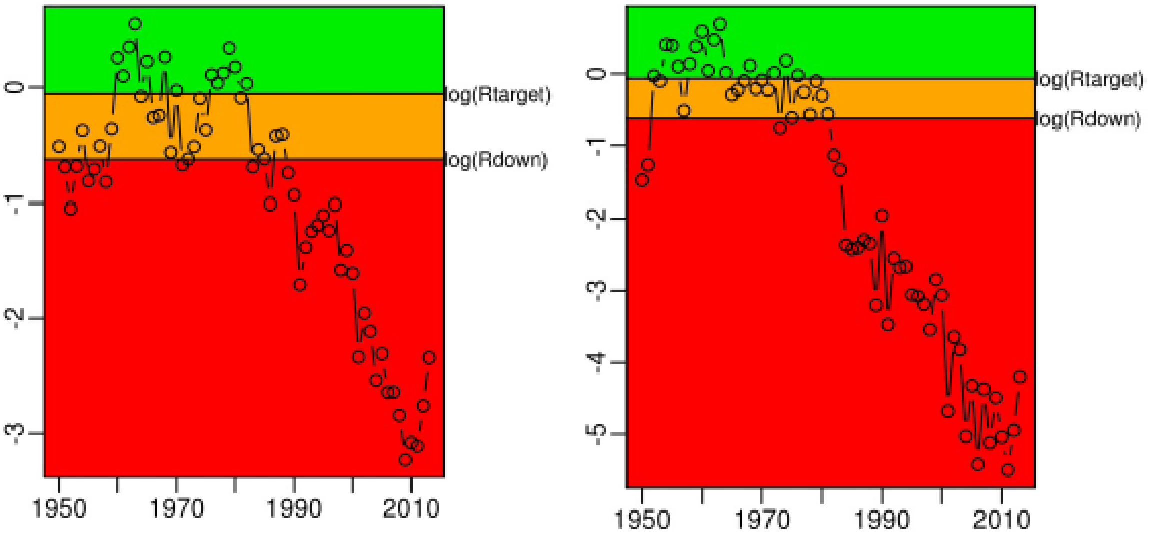

Figure 2. Trends in recruitment ("Elsewhere", left, and "North-Sea", right) of European eels with respect to healthy zone (green), cautious zone (orange) and critical zone (red). *From ICES, 2013b.*

first assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2006 and was designated as a species of "Special Concern." The status was re-examined by COSEWIC in 2012 and it was recommended to list the species as Threatened under the Canadian Species at Risk Act (similar to the U.S. Endangered Species Act). A National Management Plan for American Eel in Canada was developed by the Canadian Eel Working Group which specifies short and long term goals for recovery (DFO, 2010). One of the short-term goals of the plan is to reduce eel mortality from all anthropogenic sources by 50% relative to the 1997-2002 average. Long-term management goals include rebuilding overall abundance of the American eel in Canada to its mid-1980s levels.

Canadian commercial yellow and silver American eel fisheries occur in New Brunswick, Prince Edward Island, Nova Scotia, Newfoundland and Labrador, and Québec (Figure 3). Fishing occurs in both fresh and marine waters, but many rivers and coastal habitats remain unfished. Elver fisheries in Canada occur only in Scotia-Fundy and the south coast of Newfoundland. Overall total reported American eel landings in Canada declined through the early 1960s, increased to a peak in the late 1970s, and have since declined to the lowest level in recent history (Cairns et al, 2014). Winter recreational spear fisheries of yellow eels also occur in the Southern Gulf of St. Lawrence.

Recent management measures to meet the goals of the National Management Plan have included:

- Minimum size limits raised to 20.8 inches (Gulf region), 13.75 inches (Maritimes region) and 11.8 inches (southwestern New Brunswick, Newfoundland and Labrador)
- Reduction to seasons
- Area closures
- Buyouts of licenses
- Glass eel fisheries are not permitted in areas where fisheries exist for larger eels
- Enforcement of regulatory definitions on fyke nets

- Measures to reduce high grading
- License caps, limited entry, and license reductions
- Gear restrictions, including a 1” x ½” escapement panel
- Quota reductions, including 10% cut in glass eel fisheries

The first large-scale eel stocking experiment occurred in the Richelieu River, a tributary to Lake Champlain, in 2005. Since then, a total of seven million elvers have been stocked in Canadian waters. Stocking initiatives can be considered as a potential threat because their effects are uncertain, manifestation of some effects may only be apparent years after, and because of the documented negative effects of stocking of on other fish, particularly salmon (COSEWIC, 2012). Continuing habitat degradation, especially owing to dams and pollution, and existing fisheries in Canada and elsewhere may constrain recovery (COSEWIC, 2102).

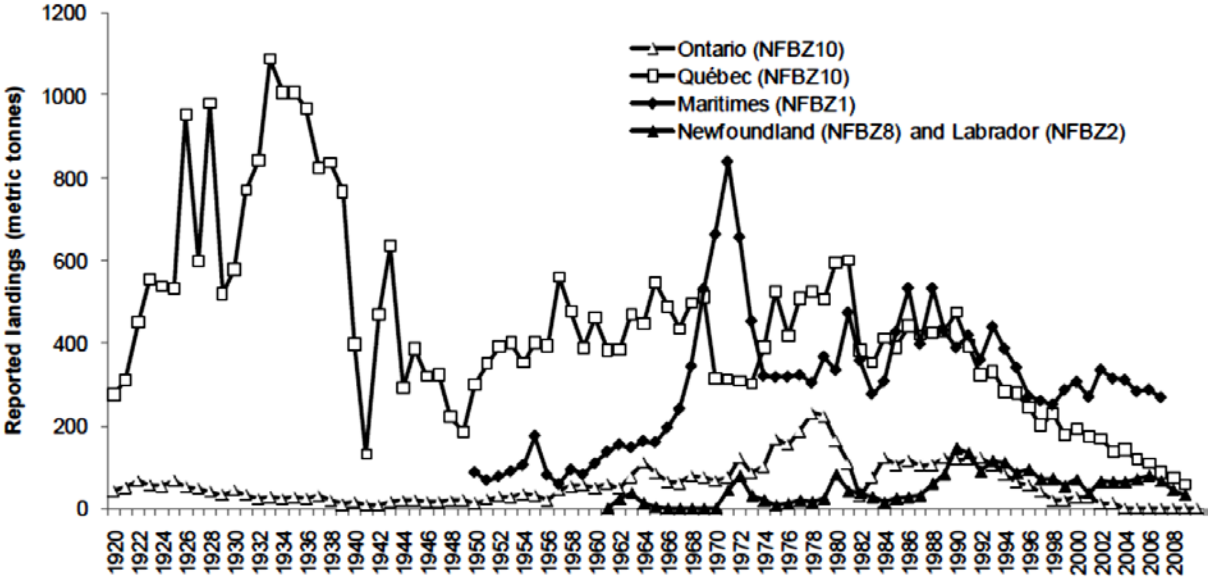

Figure 3. Reported landings of all life stages from Quebec, Ontario, the Maritime Provinces, and Newfoundland and Labrador from 1920 – 2010. *From COSEWIC, 2012.*

2.3.2. ENDANGERED SPECIES ACT CONSIDERATION

American eel were petitioned for listing as threatened under the Endangered Species Act (ESA) in April 2010 by the Center for Environmental Science, Accuracy, and Reliability (CESAR, formally the Council for Endangered Species Act Reliability). The U.S. Fish and Wildlife Service (USFWS) published a positive 90 day finding on the petition in September 2011, stating that the petition may be warranted and a status review will be conducted. CESAR filed a lawsuit in August 2012 against USFWS for failure to comply with the statues of the ESA, which specifies a proposed rule based on the status review be published within one year of the receipt of the petition. A Settlement Agreement was approved by the court in April 2013 and requires USFWS to publish a 12-month finding by September 30, 2015. The USFWS previously reviewed the status of the American eel in 2007 and found that, at that time, protection under the Endangered Species Act was not warranted.

The five factors on which listing is considered include:

1. Present or threatened destruction, modification, or curtailment of its habitat or range;
2. Over-utilization of the species for commercial, recreational, scientific, or educational purposes;
3. Disease or predation;
4. Inadequacy of existing regulatory mechanisms; and
5. Other natural or manmade factors affecting its continued existence.

2.4. STATUS OF THE STOCK

The Benchmark Stock Assessment was completed and accepted for management use in May 2012. The assessment indicated that the American eel stock has declined in recent decades and the prevalence of significant downward trends in multiple surveys across the coast is cause for concern (ASMFC, 2012). The stock is considered depleted, however no overfishing determination can be made at this time based solely on the trend analyses performed (ASMFC, 2012). The ASMFC American Eel Technical Committee (TC) and Stock Assessment Subcommittee (SAS) caution that although commercial fishery landings and effort have declined from high levels in the 1970s and 1980s (with the recent exception of the glass eel fishery), current levels of fishing effort may still be too high given the additional stressors affecting the stock such as habitat loss, passage mortality, and disease as well as potentially shifting oceanographic conditions. Fishing on all life stages of eels, particularly young-of-the-year and in-river silver eels migrating to the spawning grounds, could be particularly detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled.

In 2014 the TC and Stock Assessment Subcommittee (SAS) completed an update of the young of the year (YOY) indices included in the benchmark stock assessment. The FMP requires states and jurisdictions with a declared interest in the species to conduct an annual YOY survey for the purpose of monitoring annual recruitment of each year’s cohort. The benchmark assessment included data only through 2010. Since that time some states have heard anecdotal information about increased recruitment as well as recorded evidence of increased recruitment in their fisheries independent YOY surveys.

Based on the update of the YOY indices, the TC found no change in the YOY status from the benchmark assessment with the exception of one survey in Goose Creek, SC (Table 1). YOY trends are influenced by many local environmental factors, such as rainfall and spring temperatures. While some regions along the coast have experienced high catches in 2011, 2012, and/or 2013, other regions have experienced average or lower catches. For example in 2012, Rhode Island and Florida had below average counts, with Florida having its lowest catch of their time series; New Hampshire, New York, Virginia, and Georgia had average counts; and Maine, Connecticut, New Jersey, Delaware, and Maryland had their highest YOY catches on record. The TC stresses high YOY catches in a few consecutive years do not necessarily correspond to an increasing trend since the YOY surveys can fluctuate greatly. Additionally, due to the limited extent of sampling, trends at the state level may not be reflective of what is actually occurring statewide or coastwide. The YOY indices were only one factor in the determination of the depleted stock status for American eel, so therefore there is no recommended change in the conclusions of the benchmark assessment and the depleted stock status is still warranted. In November 2014, the International Union for the Conservation of Nature (IUCN) reviewed the status of American eel and listed the species as "endangered" on the IUCN Red List.

Region	State	Site	SA Result	Update
Gulf of Maine	ME	West Harbor Pond	NS	NS
	NH	Lamprey River	NS	NS
	MA	Jones River	NS	NS
	MA	Parker River	NS	NS
Southern New England	RI	Gilbert Stuart Dam	NS	NS
	RI	Hamilton Fish Ladder	NS	NS
	NY	Carmans River	NS	NS
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Patcong Creek	NS	NS
	DE	Millsboro Dam	NS	NS
	MD	Turville Creek	NS	NS
Chesapeake Bay	PRFC	Clarks Millpond	NS	NS
	PRFC	Gardys Millpond	NS	NS
	VA	Brackens Pond	NS	NS
	VA	Kamps Millpond	NS	NS
	VA	Warehams Pond	NS	NS
	VA	Wormley Creek	NS	NS
South Atlantic	SC	Goose Creek	NS	↓
	GA	Altamaha Canal	NS	NS
	GA	Hudson Creek	NS	NS
	FL	Guana River Dam	NS	NS

Table 1. Results of the Mann-Kendall trend analysis applied to 2012 Benchmark Stock Assessment (SA) and updated YOY indices developed from the ASMFC-mandated recruitment surveys. Trend indicates the direction of the trend if a statistically significant temporal trend was detected (P-value < α; $\alpha = 0.05$). NS = not significant.

2.5. STATUS OF THE FISHERY

The American eel fishery primarily targets yellow stage eel. Silver eels are caught during their fall migration as well. Eel pots are the most typical gear used; however, weirs, fyke nets, and other fishing methods are also employed. Yellow eels were harvested for food historically, today's fishery sells yellow eels primarily as bait for recreational fisheries. From 1950 to 2012, U.S. Atlantic coast landings ranged from a low of approximately 664,000 pounds in 1962 to a high of 3.67 million pounds in 1979 (Figure 4). After an initial decline in the 1950s, landings increased to a peak in the 1970s and early 1980s in response to higher demand from European food markets. In most regions, landings declined sharply by the late 1980s and have fluctuated around one million pounds for the past decade. The value of U.S. commercial yellow eel landings as estimated by NOAA Fisheries has varied from less than a $100,000 (prior to the 1980s) to a peak of $6.4 million in 1997.

State reported landings of yellow eels in 2013 totaled 907,671 pounds (Table 2) which represents an 17% decrease (~187,000) in landings from 2012 (1,104,429 pounds). Since 2000, yellow eel landings have increased in the Mid-Atlantic region (NY, NJ, and MD) with the exception of Delaware and the Potomac River. Additionally, yellow eel landings have declined in the New England region (ME, NH, MA, CT) with the exception of Rhode Island. Within the Southern region, since 2000 landings have declined in North Carolina but increase in Florida. In 2013, state reported landings from New Jersey, Delaware, Maryland, and Virginia each totaled over 80,000 pounds of eel, and together accounted for 86% of the coastwide commercial total landings.

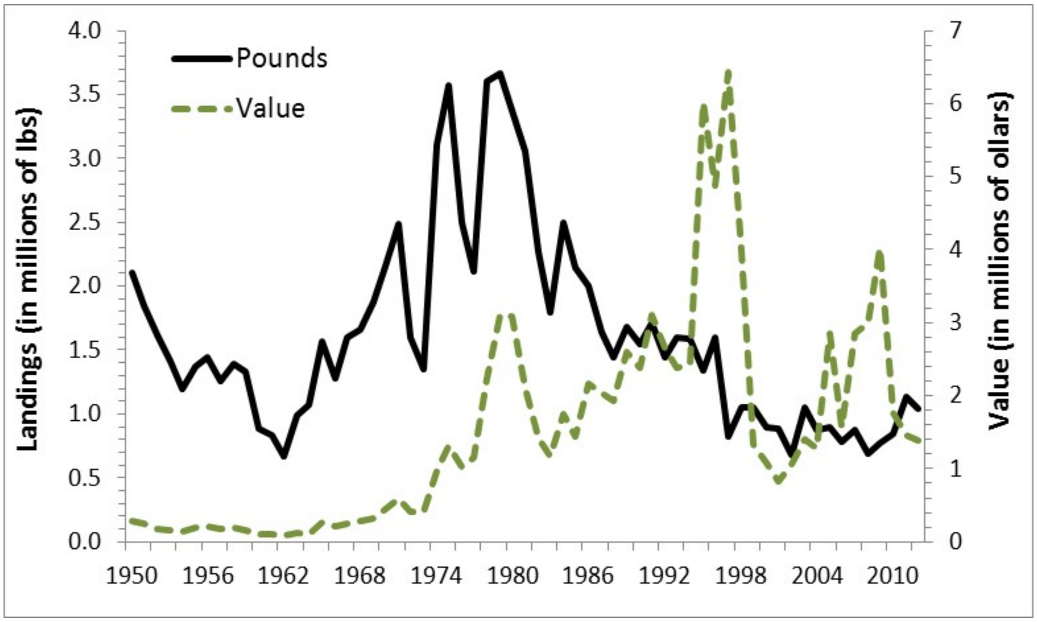

Figure 4. Total commercial landings (in pounds) and value (in millions of dollars) of yellow eels along the U.S. Atlantic Coast, 1950–2012.

Glass eel fisheries along the Atlantic coast are prohibited in all states except Maine and South Carolina. In recent years, Maine is the only state reporting significant harvest (Table 3). Harvest has increased the last few years as the market price has risen to more than $2,000 per pound, although in 2014 prices were recorded between $400 and $650 per pound. Glass eels are exported to Asia to serve as seed stock for aquaculture facilities. Landings of glass eels in 2012 were reported from Maine and South Carolina and totaled 22,215 pounds.

Because eel is managed by the states and is not a target species for the NMFS, landings information for states that rely on the NMFS estimates may be underreported. In addition, at least a portion of commercial eel landings typically come from non-marine water bodies. Even in states with mandatory reporting, these requirements may not extend outside the marine district, resulting in a potential underestimate of total landings. Despite concern about the level of under reporting, reported landings are likely indicative of the trend in total landings over time.

Table 2. Harvest (in pounds) by state of yellow eels from 1998 - 2013. * *Confidential*

Year	ME	NH	MA	RI	CT	NY	NJ	DE	MD	PRFC	VA	NC	SC	GA	FL	Total
1998	20,671	459	5,606	967	5,606	16,896	94,327	131,478	301,833	209,008	123,819	91,084		*	13,819	1,015,649
1999	36,087	245	10,281	140	10,281	7,945	90,252	128,978	305,812	163,351	183,255	99,939	*		17,533	1,054,121
2000	14,349	310	5,158	25	5,158	5,852	45,393	119,180	259,552	208,549	114,972	127,099	*		6,054	911,824
2001	9,007	185	3867	329	1,724	19,187	57,700	120,634	271,178	213,440	96,998	107,070	*	*	14,218	915,585
2002	11,616	67	3842	234	3,710	26,824	64,600	90,353	208,659	128,595	75,549	59,940	*	*	7,587	681,609
2003	15,312	36	4,047	246	1,868	3,881	100,701	155,515	346,412	123,450	121,043	172,065		*	8,486	1,053,119
2004	29,651	65	5,328	971	1,374	5,386	120,607	141,725	273,142	116,163	123,314	128,875			7,330	953,931
2005	17,189	120	3,073	0	341	25,515	148,127	110,456	378,659	103,628	66,701	49,278			3,913	907,000
2006	17,259	93	3676	1034	3,443	7,673	158,917	120,462	362,966	83,622	82,738	33,581			1,248	876,712
2007	9,309	70	2853	1230	885	15,077	164,331	131,109	309,215	97,361	56,463	34,486			7,379	829,767
2008	7,992	25	6,046	8866	6,012	15,159	140,418	80,003	381,993	71,655	84,789	24,658	*		15,624	843,762
2009	2,525	83	1217	4855	630	13,115	121,471	59,619	324,773	58,863	119,187	65,481			6,824	778,643
2010	2,624	80	277	4642	164	13,220	107,803	68,666	511,201	57,755	78,076	122,104	*	*	11,287	978,004
2011	2,700	129	368	1,521	20	56,963	129,065	90,631	715,162	29,010	103,856	61,960			25,601	1,216,986
2012	10,785	167	532	1,484	3,560	48,637	111,810	54,304	583,057	90,037	122,058	64,110		*	11,845	1,104,429
2013	1,826	106	2,499	2,244	2,638	32,573	89,300	80,811	539,775	32,290	84,385	33,980		*	17,246	919,953

Table 3. Harvest (in pounds) and value of the glass eel fishery in Maine and South Carolina from 2007 - 2013. **South Carolina landings are confidential.*

	Maine		South Carolina	
Year	**Landings**	**Value**	**Landings***	**Value**
2007	3,713	$1,287,485	No activity reported	
2008	6,951	$1,486,355	No activity reported	
2009	5,119	$519,559	No activity reported	
2010	3,158	$584,850	<500	<$100,000
2011	8,584	$7,653,331	<500	<$500,000
2012	20,764	$38,760,490	<5,000	<$2,500,000
2013	18,076	$32,926,991	<5,000	<$2,500,000

3. MANAGEMENT MEASURES

It is important to emphasize the 2012 American Eel Stock Assessment was a benchmark or baseline assessment that synthesized all available fishery-dependent and independent data, yet it was not able to construct eel population targets that could be related to sustainable fishery harvests. This is not an uncommon result of baseline stock assessments. The development of sustainable population and fishery thresholds will be a priority of future stock assessment. Despite the absence of fishery targets derived from population models, it is clear that high levels of yellow eel fishing occurred in the 1970s and 1980s in response to high prices offered from the export food market (Figure 4). For all coastal regions, peak catches in this period were followed by declining catches in the 1990s and 2000s, with some regions now at historic low levels of harvest. Given high catches in the past could have contributed to the current depleted status, it is prudent to reduce mortality while enhancing and restoring habitat. This approach is further justified in light of the public interest in eel population conservation demonstrated by two recent petitions to list American eel under the Endangered Species Act and the recent listing by the International Union for the Conservation of Nature (IUCN) as endangered on the IUCN Red List.

The provisions of this Addendum are a compliance requirement and are effective upon adoption of the Addendum as specified by the Board. Management measures include all mandatory monitoring and reporting requirements as described in this Section.

3.1 COMMERCIAL FISHERY MANAGEMENT PROGRAM

The 2012 American Eel Stock Benchmark Stock Assessment recommended mortality should be reduced on all life stages. Therefore, this addendum implements management measures to reduce overall mortality in order to maximize the conservation benefit to American eel stocks. States /jurisdictions shall maintain existing or more conservative American eel commercial fishery regulations, unless otherwise approved by the Board. States may always implement more conservative management measures.

3.1.1 GLASS EEL FISHERY MANAGEMENT PROGRAM

The following apply to the glass eel fisheries operating in Maine and South Carolina, unless otherwise noted.

Quota Management (Maine Only)
Maine's commercial glass eel quota for the 2015-2017 commercial glass eel fishing seasons will be set at 9,688 pounds annually. The quota shall be re-evaluated after three years (prior to the start of the 2018 fishing season), incorporating any information collected through Maine's life cycle monitoring program (see below), as well as other available programs, as feasible. Maine's commercial glass eel quota (9,688 pounds) may be extended through Board action. Any other modification (e.g. increase) to the quota amount will be subject to the Commission's addendum process.

Quota management provides a more reliable method to track mortality, increases accuracy of harvest data, and reduces opportunities for illegal harvest. In 2014 Maine pro-actively implemented new regulations to manage the glass eel fishery through output controls (quota management) instead of input control (gear and licenses restrictions). The state worked with industry and tribal representatives to develop a quota (11,479 pounds) that was a 35% reduction from 2012 landings. In 2014, the state landed 9,688 pounds.

Quota Overages
For any state or jurisdiction with a commercial glass eel quota, if an overages occurs in a fishing year, then that state or jurisdiction will be required to deduct the entire overage from the state's quota the following year, pound for pound.

Glass Eel Harvest Allowance Based on Stock Enhancement Programs
Any state or jurisdiction can request an allowance for commercial harvest of glass eels based on stock enhancement programs implemented after January 1, 2011. Examples of stock enhancement programs include, but are not limited to, habitat restoration projects, fish passage improvements, or fish passage construction. Fish passage projects may focus on upstream or downstream passage or both. Stock enhancement programs must show a measurable increase in glass eel passage and/or glass eel survival. Harvest shall not be restricted to the basin of restoration (i.e. harvest may occur at any approved location within the state or jurisdiction). Harvest requests shall not exceed 25% of the quantified contribution provided by the stock enhancement program.

Requests for harvest must be in writing and include a description of the: stock enhancement program, fishery requested, monitoring program to ensure harvest is not exceeded, monitoring program to ensure stock enhancement program targets are annually met, adequate enforcement capabilities, and adequate penalties for violations. The stock contribution percentage may be based on, for example, the amount of available suitable habitat that will become accessible, passage numbers, or other appropriate metrics.

Requests must be submitted to the Board by September 1st of the preceding fishing year. The Board will review and consider approval of the requests after a TC review.. After the first year of implementation the TC will evaluate the program and provide recommendations to the Board on the overall impact of and adherence to the plan. If the stock enhancement program cannot be assessed one year post-implementation, then a secondary review must occur within three years post-implementation. If changes to that habitat or fishway occurs in subsequent years, the Commission must be notified through the annual compliance report and a review of the harvest allowance may be initiated.

Reporting Requirements
Any state or jurisdiction with a commercial glass eel fishery is required to implement daily trip level reporting with daily electronic accounting to the state for both harvesters and dealers in order to ensure accurate reporting of commercial glass eel harvest. States or jurisdictions commercially harvesting less than 750 pounds of glass eels are exempt from this requirement.

Monitoring Requirements
Any states or jurisdiction with a commercial glass eel fishery must implement a fishery independent life cycle survey covering glass, yellow, and silver eels within at least one river system. If possible and appropriate, the survey should be implemented in the river system where the glass eel survey (as required under Addendum III) is being conducted to take advantage of the long term glass eel survey data collection. At a minimum the survey must collect the following information: fisheries independent index of abundance, age of entry into the fishery/survey, biomass and mortality of glass and yellow eels, sex composition, age structure, prevalence of *A. crassus,* and average length and weight of eels in the fishery/survey. Survey proposals will be subject to TC review and Board approval. States or jurisdictions commercially harvesting less than 750 pounds of glass eels are exempt from this requirement.

3.1.2 YELLOW EEL FISHERY MANAGEMENT PROGRAM

Currently, commercial yellow eel fisheries operate in all states with the exception of Pennsylvania and the District of Columbia. Management measures selected by the Board in Addendum III went into effect January 1, 2014. These measures included a 9 inch minimum size limit for both the commercial and recreational fishery and a ½ by ½ inch minimum mesh requirement for the commercial fishery.

The American Eel TC recommended commercial harvest be reduced from the 1998 – 2010 average (907,669 pounds), specifically a 12% reduction from the 1998-2010 average was seen as an acceptable precautionary approach (798,750 pounds).

Coastwide Catch Cap
The commercial yellow eel fishery is regulated through an annual coastwide catch cap set at 907,671 pounds (1998 – 2010 harvest level).

The use of a coastwide cap provides a flexible management system that responds to fluctuations in market conditions while providing a quantifiable conservation benefit to

American eels. One of the benefits of a catch cap is that it reduces the administrative and legislative burden of implementing a state specific quota system while still controlling the total amount of fishing mortality that is occurring annually. Additionally, a coastwide catch cap does not require a specific allocation by state or jurisdiction, which can be problematic due to the fluctuations in landings as a result of environmental and market conditions. However, under this system states and jurisdiction still need timely reporting in place to ensure that that the cap was not exceeded. Furthermore, a mortality cap may promote a derby style fishery, which could possibly flood the market and drive down prices.

Under the catch cap, there are two management triggers. Upon reaching either of these triggers, the Board is required to alter the management program as specified below in order to ensure the objectives of the management program are achieved.

Management Triggers

1. The coastwide catch cap is exceeded by more than 10% in a given year (998,438 pounds).
2. The coastwide catch cap is exceeded for two consecutive years, regardless of percent over.

Management Response

If either trigger is tripped, then there would be automatic implementation of a state-by-state commercial yellow eel quota. The annual coastwide quota is set at 907,669 pounds, with allocations as specified in Table 4. See Appendix A for a description on the allocation methodology. States and jurisdictions are required to approve regulations that would allow for implementation of a quota management program and timely monitoring of harvest no later than March 2016. This ensures if a management trigger is activated in the first year of implementation (2015) then the required management action could be taken. The quota management program must include a provision to address quota overages and allow quota transfers, as specified below. It is recommended monitoring and reporting requirements are sufficient to prevent repeated overages.

If the state-by-state quota system is implemented and a state or jurisdiction has an overage in a given fishing year, then the state or jurisdiction is required to reduce their following year’s quota by the same amount the quota was exceeded, pound for pound. For states that qualify for the automatic 2,000 pound quota, any overages would be deducted from the 2,000 pound allocation.

If the state-by-state quota system is implemented then any state or jurisdiction may request approval from the Board Chair or Commission Chair to transfer all or part of its annual quota to one or more states, including states that receive the automatic 2,000 pound quota. Requests for transfers must be made by individual or joint letters signed by the principal state official with marine fishery management authority for each state involved. The Chair will notify the requesting states within ten working days of the disposition of the request. In evaluating the request, the Chair will consider: if the transfer would preclude the overall annual quota from being harvested, the transfer addresses an unforeseen variation or contingency in the fishery, and if the transfer is consistent with the objects of the FMP. Transfer requests for the current fishing year must be submitted by December 31 of that fishing year.

The transfer of quota would be valid for only the calendar year in which the request is made. These transfers do not permanently affect the state-specific shares of the quota, i.e., the state-specific shares remain fixed. Once quota has been transferred to a state, the state receiving quota becomes responsible for any overages of transferred quota.

Under both the catch cap and quota systems all New York American eel landings (i.e. from both the yellow and silver eel fisheries) are included, until otherwise shown to preclude it. The Board has the ability to re-visit quota and allocation through subsequent addenda.

Table 4. Recommended Quota Allocation for the Commercial Yellow Eel Fishery. This quota would ONLY be implemented if wither management trigger is tripped.

	Initial Allocation	Final Quota
Maine	0.48%	3,907
New Hampshire	0.01%	2,000
Massachusetts	0.04%	2,000
Rhode Island	0.16%	4,642
Connecticut	0.19%	2,000
New York	4.26%	15,220
New Jersey	10.19%	94,899
Delaware	6.97%	61,632
Maryland	56.72%	465,968
PRFC	4.67%	52,358
Virginia	9.58%	78,702
North Carolina	4.94%	107,054
South Carolina		2,000
Georgia	0.11%	2,000
Florida	1.69%	13,287
Total	100%	907,669

3.1.3 SILVER EEL FISHERY MANAGEMENT PROGRAM

The following measures apply only to the commercial weir fishery in the New York portion of the Delaware River and its’ tributaries. New York was granted a one year extension from the requirements as specified under Section 4.1.3 of Addendum III:

> *Section 4.1.3: States and jurisdictions are required to implement no take of eels from September 1st through December 31st from any gear type other than baited traps/pots or spears (e.g. fyke nets, pound nets, and weirs). These gears may still be fished, however retention of eels is prohibited. A state or jurisdiction may request an alternative time frame for the closure if it can demonstrate the proposed closure dates*

encompass the silver eel outmigration period. Any requests will be reviewed by the TC and submitted to the Board for approval.

The American Eel Benchmark Stock Assessment found “fishing on out-migrating silver eels could be particularly detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled.” Conservation efforts on earlier life stages will only delay mortality and provide limited additional benefit to stock health if harvest occurs at later stages.

License Cap

The Delaware River silver eel weir fishery is restricted to nine annual permits. These permits are initially limited to those permitted participants that fished and reported landings from 2010 to 2013. Permits may be transferred thereafter.

3.1.4 STATE SPECIFIC SUSTAINABLE FISHERY MANAGEMENT PLANS FOR AMERICAN EEL

States or jurisdictions may petition the Board to allow for a state specific Sustainable Fishery Management Plan (Plan) for American Eel.

Currently, states and jurisdictions are allowed to petition the Board for an alternative management program, per Section 4.4 of the FMP. This section is not meant to replace Section 4.4 of the FMP, rather it provides guidance on specific types of alternative management the states can to request.

The objective of these programs is to allow states and jurisdictions the ability to manage their American eel fishery (glass, yellow, or silver) to both meet the needs of their current fishermen while providing conservation benefit for the American eel population. Three types of Plans (Fishing Mortality Based Plan, Transfer Plan, and Aquaculture Plan) are presented below. All plans must be submitted to the Board for their review and approval after TC review.

Fishing Mortality Based Plan

Under this scenario, states and jurisdictions may petition the Board for alternative management based on the current level of mortality that is occurring on their population. This Plan shall:

1. Require states or jurisdictions to assess, with some level of confidence, the status of eel abundance and current level of mortality (e.g. fisheries, natural, and other man-made) that is occurring on the American eel populations within their jurisdiction.
2. Once adequately documented, states or jurisdictions may allocate their fishing mortality to any American eel fishery (glass, yellow, or silver) even if the state does not currently participate in that fishery (i.e. a state would be allowed to open up a glass eel fishery if they did not currently have one due to the restrictions of the FMP). This could be applied for commercial, recreational, aquaculture industries and/or research set-aside purposes.

3. States may increase the fishing mortality rate provided it is offset by decreases in other mortality (e.g. though habitat improvements, increased fish passage, reduced turbine mortality, etc.) and there is an overall net gain to conservation (i.e. overall mortality is reduced, spawner escapement increases, etc...).

The format of the Fishing Mortality Based Plan is as follows:

1. Current regulations
2. Proposed change to regulations (e.g. request for fishery, fish passage restrictions, water quality improvements, etc...)
3. Description of fishing monitoring and enforcement capabilities
4. Description and supporting information on eel abundance and current mortality within state or jurisdiction
    a. Fishing mortality (including but not limited to commercial, recreational, sustenance, and bycatch)
    b. Natural mortality (including but not limited to predation and disease),
    c. Other man-made mortality (including but not limited to fish passage, turbines, habitat degradation, and pollution)
    d. Indices of abundance, age and size structure, and life cycle population metrics
5. Timeline for implementation of regulations, monitoring programs, or other activities
6. Description of conservation benefits of proposed regulatory changes or habitat improvements
7. Description of adaptive management program to evaluate success of proposed regulatory changes or habitat improvements

Transfer Plan

If states or jurisdictions are unable to assess the current level of mortality and abundance with certainty, and the state or jurisdiction implements quota management for at least one fishery, then a state may develop a Transfer Plan to request a transfer of quota from one fishery to another (e.g. from yellow to glass) based on the life history characteristic inherent to that area (e.g. state, river, or drainage). The request shall include: description of quota allocation by fishery; scientific analysis that the transfer will not increase overall eel fishing mortality, overall mortality, or reduce spawner escapement, with some level of confidence; description of monitoring program to ensure quota is not exceeded; and adequate enforcement capabilities penalties for violations.

Aquaculture Plan

States and jurisdictions may develop a Plan for aquaculture purposes. Under an approved Aquaculture Plan, states and jurisdictions may harvest a maximum of 200 pounds of glass eel annually from within their waters for use in domestic aquaculture facilities provided the state can objectively show the harvest will occur from a watershed that minimally contributes to the spawning stock of American eel. The request shall include: pounds requested; location, method, and dates of harvest; duration of requested harvest; prior approval of any applicable permits; description of the facility, including the capacity of the facility the glass eels will be held, and husbandry methods; description of the markets the eels will be distributed to; monitoring program to ensure harvest is not exceeded; and adequate enforcement capabilities penalties for violations. Approval of a request does not guarantee approval of a request in future years. Eels harvested under an approved Aquaculture Plan may not be sold until they reach the legal size in the jurisdiction of operations, unless otherwise specified.

All Plans are subject to TC and LEC review and Board approval. The Fishing Mortality Based Plan must be submitted by June 1st of the preceding fishing year in order to provide enough time for review for the upcoming fishing season. Transfer and Aquaculture Plans must be submitted by June 1st of the preceding fishing year and approval will be determined by the Board by September 1st. Plans will initially be valid for only one year. After the first year of implementation the TC will evaluate the program and provide recommendations to the Board on the overall impact of and adherence to the plan. If the proposed regulatory changes, habitat improvements, or harvest impact cannot be assessed one year post-implementation, then a secondary review must occur within three to five years post-implementation if the action is still ongoing.

If states use habitat improvements and changes to that habitat occurs in subsequent years, the Commission must be notified through the annual compliance report and a review of the Plan may be initiated. Any requests that include a stocking provision would have to ensure stocked eels were certified disease free according to standards developed by the TC and approved by the Board.

4. LAW ENFORCEMENT RECOMMENDATIONS

The Commission's Law Enforcement Committee has previously weighted in on the enforceability of proposed American eel management options based on the *Guidelines for Resource Managers on the Enforceability of Fishery Management Measures (July 2009).* These Guidelines rated management strategies using standard terms as follows, from least to most enforceable: Impossible, Impractical, Difficult and Reasonable.

The LEC concluded that status quo measures for all eel fisheries is impractical for enforcement, specifically for the glass eel fishery given the enforcement challenges associated with the prosecution of the glass eel fishery in those states currently closed to harvest of glass eels. A significant amount of illegal harvest of glass eels continues outside the two states where harvest is currently allowed, and illegally harvested eels are being possessed and shipped via those two states. State and federal enforcement agencies are tasked to thwart the illegal harvest and export with reduced staff and resources. Given the monetary value of glass eels and the ability to move illegally harvested eels via legal shipments, enforcement agencies do not have, and are unlikely to obtain the resources necessary to effectively monitor and control a limited glass eel harvest.

The LEC finds that a quota system would be difficult to enforce because of the variety of management strategies associated with quota implementation, enforceability depends largely on how quota systems are managed. Increased complexity of quota systems will generally reduce enforceability. The enforcement of time/area closures for the silver eel fishery is considered reasonable.

The LEC reports continuing illegal harvest of glass eels or elvers in the two states where some legal harvest is permitted, and in a number of states where any harvest of eels below a minimum size is prohibited. This is not unexpected given the high dollar value associated with the fishery. Enforcement agencies are dedicating resources to monitor and enforce regulations through stepped up patrols, coordination with local enforcement authorities, and by communicating the importance of glass eel cases to judiciary officials. Specific changes to regulations or statutes that would enhance field enforcement and/or penalties are encouraged, and those that have been implemented (in Maine, for example) have improved the outcome of arrests and convictions. Because of the cross-state nature of illegal glass eel harvest, strengthening of extradition or bail provisions for criminal violations would enhance the deterrent effect of enforcement actions.

5. COMPLIANCE

States and jurisdictions are required to approve regulations that would allow for implementation of a state-specific quota management program and timely monitoring of harvest no later than March 2016. To ensure this happens, state implementation plans that outline quota management programs and timely monitoring measures for eel fisheries are due for Board review and approval at the Commission's 2015 Annual Meeting.

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Appendix A

Determining the coastwide quota and state-by-state allocation

The coastwide quota and allocation is determined through a five step process. First, the quota is initially set at the 2010 harvest levels (978,004 pounds). This year (2010) was chosen as the baseline as it represents the last year of data that was included in the benchmark stock assessment and the assessment recommends reducing mortality from this level. Second, a 16% reduction is applied, bringing the quota to 821,523 pounds.

Third, the average landings for each states and jurisdiction from 2011 – 2013 is calculated. This time period was chosen in order to maintain the current distribution on fishing effort along the coast. The averages for each state and jurisdiction are totaled and then the percent contribution by each state is determined.

Fourth, in order to increase equity in the distribution of the quota, the following criteria is then applied to each state or jurisdictions allocation:

1. States or jurisdictions be allocated a minimum allocated quota fixed at 2,000 pounds in order to provide all state's a quota level sufficient to cover any directed or bycatch landings without creating an administrative burden. The 2,000 pounds quota is not expected to promote a notable increase in effort in the fishery.
2. No state or jurisdiction is allocated a quota that is more than 2,000 pounds above its 2010 commercial yellow eel harvest.
3. No state or jurisdiction is allocated a quota that is more than a 15% reduction from its 2010 commercial yellow eel harvest.

Through this filtering method the quota is updated to 893,909 pounds.

Lastly, the difference between this amount (893,909 pounds) and the TC recommendation (907,669 pounds) is 13,762 pounds. This difference is split equally among the states that are negatively impacted by the quota in comparison to their 2010 commercial harvest (Rhode Island, New Jersey, Delaware, PRFC, and North Carolina) with the exception of Maryland given their high allocation. Each of the specified states is allocated an equal portion of the 13,762 pounds, not to exceed their 2010 landings. This results in a final coastwide of 907,669 pounds.

Table 1. Quota and allocation calculation process.

	2010 Landings	2011-2013 Harvest Average	Initial Allocation Based on Harvest Average	Initial Quota	After Filtering Method is Applied	Final Quota
Maine	2,624	5,104	0.48%	3,943	3,907	3,907
New Hampshire	80	134	0.01%	82	2,000	2,000
Massachusetts	277	450	0.04%	329	2,000	2,000
Rhode Island	4,642	1,750	0.16%	1,314	3,946	4,642
Connecticut	164	2,073	0.19%	1,561	2,000	2,000
New York	13,220	46,058	4.26%	34,997	15,220	15,220
New Jersey	107,803	110,058	10.19%	83,713	91,633	94,899
Delaware	68,666	75,249	6.97%	57,260	58,366	61,632
Maryland	511,201	612,665	56.72%	465,968	465,968	465,968
PRFC	57,755	50,446	4.67%	38,365	49,092	52,358
Virginia	78,076	103,433	9.58%	78,702	78,702	78,702
North Carolina	122,104	53,350	4.94%	40,583	103,788	107,054
South Carolina	2			0	2,000	2,000
Georgia	103	1,162	0.11%	904	2,000	2,000
Florida	11,287	18,231	1.68%	13,802	13,287	13,287
Total	978,004	1,080,160	100%	821,523	893,909	907,669

Atlantic States Marine Fisheries Commission

2017 American Eel Stock Assessment Update

October 2017

Sustainably Managing Atlantic Coastal Fisheries

Atlantic States Marine Fisheries Commission

2017 American Eel Stock Assessment Update

Prepared by the
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EXECUTIVE SUMMARY

The management unit for American eel under the jurisdiction of Atlantic States Marine Fisheries Commission (ASMFC or Commission) includes that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida. The goal of the American Eel Fishery Management Plan (approved November 1999) is to conserve and protect the American eel resource to ensure ecological stability while providing for sustainable fisheries.

In the U.S., all life stages are subject to fishing pressure, and the degree of fishing varies. Glass eel fisheries are permitted in Maine and South Carolina. Yellow eel fisheries exist in all Atlantic Coast states with the exception of Pennsylvania. Eels are harvested for food, bait, and export markets.

During 1950 to 2016, Atlantic coastwide U.S. American eel landings ranged between approximately 664,000 pounds in 1962 and 3.67 million pounds in 1979. The highest landings in the time series occurred from the mid-1970s to the early 1980s after which they declined. Since the 1990s, landings have been lower than historical landings but they have been stable in recent decades.

Very few fishery-independent surveys target American eels (with the exception of the state-mandated young-of-year surveys and a few surveys in Maryland). All fishery-independent surveys used in the 2012 benchmark stock assessment were updated for this report, with some noted exceptions, and most were standardized using a generalized linear model to account for changes in catchability of American eels. Regional indices were also developed for both YOY and yellow eel stages.

Trend analyses of abundance indices provided evidence of neutral or declining abundance of American eels in the U.S in recent decades. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant downward trends in some indices. The Mann-Kendall test detected a significant downward trend in 6 of the 22 YOY indices, 5 of the 15 yellow eel indices, 3 of the 9 regional trends, and the 30-year and 40-year yellow-phase abundance indices. The remaining surveys tested had no trend, except for two which had positive trends. The Manly meta-analysis showed a decline in at least one of the indices for both yellow and YOY life stages. For the ARIMA results, the probabilities of being less than the 25th percentile reference points in the terminal year for each of the surveys were similar to those in ASMFC 2012 and currently 3 of the 14 surveys in the analysis have a greater than 50% probability of being less than the 25th percentile reference point. Overall, the occurrence of some significant downward trends in surveys across the coast remains a cause for concern.

Reference points for determining the stock status of American eel in the U.S. in ASMFC 2012 were developed using the Depletion-Based Stock Reduction Analysis (DB-SRA) model which was not accepted for management use by the Peer Review Panel. The DB-SRA was not updated for this report because the Panel recommended it be further developed which was outside the

guidelines of a stock assessment update. Therefore neither reference points nor stock status could be determined quantitatively by this stock assessment update. Compared to the 2012 benchmark stock assessment, the ARIMA had similar results and there were more significantly downward trends in indices as indicated by the Mann-Kendall test in this update. The trend analysis and stable low landings support the conclusion that the American eel population in the assessment range is similar to five years ago and remains depleted.

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1 INTRODUCTION

The purpose of this assessment was to update the 2012 American eel (*Anguilla rostrata*) benchmark stock assessment (ASMFC 2012) with recent data from 2010-2016. No changes in structure were made to the index standardization or modeling approaches. The 2012 benchmark stock assessment and this stock assessment update for American eel was initiated by the Atlantic States Marine Fisheries Commission (ASMFC or Commission) American Eel Management Board, prepared by the ASMFC American Eel Stock Assessment Subcommittee (SAS), and reviewed and approved by the ASMFC American Eel Technical Committee (TC) as part of the interstate fisheries management process.

1.1 Fisheries Management

The ASMFC American Eel Management Board (Board) first convened in November 1995 and finalized the Interstate Fishery Management Plan (FMP) for American Eel in November 1999 (ASMFC 2000a). The goal of the FMP is to conserve and protect the American eel resource to ensure ecological stability while providing for sustainable fisheries. The FMP requires all states and jurisdictions to implement an annual young-of-year (YOY) abundance survey to monitor annual recruitment of each year's cohort (ASMFC 2000a, 2000b). In addition, the FMP requires a minimum recreational size and possession limit and a state license for recreational fishermen to sell eels. The FMP requires that states and jurisdictions maintain existing or more conservative American eel commercial fishery regulations for all life stages, including minimum size limits. Each state is responsible for implementing management measures within its jurisdiction to ensure the sustainability of its American eel population.

In August 2005, the Board directed the American Eel Plan Development Team (PDT) to initiate an addendum to establish a mandatory catch and effort monitoring program for American eel. The Board approved Addendum I at the February 2006 Board meeting.

In January 2007, the Board initiated a draft addendum with the goal of increasing escapement of silver eels to the spawning grounds. In October 2008, the Board approved Addendum II, which placed increased emphasis on improving the upstream and downstream passage of American eel. The Management Board chose to delay action on management measures in order to incorporate the results of the 2012 stock assessment.

In August 2012, the Board initiated Draft Addendum III with the goal of reducing mortality on all life stages of American eel. The addendum was initiated in response to the findings of the 2012 Benchmark stock assessment, which declared American eel stock along the US East Coast as depleted. The Management Board approved Addendum III in August 2013.

Addendum III requires states to reduce the yellow eel recreational possession limit to 25 eel/person/day, with the option to allow an exception of 50 eel/person/day for party/charter employees for bait purposes. The recreational and commercial size limit increased to a minimum of 9”. Eel pots are required to be constructed with a minimum of ½” by ½” mesh size. The glass eel fishery is required to implement a maximum tolerance of 25 pigmented eels per pound of glass eel catch. The silver eel fishery is prohibited in all states from September 1st to December 31st from any gear type other than baited traps/pots or spears. The addendum also set minimum monitoring standards for states and required dealer and harvester reporting in the commercial fishery.

In October 2014, the Board approved Addendum IV. The addendum was also initiated in response to 2012 American Eel Benchmark Stock Assessment and the need to reduce mortality on all life stages. The Addendum established a coast-wide cap of 907,671 pounds of yellow eel, reduced Maine's glass eel quota to 9,688 pounds (2014 landings), and allowed for the continuation of New York's silver eel weir fishery in the Delaware River. For yellow eel fisheries, the coast-wide cap was implemented starting in the 2015 fishing year and established two management triggers: (1) if the cap is exceeded by more than 10% in a given year, or (2) the coast-wide quota is exceeded for two consecutive years regardless of the percent overage. If either one of the triggers are met then states would implement state-specific allocation based on average landings from 1998-2010 with allocation percentages derived from 2011-2013.

1.1.1 Management Unit Definition

The American eel is a catadromous species in North America that historically occurred in all major rivers from Canada through Brazil. The management unit for American eels under the jurisdiction of ASMFC includes that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

1.1.1.1 Commercial Fishery Management

1.1.1.1.1 Glass Eel / Elver Fishery

Glass eel and elver harvest along the Atlantic coast is prohibited in all states except Maine and South Carolina. In recent years, Maine was the only state reporting substantial glass eel or elver harvest. Maine implemented regulatory changes that increased elver and large eel license fees in 1996. In addition to generating revenue for enforcement and eel research, these changes set both a harvest season and closures during the harvest season. The amount of gear, type, and configuration was limited to control fishing effort. Additional measures included restrictions on allowable fishing areas, number of license holders, and a prohibition on fishing within 46 m of a dam (CAEMM 1996). South Carolina could not determine participation in the elver and glass eel fishery in coastal waters until a limited entry permit system was instituted in 1996 (B. McCord, South Carolina Department of Natural Resources, pers. comm.). Ten permits are available to both in-state and out-of-state residents. Permit holders abide by monthly effort controls and must report their harvest. There was interest in developing commercial glass eel fisheries in Connecticut, New Jersey, Virginia, and Florida. Connecticut regulations were minimal until 1996 when the state defined the glass eel as less than 10 cm in length, instituted a glass eel fishing season with a weekly closed period, limited traps, and required monthly catch reporting logbooks. Connecticut prohibited the take or attempted take of glass eels, elvers, and silver eels in 2002. The glass eel and elver fishery in New Jersey was unregulated prior to 1997 when a fishery season was allowed for dip nets only for that one year, followed by full closure in 1998. In Virginia, a six-inch minimum size was passed in 1977. Florida passed regulations in 1998 such that the eel fisheries operate under gear restrictions that prevent the landings of eels under six inches.

Prior to the implementation of the FMP, Maine was the only state compiling glass eel and elver fishery catch statistics. Under the FMP, all states are now required to submit fishery-dependent information. Given the high value, poaching of glass eels and elvers is known to be a serious problem in several states, but enforcement of the regulations is limited due to the nature of the fishery (very mobile, nighttime operation, high value for product, low administrative priority). Addendum IV (ASMFC 2014) to the FMP allows approved Aquaculture Plans from states and jurisdictions to harvest up to 200 pounds of glass eel annually from within their state waters for use in domestic aquaculture activities. The American Eel Farm (AEF) in North Carolina is the only facility to have applied and been approved for domestic aquaculture, which they have done annually since 2016. Fishing did not take place in 2016 due to permitting issues in North Carolina. In 2017, a total of 0.25 pounds of glass eels were harvested of the 200 pound quota. North Carolina Division of Marine Fisheries submitted an amended plan on behalf of AEF for 2018-2020 which was approved by the Board in August 2017.

1.1.1.1.2 Yellow / Silver Eel

The yellow American eel fishery in Maine occurs in both inland and tidal waters. Large eel fisheries in southern Maine are primarily coastal pot fisheries managed under a license requirement, minimum size limit, and gear and mesh size restrictions. New Hampshire has monitored its yellow eel fishery since 1980; effort reporting in the form of trap haul set-over days for pots or hours for other gears has been mandatory since 1990. Small-scale, commercial eel fisheries occur in Massachusetts and Rhode Island and are mainly conducted in coastal rivers and embayments with pots during May through November. Connecticut has a similar small-scale, seasonal pot fishery for yellow eels in the tidal portions of the Connecticut and Housatonic rivers (S. Gephard, Connecticut Department of Energy and Environmental Protection, pers. comm.). All New England states presently require commercial eel fishing licenses and maintain trip level reporting.

Licensed eel fishing in New York occurred primarily in Lake Ontario (prior to the 1982 closure), the Hudson River, the upper Delaware River (Blake 1982), and in the coastal marine district. A slot limit (greater than 6 inches and less than 14 inches to limit PCB exposure) exists for eels fished in the tidal Hudson River (from the Battery to Troy and all tributaries upstream to the first barrier), strictly for use as bait or for sale as bait only. Due to PCB contamination of the main stem, commercial fisheries have been closed on the freshwater portions of the Hudson River and its tributaries since 1976. The fishery in the New York portion of the Delaware River consists primarily of silver eels collected in a weir fishery. In 1995, New York approved a size limit in marine waters. New Jersey fishery regulations require a commercial license, a minimum mesh, and a minimum size limit. A minimum size limit was set in Delaware in 1995. Delaware mandated catch reporting in 1999 and more detailed effort reporting in 2007.

Maryland, Virginia, and Potomac River Fisheries Commission have primarily pot fisheries for American eels in Chesapeake Bay. Large eels are exported whereas small eels are used for bait in the crab trotline fishery. Catch reports were not required in Virginia prior to 1973 and Maryland did not require licenses until 1981. Effort reporting was not required in Maryland until 1990. The Potomac River Fisheries Commission has had harvester reporting since 1964, and has collected eel pot effort since 1988.

North Carolina has a small, primarily coastal pot fishery. A trip ticket system began in 1994 and a commercial logbook system began in 2007. The majority of landings come from the Albemarle Sound area and additional landings reported from the Pamlico Sound and “other areas.” No catch records are maintained for freshwater inland waters. Landings for “other areas” reported by the state come from southern waterbodies under the jurisdiction of NCDMF. South Carolina instituted a permitting system over ten years ago to document total eel gear and commercial harvest. Traps, pots, fyke nets, and dip nets are permitted in coastal waters. Fishing for eels in coastal waters is often conducted under the guise of fishing for crabs.

American eel fishing in Georgia was restricted to coastal waters prior to 1980 when inland fishing was permitted (Helfman et al. 1984). Catch, but not effort, data are available because no specific license is required to fish eels. The Florida pot fishery has a minimum mesh size requirement in the fishery and it is operated under a permit system.

Current commercial fisheries regulations can be found in Table 1.

1.1.1.2 Recreational Fishery

Few recreational anglers directly target American eels and most landings are incidental when anglers are fishing for other species. Eels are often purchased by recreational fishermen for use as bait for larger sport fish such as striped bass, and some recreational fishermen may catch their own eels to use as bait. Current recreational management regulations can be found in Table 2.

1.2 Stock Assessment History

In 2005, a stock assessment for American eel was conducted by the ASMFC and reviewed by a panel of independent experts (ASMFC 2005). The peer review panel recognized sufficient shortcomings with the assessment to warrant additional action prior to its use for future technical and management purposes (ASMFC 2006a). The 2005 stock assessment was not accepted by the Board; therefore, the stock status of American eel was deemed unknown by the ASMFC.

At the February 22, 2006 meeting of the Board, the American Eel Stock Assessment Subcommittee (SAS) and Technical Committee (TC) were tasked with reviewing the recommendations from the peer review advisory report and recommending a follow-up plan. Subsequently, a report was issued in October of 2006 containing updated datasets and the short-term analyses suggested by the review panel (ASMFC 2006b).

The 2012 benchmark stock assessment represented the most recent work performed by the ASMFC to ascertain stock status since 2006. Analyses and results indicated that the American eel stock had declined and that there were significant downward trends in multiple surveys across the coast. It was determined that the stock was depleted but no overfishing determination could be made based on the analyses performed. This report is an update to the 2012 benchmark stock assessment report.

1.3 Petitions for ESA Listing

In response to the extreme declines in American eel abundance in the Saint Lawrence River-Lake Ontario portion of the species' range (personal comm., Dr. John Casselman, DFO), the ASMFC requested that the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS) conduct a status review of American eels in 2004. The ASMFC also requested an evaluation of a Distinct Population Segment (DPS) listing under the Endangered Species Act (ESA) for the Saint Lawrence River/Lake Ontario and Lake Champlain/Richelieu River portion of the species range, as well as an evaluation of the entire Atlantic coast American eel population. A preliminary status review conducted by USFWS determined that American eel was not likely to meet the requirements of DPS determinations. However, the USFWS initiated a coastwide status review of the American eel in coordination with the NMFS and ASMFC. At this same time, two private citizens submitted a petition to the USFWS and NMFS to list American eel under the ESA.

In February 2007, the USFWS announced the completion of a Status Review for American Eel (50 CFR Part 17; USFWS 2007). The report concluded that protecting eels as an endangered or threatened species was not warranted. The USFWS did note that while the species' overall population was not in danger of extinction or likely to become so in the foreseeable future, the eel population has "been extirpated from some portions of its historical freshwater habitat over the last 100 years… [and the species abundance has declined] likely as a result of harvest or turbine mortality, or a combination of factors".

In 2010, the Center for Environmental Science Accuracy and Reliability filed a petition to the USFWS to consider listing the American eel on the endangered species list. The proposal was based on new information that had become available since the last status review. In September 2011, the USFWS published a positive 90-Day Finding, which stated that the petition contained enough information to warrant conducting a status review (USFWS 2011).

In 2015, USFWS announced that the American eel population is stable and protection under ESA was not warranted although the agency did recommend continuing efforts to maintain healthy habitats, monitor harvest levels, and improve river passage (USFWS 2015). Conversely, the International Union for the Conservation of Nature (IUCN) listed American eel as "Endangered" on the Red List in 2014 (Jacoby et al. 2014). While this has no legal implications, it is an important metric and the Commission remains committed to closely monitoring this species and making management adjustments as necessary.

2 LIFE HISTORY

American eels are found from the southern tip of Greenland, Labrador and the northern Gulf of St. Lawrence in the north, south along the Atlantic and Gulf coasts of North America and eastern Central America to the northeast coast of South America, and into the inland areas of the Mississippi and Great Lakes drainages (Tesch 1977). The American eel is regarded as a single, panmictic breeding population. American eels are found in a variety of habitats throughout their life cycle, including the open ocean, large coastal tributaries, small freshwater streams, and lakes and ponds. They are opportunistic feeders that will eat, depending on their life stage, phytoplankton, zooplankton, insects, crustaceans, and fish. Individuals grow in freshwater or estuarine environments for anywhere from 3 to 30 or more years before maturing and returning to the ocean as adults to spawn and die.

American eels are confronted with many environmental and human-induced stressors which affect all life stages and may reduce survival. Since all anthropogenic eel mortality is pre-spawning, reproduction can be reduced by these cumulative pressures. Commercial harvest occurs at all American eel life stages (glass, elver, yellow, and silver). Blockages and obstructions that limit upstream migration of American eels have reduced habitat availability and limited the range of the species. Dams may also limit or delay downstream movements of spawning adults. Additionally, downstream mortality may be caused by hydroelectric facilities by impingement or turbine passage. Freshwater habitat degradation resulting in reduced food productivity increases mortality of the freshwater life stages. Predation by fish, birds, and mammals can impact eel populations during all life stages. The non-native swim bladder parasite, *Anguillicoloides crassus*, can decrease swimming ability and reduce the silver eel's ability to reach the spawning grounds. Contaminants also may reduce the reproductive success of American eels because they have a high contaminant bioaccumulation rate (Couillard et al. 1997). Oceanographic changes influencing larval drift and migration may reduce year-class success. American eel, as a panmictic species, could be particularly vulnerable to drastic oceanic variations. An understanding of the requirements of the American eel's different life stages is needed to protect and manage this species.

The following sections have been condensed and also updated with new research since the 2012 benchmark assessment report. Refer to ASMFC 2012 for more a more detailed discussion of life history.

2.1 Stock Definitions

The American eel is a panmictic species, with a single spawning stock that reproduces in the Sargasso Sea. Eel larvae (leptocephali) are broadly dispersed by ocean currents along the Atlantic coasts of northern South, Central, and North America. Genetic research indicates that there is no reproductive isolation of American eels migrating from the Atlantic Coast. Further, any genetic differentiation is a result of natural selection upon a particular cohort within a geographic area rather than actual genetic differences within the species (Avise et al. 1986; Wirth and Bernatchez 2003; Cote et al. 2009).

2.2 Migration Patterns

American eels may travel thousands of miles in their lifetime. They are a catadromous fish that spawn in the Sargasso Sea, and the larvae drift on ocean currents until they reach the eastern seaboard of North America. Young eels (glass or elver stage) actively swim upstream to reach estuarine and freshwater habitats, sometimes hundreds of miles upriver. The young eels spend between 3 and 30 or more years in estuarine or freshwater habitats before maturing and migrating back downstream and to the Sargasso Sea to spawn. Since the 2012 assessment, a study on chemical cues was published indicating that diluted odors emitted by glass eels were detected by other glass eels in a laboratory setting and suggested coordinated inland migration (Schmucker et al. 2016). This was expanded by Galbraith et al. (2017) to suggest that cues may be life-stage specific so that one year class of glass eels moving inland may be responding to cues from the previous year class as guidance.

2.3 Life Cycle

American eels undergo six distinct life stages. The life cycle begins when the eggs hatch and leptocephali (larvae) are carried by ocean currents from the spawning grounds in the Sargasso Sea. The prevailing currents along coastal areas disperse the leptocephali, which metamorphose into glass eels on the continental shelf. Glass eels move toward inland areas and become pigmented elvers before or during their entry into coastal estuaries. Elvers and yellow eels settle in habitats ranging from estuaries to far upstream freshwater reaches. Eels reach the silver stage at maturity and return to the Sargasso Sea, where they spawn and die.

2.4 Life History Characteristics

2.4.1 Age

The age of American eels can be determined by taking transverse sections of the sagittal otoliths. Two otolith processing techniques (embedding and sectioning or grinding and polishing) are accepted ageing methods by the ASMFC (ASMFC 2001). American eel otolith ageing methods have been described by Liew (1974), Chisnall and Kalish (1993), and Oliveira (1997). Since the 2012 benchmark stock assessment, the ASMFC organized an American eel otolith sample exchange. This project determined that laboratories and state agencies that age American eel along the Atlantic coast were using different processing and reading methods that resulted in a high degree of imprecision and bias across laboratories and readers (ASMFC 2017). Because of these results, the ASMFC will hold an ageing workshop for American eel in January 2018 to standardize sample preparation and reading protocols for agers.

2.4.2 Growth

Slower growth occurs in more northern portions of the American eel’s distribution compared to the south (Helfman et al. 1984; Richkus and Whalen 1999; Jessop 2010). Male maximum size is the same throughout their distribution (Jessop 2010) However, female eels reach a larger maximum size in the northern portion of their range compared to the south (Jessop 2010). Eel growth is related to seasons, with most growth occurring during spring through fall and very little growth in the winter (Helfman et al. 1984). The shorter growing seasons in the higher latitudes may explain why eels experience slower growth in the northern portions of their range. Growth rates are highly variable among fish within the same watershed and of the same sex thus total length is not an accurate predictor of age.

2.4.3 Reproduction

The sex of American eels can be determined by gross morphological examination (Vladykov 1967; Krueger and Oliveira 1997). Differentiation between sexes occurs in the yellow eel stage of American eels and maturity-at-length varies by sex and latitude (Dolan and Power 1977; Oliveira and McCleave 2000; Goodwin and Angermeier 2003; Morrison and Secor 2003; Tremblay 2009). Sex ratios by location are also variable with males found more commonly in downriver sites and females more common in upriver sites (Facey and Helfman 1985; Helfman et al. 1983; Krueger and Oliveira 1999; Oliveira and McCleave 2000; Goodwin and Angermeier 2003; Davey and Jellyman 2005) and Oliveira and McCleave (2000) found that yellow eels >400 mm and silver eels >425 mm were exclusively female. Sex-linked migration patterns are another possible explanation for why male American eels are typically found in coastal habitats while females tend to be found in more upstream areas (Jessop 2010). Females are found in habitats that are less densely populated with eels so sex may not be a function of density dependence but rather that female eels migrate further upstream than males (Jessop 2010). Fecundity estimates are higher in the northern portion of the eel’s range because of the larger sizes of migrating female eels from northern areas (Barbin and McCleave 1998). American eels are thought to spawn in the Sargasso Sea during late winter through spring, but spawning has never been observed. It is also unknown if they have paired or group spawning. Because no spent eel has ever been documented, it is assumed that American eels are semelparous.

2.4.4 Food Habits

American eel diet varies greatly depending on life stage and habitat. American eel leptocephali and glass eel feeding habits have not been reported. However, the dentition and gape of the mouth suggest that they are capable of feeding on individual zooplankton and phytoplankton. Prey size increases as eels grow, with elvers and small yellow eels consuming mostly benthic macroinvertebrates and larger yellow eels switching primarily to crayfish and fish. Silver eels are thought not to eat during their migration to the Sargasso Sea.

2.4.5 Natural Mortality

Very little is known about the natural mortality of American eels. Since eels are highly fecund (Wenner and Musick 1974; Barbin et al. 1998; Tremblay 2009), natural mortality is likely very high, particularly during the early life stages. Eel survival is likely impacted by changes in oceanographic conditions, predation, and the spread of the non-native swim bladder nematode *Anguillicoloides crassus*. ASMFC 2012 describes each of these threats to the American eel in detail, with recent studies adding information regarding *A. crassus*. Waldt et al. (2013) found that nearly 50% of American eels in a Hudson River tributary in New York were infected during the fall of 2009. Zimmerman and Welsh (2012) confirmed the presence of *A. crassus* in the upper Potomac River watershed and found that length-at-age was lower in previously infected American eels than those uninfected, potentially reducing reproductive capabilities. Hein et al. (2014) reevaluated *A. crassus* infection in South Carolina where the American eel population has been declining since 2001 and the infection was first reported nearly 20 years ago. That study found that parasite prevalence was higher in South Carolina than in New York and Chesapeake Bay and possibly has been increasing over time. Additionally, the authors suggest that milder winters due to climate change could increase infection.

2.4.6 Incidental Mortality

Incidental mortality, caused by anthropogenic activities other than harvest, can be attributed to habitat alterations and restrictions as well as mechanical and chemical injuries. Inland habitat alterations and restrictions come primarily in the form of barriers to upstream migration for American eels. These can either be physical (dams) or chemical (areas of poor water quality) factors that limit habitat use by eels. This compression of range through habitat restrictions may increase the level of predation mortality or contribute to density dependent effects on growth or reproductive success. The location and number of dams may restrict eel distribution by limiting upstream movements (Levesque and Whitworth 1987; Goodwin and Angermeier 2003; Verreault et al. 2004; Machut et al. 2007; Hitt et al. 2012) and could impact the total number, size distribution, and number of eggs produced from a river system (Sweka et al. 2014).

3 HABITAT DESCRIPTION

3.1 Brief Overview

Section 3 provides a short description of American eel habitat use. A detailed review of American eel habitat requirements can be found in the Atlantic Coast Diadromous Fish Habitat document (Greene et al. 2009). Habitat descriptions by life history stage can be found in Section 3 of ASMFC 2012.

American eels exhibit a highly complex catadromous life cycle and are found in marine, brackish, and freshwater habitats (Adams and Hankinson 1928; Facey and LaBar 1981; Facey and Van Den Avyle 1987; Helfman et al. 1984). Habitat types used by different phases of eels include open ocean, estuaries, rivers, streams, lakes (including land-locked lakes), and ponds (Facey and Van Den Avyle 1987).

American eel habitat associations and requirements vary by life stage. After hatching in winter and spring in the Sargasso Sea, larval American eels passively migrate to the continental shelf along the east coast of North America where they metamorphose into glass eels (Greene et al. 2009). After developing pigment (becoming elvers), some eels start migrating upstream into freshwater while others remain in coastal rivers and estuaries. Upstream migration may continue throughout the yellow phase as well. During maturation, silver eels migrate downstream to the ocean and return to the Sargasso Sea to spawn before dying (Haro and Krueger 1991).

4 FISHERY DESCRIPTION

The American eel fishery has a long history in the U.S., and a description of the current and documented historical fisheries can be found in ASMFC 2012. A summary follows and includes any new or updated information.

4.1 Commercial Fisheries

4.1.1 Glass Eel Fishery

Glass eel fisheries along the Atlantic coast are prohibited in all states except Maine and South Carolina. Over the last seven years, there has been an increase in the demand for glass eel due to concerns over population levels of European and Japanese eels, as well as tighter restrictions on the export of European eel. Harvest, by dip net or fyke net, has increased as the average market price has risen to over $1,000 per pound with peaks exceeding $2,000 per pound. The highest value reported in Maine in the last five years was $40.38 million in 2012 for 21,611 pounds ($1,868 per pound). Since the implementation of Addendum IV (ASMFC 2014), Maine's glass eel quota has been set at 9,688 pounds (a 17.5% reduction from the 2014 quota). In 2017, preliminary landings indicate 9,282 pounds of glass eels were sold for a value of $12.08 million ($1,301 per pound).

4.1.2 Yellow Eel Fishery

Historically and currently, the majority of commercial landings come from the yellow eel fishery. Accounts of eel harvest date back to colonial times, with some commercial fishery harvest records available beginning in the late 1880s, but consistent record keeping began in 1950. After an initial decline in the 1950s, commercial yellow eel landings increased to a peak of 3.67 million pounds in 1979, declined again in the 2000s, and have exceeded one million pounds three times since 2004. Addendum IV (2014) implemented a coastwide cap of 907,671 pounds and two management triggers: (1) the coastwide cap is exceeded by more than 10% in a given year and (2) the coastwide cap is exceeded for two consecutive years, regardless of the percent over. If triggered, there is an automatic implementation of state-by-state quota as laid out in Addendum IV. In 2016, U.S. Atlantic coast preliminary yellow eel landings totaled 928,358 pounds which is above the cap although these landings are not final. Management triggers will be evaluated once landings are final. Eel pots are the typical gear used in the commercial yellow eel fishery; however, weirs, fyke nets, and other fishing methods are also employed. Although yellow eel were harvested for food historically, today's fishery sells yellow eel primarily as bait for recreational fisheries.

4.1.3 Silver Eel Fishery

Since the approval of Addendum IV (2014), silver eel fisheries are only permitted on a limited basis in the Delaware River (NY). The Delaware River eel weir fishery is restricted to nine annual permits which were initially limited to those who fished and reported landings from 2010 to 2013.

4.1.4 Bait Fishery

The use of harvested American eels for bait in other fisheries is not well-described, although it does not appear to have been common before the 20th century nor had the relative importance of food markets. Eel harvesting in the South Atlantic Bight prior to the 1970s was focused primarily on harvesting eels for live bait in sport fisheries and secondarily as bait for blue crab pots (Van Den Avyle 1984). Harvesting eels for crab trotline bait was important in the Maryland eel fishery in the 20th century (Foster and Brody 1982). The proportion of the eel harvest sold for bait declined with the advent of the overseas food market in the 1960s, and this disposition declined further as the increased use of crab pots reduced the need for baited trotlines (Lane 1978).

A more recent development in the marketing of U. S. caught American eels is the use of eels as bait in recreational striped bass, cobia, and catfish fisheries. Several references that summarize U.S. eel fisheries prior to the 1990s (Fahay 1978; Lane 1978; Van Den Avyle 1984) do not mention this harvest disposition, and more recent references mention the practice with no details (Haro et al. 2000; Collette and Klein-MacPhee 2002). It is likely that the practice of rigging eels for striped bass angling originated early in the 20th century but did not become widespread until recently. Presently, the use of eels as striped bass bait is probably the dominant use of harvested eels in New England and comprises a larger proportion of the Chesapeake Bay eel fishery than any time previous. U.S. eel fishery data does not have the resolution to separate striped bass bait from other dispositions. Commercial eel fishery reporting since the implementation of the ASMFC eel management plan in 2001 has improved and could provide information on this recent development.

4.1.5 Exports

The weight and value of U.S. domestic exports of American eels from selected districts along the Atlantic coast for 1981–2016 were provided by the NMFS (1981–1988; Fisheries Statistics Division, Silver Spring, MD, pers. comm.) and the United States International Trade Commission (USITC) DataWeb (1989–2016; pers. comm.). Export values were converted to 2016 dollar values using conversion factors based on the annual average consumer price index (CPI) values, which were obtained from the U.S. Bureau of Labor Statistics (pers. comm.).

Prior to 1989, exports were classified as either fresh/frozen or live. Since 1989, the fresh/frozen group has been separated into two categories—fresh (or fresh or chilled) and frozen. Live export weight data for American eels were not available for the 1989–1992 time period, likely due to differences in reporting requirements during those years (A. Lowther, NOAA Fisheries, pers. comm.; M. Savage, USITC, pers. comm.).

Domestic exports of American eels from the Atlantic coast ranged from 229,000 to over 6.1 million pounds per year from 1981 through 2016 (Figure 1). Live eels comprised the majority (>50%) of exports in 1983–1988, 1993, 1999, and 2003–2005. From 2006-2011, exports of fresh and frozen eels accounted for an average of 75% of the total eel exports per year. The reason that the magnitude of domestic exports exceeds commercial landings in some years may be that export landings records include significant quantities of hagfish misreported as American eel. Since 2011, there have been no fresh or frozen American eel exports and 100% of the exports came from live American eel.

The value of American eel exports ranged from $2.0 to $39.6 million per year over the time series (Figure 1). Export values decreased during the earliest years in the time series and then generally increased to the peak observed in 1997. The value of exports substantially dropped following the 1997 peak but has shown a generally increasing trend through 2011 after which there were no fresh or frozen American eels exported.

The value per pound of exported American eels classified as live was above the value per pound of fresh and frozen eels (combined) throughout the time series (Figure 2). The value per pound of fresh and frozen eels ranged from $0.81 to $5.47 per pound per year from 1981 to 2016. The value per pound of fresh and frozen eels has exhibited a general decline over the time series except for one peak in 2003. The value per pound of live exports has varied over the available time series, ranging from $2.78 to $73.41 per pound per year.

4.2 Commercial Catch-Per-Unit-Effort

Fishery-dependent catch-per-unit-effort (CPUE) was available in some states, but following a review of these data by the SAS they were not considered indicative of trends in the stock as a whole and therefore were not updated for this stock assessment report. Note that fishery-dependent CPUE is almost exclusively composed of positive trips only; trip reports with zero eels caught are rare because most agencies do not require reports of zero catches. Furthermore, differences in baiting practices and bait preference vary geographically and that can confound the accuracy of commercial CPUE.

4.3 Recreational Fisheries

Studies and reports that summarize U.S. eel fisheries provide little information on targeted recreational eel fisheries (Bigelow and Schroeder 1953; Fahay 1978; Lane 1978; and Van Den Avyle 1984). The practice of spearing or gigging eels buried in the mud during winter is an eel fishing method that was developed for subsistence fishing but came to have both commercial and sportfishing appeal in the 19th century until recently. Eels are encountered over much of their U.S. range by recreational anglers as bycatch. Van Den Avyle (1984) reported that no major sport fishery for American eels occurred in coastal rivers of the South Atlantic Bight, but incidental catches were made by anglers in estuaries and rivers. Despite the incidental nature of eel hook-and-line catches, the Marine Recreational Information Program (MRIP) does encounter enough observations to generate catch estimates that indicate widespread and common presence as a bycatch species. Starting with 1981 estimates, the MRIP survey for all major eastern U.S. regions show higher catch estimates in the 1980s than in the 2000s on average.

There is also a subsistence component to the American eel fishery. The harvest of American eels as a food source for subsistence has been portrayed as having importance for Native Americans and European settlers in North America with declining importance after the 19th century. Most accounts are anecdotal and entail brief references in popular literature. It is likely that changes in eel abundance and demand have diminished this practice in the 20th century resulting in declining cultural importance of eels in coastal communities.

4.4 Gulf of Mexico

A small portion of U.S. landings are attributed to the Gulf of Mexico. Landings records in this region were historically collected by the NMFS but have been administered by the Gulf States Marine Fisheries Commission since 1985 (D. Bellais, GSMFC, pers. comm.). Between 1950 and 1999, landings in the Gulf of Mexico ranged between approximately 200 pounds in 1994 and 28,000 pounds in 1985 (Figure 3). Landings reported since 1999 have been negligible and are thus confidential (R. Maxwell, LA DWF, pers. comm.). Fahay (1978) reported total U.S. landings of American eels during 1955–1973 with minor landings registered from the U.S. Gulf of Mexico region during about half of those years but never exceeded 1% of total U.S. landings. Note that the Gulf States (including western Florida) are under the jurisdiction of the Gulf States Marine Fisheries Commission and are not subject to ASMFC-led interstate fisheries management.

4.5 Fisheries Outside the United States

Because of the panmictic status of American eel, fisheries outside the jurisdiction of the United States are relevant to ASMFC management efforts, although they are not subject to management regulations implemented though the ASMFC. Brief descriptions of Canadian eel fisheries and fisheries at locations south of the United States are provided below for perspective on activity at the northern and southern ends of American eel's range. Information on commercial eel landings in Canada and other western Atlantic countries was obtained from the Department of Fisheries and Oceans (DFO) Canada (DFO, pers. comm.) and the Fisheries Department of the Food and Agriculture Organization (FAO) of the United Nations (FAO, pers. comm.), respectively.

4.5.1 Commercial Fisheries in Canada

For a description of American eel fisheries in Canada, refer to ASMFC 2012.

Fisheries and Oceans Canada, or the DFO, Statistical Services Unit maintains fisheries data for Canada. These data were available for 1972–present. Data from Canada's marine and freshwater commercial fisheries are available via online tables that are summarized by species, province, and region (e.g., Scotia-Fundy vs. Gulf). Trends in seafisheries records from 1972 to 2015 indicate a steady decline in commercial eel landings since the early 1990s, with the exception of 2012-2013 (Figure 4). Available freshwater fisheries records cover a shorter time span (1990–2015) during which time there has been a steady decline since 2000, with the exception of 2013-2014 (Figure 5). However, freshwater landings records may be less reliable than seafisheries records and it is unclear whether overlap in reporting between freshwater fisheries and seafisheries occurs.

4.5.2 Commercial Fisheries in Central and South America

Studies and reports that summarize U.S. American eel fisheries provide no information on commercial eel fisheries in Mexico or the Caribbean Islands other than mentioning that the

American eel's range does extend to these regions (Bigelow and Schroeder 1953; Fahay 1978; Lane 1978; and Van Den Avyle 1984). Annual landings between 1950 and 2015 are available by country and major fishing area from the Food and Agriculture Organization (FAO) of the United Nations Fishery Global Statistics Program of the Fisheries Data, Information, and Statistics Unit (FIDI) via online tables. Mexico, the Dominican Republic, and Cuba reported a small amount of landings (primarily from in-river fisheries) from 1975-2010, although there are several missing values or years of no landings (Figure 6). There was an increase in landings, or reported landings, for 2011-2012 from Mexico and the Dominican Republic. From 2013-2015, landings remained high for the Dominican Republic but not Mexico. It is unknown whether these reports are comprehensive.

5 DATA SOURCES

For this assessment update report, the SAS updated the commercial and recreational landings through 2016. Fishery independent survey data that was used in the trend analyses in ASMFC 2012 was also updated, including state-mandated YOY surveys, non-mandated YOY surveys, yellow eel surveys, and biological data sets used in the growth analysis. Efforts were made to maintain consistency with the benchmark in terms of the data sources and treatment, but this was not always possible. Differences between the benchmark and this update are noted as appropriate.

5.1 Fishery-Dependent

5.1.1 Commercial Fisheries

The FMP for American eel requires states to report commercial harvest by life stage, gear type, month, and region as defined by the states (ASMFC 2000a). During development of the benchmark assessment, not all states were able to provide this level of information, and this remains a challenge for this update.

5.1.1.1 Atlantic Coast

Historical commercial landings data from 1888 to 1940 were transcribed from online U.S. Fish and Fisheries Commission Annual reports (NOAA Central Library Data Imaging Project, pers. comm.).

Commercial landings data collected since the 1900s were obtained from the Atlantic Coastal Cooperative Statistics Program (ACCSP). Since 1950, most landings information on the East Coast has been collected by NMFS through dealer and/or fisherman reporting under a state-federal cooperative program. All historical NMFS data are now housed at ACCSP. Prior to the 1990s, information was summarized annually or monthly; more detailed information became available as states individually began adopting harvester reports (e.g., trip ticket systems or logbooks).

During 1950 to 2016, Atlantic coastwide U.S. American eel landings ranged between approximately 664,000 pounds in 1962 and 3.67 million pounds in 1979 (Figure 7). The highest landings in the time series occurred from the mid-1970s to the early 1980s. Beginning in 1984, landings begun to steadily decline. While landings since the 1990s have been lower than historical landings, they have been stable in recent decades.

Geographic regions used in the 2005 assessment (North, Mid-, and South Atlantic) exhibited differing trends and magnitudes in their eel fisheries (Figure 8). The majority of landings were reported in the Mid-Atlantic (New Jersey to Virginia), followed by the South Atlantic (North Carolina to Florida) and North Atlantic (Maine to New York). Since the coastwide landings peak in the 1970s and 1980s, North and South Atlantic landings have been minimal compared with Mid-Atlantic region landings.

A new set of watershed-based geographic regions were created for the 2012 assessment: Gulf of Maine, Southern New England, Hudson River, Delaware Bay/Mid-Atlantic Coast Bays, Chesapeake Bay, and the South Atlantic (Figure 9). The temporal extent to which landings could be assigned by region (i.e., divide landings within a state like Massachusetts or Maryland) could not be replicated for this update from the available commercial landings data set.

The value of U.S. commercial American eel landings as estimated by NMFS has varied between a few hundred thousand dollars (prior to the 1980s) and a peak of $40.6 million in 2012 (Figure 10). Total landings value declined again in 2014 from the large values from the previous two years but still remained high compared to the rest of the time series.

Since 1950, the majority (79%) of American eel landings were caught in pots and traps (Figure 11). Fixed nets (e.g., weirs, pound nets) accounted for about 7% of the landings. Approximately 5% of landings were caught using other gears (non-pot/trap or fixed net). About 9% of landings are reported with unknown gear type. Throughout the time series, pots and traps were the dominant gear reported for most eel landings (Figure 12).

Potential Biases

There are several potential biases present in the commercial data set. ACCSP validated the yellow American eel landings with each state partner, although several member states used their compliance reports rather than state data and therefore the numbers were not thoroughly validated in all cases. Additionally, Virginia and Maryland have different methods of dealing with PRFC data where Virginia includes those data and Maryland does not in their totals. As identified in ASMFC 2012, at least a portion of commercial American eel landings typically come from non-marine water bodies. Even in states with mandatory reporting, these requirements may not extend outside the marine district, resulting in a potential underestimate of total landings. Misreporting between conger eel, hagfish, slime eel, and American eel can occur, i.e. bycatch caught and reported from trawl gear. Despite these potential biases, the SAS felt that these landings represented the best data available and were indicative of the trend in total landings over time.

5.1.1.2 State-specific data collection

Refer to ASMFC 2012 for a description of state-specific data collection for dealer and harvester reporting. Data collection and reporting on commercial landings at the state level have changed since ASMFC 2012 due to recent addenda to the FMP and efforts by the states to improve on the accuracy of landings information. Specifically, Addendum IV (ASMFC 2014) - which stipulated the potential for state by state quota management for yellow eel if the coast wide cap is exceed by the management triggers- required all states with a yellow eel fishery to develop an implementation plan detailing the 1) current reporting structure for eels, 2) type of reporting used for monitoring quota, 3) a mechanism to account for quota overages, 4) a mechanism for quota transfers, 5) any additional management measures planned to control harvest. Table 3 indicates current reporting structure within states/jurisdictions.

5.1.2 Recreational Fisheries

5.1.2.1 Data Collection

The primary source of recreational fishery statistics for the Atlantic coast is the National Marine Fisheries Service's Marine Recreational Information Program (MRIP), formerly the Marine Recreational Fishery Statistics Survey (MRFSS) program. These programs collected data on marine recreational fishing to estimate statistics characterizing the catch and effort in marine recreational fisheries. Recreational fisheries statistics for American eels were obtained from the MRIP online data query. Catch estimates from MRIP have been available since 2004. Previous to 2004, only catch estimates from MRFSS are available. The method developed by MRIP to calibrate 1981-2003 MRFSS estimates was used in this assessment (SEDAR 2016).

5.1.2.2 Development of Estimates

Estimates of harvest in terms of numbers are available for all three catch types (Type A, B1, and B2). Weight estimates are only available for recreational harvest (Type A+B1). Annual length-frequency distributions of American eels sampled by the MRFSS were calculated using the Type A biological sampling data. These data were available for 1981 through 2016.

5.1.2.3 Estimates

Recreational harvest (Type A + B1) of American eels along the Atlantic coast ranged from 3,062 to 220,596 eels per year during 1981 through 2016. In terms of weight, recreational eel harvest ranged from 497 to 218,269 pounds per year during the same time period (Table 6). American eel recreational harvest demonstrated an overall decline over the available time series, with some large peaks in the mid-1980s, early 1990s, and 2010 (Figure 13). The number of American eels released alive by recreational anglers ranged from a low of 26,707 eels in 1997 to a high of 157,189 eels in 2003. Live releases of American eels generally declined from the late 1980s through the late 1990s to early 2000s. Numbers of live releases have since increased from 2002-2014. Both 2015-2016 indicate lower numbers of live releases.

The precision of the estimated harvest numbers, measured as proportional standard error (PSE), exceeded 50% in 29 of the 36 years for which estimates were available (Table 6). The precision of harvest weight estimates exceeded 50% in 18 of the 34 years with PSE calculations. In some years, the sampling data were insufficient to allow calculation of precision of harvest weight. Estimates of the number of American eels released alive had higher precision than the harvest estimates, with PSE values exceeding 50% in 8 of the 36 years.

The low precision associated with the recreational fishery statistics is due to the limited numbers of American eels that have been encountered during surveys of recreational anglers along the Atlantic Coast (Table 4 and Table 5). These limited numbers are partly due to the design of the MRFSS/MRIP survey, which does not include the areas and gears assumed to be responsible for the majority of recreational fishing for American eels. As such, the recreational fishery statistics for American eels provided by MRFSS should be interpreted with caution.

The lengths reported for American eels sampled (Type A catch) ranged from 20 mm to 1,100 mm during 1981 to 2016 (Figure 14). Smaller recorded lengths are likely recording errors or species misidentifications.

5.2 Fishery-Independent Surveys and Studies

This section summarizes survey data and studies used to inform the stock assessment. All fishery-independent surveys used in ASMFC 2012 were evaluated using a standard set of criteria (see Appendix 2 in ASMFC 2012) that resulted in data-based decisions to inform the analytical framework (primary assumptions regarding the error structure) for each survey independently. Application of these criteria resulted in nearly all surveys being standardized (unless otherwise noted) using a generalized linear model (GLM) to account for changes in catchability of eel. Only the surveys that were used in the trend analyses in the benchmark assessment were updated in this report. Some state-mandated YOY surveys were excluded from trend analysis in ASMFC 2012 because they did not have at least 10 years of data but have been included in this update if the survey met that requirement. The same methods were used as ASMFC 2012, although differences in GLM standardization are described below.

5.2.1 Young-of-Year Abundance Surveys

5.2.1.1 Development of Indices

For a description of the coastwide mandatory state YOY and non-mandated survey methods, sampling intensity, biological sampling, and potential biases refer to ASMFC 2012 section 5.2.1.1. Annual indices of relative YOY abundance were calculated using the protocol outlined in Appendix 2 of ASMFC 2012. The YOY indices developed for ASMFC 2012 were from surveys that were sampled for at least 10 years as of 2010. For this update, three more surveys had reached the 10 year requirement: Connecticut’s Ingham Hill site, Rhode Island’s Hamilton Fish Ladder, and Virginia’s Wareham’s Pond. Conversely, three YOY indices were not updated through 2016 due to the sampling site being moved (PRFC’s Clark’s Millpond and South Carolina’s Goose Creek) or no longer sampled (Georgia’s Altamaha Canal). While these sites were not updated, they were still included in analyses and correlations. ASMFC 2012 categorized NC’s Beaufort Bridgenet Ichthyoplankton Sampling Program (which ASMFC referred to as the Beaufort Inlet Ichthyoplankton Survey) as non-mandated, when it in fact serves as the state’s mandated YOY survey so that has been corrected for this report. Additionally, data was only available through 2007 when it was included in analyses for this update (Figure 31). The data was later updated through 2013 but the analyses were already completed.

The availability of potential covariates varied among sites and years. Though the ASMFC YOY survey protocol requires that states record effort, water temperature, water level, and discharge (ASMFC 2000b), effort and water temperature were the only auxiliary variables consistently available for all sites. Additional variables were considered as covariates in the GLM analysis if the data were available in all years for a particular site.

Spearman's rank correlation coefficient, ρ, and the associated probability were calculated for all pairs of YOY indices to assess the degree of association among the indices. Indices were considered significantly correlated at $\alpha = 0.10$.

5.2.1.2 Estimates

Annual recruitment indices were computed for nineteen sites sampled as part of the ASMFC-mandate, as well as three indices that are not required by ASMFC (Table 7). Water temperature was found to be a significant covariate affecting catchability for most survey sites. Note that effort was not determined to be a significant covariate in the models for any of the survey sites. Most of the survey data were best characterized using a model that had negative binomial errors. For some sites, a stable generalized linear model could not be developed, so arithmetic mean catch per unit effort was used as an index of abundance.

Trends in the YOY indices were variable within and among survey sites (Figure 15–Figure 31). The degree of correlation between survey sites varied and all were either not significant or were significant and positively correlated (Table 8). While there is still not a lot of agreement among YOY sites, there is an improvement since ASMFC 2012. In this update, of the 22 significant relationships, all were positive. In the benchmark stock assessment, there were 13 significant relationships, ten positive and three negative. In addition, at the regional level there were 5 significant relationships between regions, all of which were positive. It should be noted that ASMFC 2012 incorrectly categorized the Beaufort Bridgenet Ichthyoplankton Sampling Program (BBISP) as non-mandated so it was not included in the correlations at that time but is included in the correlations for this report.

In the Gulf of Maine region, two YOY indices were significantly positively correlated - West Harbor Pond (Maine; Figure 15) and Lamprey River (New Hampshire; Figure 16) (Table 8). Both of these indices show low abundances in the beginning of the time series with peaks in the early 2010s. In the Southern New England region, there were two pairs of sites that were significantly positively correlated —Gilbert Stuart Dam (Rhode Island; Figure 18) and Hamilton Fish Ladder (Rhode Island; Figure 19) and Gilbert Stuart Dam (Rhode Island) and Carman's River (New York; Figure 21) (Table 8). All three of these indices show low abundances in the early and mid-2000s with small increases in the early and mid-2010s. In the Delaware Bay and Mid-Atlantic Coastal Bays and Chesapeake Bay regions, there were no significant relationships between YOY surveys (Table 8). One significant correlation was detected among the YOY indices in the South Atlantic region. The YOY indices for Goose Creek (South Carolina; Figure 32) and Guana River Dam (Florida; Figure 34) were significantly and positively correlated (Table 8). Both of these indices show a peak in recruitment in 2001 and 2005 and then a decline for the remaining years in the time series.

5.2.2 Yearling, Elver, and Yellow Eel Abundance Surveys

5.2.2.1 Development of Indices

Several surveys were developed into abundance indices for yearling, elver, and yellow American eel life stages from Connecticut to South Carolina. For a full description of these survey methods, sampling intensity, biological sampling, and potential biases refer to ASMFC 2012. Abundance indices from these surveys were standardized using the same methods as the benchmark. During the GLM standardization, there were some differences in the covariates used in the model. Table 9 summarizes the GLM model used and significant covariates. Below are some additional notes on each survey.

CTDEP Electrofishing

Elver & yellow eel index: A population estimate was derived using maximum weighted likelihood by CTDEP. The site was not sampled in 2013 and then moved to a new site for 2015-2016. Due to the change in site, the SAS decided to abbreviate this time series to 2014 (Figure 37).

NY Western Long Island Survey

Yellow eel index: A full model that predicted catch as a function of year, month, and latitude as factors was compared with nested submodels using AIC. The full model with a negative binomial error structure was selected because it produced the lowest AIC. The model was unchanged from the previous benchmark assessment, although latitude was used instead of system, and updated through 2016. The time series peaked to its highest value in 1985 and has declined since then, remaining low until the terminal year (Figure 38).

NYDEC Alosine Beach Seine Survey

Elver & yellow eel index: A full model that predicted catch as a function of year, month, river mile, water temperature, latitude, and longitude was compared with nested submodels using AIC. The model that included year, month, and river mile with a negative binomial error structure was selected because it produced the lowest AIC. The model was changed from the previous benchmark assessment, which had year, month, river mile, and water temperature as covariates. The index is variable with higher peaks in the early part of the time series and low but stable values in the later part of the time series (Figure 39).

NYDEC Striped Bass Beach Seine Survey

Elver & yellow eel index: A full model that predicted catch as a function of year, month, river mile, water temperature, latitude, and longitude was compared with nested submodels using AIC. The model that included year, month, and longitude with a negative binomial error structure was selected because it produced the lowest AIC. The model was changed from the previous benchmark assessment, which had year, month, river mile, and water temperature as covariates. The index is variable with higher peaks in the early part of the time series and declining but stable values in the later part time series. There was a notable peak in abundance in 2015 which was followed by the lowest point in the time series in 2016 (Figure 40).

HRE Monitoring Program

Yearling & older eel index: A full model that predicted catch as a function of year, month, station, river mile, tide, temperature, depth, tow volume, gear, and strata was compared with nested submodels using AIC. The model that included year, month, strata, river mile, and tow volume with a negative binomial error structure was selected because it produced the lowest AIC and good model diagnostics. The model formula for the previous benchmark assessment was the same but also included gear which was no longer significant for this update.

NYDEC provided the SAS with the HRE Monitoring Program data set through 2013. Because this data set is maintained by a utility company, the SAS submitted an additional request to HRE to obtain 2014-2016 due to data confidentiality concerns. The data set was updated through 2015, although it was received too late to be incorporated into the trend analysis and regional indices. Biologists for the HRE Monitoring Program expressed concern that the length cutoff between YOY and yearling+ was not accurate in the data set provided by NYDEC. Additionally, they were concerned that some of the covariates may not have been converted correctly. The updated data set represents the most complete and accurate data set and is included in this report despite not being used in the analyses. For the analyses and regional indices, the previous data set provided by NYDEC through 2013 was used. The GLM model for both the 1974-2013 and the 1974-2015 data sets was the same, as was the general pattern of the time series, although the scale was different (Figure 41). Abundance was highest during the early years of the time series, after which it dropped abruptly and then rebounded within the first decade. A more gradual declined followed from the mid-1980s through the early 2000s. Since then, abundance has gradually increased, but is still below levels seem in the mid-1980s.

NJDFW Striped Bass Seine Survey

Yellow eel index: A full model that predicted catch as a function of year, month, water temperature, and salinity was compared with nested submodels using AIC. The model that included year, water temperature, and salinity with a negative binomial error structure was selected. The model was unchanged from the previous benchmark assessment although salinity was not significant this time but it was retained for consistency. The index exhibited some high abundance in the early time series but otherwise a stable abundance throughout (Figure 42).

Delaware 16’ Trawl Survey

Elver & yellow eel index: A full model that predicted catch as a function of year, month, surface temperature, and surface salinity was compared with nested submodels using AIC. The full model that included year, month, surface temperature, and surface salinity with a negative binomial error structure was selected. The model was unchanged from the previous benchmark assessment although surface temperature was not significant this time but it was retained for consistency. Abundance declined in the 1980s, increased in the 1990s, declined until about 2005, after which it has been relatively stable (Figure 43).

PSEG Trawl

Elver & yellow eel index: A full model that predicted catch as a function of year, month, bottom salinity, and strata was compared with nested submodels using AIC. Consistent sampling was conducted every year since 1998 so the time series was abbreviated from the previous assessment. Also, the stations have changed over time. Attempts were made to replicate the covariates from ASMFC 2012, but that model used only the months April-June when there are still consistent catches July-October. Additionally, the previous model used strata 7-9, but this update used 6-8. The model that included year, month, and bottom salinity with a negative binomial error structure was selected because it produced the lowest AIC. The model was unchanged from the previous benchmark assessment, although the months and strata used were different. The abundance index was variable in the late 1990s and early 2000s and then steady through mid-2010s. There were peaks in 2013 and 2016 (Figure 44).

Pennsylvania Area 6 Electrofishing

Elver index: A full model that predicted catch as a function of year, month, site, and tow duration was compared with nested submodels using AIC. The model that included year and site with a negative binomial error structure was selected because it produced the lowest AIC. The model was unchanged from the previous benchmark assessment. There were peaks of abundance in 2001 and 2015 and low abundance in 2002 and 2016, otherwise the index indicates steady abundance (Figure 45).

MDDNR Striped Bass Seine Survey

Yellow eel index: A full model that predicted catch as a function of year, month, and salinity was compared with nested submodels using AIC. The full model that included year, month, and salinity with a negative binomial error structure was selected because it produced the lowest AIC. The model was unchanged from the previous benchmark assessment. Abundance was high in 1965, 1975, 2003, and 2005 and low in the early 1970s, early and mid-1990s, mid-2000s, and early 2010s (Figure 46).

VIMS Juvenile Striped Bass Seine Survey

Yellow eel index: A full model that predicted catch as a function of year, month, station type, system, and salinity was compared with nested submodels using AIC. This data set was analyzed for two time periods: long (1967-2016; Figure 47) and short (1989-2016; Figure 48). The model with a negative binomial error structure was selected because it produced the lowest AIC for both long and short indices. The long model was unchanged from the previous benchmark assessment with only system as a covariate. The short model used station type whereas the benchmark assessment also had salinity as a significant covariate. Both indices are variable. The longer time series shows high abundance in 1968 and 1971, followed by low abundance and some missing values. The index is low through the late 1980s and early 1990s and then variable with some peaks in abundance in the last decade (Figure 47). The shorter time series shows a more stable abundance through time with some peaks in 1997, 2009, and 2012 and low values in 1996, 2003, 2005, and 2013 (Figure 48).

North Anna Electrofishing Survey

Elver and yellow eel index: Updated data through 2016 from this survey was not provided for this assessment and therefore the index from the benchmark was used in analyses and regional indices. The abundance index indicates low values through the 1990s to 2002. Following a missing value point in 2003, the index shows increased abundance, ending with the highest value in the terminal year of 2009 (Figure 49).

NCDMF Estuarine Trawl Survey

Elver & yellow eel index: A full model that predicted catch as a function of year, month, water temperature, salinity, dissolved oxygen, depth, latitude, longitude, and bottom type was compared with nested submodels using AIC. The model that included year, latitude, longitude, and bottom type with a negative binomial error structure was selected. The model was unchanged from the previous benchmark assessment. The abundance index shows a lot of variability with the highest values in 1990-1991 and 2011-2012 and the lowest values in 2000, 2009, 2013, and 2016 (Figure 50).

SC Electrofishing Survey

Elver & yellow eel index: A full model that predicted catch as a function of year, month, strata, water temperature, salinity, and tide was compared with nested submodels using AIC. The full model with a negative binomial error structure was selected. The model was unchanged from the previous benchmark assessment. The abundance index indicates steady abundance throughout the time series with one larger peak in 2003 (Figure 51).

Spearman's rank correlation coefficient, ρ, and the associated probability were calculated for all pairs of yellow American eel indices to assess the degree of association among the indices. Indices were considered significantly correlated at α= 0.10. The degree of correlation between survey sites varied and all were either not significant or were significant and positively correlated (Table 10). Surveys in the Hudson River region were positively correlated with many Southern New England and other Hudson River surveys. Only the New Jersey Striped Bass Seine Survey and the Delaware trawl were positively correlated with each other in the Delaware Bay/Mid-Atlantic region. In the Chesapeake Bay region, only the MDDNR Striped Bass Seine Survey and North Anna Electrofishing survey were positively correlated while the other surveys did not have a significant relationship. The two surveys available in the South Atlantic region were not significantly correlated with each other.

6 ASSESSMENT

6.1 Coastwide Abundance Indices

Indices of coastwide abundance for YOY and yellow-phase American eel were developed by combining data from multiple surveys along the coast. Detailed information describing the surveys included in the coastwide indices and the methods for calculating them can be found in ASMFC 2012.

6.1.1 Development of Estimates

Coastwide Recruitment

All ASMFC-mandated YOY abundance surveys and the two non-mandated YOY abundance surveys were used to assess coastwide recruitment. Two coastwide indices of American eel recruitment were computed—a short-term index and a long-term index. The short- and long-term indices were developed by combining individual standardized indices into a single, coastwide index using the generalized linear modeling approach (ASMFC 2012 Appendix 2). The short-term recruitment index was based on the standardized indices developed from the ASMFC-mandated annual YOY surveys. The time period used for generating the short-term coastwide recruitment index was 2000 to 2016. The long-term recruitment index was based on the Beaufort Bridgenet Ichthyoplankton Sampling Program (referred to incorrectly as the Beaufort Inlet Ichthyoplankton Survey and miscategorized as non-mandated in ASFMC 2012) and the non-mandated HRE Monitoring Program and Little Egg Inlet Ichthyoplankton Survey standardized indices. The covariates considered for inclusion in the model for the short- and long-term indices were year, region, and survey site. The time period used for generating the long-term coastwide recruitment index was 1988 to 2013. This time period was selected so that index values from at least two of the long-term YOY surveys were available for every year included in the combined index.

Coastwide Yellow-Phase Abundance

The surveys used to develop the coastwide yellow-phase abundance indices were: NY Western Long Island Survey, HRE Monitoring Program, NYDEC Alosine and Striped Bass Beach Seine Surveys, New Jersey Striped Bass Seine Survey, Delaware Juvenile Finfish Trawl Survey, PSEG Trawl Survey, Pennsylvania's Area 6 Electrofishing Survey, Maryland Striped Bass Seine Survey, North Anna Electrofishing Survey, VIMS Juvenile Striped Bass Seine Survey, NCDMF Estuarine Trawl Survey, and South Carolina's Electrofishing Survey. Although these surveys catch yellow stage eels, it should be noted that some portion of the catch in these surveys may include elvers as well.

Three indices of coastwide, yellow-phase abundance were computed using different time series lengths—twenty, thirty, and forty-plus years. The indices were developed by combining individual standardized indices into coastwide indices using the generalized linear modeling approach (ASMFC 2012 Appendix 2). The 40-plus-year coastwide index of yellow-phase abundance was based on the HRE Monitoring Program, MDDNR Striped Bass Seine Survey, and VIMS Juvenile Striped Bass Seine Survey (long time series) standardized indices. In ASMFC 2012, PSEG trawl was included in this index but it was omitted for this update because the time series length changed due to data concerns. Conversely, the HRE Monitoring Program survey was added since it now has enough years of data to be included in the 40-year index. The 1974–2016 time period was used for the 40-plus index because it was the longest time series that could be used for which at least two of the 40-plus-year indices were available for every year included.

The 30-year coastwide, yellow-phase abundance index included the same survey indices as the 40-plus index as well as the NY Western Long Island Survey, NYDEC Alosine Beach Seine Survey, NYDEC Striped Bass Beach Seine, New Jersey Striped Bass Seine Survey, and Delaware Trawl Survey. The 20-year index included the same survey indices as the 30-year index except for the VIMS Juvenile Striped Bass Seine Survey long time series index. Instead, the 20-year yellow-phase abundance index included the short time series index developed from the VIMS Juvenile Striped Bass Seine Survey. In addition, the 20-year index included the PSEG Trawl Survey, Pennsylvania's Area 6 Electrofishing Survey, North Anna Electrofishing Survey, NCDMF Estuarine Trawl Survey, and SC Electrofishing Survey standardized indices.

6.1.2 Estimates

Coastwide Recruitment

The short- and long-term YOY recruitment indices were developed assuming a lognormal error structure. The final model for both indices included year and region as covariates.

The short-term, coastwide recruitment index was variable (Figure 52). The index begins with low abundance and then increases to a high in 2002. Following that peak, the index declines through 2004 and then has a slight uptick and remained stable through the mid and late-2000s. Abundance increased from 2009 to the highest value in the series in 2012 and has declined slightly since then.

The long-term, coastwide index was variable, with low values in 1991 and 2010 and high values in 1988, the mid-1990s, and 2008 (Figure 53).

Coastwide Yellow-Phase Abundance

The coastwide, yellow-phase abundance indices were developed assuming a lognormal error structure. The final model for all three indices included year and survey site as covariates.

The 40-plus yellow-phase index for the coast began with higher abundances in the mid-1970s and a decline through the 1980s (Figure 54). Abundance has been stable since the 1990s. The time series demonstrates inter-annual variability and while values have been lower since the mid-1970s, the trend appears stable in recent decades. The 30-year coastwide index of yellow-phase American eel abundance also exhibits a decline from the beginning of the time series to the early 1990s (Figure 55). The 30-year index show little variability or trend throughout the rest of the time series. The 20-year index of yellow-phase abundance shows limited variability and a no discernable trend (Figure 56). Of the three coastwide, yellow-phase abundance indices, the 20-year and 40-year indices were negatively correlated with each other but not significantly (ρ=-0.152; P=0.742). The 30-year index was positively correlated with both of the 20-year (ρ=0.383; P<0.10) and 40-year (ρ=0.493; P<0.10) indices.

6.2 Regional Abundance Indices

Indices of regional abundance for YOY and yellow-stage American eel were developed for each of the regions by combining data from relevant surveys within each region (Table 11). Note that the regional indices labeled as yellow-stage indices actually reflect the relative abundance of both yellow-stage eels and elvers, in most cases (see Table 9).

6.2.1 Development of Estimates

Region-specific indices of YOY and yellow-stage relative abundance were computed for each of the six geographic regions where data were available. Indices of YOY and yellow-stage American eel abundance were developed by combining individual standardized indices (Table 7 and Table 9) using the generalized linear modeling approach (ASMFC 2012 Appendix A). The time period for each regional index was selected so that index values from at least two of the surveys included were available for every year included in the combined index. The surveys used in the development of the regional YOY and yellow-stage indices and the time periods of those indices are listed in Table 11.

Spearman's rank correlation coefficient, ρ, and the associated probability were calculated for all pairs of regional YOY indices and all pairs of regional yellow-stage indices to assess the degree of association among the indices. The correlation analysis was also applied to evaluate the degree of association between the yellow-stage indices and the YOY indices within each region. The YOY indices were lagged by 0–4 years for comparison to the yellow-stage indices. Indices were considered significantly correlated at $\alpha = 0.10$.

6.2.2 Estimates

All region-specific YOY and yellow-stage indices of American eel abundance were modeled assuming lognormal error structures and the final models all included year and state as covariates. The Chesapeake Bay's yellow eel index also included gear. The Hudson River region YOY index was based on a single recruitment index because only one such index was available for the region (Table 11). No yellow-stage indices of American eel abundance were available for the Gulf of Maine so a yellow-stage index could not be developed for the Gulf of Maine. There were two yellow eel abundance indices in the Southern New England region, CTDEP Electrofishing Survey and the NY Western Long Island Survey, but a regional yellow eel abundance survey was not developed due to concerns using a population estimate (CTDEP Electrofishing) and a standardized abundance index (NY Western Long Island Survey) together. Additionally, the CTDEP Electrofishing Survey had an abbreviated time series due to a year that wasn't sampled and then a change in the site location.

The regional YOY and yellow-stage indices of American eel abundance are depicted in Figure 57 and Figure 58. Both the YOY and yellow-stage regional indices are variable among years. All the YOY indices, except in the Delaware Bay and Hudson River regions, are characterized by relatively large standard errors. This is partly due to the differences in the magnitudes of the index values among surveys that were combined in developing the region-specific indices.

Among the regional YOY indices for American eel, the Hudson River and Delaware Bay/Mid-Atlantic Coastal Bays indices were found to be significantly and positively correlated with Gulf of Maine indices (Table 12). Significant, positive correlations were also detected between the Delaware Bay/Mid-Atlantic Coastal Bay regional index and the Southern New England and

Hudson River YOY regional indices. The Hudson River was also positively correlated with the South Atlantic YOY regional index. There were no statistically significant correlations detected among the region-specific yellow-stage indices (Table 13). Some significant correlations were detected between the region-specific yellow-stage and lagged YOY indices (Table 14). The Hudson River yellow-stage index was significantly correlated with the Hudson River YOY index that was lagged by one, two, three, and four years. The Chesapeake Bay yellow-stage index was significantly and positively correlated with the Chesapeake Bay YOY index that was lagged by two years. The South Atlantic yellow-stage index was significantly and positively correlated with the South Atlantic YOY index that was lagged one, two, and four years.

6.3 Analyses of Life History Data

6.3.1 Growth Meta-Analysis

6.3.1.1 Methods

Biological data for American eel were compiled from a number of past and on-going research programs along the Atlantic Coast and classified into one of the six geographic regions used in the assessment. These data, updated through 2016, were used to model both the length-weight and age-length relationship for American eel. The relation of length in millimeters to weight in grams was modeled using the allometric length-weight function. Length-weight parameters were estimated by region, sex, and for all data pooled together. The analysis of the residual sum of squares (ARSS) method was performed to compare the length-weight curves among regions and between sexes (Chen et al. 1992; Haddon 2001). The ARSS method provided a procedure for testing whether two or more nonlinear curves are coincident (i.e., not statistically different). Values were considered statistically significant at $\alpha < 0.05$.

Linear regression was used to model the relation of age in years to length in millimeters by region, sex, and for all data pooled together. A test for coincident regressions was applied to test for differences in the regressions among regions and between sexes (Zar 1999). Values were considered statistically significant at $\alpha < 0.05$. The age-length relationship for American eel was also described through the von Bertalanffy model, which is given by:

$$Lt = L\infty\ [1 - e^{-K\ (t-t0)}]$$

where Lt is length at age t, $L\infty$ is the theoretical asymptotic average length (if $K > 0$), K is growth rate at which the asymptote is approached, and $t0$ is the hypothetical age at which length is zero. Model fits were first evaluated based on convergence status; models that did not successfully converge were removed from consideration for the associated dataset.

6.3.1.2 Results

The length-weight model successfully converged and parameters estimated for each of the six regions, by sex, and for all data pooled (Table 15; Figure 59). The results of the ARSS indicated that there were statistically significant differences in the length-weight relationship between at least two regions ($F_{10,\ 68{,}276} = 293$, $P < 0.001$). However, parameter estimates were very similar in five of the six regions particularly in the Delaware Bay/ Mid Atlantic Coastal Bays, Chesapeake Bay, and South Atlantic. Parameter estimates were most different in the Southern New England region, which may be due to an extremely small sample size (N=166) and range of length-weights available in the dataset. The fit of the length-weight function to all pooled data was dominated by data from the Chesapeake Bay region, which was the source of more than 55% of the length and weight biological samples. The results of the ARSS indicated no sex specific significance between estimated length-weight parameters ($F_{2,\ 6{,}687} = 0.91$, $P = 0.40$; Figure 60).

The parameters estimated from the linear regression of length on age for the various dataset configurations are presented in Table 16. There are statistically significant differences in the age-length relation among regions based on the results of the test for coincident regressions ($F_{10,\ 17{,}402} = 754$, $P < 0.0001$). The final parameter estimates suggested distinct differences in growth patterns between the northernmost regions (Hudson River, Southern New England, Gulf of Maine) and southernmost regions (Del Bay/Mid-Atlantic Coastal Bays, Chesapeake Bay, South Atlantic) (Table 16; Figure 61). The fastest growth in length with age occurred in the Delaware Bay/Mid-Atlantic Coastal Bays region. The test for coincident regressions also detected significant differences in the age-length regressions between sexes ($F_{2,\ 5{,}932} = 1{,}520$, $P < 0.0001$; Figure 62). The results suggested the rate of growth in length with age is faster in females than males (Table 16; Figure 62).

Parameters were estimated from the von Bertalanffy model to further examine the age-length relationship of American eel by region and by sex (Table 17). The model failed to converge for the Southern New England region and for males. The clear differences in growth between the northernmost and southernmost regions determined from the linear regression analysis were not apparent in the parameter estimates derived from the von Bertalanffy model. However, the growth coefficient (K) was the highest in the South Atlantic region and the lowest in the Gulf of Maine.

Significant variation in length at age and a broad overlap in lengths across multiple age groups were observed in the data even within a regional analysis. Pooled data for all regions amplified these variations in length at age. These analyses confirm the relationship between age and length for American eel is not well defined and that age is a poor predictor of length for American eel. Ageing error and uncertainty around ageing estimates may also play an additional role in the weak relationship of length and age.

6.4 Trend Analyses

6.4.1 Power Analysis

Power analysis was performed on all fishery-independent American eel surveys as a means to evaluate the precision of abundance indices.

6.4.1.1 Methods

Power analysis followed methods described in Gerrodette (1987) for both potential linear and exponential trends. A linear trend can be modeled as $A_i = A_1[1+r(i-1)]$ and an exponential trend as $A_i = A_1(1+r)^{i-1}$ where A_i is the abundance index in year *i*, A_1 is the abundance index in year 1, and *r* is a constant increment of change as a fraction of the initial abundance index A_1. The overall fractional change in abundance over n years can be expressed as $R = r(n-1)$.

If α and β are the probabilities of type 1 and type 2 errors respectively, the power of a linear trend (1 – β) assuming CV ~ 1/√*A* can be determined by satisfying the equation:

$$r^2 n(n-1)(n+1) \geq 12CV_1^2\left(z_\alpha + z_\beta\right)^2 \left\{1 + \frac{3r}{2}(n-1)\left[1 + \frac{r}{3}(2n-1) + \frac{r^2}{6}n(n-1)\right]\right\}$$

and the power of an exponential trend can be determined by satisfying the equation:

$$[\ln(1+r)]^2 n(n-1)(n+1) \geq 12\left(z_\alpha + z_\beta\right)^2 \left\{\frac{1}{n}\sum \ln\left[CV_1^2(1+r)^{i-1} + 1\right]\right\}$$

where CV_1 is an estimate of the coefficient of variation of the survey. For each of the surveys, the median CV of the survey was calculated over the entire time series of the survey and used as an estimate of CV_1. Power was then calculated for an overall change (*R*) of ±50% over a 10 year time period (*r* = 0.056) for both a linear and exponential trend.

6.4.1.2 Results

Median CVs of the surveys ranged from 0.04 to 5.50. Resulting estimates of power were a function of CVs with those surveys having low CVs having high power, and those surveys having high CVs having low power. Power values ranged from 0.06 to 1.00 (Table 18). For all surveys, there is greater power to detect a decreasing trend compared to an increasing trend which is a property of surveys whose CV~1⁄√A. There was very little difference in power between linear and exponential trends. The values of power presented in Table 18 can be interpreted as the probability of detecting a given linear or exponential trend of ±50% over a ten year period if it actually occurs. Many surveys decreased the median CV values with the additional years of data since ASMFC 2012 and therefore increased the power associated with that survey. These values do not reflect a retrospective power analysis and a survey with a low power value may still be capable of detecting a statistically significant trend if given enough years of data or the change over time is very large.

6.4.2 Mann-Kendall Analysis

6.4.2.1 Methods

The Mann-Kendall trend analysis is a non-parametric test for monotonic trend in time-ordered data (Gilbert 1987). The null hypothesis is that the time series is independent and identically distributed—there is no significant trend across time. The test allows for missing values and can account for tied values if present.

The Mann-Kendall test was applied to all local, regional, and coastwide indices of relative abundance computed in this assessment. This included four new local YOY indices; Hamilton Fish Ladder, Gilbert Stuart Dam, Ingham Hill, Carman's River, HRE Monitoring Program, and

Little Egg Inlet Ichthyoplanton. There were no new yellow eel indices. Two regional indices were not analyzed because only one index in the region had been updated to 2016.

A two-tailed test was used to test for the presence of either an upward or downward trend over the entire time series. Trends were considered statistically significant at $\alpha = 0.05$.

6.4.2.2 Results

Local Indices

No significant temporal trends were detected among the YOY indices developed from the ASMFC-mandated recruitment surveys when the analysis was done in the last benchmark (Table 19). Of the two YOY surveys that are not ASMFC-mandated, the Little Egg Inlet had no trend and the HRE Monitoring Program had a declining trend in ASMFC 2012. In this update, six of the 22 indices showed significant negative trends. This included many of the new indices, of which 3 showed significant declining trends.

The Mann-Kendall test found statistically significant trends in six of the 15 other individual yellow eel indices evaluated; all but one of which was negative (Table 20). Since the last benchmark two significant downward trends became non-significant, while two significant upward trends also became non-significant.

Regional Indices

Of the nine regional indices, significant trends were seen in four; one positive and 3 negative (Table 21). One of the negative trends, the YOY for the South Atlantic, was not significant during the last benchmark, but is now a significantly declining trend with this update.

Coastwide Indices

The Mann-Kendall test detected two significant trends among the coastwide indices (Table 21). Both the 30-year and 40-year yellow-phase abundance indices exhibited a significant downward trend. The 40 year was not significantly declining in the last benchmark, but is with this update. The starting year of this index was 1967 in ASMFC and it is now 1974 for this update, so the loss of the beginning years may influence this declining trend.

6.4.3 Manly Analysis

A meta-analysis was conducted to determine if there was consensus among fishery-independent survey indices for a coastwide decline in American eel. Meta-analysis is a statistical approach that combines the results from independent datasets to determine if the datasets are showing the same patterns. The meta-analysis techniques employed in this analysis are described by Manly (2001).

6.4.3.1 Methods

American eel surveys were grouped according to life stages (yellow vs. YOY) and one-tailed *p*-values from the Mann-Kendall test for trend were used in the meta-analysis (Manly 2001). Two meta-analysis techniques were used.

Fisher's method tests the hypothesis that at least one of the indices showed a significant decline through time. The test statistic was calculated as $S_1 = -2\sum\log_e(p_i)$, where p_i is the one-tailed *p*-value that tests for a negative trend from the *i*th index. The one tailed *p*-value is used because we are interested in whether the index has declined through time. If the null hypothesis is true for a test of significance, then the *p*-value from the test has a uniform distribution between 0 and 1, and if *p* has a uniform distribution, then $-2\log_e(p)$ has a chi-square distribution with 2 degrees of freedom. The test statistic, S_1, is then compared to a chi-square distribution with $2n$ degrees of freedom, where *n* equals the number of independent surveys considered.

The Liptak-Stouffer method tests the hypothesis that there is consensus for a decline supported by the entire set of indices. The individual one-tailed *p*-values were converted to z-scores. If the null hypothesis is true for all indices, the z-scores are distributed as a normal random variable with mean equal to 0 and variance equal to $1/\sqrt{n}$. This allows for weighting the results from the indices differently. The test statistic is $S_2 = \sum w_i z_i / \sqrt{\sum w_i^2}$ where w_i is the weight of the *i*th index. In this analysis, the number of years of survey data was used as the weight for the *i*th index. A level of $\alpha = 0.05$ was used in meta-analyses for tests of significance.

6.4.3.2 Results

At least one of the indices for both life stages showed a decline though time (yellow eels: $S_1 = 115.88$, $P < 0.01$; YOY eels: $S_1 = 95.22$, $P < 0.01$; Table 22). Also, there was consensus for a decline for both life stages through time (yellow eels: $S_2 = -5.05$, $P < 0.01$; YOY eels: $S_2 = -16.03$, $P < 0.01$).

6.4.4 ARIMA

Fishery-independent surveys for American eel can be quite variable, making inferences about population trends uncertain. Time series of abundance indices can be influenced by true changes in abundance, within survey sampling error, and varying catchability over time. One approach to minimize measurement error in the survey estimates is by using autoregressive integrated moving average models (ARIMA, Box and Jenkins 1976). The ARIMA approach derives fitted estimates of abundance over the entire time series whose variance is less than the variance of the observed series (Pennington 1986). This approach is commonly used to gain insight in stock assessments where enough data for size or age-structured assessments (e.g., yield per recruit, catch at age) is not yet available.

Helser and Hayes (1995) extended Pennington's (1986) application of ARIMA models to fisheries survey data to infer population status relative to an index-based reference point. This methodology yields a probability of the fitted index value of a particular year being less than the reference point [$p(\text{index}_t < \text{reference})$]. Helser et al. (2002) suggested using a two-tiered approach when evaluating reference points whereby not only is the probability of being below (or above) the reference point estimated, the statistical level of confidence is also specified. The confidence level can be thought of as a one-tailed α-probability from typical statistical hypothesis testing. For example, if the $p(\text{index}_t < \text{reference}) = 0.90$ at an 80% confidence level, there is strong evidence that the index of the year in question is less than the reference point.

This methodology characterizes both the uncertainty in the index of abundance and in the chosen reference point. Helser and Hayes (1995) suggested the lower quartile (25th percentile) of the fitted abundance index as the reference point in an analysis of Atlantic wolfish (*Anarhichas lupus*) data. The use of the lower quartile as a reference point is arbitrary, but does provide a reasonable reference point for comparison for data with relatively high and low abundance over a range of years.

6.4.4.1 Methods

The purpose of this analysis was to fit ARIMA models to time series of eel abundance indices to infer the status of the population(s). The ARIMA model fitting procedure of Pennington (1986) and bootstrapped estimates of the probability of being less than an index-based reference point (25th percentile, Helser and Hayes 1995) were coded in R (R code developed by Gary Nelson, Massachusetts Division of Marine Fisheries). Index values were log*e* transformed (log*e*[index + 0.01] in cases where “0” values were observed) prior to ARIMA model fitting. The reported probabilities of being less than the 25th percentile reference point correspond to 80% confidence levels. Only time series with 20 or more years of index values were used in ARIMA modeling because the 25th percentile reference point can be unstable with few observations. The one exception to the 20 year criteria was the PSEG trawl survey which had 19 years of data included. In the previous 2012 stock assessment, the PSEG trawl survey had 38 years of data at that time, but it was truncated for this assessment update to account for methodology and sampling changes over the years.

6.4.4.2 Results

Fourteen surveys were used in ARIMA modeling (Table 23). Two surveys that were included in this assessment update that were not included in the 2012 stock assessment were the Little Egg Inlet and the Beaufort Bridgenet Ichthyoplankton surveys. These surveys were added to the ARIMA modeling because they now each had >20 years of data available.

Trends in fitted ARIMA values varied both within and among regions. In the Chesapeake Bay region, the long VIMS Juvenile Striped Bass Seine Survey for yellow eels showed a consistent increase since 2008, but the short VIMS Juvenile Striped Bass Seine Survey and the Maryland Striped Bass Seine Survey showed stable trends in recent years (Figure 63). Trends in the Delaware Bay/Mid-Atlantic region did not show any directional trends in recent years (Figure 64). Surveys in the Hudson River region generally showed continued decreasing trends except for the Hudson River Estuary Monitoring Program which has shown a consistent increase since the early 2000’s (Figure 65). Both surveys in the South Atlantic region showed somewhat decreasing trends, but there was also a relatively high degree of annual variation in these surveys (Figure 66).

Overall, the probabilities of being less than the 25th percentile reference points in the terminal year (2016 in most cases) for each of the surveys were similar to those probabilities found for year 2010 (the last year of data used in the 2012 stock assessment; Table 23). This indicates relatively stable indices. One large difference between 2010 and 2016 was the NYDEC Alosine

Beach Seine survey in which the probability of being less than the 25th percentile reference point increased from 0.344 in 2010 to 0.720 in 2016. This is indicative of the continued decline of elver and yellow eels in this survey since the last stock assessment. In total, 3 of the 14 surveys included in the ARIMA modeling had greater than a 0.50 probability of being less than the 25th percentile reference point in the terminal year of the survey.

The 2012 Peer Review Panel noted that ARIMA is sensitive to the first data point in the time series and they suggested that trends be interpreted with caution, which is why this analysis is not used for developing reference points for American eel management but rather as one of the trend analyses used to draw general conclusions about the status of the stock.

6.5 Other Modeling Approaches

Several other modeling approaches were explored in ASMFC 2012 that were not updated for this report including a suite of models used by ICES (Study Leading to Informed Management of Eels or SLIME), Surplus Production Models (SPM; both age-structured and catch-free), Traffic Light Analysis (TLA), and Depletion-Based Stock Reduction Analysis (DB-SRA). The SLIME model was deemed inappropriate to the needs of the ASMFC for managing American eel. The SPMs did not find stable solutions and the TLA produced results that were difficult to interpret and therefore were not endorsed for management use by the Peer Review Panel in 2012. The Panel suggested that the TLA continue to be explored to incorporate more data, so while it could inform management decision-making in the future additional work on that model would require a peer review so it was not updated for this report. The Peer Review Panel endorsed the DB-SRA model for assessing American eel but had a number of concerns about the model (American Eel Stock Assessment Peer Review Report in ASMFC 2012). The Panel was impressed with the development of DB-SRA but ultimately were not comfortable using it to develop reference points or determine stock status without further refinements. Because further developing the DB-SRA would require a peer review for it to be used for management, the SAS did not update the model for this update report.

7 STOCK STATUS DETERMINATION

7.1 Status Determination Criteria and Current Stock Status

Reference points for determining the stock status of American eel in the U.S. in ASMFC 2012 were developed using the DB-SRA model which was not accepted for management use by the Peer Review Panel. The American Eel Technical Committee recommended that stock status was declared depleted based on trend analysis and the biomass trends estimated by the DB-SRA as recommended by the Peer Review Panel. The DB-SRA was not updated for this report because the Panel recommended it be further developed which was outside the guidelines of a stock assessment update. Therefore neither reference points nor stock status could be determined quantitatively by this stock assessment update. The trend analyses were updated and a discussion of overall trends follows in Section 8. Overall, the results in this update are very similar to the results in ASMFC 2012 and therefore the SAS and TC concluded the stock remains depleted.

8 DISCUSSION AND CONCLUSIONS

The data evaluated in this assessment provide evidence of neutral or declining abundance of American eel in the U.S in recent decades. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant declining trends in some indices over the time period examined. The Mann-Kendall test detected a significant declining trend in six of the 22 YOY indices, five of the 15 yellow eel indices, three of the nine regional trends, and the 30-year and 40-year yellow-phase abundance index. The remaining surveys tested had no trend, except for the North Anna Electrofishing and the regional Chesapeake Bay yellow eel indices which had a positive trend (although it should be noted that the North Anna Electrofishing survey was not updated from ASMFC 2012). These two surveys also had an increasing trend in ASMFC 2012, but the other two surveys that had an increasing trend in ASMFC 2012 (CTDEP Electrofishing Survey and PSEG Trawl Survey) now have no significant trend, noting that the time frame for the PSEG Trawl Survey changed since ASMFC 2012. The Manly meta-analysis showed a decline in at least one of the indices for both yellow and YOY life stages. Also, there was consensus for a decline for both life stages through time. Conclusions from the Manly meta-analysis results were the same as those in ASMFC 2012.

In ASMFC 2012, the ARIMA results indicated decreasing trends in the Hudson River and South Atlantic regions. For this update, the results of the ARIMA are the same except for the HRE Monitoring Program in the Hudson River region which has been increasing in recent years. Survey indices from the Chesapeake Bay and Delaware Bay/Mid-Atlantic Coastal Bays regions showed no consistent increasing or decreasing trends in ASMFC 2012, but now the Chesapeake Bay region surveys have increasing or stable trends and the Delaware Bay exhibits no directional trends in recent years. The probabilities of being less than the 25th percentile reference points in the terminal year for each of the surveys were similar to those in ASMFC 2012 and currently 3 of the fourteen surveys in the analysis have a greater than 50% probability of being less than the 25th percentile reference point.

ASMFC 2012 concluded that significant downward trends in some surveys across the coast was cause for concern. The trend analysis results in this stock assessment update are consistent with the ASMFC 2012 results, with few exceptions. Despite downward trends in the indices, commercial yellow American eel landings have been stable in the recent decades along the Atlantic coast (U.S. and Canada) although landings still remain much lower than historical landings. Compared to ASMFC 2012, there are more significantly downward trends in indices as indicated by the Mann-Kendall test and similar results for the ARIMA. This trend analysis and stable low landings support the update conclusion that the American eel population in the assessment range is similar to five years ago and remains depleted.

9 RESEARCH RECOMMENDATIONS

The following research recommendations are based on input from the ASMFC American Eel TC and SAS during the 2012 benchmark stock assessment and many remain relevant for this update stock assessment. A single asterisk (*) denotes short-term recommendations and two asterisks (**) denote long-term recommendations. Recommendations formatted in **bold** identify improvements needed for the next benchmark assessment. Notes have been added for this report regarding work that has been addressed or initiated since ASMFC 2012.

Data Collection

Fisheries Catch and Effort

- **Improve accuracy of commercial catch and effort data (NOTE: Some progress was made on this recommendation through Addenda III and IV)**
  - Compare buyer reports to reported state landings* (NOTE: Initiated in NY by NYDEC)
  - Improve compliance with landings and effort reporting requirements as outlined in the ASMFC FMP for American eel (see ASMFC 2000a for specific requirements)* (NOTE: Initiated in NY by NYDEC and NJ by NJDFW)
  - Require standardized reporting of trip-level landings and effort data for all states in inland waters; data should be collected using the ACCSP standards for collection of catch and effort data (ACCSP 2004 and initiated in NY by NYDEC)*
- Estimate catch and effort in personal-use and bait fisheries (NOTE: Initiated in NJ by NJDFW)
  - Monitor catch and effort in personal-use fisheries that are not currently covered by the MRFSS or commercial fisheries monitoring programs*
  - Implement a special-use permit for use of commercial fixed gear (e.g., pots and traps) to harvest American eels for personal use; special-use permit holders should be subject to the same reporting requirements for landings and effort as the commercial fishery**
  - Improve monitoring of catch and effort in bait fisheries (commercial and personal-use)*
- Estimate non-directed fishery losses
  - Recommend monitoring of discards in targeted and non-targeted fisheries*
  - Continue to require states to report non-harvest losses in their annual compliance reports*
- **Characterize the length, weight, age, and sex structure of commercially harvested American eels along the Atlantic Coast over time**
  - Require that states collect biological information by life stage (potentially through collaborative monitoring and research programs with dealers) including length, weight, age, and sex through fishery-dependent sampling programs; biological samples should be collected from gear types that target each life stage; at a minimum, length samples should be routinely collected from commercial fisheries* (NOTE: Initiated in Chesapeake Bay sites (VMRC) and in NY, NJ, DE, MD by NYDEC, NJDFW, DEDFW, and MDDNR respectively)
  - Finish protocol for sampling fisheries; SASC has draft protocol in development*
- Improve estimates of recreational catch and effort
  - Collect site-specific information on the recreational harvest of American eels in inland waters; this could be addressed by expanding the MRIP into inland areas**

- Improve knowledge of fisheries occurring south of the U.S. and within the species' range that may affect the U.S. portion of the stock (i.e., West Indies, Mexico, Central America, and South America)**

Socioeconomic Considerations

- Perform economics studies to determine the value of the fishery and the impact of regulatory management**
- Improve knowledge regarding subsistence fisheries
  - Review the historic participation level of subsistence fishers and relevant issues brought forth with respect to those subsistence fishers involved with American eel**
  - Investigate American eel harvest and resource by subsistence harvesters (e.g., Native American tribes, Asian and European ethnic groups)**

Distribution, Abundance, & Growth

- **Improve understanding of the distribution and frequency of occurrence of American eels along the Atlantic Coast over time** (see Cairns et al. 2017 for a description of the distribution of American eels from Canada to Florida)
  - Maintain and update the list of fisheries-independent surveys that have caught American eels and note the appropriate contact person for each survey* (NOTE: Work being done in NY by NYDEC and NJ by NJDFW)
  - Request that states record the number of eels caught by fishery-independent surveys; recommend states collect biological information by life stage including length, weight, age, and sex of eels caught in fishery-independent sampling programs; at a minimum, length samples should be routinely collected from fishery-independent surveys* (NOTE: NYDEC began this in 2014; NJDFW collects numbers and lengths; VIMS collects numbers, lengths, weights, ages, and disease status; NCDMF collects numbers and lengths; work being done through FL FWC and a freshwater electrofishing survey)
  - Encourage states to implement surveys that directly target and measure abundance of yellow- and silver-stage American eels, especially in states where few targeted eel surveys are conducted** (NOTE: MA, MD, and NJ yellow eel survey began in 2015 by MADMF, MDDNR, and NJDFW)
  - A coastwide sampling program for yellow and silver American eels should be developed using standardized and statistically robust methodologies**
- Improve understanding of coastwide recruitment trends
  - Continue the ASMFC-mandated YOY surveys; these surveys could be particularly valuable as an early warning signal of recruitment failure* (NOTE: All states have a state-mandated YOY survey except for GA)
  - Develop proceedings document for the 2006 ASMFC YOY Survey Workshop; follow-up on decisions and recommendations made at the workshop*

  - Examine age at entry of glass eel into estuaries and freshwater** (NOTE: see Pratt et al. 2014)
  - Develop monitoring framework to provide information for future modeling on the influence of environmental factors and climate change on recruitment**

- Improve knowledge and understanding of the portion of the American eel population occurring south of the U.S. (i.e., West Indies, Mexico, Central America, and South America)**

<u>Future Research</u>

Biology

- Improve understanding of the leptocephalus stage of American eel
  - Examine the mechanisms for exit from the Sargasso Sea and transport across the continental shelf** (NOTE: see Rypina et al 2014)
  - Examine the mode of nutrition for leptocephalus in the ocean**

- Improve understanding of impact of contaminants as sources of mortality and non-lethal population stressors
  - Investigate the effects of environmental contaminants on fecundity, natural mortality, and overall health**
  - Research the effects of bioaccumulation with respect to impacts on survival and growth (by age) and effect on maturation and reproductive success**

- **Improve understanding of impact of *Anguillicoloides crassus* on American eel**
  - Investigate the prevalence and incidence of infection by the nematode parasite *A. crassus* across the species range* (NOTE: Initiated in NC with a Roanoke study and in FL, work currently underway in the Chesapeake Bay through Z. Warshafsky's graduate work at VIMS, see also Zimmerman and Welsh 2012, Campbell et al. 2013, Denny et al. 2013, Waldt et al. 2013, Hein et al. 2014)
  - Research the effects of the swim bladder parasite *A. crassus* on the American eel's growth and maturation, migration to the Sargasso Sea, and the spawning potential* (NOTE: work currently underway in the Chesapeake Bay through Z. Warshafsky's graduate work at VIMS, see also Zimmerman and Welsh 2012)
  - Investigate the impact of the introduction of *A. crassus* into areas that are presently free of the parasite**

- **Improve understanding of spawning and maturation**
  - Investigate relation between fecundity and length and fecundity and weight for females throughout their range**
  - Identify triggering mechanism for metamorphosis to mature adult, silver eel life stage, with specific emphasis on the size and age of the onset of maturity, by sex; a maturity

  - schedule (proportion mature by size or age) would be extremely useful in combination with migration rates**
  - Research mechanisms of recognition of the spawning area by silver eel, mate location in the Sargasso Sea, spawning behavior, and gonadal development in maturation**
  - Examine migratory routes and guidance mechanisms for silver eel in the ocean**

- Improve understanding of predator-prey relationships**

- Investigating the mechanisms driving sexual determination and the potential management implications**

Passage & Habitat

- **Improve upstream and downstream passage for all life stages of American eels (NOTE: Initiated in ME, also see Hitt et al. 2012, Gardner et al. 2013)**
  - Develop design standards for upstream passage devices for eels. The ASMFC 2011 Eel Passage Workshop (ASMFC 2013) made contributions to this goal.
  - Investigate, develop, and improve technologies for American eel passage upstream and downstream at various barriers for each life stage; in particular, investigate low-cost alternatives to traditional fishway designs for passage of eel** (NOTE: MADMF designed and deployed a gravity fed eel pass)

- Improve understanding of the impact of barriers on upstream and downstream movement (NOTE: Sweka et al. 2014 used an egg per recruit model to evaluate the costs/benefits to reproductive output with transport of eels upstream of hydroelectric dams and found that without downstream passage, transporting eels upstream resulted in a net loss of reproductive output.)
  - Evaluate the impact, both upstream and downstream, of barriers to eel movement with respect to population and distribution effects; determine relative contribution of historic loss of habitat to potential eel population and reproductive capacity**
  - Recommend monitoring of upstream and downstream movement at migratory barriers that are efficient at passing eels (e.g., fish ladder/lift counts); data that should be collected include presence/absence, abundance, and biological information; provide standardized protocols for monitoring eels at passage facilities; coordinate compilation of these data; provide guidance on the need and purpose of site-specific monitoring**
  - Use the information gained from the above evaluation and monitoring of barriers to American eel passage to develop metrics for prioritizing passage restoration projects.

- **Improve understanding of habitat needs and availability**
  - Assess characteristics and distribution of American eel habitat and value of habitat with respect to growth and sex determination; develop GIS of American eel habitat in U.S.**
  - Assess available drainage area over time to account for temporal changes in carrying capacity; develop GIS of major passage barriers**

  - Improve understanding of freshwater habitat and water quality thresholds for American eel.
- Improve understanding of within-drainage behavior and movement and the exchange between freshwater and estuarine systems**
- Improve estimates of mortality associated with upstream and downstream passage
  - Monitor non-harvest losses such as impingement, entrainment, spill, and hydropower turbine mortality* (NOTE: Data available for the Susquehanna and Shenandoah Rivers from Eyler et al. 2016 and USFWS 2012.)
- Evaluate eel impingement and entrainment at facilities with NPDES authorization for large water withdrawals; quantify regional mortality and determine if indices of abundance could be established as specific facilities** (NOTE: Data available for the Delaware River through work done by the Delaware River Basin Fish and Wildlife Management Cooperative)
- Investigate best methods for reintroducing eels into a watershed; examine approaches for determining optimum density* (Note: Data available from the Roanoke Rapids and Susquehanna River through a project with Dominion Energy and USFWS-Maryland Fish and Wildlife Conservation Office, respectively)

<u>Assessment Methodology & Management Support</u>

- Coordinate monitoring, assessment, and management among agencies that have jurisdiction within the species' range (e.g., ASMFC, GLFC, Canada DFO)**
- Perform a joint U.S.-Canadian stock assessment*
- Perform periodic stock assessments (every 5–7 years) and establish sustainable reference points for American eel are required to develop a sustainable harvest rate in addition to determining whether the population is stable, decreasing, or increasing
  - Develop new assessment models (e.g., delay-difference model) specific to eel life history and fit to available indices**
  - **Conduct intensive age and growth studies at regional index sites to support development of reference points and estimates of exploitation* (NOTE: Initiated in the Chesapeake Bay by MDDNR which has collected age information on selected tributaries since 1998)**
  - Develop GIS-type model that incorporates habitat type, abundance, contamination, and other environmental factors**
  - Develop population targets based on habitat availability at the regional and local level**
- Implement large-scale (coastwide or regional) tagging studies of eels at different life stages; tagging studies could address a number of issues including:
  - Natural, fishing, and discard mortality; survival**
  - Growth**

- Passage mortality**
- Movement, migration, and residency**
- Validation of ageing methods**
- Reporting rates**
- Tag shedding or tag attrition rates**

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11 TABLES

Table 1. Commercial fishery regulations for American eels as of 2016, by state. For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

State	Min Size Limit	License/Permit	Other
ME	Glass no min size	Daily dealer reports/swipe card program; monthly harvester report of daily landings. Tribal permit system in place for some Native American groups.	Harvester license lottery system.
	Yellow 9”	Harvester/dealer license and monthly reporting. Tribal permit system in place for some Native American groups.	Seasonal closures. Gear restrictions. Weekly closures.
NH	9”	Commercial saltwater license and wholesaler license. No dealer reports. Monthly harvester reporting includes dealer information.	Gear restrictions in freshwater.
MA	9"	Commercial permit with annual catch report requirement. Registration for dealers with purchase record requirement. Dealer/harvester reporting.	Traps, pots, spears, and angling only. Mesh restrictions.
RI	9"	Commercial fishing license. Dealer/harvester reporting.	Gear restrictions.
CT	9"	Commercial license (not required for personal use). Dealer/harvester reporting.	Gear restrictions.
NY	9"	Harvester/dealer license and reporting.	Gear restrictions. Maximum limit of 14” in some rivers.
NJ	9"	License required. No dealer reports. Monthly harvester reporting includes dealer information.	Gear restrictions.
PA	NO COMMERCIAL FISHERY		
DE	6"	Harvester reporting, no dealer reporting. License required.	Commercial fishing in tidal waters only. Gear restrictions.
MD	9"	Dealer/harvester license and monthly reporting.	Prohibited in non-tidal waters. Gear restrictions. Commercial crabbers may fish 50 pots per day, must submit catch reports.
DC	NO COMMERCIAL FISHERY		
PRFC	9"	Harvester license and daily reporting due weekly. No dealer reporting.	Gear restrictions.
VA	9"	Harvester license required. Dealer/harvester monthly reporting.	Mesh size restrictions on eel pots. Seasonal closures.

Table 1. ***Continued*****.**

State	**Min Size Limit**	**License/Permit**	**Other**
NC	9"	Standard Commercial Fishing License for all commercial fishing. Dealer/harvester monthly combined reports on trip ticket.	Mesh size restrictions on eel pots. Seasonal closures.
SC	Glass no min size	Fyke and dip net only permitted. Dealer/harvester monthly combined reports on trip ticket.	Max 10 individuals. Gear and area restrictions.
	Yellow 9”	Pots only permitted. Dealer/harvester monthly combined reports on trip ticket.	Gear restrictions.
GA	9"	Personal commercial fishing license and commercial fishing boat license. Dealer/harvester monthly combined reports on trip ticket.	Gear restrictions on traps and pots. Area restrictions.
FL	9”	Permits and licenses. Harvester reporting. No dealer reporting.	Gear restrictions.

Table 2. Recreational fishery regulations for American eels as of 2016, by state. For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

State	Size Limit	Possession Limit	Other
ME	9"	25 eels/person/day	Gear restrictions. License requirement and seasonal closures (inland waters only). Bait limit of 50 eels/day for party/charter boat captain and crew.
NH	9"	25 eels/person/day	Coastal harvest permit needed if taking eels other than by angling. Gear restrictions in freshwater.
MA	9"	25 eels/person/day	Nets, Pots, traps, spears, and angling only; mesh restrictions.
RI	9"	25 eels/person/day	
CT	9"	25 eels/person/day	
NY	9”	25/eels/person/day	Maximum limit of 14” in some rivers. Bait limit of 50 eels/day for party/charter boat captain and crew.
NJ	9"	25 eels/person/day	Bait limit of 50 eels/day for party/charter boat captain and crew.
PA	9"	25 eels/person/day	Gear restrictions. Bait limit of 50 eels/day for party/charter boat captain and crew.
DE	6"	50 eels/person/day	Two pot limit/person.
MD	9"	25 eels/person/day	Gear restrictions.
DC	9"	10 eels/person/day	
PRFC	9"	25 eels/person/day	
VA	9"	25 eels/person/day	Recreational license. Two pot limit. Mandatory annual catch report. Gear restrictions. Bait limit of 50 eels/day for party/charter boat captain and crew.
NC	9"	25 eels/person/day	Gear restrictions. Non-commercial special device license. Two eel pots allowed under Recreational Commercial Gear license. Bait limit of 50 eels/day for party/charter boat captain and crew.
SC	9”	25 eels/person/day	Gear restrictions. Permits and licenses. Two pot limit
GA	9”	25 eels/person/day	
FL	9”	25 eels/person/day	Gear restrictions. Wholesale/Retail purchase exemption applies to possession limit for bait.

Table 3. Summary of current state/jurisdiction reporting structure for commercial eel landings and quota management per Addendum VI requirements.

State	Rulemaking Process	Rulemaking Timeframe	Reporting to monitor quota	Overages and Transfers	Additional Measures Planned
Maine	DMR Authority	up to 100 days	Monthly harvester. Likely to use swipe card system	Y	Possible seasons and days out by 2017
New Hampshire	Director Authority	at least 1 month	Monthly harvester	Y	None, but can if needed
Massachusetts	MF Advisory Commission	by March 2016	Weekly dealer (personal bait not counted)	Y	Close H&L gear Sept 1-Dec 31
Rhode Island	Director Authority	30 day public comment	Dealer twice a week	Y	None, but can if needed
Connecticut	DEEP Authority	10 days public notice	Monthly harvester	Y	None, but can if needed
New York	DEC Authority	6 months	Monthly harvester (river/marine) and weekly dealer (marine)	Y	Closed pot fishery on Delaware River. Need adjustment to quota through transfers or management addendum.
New Jersey	Commissioner/Council Rulemaking	3-4 months	Monthly harvester	Y	Limited entry based on 2007-2014 harvest. Possible pot maximum, and seasons. Some through notice process while others up to two years.
Delaware	Legislature (resumes in Jan 2016)	Legislature Session Jan-June	Daily harvester	Legislature	None, but can if needed
Maryland	DNR Authority	100 days or 48h with public notice authority	Daily harvester	Y	Harvester permit by 03/2016 with reporting requirement
PRFC	PRFC Authority	1-2 months	Weekly harvester	Y	None, but can if needed
Virginia	VMRC Authority	1 month	Monthly harvester with dealer check	Y	Possible seasonal closures and possession limits. Quota trigger to implement weekly/daily dealer reports.
North Carolina	NCDMF Authority	Immediate	Monthly dealer and harvester log books	Y	Proactive reporting trigger program to weekly/daily and closure at 85% of quota.
South Carolina	Legislature, but permitting authority	Permit cycle June 30	Monthly harvester and dealer	Y	Possible gear restrictions, seasons, catch limits, or closure
Georgia	Natural Resources Authority	Up to 90 days	Monthly harvester and dealer	Y	Likely close eel commercial fishery if state by state quotas are implemented
Florida	Executive Order Rulemaking	Governor-commission meets 5 times a year	Monthly harvester, weekly harvester when 50% quota is reached	Y	None, but can if needed. Issue of harvester selling to dealers outside the state and potential double counting of quota

Table 4. Numbers of American eel samples reported by the MRIP/MRFSS angler-intercept survey and at-sea headboat survey, by catch type, 1981–2016.

Year	Type A	Type B1		Type B2	
	Intercept	Intercept	Headboat	Intercept	Headboat
1981	22	75		94	
1982	75	44		43	
1983	28	19		73	
1984	28	12		26	
1985	53	17		91	
1986	62	41		138	
1987	16	34		49	
1988	35	36		74	
1989	57	31		150	
1990	36	16		154	
1991	113	30		123	
1992	13	25		101	
1993	224	40		101	
1994	98	48		89	
1995	23	6		96	
1996	18	29		77	
1997	9	8		50	
1998	7	3		84	
1999	4	7		70	
2000	7	5		43	
2001	1	8		44	
2002	6	10		79	
2003	16	16		155	
2004	13	16		99	
2005	7	3		65	
2006	7	3		76	
2007	39	7		73	
2008	4	5		66	
2009	9	4		75	
2010	14	22		117	
2011	2	4		91	
2012	11	42		119	
2013	10	5		99	
2014	5	12		99	
2015	1	6		100	
2016	7	20		92	

Table 5. Numbers of American eels available for biological sampling in the MRIP/ MRFSS angler-intercept survey and at-sea headboat survey, by survey component, 1981–2016.

Year	Intercept (Type A)		Headboat (Type B2)
	Weighed	Measured	Measured
1981	21	21	
1982	46	49	
1983	16	16	
1984	22	22	
1985	30	27	
1986	25	18	
1987	13	10	
1988	28	27	
1989	47	29	
1990	12	17	
1991	37	35	
1992	3	3	
1993	15	32	
1994	21	13	
1995	2	2	
1996	5	5	
1997	7	7	
1998	3	4	
1999	1	2	
2000	7	7	
2001	0	1	
2002	1	2	
2003	0	2	
2004	11	13	
2005	4	6	1
2006	3	3	1
2007	3	4	6
2008	2	3	8
2009	4	4	1
2010	6	6	2
2011	1	0	1
2012	5	5	1
2013	3	6	2
2014	1	4	0
2015	0	1	0
2016	3	4	2

Table 6. **Estimates of recreational fishery harvest and released alive for American eels along the Atlantic coast, 1981–2015. The precision of each estimate, measured as proportional standard error (PSE), is also given. Estimates for 1981-2003 have been calibrated to MRIP from MRFSS.**

Year	Harvest (Type A+B1)				Released Alive (Type B2)	
	Numbers	PSE[Num]	Weight (lbs)	PSE[Weight]	Numbers	PSE[Num]
1981	117,583	53.6	99,918	46.2	117,131	53.2
1982	197,724	62.6	130,815	44.3	85,001	64.6
1983	120,777	82.8	105,986	60.2	83,688	40.4
1984	81,524	54.1	78,306	47.6	49,277	60.7
1985	220,596	77.8	218,269	30.4	85,031	47.9
1986	138,583	56.6	112,388	39.7	120,993	35.4
1987	51,714	63.8	38,972	51.7	65,609	50.7
1988	85,483	52.3	41,166	32.6	104,581	52.8
1989	68,748	50.7	92,589	34.8	113,377	30.9
1990	33,324	55.9	18,239	45.8	99,998	31.0
1991	106,427	62.9	79,603	42.2	80,022	42.4
1992	42,846	70.7	2,717	28.2	55,788	48.2
1993	97,664	75.1	60,714	61.0	87,265	40.7
1994	67,999	63.1	34,420	53.1	70,089	32.3
1995	12,598	108	1,304	28.2	64,478	45.4
1996	28,149	67.4	8,765	56.9	56,131	34.3
1997	21,256	111	9,118	61.8	26,707	43.3
1998	8,543	80.6	4,625	88.0	57,803	41.8
1999	7,739	87.4	497	28.2	56,574	95.1
2000	37,084	144	18,398	92.2	48,119	52.9
2001	14,798	149			30,739	40.0
2002	7,625	74.7	812	28.2	47,952	31.8
2003	42,582	119			157,189	33.5
2004	41,286	61.4	41,191	65.2	74,653	24.6
2005	5,217	48.4	4,309	54.3	63,939	40.8
2006	19,389	53.6	15,917	49.2	99,974	42.2
2007	40,676	60.1	46,700	85.4	113,424	47.3
2008	3,062	46.0	1,245	61.4	62,625	34.5
2009	9,890	57.6	6,616	62.4	92,399	31.3
2010	129,803	78.7	31,518	64.1	90,437	28.6
2011	6,860	51.4	5,314	73.3	81,848	28.5
2012	38,493	49.0	11,999	52.1	143,868	34.1
2013	8,833	48.9	6,030	36.1	115,359	25.5
2014	5,974	47.6	7,684	61.4	148,598	53.1
2015	4,077	48.7	10,855	59.8	54,227	24.2
2016	63,946	18.8	107,480	18.0	60,589	39.6

Table 7. Summary of GLM analyses used to standardize YOY indices developed from the ASMFC-mandated and non-mandated (indicated with an * next to the survey name) recruitment surveys. Phi is the overdispersion parameter. For GLM standardized indices, the response variable was American eel catch. If a GLM wasn't applied, a nominal index was computed; nominal indices computed as ratio estimators.

Region	State	Site	Years	Gear	GLM?	Error	Predictors	Phi
Gulf of Maine	ME	West Harbor Pond	2001-2016	Irish Elver Ramp	N			
	NH	Lamprey River	2001-2016	Irish Elver Trap	Y	NB	Year+WaterTemp	1.48
	MA	Jones River	2001-2016	Sheldon Elver Trap	Y	NB	Year+Discharge	1.08
Southern New England	CT	Ingham Hill	2007-2016	Irish Elver Ramp	N			
	RI	Gilbert Stuart Dam	2000-2016	Irish Elver Ramp	Y	NB	Year+WaterTemp+WaterLevel	1.38
	RI	Hamilton Fish Ladder	2004-2016	Irish Elver Ramp	Y	NB	Year+WaterLevel	1.43
	NY	Carman's River	2000-2016	Fyke Net	Y	NB	Year+WaterTemp	1.74
Hudson River	NY	HRE Monitoring *	1974-2013	Epibenthic Sled and Tucker Trawl	Y	Delta-gamma	Year + Month + Strata + Rivermile + Volume	0.66
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Patcong Creek	2004-2016	Fyke Net	N			
	NJ	Little Egg Inlet Ichthyoplankton *	1992-2015	Plankton Net	Y	NB	Year + Month + Flow meter + River discharge	1.07
	DE	Millsboro Dam	2000-2016	Fyke Net	Y	NB	Year+Discharge	1.76
	MD	Turville Creek	2000-2016	Irish Elver Ramp	N			
Chesapeake Bay	PRFC	Clark's Millpond	2000-2013	Irish Elver Ramp	N			
	PRFC	Gardy's Millpond	2000-2016	Irish Elver Ramp	N			
	VA	Bracken's Pond	2000-2016	Irish Elver Ramp	N			
	VA	Kamp's Millpond	2000-2016	Irish Elver Ramp	N			
	VA	Wareham's Pond	2003-2016	Irish Elver Ramp	Y	NB	Year+WaterTemp	1.31
	VA	Wormley Creek	2001-2016	Irish Elver Ramp	Y	NB	Year+WaterTemp	1.54
South Atlantic	NC	Beaufort Bridgenet Ichthyoplankton	1987-2007	Plankton Net	Y	NB	Year + Month + River discharge	1.27
	SC	Goose Creek	2000-2015	Fyke Net	Y	NB	Year+WaterTemp	1.09
	GA	Altamaha Canal	2001-2010	Fyke Net	Y	LN	Year+WaterTemp	1.11
	FL	Guana River Dam	2001-2016	Dip Net	N			

Table 8. Spearman's rank correlation between YOY indices developed from the ASMFC-mandated recruitment surveys. Values formatted in bold and italicized font are statistically significant at α < 0.10. NC's Beaufort Bridgenet Ichthyoplankton Sampling Program (BBISP) and CT's Ingham Hill indices only overlap for one year and therefore are "NA" in the table.

	Region	Gulf of Maine			Southern New England				Delaware Bay/Mid-Atl			Chesapeake Bay						South Atlantic		
Region	Survey Site	West Harbor Pond (ME)	Lamprey River (NH)	Jones River (MA)	Ingham Hill (CT)	Gilbert Stuart Dam (RI)	Hamilton Ladder (RI)	Carman's River (NY)	Patcong Creek (NJ)	Mills-boro Dam (DE)	Turville Creek (MD)	Clarks Millpond (PRFC)	Gardys Millpond (PRFC)	Brackens Pond (VA)	Kamps Millpond (VA)	Warehams Pond (VA)	Wormley Creek (VA)	BBISP (NC)	Goose Creek (SC)	Altamaha Canal (GA)
Gulf of Maine	Lamprey River (NH)	***0.532***																		
	Jones River (MA)	-0.362	-0.503																	
Southern New England	Ingham Hill (CT)	0.079	-0.224	***0.455***																
	Gilbert Stuart Dam (RI)	***0.418***	***0.476***	-0.288	0.236															
	Hamilton Fish Ladder (RI)	0.220	0.363	-0.467	-0.030	***0.505***														
	Carman's River (NY)	***0.506***	***0.535***	-0.359	0.127	***0.502***	0.319													
Delaware Bay/Mid-Atl	Patcong Creek (NJ)	0.343	***0.446***	0.032	0.183	0.332	-0.266	0.224												
	Millsboro Dam (DE)	***0.432***	***0.585***	-0.253	0.042	***0.368***	***0.434***	0.294	0.265											
	Turville Creek (MD)	0.029	-0.109	-0.203	0.176	0.157	0.049	-0.233	-0.335	0.294										
Chesapeake Bay	Clarks Millpond (PRFC)	-0.332	-0.326	0.132	0.115	-0.103	-0.462	0.118	0.009	-0.221	-0.005									
	Gardys Millpond (PRFC)	0.276	0.106	0.094	0.188	0.230	0.115	0.324	-0.091	0.211	0.002	-0.235								
	Brackens Pond (VA)	-0.179	-0.321	***0.685***	***0.564***	0.228	-0.154	-0.162	-0.029	0.032	0.235	0.208	-0.096							
	Kamps Millpond (VA)	***0.597***	0.256	-0.132	0.127	0.206	0.093	0.162	0.053	0.145	0.174	0.115	0.061	0.074						
	Warehams Pond (VA)	0.126	0.258	0.005	0.000	0.330	0.126	-0.049	0.343	-0.297	0.126	-0.511	0.077	-0.038	-0.104					
	Wormley Creek (VA)	-0.385	0.171	-0.071	-0.224	0.109	-0.005	-0.218	-0.118	0.206	0.194	0.335	-0.300	0.162	0.103	-0.291				
South Atlantic	BBISP (NC)	***0.679***	0.107	-0.286	NA	0.214	0.400	0.452	0.071	-0.452	-0.429	0.214	0.119	-0.452	***0.786***	-0.700	-0.429			
	Goose Creek (SC)	0.021	-0.271	***0.496***	0.183	-0.288	-0.112	-0.259	-0.132	-0.141	-0.379	-0.144	0.021	0.074	0.221	-0.434	0.061	0.476		
	Altamaha Canal (GA)	-0.079	0.164	0.309	0.600	-0.345	0.107	-0.212	-0.006	***0.455***	-0.067	-0.442	-0.067	0.236	0.103	0.000	0.297	-0.536	0.394	
	Guana River Dam (FL)	-0.147	-0.456	***0.491***	-0.455	-0.115	-0.280	-0.371	-0.275	-0.388	-0.094	0.085	0.100	0.203	0.215	-0.115	0.124	0.286	***0.629***	-0.200

Table 9. Summary of GLM analyses used to standardize fisheries-independent indices developed from elver and yellow eel American eel surveys. Phi is the overdispersion parameter.

Region	State	Survey	Location	Years	Gear	Life Stage(s)	GLM ?	Error	Predictors	Phi
Southern New England	CT	CTDEP Electrofishing Survey	Farmill River	2001-2014	Electrofishing	Elver & Yellow	N			
	NY	NY Western Long Island Survey	Western Long Island	1984-2016	Seine	Yellow	Y	NB	Year + Month + Lat	0.48
Hudson River	NY	HRE Monitoring Program	Hudson River	1974-2013	Epidbenthic Sled and Tucker Trawl	Yearling & older	Y	NB	Year + Gear + Month + Strata + Rivermile +Volume	1.91
	NY	NYDEC Alosine Beach Seine Survey	Hudson River	1980-2016	Seine	Elver & Yellow	Y	NB	Year + Month + Rivermile	1.23
	NY	NYDEC Striped Bass Beach Seine Survey	Hudson River	1980-2016	Seine	Elver & Yellow	Y	NB	Year + Month + Longitude	1.31
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	NJDFW Striped Bass Seine	Delaware River	1980-2016	Seine	Yellow	Y	NB	Year + Water temp + Salinity	1.02
	DE	Delaware Trawl Survey	Delaware River	1982-2016	Trawl	Elver & Yellow	Y	NB	Year + Month + Surf_Temp + Surf_Sal	2.18
	DE	PSEG Trawl Survey	Delaware River	1998-2016	Trawl	Elver & Yellow	Y	NB	Year + Month + Bot_S	1.95
	PA	Area 6 Electrofishing Survey	Delaware River	1999-2016	Electrofishing	Elver	Y	NB	Year + Site	1.16

Table 9. ***Continued*.**

Region	State	Survey	Location	Years	Gear	Life Stage(s)	GLM ?	Error	Predictors	Phi
Chesapeake Bay	MD	MDDNR Striped Bass Seine	Chesapeake Bay	1966-2016	Seine	Yellow	Y	NB	Year + Month + Salinity	0.95
	VA	North Anna Electrofishing Survey	North Anna River	1990-2009	Electrofishing	Elver & Yellow	Y	NB	Year+GearType+TimePeriod+Station	1.20
	VA	VIMS Juvenile Striped Bass Seine Survey - long	Lower Ches Bay & Trib	1967-2016	Seine	Yellow	Y	NB	Year + SYSTEM	1.69
	VA	VIMS Juvenile Striped Bass Seine Survey - short	Lower Ches Bay & Trib	1989-2016	Seine	Yellow	Y	NB	Year + STATION TYPE	1.38
South Atlantic	NC	NCDMF Estuarine Trawl Survey	NC waters	1989-2016	Trawl	Elver & Yellow	Y	NB	Year + Lat + Lon + Bottomtype	1.29
	SC	SC Electrofishing Survey	SC waters	2001-2016	Electrofishing	Elver & Yellow	Y	NB	Year + Strata + Water temp + Salinity + Tide Stage	1.10

Table 10. Spearman's rank correlation between yellow American eel indices. Values formatted in bold and italicized font are statistically significant at α < 0.10.

	Region	**S. New England**		**Hudson River**			**Delaware Bay/Mid-Atl**				**Chesapeake Bay**			**South Atlantic**
Region	Survey Site	CTDEP (CT)	W. Long Island (NY)	HRE Monitoring (NY)	NYDEC Alosine Beach Seine (NY)	NYDEC Striped Bass Beach Seine (NY)	NJDFW Striped Bass Seine (NJ)	Delaware Trawl (DE)	PSEG Trawl Survey (DE)	Area 6 Electrofishing (PA)	MDDNR Striped Bass Seine (MD)	North Anna (VA)	VIMS Juvenile Striped Bass Seine —short (VA)	NCDMF Estuarine Trawl Survey (NC)
S. New England	W. Long Island Study (NY)	-0.254												
Hudson River	HRE Monitoring (NY)	***0.406***	***0.440***											
	NYDEC Alosine Beach Seine (NY)	0.091	***0.279***	***0.284***										
	NYDEC Striped Bass Beach Seine (NY)	0.168	***0.492***	***0.726***	***0.290***									
Delaware Bay/Mid-Atl	NJDFW Striped Bass Seine (NJ)	0.147	0.129	-0.033	***0.237***	0.085								
	Delaware Trawl (DE)	-0.063	-0.162	-0.087	0.120	0.171	***0.296***							
	PSEG Trawl Survey (DE)	-0.217	-0.203	0.158	-0.275	-0.235	-0.226	0.198						
	Survey (PA)	***0.706***	0.087	***0.493***	-0.183	0.110	-0.042	-0.187	-0.028					
Chesapeake Bay	Seine (MD)	-0.007	0.105	0.047	0.131	0.184	0.099	***0.296***	0.096	-0.247				
	North Anna (VA)	***0.857***	-0.171	-0.337	0.147	-0.377	***0.575***	-0.107	0.264	***0.455***	***0.389***			
	VIMS Juvenile Striped Bass Seine —short (VA)	***0.552***	-0.077	-0.201	-0.083	0.057	-0.055	0.117	-0.175	0.115	0.139	0.072		
South Atlantic	NCDMF Estuarine Trawl Survey (NC)	0.098	0.024	***0.461***	0.111	***0.426***	-0.346	-0.098	-0.056	-0.218	-0.445	-0.491	-0.006	
	SC Electrofishing Survey (SC)	-0.217	***0.534***	-0.436	0.168	-0.238	***0.382***	***0.468***	***0.388***	-0.174	0.206	-0.167	-0.282	-0.491

Table 11. Summary of surveys used in development of region-specific indices of American eel relative abundance. Asterisks (*) denote the ASMFC-mandated recruitment surveys. A Southern New England regional yellow eel index was not developed due to concerns about the indices in that region, see section 6.2.2 for more information.

Region	Life Stage	Time Period	Survey
Gulf of Maine	YOY	2001–2016	West Harbor Pond (ME) *
			Lamprey River (NH) *
			Jones River (MA) *
	Yellow		*none available*
Southern New England	YOY	2000–2016	Gilbert Stuart Dam (RI) *
			Hamilton Fish Ladder (RI) *
			Ingham Hill (CT) *
			Carman's River (NY) *
	Yellow	2000–2012	CTDEP Electrofishing Survey (CT)
			NY Western Long Island Survey (NY)
Hudson River	YOY	1974–2013	HRE Monitoring Program (NY)
	Yellow	1980–2015	HRE Monitoring Program (NY)
			NYDEC Alosine Beach Seine Survey (NY)
			NYDEC Striped Bass Beach Seine Survey (NY)
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	2000–2016	Millsboro Dam (DE) *
			Patcong Creek (NJ) *
			Little Egg Inlet Ichthyoplankton Survey (NJ)
			Turville Creek (MD) *
	Yellow	1999–2015	NJDFW Striped Bass Seine (NJ)
			Delaware Trawl Survey (DE)
			PSEG Trawl Survey (DE)
			Area 6 Electrofishing Survey (PA)
Chesapeake Bay	YOY	2000–2016	Clark's Millpond (PRFC) *
			Gardy's Millpond (PRFC) *
			Bracken's Pond (VA) *
			Kamp's Millpond (VA) *
			Warehams Pond (VA) *
			Wormley Creek (VA) *
	Yellow	1990–2009	MDDNR Striped Bass Seine (MD)
			North Anna Electrofishing Survey (VA)
			VIMS Juvenile Striped Bass Seine Survey—short (VA)
South Atlantic	YOY	2000–2015	Beaufort Bridgenet Ichthyoplankton (NC) *
			Goose Creek (SC) *
			Altamaha Canal (GA) *
			Guana River Dam (FL) *
	Yellow	2001–2016	NCDMF Estuarine Trawl Survey (NC)
			SC Electrofishing Survey (SC)

Table 12. Spearman's rank correlation between regional YOY indices for American eel. Values formatted in *bold and italicized* font are statistically significant at α < 0.10.

	Gulf of Maine	Southern New England	Hudson River	Delaware Bay/Mid-Atlantic	Chesapeake Bay
Southern New England	0.053				
Hudson River	***0.500***	0.345			
Delaware Bay/Mid-Atlantic	***0.535***	***0.417***	***0.486***		
Chesapeake Bay	0.050	0.096	0.244	0.029	
South Atlantic	0.221	-0.285	***0.415***	-0.141	0.091

Table 13. Spearman's rank correlation between regional yellow-phase indices for American eel. Values formatted in *bold and italicized* font. None of the values are statistically significant at α < 0.10.

	Hudson River	Delaware Bay/ Mid-Atlantic Coastal Bays	Chesapeake Bay
Delaware Bay/ Mid-Atlantic Coastal Bays	-0.026		
Chesapeake Bay	-0.367	0.227	
South Atlantic	-0.372	-0.215	-0.050

Table 14. Spearman's rank correlation coefficients (ρ) and associated *P*-values from correlation of region-specific yellow-phase indices and lagged YOY indices for American eel. Values formatted in *bold and italicized* font are statistically significant at α < 0.10. There was no regional yellow eel index for Gulf of Maine or Southern New England.

Region	Yellow vs.	Lag (years)	*ρ*	*P* > |*ρ*|
Hudson River	YOY	0	0.011	0.477
		1	***0.269***	0.087
		2	***0.277***	0.085
		3	***0.476***	0.008
		4	***0.521***	0.004
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	0	0.199	0.222
		1	0.194	0.228
		2	-0.126	0.684
		3	0.039	0.446
		4	0.349	0.110
Chesapeake Bay	YOY	0	-0.370	0.861
		1	-0.091	0.612
		2	***0.734***	0.005
		3	0.137	0.328
		4	-0.024	0.536
South Atlantic	YOY	0	0.300	0.138
		1	***0.714***	0.003
		2	***0.473***	0.053
		3	0.364	0.123
		4	***0.573***	0.035

Table 15. Parameter estimates (standard errors in parentheses) of the allometric length (mm)-weight (g) relation fit to available data for American eel by region, sex, and all data pooled. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

	Subset	**n**	***a***	***b***
None	All	68,334	4.05E-7 (1.324E-8)	3.25 (0.00509)
Region	Gulf of Maine	3,420	6.49E-7 (3.574E-8)	3.17 (0.00843)
	Southern New England	166	5.10E-5 (4.10E-5*)	2.52 (0.1236)
	Hudson River	2,249	1.27E-6 (1.956E-7)	3.06 (0.0240)
	Del Bay/Mid-Atl Coastal Bays	11,270	3.48E-7 (1.972E-8)	3.26 (0.00886)
	Chesapeake Bay	38,161	3.25E-7 (1.589E-8)	3.28 (0.00757)
	South Atlantic	13,068	3.32E-7 (3.403E-8)	3.29 (0.0161)
Sex	Male	2,643	5.81E-7 (3.301E-8)	3.19 (0.00958)
	Female	4,049	6.81E-7 (4.003E-8)	3.16 (0.00912)

Table 16. Parameter estimates (standard errors in parentheses) for the linear regression of length (mm) on age (years) fit to available data for American eel by region, sex, and all data pooled. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	***Intercept***	***Slope***
None	All	17,414	338 (1.55)	8.77 (0.224)
Region	Gulf of Maine	2,356	87.5 (2.96)	23.5 (0.271)
	Southern New England	475	192 (18.7)	14.5 (1.57)
	Hudson River	875	238 (7.68)	13.7 (0.556)
	Del Bay/Mid-Atl Coastal Bays	4,815	278 (3.61)	29.4 (0.847)
	Chesapeake Bay	7,734	263 (2.85)	28.1 (0.556)
	South Atlantic	1,159	331 (9.47)	26.0 (1.92)
Sex	Male	2,423	295 (1.50)	3.39 (0.172)
	Female	3,513	358 (2.86)	7.65 (0.27)

Table 17. Parameter estimates (standard errors in parentheses) of the von Bertalanffy age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_∞ represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	Subset	n	L_∞	K	T_0
None	All	17,414	434 (1.78)	0.515 (0.018)	-0.34 (0.080)
Region	Gulf of Maine	2,356	1,397 (191.1)	0.022 (0.004)	-2.15 (0.254)
	Southern New England	475	*failed to converge*		
	Hudson River	875	484 (5.36)	0.230 (0.013)	0.35 (0.139*)
	Del Bay/Mid-Atl Coastal Bays	4,815	585 (26.98)	0.179 (0.027)	-2.52 (0.421)
	Chesapeake Bay	7,734	1366 (380.1)	0.030 (0.012*)	-6.84 (0.803)
	South Atlantic	1,159	569.9 (26.31)	0.263 (0.056)	-1.67 (0.623*)
Sex	Male	2,423	*failed to converge*		
	Female	3,513	668 (85.70)	0.035 (0.013*)	-20.96 (4.645)

Table 18. Result of power analysis for linear and exponential trends in American eel abundance indices over a ten-year period. Power was calculated according to methods in Gerrodette (1987).

Region	Life Stage	Survey	State	Median CV	Linear trend		Exponential Trend	
					50%	-50%	50%	-50%
Gulf of Maine	YOY	YOY Survey--Jones River	MA	0.347	0.33	0.46	0.34	0.48
	YOY	YOY Survey--Lamprey River	NH	0.316	0.37	0.52	0.38	0.54
	YOY	YOY Survey - West Harbor Pond	ME	33.245	0.05	0.05	0.07	0.08
Southern New England	Elver & Yellow	CTDEP Electrofishing	CT	0.043	1	1	1	1
	Yellow	NY Western Long Island Survey	NY	1.061	0.1	0.13	0.12	0.16
	YOY	YOY Survey - Carman's River	NY	0.19	0.7	0.87	0.7	0.88
	YOY	YOY Survey - Gilbert Stuart Dam	RI	0.205	0.64	0.83	0.65	0.84
	YOY	Hamilton Fish Ladder	RI	0.205	0.64	0.83	0.65	0.84
	YOY	Ingham Hill	CT	0.455	0.23	0.32	0.24	0.35
Hudson	Elver & Yellow	NYDEC Alosine Beach Seine	NY	0.176	0.76	0.91	0.76	0.92
	Elver & Yellow	NYDEC Striped Bass Beach Seine	NY	0.231	0.56	0.74	0.56	0.76
	Yearling +	HRE Monitoring Program	NY	0.067	1	1	1	1
	YOY	HRE Monitoring Program	NY	0.111	0.98	1	0.98	1
Delaware Bay/Mid-Atlantic Coastal Bays	Elver	Area 6 Electrofishing	PA	0.182	0.73	0.9	0.74	0.9
	Elver & Yellow	Delaware Trawl Survey	DE	0.222	0.58	0.77	0.59	0.78
	Elver & Yellow	PSEG Trawl Survey	DE	0.265	0.47	0.66	0.46	0.64
	Yellow	NJ Striped Bass Seine Survey	NJ	0.501	0.21	0.28	0.22	0.31
	YOY	Little Egg Inlet Ichthyoplankton Survey	NJ	0.18	0.74	0.9	0.74	0.91
	YOY	YOY Survey--Millsboro Dam	DE	0.295	0.4	0.56	0.41	0.58
	YOY	YOY Survey--Patcong Creek	NJ	1.391	0.09	0.1	0.1	0.14
	YOY	YOY Survey--Turville Creek	MD	5.5	0.06	0.06	0.08	0.09

Table 18. *Continued.*

Region	Life Stage	Survey	State	Median CV	Linear trend		Exponential Trend	
					+50%	-50%	+50%	-50%
Chesapeake Bay	Elver & Yellow	North Anna Electrofishing Survey	VA	0.238	0.54	0.72	0.54	0.74
	Yellow	MD Striped Bass Seine Survey	MD	0.621	0.16	0.22	0.18	0.25
	Yellow	VIMS Juvenile SB Seine Survey--long	VA	0.698	0.15	0.19	0.16	0.22
	Yellow	VIMS Juvenile SB Seine Survey--short	VA	0.472	0.22	0.30	0.23	0.33
	YOY	YOY Survey--Brackens Pond	VA	0.638	0.16	0.21	0.17	0.24
	YOY	YOY Survey—Clark’s Millpond	PRFC	0.004	1.00	1.00	1.00	1.00
	YOY	YOY Survey—Gardy’s Millpond	PRFC	0.005	1.00	1.00	1.00	1.00
	YOY	YOY Survey—Kamp’s Millpond	VA	0.052	1.00	1.00	1.00	1.00
	YOY	YOY Survey--Wormley Creek	VA	0.250	0.50	0.69	0.51	0.70
	YOY	Wareham’s Pond	VA	0.246	0.51	0.70	0.52	0.71
South Atlantic	Elver & Yellow	NCDMF Estuarine Trawl Survey	NC	0.507	0.20	0.28	0.22	0.31
	Elver & Yellow	SC Electrofishing Survey	SC	0.131	0.93	0.99	0.93	0.99
	YOY	YOY Beaufort Bridgenet Ichthyo.	NC	0.216	0.60	0.79	0.61	0.80
	YOY	YOY Survey - Altamaha Canal	GA	0.320	0.36	0.50	0.37	0.53
	YOY	YOY Survey--Goose Creek	SC	0.205	0.64	0.83	0.65	0.84
	YOY	YOY Survey--Guana River Dam	FL	0.013	1.00	1.00	1.00	1.00

Table 19. Results of the Mann-Kendall trend analysis applied to YOY indices. S is the Mann-Kendall statistic, D is the Denominator, *P*-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected (*P*-value < α; α = 0.05). NS = not significant. "-" indicates an index which was not available during the last benchmark but was included in the 2017 update because it now has at least 10 years of data.

Region	State	Location	Gear	Time Period	n	*T*	*D*	*S*	*P*-value	Trend 2012	Trend 2016
Gulf of Maine	ME	West Harbor Pond	Irish Elver Ramp	2001–2016	16	0.283	120	33.96	0.137	NS	NS
	NH	Lamprey River	Irish Elver Trap	2001–2016	16	0.350	120	42.00	0.065	NS	NS
	MA	Jones River	Sheldon Elver Trap	2001–2016	16	-0.533	120	-63.96	0.005	NS	↓
Southern New England	RI	Hamilton Fish Ladder	Irish Elver Ramp	2004-2016	13	0.282	78	22.00	0.200	-	NS
	RI	Gilbert Stuart Dam	Irish Elver Ramp	2000–2016	17	0.162	136	22.03	0.387	NS	NS
	CT	Ingham Hill	Irish Elver Ramp	2007-2016	10	-0.244	45	-10.98	0.371	-	NS
	NY	Carman's River	Fyke Net	2000–2016	17	0.044	136	6.00	0.840	NS	NS
	NY	HRE Monitoring	Epibenthic sled & tucker trawl	1974-2013	34	-0.422	561	-236.74	0.000	↓	↓
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Little Egg	Plankton Net	1992-2015	24	-0.355	276	-97.98	0.016	NS	↓
	NJ	Patcong Creek	Fyke Net	2004–2016	12	0.217	120	26.04	0.260	NS	NS
	DE	Millsboro Dam	Fyke Net	2000–2016	17	0.191	136	25.98	0.303	NS	NS
	MD	Turville Creek	Irish Elver Ramp	2000–2016	17	0.176	136	23.94	0.343	NS	NS

Table 19. ***Continued.***

Region	State	Location	Gear	Time Period	n	*T*	*D*	*S*	*P*-value	Trend 2012	Trend 2016
Chesapeake Bay	PRFC	Clark's Millpond	Irish Elver Ramp	2000–2016	17	-0.147	136	-19.99	0.434	NS	NS
	PRFC	Gardy's Millpond	Irish Elver Ramp	2000–2016	17	-0.191	136	-25.98	0.303	NS	NS
	VA	Warehams Pond	Irish Elver Ramp	2003-2016	13	0.308	78	24.02	0.161	-	NS
	VA	Bracken's Pond	Irish Elver Ramp	2000–2016	17	-0.324	136	-44.06	0.077	NS	NS
	VA	Kamp's Millpond	Irish Elver Ramp	2000–2016	17	-0.044	136	-6.00	0.837	NS	NS
	VA	Wormley Creek	Irish Elver Ramp	2001–2016	17	-0.100	120	-12.00	0.620	NS	NS
South Atlantic	NC	Beaufort Bridgenet Ichthyo	Plankton Net	1987-2007	21	-0.343	210	-72.03	0.032	NS	↓
	SC	Goose Creek	Fyke Net	2000–2015	16	-0.433	120	-51.96	0.022	NS	↓
	GA	Altamaha Canal	Fyke Net	2001–2010	10	-0.333	45	-14.99	0.211	NS	NS
	FL	Guana River Dam	Dip Net	2001–2016	16	-0.343	210	-72.03	0.032	NS	↓

Table 20. Results of the Mann-Kendall trend analysis applied to yellow eel indices. S is the Mann-Kendall statistic, D is the Denominator, *P*-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected (*P*-value < α; α = 0.05). NS = not significant. The length range of observed American eels is shown in parentheses after the life stage if the information was available.

Region	Survey	Gear	Life Stage	Time Period	n *	*T*	*D*	*S*	*P*-value	Trend 2012	Trend 2017
Southern New England	CTDEP Electrofishing Survey	Electrofishing	Elver & Yellow (50–590 mm)	2001–2014	11	0.273	66	18.018	0.244	↑	NS
	NY Western Long Island Survey	Seine	Yellow (35–770 mm)	1984–2016	32	-0.49	499.744	-244.87	0.000	↓	↓
Hudson River	HRE Monitoring Program	Epibenthic Sled and Tucker Trawl	Yearling and Older	1974–2013	39	-0.526	780	-410.28	0.000	↓	↓
	NYDEC Alosine Beach Seine	Seine	Elver & Yellow	1980–2016	36	-0.42	666	-410.28	0.000	↓	↓
	NYDEC Striped Bass Beach Seine	Seine	Elver & Yellow	1980–2016	36	-0.523	666	-279.72	0.000	↓	↓
Delaware Bay/ Mid-Atlantic Coastal Bays	NJDFW Striped Bass Seine Survey	Seine	Yellow (50–750 mm)	1980–2016	36	-0.0631	666	-42.025	0.592	NS	NS
	Delaware Trawl Survey	Trawl	Elver & Yellow (55–690 mm)	1982–2016	34	-0.153	595	-91.035	0.201	NS	NS
	PSEG Trawl Survey	Trawl	Elver & Yellow (97–602 mm)	1998–2016	18	0.158	171	27.018	0.363	↑	NS[1]
	Area 6 Electrofishing	Electrofishing	Elver	1999–2016	17	0.216	153	33.048	0.225	NS	NS
	MDDNR Striped Bass Seine Survey	Seine	Yellow (77–687 mm)	1966–2016	50	-0.111	1274.5	-141.47	0.252	NS	NS

Table 20. ***Continued.***

Region	Survey	Gear	Life Stage	Time Period	n *	*T*	*D*	*S*	*P*-value	Trend 2012	Trend 2017
Chesapeake Bay	North Anna Electrofishing Survey	Electrofishing	Elver & Yellow (32–726 mm)	1990–2009	19	0.626	171	107.046	0.000	↑	↑[1]
	VIMS Juvenile Striped Bass Seine Survey—long	Seine	Yellow	1989–2016	49	0.00753	929.354	6.99803	0.951	NS	NS
	VIMS Juvenile Striped Bass Seine Survey—short	Seine	Yellow	1967–2016	27	-0.135	377.499	-50.962	0.323	↓	NS
South Atlantic	NCDMF Estuarine Trawl Survey	Trawl	Elver & Yellow (26–921 mm)	1989–2016	27	-0.296	378	-111.89	0.028	↓	↓
	SC Electrofishing Survey	Electrofishing	Elver & Yellow (44–890 mm)	2001–2016	15	-0.367	120	-44.04	0.053	↓	NS

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[1] The timeframe for the PSEG trawl survey changed from 1970-2010 in ASFMC 2012 to 1998-2016 in this update report. The North Anna Electrofishing survey was not updated for this report with data from 2010-2016 and therefore the trend remains the same. Refer to Section 5.2.2. for information on survey and standardization changes.

Table 21. Results of the Mann-Kendall trend analysis applied to regional and coastwide indices of American eel abundance. S is the Mann-Kendall statistic, D is the Denominator, P-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected (P-value < α; α = 0.05). NS = not significant. "-" are indices that were not updated.

Region	Life Stage	Time Period	n	*T*	*D*	*S*	*P*-value	2012 Trend	2017 Trend
Gulf of Maine	YOY	2001–2016	15	0.017	120	2.004	0.964	NS	NS
Southern New England	YOY	2000–2016	16	0.118	136	16.05	0.537	NS	NS
	Yellow	2001–2010	9			0		NS	-
Hudson River	YOY	1974–2009	35			0		↓	-
	Yellow	1980–2016	36	-0.527	665	-351	0.000	↓	↓
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	2000–2016	16	0.191	136	25.98	0.303	NS	NS
	Yellow	1999–2016	17	0.203	153	31.06	0.256	NS	NS
Chesapeake Bay	YOY	2000–2016	16	0.015	136	1.999	0.967	NS	NS
	Yellow	1990–2009	19	0.621	190	118	0.000	↑	↑
South Atlantic	YOY	2001–2015	14	-0.433	120	-52	0.022	NS	↓
	Yellow	2001–2016	15	-0.4	120	-48	0.034	↓	↓
Atlantic Coast	YOY (short-term)	2000–2016	16	0.118	136	16.05	0.537	NS	NS
	YOY (long-term)	1987–2013	26	-0.237	325	-77	0.094	NS	NS
	Yellow (40+ year)	1974–2016	42	-0.391	903	-353	0.000	NS	↓
	Yellow (30-year)	1987–2016	29	-0.333	435	-145	0.010	↓	↓
	Yellow (20-year)	1997–2016	19	-0.211	190	-40.1	0.206	NS	NS

Table 22. Results of the meta-analysis to synthesize trends for American eel. The meta-analysis techniques are from Manly (2001) where S_1 tests whether at least one of the datasets shows a significant decline through time and S_2 tests whether there is consensus among the datasets for a decline. S_2 incorporates a weight equal to the number of years of the survey, n. The value of p represents the one-tailed p-value from the Mann-Kendall nonparametric test for a decreasing trend through time.

Life Stage	Survey	n	p	Meta-analysis statistics	
Yellow	Area 6 Electrofishing	17	0.887		
	CTDEP Electrofishing Survey	11	0.878		
	NYDEC Alosine Beach Seine	36	0.000	S_1:	115.88
	NYDEC Striped Bass Beach Seine	36	0.000	df:	30
	Delaware Trawl Survey	34	0.101	$P(X^2>S_1\|df)$:	<0.01
	PSEG Trawl Survey	18	0.819		
	North Anna Electrofishing Survey	19	1.000	S_2:	-5.05
	NCDMF Estuarine Trawl Survey	27	0.142	$P(Z>S_2)$:	<0.01
	SC Electrofishing Survey	16	0.026		
	HRE Monitoring	39	0.000		
	NY Western Long Island Survey	32	0.000		
	NJDFW Striped Bass Seine Survey	36	0.296		
	MD Striped Bass Seine Survey	50	0.126		
	VIMS Juvenile Striped Bass Seine --short	19	0.476		
	VIMS Juvenile Striped Bass Seine--long	49	0.838		
YOY	West Harbor Pond	16	0.932		
	Lamprey River	16	0.968		
	Jones River	13	0.003	S_1:	95.22
	Hamilton Fish Ladder	13	0.900	df:	42
	Gilbert Stuart Dam	17	0.807	$P(X^2>S_1\|df)$:	<0.01
	Ingham Hill	10	0.186		
	Carman's River	17	0.580	S_2:	-16.03
	HRE Monitoring	34	0.000	$P(Z>S_2)$:	<0.01
	Little Egg Inlet Ichthyoplankton Survey	24	0.008		
	Patcong Creek	12	0.870		
	Millsboro Dam	17	0.849		
	Turville Creek	17	0.829		
	Clarks Millpond	17	0.217		
	Gardys Millpond	17	0.152		
	Brackens Pond	17	0.039		
	Kamps Millpond	17	0.419		
	Wormley Creek	17	0.310		
	Beaufort Bridgenet Ichthyoplankton	21	0.016		
	Goose Creek	16	0.011		
	Altamaha Canal	10	0.106		
	Guana River Dam	16	0.016		

Table 23. Summary statistics from ARIMA model fits to American eel surveys with 20 or more years of data. $Q_{0.25}$ is the 25th percentile of the fitted values; P(<0.25) is the probability of the of the survey being below $Q_{0.25}$ in 2010 or in the terminal year with 80% confidence; r1–r3 are the first three autocorrelations; θ is the moving average parameter; SE is the standard error of θ; and σ^2c is the variance of the index. P(<0.25) in 2010 is included for comparison purposes of the status of the survey from the 2012 benchmark assessment.

Region	Survey	Life Stage	Years	$Q_{0.25}$	P(<0.25) in 2010	P(<0.25) in terminal year	n	r1	r2	r3	θ	SE	σ^2_c
Hudson River	NY Western Long Island Survey	Yellow	1984 - 2016	-4.27	0.462	0.412	33	-0.26	-0.08	-0.06	0.41	0.15	0.65
	HRE Monitoring Program	YOY	1974 - 2013	-2.23	0.516	0.544	34	-0.06	-0.11	-0.29	0.78	0.14	0.28
	HRE Monitoring Program	Yearling and Older	1974 - 2013	-1.62	0.034	0.003	40	-0.14	-0.28	0.39	0.32	0.14	0.26
	NYDEC Alosine Beach Seine	Elver & Yellow	1980 - 2016	-1.33	0.344	0.72	37	-0.38	0.01	-0.06	0.66	0.13	0.25
	NYDEC Striped Bass Beach Seine	Elver & Yellow	1980 - 2016	-1.37	0.286	0.446	37	-0.08	-0.19	-0.1	0.72	0.11	0.33

Table 23. ***Continued.***

Region	Survey	Life Stage	Years	$Q_{0.25}$	P(<0.25) in 2010	P(<0.25) in terminal year	n	r1	r2	r3	θ	SE	σ^2_c
Delaware Bay/Mid-Atlantic Coastal Bays	Little Egg Inlet Ichthyoplankton Survey	YOY	1992 - 2015	-0.01	0.722	0.755	24	0.03	-0.51	-0.12	0.25	0.32	0.17
	NJDFW Striped Bass Seine Survey	Yellow	1980 - 2016	-2.75	0	0	37	-0.24	-0.33	0.05	1	0.1	0.59
	Delaware Trawl Survey	Elver & Yellow	1982 - 2016	-1.98	0.479	0.242	35	-0.54	0.43	-0.28	0.54	0.14	0.41
	PSEG Trawl Survey	Elver & Yellow	1998 - 2016	-0.12	0.002	0	19	-0.85	0.7	-0.62	1	0.19	0.28
Chesapeake Bay	MD Striped Bass Seine Survey	Yellow	1966 - 2016	-2.24	0.155	0.202	51	-0.29	0.01	-0.07	0.58	0.17	1
	VIMS Juvenile SB Seine Survey - short	Yellow	1989 - 2016	-2.37	0.085	0.066	28	-0.69	0.23	0.01	1	0.13	0.33
	VIMS Juvenile SB Seine Survey - long	Yellow	1967 - 2016	-3.2	0.006	0.009	44	-0.35	-0.34	0.21	0.63	0.12	0.88
South Atlantic	Beaufort Bridgenet Ichthyoplankton	YOY	1987 - 2007	-1.12		0.454	21	-0.43	-0.12	0.1	0.74	0.17	0.52
	NCDMF Estuarine Trawl Survey	Elver & Yellow	1989 - 2016	-2.09	0.192	0.284	28	-0.28	-0.31	0.18	0.85	0.11	0.64

12 FIGURES

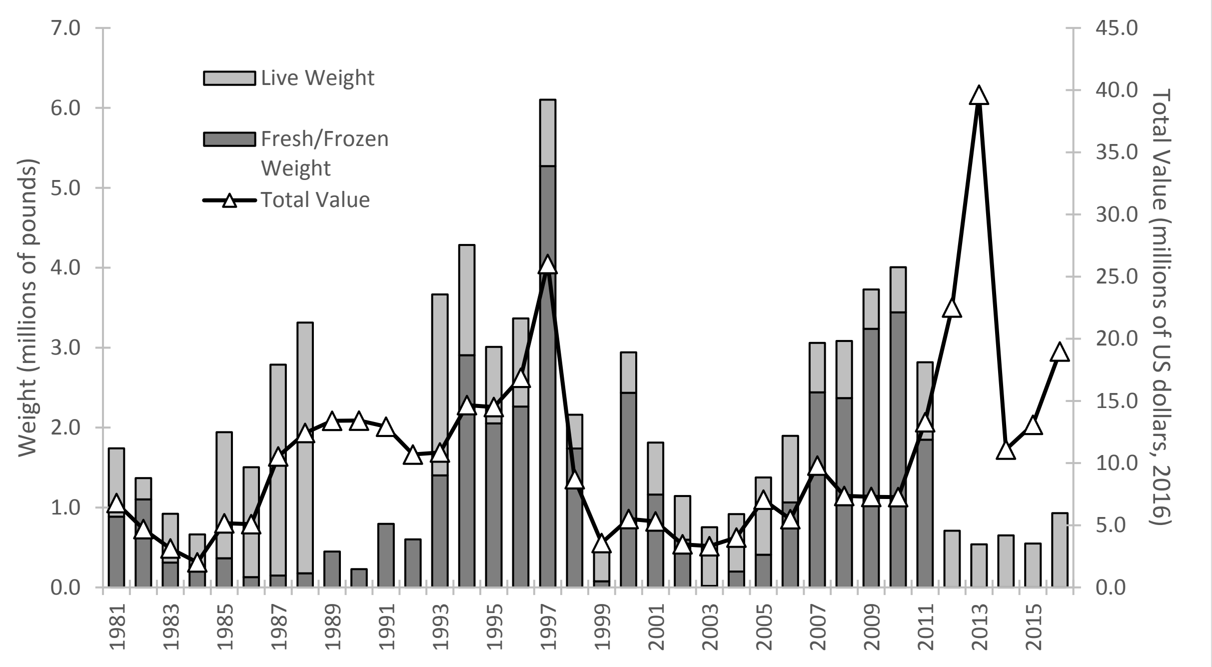

Figure 1. Annual U.S. domestic exports of American eels from districts along the Atlantic coast, 1981–2016. Note that the weights of live exports were not available for 1989 to 1992 and there were no fresh/frozen weight after 2011.

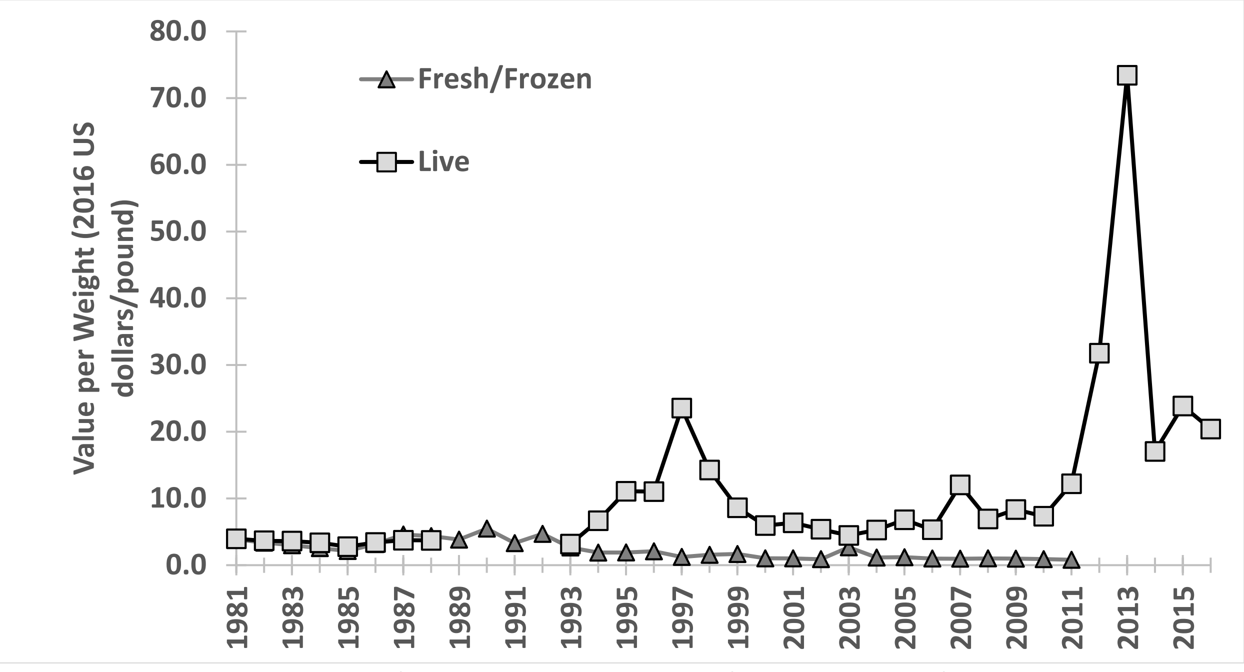

Figure 2. Value per weight of U.S. domestic exports of American eels from districts along the Atlantic Coast, 1981-2016. Note that there was no data for fresh/frozen after 2011.

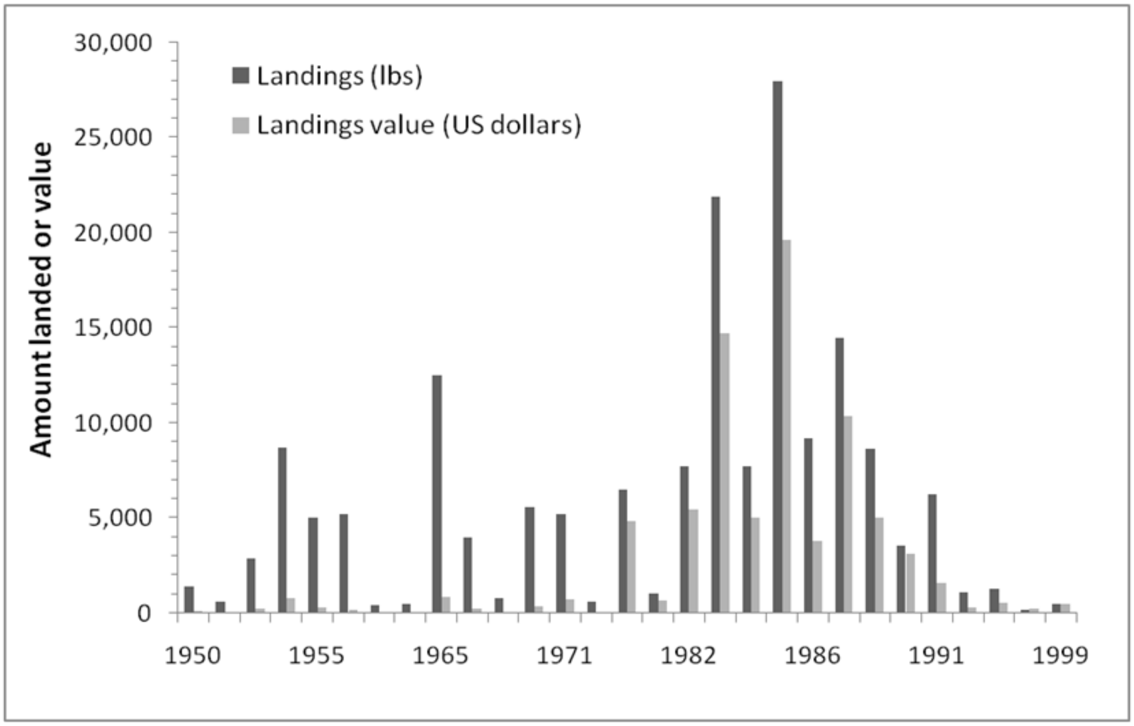

Figure 3. Total weight and value of American eel commercial landings in the Gulf of Mexico, 1950–1999. Recent landings are confidential.

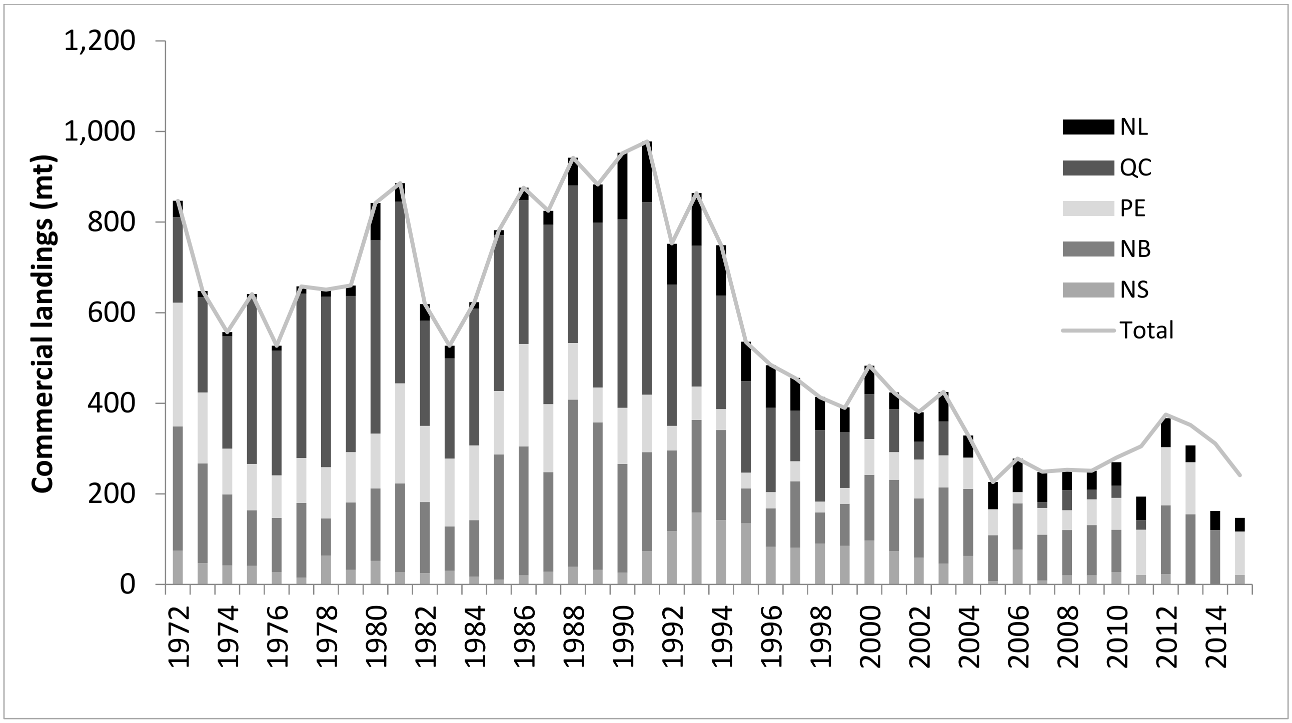

Figure 4. Annual commercial seafisheries landings (live weight) of American eel along Canada's Atlantic Coast summarized by province, 1972–2015. In recent years, some provinces' landings have been confidential so total landings has been provided as a line.

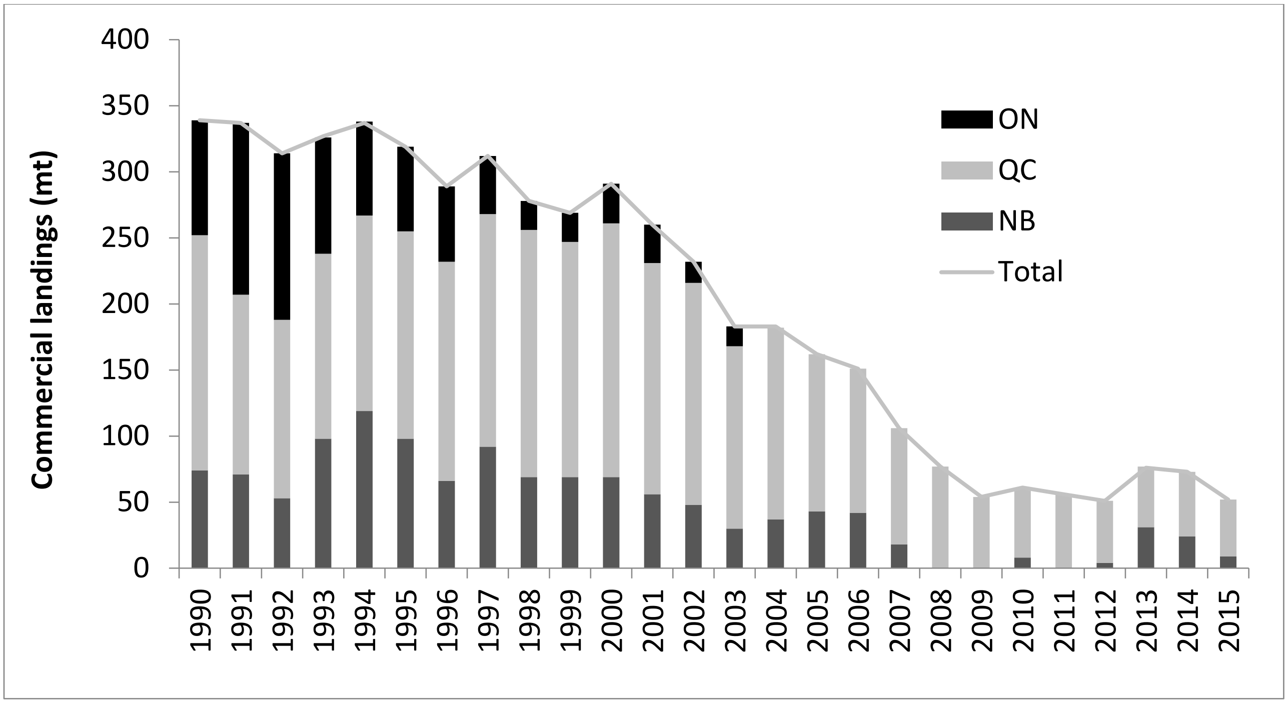

Figure 5. **Annual commercial freshwater landings (live weight) of American eel along Canada's Atlantic Coast summarized by province, 1990–2015.**

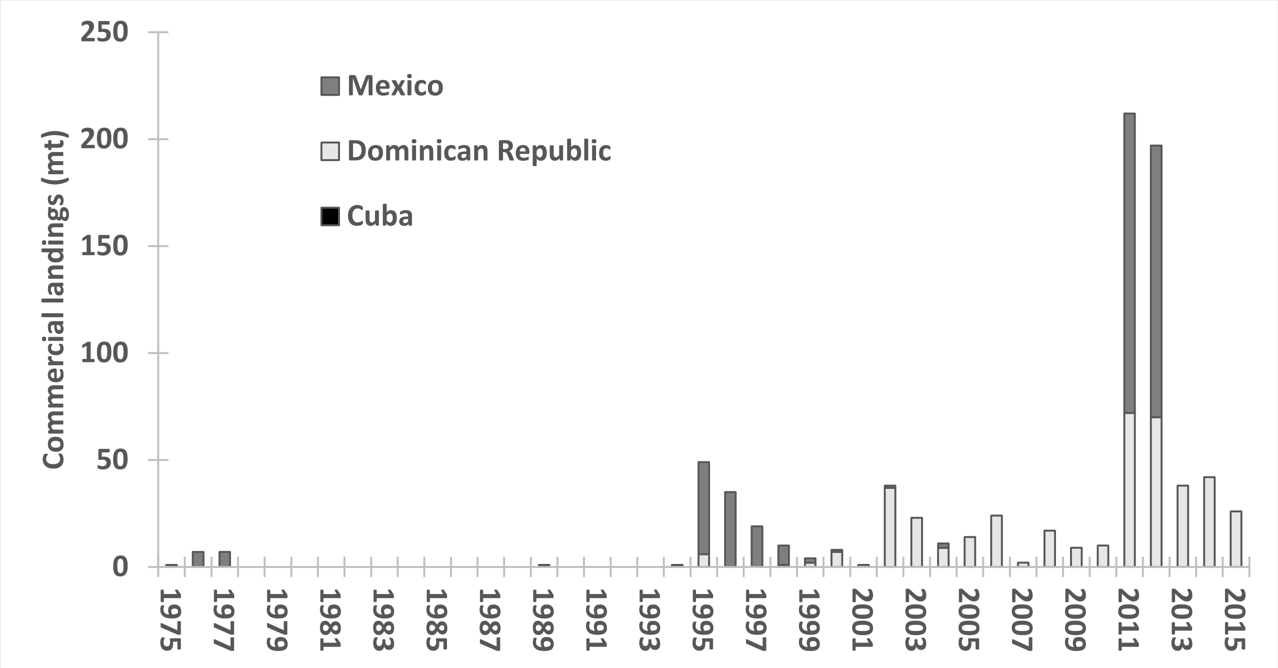

Figure 6. **Annual commercial landings (live weight) of American eel reported by the FAO from Central and South America, 1975–2015. No landings were reported between 1950-1974, 1978-1988, and 1990-1993. Cuba's only reported American eel landings were 1 mt in 1989 and 1 mt in 1994.**

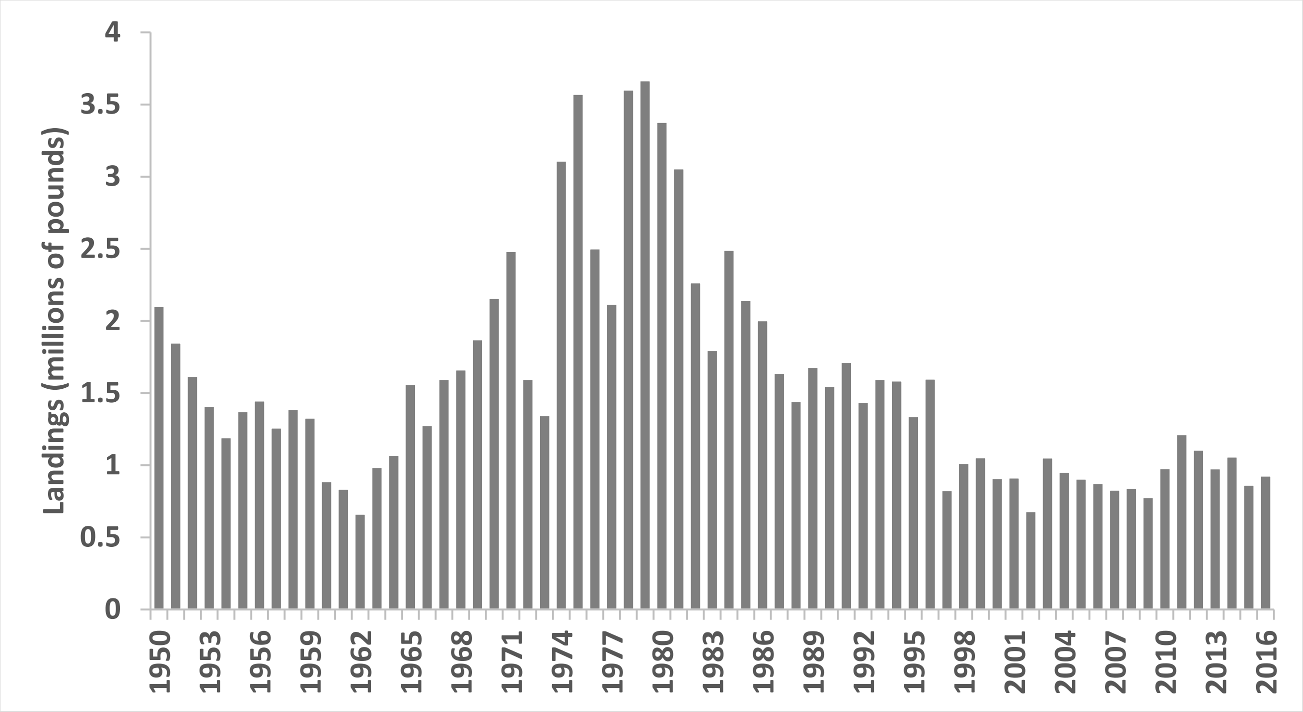

Figure 7. **Total commercial landings of American eel along the U.S. Atlantic Coast, 1950–2016. Landings in 2016 are preliminary.**

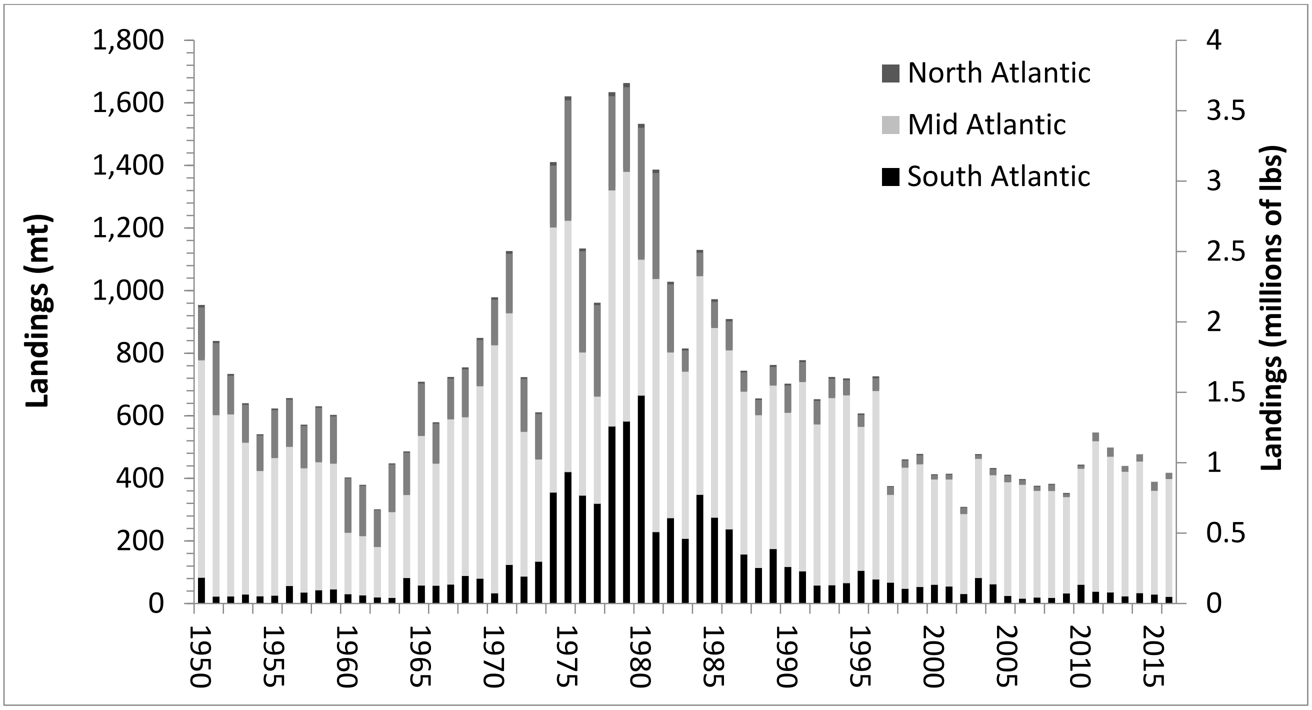

Figure 8. **Total commercial landings of American eel by old geographic region along the U.S. Atlantic Coast, 1950–2016. Landings in 2016 are preliminary.**

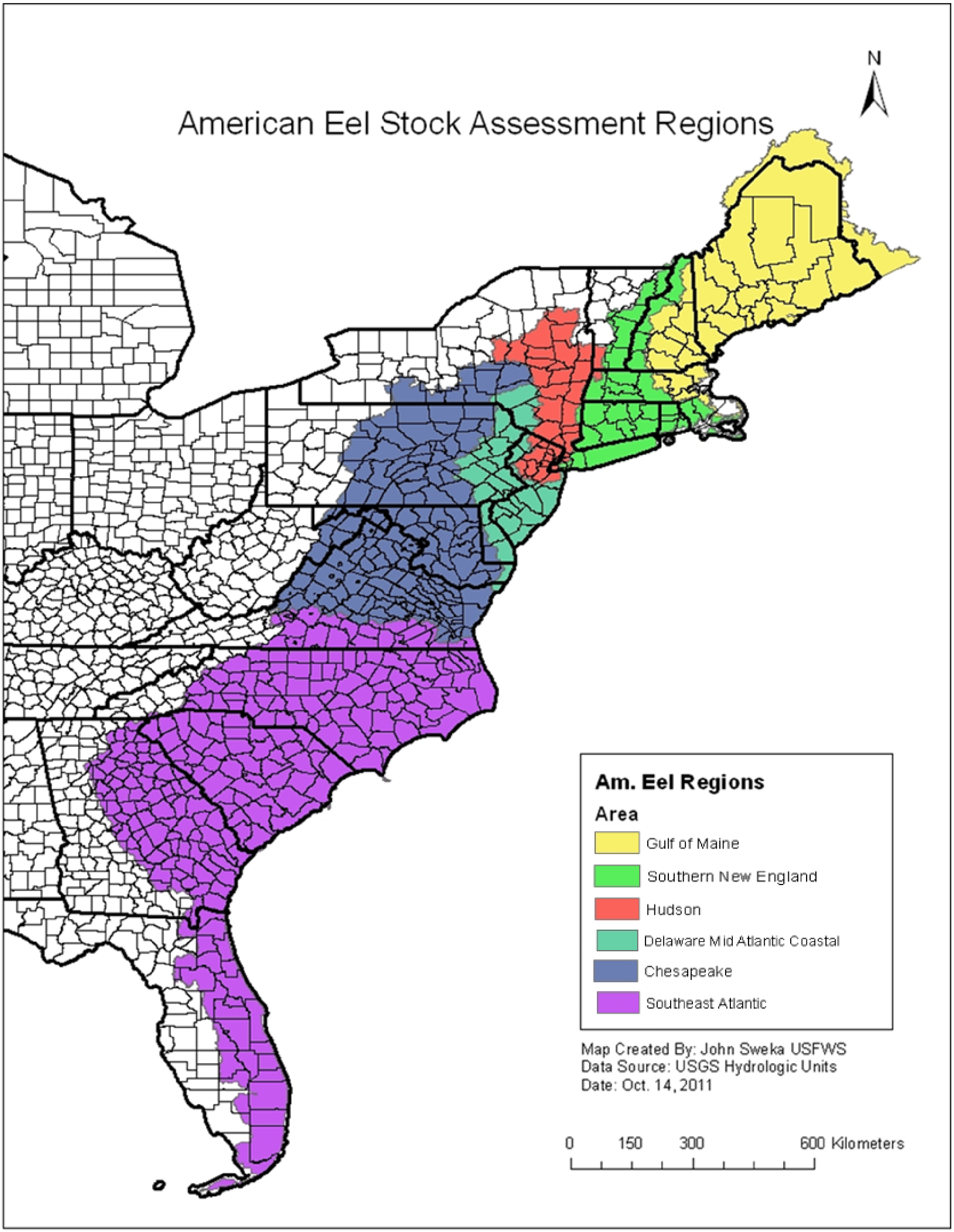

Figure 9. Watershed-based geographic regions used in the 2012 benchmark stock assessment.

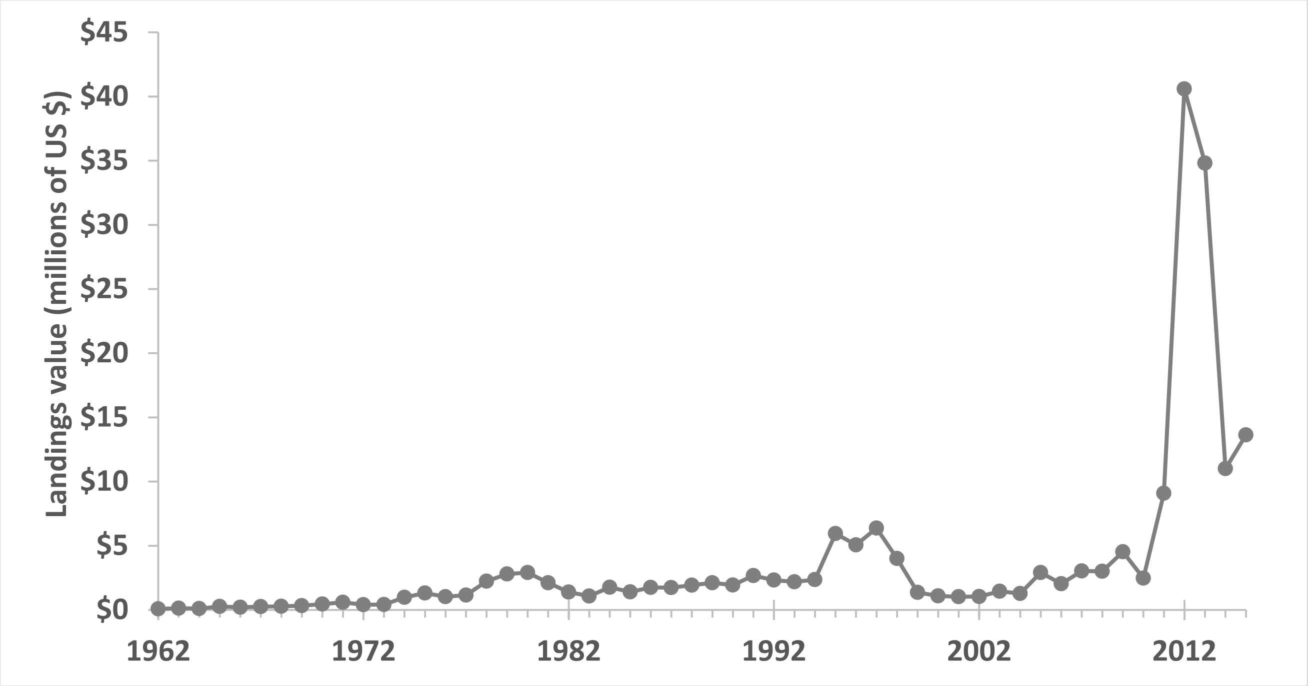

Figure 10. Estimated value of U.S. American eel landings, 1962–2015.

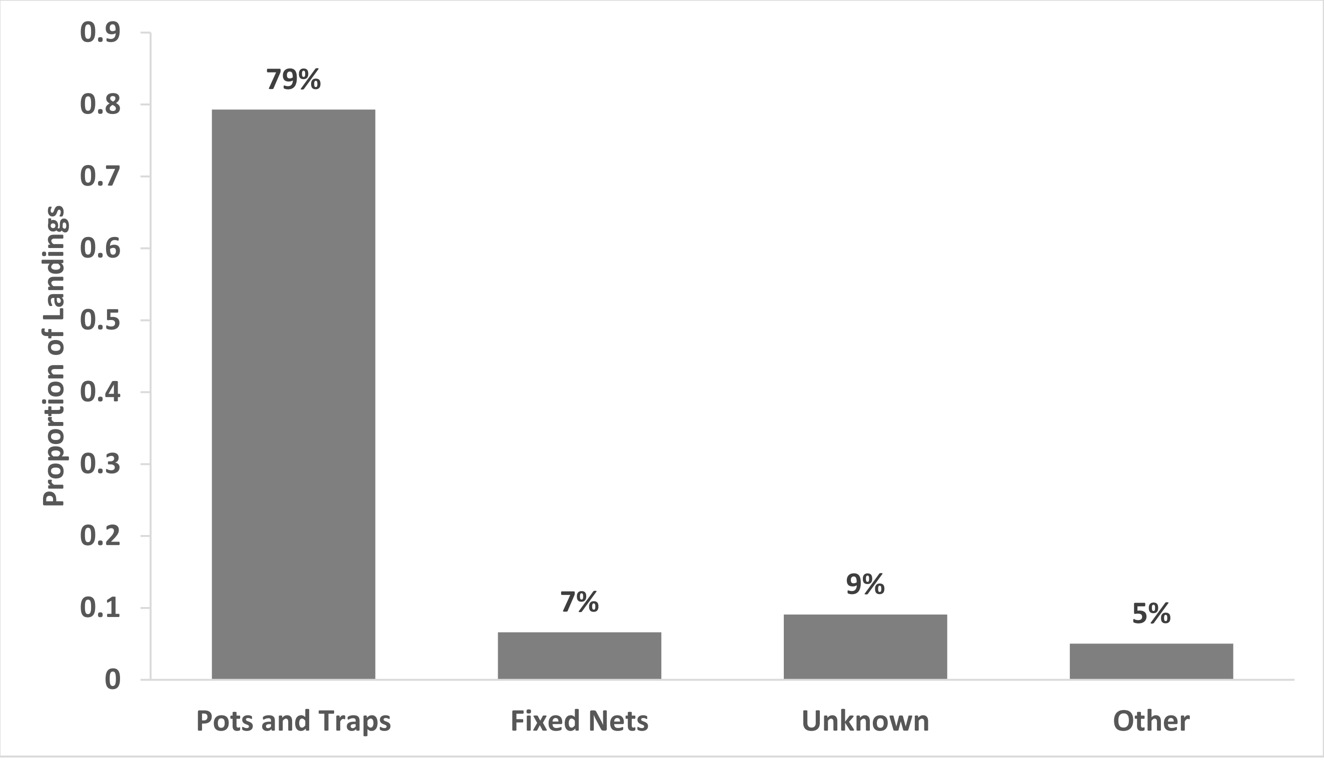

Figure 11. Proportion of Atlantic coast commercial landings by general gear type, 1950–2016.

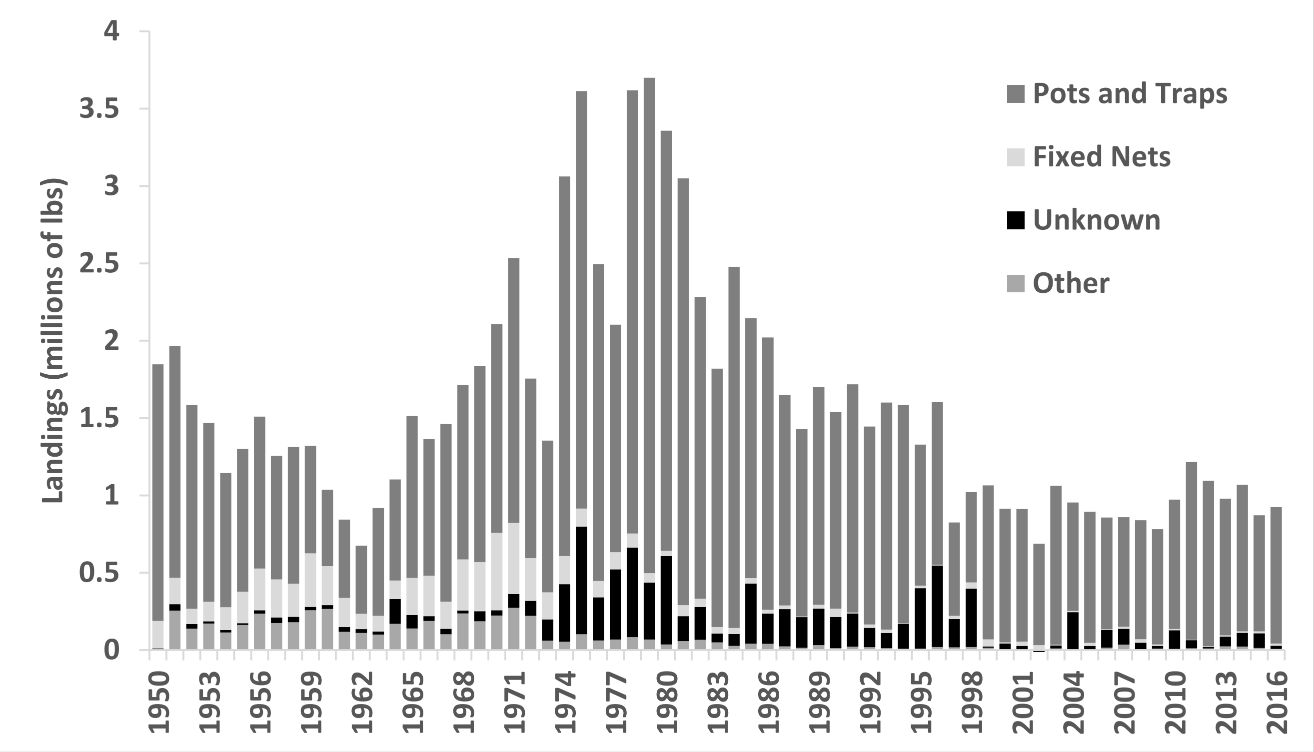

Figure 12. Trends in the proportion of Atlantic coast commercial landings by general gear type, 1950-2016. Landings in 2016 are preliminary.

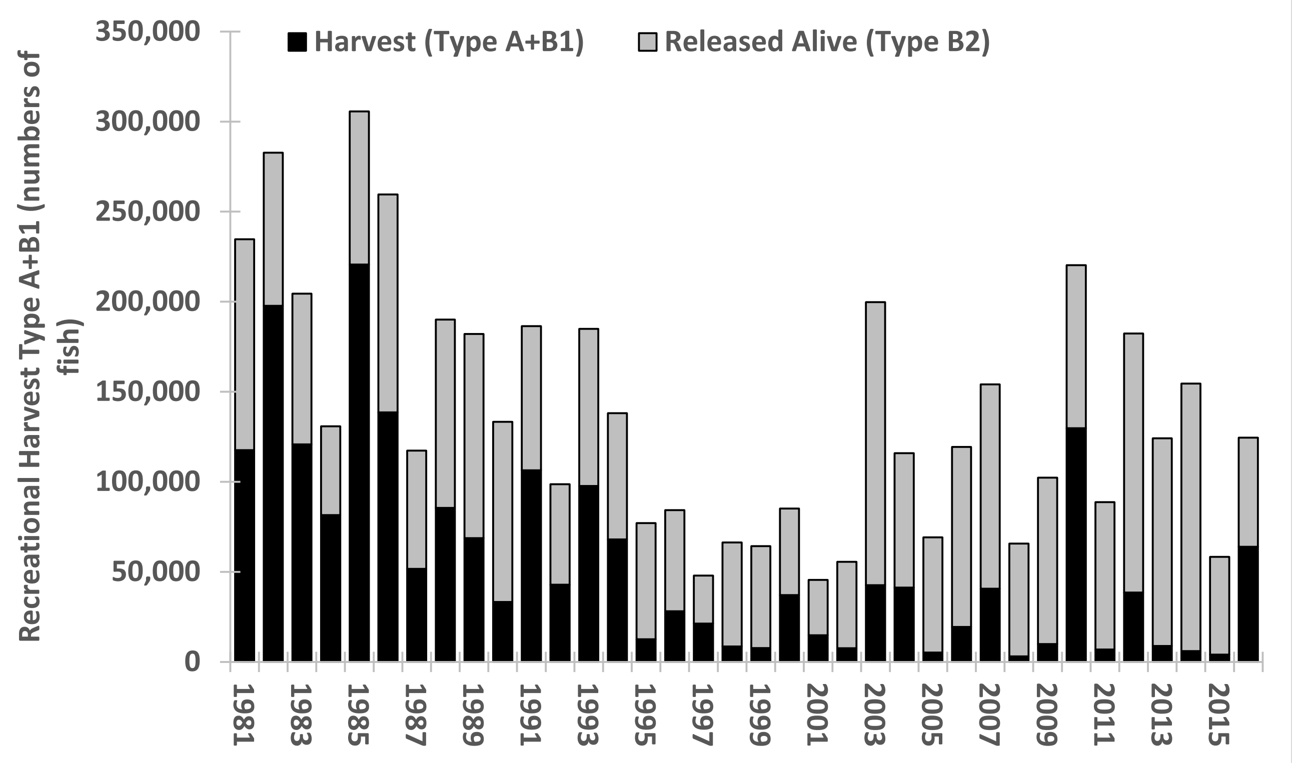

Figure 13. Recreational harvest and releases for American eel 1981-2016. Estimates for 1981-2003 have been calibrated to MRIP from MRFSS.

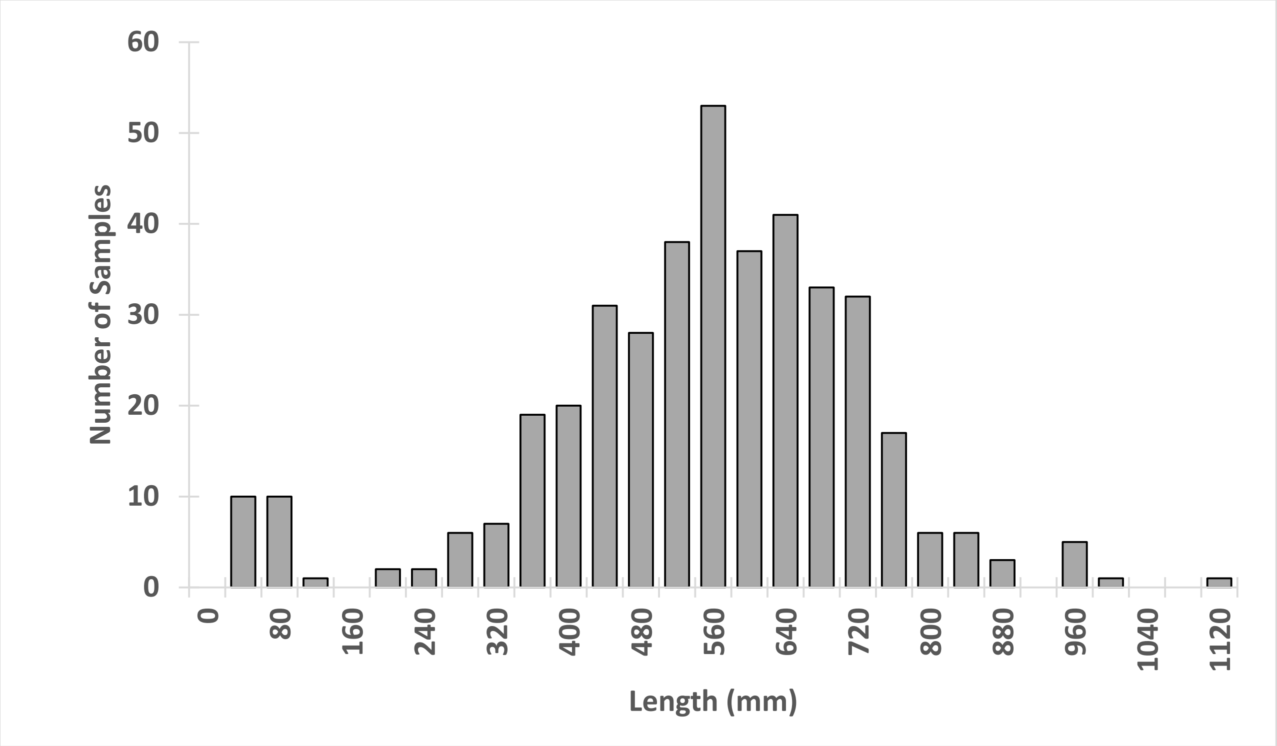

Figure 14. Length-frequency of American eels sampled by the MRFSS angler-intercept survey (Type A catch), 1981–2016. It was noted by the SAS that small lengths may represent a species misidentification.

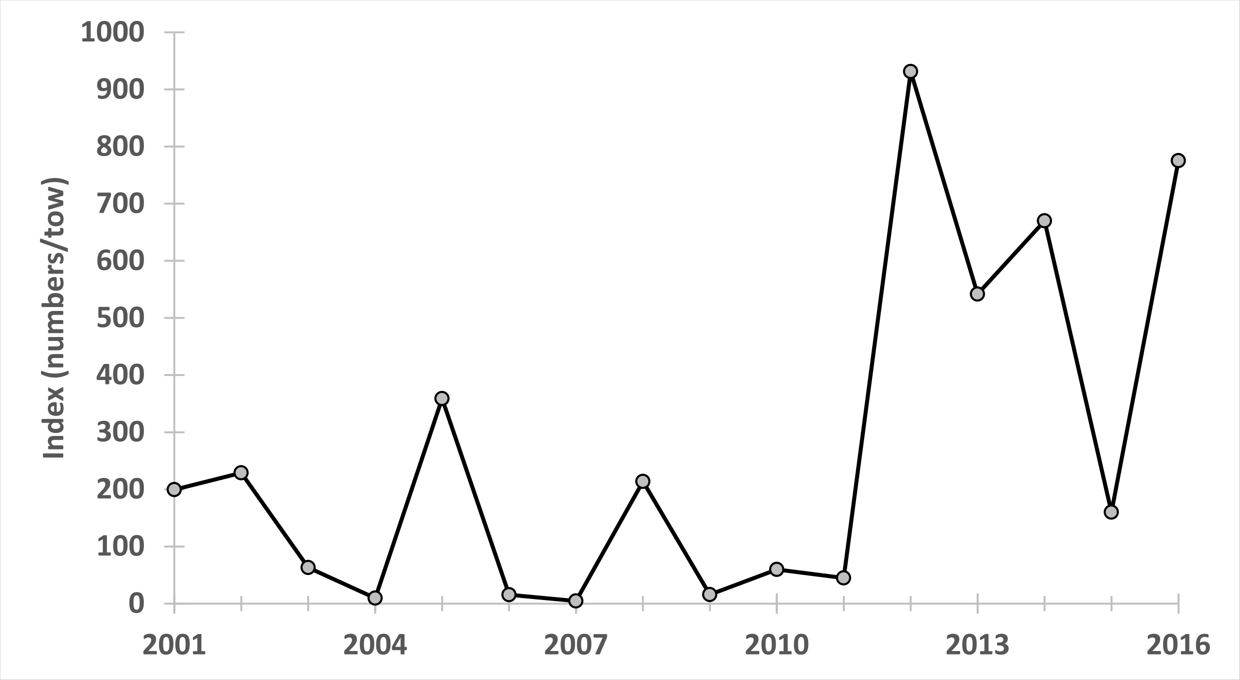

Figure 15. GLM-standardized index of abundance for YOY American eels caught by Maine's annual YOY survey in West Harbor Pond, 2001–2016. The error bars were omitted from the graph because there were several very large values. See text for more discussion on this.

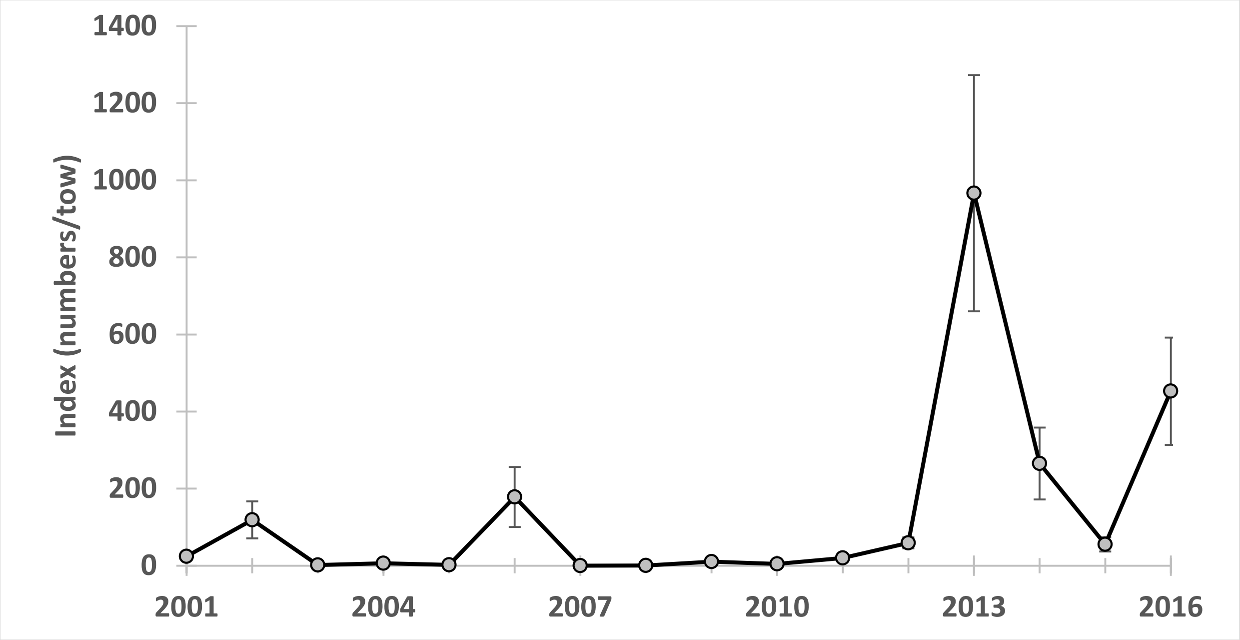

Figure 16. GLM-standardized index of abundance for YOY American eels caught by New Hampshire's annual YOY survey in the Lamprey River, 2001–2016. The error bars represent the standard errors about the estimates.

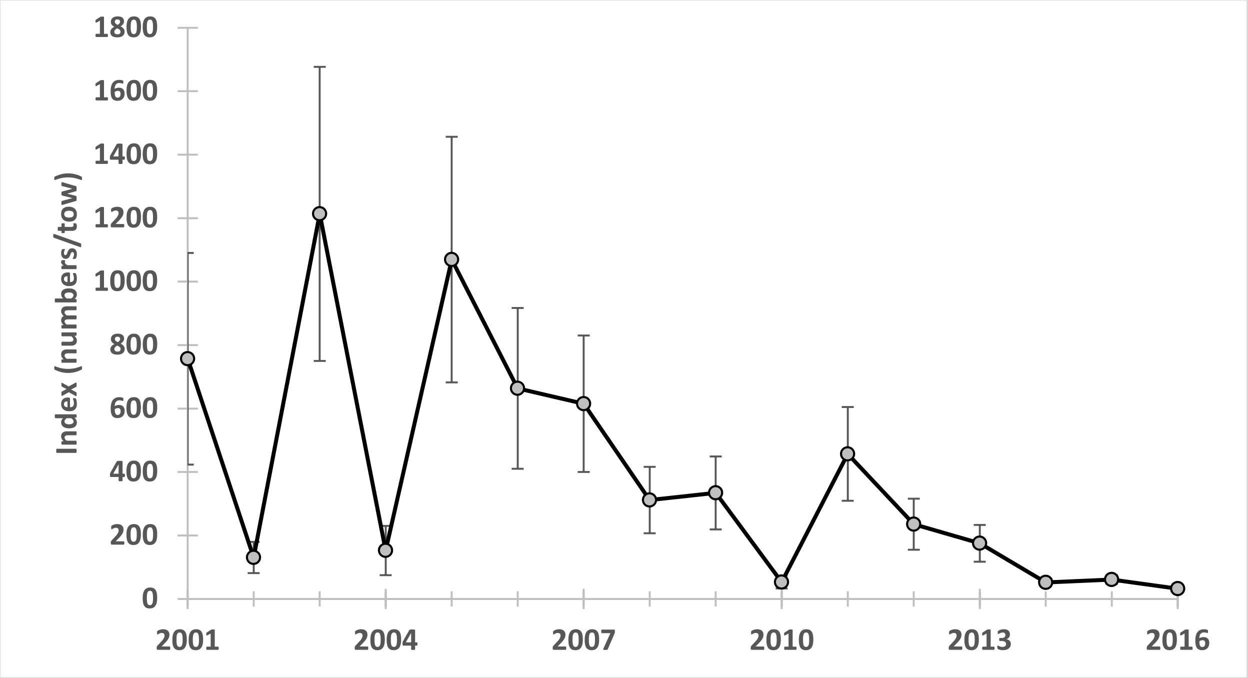

Figure 17. GLM-standardized index of abundance for YOY American eels caught by Massachusetts' annual YOY survey in the Jones River, 2001–2016. The error bars represent the standard errors about the estimates.

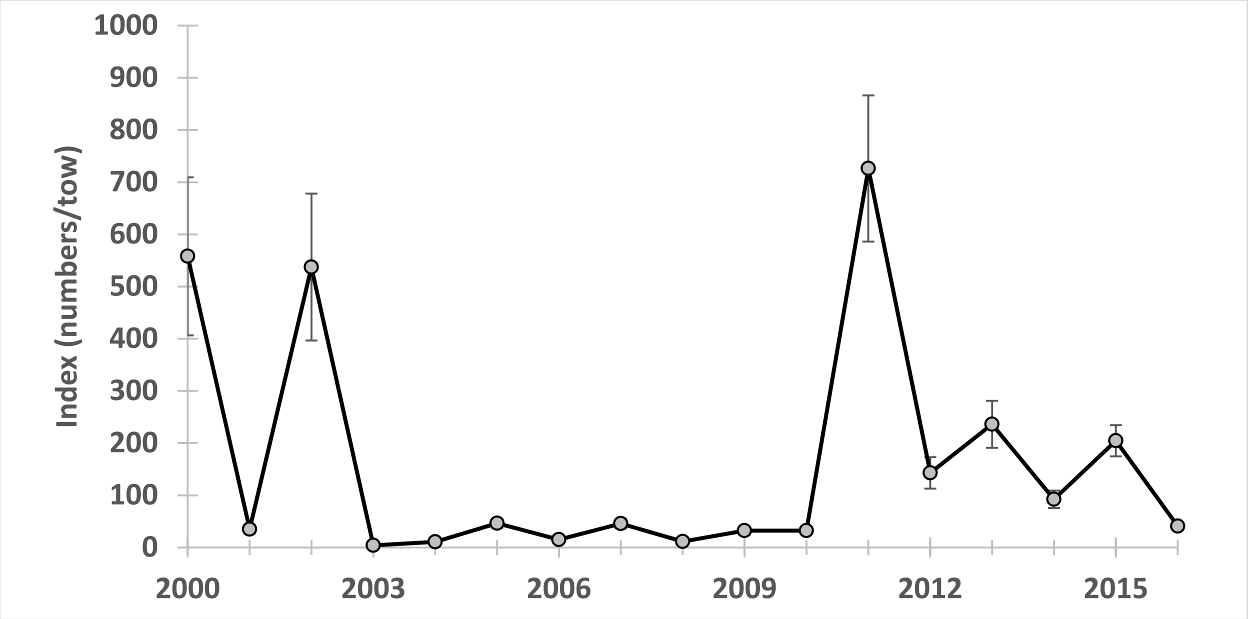

Figure 18. GLM-standardized index of abundance for American eels caught by Rhode Island's annual YOY survey near Gilbert Stuart Dam, 2000–2016. The error bars represent the standard errors about the estimates.

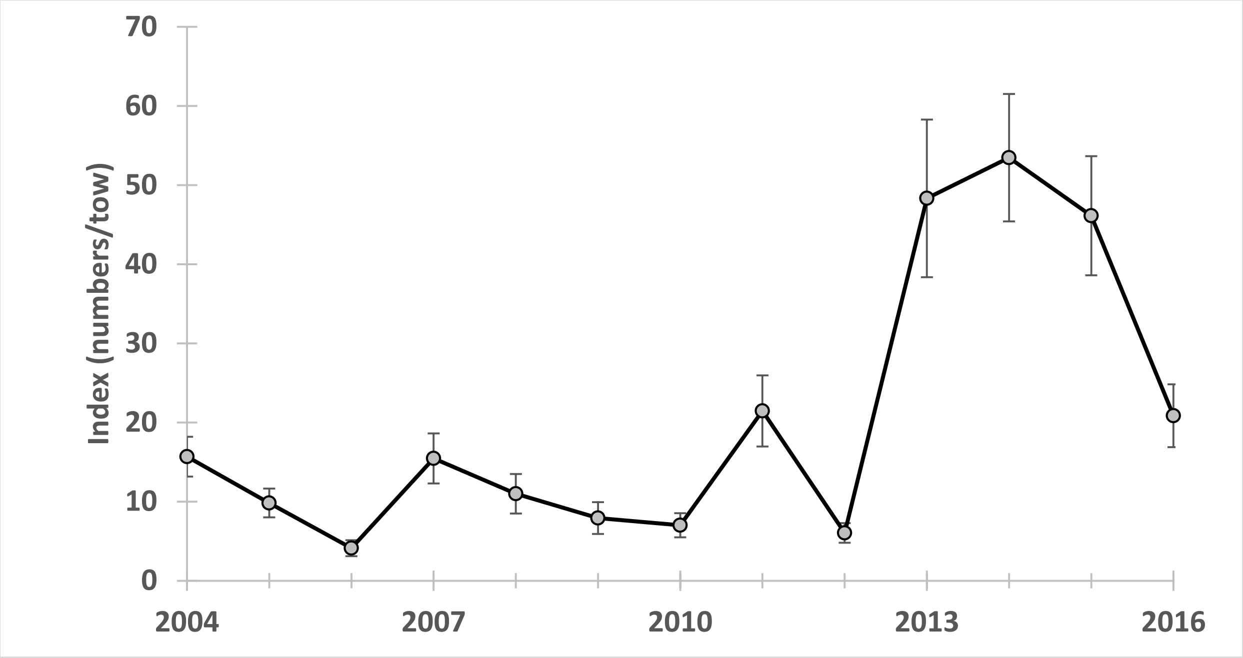

Figure 19. GLM-standardized index of abundance for American eels caught by Rhode Island's annual YOY survey at Hamilton Fish Ladder, 2004–2016. The error bars represent the standard errors about the estimates.

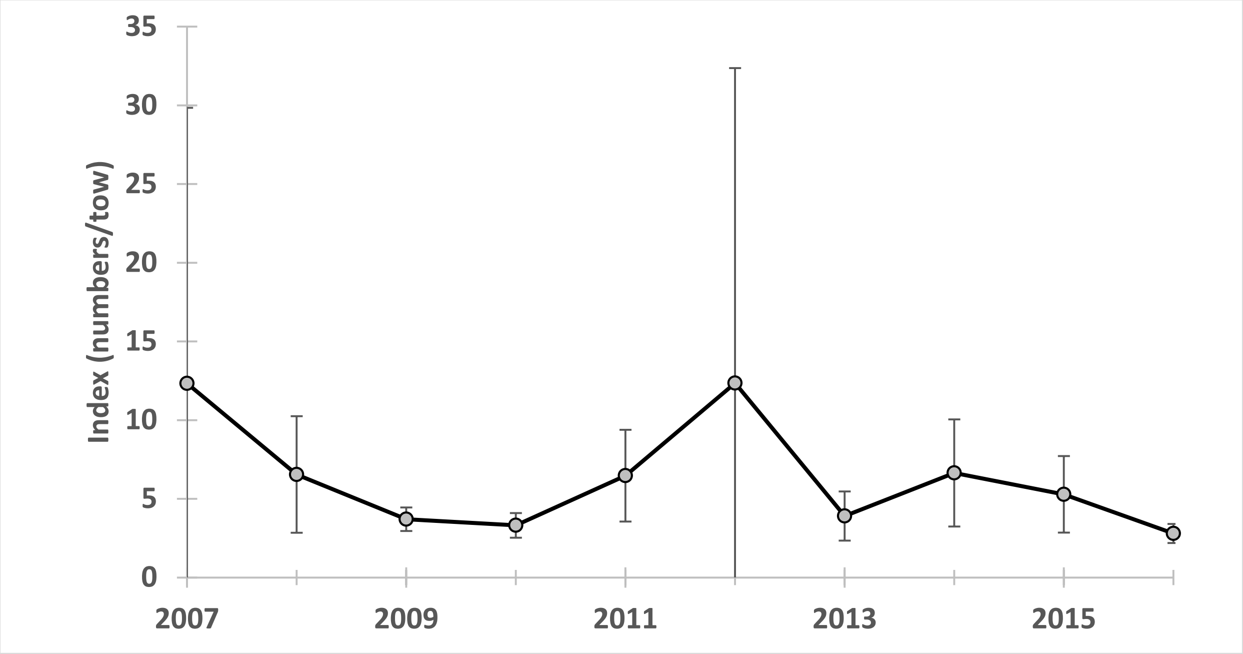

Figure 20. GLM-standardized index of abundance for American eels caught by Connecticut's annual YOY survey at Ingham Hill, 2007–2016. The error bars represent the standard errors about the estimates.

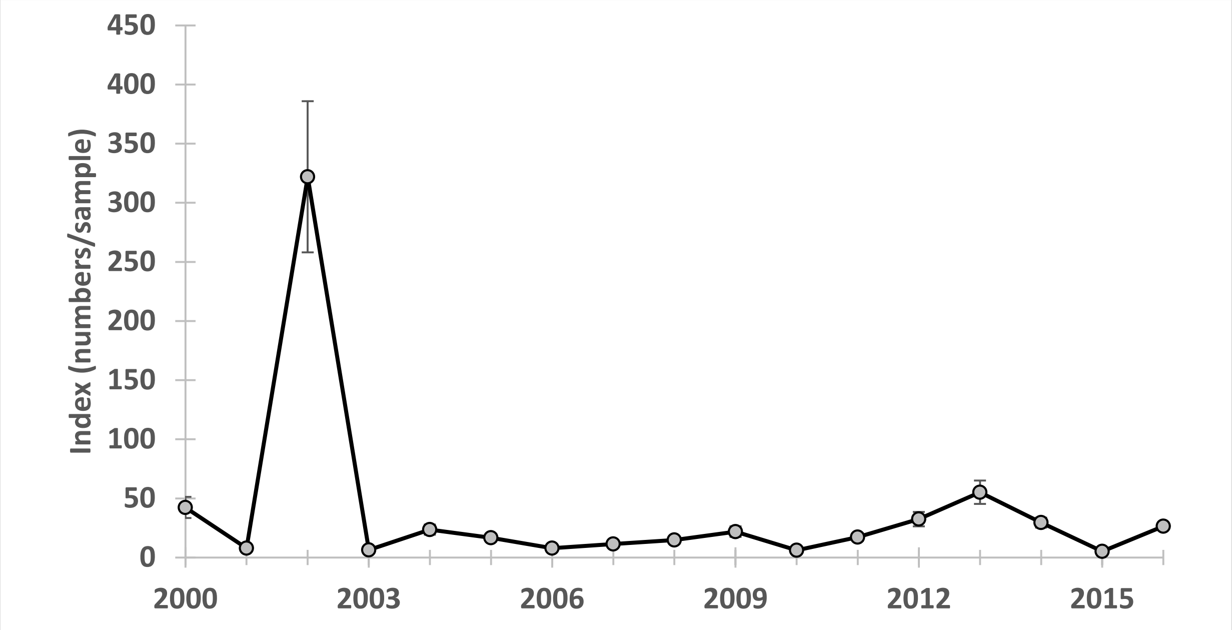

Figure 21. **GLM-standardized index of abundance for American eels caught by New York's annual YOY survey in Carman's River, 2001–2016. The error bars represent the standard errors about the estimates.**

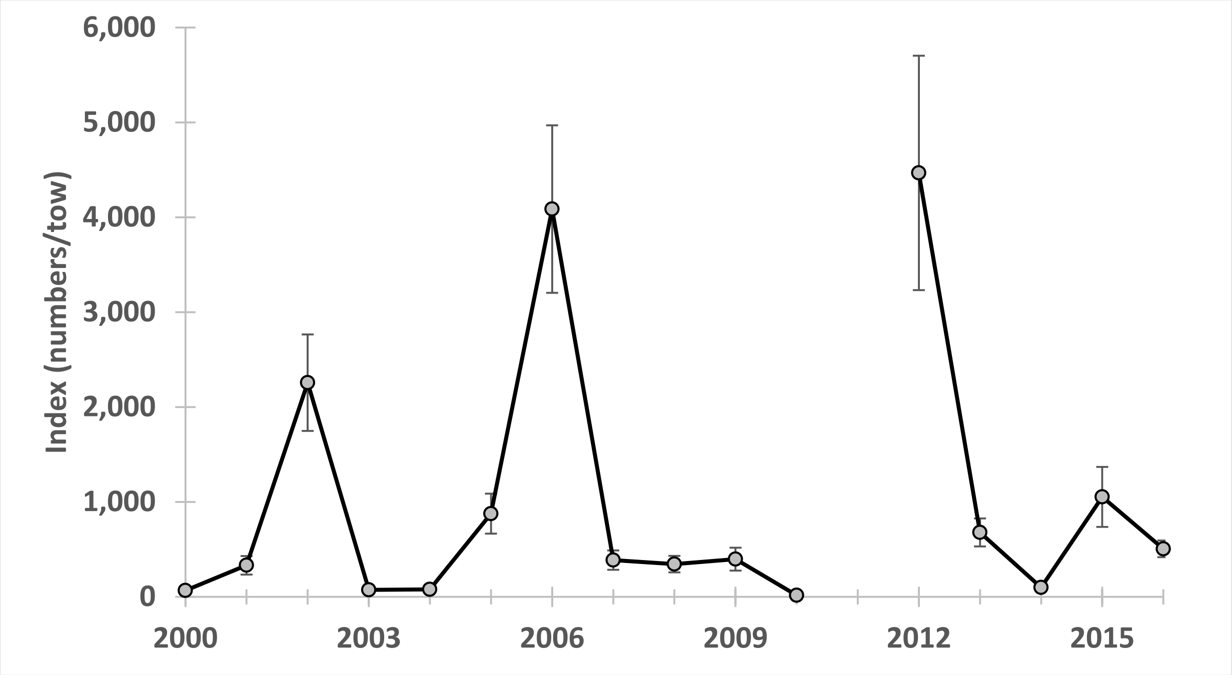

Figure 22. **GLM-standardized index of abundance for YOY American eels caught by New Jersey's annual YOY survey in Patcong Creek, 2000–2016. The error bars represent the standard errors about the estimates.**

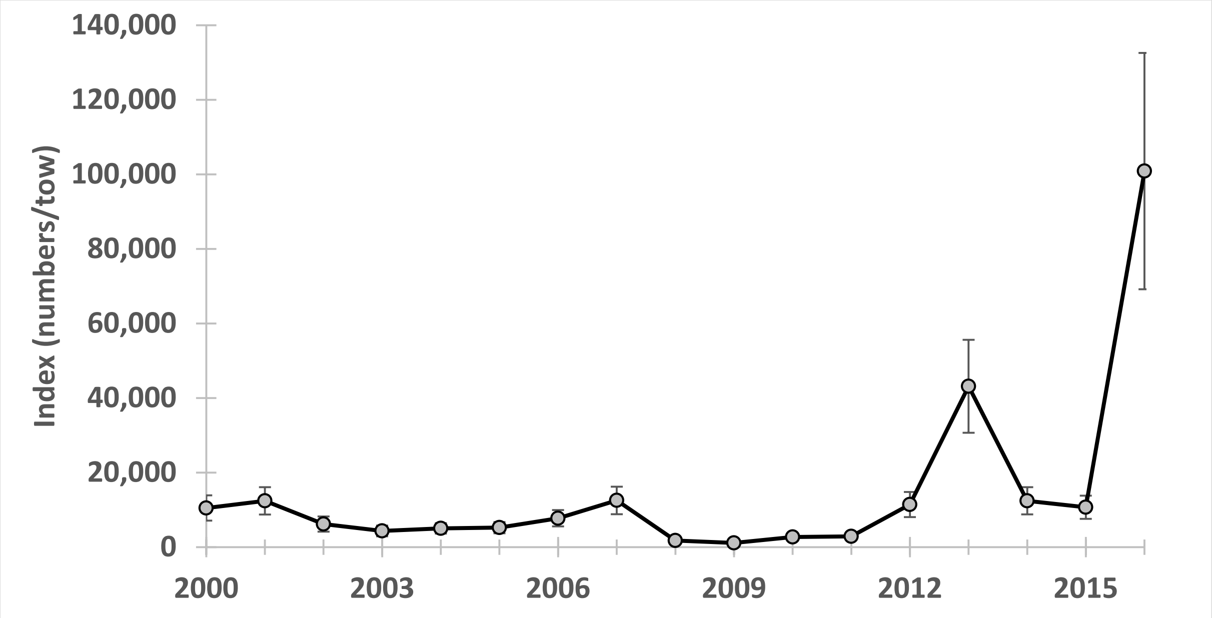

Figure 23. GLM-standardized index of abundance for American eels caught by Delaware's annual YOY survey near the Millsboro Dam, 2000–2016. The error bars represent the standard errors about the estimates.

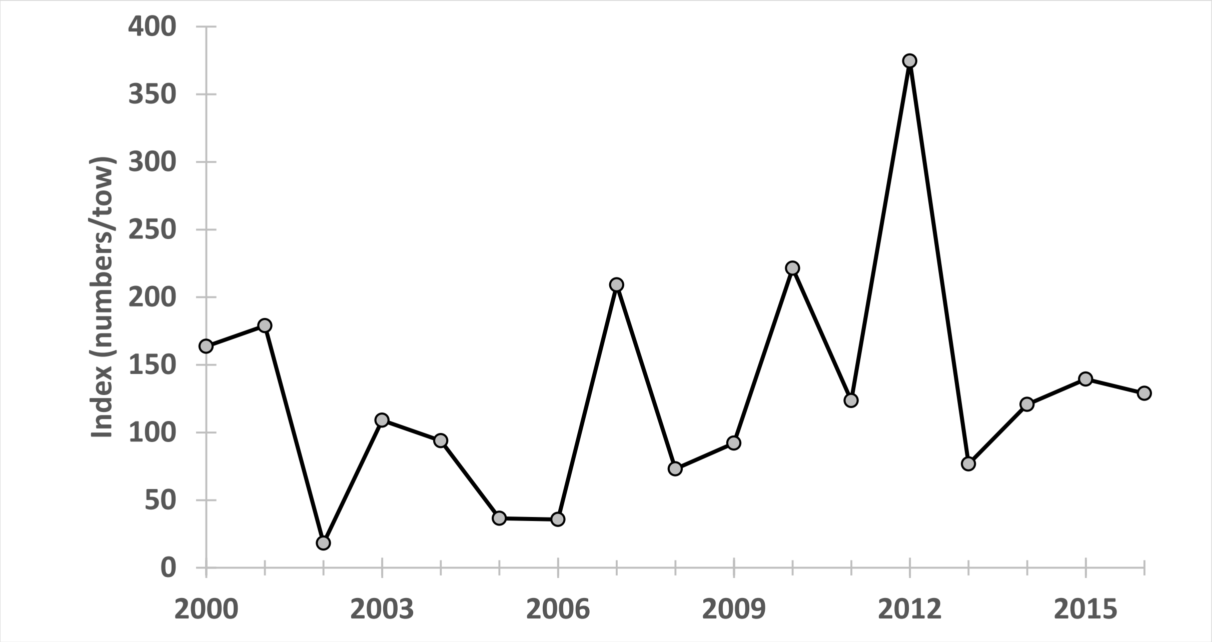

Figure 24. Annual index of abundance for American eels caught by Maryland's annual YOY survey in Turville Creek, 2000–2016. The error bars were omitted from the graph because there were several very large values. See text for more discussion.

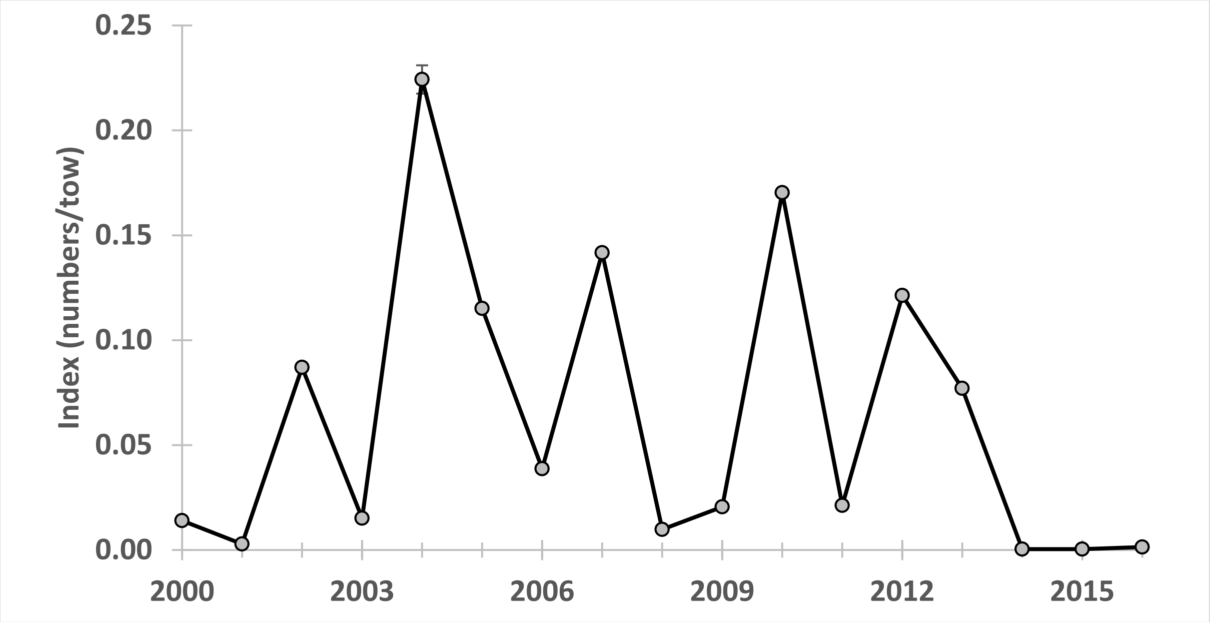

Figure 25. GLM-standardized index of abundance for American eels caught by PRFC's annual YOY survey in Clark's Millpond, 2000–2016. The error bars represent the standard errors about the estimates.

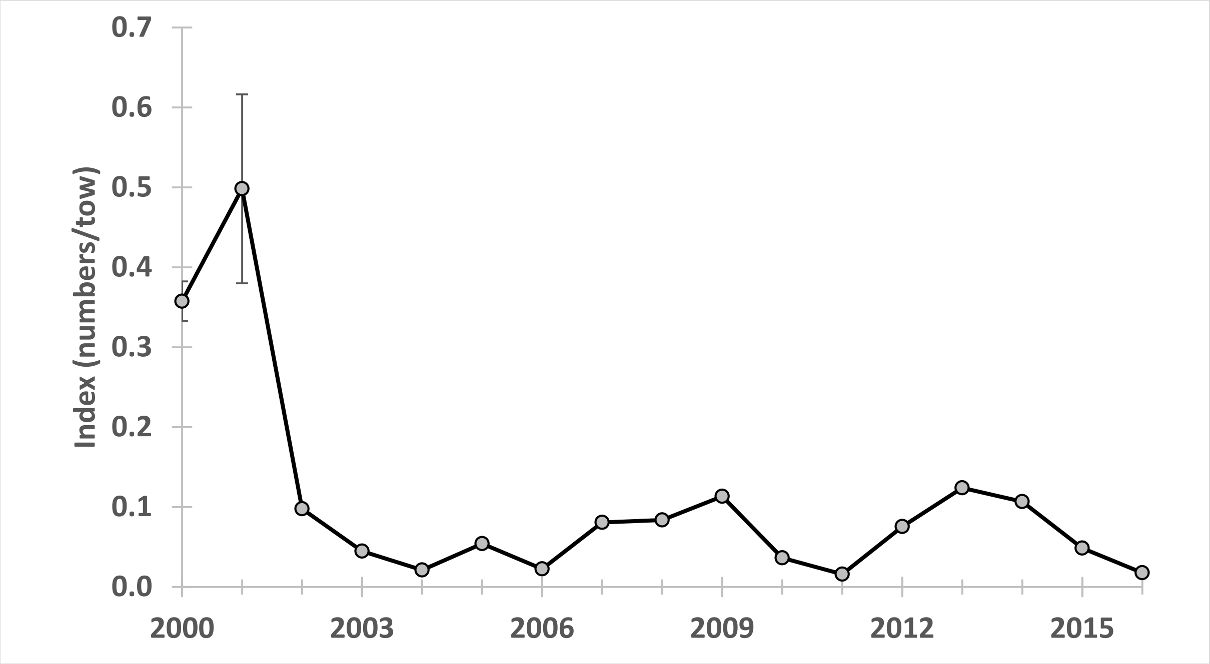

Figure 26. GLM-standardized index of abundance for American eels caught by PRFC's annual YOY survey in Gardy's Millpond, 2000–2016. The error bars represent the standard errors about the estimates.

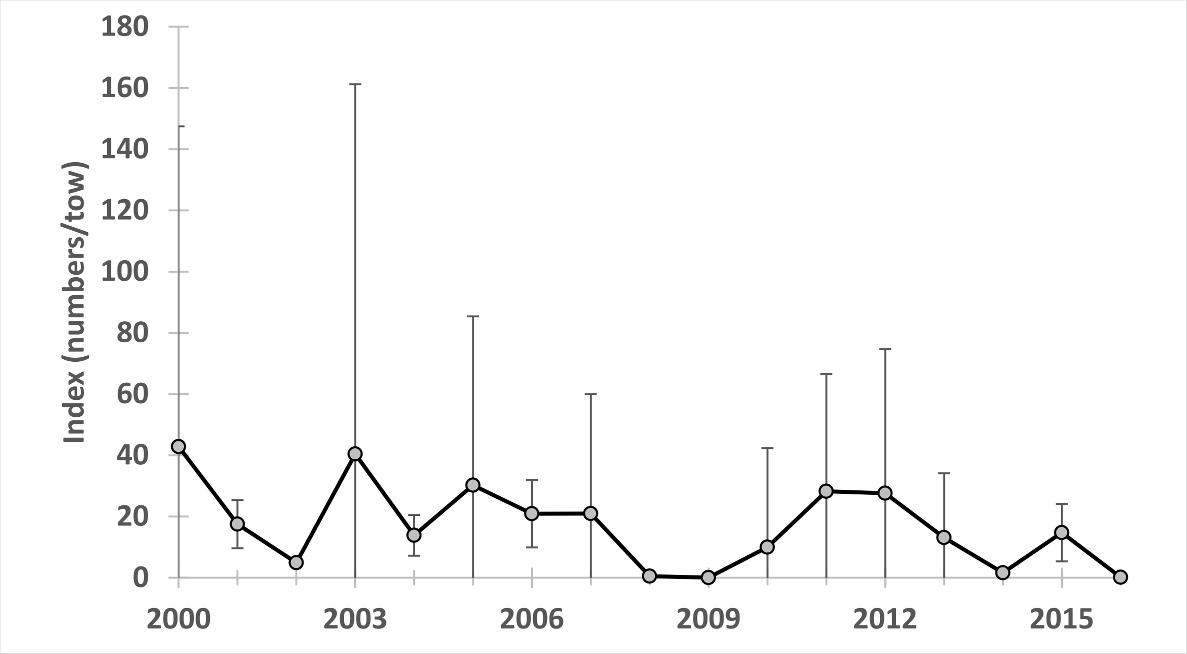

Figure 27. **Annual index of abundance for American eels caught by Virginia's annual YOY survey in Bracken's Pond, 2000–2016. The error bars represent the standard errors about the estimates.**

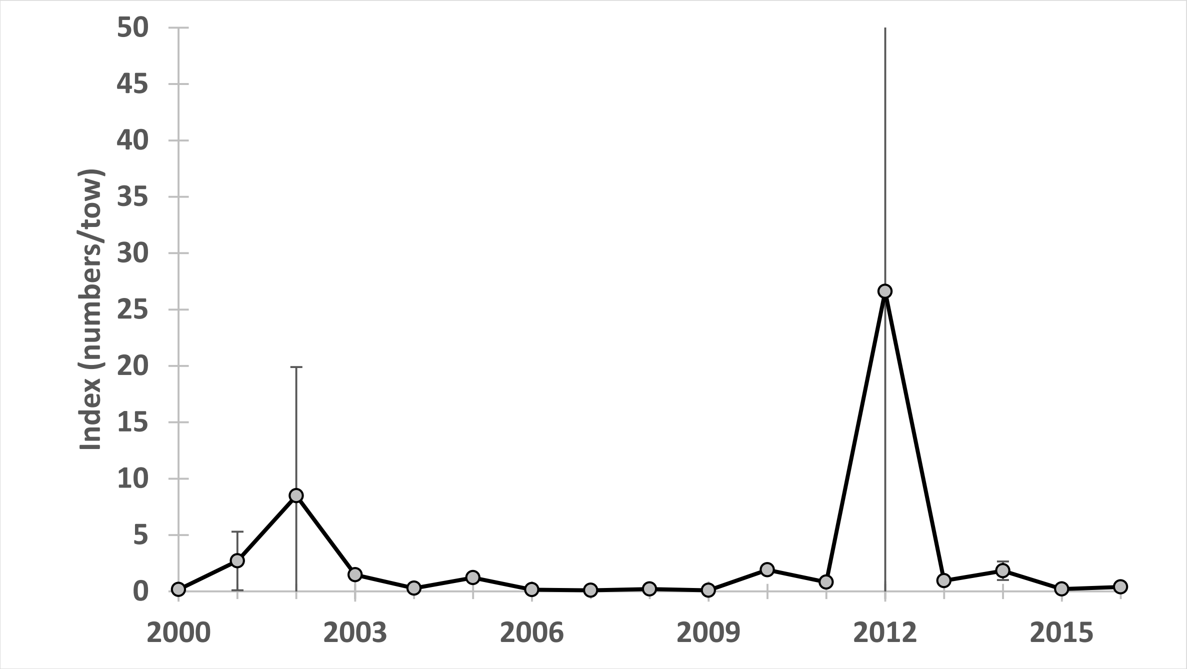

Figure 28. **GLM-standardized index of abundance for American eels caught by Virginia's annual YOY survey in Kamp's Millpond, 2000–2016. The error bars represent the standard errors about the estimates.**

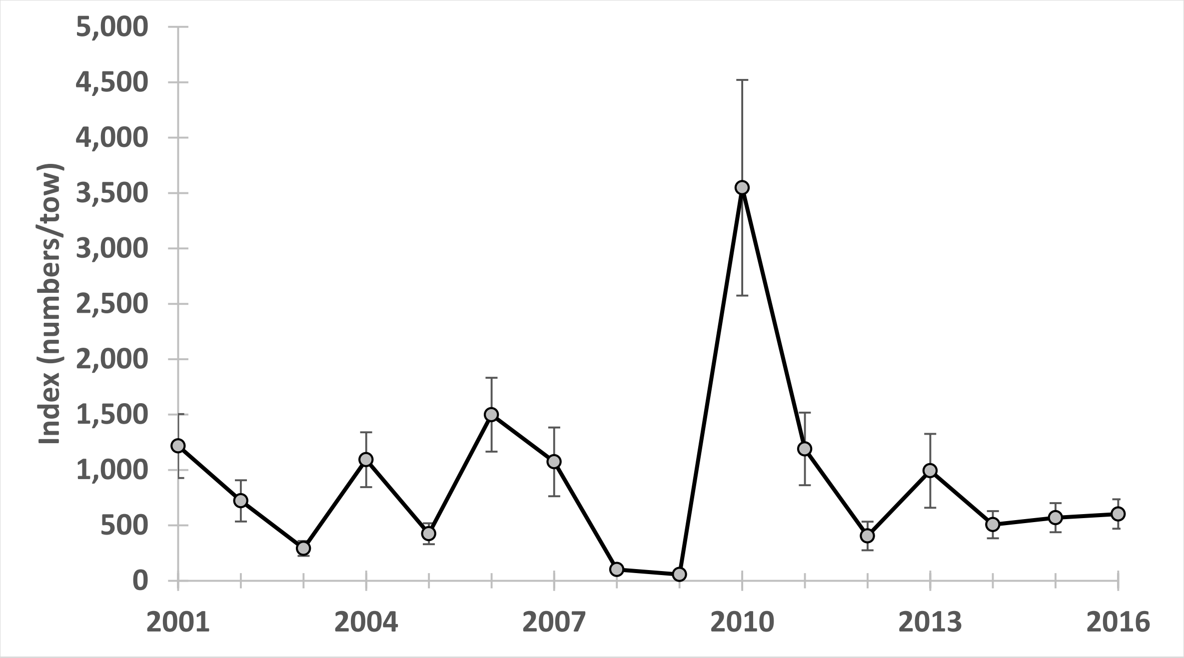

Figure 29. GLM-standardized index of abundance for American eels caught by Virginia's annual YOY survey in Wormley Creek, 2001–2016. The error bars represent the standard errors about the estimates.

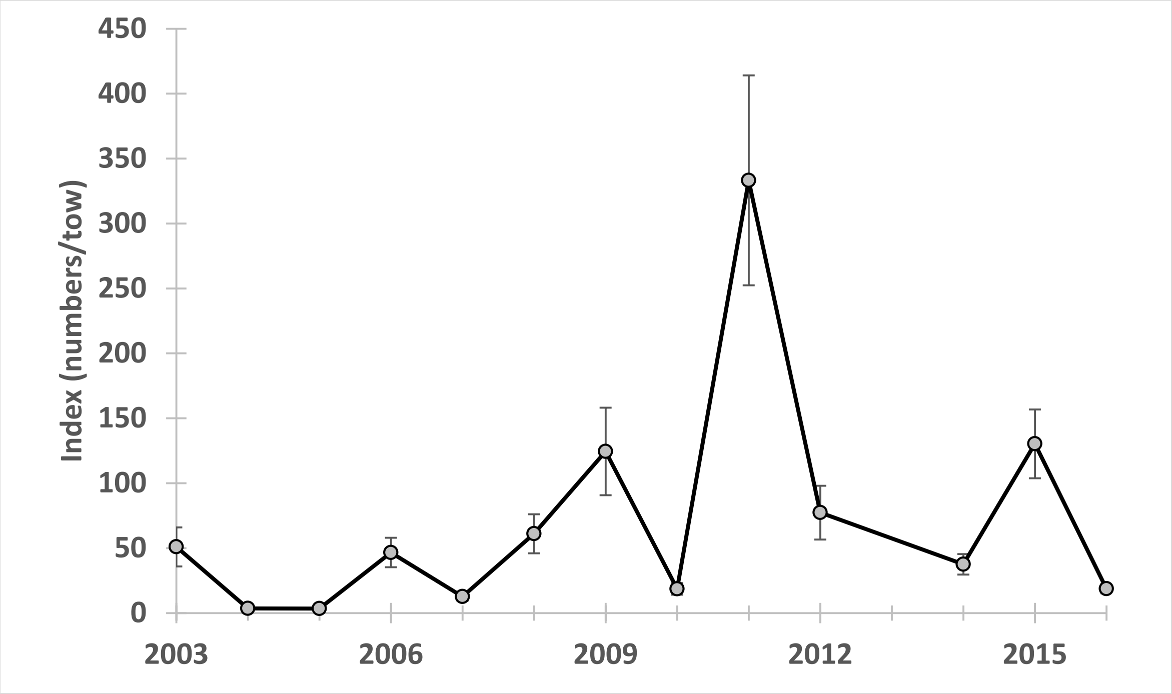

Figure 30. GLM-standardized index of abundance for American eels caught by Virginia's annual YOY survey in Wareham's Pond, 2003–2016. The error bars represent the standard errors about the estimates.

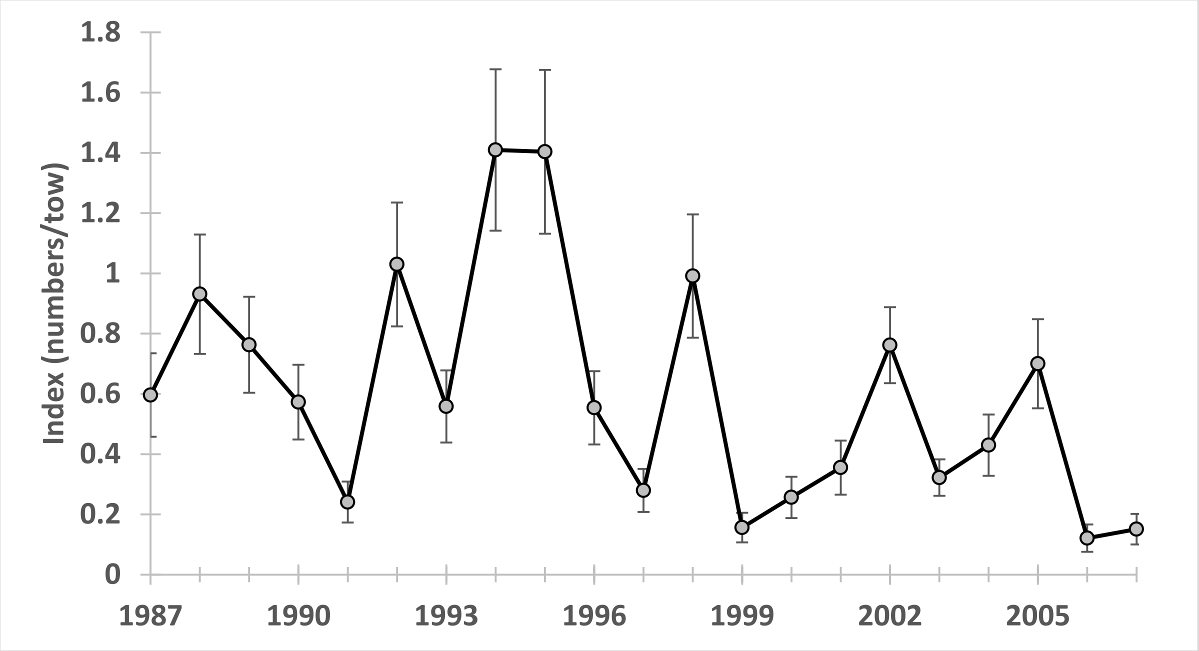

Figure 31. **GLM-standardized index of abundance for YOY American eels caught by North Carolina's Beaufort Bridgenet Ichthyoplankton Sampling Program (BBISP) conducted by NOAA, 1987–2007. The error bars represent the standard errors about the estimates.**

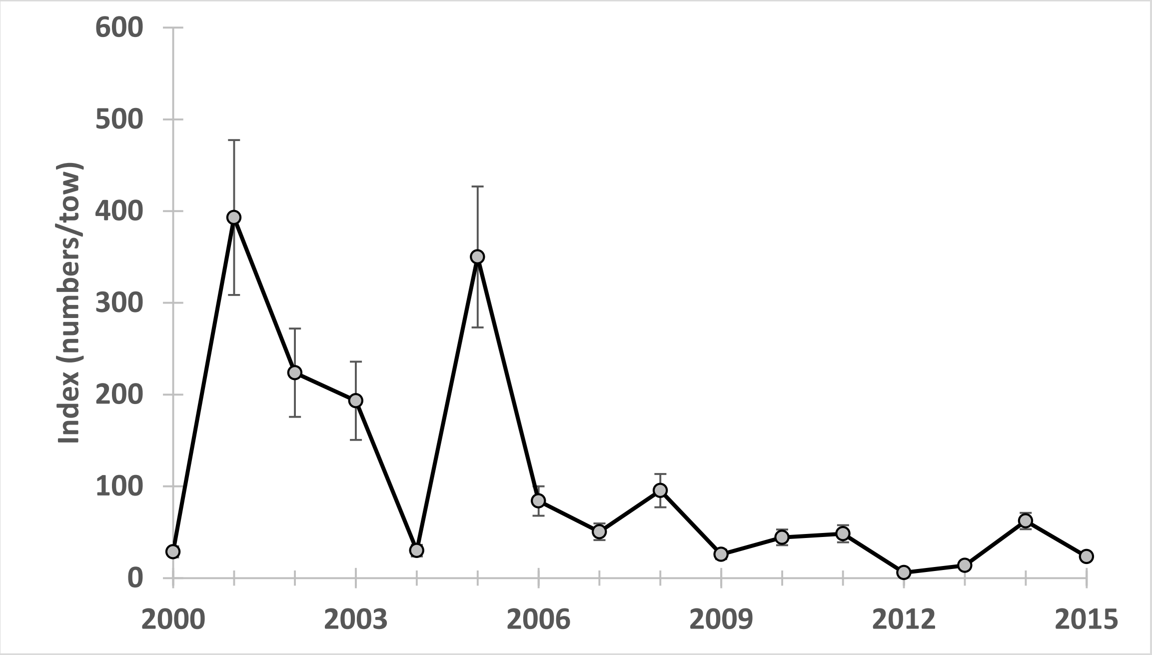

Figure 32. **GLM-standardized index of abundance for American eels caught by South Carolina's annual YOY survey in Goose Creek, 2000–2015. The error bars represent the standard errors about the estimates.**

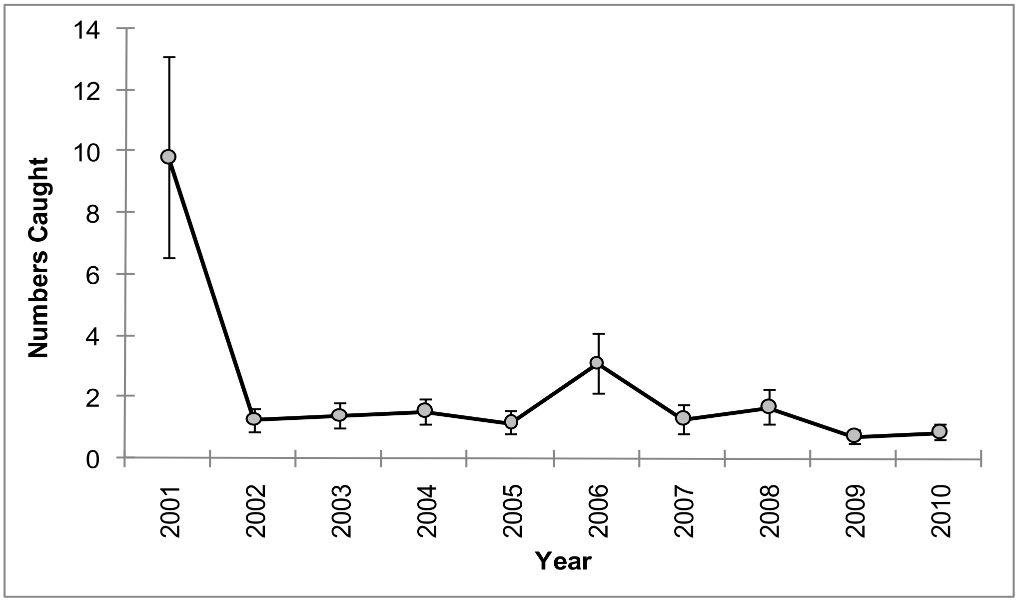

Figure 33. GLM-standardized index of abundance for American eels caught by Georgia's annual YOY survey near the Altamaha Canal, 2001–2010. The error bars represent the standard errors about the estimates. This index was not updated because the site was discontinued.

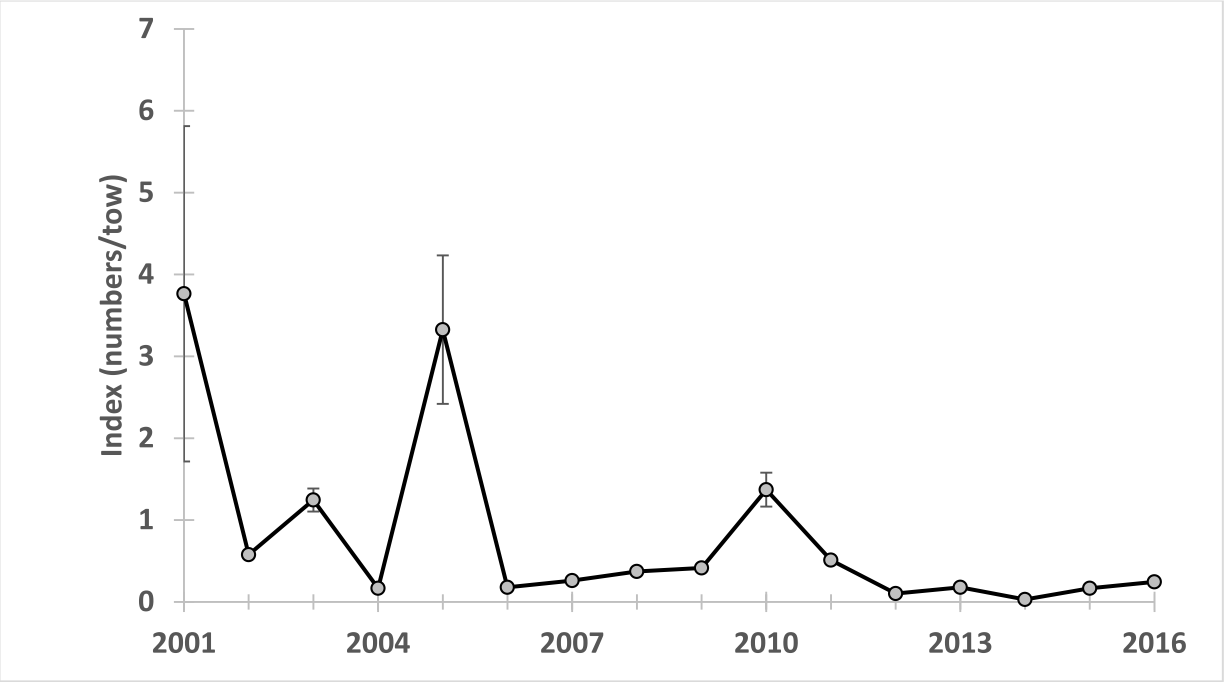

Figure 34. Annual index of abundance for American eels caught by Florida's annual YOY survey near Guana River Dam, 2001–2016. The error bars represent the standard errors about the estimates.

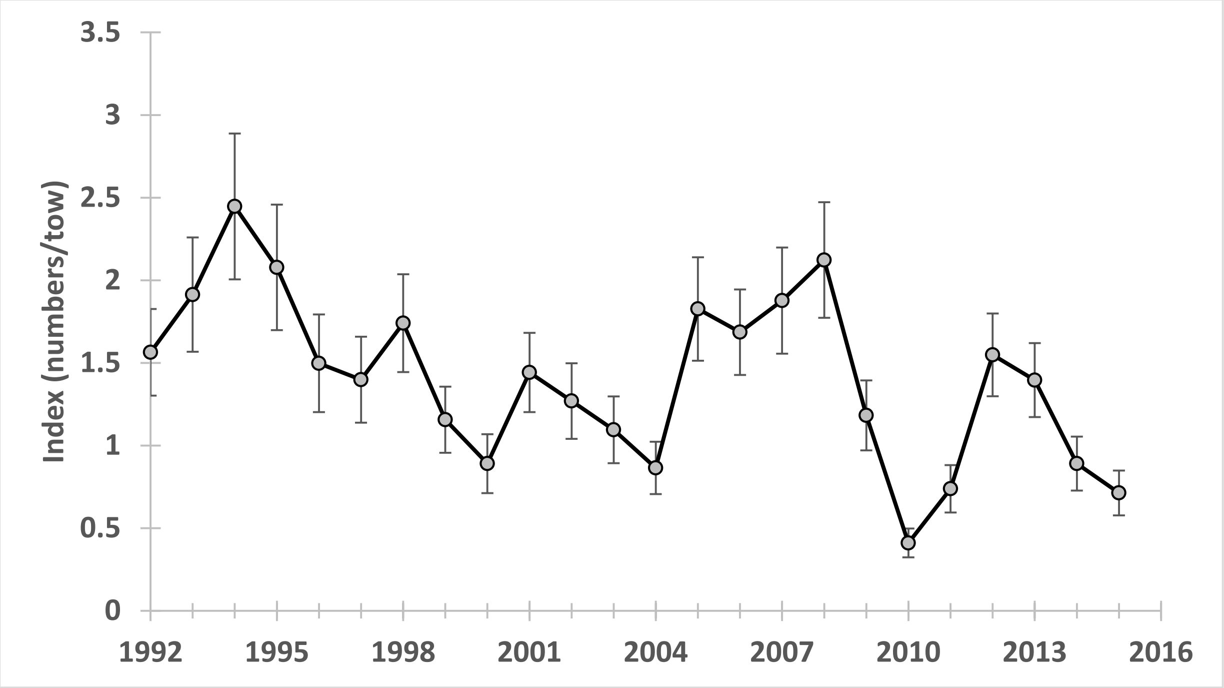

Figure 35. GLM-standardized index of abundance for YOY American eels caught by the Little Egg Inlet Ichthyoplankton Survey, 1992–2016. The error bars represent the standard errors about the estimates.

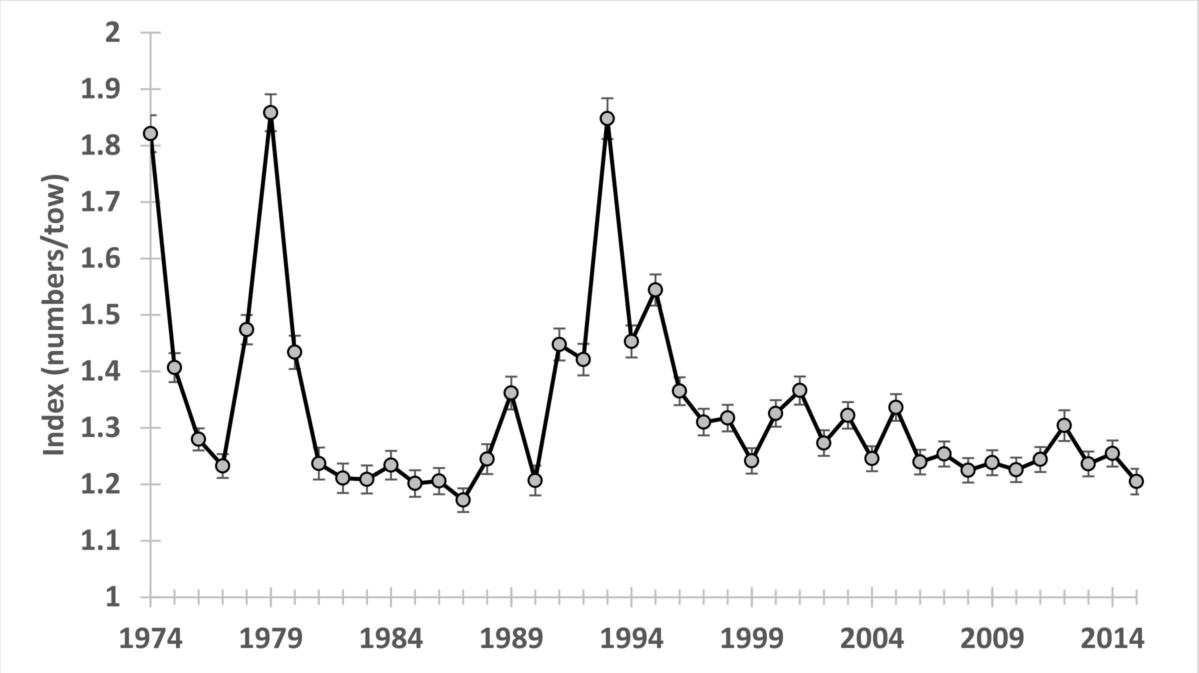

Figure 36. GLM-standardized index of abundance for YOY American eels caught by the Hudson River Estuary Monitoring Program's Ichthyoplankton Survey, 1974–2015. The error bars represent the standard errors about the estimates.

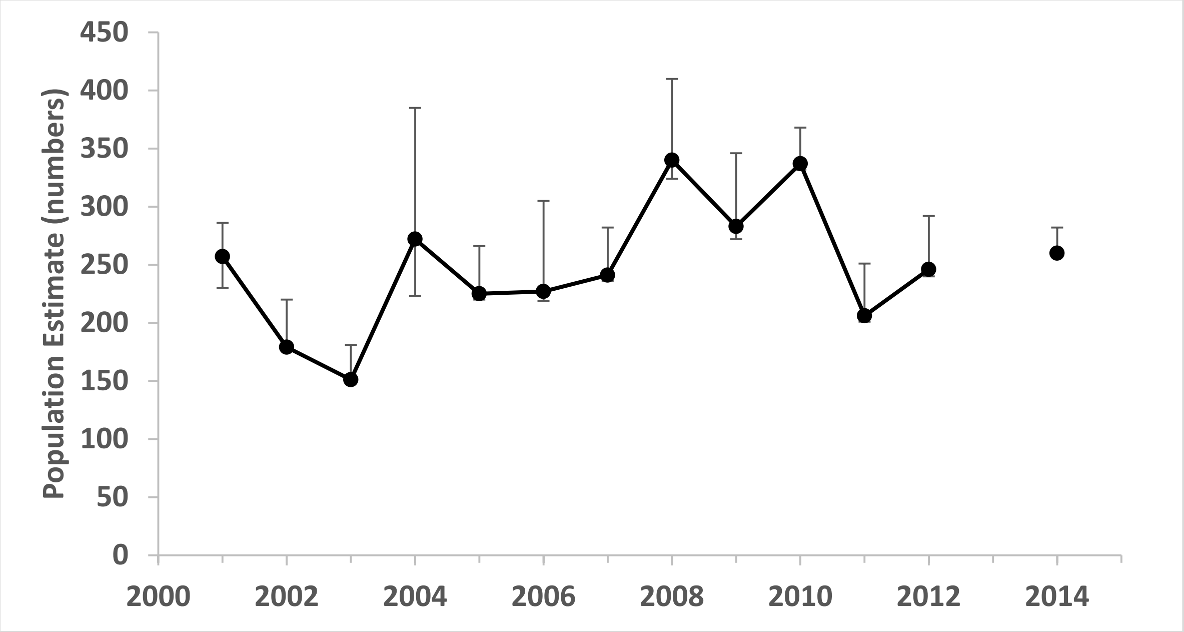

Figure 37. Annual index of abundance for American eels caught by the CTDEP Electrofishing Survey in the Farmill River, 2001–2014. The error bars represent 95% confidence intervals.

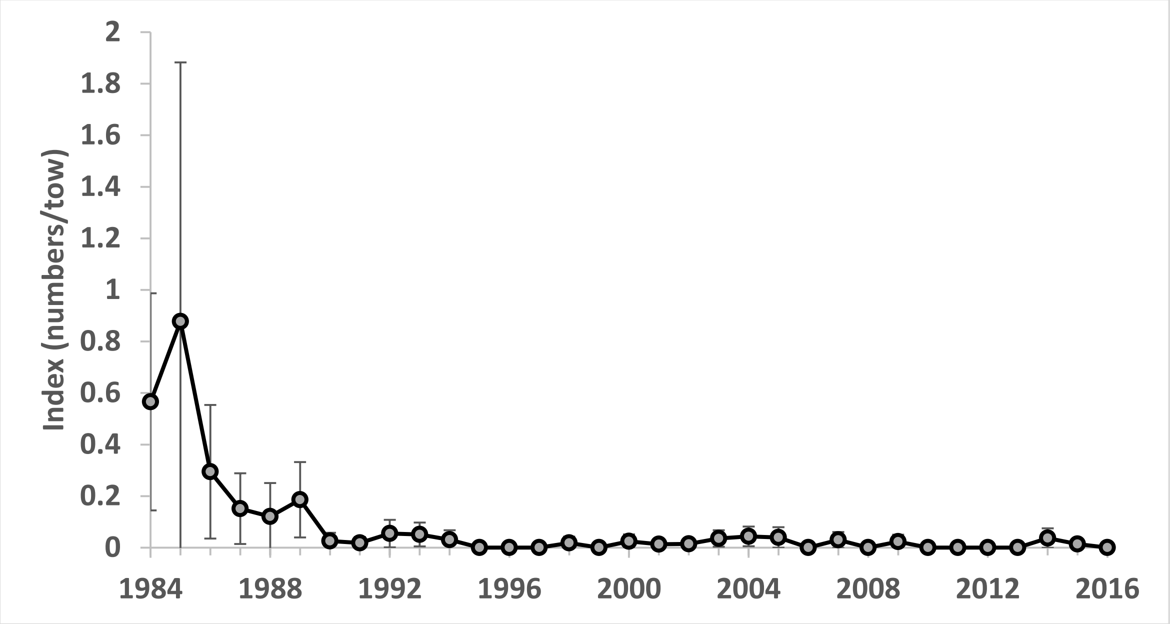

Figure 38. GLM-standardized index of abundance for American eels caught by the NY Western Long Island Survey, 1984–2016. The error bars represent the standard errors about the estimates.

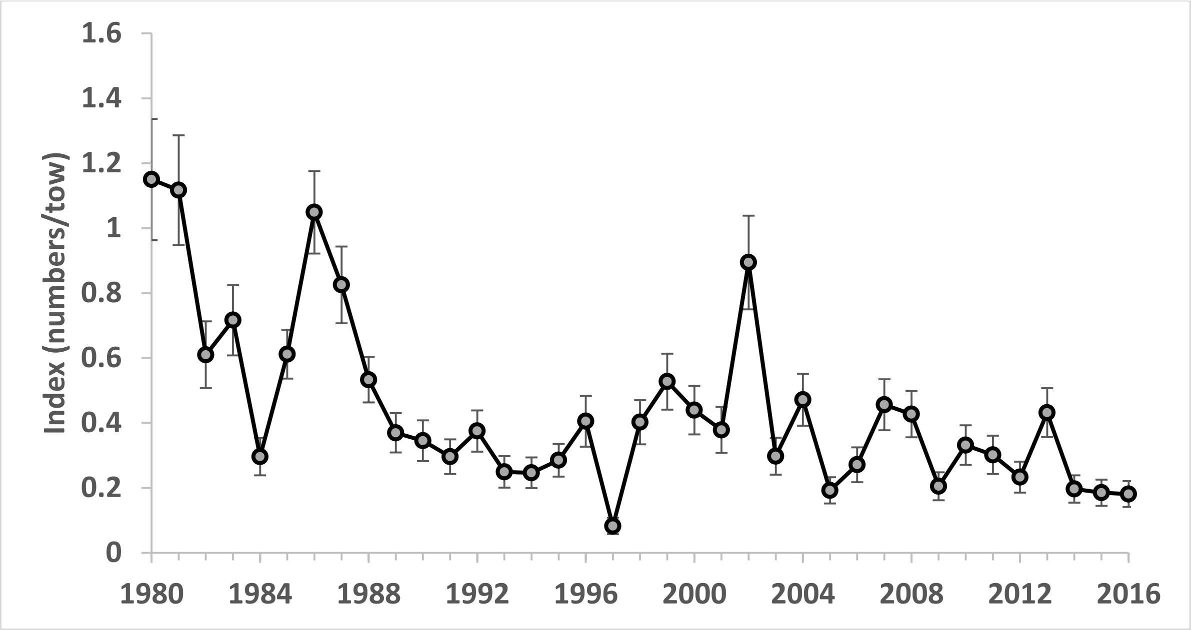

Figure 39. Annual index of abundance for American eels caught by the NYDEC Alosine Beach Seine Survey, 1980–2016. The error bars represent the standard errors about the estimates.

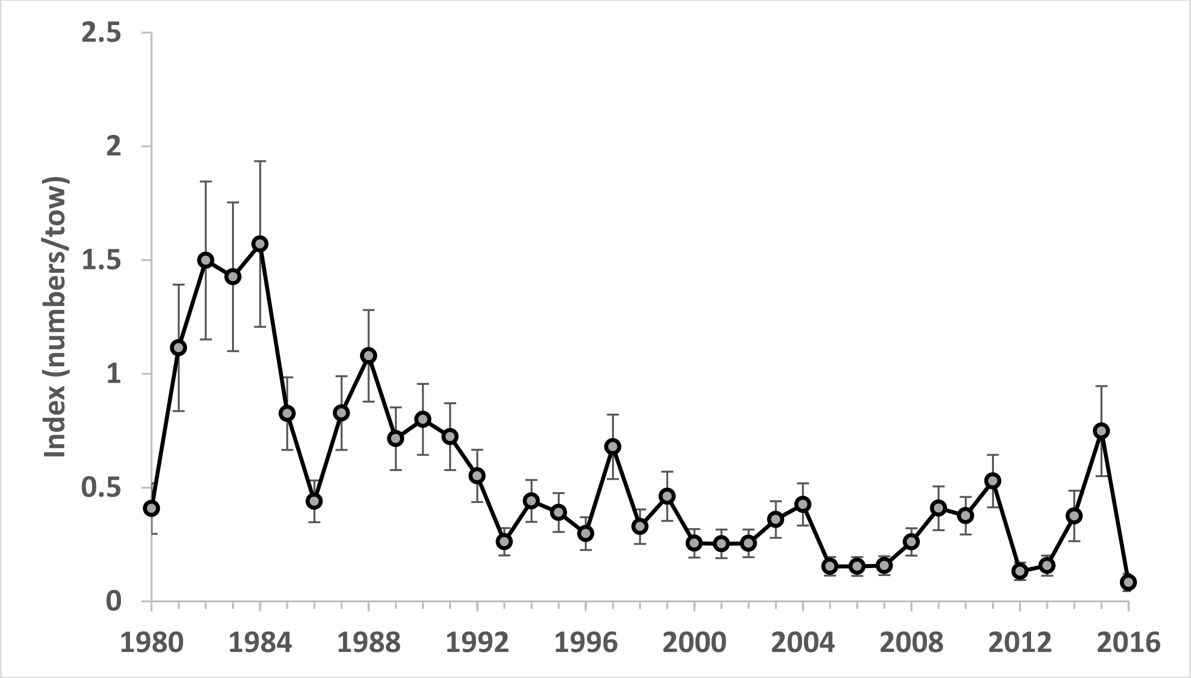

Figure 40. Annual index of abundance for American eels caught by the NYDEC Striped Bass Beach Seine Survey, 1980–2016. The error bars represent the standard errors about the estimates.

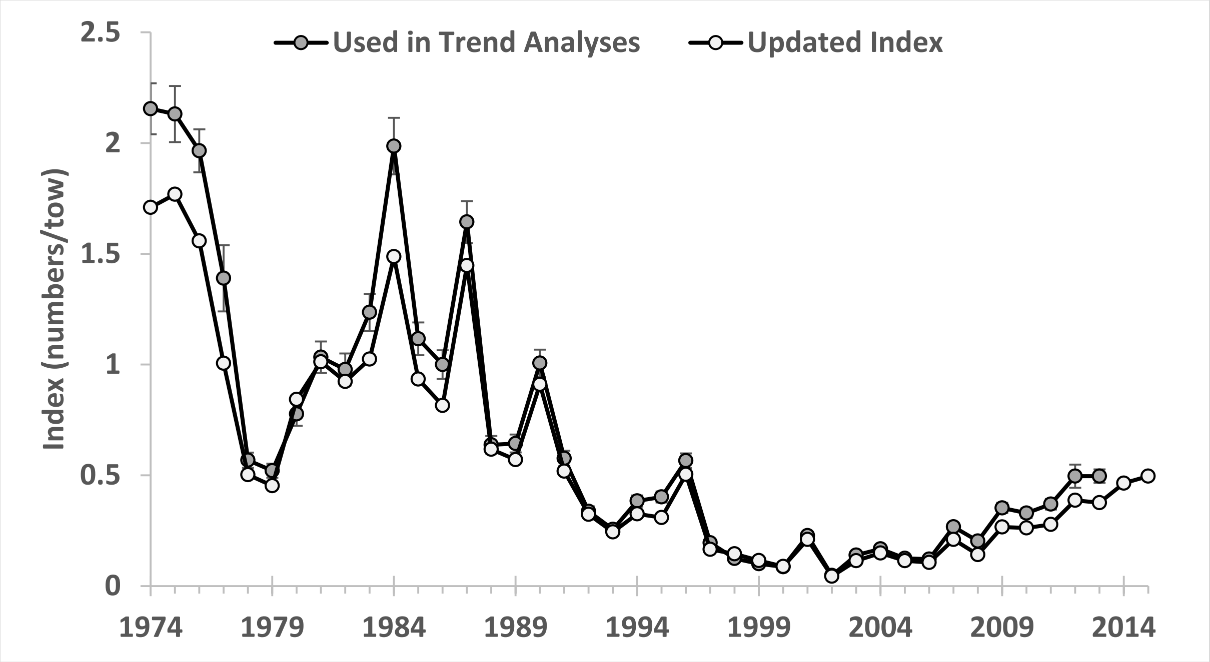

Figure 41. GLM-standardized index of abundance for yearling and older American eels caught by the HRE Monitoring Program. The error bars represent the standard errors about the estimates. Refer to section 5.2.2.1 for index discussion.

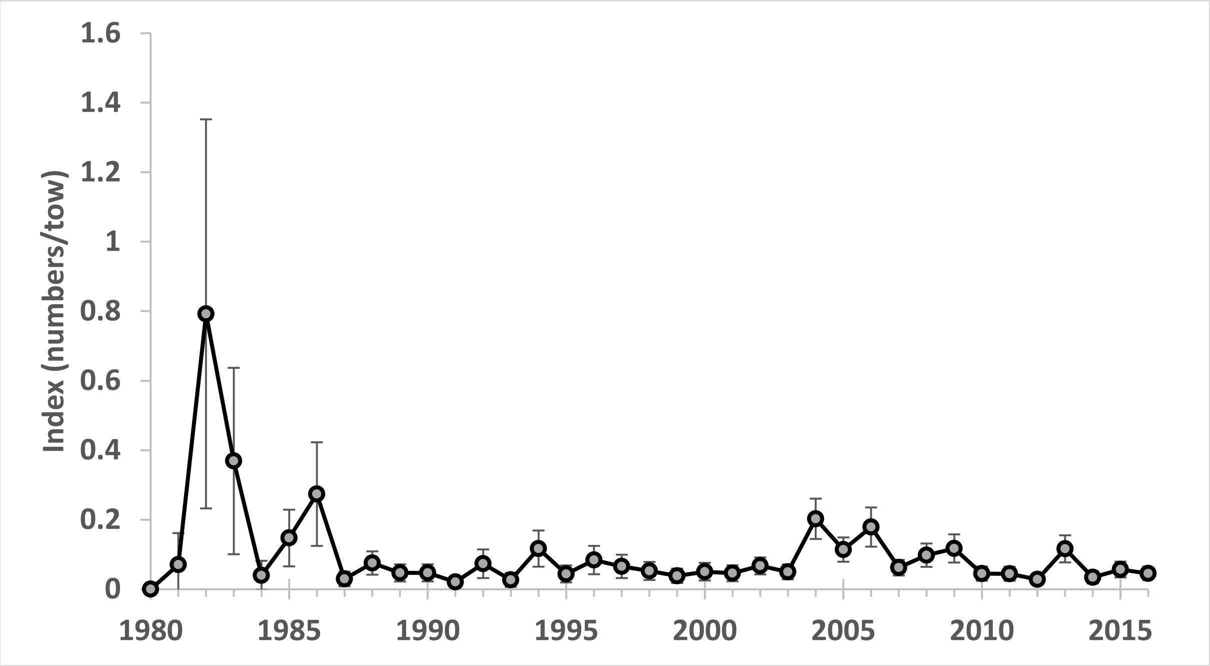

Figure 42. GLM-standardized index of abundance for American eels caught by NJDFW's Striped Bass Seine Survey, 1980–2016. The error bars represent the standard errors about the estimates.

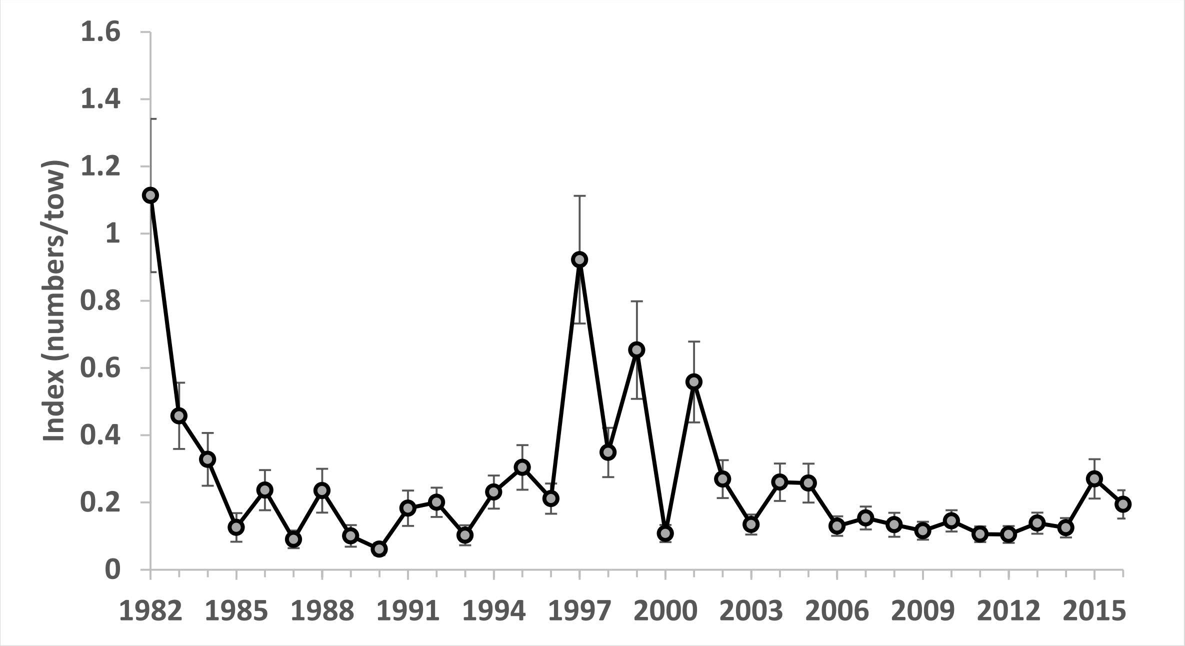

Figure 43. **GLM-standardized index of abundance for American eels caught by the Delaware Trawl Survey, 1982–2016. The error bars represent the standard errors about the estimates.**

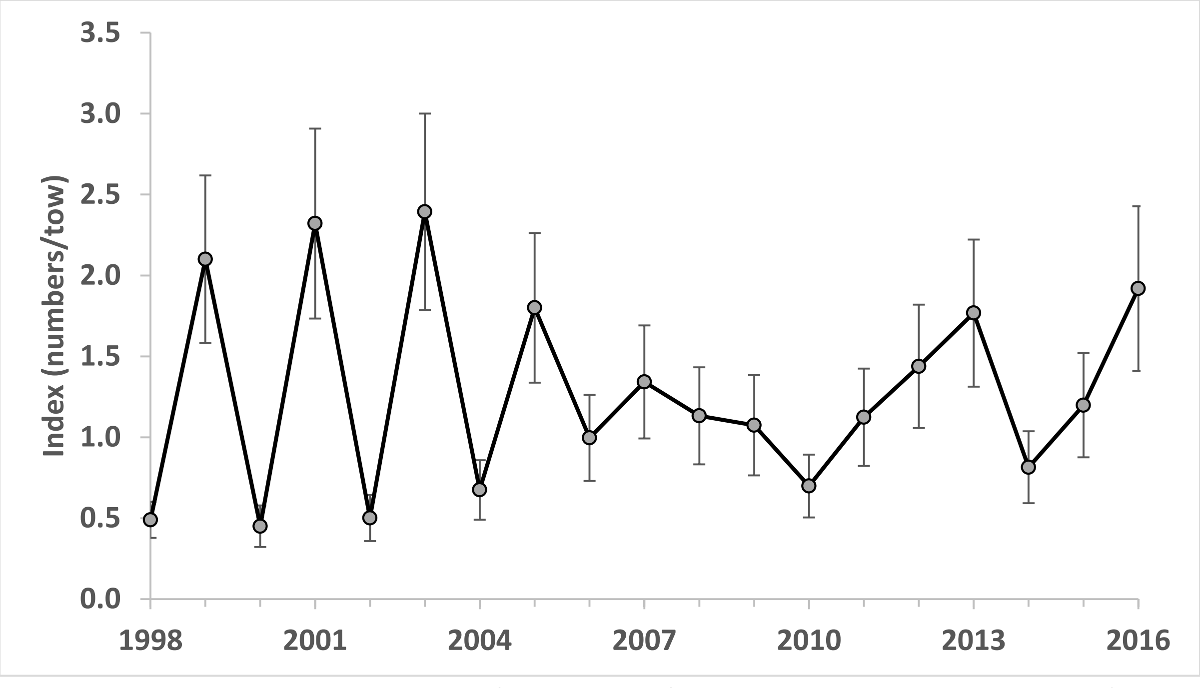

Figure 44. **GLM-standardized index of abundance for American eels caught by PSEG's Trawl Survey, 1998-2016. The error bars represent the standard errors about the estimates.**

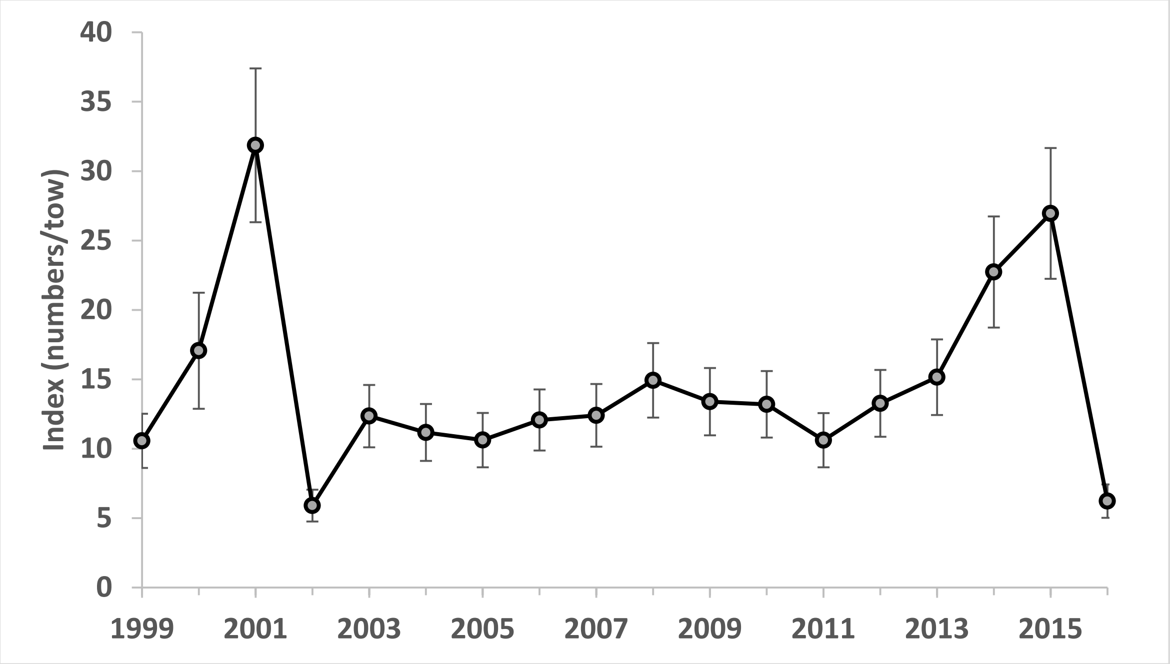

Figure 45. GLM-standardized index of abundance for American eels caught by the Area 6 Electrofishing Survey, 1999–2016. The error bars represent the standard errors about the estimates.

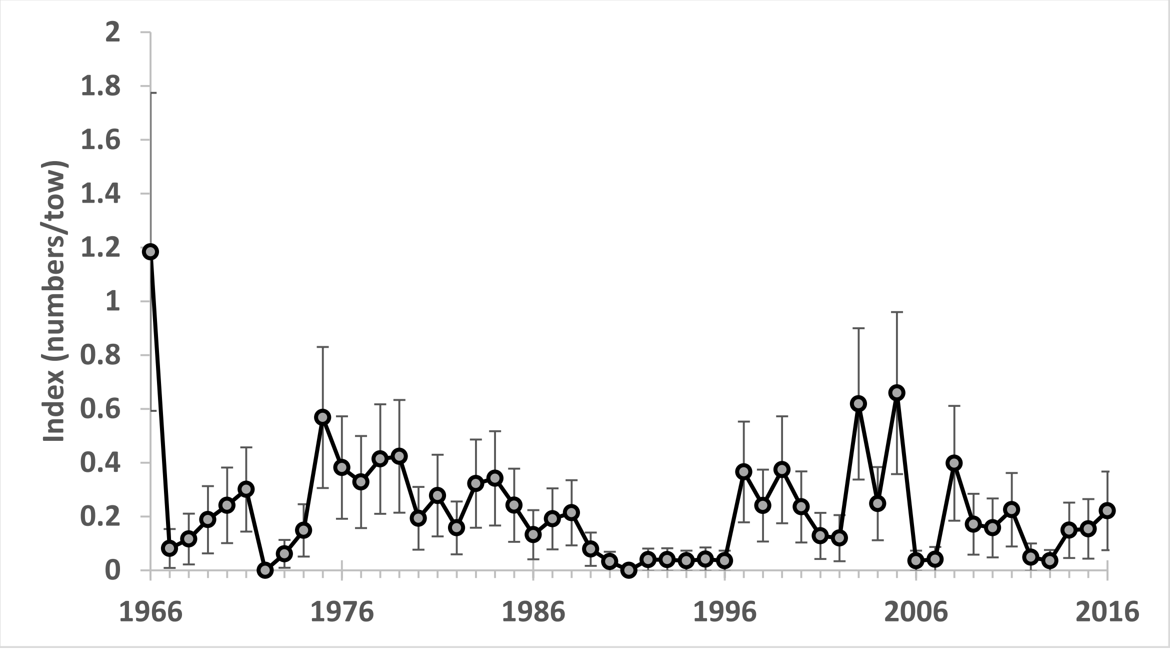

Figure 46. GLM-standardized index of abundance for American eels caught by the MDDNR Striped Bass Seine Survey, 1966–2016. The error bars represent the standard errors about the estimates.

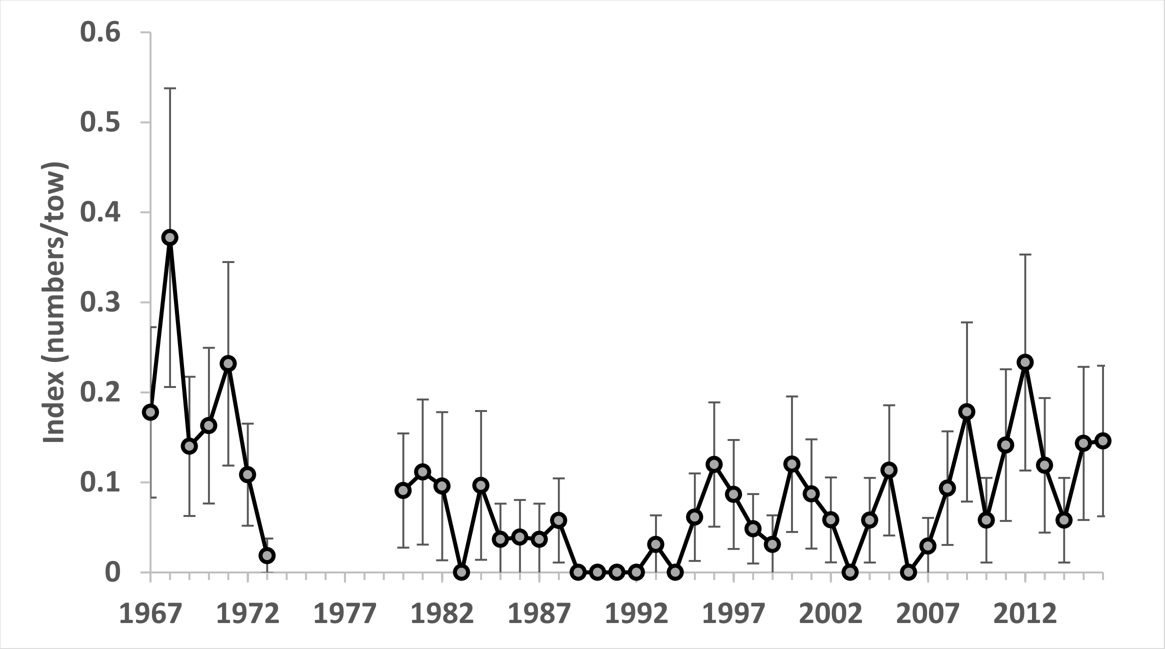

Figure 47. GLM-standardized index of abundance for American eels caught by the VIMS Juvenile Striped Bass Seine Survey, 1967–2016. The error bars represent the standard errors about the estimates.

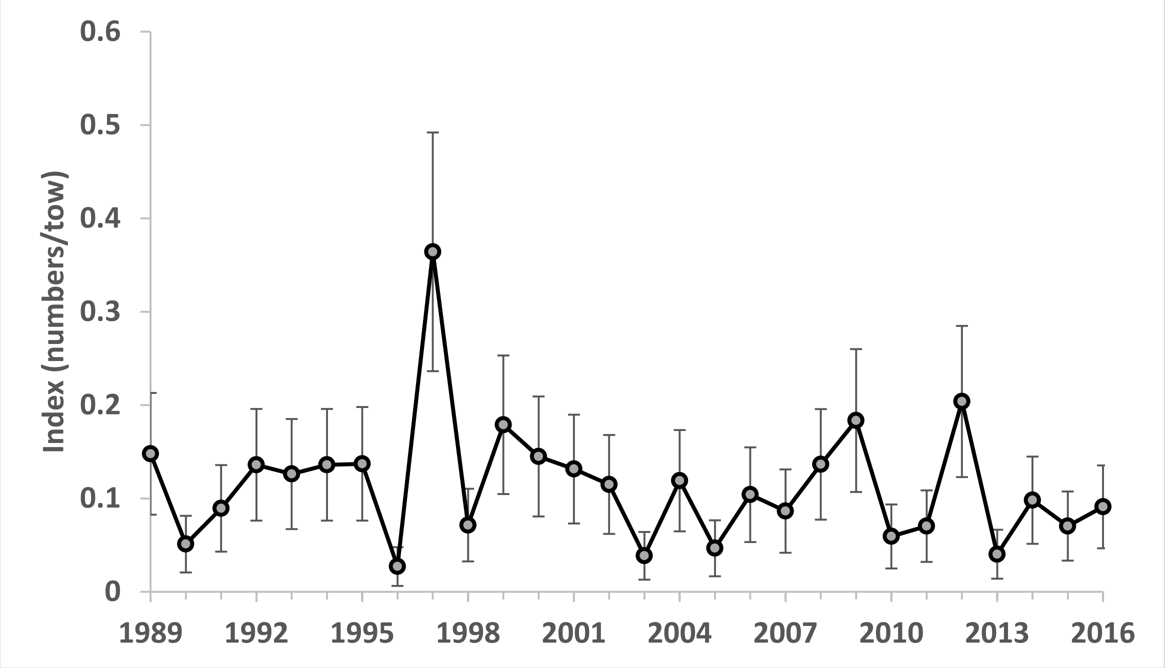

Figure 48. GLM-standardized index of abundance for American eels caught by the VIMS Juvenile Striped Bass Seine Survey, 1989–2016. The error bars represent the standard errors about the estimates.

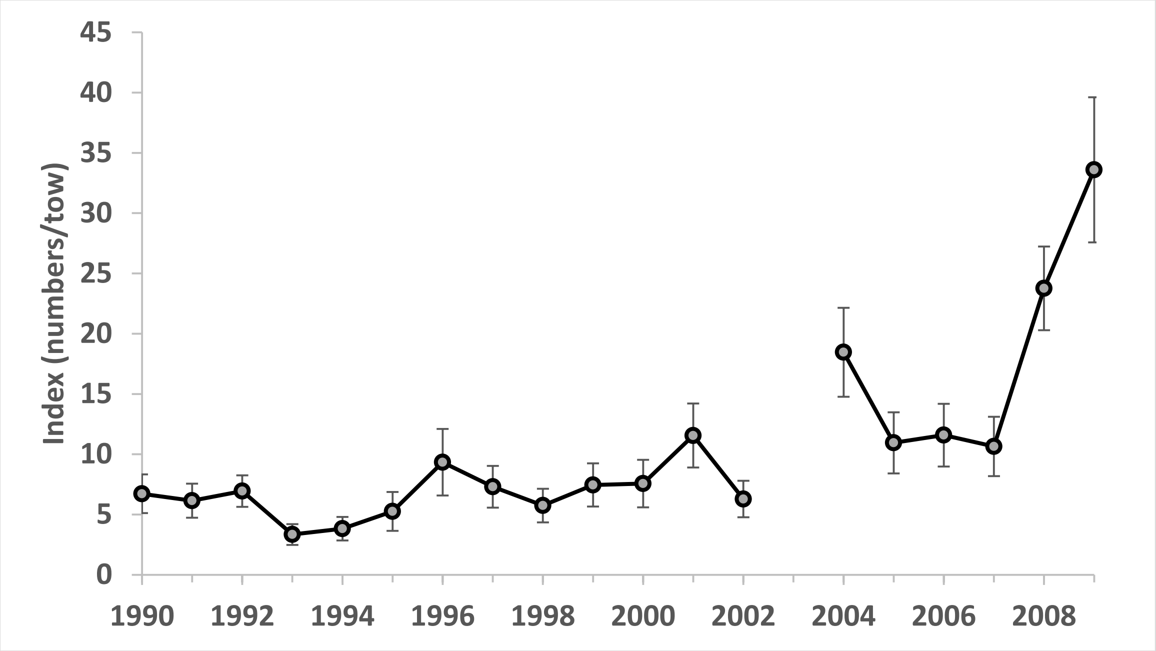

Figure 49. GLM-standardized index of abundance for American eels caught by the North Anna Electrofishing Survey, 1990–2009. The error bars represent the standard errors about the estimates.

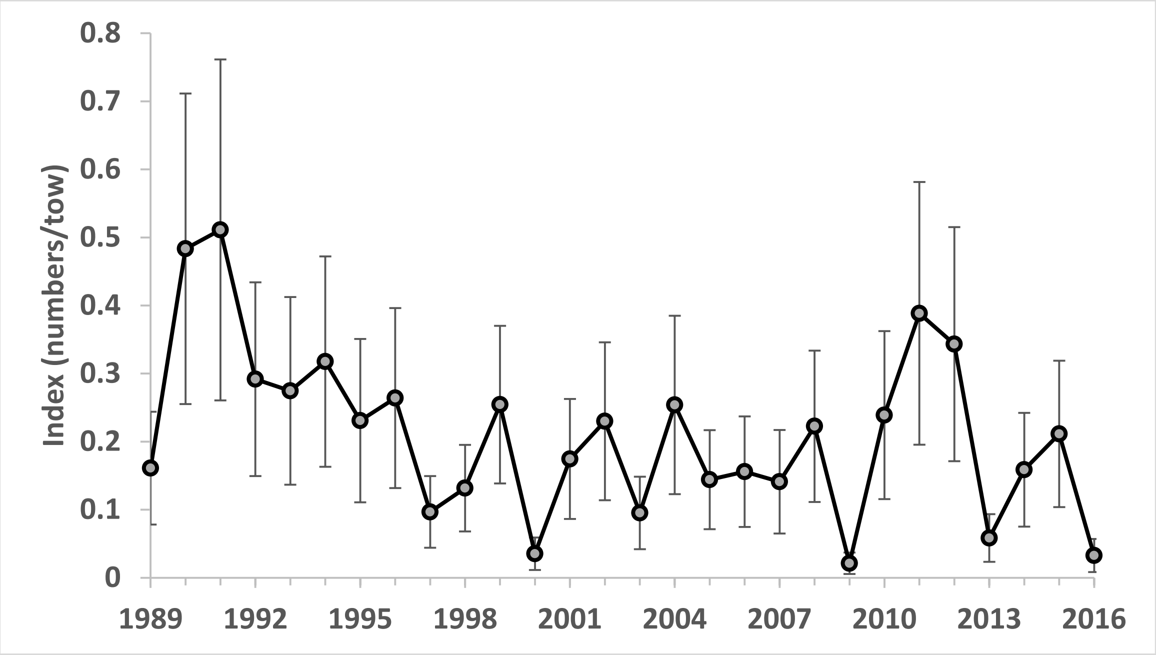

Figure 50. GLM-standardized index of abundance for American eels caught by the NCDMF Estuarine Trawl Survey, 1989–2016. The error bars represent the standard errors about the estimates.

Figure 51. GLM-standardized index of abundance for American eels caught by the SC Electrofishing Survey, 2001–2016. The error bars represent the standard errors about the estimates.

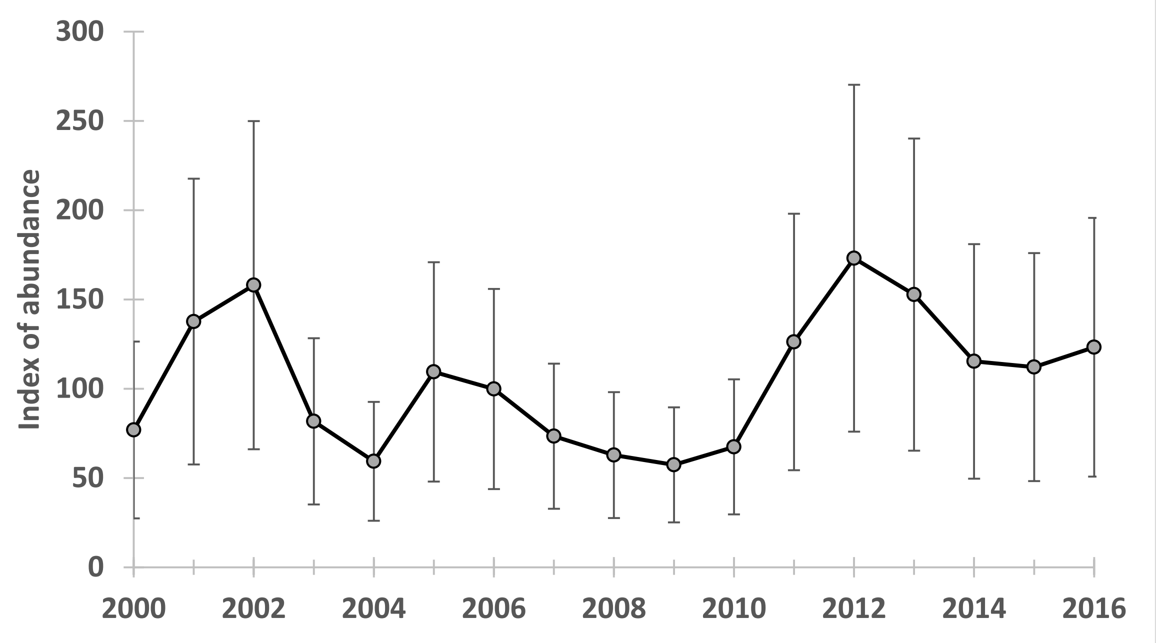

Figure 52. GLM-standardized, short-term index of abundance for YOY American eels along the Atlantic Coast, 2000–2016. The error bars represent the standard errors about the estimates.

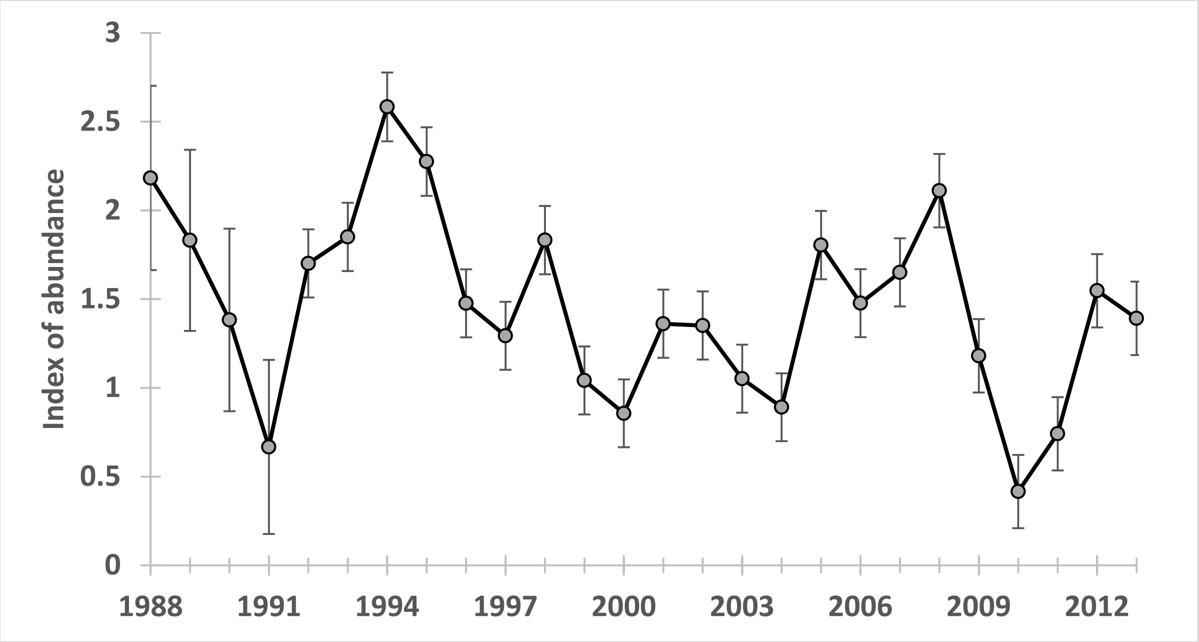

Figure 53. GLM-standardized, long-term index of abundance for YOY American eels along the Atlantic Coast, 1988–2013. The error bars represent the standard errors about the estimates.

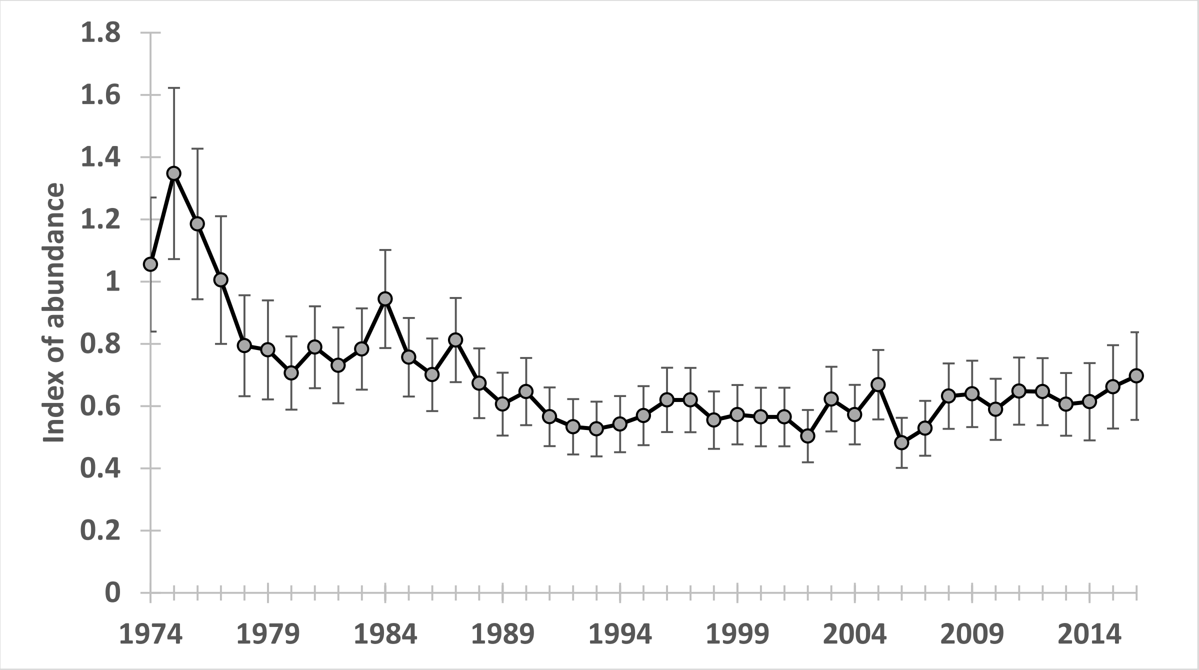

Figure 54. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1974–2016 (40-plus-year index). The error bars represent the standard errors about the estimates.

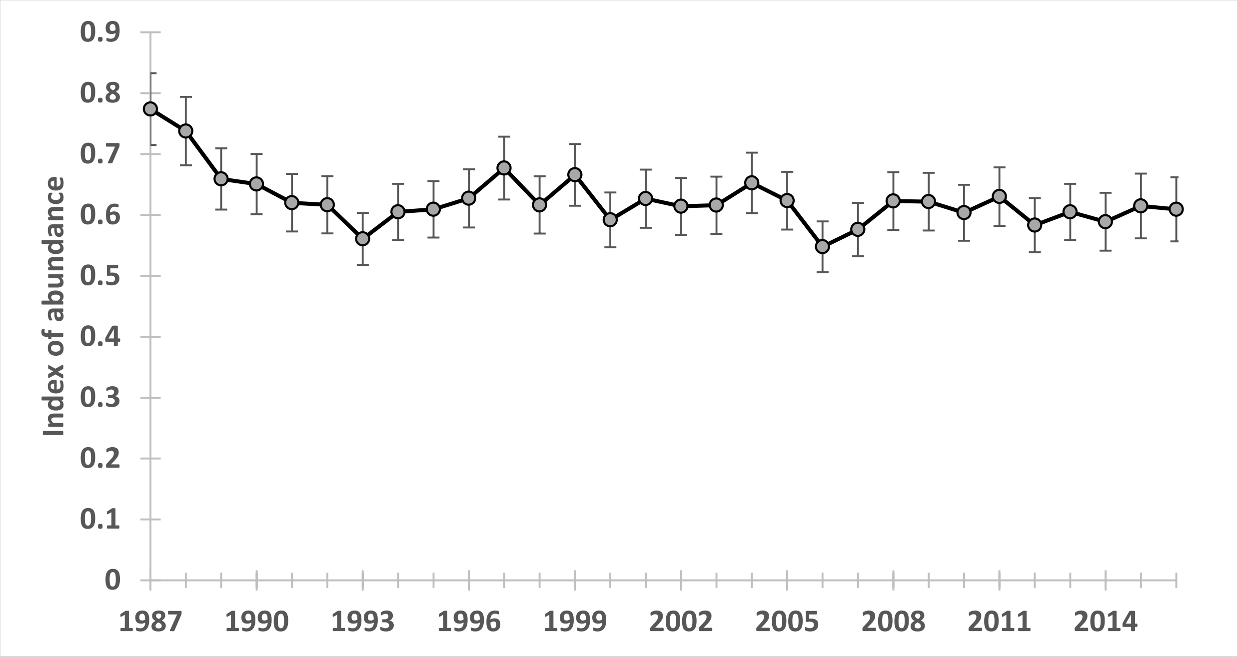

Figure 55. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1987–2016 (30-year index). The error bars represent the standard errors about the estimates.

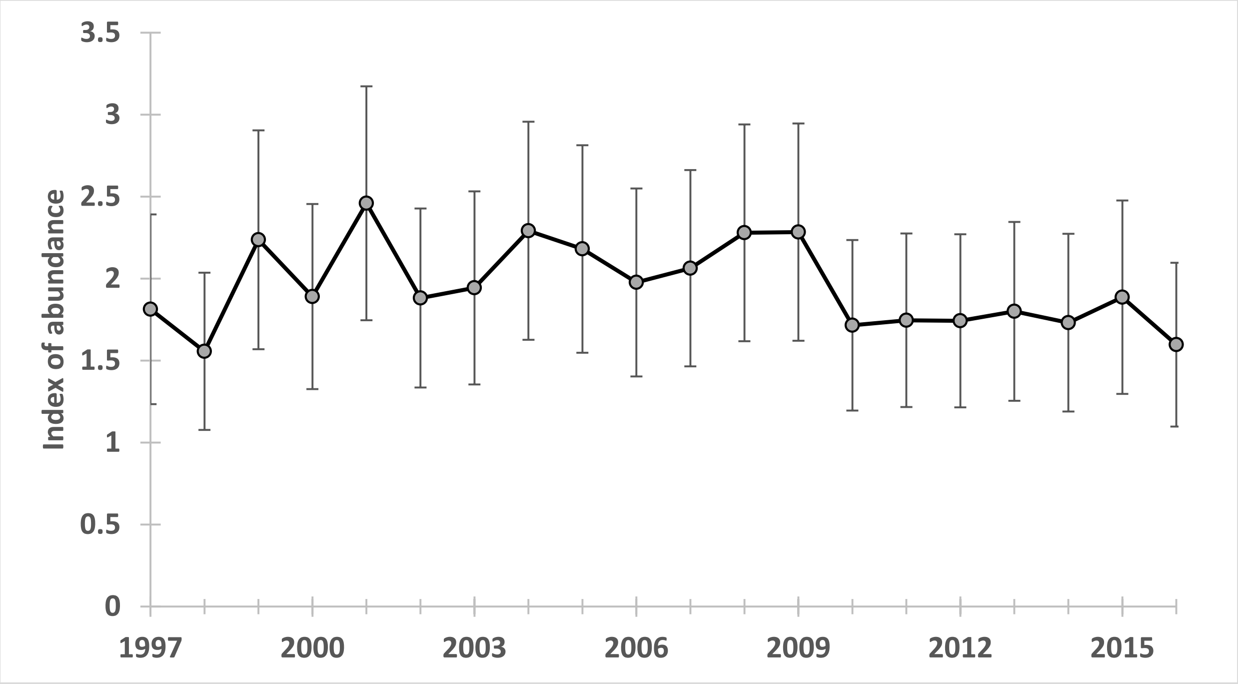

Figure 56. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1997–2016 (20-year index). The error bars represent the standard errors about the estimates.

Figure 57. Regional indices of YOY abundance for American eels. The error bars represent the standard errors about the estimates. For the South Atlantic, the standard errors were small and do not show up on the graph.

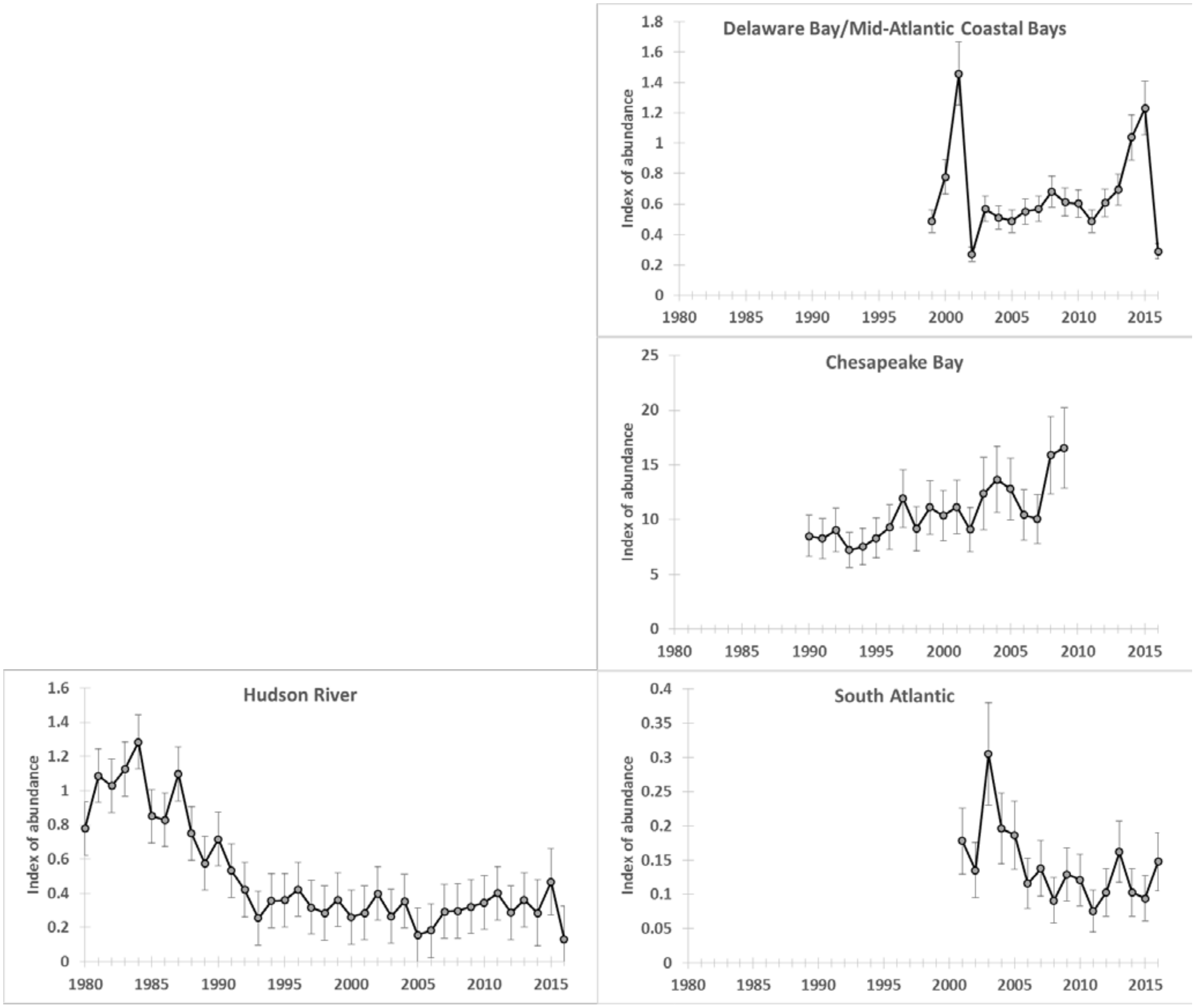

Figure 58. Regional indices of yellow-stage abundance for American eels. The error bars represent the standard errors about the estimates.

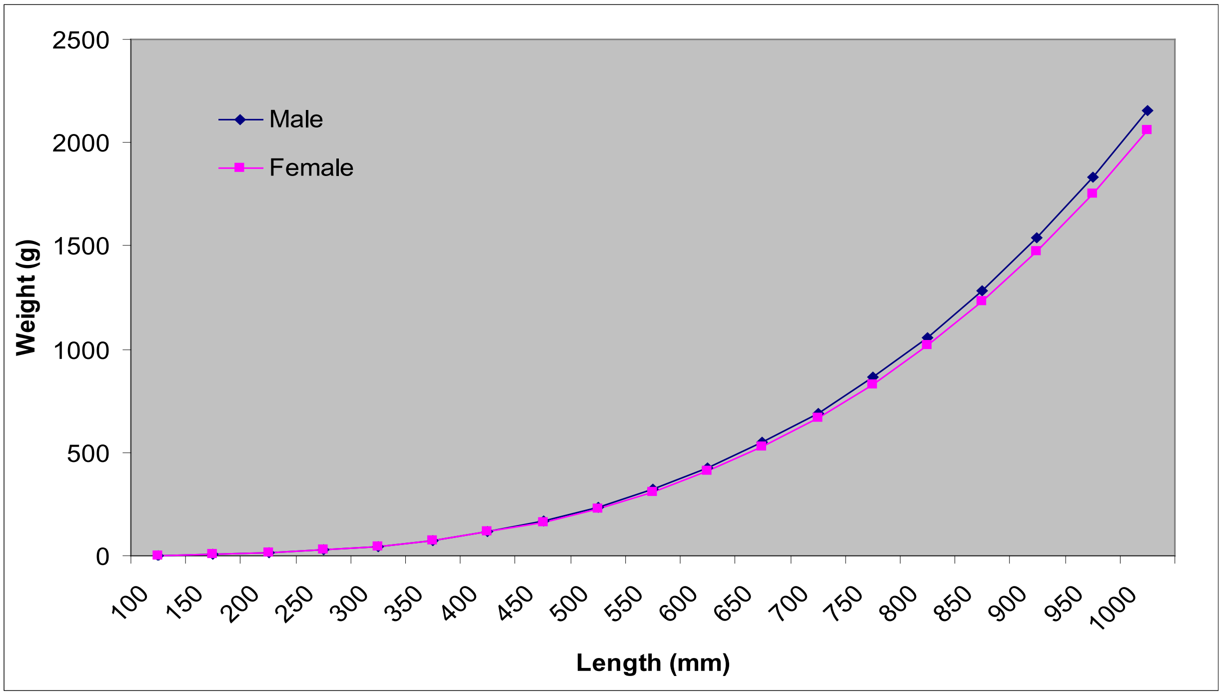

Figure 59. **Predicted total length-weight relation for American eel based on available data, by sex.**

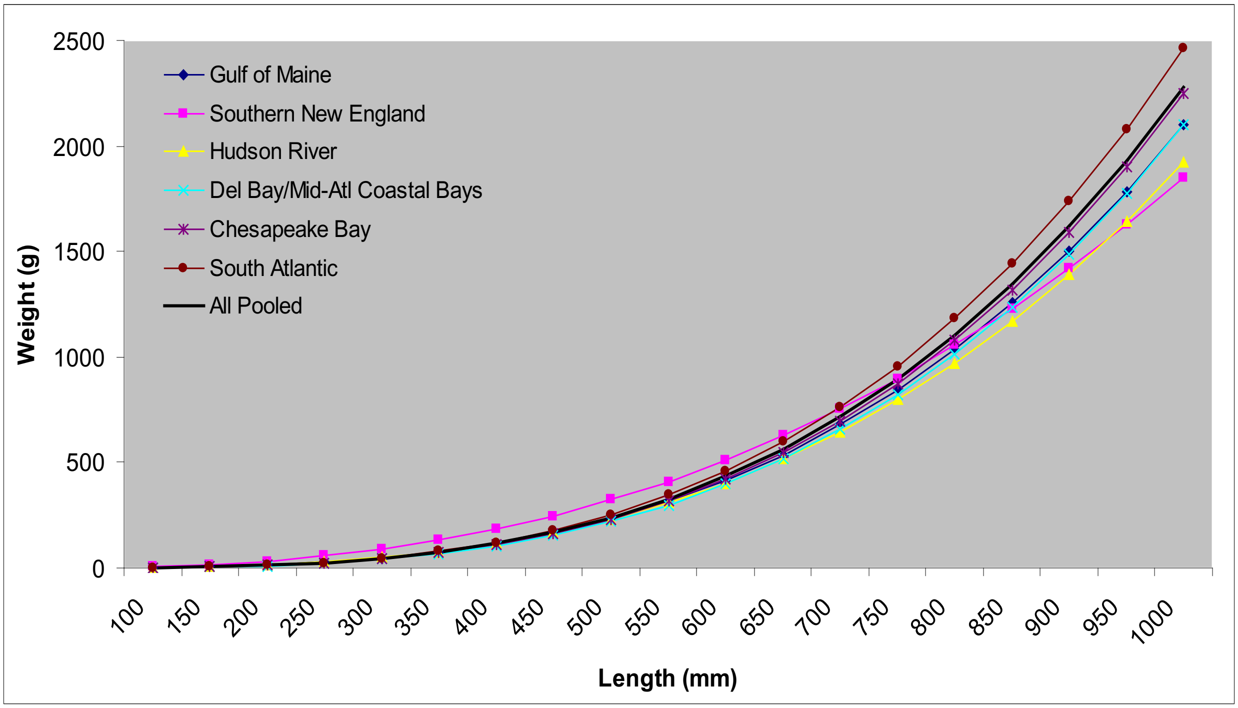

Figure 60. **Predicted total length-weight relation for American eel based on available data, by region and all pooled.**

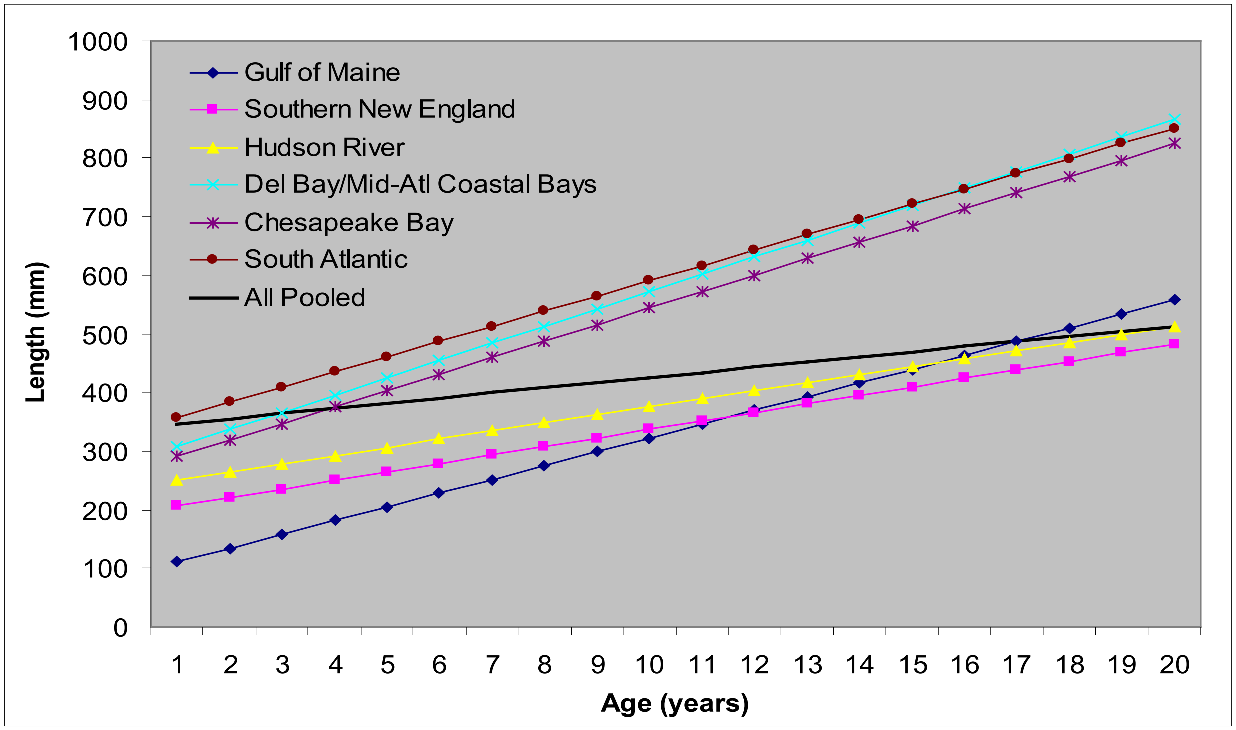

Figure 61. **Predicted linear age-length relation for American eel based on available data, by region and all pooled.**

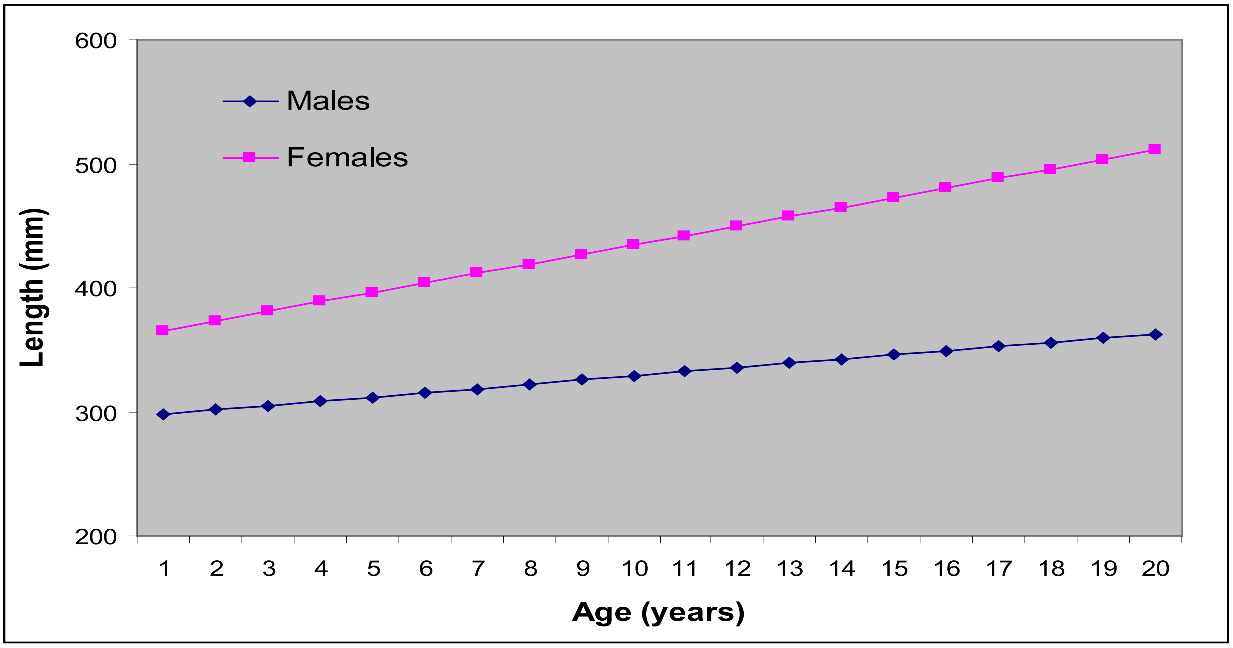

Figure 62. **Predicted linear age-length relation for American eel based on available data, by sex.**

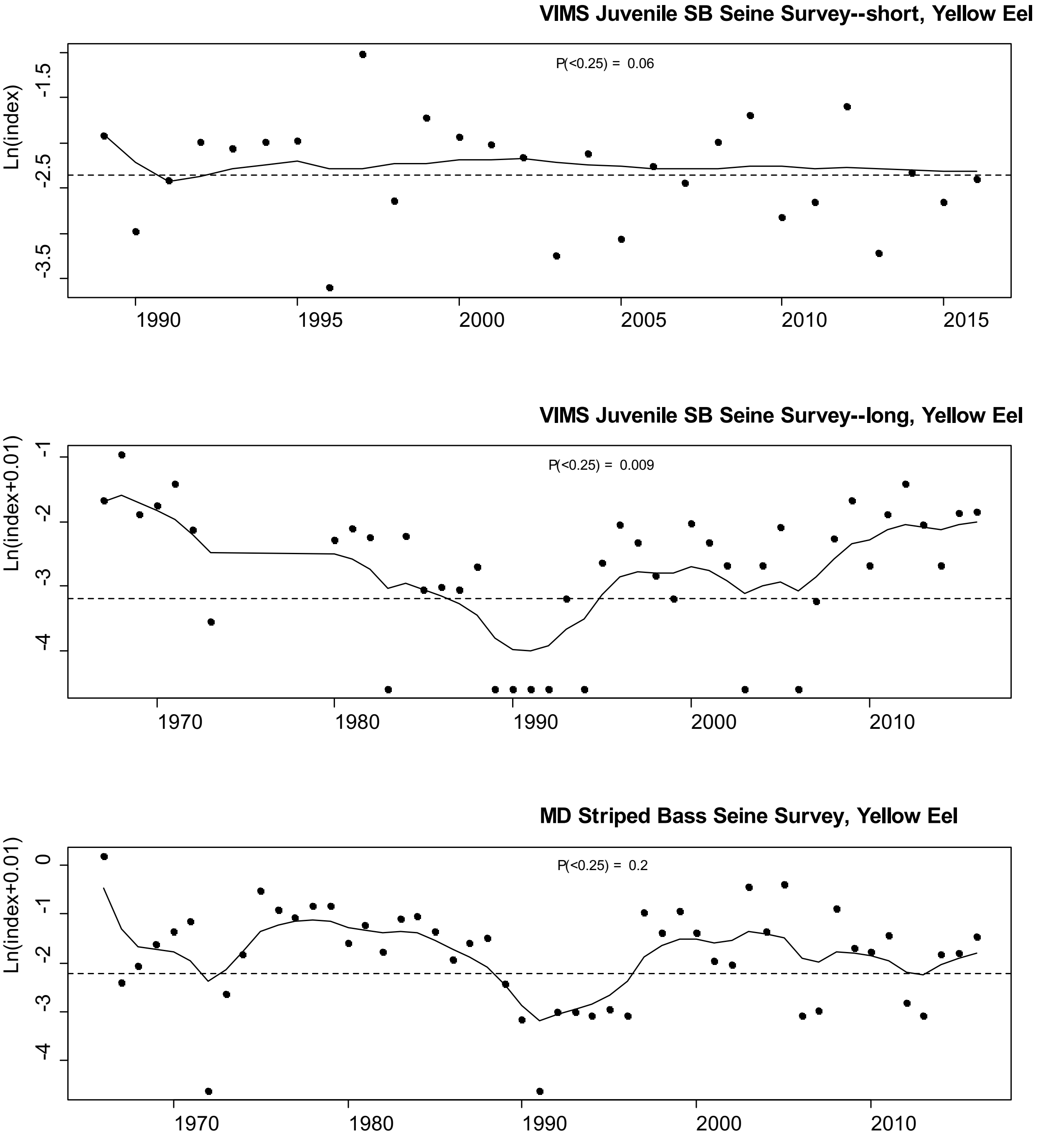

Figure 63. ARIMA model fits to American eel surveys from the Chesapeake Bay region. The dotted line represents the 25th percentile of the fitted values and *P*(<0.25) is the probability of the terminal year of the survey being less than the 25th percentile of the values.

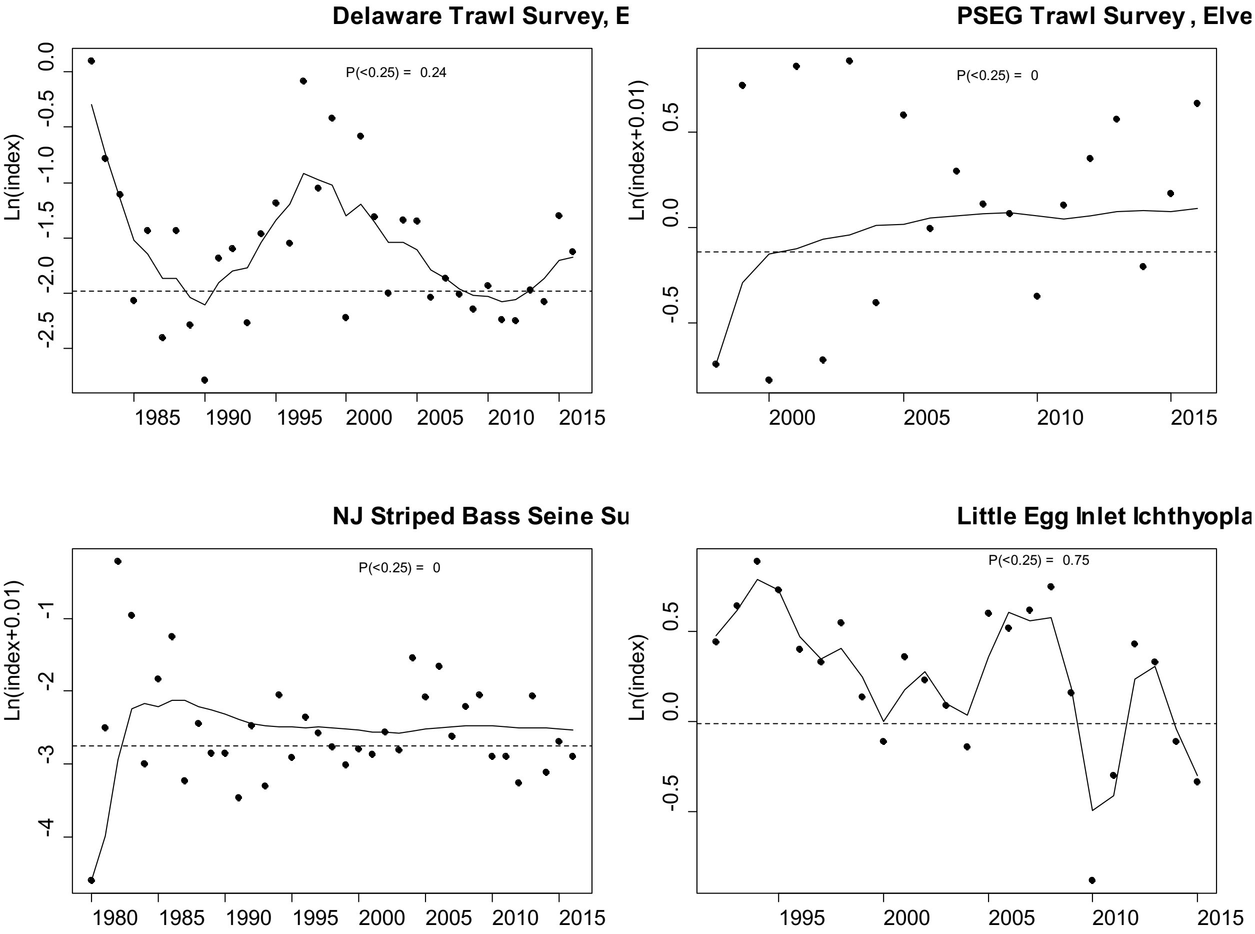

Figure 64. ARIMA model fits to American eel surveys from the Delaware Bay/Mid-Atlantic Coastal Bays region. The dotted line represents the 25th percentile of the fitted values and *P*(<0.25) is the probability of the terminal year of the survey being less than the 25th percentile of the fitted values.

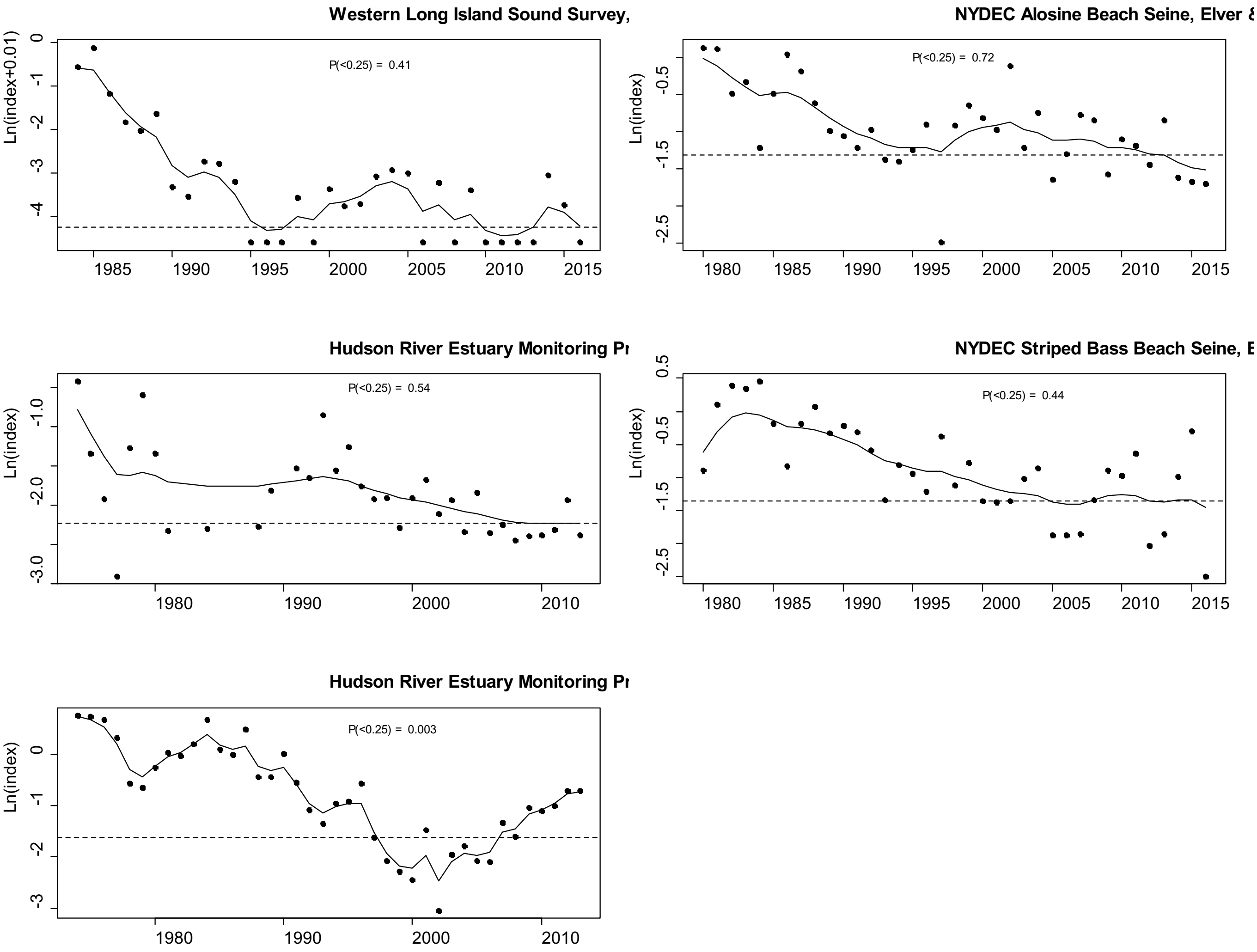

Figure 65. ARIMA model fits to American eel surveys from the Hudson River region. The dotted line represents the 25th percentile of the fitted values and *P*(<0.25) is the probability of the terminal year of the survey being less than the 25th percentile of the fitted values.

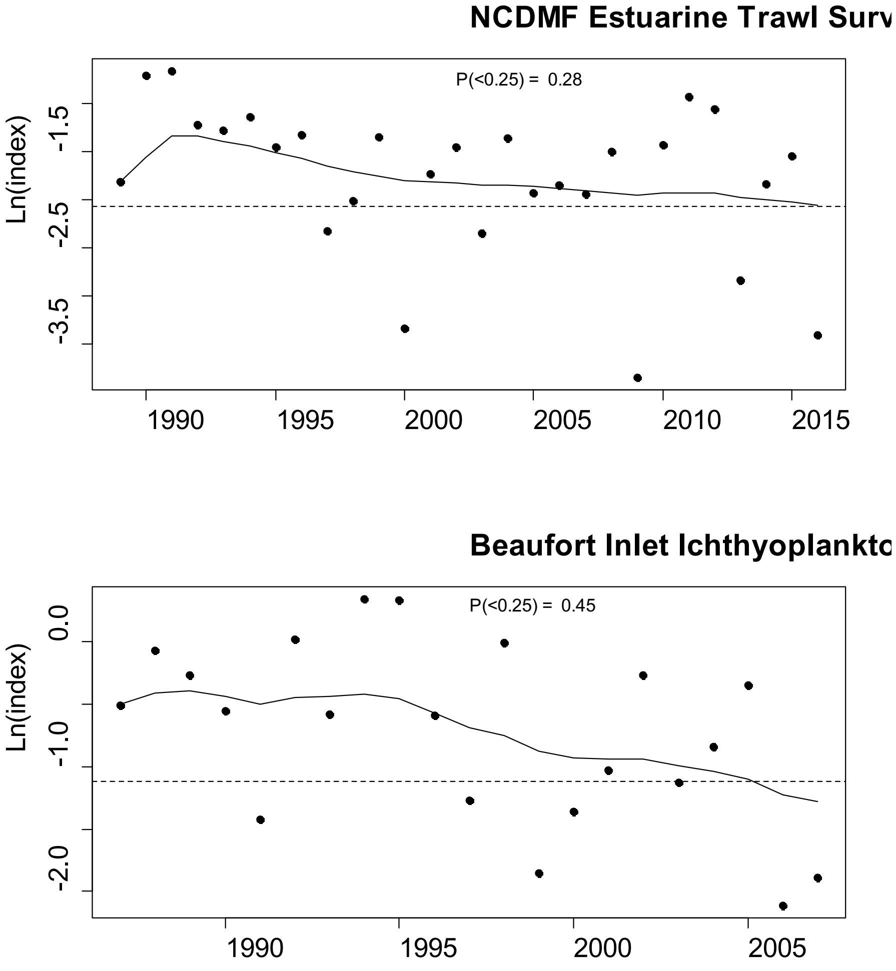

Figure 66. **ARIMA model fits to American eel surveys from the South Atlantic region. The dotted line represents the 25th percentile of the fitted values and *P*(<0.25) is the probability of the terminal year of the survey being less than the 25th percentile of the fitted values.**

Atlantic States Marine Fisheries Commission

ADDENDUM V TO THE INTERSTATE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL

Commercial Yellow and Glass/Elver Eel Allocation and Management

Approved August 2018

Revised October 2019 (to include Appendix on Coastwide Cap Policy)

Sustainable and Cooperative Management of Atlantic Coastal Fisheries

Table of Contents

1.0 Introduction

The Atlantic States Marine Fisheries Commission (Commission) has coordinated interstate management of American eel (*Anguilla rostrata*) from 0-3 miles offshore since 2000. American eel is currently managed under the Interstate Fishery Management Plan (FMP) and Addenda I-V to the FMP. Management authority in the exclusive economic zone (EEZ) from 3-200 miles from shore lies with NOAA Fisheries. The management unit is defined as the portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

This Addendum establishes a new commercial coastwide landings cap for the yellow eel fishery; new management triggers to evaluate the yellow eel coastwide cap; and a process for addressing overages and reductions if the coastwide cap is exceeded. Lastly, the Addendum outlines new criteria for evaluating glass eel aquaculture proposals.

2.0 Overview

2.1 Statement of Problem

The Commission's Interstate Fisheries Management Program Charter establishes fairness and equity as guiding principles for the conservation and management programs set forth in the Commission's FMPs. The American Eel Management Board (Board) has strived to achieve these principles through the commercial allocation program outlined in Addendum IV (2014) to the American Eel FMP. Addendum IV had set an annual commercial coastwide landings quota (referred to as the coastwide cap) of 907,671 pounds that included two management triggers:

1. The coastwide cap is exceeded by more than 10% in a given year (998,438 pounds); or
2. The coastwide cap is exceeded for two consecutive years, regardless of percent overage. Exceeding one of the two triggers would result in automatic implementation of state-by-state quotas.

Since the implementation of Addendum IV, states have raised several concerns about the current management structure, including the management trigger provision. A second-year overage, of any amount, is troublesome to some jurisdictions given the inherent uncertainty of the landings data. The FMP requires states to report commercial landings by life stage, gear type, month, and region; although not all states were able to provide this level of information for either the benchmark (2012) or updated (2017) stock assessment. In addition to not always having a complete data set to distinguish landings by life stage, there are other potential biases present in the commercial yellow eel data set including: 1. At least a portion of commercial American eel landings are from non-marine waters, and even with mandatory reporting, requirements do not always extend outside marine districts. 2. Misreporting between conger eel, hagfish, slime eel, and American eel has been known to occur. Despite these uncertainties, the commercial landings do represent the best data available and are indicative of the trend of total landings over time.

At the time of drafting the Addendum, estimated landings indicate the coastwide cap was exceeded by less than 10% in 2016. Many expressed concern that a small overage in 2017 could result in significant economic consequences for multiple jurisdictions. States also expressed concern the current coastwide cap was set independent of any ability to quantify the amount of change in landings necessary to affect fishing mortality rates and spawning stock status. Neither of those stock status elements are currently calculated for American eel due to a lack of data. In addition, states expressed concern that moving to state-specific quotas for the American eel yellow life stage fishery would create a new administrative burden. Finally, equitable allocation of this resource is particularly difficult given the variation in availability and the market demand for eels up and down the Atlantic coast.

Lastly, Addendum IV specified an annual glass eel commercial quota for Maine of 9,688 pounds for the 2015-2017 fishing seasons, and that it be re-evaluated after 3 years (prior to the start of the 2018 fishing season). In October 2017, the Board specified a glass eel commercial quota for Maine of 9,688 pounds for the 2018 fishing season. The state of Maine has expressed interest in increasing it's their glass eel quota, which requires a new addendum.

2.2 Background

American eel inhabit fresh, brackish, and coastal waters along the Atlantic, from the southern tip of Greenland to Brazil. American eel eggs are spawned and hatch in the Sargasso Sea. After hatching, leptocephali (the larval stage) are transported to the coasts of North America and the upper portions of South America by ocean currents. Leptocephali then transform into glass eels via metamorphosis. In most areas, glass eel enter nearshore waters and begin to migrate up-river, although there have been reports of leptocephali found in freshwater in Florida. Glass eels settle in fresh, brackish, and marine waters, where they undergo pigmentation, reaching the elver life stage. Elvers subsequently mature into the yellow eel phase, most by the age of two years.

The Commission's American Eel Board first convened in November 1995 and finalized the FMP for American Eel in November 1999. The goal of the FMP is to conserve and protect the American eel resource to ensure its continued role in its ecosystems while providing the opportunity for commercial, recreational, scientific, and educational uses. The FMP requires all states and jurisdictions to implement an annual young-of-year (YOY) abundance survey to monitor annual recruitment of each year's cohort. In addition, the FMP requires a minimum recreational size and possession limit and a state license for recreational harvesters to sell eels. The FMP requires that states and jurisdictions maintain existing or more conservative American eel commercial fishery regulations for all life stages, including minimum size limits. Each state is responsible for implementing management measures within its jurisdiction to ensure the sustainability of its American eel population.

Since the FMP was approved in 1999, it has been modified four times. Addendum I (approved in February 2006) established a mandatory catch and effort monitoring program for American eel. Addendum II (approved in October 2008) made recommendations for improving upstream and downstream passage for American eels. Most recently, Addendum III (approved in August 2013) made changes to the commercial fishery, specifically implementing restrictions on pigmented eels, increasing the yellow eel size limit from 6 to 9 inches, and reducing the recreational creel limit from 50 fish to 25 fish per day. In October 2014, the Board approved Addendum IV which set goals of reducing overall mortality and maximizing the conservation benefit to American eel stocks (ASMFC 2014). The Addendum established a coastwide cap of 907,671 pounds of yellow eel, reduced Maine's glass eel quota to 9,688 pounds (2014 landings), and allowed for the continuation of New York's silver eel weir fishery in the Delaware River. For yellow eel fisheries, the coastwide cap was implemented starting in the 2015 fishing year and established two management triggers: (1) if the coastwide cap is exceeded by more than 10% in a given year, or (2) the coastwide cap is exceeded for two consecutive years regardless of the percent overage. If either one of the triggers are met then states would implement state-specific allocations based on average landings from 1998-2010 with allocation percentages derived from 2011-2013.

The following objectives were addressed through Addendum V:

1. Examined Maine's glass/elver eel quota based on updated information but made no changes to the state's quota;
2. Revised the yellow eel coastwide cap and management triggers based on recent fishery performance and updated landings data, and to ensure the overarching goal of the FMP - *to conserve and protect the American eel resource to ensure its continued role in the ecosystems while providing the opportunity for its commercial, recreational, scientific, and educational use* - is met; and
3. Resolved potential inequities in allocation by removing state-by-state quotas for the yellow eel fishery.

2.3 Description of the Fishery

2.3.1 Glass Eel/Elver Fishery

Life stage glass and elver eel harvest along the Atlantic coast is prohibited in all states except Maine and South Carolina. Prior to the implementation of the FMP, Maine was the only state compiling glass eel and elver fishery catch statistics. Under the FMP, all states are now required to submit fishery-dependent information. In recent years, Maine was the only state reporting substantial glass eel or elver harvest.

Maine Glass Eel/Elver Fishery

Since the implementation of the 9,688 pound glass eel quota for Maine in 2015 through Addendum IV, landings have tracked close to the quota. In both 2016 and 2017, landings were 97% and 96% of the quota, respectively, after being much lower in 2015 (5,260 pounds).

Table 1. Maine's Glass/Elver Eel Landings 2007-2017 (Source: ACCSP)

Year	Landings	Value
2007	3,714	$1,287,479
2008	6,951	$1,486,353
2009	5,199	$514,629
2010	3,158	$592,405
2011	8,585	$7,656,345
2012	21,610	$38,791,627
2013	18,081	$32,926,991
2014	9,688	$8,440,333
2015	5,260	$11,389,891
**2016	9,399	$13,388,040
**2017	9,282	>$12,000,000

**Preliminary landings

In 2012, Maine's glass eel landings hit an all-time high of 21,610 pounds with a landed value of over $38 million. This huge spike in price per pound created a gold rush mentality that brought with it poaching problems that most thought Maine could not overcome, and there was a call to close the fishery all together. Over the next two years, the Maine Department of Marine Resources (ME DMR) responded by instituting a voluntary reduction in harvest of 35% from the 18,076 pounds that was landed in 2013. This established the first glass eel quota for Maine at 11,749 pounds. Maine instituted individual fishing quotas, and penalties were moved from civil to criminal and included a "two-strike" provision where a harvester license would be permanently revoked. Also in 2013, ME DMR developed a swipe card program that allow dealers to enter daily landings data and allow ME DMR to analyze that data within 24 hours of receipt, as well as serve as a fishery management tool to implement an individual fishing quota (IFQ) for harvesters. The Program was expanded in 2015 to include dealer-to-dealer transactions. With the implementation of Addendum IV, the elver quota was cut another 11%, reducing Maine's glass eel quota to 9,688 pounds. Since the implementation of the 9,688 pound glass eel quota, landings have tracked close to the quota with the exception of 2015 where a late spring with ice and high water contributed to a drop in landings down to 5,260 pounds.

Since 2014, ME DMR has effectively track the IFQs of approximately 900 harvesters, as well as the overall quota. In a two-year period, over 23,000 daily landings reports did not need to be key-entered by ME DMR staff due to the Swipe Card System, and only two card failures were reported. In addition, the number of fishery-related infractions reported by the Marine Patrol dropped from over 200 in 2013 to under 20 in 2014 through 2016. The addition of the dealer-to-dealer swipe card program resulted in a difference of just over 120 pounds (approximately 2%) between what dealers reported purchasing directly from harvesters to what was exported from Maine dealers in 2015. These 120 pounds is likely attributed to shrinkage (die off between initial purchases to final shipment) and did not raise concerns.

Given the high market value, poaching of glass eels and elvers is known to be a serious problem in several states. Enforcement of the regulations is challenging due to the nature of the fishery (very mobile, nighttime operation, and high value for product). However, the recent cooperation between the State's enforcement agencies and the U.S. Fish and Wildlife Service remains a high priority and has resulted in several convictions for violation of the Lacey Act.

Aquaculture

Addendum IV to the FMP also allows approved Aquaculture Plans from states and jurisdictions to harvest up to 200 pounds of glass/elver eel annually from within their state waters for use in domestic aquaculture activities. Aquaculture Plans have been approved for North Carolina since 2016 and Maine starting in 2018 (2019 fishing season).

2.3.2 Yellow Eel Fishery

Coastwide Description

Yellow eel landings have varied considerably over the years due to a combination of market trends and availability. These fluctuations are evident both within states and jurisdictions, as well as at a regional level. Such fluctuations pose significant management challenges with regard to balancing sustainable landings and access to the resource with economic considerations. Over the last 19 years, total coastwide landings have ranged from a low of approximately 717,698 pounds in 2002 to a high of approximately 1,189,455 pounds in 2011. State reported landings of yellow/silver eels in 2016 totaled 943,808 pounds (Table 2), which represent an 9% increase in landings from 2015 (868,122 pounds). 2016 yellow eel landings increased in Maine, Rhode Island, Connecticut, Maryland through Virginia, and Florida but decreased in all other states and jurisdictions.

Table 2. State-by-state Yellow Eel Landings: 1998-2016. Source: Personal Communication from State and Jurisdictions, January 2018.

Year	ME	NH	MA	RI	CT	NY	NJ	DE	MD	PRFC	VA	NC	SC	GA	FL	Total
1998	0	Time series average of less than 400 pounds	3,456	967	5,606	16,867	94,327	131,478	301,833	209,008	123,837	91,084	Time series average of less than 400 pounds	Time series average of less than 400 pounds	13,819	992,741
1999	0		3,456	140	10,250	7,882	90,252	128,978	305,812	163,351	183,255	99,939			17,533	1,011,093
2000	0		2,976	25	4,643	5,824	45,393	119,180	259,552	208,549	114,972	127,099			6,054	894,577
2001	9,007		3,867	14,357	1,724	18,192	57,700	121,515	271,178	213,440	97,032	107,070			14,218	929,523
2002	11,617		3,949	22,965	3,710	30,930	64,600	99,529	208,659	128,595	75,549	59,940			7,587	717,698
2003	15,312		4,047	24,883	1,868	8,296	100,701	155,516	346,412	123,450	121,091	172,065			8,486	1,082,614
2004	29,646		5,328	19,858	1,374	5,354	120,607	137,489	273,142	116,263	123,812	128,875			7,330	969,318
2005	17,189		3,073	22,001	337	27,726	148,127	111,200	378,659	103,628	66,956	49,278			3,913	932,087
2006	27,489		3,676	1,034	3,443	10,601	158,917	123,994	362,966	83,622	82,756	33,581			1,248	894,192
2007	14,251		2,853	1,230	935	14,881	169,902	139,647	343,141	97,361	56,512	37,937			7,379	886,470
2008	3,882		3,297	8,866	6,046	15,025	137,687	80,002	381,993	71,655	84,031	23,833			15,624	832,475
2009	2,285		1,217	4,855	435	12,676	118,533	59,619	335,575	58,863	117,974	65,481			6,824	784,420
2010	2,605		322	3,860	167	12,179	105,089	69,355	524,768	57,755	77,263	122,104			11,287	986,937
2011	2,666		368	2,038	60	36,451	120,576	92,181	715,162	29,010	103,222	61,960			25,601	1,189,455
2012	12,775		462	1,484	2,228	35,603	113,806	54,304	590,412	90,037	121,605	64,110			11,845	1,100,881
2013	4,596		2,499	2,244	546	42,845	90,244	82,991	587,872	32,290	100,379	33,980			15,059	997,052
2014	4,320		3,903	2,353	1,390	38,143	91,225	62,388	619,935	49,293	109,537	60,755			14,092	1,057,467
2015	3,559		2,255	1,538	2,271	50,194	88,828	44,708	493,043	31,588	86,715	57,791			5,632	868,122
2016	4,509		1,705	2,651	2,445	36,371	67,422	44,558	583,578	58,223	96,336	39,911			6,034	943,808

Note: Due to data confidentiality rules, annual landings for New Hampshire, South Carolina, and Georgia are not shown rather the time series landings average of less than 400 pounds.

State-by-State Descriptions
The yellow eel fishery in Maine occurs in both inland and tidal waters. Yellow eel fisheries in southern Maine are primarily coastal pot fisheries managed under a license requirement, minimum size limit, and gear and mesh size restrictions. New Hampshire has monitored its yellow eel fishery since 1980; reporting effort in the form of trap haul set-over days for pots or hours for other gears has been mandatory since 1990. Small-scale, commercial eel fisheries occur in Massachusetts and Rhode Island and are mainly conducted in coastal rivers and embayments with pots during May through November. Connecticut has a similar small-scale, seasonal pot fishery for yellow eels in the tidal portions of the Connecticut and Housatonic rivers. All New England states presently require commercial fishing licenses to harvest eels and maintain trip-level reporting.

Licensed eel fishing in New York occurs primarily in the Hudson River, the upper Delaware River (Blake 1982), and in the coastal marine district; prior to a closure starting fishing also occurred in Lake Ontario. A slot limit (greater than 9 inches and less than 14 inches to limit PCB exposure) exists for eels fished in the tidal Hudson River (from the Battery to Troy and all tributaries upstream to the first barrier), strictly for use as bait or for sale as bait only. Due to PCB contamination of the main stem, commercial fisheries have been closed on the freshwater portions of the Hudson River and its tributaries since 1976. The fishery in the New York portion of the Delaware River consists primarily of silver eels collected in a weir fishery. In 1995, New York approved a size limit in marine waters. New Jersey fishery regulations require a commercial license, a minimum mesh, and a minimum size limit. A minimum size limit was set in Delaware in 1995. Delaware mandated catch reporting in 1999 and more detailed effort reporting in 2007.

Maryland, Virginia, and Potomac River Fisheries Commission have primarily pot fisheries for American eels in Chesapeake Bay. Large eels are exported whereas small eels are used for bait in the crab trotline fishery, except in Virginia. Ninety-five percent of all American eel harvest in Virginia is by pots, and eel pots are the major pot gear. Virginia implemented a voluntary buyer reporting system in 1973 and a mandatory harvester reporting system, for all seafood species began in 1993. Since 1991, it has been mandatory that eel pots are equipped with mesh that cannot be less than one-half inch (1/2") by one-half inch (1/2"), with at least one unrestricted 4-inch by 4-inch square escape panels consisting of 1/2-inch by 1-inch mesh, regardless of pot shape. Maryland did not require licenses until 1981. Effort reporting was not required in Maryland until 1990. The Potomac River Fisheries Commission has had harvester reporting since 1964, and has collected eel pot effort since 1988.

North Carolina has a small, primarily coastal pot fishery that fluctuates with market demands. The majority of landings come from the Albemarle Sound area, with additional landings reported from the Pamlico Sound and “other areas.” No catch records are maintained for freshwater inland waters, and no sale of eels harvested from these waters is permitted. Landings for “other areas” reported by the state come from southern waterbodies under the jurisdiction of North Carolina Department of Marine Fisheries. South Carolina instituted a permitting system over ten years ago to document total eel gear and commercial landings. Pots and traps are permitted in coastal waters for the yellow eel life stage fishery; fyke nets and dip nets are permitted for glass eels.

American eel fishing in Georgia was restricted to coastal waters prior to 1980 when inland fishing was permitted (Helfman et al. 1984). Landings data are available for the state, but effort data is currently not. The state implemented a new specific license endorsement to fish eels in 2017. The Florida pot fishery has a minimum mesh size requirement in the fishery and it is operated under a permit system.

2.4 Status of the Stock

The last peer reviewed and accepted benchmark stock assessment was approved for management use in 2012. Analyses and results indicated the American eel stock had declined and there were significant downward trends in multiple surveys across the coast. It was determined the stock was depleted but no overfishing determination could be made based on the analyses performed.

The 2012 benchmark stock assessment was updated in 2017 with data through 2016. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant downward trends in some indices. The Mann-Kendall test detected a significant downward trend in six of the 22 YOY indices, 5 of the 15 yellow eel indices, 3 of the 9 regional indices, and the 30-year and 40-year yellow-phase abundance indices. The remaining surveys tested had no trend, except for two which had positive trends. The Manly meta-analysis showed a decline in at least one of the indices for both yellow and YOY life stages. For the ARIMA results, the probabilities of being less than the 25th percentile reference points in the terminal year for each of the surveys were similar to those in the 2012 benchmark assessment and currently three of the 14 surveys in the analysis have a greater than 50% probability of the terminal year of each survey being less than the 25th percentile reference point. Overall, the occurrence of some significant downward trends in surveys across the coast remains a cause for concern and the assessment maintained the stock remains depleted.

3.0 Management Program

3.1 Maine Glass Eel Quota

The Maine glass eel quota is set at 9,688 pounds. This quota level was specified based on the state's 2014 landings. The following components of Addendum IV's commercial glass/elver eel fishery management program remain unchanged:

- **Quota Overages**: For any state or jurisdiction managed with a commercial glass/elver eel quota, if an overage occurs in a fishing year, that state or jurisdiction will be required to deduct their entire overage from their quota the following year, on a pound for pound basis.

- **Reporting Requirements**: Any state or jurisdiction with a commercial glass eel fishery is required to implement daily trip-level reporting with daily electronic accounting to the state for both harvesters and dealers in order to ensure accurate reporting of commercial glass eel harvest. The State of Maine's swipe card system is used by the state as a dealer report. Harvesters in Maine are currently reporting monthly via paper report submission. States or jurisdictions commercially harvesting less than 750 pounds of glass eels are exempt from this requirement.

- **Monitoring Requirements**: Any state or jurisdiction with a commercial glass eel fishery must implement a fishery-independent life cycle survey covering glass/elver, yellow, and silver eels within at least one river system. If possible and appropriate, the survey should be implemented in the river system where the glass eel survey (as required under Addendum III) is being conducted to take advantage of the long-term glass eel survey data collection. At a minimum the survey must collect the following information: fishery-independent index of abundance, age of entry into the fishery/survey, biomass and mortality of glass and yellow eels, sex composition, age structure, prevalence of *Anguillicoloides crassus (*invasive nematode*)*, and average length and weight of eels in the fishery/survey. Survey proposals will be subject to Technical Committee (TC) review and Board approval. States or jurisdictions commercially harvesting less than 750 pounds of glass eels are exempt from this requirement.

- **Glass Eel Harvest Allowance Based on Stock Enhancement Programs**: Any state or jurisdiction can request an allowance for commercial harvest of glass eels based on stock enhancement programs implemented after January 1, 2011, subject to TC review and Board approval. Provisions of the stock enhancement program include: demonstration that the program has resulted in a measurable increase in glass eel passage and/or survival; harvest shall not be restricted to the basin of restoration (i.e. harvest may occur at any approved location within the state or jurisdiction); and harvest requests shall not exceed 25% of the quantified contribution provided by the stock enhancement program. See Addendum IV for more detail on specific stock enhancement program examples.

3.2 Glass Eel Aquaculture Plan Provisions

The Aquaculture Plan proposal requirements have been modified based on the following criteria (as recommended by the TC):

States and jurisdictions may develop a Plan for aquaculture purposes. Under an approved Aquaculture Plan, states and jurisdictions may harvest a maximum of 200 pounds of glass eels annually from within their waters for use in domestic aquaculture facilities. Site selection for harvest will be an important consideration for applicants and reviewers. Suitable harvest locations will be evaluated with a preference to locations that have:

1. Established or proposed glass eel monitoring;
2. Are favorable to law enforcement; and
3. Watershed characteristics that are prone to relatively high mortality rates.

Watersheds known to have features (ex. impassible dams, limited area of upstream habitat, limited water quality of upstream habitat, and hydropower mortality) that would be expected to cause lower eel productivity and/or higher glass eel mortality will be preferred targets for glass eel harvest. This is not an exclusive requirement, because there will be coastal regions with interest in eel aquaculture where preferred watershed features do not occur or are not easily demonstrated. In all cases, the applicant should demonstrate the above three interests were prioritized and considered.

The following components of Addendum IV's Aquaculture Plan provisions remain unchanged:

- Approval of a request does not guarantee approval of a request in future years. Eels harvested under an approved Aquaculture Plan may not be sold until they reach the legal size in the jurisdiction of operations, unless otherwise specified.

- All Plans are subject to TC and Law Enforcement Committee review and Board approval. The Fishing Mortality Based Plan must be submitted by June 1st of the preceding fishing year in order to provide enough time for review for the upcoming fishing season. Transfer and Aquaculture Plans must be submitted by June 1st of the preceding fishing year and approval will be determined by the Board by September 1st. Plans will initially be valid for only one year. After the first year of implementation the TC will evaluate the program and provide recommendations to the Board on the overall impact of and adherence to the plan. If the proposed regulatory changes, habitat improvements, or harvest impact cannot be assessed one year post-implementation, then a secondary review must occur within three to five years post-implementation if the action is still ongoing. If states use habitat improvements and changes to that habitat occurs in subsequent years, the Commission must be notified through the annual compliance report and a review of the Plan may be initiated. Any requests that include a stocking provision would have to ensure stocked eels were certified disease free according to standards developed by the TC and approved by the Board.

3.3 Yellow Eel Coastwide Cap, Management Trigger, and Allocation

3.3.1 Yellow Eel Coastwide Landings Cap

The coastwide yellow eel landings cap is 916,473 pounds, which is the coastwide average landings during the years of 1998 through 2010 (based on revised landings information through 2016 as of January 2018). This timeframe was also the period covered by the 2012 benchmark stock assessment.

3.3.2 Yellow Eel Coastwide Cap Management Trigger

Starting in 2019, the coastwide landings are annually evaluated against a two-year management trigger. If the coastwide cap is exceeded by 10% (10% of the coastwide cap = 91,647 pounds; coastwide cap + 10%= **1,008,120 pounds**) for two consecutive years, the Board is required to alter the management program as specified below to ensure the objectives of the management program are achieved.

3.3.3 Allocation

The yellow eel fishery is managed without state-specific quotas through adaptive management. If the management trigger is tripped, only states with landings greater than 1% of the coastwide landings, in the year(s) when the management trigger is tripped, will be responsible for reducing their landings to achieve the coastwide cap in the subsequent year. States with landings greater than 1% of the coastwide landings will work collectively to achieve an equitable reduction to the coastwide cap. For states with landings less than 1% of the coastwide landings, if in subsequent years a state's landings exceeds 1% of the coastwide landings after reductions have been applied, that state must reduce their individual state landings in the subsequent year to return to the less than 1% level.

More details on the process the Management Board will undertake to respond to overages of the coastwide cap are outlined in Appendix I.

3.4 Timeframe for Addendum Provisions

Specific to the Maine glass eel quota of 9,688 pounds, the quota level will be set for three years moving forward (starting in the 2019; from 2019-2021), and can be revisited before year four (2022). If the Board decides to maintain Maine's glass eel quota at 9,688 pounds, the quota can be extended for an additional three years (2022-2024) without requiring a new addendum. If there is a desire to increase Maine's glass eel quota from the specified level above, a new addendum will be required.

All other management provisions will remain in place until a new or different management program implemented through the Commission management process.

4.0 Compliance

The implementation deadline for this Addendum is January 1, 2019. Starting January 1, the yellow eel coastwide cap will be 916,473 pounds and the management trigger will be two years of exceeding coastwide cap by 10% (1,008,120 pounds).

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Appendix

Policy to Address Coastwide Cap Overages for the Yellow Eel Commercial Fishery

This appendix is intended to provide guidance to the Board in the event that the coastwide cap of 916,473 pounds of American eel is exceeded in a given year. Sections 3.3.2 and 3.3.3 of this Addendum state the following regarding the management trigger and the response:

3.3.2 Yellow Eel Coastwide Cap Management Trigger
Starting in 2019, the coastwide landings are annually evaluated against a two-year management trigger. If the coastwide cap is exceeded by 10% (10% of the coastwide cap = 91,647 pounds; coastwide cap + 10% = 1,008,120 pounds) for two consecutive years, the Board is required to alter the management program as specified below to ensure the objectives of the management program are achieved.

3.3.3 Allocation
The yellow eel fishery is managed without state-specific quotas through adaptive management. If the management trigger is tripped. Only states with landings greater than 1% of the coastwide landings, in the year(s) when the management trigger is tripped, will be responsible for reducing their landings to achieve the coastwide cap in the subsequent year. States with landings greater than 1% of the coastwide landings will work collectively to achieve an equitable reduction to the coastwide cap. For states with landings less than 1% of the coastwide landings, if in subsequent years a state's landings exceeds 1% of the coastwide landings after reductions have been applied, that state must reduce their individual state landings in the following year to return to the less than 1% level[1]*.*

A management objective under this Addendum is to manage landings to the coastwide cap (cap). Annual landings are not finalized until the spring of the following fishing year. Therefore, if an overage occurs, a year lag time will likely occur before full action is taken to reduce harvest to the cap. For example, a cap overage in 2019 would not be determined until 2020, and action would likely be delayed until 2021 since some states do not have authority to act within the same fishing year when the overage is determined.

One way to proactively manage the yellow eel fishery is to closely monitor landings and encourage states to take voluntary action when it is clear an overage has occurred in the previous year. By engaging with states before the management trigger is tripped, but after landings have exceeded the cap, a lengthy addendum process can be avoided and more immediate action can be taken to ensure the fishery is managed to the cap. This proactive approach encourages vigilance and voluntary action in the first year of an overage, and provides opportunity for collaborative, rapid action to prevent an overage in the second consecutive year, thereby preventing the triggering of mandatory management action through an addendum.

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[1] To clarify, reduction measures apply when the management trigger is tripped. States are not held to a landings level until coastwide landings have exceeded the coastwide cap.

Thus, to improve the expediency in reacting to an overage, it is recommended that preliminary commercial yellow eel landings from the ACCSP Data Warehouse be made available for the Board's consideration prior to the ASMFC Spring Meeting, annually. Based on the preliminary data review, if it's determined the cap has likely been exceeded in one year the Board will convene a work group (WG) consisting (at a minimum) of one representative from each state/jurisdiction that harvested more than 1% of the coastwide landings in the year of the overage. The charge of the WG is to consider the overage relative to the decision trees (Figure 1) and determine if and how the Board should recommend voluntary action by those states that harvested more than 1% of the coastwide landings (1% states).

Response Strategy When Cap is exceeded in One Year

Once convened by the Board, the WG will review the magnitude and the pattern of the overage relative to the decision trees (Figures 1-3) to determine the need for voluntary action. "Pattern" refers to whether landings of American eel increased in all states or in some states while harvest decreased in others. "Magnitude" refers to the extent of the overage and, for individual states, the amount of harvest increase relative to the previous year. It will be important for the WG to examine potential reasons for increasing harvest, such as increased effort, increased availability of eels, improved market conditions, etc. Once the Board recommends states decrease landings it will be up to the states to take action.

States may utilize (but are not restricted to) the following voluntary methods to reduce eel harvest as considered by the Board in Draft Addendum II (2007):

- Seasonal restrictions,
- Gear limits, and
- Size limits.

Note: Harvest reductions were not approved by the Board and were not included in Addendum II (2008).

Seasonal restrictions are the simplest method of reducing harvest, but there was strong opposition to the seasonal restrictions from the Advisory Panel when proposed in Draft Addendum II. However, those seasonal closures were designed to increase escapement of silver eels and occurred in the fall during times of maximal fishing effort, so it is conceivable that a seasonal closure could be designed that would reduce harvest without imposing a severe hardship on the fishery. The Board considered a maximum size limit as a method to allow more escapement of silver eels and increase eggs-per-recruit (EPR). A range of size limits were presented in the Draft Addendum ranging from a 19" maximum size limit, which was estimated to increase EPR by 138%, but at a reduction of 40% to the harvest, to a 23" maximum size, which only increased EPR by 3.8% and reduced harvest by less than 10%. A larger minimum size also will reduce harvest if harvest reduction is the sole goal. Size limits could either be enforced by gear modifications or by grading the eels on the water. Gear modifications can impose a large financial burden on harvesters, depending on the number of pots fished and length limit. If a minimum length is used, eel pots can be modified by installing an escape panel of a mesh size that would only retain eels above the minimum length. If a maximum eel length is used, the funnel(s) on the eel pots can be modified by restricting the circumference. A grader can also be used to comply with length limits at a lower cost to the harvesters than gear modification. Grader bars can be set to pass all eels below a minimum length or to hold all eels above a maximum length. Although the Advisory Panel favored grading for complying with a maximum length limit during the Draft Addendum II deliberations, the Law Enforcement Committee thought on-water enforcement of the length limit by grading would be difficult.

Response Strategy if the Two-Year Management Trigger is Tripped

If a review of landings at the Commission's Spring Meeting indicate the two-year management trigger has been met, the Board will initiate an addendum to reduce landings to or below the cap. A Plan Development Team (PDT) will be convened to draft the addendum (Table 1). The PDT will consider a variety of actions to reduce harvest back to the cap, including but not limited to: (1) an equal percent reduction taken only from the 1% states whose harvest increased in the overage year(s); (2) an equal percent reduction taken from all 1% states regardless of whether their harvest increased or decreased; (3) each 1% state takes a base reduction that is less than the total reduction needed, and the remainder of the reduction is taken only by those 1% states who had substantially increased harvest leading up to the overage year. The PDT should consider the impacts of calculating a reduction in harvest from a single overage year, the 2 years over which the trigger was reached or from a baseline within the last 5 years using a maximum of 3 years that ensures equitable reductions.

Once action is taken to reduce harvest to the cap (either voluntary after the first year of an overage or required after the management trigger is tripped), actions will remain in place until the coastwide harvest returns to a level that is at or below the cap. At this point, states may propose adjustments to the Board recognizing the process will begin again if another year's overage occurs or a management action is enacted.

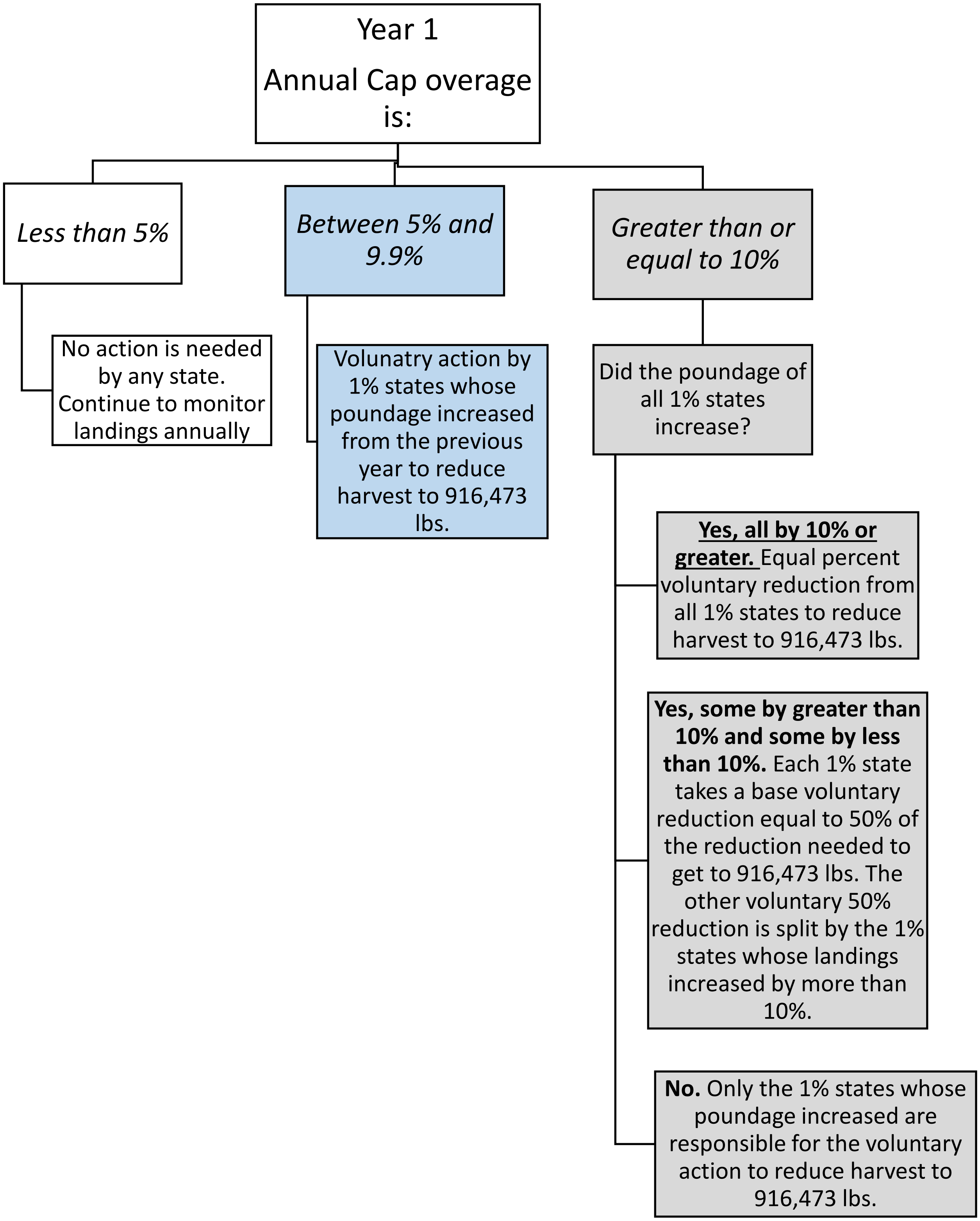

Figure 1. Decision Tree for Management Response to Cap Overage in Year 1

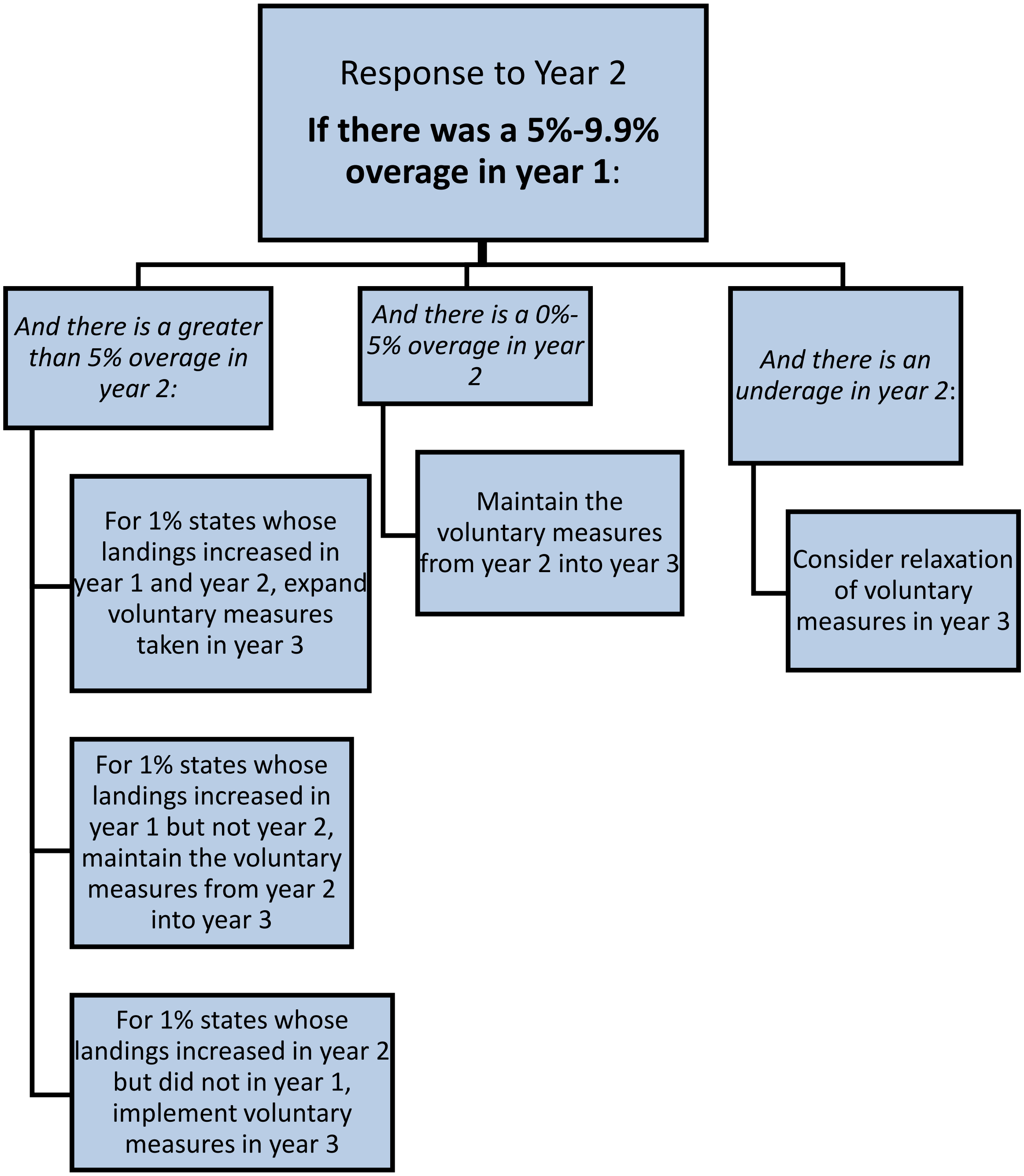

Figure 2. Decision Tree for Management Response in Year 3 if Overage is less than 10% in Year 1

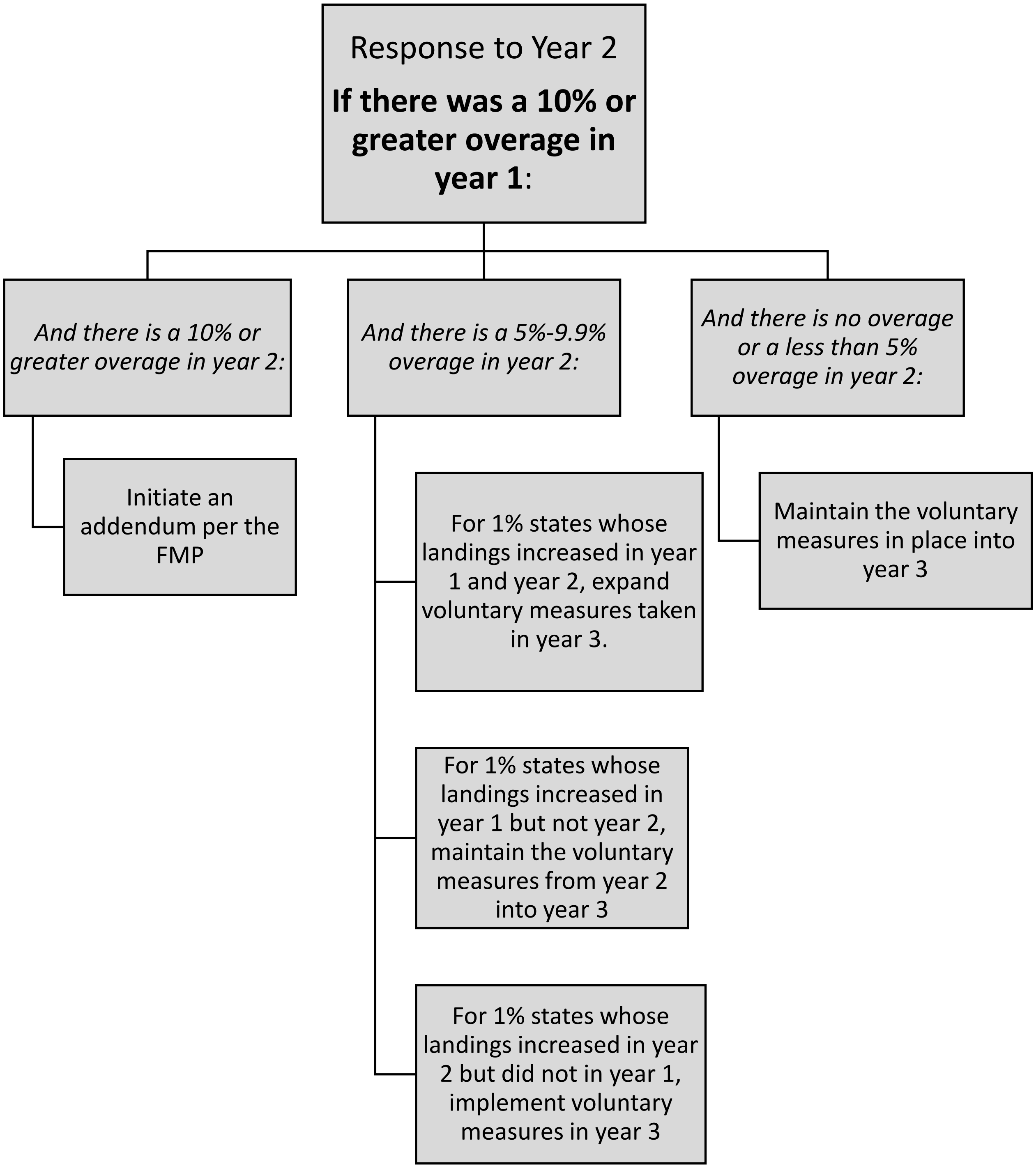

Figure 3. Decision Tree for Management Response in Year 3 if Overage is more than 10% in Year 1

Appendix Table 1. Example Timeline if Two Year Management Trigger is Tripped Based on Decision Trees

Date	Action
Spring 2020	Board review 2019 landings. It is determined an overage => 10% of the cap occurred. Board convenes workgroup (WG).
Summer 2020	WG reviews the overage relative to decision trees and develops report with recommended action for Board consideration.
August 2020	Board considers WG report and recommends states take voluntary action as soon as possible. Voluntary measures are implemented as soon as possible for 2020 fishing year.
Spring 2021	Board reviews 2020 landings. It is determined an overage =>10% of the cap occurred. **Management trigger tripped**. Board initiates Addendum.
Summer 2021	Staff and PDT develop Draft Addendum.
August 2021	Board approves Draft Addendum for public comment.
Fall 2021	Public comment period for Draft Addendum.
October 2021	Board finalizes and approves Addendum.
January 2022	Addendum implemented.

Atlantic States Marine Fisheries Commission

Healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015.

Addendum I

To the Interstate Fishery Management Plan for American Eel

Approved February 22, 2006
Edited on January 5, 2007 to Reflect February 2006 American Eel Management Board Proceedings

CH 2 ASMFC 2006a

INTRODUCTION

Complete catch and effort data is needed in order to properly assess and manage the American eel population. The 2006 peer review highlighted a lack of eel catch and effort data as a major impediment to completing a quantitative stock assessment. To collect the necessary data for future stock assessments, the American Eel Technical Committee has recommended that, at a minimum, states be required to provide accurate catch and effort data. Specifically, the Technical Committee has recommended implementation of a specific eel harvester permit/license for each state, with each license requiring reporting of catch and effort. The permit/license should be required for all eel harvesters, including those who harvest eels for use as bait. Further, the Technical Committee has recommended a specific eel report and license/permit from dealers, including bait dealers. Harvester and dealer reports must differentiate between the amount of eels used or sold for food (human consumption) and the amount of eels used or sold for bait.

Based on these recommendations, the American Eel Management Board developed and subsequently approved this Addendum in order to establish a mandatory catch and effort monitoring program for American eel. The management measures in this Addendum are to be implemented by January 1, 2007 (states that are required to implement new management measures through legislation are permitted an addition six months to implement the Addendum).

Background

American eel inhabit fresh, brackish, and coastal waters along the Atlantic from the southern tip of Greenland to northeastern South America. The species is catadromous, spawning only in the Sargasso Sea and then migrating toward land and into freshwater, where it spends the majority of its life. After hatching and ocean drift, initially in the pre-larval stage and then in the leptocephalus phase, metamorphosis occurs. In most areas, glass eel enter the nearshore area and begin to migrate up-river, although there have been reports of leptocephali found in freshwater in Florida. Eel are found in the marine environment during various parts of their life cycle. Elvers, yellow eel, and silver eel make extensive use of freshwater systems. Therefore, a comprehensive eel management plan and set of regulations must consider the various unique life stages and the diverse habitats of American eel, in addition to society's interest and use of this resource.

American eel (*Anguilla rostrata*) occupy a significant and unique niche in the Atlantic coastal reaches and its tributaries. Historically, American eel were very abundant in East Coast streams, comprising more than 25 percent of the total fish biomass. Eel abundance declined from historic levels but remained relatively stable until the 1970s. More recently, fishermen, resource managers, and scientists postulated a further decline in abundance based on harvest information and limited assessment data. This resulted in the development of the Atlantic States Marine Fisheries Commission (ASMFC) Interstate Fishery Management Plan (FMP) for American Eel. The goals of the FMP are:

1. Protect and enhance the abundance of American eel in inland and territorial waters of the Atlantic states and jurisdictions, and contribute to the viability of the American eel spawning population; and
2. Provide for sustainable commercial, subsistence, and recreational fisheries by preventing over-harvest of any eel life stage.

In support of this goal, the following objectives were included in the FMP:

- Improve knowledge of eel utilization at all life stages through mandatory reporting of harvest and effort by commercial fishers and dealers, and enhanced recreational fisheries monitoring.

- Increase understanding of factors affecting eel population dynamics and life history through increased research and monitoring.
- Protect and enhance American eel abundance in all watersheds where eel now occur.
- Where practical, restore American eel to those waters where they had historical abundance but may now be absent by providing access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel.
- Investigate the abundance level of eel at the various life stages necessary to provide adequate forage for natural predators and support ecosystem health and food chain structure.

Status of the Stock

Current stock status for American eel is poorly understood due to limited and non-uniform stock assessment efforts and protocols across the range of this species. Reliable indices of abundance of this species are scarce. Limited data from indirect measurements (harvest by various gear types and locations) and localized stock assessment information are currently collected.

Although eel have been continuously harvested, consistent data on harvest are often unavailable. Harvest data are often a poor indicator of abundance because harvest is dependent on demand and may consist of annually changing mixes of year classes. Most of the data collections were of short duration and were not standardized between management agencies. Harvest data from the Atlantic coastal states (Maine to Florida) indicate that harvest has declined after a peak in the mid-1970s. Annual eel catch ranged from 913,251 pounds to 3,626,936 pounds between 1970 and 2000. The lowest harvest (between 1970 and 2001) was 898,459 pounds and occurred in 2001. Because fishing effort data is unavailable, finding a correlation between population numbers and landings data is problematic.

As stated in Section 2 of the FMP, the purpose of this management program is to reverse any local or regional declines in abundance and institute consistent fishery-independent and dependent monitoring programs throughout the management unit.

In 2003, declarations from the International Eel Symposium (AFS 2003, Quebec City, Quebec, Canada) and the Great Lakes Fishery Commission (GLFC) highlighted concerns regarding the health of American eel stock. Available data point to decreasing recruitment combined with localized declines in abundance. This information is cause for concern and represents an opportunity for cooperation with other entities such as the GLFC to preserve the American eel stock.

In 2005, the ASMFC American Eel Stock Assessment Subcommittee (SASC) conducted a stock assessment for American eel. This assessment was reviewed by the ASMFC American Eel Technical Committee and underwent an independent peer review in December 2005. The results of the peer review can be found on the Atlantic States Marine Fisheries Commission website, www.asmfc.org.

Status of the Fishery

American eel currently support important commercial fisheries throughout their range. Fisheries are executed in rivers, estuaries, and the ocean. Commercial fisheries for glass eel and elver exist in Maine, South Carolina, and Florida (though in South Carolina and Florida, no commercial glass eel or elver landings were recorded in 2004), whereas yellow and silver eel fisheries exist in all states and jurisdictions with the exception of Pennsylvania and the District of Columbia.

Commercial

Commercial landings decreased from the high of 1.8 million pounds in 1985 to a low of 649,000 pounds in 2002. Landings in 2004 totaled 921,896 pounds. The States of New Jersey, Delaware, Maryland, Virginia, and North Carolina each landed over 100,000 pounds of eel, and together accounted for 88 percent of the coastwide commercial total landings in 2004.

Recreational

Few recreational anglers directly target eel. Hook and line fishermen, for the most part, catch eel incidentally when fishing for other species. The National Marine Fisheries Service (NMFS) Marine Recreational Fisheries Statistics Survey (MRFSS), which has surveyed recreational catch in ocean and coastal county waters since 1981, shows a declining trend in the catch of eel during the latter part of the 1990s. According to MRFSS[1], 2004 recreational catch was 112,001 fish, which represents a slight decrease in number of fish from 2003 (156,381 fish). New Jersey and Delaware combined represented 40 percent of the recreational American eel catch, and New York and Delaware combined represented 62 percent of the recreational American eel harvest in 2004. About 79 percent of the eel caught were released alive by anglers in 2004 (MRFSS 2004; total recreational harvest was 23,442 fish). Eel are often purchased by recreational fishermen for use as bait for larger gamefish such as striped bass, and some recreational fishermen may catch eels and utilize them as bait.

STATEMENT OF THE PROBLEM

The American Eel FMP includes a requirement for states to institute licensing and reporting mechanisms to ensure that annual effort (including total units of gear deployed) and landings information by life stage (glass eel/elver, yellow eel, and silver eel) are provided by harvesters and/or dealers. The stock assessment also recommends improved catch and effort reporting for improvement of future stock assessments. In addition, the ACCSP calls for a comprehensive permit/license system for all commercial dealers and fishermen.

The FMP requires states to report the following information each year:

Commercial Fishery
- Estimates of directed harvest, by month, by region as defined by the states
  - Pounds landed by life stage and gear type (defined in advanced by ASMFC)
  - Biological data taken from representative sub-samples to include sex ratio and age structure (for yellow/silver eels), length, and weight, if available
  - Estimated percent of harvest going to food versus bait
- Estimates of export by season (provided by dealers)
- Harvest data provided as CPUE (by life stage and gear type)
- Permitted catch for personal use, if available

Recreational Fishery
- Estimate of recreational harvest by season, if available
  - Biological data taken from representative sub-samples to include sex ratio, age structure, length, and weight, if available

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[1] MRFSS data for American eel are unreliable. 2004 Proportional Standard Error (PSE) values for recreational harvest in Connecticut, New York, New Jersey, Delaware, Maryland, and North Carolina are 100, 74.1, 100, 47.3, 83.5, and 100, respectively.

The 2005 stock assessment for American eel was still in draft form during development of this Addendum. An independent peer review of the stock assessment was held in December 2005. In the stock assessment, the Stock Assessment Subcommittee recommended the following for improving future stock assessments:

- Improve catch and effort monitoring by requiring trip-level landing and effort data by state.
- Require states to report catch and effort in standardized units.
- Require effort to be reported by gear type, the number of units of gear fished per person per trip, including soak time or fishing time. States should be required to report these effort data annually.
- Require states to implement commercial eel harvest and dealer permits as a measure of participation.

The ACCSP commercial data collection program includes a trip-based system with all fishermen and dealers required to report a minimum set of standard data elements (refer to the ACCSP Program Design Document for details). Commercial fishermen and dealer reports should be submitted after the 10th of each month.

Any marine fishery products landed in any state must be reported by a dealer or a marine resource harvester acting as a dealer in that state. Any marine resource harvester or aquaculturist who sells, consigns, transfers, or barters marine fishery products to anyone other than a dealer would themselves be acting as a dealer and would therefore be responsible for reporting as a dealer.

The ACCSP recreational data collection program for private/rental and shore modes of fishing is conducted through a combination telephone and intercept survey. Recreational effort data are collected through a telephone survey with random sampling of households until such time as a more comprehensive universal sampling frame is established. Recreational catch data are collected through an access-site intercept survey. A minimum set of standard data elements is collected in both telephone and intercept surveys (refer to the ACCSP Program Design Document for details). The ACCSP will implement research and evaluation studies to expand sampling and improve the estimates of recreational catch and effort.

States currently have varying types of commercial license structures and reporting requirements. Specifics of the existing state programs are summarized in Table 1 for the commercial fishery and Table 2 for the recreational fishery. All states except New Jersey and Rhode Island have implemented mandatory reporting for the commercial fishery, but the level of reporting varies from daily to monthly to annually/by season. Units of effort collected through these reporting programs include per month, pounds per unit of gear per day, and eels per pot-hour. Some states have a specific eel license, but a general commercial fishing license is the most common license type.

For the recreational sector, many states have a freshwater recreational fishing license but few require a saltwater recreational fishing license. Virginia has a recreational eel pot license with mandatory reporting, but no reporting is required for a saltwater license, which allows the license holder to use up to two eel pots. North Carolina has a Recreational Commercial Gear License, and 33 percent of license holders are surveyed each year to obtain an estimate of recreational catch and effort. The remaining states do not currently have recreational mandatory reporting.

The ASMFC American Eel Technical Committee noted that a large percentage of eel catch and effort takes place in inland areas under the jurisdiction of multiple state agencies. Full implementation of this Addendum will require cooperation and communication between state agencies to ensure coverage in all areas where eel harvest occurs.

Table 1. American eel commercial reporting and license requirements by state as of November 2005.

State	Commercial Mandatory Reporting?	Schedule of Commercial Reporting?	Commercial Effort Type Reported	Commercial License Type	Dealer or Harvest Data	Gear Types
ME - elver fishery	yes	season report	total pounds/month reported, pounds/net by month calculated assuming all gear fished	specific elver license	dealer	dip net, mostly fyke net
ME - pot fishery	yes	season report	pounds/month, pots fished, and days fished reported	specific license	harvest	pot
ME - weir fishery	yes	season report	pounds/month reported, days fished reported, pounds/weir/day calculated	specific license	harvest	weir
NH	yes	monthly reports with daily information	pounds landed, hours or days gear fished	general commercial license	harvest	pot
MA	yes	annual catch reports	pounds/pot/night (beginning in 2003)	general commercial license, specific endorsement for eel	harvest	pot
RI	no	n/a	n/a	multipurpose license	IVR system	pot
CT	yes	monthly reports with daily information	pounds/day	general commercial license		harvest
NY - marine district	yes	VTR	catch (pounds)/trip	general commercial license	VTR and IVR	pot
NY - inland	yes	season report	catch/unit of gear/day	each piece of gear is licensed	harvest	weir and pot
NJ	no	n/a	n/a	general commercial license	none	pot
PA	n/a	no commercial fishery	n/a	n/a	n/a	n/a
DE	yes	monthly	pounds landed, pots fished/day	specific eel license	harvest	pot
MD	yes	monthly reports with daily information	pounds/pot/area/day	general commercial license	harvest	pot
DC	n/a	no commercial fishery	n/a	n/a	n/a	n/a
PRFC			pounds/license, pounds/pot, pounds/day			pot
VA	yes	monthly reports with daily information	soak time for gear used, number of pots fished, pounds landed, water body	each gear has a specific license (including eel pots), dealer license required to purchase from harvester	harvester or dealer	mainly eel, fish and peeler pots
NC	yes	trip level	per trip (per purchase)	standard commercial fishing license (SCFL)	trip ticked (since 1994)	pot
SC	yes	monthly reports with daily information	eels/pot-hour	general freshwater commercial license, general saltwater commercial license	harvest	pot, dip net, fyke net
GA	yes	monthly reports with daily information	eels/pot-hour	commercial fishing license, commercial boating license	harvest	pot, trap
FL	yes	monthly	pounds/pot/day (since 2003)	specific permit for those who use HSC as bait (until July 2006), all commercial harvesters have a generic commercial license, specific eel permit will be required 7-1-06	harvest	pot

Table 2. American eel recreational reporting and license requirements by state as of November 2005.

State	Recreational License Type	Recreational Reporting?
ME - elver fishery	n/a (no recreational fishing for elvers)	n/a
ME - pot fishery	none	
ME - weir fishery	n/a (no recreational weir fishing)	n/a
NH	coastal harvest license (saltwater) for pot/trap gear, freshwater fishing license for hook and line	coastal harvest report (saltwater) if using gear other than hook and line
MA	none	none
RI	no saltwater recreational license	none
CT	no saltwater recreational license	none
NY - marine district	no saltwater recreational license	none
NY - inland	recreational license above first dam impassable to fish	none
NJ	no saltwater recreational license	none
PA	freshwater fishing license required	
DE	no saltwater recreational license	none
MD	tidal recreational license, non-tidal recreational license	none
DC	recreational fishing license	
PRFC		
VA	saltwater fishing license, freshwater fishing license, recreational eel pot license	saltwater license allows 2 eel pots with no reporting requirement (as of July 2005), no reporting for freshwater license, mandatory reporting for recreational eel pot license
NC	Recreational Commercial Gear License in marine waters, inland recreational license through WRC	RCGL survey: 33% of license holders, survey asks total # of trips/month, avg. # eel pots/trip, water body most often fished, catch information, species, # kept, # released
SC	tag required to use commercial gear in freshwater, saltwater recreational fishing license	none
GA	general state recreational fishing license (freshwater and saltwater)	none
FL	general state recreational fishing license	none

MANDATORY CATCH AND EFFORT MONITORING PROGRAM

Following the recommendations of the American Eel Stock Assessment Subcommittee, Technical Committee, and Advisory Panel, the Management Board requires, through this Addendum, a catch and effort monitoring program for American eel. States and jurisdictions have the following options:

Option 1

A permit allowing commercial harvest with mandatory reporting of eel catch and effort, applicable only to the commercial sector of the eel fishery.

Option 2

A dealer permit with a mandatory purchase-reporting requirement.

The eel permit and reporting program is to be implemented in all areas, freshwater and saltwater, where eel are harvested to provide a complete picture of catch and effort for the commercial fishery and useful data for stock assessments. Permits are to be issued with a requirement to report eel catch and effort on a trip-level basis. Completion of reporting is to be a condition of permit renewal. Reports should include soak time, number of units of gear fished, and pounds landed by life stage.

Efforts to collect catch and effort data should be consistent with the ACCSP standards listed above.

MANAGEMENT PROGRAM IMPLEMENTATION

Implementation Schedule

The implementation deadline for Addendum I is January 1, 2007. States that are required to pass new regulations in their legislatures are permitted an additional six months to implement the Addendum, if needed. State implementation plans are due to the ASMFC by May 1, 2006. Upon receipt, the American Eel Technical Committee will review the implementation plans and provide feedback to the American Eel Management Board. The earlier deadline is intended to allow additional time for the states to make changes to their plans prior to the implementation process.

Atlantic States Marine Fisheries Commission

ADDENDUM II TO THE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL

ASMFC Vision Statement:
Healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015.

Approved October 23, 2008

INTRODUCTION

The Atlantic States Marine Fisheries Commission's American Eel Management Board initiated the development of Addendum II in January 2007 to propose measures that would facilitate escapement of silver eels during or just prior to their spawning migration as a means to improve American eel recruitment and abundance. Although the available data for American eel in the U.S. have not been sufficient to perform a reliable quantitative assessment of the population size or fishing mortality rates (ASMFC 2001, 2006), there has been evidence that the stock has declined and is at or near low levels (ASMFC 2000, 2001, 2006; USFWS 2007). The Management Board asked the Technical Committee (TC) and Advisory Panel (AP) to consider closed seasons, gear restrictions, size limits or a combination of these measures to reduce the harvest of emigrating eels. The public comment draft of Addendum II proposed these management measures, as well as recommendations for increased protection of American eels during their upstream and downstream migration.

This Addendum recommends stronger regulatory language to improve upstream and downstream passage of American eel to state and federal regulatory agencies. As such, there is no implementation schedule and there are no new compliance requirements. Member states are still required to submit annual compliance reports by September 1. This Addendum does not alter any other provisions from the Interstate Fishery Management Plan (FMP) and makes no changes to Addendum I to the FMP.

Background

The American eel occupies fresh, brackish, and coastal waters along the Atlantic from the southern tip of Greenland to northeastern South America. The species is catadromous, spending the majority of life in freshwater, but migrating to the Sargasso Sea to spawn. Newly hatched eels drift on oceans currents, eventually entering nearshore areas where they migrate up-river. Therefore, a comprehensive eel management plan and comprehensive set of regulations must consider the various unique life stages and the diverse habitats used, in addition to society's interest and use of this resource.

American eel (*Anguilla rostrata*) occupy a significant and unique niche in the Atlantic coastal reaches and its tributaries. Historically, American eel were very abundant in East Coast streams, comprising more than 25 percent of the total fish biomass. Eel abundance declined from historic levels but remained relatively stable until the 1970s. More recently, fishermen, resource managers, and scientists postulated a further decline in abundance based on harvest information and limited assessment data. This resulted in the development of the Atlantic States Marine Fisheries Commission FMP for American Eel. The goals of the FMP are:

1. Protect and enhance the abundance of American eel in inland and territorial waters of the Atlantic States and jurisdictions and contribute to the viability of the American eel spawning population; and
2. Provide for sustainable commercial, subsistence, and recreational fisheries by preventing overharvest of any eel life stage.

In support of these goals, the following objectives were included in the FMP:

- Improve knowledge of eel utilization at all life stages through mandatory reporting of harvest and effort by commercial fishers and dealers, and enhanced recreational fisheries monitoring.
- Increase understanding of factors affecting eel population dynamics and life history through increased research and monitoring.
- Protect and enhance American eel abundance in all watersheds where eel now occur.
- Where practical, restore American eel to those waters where they had historical abundance but may now be absent by providing access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel.
- Investigate the abundance level of eel at the various life stages, necessary to provide adequate forage for natural predators and support ecosystem health and food chain structure.

Status of the Stock

Current stock status (i.e., overfished or not overfished) for American eel is poorly understood due to limited and non-uniform stock assessment efforts and protocols across the species' range. No range-wide estimate of abundance exists and reliable indices of abundance of this species are scarce. Information on demographic structure is lacking and difficult to determine because the American eel is a single population (termed *panmixia*) with individuals randomly spread over an extremely large and diverse geographic range, with growth rates and sex ratios environmentally dependent. At present, limited data (fishery-dependent and independent) from indirect measurements (harvest by various gear types and locations) and localized direct stock assessment information are collected.

In 2003, declarations from the International Eel Symposium (AFS 2003, Quebec City, Quebec, Canada) and the Great Lakes Fishery Commission (GLFC) highlighted concerns regarding the health of American eel stock. Canada has recently applied the "Special Concern" designation to American eel. Available data attributes the population drop to decreasing recruitment combined with localized declines in abundance. This information is cause for concern and represents an opportunity for cooperation with other entities such as the GLFC to preserve the American eel stock.

The most recent peer reviewed stock assessment was presented to the Commission's American Eel Management Board in February 2006. The stock assessment did not meet some of the terms of reference according to the Terms of Reference and Advisory Report to the American Eel Stock Assessment Peer Review (ASMFC 2006). In May 2006, the Board tasked the American Eel Stock Assessment Subcommittee (SASC) with following up on specific recommendations in the peer review report to improve the 2005 stock assessment. The SASC follow-up to the Terms of Reference and Advisory Report to the American Eel Stock Assessment Peer Review was presented to the Board in October 2006. This report was inconclusive regarding the status of the stock. In their follow-up report, the SASC created a coastwide index for American eel using yellow eel indices that are monitored along the Atlantic Coast, both in the United States and Canada, and combing them with General Linear

Modeling (GLM). The SASC's report included a suggestion that the coastwide yellow eel GLM index could be used as a management trigger and would be a means to monitor coastwide, yet act locally.

In reaction to the extreme declines in eel abundance the Saint Lawrence River-Lake Ontario portion of the species' range, the Commission requested in 2004 that the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS) conduct a status review of American eel. In February 2007, the USFWS announced the completion of a Status Review for American eel. The report concluded that protecting eel as an endangered or threatened species is not warranted. The USFWS did note that while the species' overall population is not in danger of extinction or likely to become so in the foreseeable future, the eel population has "been extirpated from some portions of its historical freshwater habitat over the last 100 years…[and the species abundance has declined] likely as a result of harvest or turbine mortality, or a combination of factors" (50 CFR Part 17).

Following the 2005 stock assessment, Terms of Reference and Advisory Report to the American Eel Stock Assessment Peer Review, and Stock Assessment Subcommittee's 2006 report, the Board initiated this Addendum to consider management options to halt the current decline in yellow eel abundance.

Status of the Fishery

American eel currently support important commercial fisheries throughout their range. Fisheries are executed in rivers, estuaries, and ocean. Commercial glass eel harvest is legal in Maine and South Carolina, although reported landings are minimal in South Carolina. Yellow and silver eel fisheries exist in all states and jurisdictions with the exception of Pennsylvania and the District of Columbia. South Carolina and Georgia recorded no commercial yellow or silver eel landings in 2007.

Commercial

Commercial landings decreased from a high of 1.8 million pounds in 1985 to a low of 641,000 pounds in 2002. Landings of yellow and silver eel in 2007 totaled 834,500 pounds.[1] New Jersey and Delaware each reported landings over 100,000 pounds of eel and Maryland reported landings over 300,000 pounds in 2007. Combined, these three states accounted for 73% of the coastwide commercial landings. Massachusetts, Pennsylvania, Georgia, Florida, and the District of Columbia were granted *de minimis* status for the 2007 commercial fishing year. D*e minimis* is approved if a member states' commercial landings of yellow and silver eel for the previous year is less than 1% of the coastwide landings for the same year. Additionally, member states must request *de minimis* status.

Recreational

Few recreational anglers directly target eel and most landings are incidental when anglers are fishing for other species. Eel are often purchased by recreational fishermen for use as bait for larger sport fish such as striped bass, and some recreational fishermen may catch their own eel to utilize as bait. The NMFS Marine Recreational Fisheries Statistics Survey (MRFSS)

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[1] Harvest data for 2007 comes from the 2008 State Compliance Reports. The landings are preliminary and some are incomplete.

shows a declining trend in the catch of eel during the latter part of the 1990s. According to MRFSS[2], 2007 recreational total catch was 140,372 fish, which represents a 63% increase in number of fish from 2006 (86,024 fish). About 59% of the eel caught were released alive by the anglers. MRFSS 2007 total recreational harvest was 57,986 fish.

For current commercial and recreational regulations for American eel by state, please see Appendix I.

STATEMENT OF THE PROBLEM

While the status of the American eel stock is uncertain, the latest stock assessment information indicates that the abundance of yellow eel (a juvenile life stage) has declined in the last two decades and the stock is at or near low levels. Further, relative abundance is likely to continue to decline unless mortality decreases and recruitment increases. The American Eel Management Board directed the American Eel Plan Development Team (PDT) to develop potential management measures for American eel that would facilitate an increase in the number of adult American eel (also known as silver eel) that are able to move from fresh and estuarine water to the ocean—also known as out-migrate—and spawn. The recommended management measures included gear and size restrictions, seasonal closures, and a recommendation to protect the upstream and downstream migration of American eel.

The Board initiated this Addendum based on a concern for the American eel population and sought public comment on measures that would facilitate escapement of silver eel on their spawning migration with the intent of halting any further declines in juvenile recruitment and eel abundance. The Board chose not to implement any additional restrictions on the fishery at this time and requested that a new stock assessment be initiated to better understand the stock status. The primary objective of this document is to recommend stronger regulatory language to improve upstream and downstream passage of American eel to state and federal regulatory agencies.

PROPOSED MANAGEMENT OPTIONS from the PUBLIC COMMENT DRAFT of ADDENDUM II

Gear restrictions, size limits, and seasonal closures employed individually or in combination can protect out-migrating silver eels by allowing more silver eel to reach the Sargasso Sea and spawn. American eel larvae and glass eel recruit to estuaries and freshwater at random; it is predicted that increased escapement from any part of the species' range has the potential to benefit the species throughout the entire range. While operating under the theory that allowing more silver eel to escape will result in increased juvenile recruitment, the PDT recognizes that several factors can influence the amount of silver eels that are allowed to out-migrate, including:

1. The time duration in which silver eel out-migrate;

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[2] MRFSS Data for American Eel are unreliable. 2007 Proportional Standard Error (PSE) values for recreational harvest in Massachusetts, Rhode Island, New Jersey, Delaware, Virginia, and South Carolina are 100, 84.3, 70.2, 100.4, 100 and 100 respectively.

2. The portion of the out-migration period that is covered by the closed season;
3. The maximum size eel that gear can catch;
4. The maximum size eel that harvesters are allowed to possess.

The Board chose to delay action on commercial fishery management measures in order to incorporate the results of the upcoming stock assessment, which will present new and updated information on American eel stock status, including the long-term young-of-the-year index being conducted by the states. In addition, the Board received substantial public comment and advice from its Advisory Panel that further restrictions on American eel harvest would significantly impact fishermen. The states will revisit management measures upon completion of the American eel stock assessment.

RECOMMENDATIONS FOR IMPROVING UPSTREAM AND DOWN STREAM PASSAGE OF AMERICAN EEL

There are multiple factors that influence the American eel population across its range, as well as factors that influence their local abundance. Such factors include barriers to upstream and downstream migration, loss of habitat, and natural oceanographic conditions. On the Atlantic and Gulf coasts, 33,663 dams potentially hinder American eel movement. Of these dams, 1,511 (4.5 percent) are for hydropower (50 CFR Part 17).

Recommendations for Federal Energy Regulatory Commission Relicensing

The Commission recognizes that many factors influence the American eel population, including harvest, barriers to migration, habitat loss, and natural climatic variation. The Commission's authority, through its member states, is limited to controlling commercial and recreational fishing activity; however, to further promote the rebuilding of the American eel population, the Commission strongly encourages member states and jurisdictions, as well as the U.S. Fish and Wildlife Service, to consider and mitigate, if possible, other factors that limit eel survival. Specifically, the Commission requests that member states and jurisdictions request special consideration for American eel in the Federal Energy Regulatory Commission relicensing process. This consideration should include, but not be limited to, improving upstream passage and downstream passage, and collecting data on both means of passage.

Recommendations for Improving American Eel Passage at Non-Federally Licensed Dams

Of the 33,663 dams located on the Atlantic and Gulf Coasts that potentially hinder American eel movement, 95% are not licensed by the federal government. Therefore, the states should strive to remove these obstructions where feasible. If removal is not feasible, then upstream and downstream passage should be improved to provide access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel consistent with the goal of the FMP.

APPENDIX I

Table A1. Commercial Regulations by State*

State	Size Limit	License/Permit	Other
ME		· Harvester and dealer license · Dealer reporting	· Seasonal closures · Gear restrictions
NH	6"	· Commercial saltwater license · Coastal harvest permit · Monthly trip level catch & effort reporting of harvest	· 50/day for bait · Gear restrictions in freshwater
MA	6"	· Commercial permit with annual catch report requirement · Registration and reporting for all eel buyers	· Nets, pots, spears, and angling only · Mesh restrictions · Coastal towns may have additional requirements
RI	6"	· Commercial fishing license required for the sale of American eel · Quarterly reporting	
CT	6"	· Commercial license with dealer reporting	· Gear restrictions
NY	6”	· Commercial harvester and dealer license and harvester reporting	· Gear restrictions
NJ	6"	· License required · Monthly reporting for eel pot license	· Gear restrictions
PA		· No commercial fishery	
DE	6"	· License required · Monthly reporting with catch and effort	· Commercial fishing in tidal waters only
MD	6"	· Licensed required with monthly reporting.	· Prohibited in non-tidal waters · Gear restrictions · Commercial crabbers 50 eel pots/day max no harvest limit
DC		· No commercial fishery	
PRFC	6"	· Eel license · Harvester weekly reporting w/daily effort	· Gear restrictions
VA	6"	· License with two-year delayed entry system · Mandatory monthly reporting (at trip level)	· Mesh size restrictions on eel pots
NC	6"	· Standard Commercial Fishing License for all commercial fishing	· Mesh size restrictions on eel pots · Bait limit of 50 eels/day
SC		· Permits by gear and area fished · Mandatory monthly reporting · License for all commercial fishing and sale	· Various gear restrictions
GA	6"	· Personal commercial fishing license and commercial fishing boat license · Harvester/dealer reporting required	· Gear restrictions on traps and pots
FL		· Commercial fishing license · Mandatory permit for all commercial eel harvesters · Mandatory trip and monthly sales summary reporting for permittees	· Gear restrictions

* For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

Table A2. Recreational Regulations by State*

State	Size Limit	Possession Limit	Other
ME	6"	50 eels/person/day	· Gear restrictions · License requirement and seasonal closures (inland waters only)
NH	6"	50 eels/person/day	· Coastal harvest permit needed if taking eels other than by angling · Gear restrictions in freshwater.
MA	6"	50 eels/person/day	· Nets, pots, spears, and angling only · Mesh restrictions · Coastal towns may have additional requirements
RI	6"	50 eels/person/day	
CT	6"	50 eels/person/day	
NY	6”	50 eels/person/day	· Additional length restrictions in specific inland waters
NJ	6"	50 eels/person/day	
PA	6"	50 eels/person/day	· Gear restrictions
DE	6"	50 eels/person/day	· Two pot limit/person
MD	6"	No possession limit in tidal areas (hook & line); 25/person/day w/10 eel pot max for rec. crabber in tidal; 25/person/day in non-tidal	· Gear restrictions
DC	6"	10 eels/person/day	· Five trap limit
PRFC	6"	50 eels/person/day	· Recreational license
VA	6"	50 eels/person/day	· Recreational license, no reporting · Recreational commercial gear license, annual report required · Two eel pot limit (both licenses) · Mandatory annual catch report for eel pot license · Mesh size restrictions on eel pots
NC	6"	50 eels/person/day	· Gear restrictions · Noncommercial special device license, allowed two eel pots under Recreational Commercial Gear license
SC	None	None	· Gear restrictions
GA	None	None	
FL	None	None	· Mesh size and funnel opening restrictions on eel pots

* For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

Stock Assessment Report No. 12-01
of the

Atlantic States Marine Fisheries Commission

American Eel Benchmark Stock Assessment

Accepted for Management Use May 2012

Working towards healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015

Preface

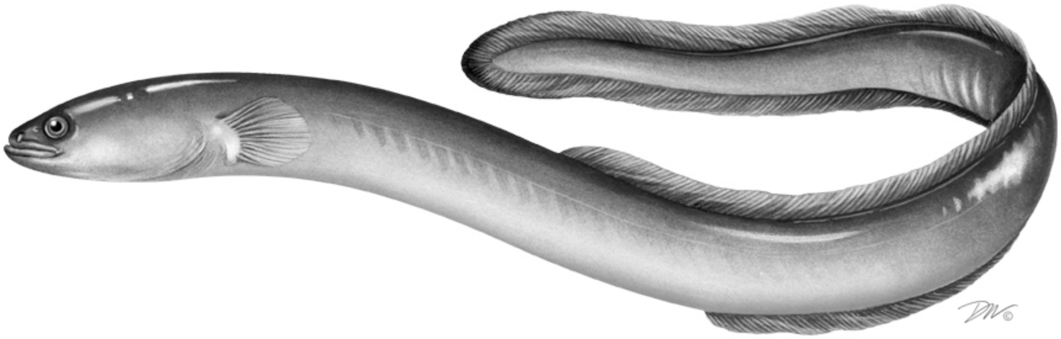
The 2012 Benchmark Stock Assessment of American Eel occurred through an Atlantic States Marine Fisheries Commission (ASMFC) external peer review process. ASMFC organized and held Data Workshops on September 14-16, 2009 and June 21-24, 2010. Assessment Workshops were held on May 23-26, 2011 and August 22-25, 2011. Participants of the Data and Assessment Workshops included the ASMFC American Eel Stock Assessment Subcommittee and Technical Committee, as well as invited individuals from state and federal partners. ASMFC coordinated a Peer Review Workshop from March 16 – 17, 2012. Participants included members of the American Eel Stock Assessment Subcommittee and a Review Panel consisting of four reviewers appointed by ASMFC.

Terms of Reference and Advisory Report of the Peer Review Panel (PDF Pages 3-35)
The Terms of Reference Report provides a detailed evaluation of how each Terms of Reference was addressed by the Stock Assessment Subcommittee, including the Panel's findings on stock status and future research recommendations. The Advisory Report provides an summary of the stock assessment results supported by the Review Panel.

American Eel Stock Assessment Report for Peer Review (PDF Pages 36-338)
This report describes the background information, data used, and analysis for the assessment submitted by the Technical Committee to the Review Panel. It contains a coastwide and regional analysis and comparison of American eel populations.

A publication of the Atlantic States Marine Fisheries Commission pursuant to National Oceanic and Atmospheric Administration Award No. NA10NMF4740016

Terms of Reference & Advisory Report of the American Eel Stock Assessment Peer Review



Conducted on
March 12-13, 2012
Raleigh, North Carolina

Prepared by the
ASMFC American Eel Stock Assessment Peer Review Panel

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Atlantic States Marine Fisheries Commission

Working towards healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015

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Preface

Summary of the ASMFC Peer Review Process

The Stock Assessment Peer Review Process, adopted in October 1998 and revised in 2002 and 2005 by the Atlantic States Marine Fisheries Commission (ASMFC or Commission), was developed to standardize the process of stock assessment reviews and validate the Commission's stock assessments. The purpose of the peer review process is to: (1) ensure that stock assessments for all species managed by the Commission periodically undergo a formal peer review; (2) improve the quality of Commission stock assessments; (3) improve the credibility of the scientific basis for management; and (4) improve public understanding of fisheries stock assessments. The Commission stock assessment review process includes an evaluation of input data, model development, model assumptions, scientific advice, and a review of broad scientific issues, where appropriate.

The Benchmark Stock Assessments: Data and Assessment Workshop and Peer Review Process report outlines options for conducting an external peer review of Commission managed species. These options are:

1. The Stock Assessment Workshop/Stock Assessment Review Committee (SAW/SARC) conducted by the National Marine Fisheries Service (NMFS), Northeast Fisheries Science Center (NEFSC).
2. The Southeast Data and Assessment Review (SEDAR) conducted by the National Marine Fisheries Service, Southeast Fisheries Science Center (SEFSC).
3. The Transboundary Resources Assessment Committee (TRAC) reviews stock assessments for the shared resources across the USA-Canada boundary and is conducted jointly through the National Marine Fisheries Service and the Canada Department of Fisheries and Oceans (DFO).
4. A Commission stock assessment Peer Review Panel conducted by 3-5 stock assessment biologists (state, federal, university). The Commission Review Panel will include scientists from outside the range of the species to improve objectivity.
5. A formal review using the structure of existing organizations (i.e. American Fisheries Society, International Council for Exploration of the Sea, or the National Academy of Sciences).

Twice annually, the Commission's Interstate Fisheries Management Program (ISFMP) Policy Board prioritizes all Commission managed species based on species management board advice and other prioritization criteria. The species with highest priority are assigned to a review process to be conducted in a timely manner.

In March 2012, the Commission convened a Stock Assessment Peer Review Panel comprised of scientists with expertise in stock assessment methods and/or diadromous species and their life history. The review of the American eel stock assessment was conducted at the Doubletree Brownstone Hotel in Raleigh, North Carolina from March 12-13, 2012. Prior to the Review Panel meeting, the Commission provided the Review Panel Members with an electronic copy of the 2012 American Eel Stock Assessment Report.

The review process consisted of an introductory presentation of the completed 2012 stock assessment. Each presentation was followed by general questions from the Panel. The second day involved a closed-door meeting of the Review Panel during which the documents and presentations were reviewed and a report prepared.

The report of the Review Panel is structured to closely follow the terms of reference provided to the stock assessment team.

Acknowledgements

The Review Panel thanks members of the American Eel Stock Assessment Subcommittee and Technical Committee, as well as staff of the Atlantic States Marine Fisheries Commission, particularly Patrick Campfield, for support during the review process.

Photo: Travis Elsdon

Introduction

The American eel *Anguilla rostrata* is one of 15 species in the family Anguillidae. All are characterized by great adaptability to a wide range of aquatic ecosystems, and consequently are found around the globe. All reproduce at sea and are at least facultatively catadromous, meaning they use inland habitats. Their complex life history is documented well in the 2012 American Eel Stock Assessment report. Of note is the fact that the American eel, from its northern limit in Greenland down to its southern limit in French Guiana, is one population.

In New Zealand, anguillid eels are revered as spirits as much as they are prized as food (Prosek 2010). In traditional North American Indian cultures, the same is true. The Iroquois Confederacy in New York State has an Eel Clan; many of the governing leaders are recruited from this clan. However, today, the American eel is all but extirpated from Lake Ontario drainages, and most members of the Eel Clan have never seen a live eel (J. Shenandoah, Onondaga Nation elder, personal communication).

Eels were formerly extremely abundant in inland waters of eastern North America, colonizing lakes, rivers, streams, and estuaries. In Onondaga Lake in New York State, 17th century Jesuit missionaries noted with wonder that "...the eel is so abundant that a thousand are sometimes speared by a single fisherman in a night..." (Clark 1849). American eels penetrated the major Atlantic waterways of North America, reaching the Great Lakes via the St. Lawrence River and the mid-western American states via the Mississippi as far as Minnesota (Eddy and Underhill 1974). Coastal eel abundances were very high, and during the spring, runs of recruiting glass eels would form "walls of glass" as they ascended barriers. Eel fisheries flourished well into the early 20th century.

Eels were once an important food fish in the U.S., but today are mainly sold as bait or exported to Europe and Asia, where demand continues to be high. Declines in European and Asian eels drive the export fishery, and in particular, the export market for glass eels has commanded prices exceeding $2000/lb this year.

The American eel stock status is depleted. The seeds of the current depletion lay in part in a fishing up/fishing down' episode that occurred on American eels in the 1970s into the 1980s as export demand rose. Roughly during the same period, river damming intensified and hydroelectric facilities on dams caused additional mortality. A suite of stressors including habitat loss from dams or urbanization, turbine mortality, the non-native swim-bladder parasite *Anguillicolla*, toxic pollutants, and climate change are all factors that act in concert with fishing mortality on American eel. Through a series of data analyses and modeling, the SASC has documented this depletion. The following Peer Review Report discusses the SASC stock assessment findings, comments on strengths and weaknesses, and makes recommendations for additional data needs and future assessments.

Terms of Reference for the American Eel Stock Assessment Peer Review

1. Evaluate the thoroughness of data collection and the presentation and treatment of fishery-dependent and fishery-independent data in the assessment, including the following but not limited to:

1. **Consideration of data strengths and weaknesses (e.g., temporal and spatial scale, gear selectivities, ageing accuracy, sample size),**
2. **Presentation of data source variance (e.g., standard errors),**
3. **Justification for inclusion or elimination of available data sources,**
4. **Calculation and/or standardization of abundance indices.**

In accordance with recommendations from the 2005 stock assessment peer review (ASMFC 2006), more up-to-date information was included as regards biology, life history, and habitat use in continental waters. Updated information was also provided on fishing regulations (commercial and recreational), as well as on an ongoing review by the U.S. Fish and Wildlife Service for possible listing under the Endangered Species Act.

Trends in catch and value of catch were included. Of special note is the current market price of glass eels, which rose to over $2000/lb this spring (NYT 2012). This fishery is regulated in the state of Maine, but the SASC noted that poaching (unlicensed fishing) is a serious concern in many states, including Maine. In other fisheries, which are largely for bait (domestic usage) or export, there are uncertainties in catches particularly for recreational fishing and for data prior to the standardized record-keeping of the 1950s. *Nevertheless, the Panel felt the data collection and data quality analyses conducted were adequate*.

The American eel Stock Assessment Subcommittee (SASC) canvassed and assessed all available and known data sets (the Panel noted there were some that were unknown to the SASC). Over 100 data sets, comprising fishery-independent and dependent studies, were found and assessed (Appendix A of the stock assessment report). Fishery-dependent data were examined for trends but not included in the analyses due to problems with series standardization. Fishery-independent data sets were excluded if they were less than 10 years in length, if eels were sparsely reported, if there was bias due to catchability, or if sampling protocols or sites were inconsistent. *Given what was available, the Panel felt the data were sufficient to perform the necessary assessments*.

However, some potentially useful data sets were unavailable to the assessment due to data processing lags, unknown errors, or legal issues. One such data set that would have been of particular utility is a 30+ year Juvenile Fish and Blue Crab Trawl Survey conducted by the Virginia Institute of Marine Science (VIMS). Raw survey data were requested but not provided by VIMS. Index values were provided, although there was apparently a processing error in the data base, such that all size classes (pre-recruits, recruits, and post-recruits) appeared to have identical indices of abundance. The Panel judged index values to be erroneous based on length-frequency data provided for selected years of the survey. *The Panel recommends additional effort be made to obtain raw data, and/or reconcile the size-based abundance data with length frequency data in the VIMS Trawl Survey, so that this valuable data set can be used in future assessments*.

The data sets were analyzed for trends by grouping them into six regions, loosely defined as hydrologic units. This approach worked better in data-rich regions of the north and mid-Atlantic, but North Carolina, South Carolina, Georgia, and Florida were grouped as a single unit, due to data paucity, making it difficult to discern trends in this region.

The assessment considered length-weight relationships, age-length relationships, sex ratios, and growth models (6 in total, evaluated with AIC). Length-weight relationships varied by region, with the highest weight per length in the southeastern region and lowest in the Hudson River. As is the case for European eel, age at length shows great variation, but there are significant regional trends: higher lengths at age in the north and lower lengths at age in the South Atlantic. These are likely affected by habitat and environment, as is also the case with sex ratio. As for growth, no single model stood out as “best,” due in part to the data sets available. Notwithstanding this, these analyses confirmed that there is only a weak relationship of age with length in American eel.

As recommended in the 2005 peer review (ASMFC 2006), trend analyses were performed after first standardizing the data sets by generalized linear modeling (GLM; protocols documented in Appendix B of stock assessment report). The Panel noted that while this is a reasonable approach, the variance in the indices is likely understated. GLMs were applied to individual datasets to standardize the indices of abundance, and then those estimates were input into another GLM to produce regional or coastwide estimates of abundance. Datasets from individual surveys could not be combined into a single GLM due to the different covariates measured in the individual surveys. The Panel felt that doing a trend analysis on a regional or coastwide GLM estimate of abundance (which is based on GLM estimates of abundance of individual surveys) masked the uncertainty in these trends. It was suggested that hierarchical GLM may be used in future assessments to explore relationships across regions where covariate data exist. This may allow for determination of the level of unquantified uncertainty in the current approach.

Regional and coast-wide abundance indices (GLM-standardized) of young-of-year (YOY) and older eels were developed by combining individual data sets. Trends were shown with standard error bars about the estimates. Region to region, some areas exhibited clear declines (Hudson, southern states) while others exhibited little or no trend (e.g., Delaware and Mid-Atlantic coastal bays). However, to some extent this was confounded by the length of the time series and the availability of the regional data sets.

Power analysis, Mann-Kendall tests, meta-analyses, and ARIMA models were used to examine trends in the data and were useful as exploratory tools. The Panel was concerned that the ARIMA approach depended heavily on the first data point in any given time series, as this often defined the resulting observed trend even when immediately adjacent data points showed the opposite trend. Caution should therefore be taken with interpreting ARIMA-based indices. Nevertheless, taken in the aggregate, a number of these analyses showed evidence of a long-term decline in eel abundance.

The ‘traffic light approach’ (TLA; cf. Caddy 1998, 1999) was also used to explore trends in the various data sets. The TLA provides a framework to communicate trends in disparate data sets to stakeholders and the general public. The SASC used the TLA to summarize the trends in abundance indices, color coding them by region and year as ‘green’ (metric above 75th percentile), ‘yellow’ (between 25th and 75th percentile), and ‘red’ (below the 25th percentile of the data). This yielded complex spatial and temporal patterns in the indices that were difficult to interpret. The Panel noted the TLA could be used to put the abundance indices in the broader context of trends in the environment (e.g. regional temperatures and salinities), the eel’s biology (e.g. growth, condition, and early life history) and loss of its habitat (e.g. dam construction).

As required by ASMFC mandate, states must now monitor YOY eels. Data sets were analyzed but few trends were found, likely because the monitoring programs were only relatively recently implemented. Other, longer term ichthyoplankton data (Little Egg Inlet, NJ and Beaufort Inlet, NC) could be of interest. These data are of leptocephali just encountering the coast, and hence may be more a measure of inter-annual variability in offshore recruitment from the Sargasso. Although Sullivan et al. (2006) found little concordance in these data, the GLM normalized data (presented in Figures 5.35 and 5.36 of the stock assessment report) showed a high degree of temporal concordance (Figure 1), Although the Beaufort data are truncated to 2003 (due to lack of resources to process the samples), the strong concordance suggests the Beaufort site might show trends similar to Little Egg Inlet in recent years (a marked decline in abundance after 2008).

In summary, following the recommendations of the 2005 stock assessment peer review, many data sets and ancillary information were gathered; uncertainties quantified; trends examined in multiple ways; and strengths and weaknesses of data and approaches were pointed out. *The Panel considers that a credible analysis of the available data was undertaken by the SASC.*

2. Evaluate the methods and models used to estimate population parameters (e.g., *F*, biomass, abundance), including but not limited to:

1. **Evaluate the choice and justification of the preferred model(s). Was the most appropriate model (or model averaging approach) chosen given available data and life history of the species?**
2. **If multiple models were considered, evaluate the analysts’ explanation of any differences in results.**
3. **Evaluate model parameterization and specification (e.g., choice of CVs, effective sample sizes, likelihood weighting schemes, calculation/specification of *M*, stock-recruitment relationship, time-varying parameters, plus group treatment).**

The SASC considered a range of potential population models, most of which have been designed for use in data-poor situations. These included Catch Curve Analysis, Depletion-Corrected Average Catch (DCAC), Depletion-Based Stock Reduction Analysis (DB-SRA), Surplus Production Models (SPM; both age-structured and catch-free), An Index Method (AIM), Collie-Sissenwine, Survival Estimation in Non-Equilibrium situations (SEINE), and a suite of models used by ICES (Study Leading to Informed Management of Eels or SLIME). A number of these models were not pursued due to the lack of appropriate input information. Other models were considered inappropriate to the eel management needs of the ASMFC. For instance, the SLIME suite of models is generally designed to meet Northeast Atlantic–specific management requirements (i.e., provide estimates of escapement). The remaining models were pursued at some level. Surplus production models were attempted using the various regional and coast-wide yellow-stage indices of abundance but stable solutions could not be found. An AIM model was attempted for the Delaware Bay/Mid-Atlantic Coastal Bays and Chesapeake Bay regions but only one of the survey indices exhibited a correlation with the catch, and thus the method was not pursued. The SEINE model relies on a time series of data of sufficient length (greater than10 years), which is generally lacking.

The DB-SRA, which is an evolution of the DCAC, was thus pursued for application to American eels. The eel DB-SRA assumes that stock dynamics follow a hybrid of a Schaefer and Pella-Tomlinson-Fletcher surplus production functions (Dick and MacCall, 2011). It was noted that the Pacific Fishery Management Council requested a formal review of the DB-SRA model, along with others, and determined that it generally performs well in data-poor situations (Dorn, 2011). The model is applicable to a stock which has a time series of catch and for which productivity is not dominated by recruitment variability. Dorn (2011) noted that the performance (in terms of the federal Overfishing Limit or OFL estimation) was robust across a wide range of scenarios explored in simulation studies. In addition, the OFLs estimated by the model were generally lower than the "true" estimates, suggesting they are biased towards lower risk. The model has a number of advantages – it has minimal input data requirements, has a means to explore uncertainties, and allows determination of stock status in relation to derived reference points.

The Panel endorsed the SASC's selection of the DB-SRA model for use in the American eel stock assessment but had a number of concerns. The model's production function is designed for Pacific finfish and may not be appropriate for east coast American eel. In its current configuration, the model is restricted to describing eel stock dynamics during the freshwater / estuarine life history stages, with no consideration of the marine stage. Thus, it cannot respond to the dynamics of eel stock components that reside elsewhere (e.g. in Canadian and Caribbean waters, or in offshore marine waters). The assumption is made that the dynamics of non-US eel stock components follow those modeled for the US component. This assumption is violated in Canadian waters as some eel fisheries (e.g. Ontario) are currently closed (DFO, 2010). The model makes the assumption that there is negligible error in the catch. The SASC noted a number of issues with the historical catch which puts this assumption in doubt. In order to compensate for a lack of data, the DB-SRA requires a number of assumptions on stock dynamics, including natural mortality

(M), the F_{MSY} / M ratio, the B_{MSY} / K ratio, and finally the $B_{CURRENT}$ / K (or depletion) ratio. Input estimates of the carrying capacity (K) are varied to determine the K which provides the desired current depletion ratio. Virtually all the parameters of the stock's production dynamics are defined based upon an expert judgment process. Therefore, careful consideration needs to be given to the selection of these inputs. The Panel was satisfied that the SASC chose appropriate estimates of input parameters, based upon knowledge of eel life history and analogy to other finfish stocks.

The DB-SRA model of Dick and MacCall (2011) does not incorporate observation on abundance indices either through a least squares or likelihood function to optimize the search of the input parameters. Thus, issues of effective sample size, likelihood weighting schemes, and so on, are not relevant to the current model. Subsequent to the review meeting, one of the panelists (J. Wiedenmann) was informed by the co-creator of DB-SRA (E. Dick) that the PFMC does not use the model to assess stock status and that it is only used to estimate yield under an assumed estimate of current depletion. This usage may be due to the lack of an optimization function in the model. However, in a form of optimization (see below), the SASC used the 1990–2010 coastwide eel biomass indices to help inform the input distribution of $B_{CURRENT}$/K. The Panel felt this was an important innovation to the DB-SRA formulation introduced by the SASC, and represents a step toward more formal model fitting.

Another innovation introduced by the SASC was the incorporation of M in two time periods (1880–1969 and 1970–2010) to model the effects of habitat loss on stock productivity. Dam construction on the US east coast was considerable prior to 1970 which limited habitat availability to eels. The Panel considered that while adjustment of the model's production function due to habitat loss was necessary, it may be more appropriate to do this through a change to K. During the review meeting, a model change was made in which K varied between two time stanzas, with it being 75% of the historical K since 1970. Preliminary runs indicated the M and K adjusted models provided similar outputs. While the Panel accepted these adjustments to address the impact of habitat loss on the eel production function, it encouraged further explorations of these relationships.

The average age of maturity was assumed to be 8, which was used as the time lag between stock production and fishery exploitation. The Panel noted this may be too short a period: 4 is the age of recruitment to the fishery, the larval stage is 1-½ years duration and it takes about 4 years for a larva to grow to the silver (exploited) stage. Further analyses are encouraged to explore the sensitivity of the estimated reference points and stock status to changes in the age of maturity.

Notwithstanding the issues with the DB-SRA model, the Panel considered that the SASC undertook an appropriate selection process, adequately derived the range of input parameters and undertook innovative model adjustments to addresses issues specific to American eels.

3. Evaluate the diagnostic analyses performed, including but not limited to:

a. **Sensitivity analyses to determine model stability and potential consequences of major model assumptions**
b. **Evaluate the methods used to characterize uncertainty in estimated parameters. Ensure the implications of uncertainty in technical conclusions are clearly stated.**

For the DB-SRA, a number of sensitivity analyses were conducted to explore model stability and the impacts of different model assumptions. A thorough exploration of the sensitivity of results to model inputs and assumptions was conducted. In total, 14 sensitivity runs were reported within the assessment, although additional runs were explored at the stock assessment review meeting.

A thorough explanation of DB-SRA is provided in response to ToR 2. For all of the input distributions (M, F_{Msy}/M, B_{MSY}/K, and $B_{CURRENT}$ /K), the SASC assumed uniform distributions. Different ranges were explored for BCURRENT /K, but not for M, FMSY/M, BMSY/K. While the Panel agreed with the general ranges for the M, FMSY/M, and BMSY/K, it felt that an exploration of broader ranges, at least during initial runs, could better describe plausible values.

The sensitivity runs can be grouped into 2 broad categories: runs with a single M-stanza, and runs with a double-M stanza. DB-SRA assumes productivity (in relation to biomass) is constant through time, and the single M-stanza run assumes no change in productivity. Within the single M-stanza runs, model sensitivity to the magnitude and duration of early catches (pre-1900), as well as the effect of starting the model at different time periods (1880, 1925, 1970) was explored. It was acknowledged in the assessment that productivity has likely declined for American eels, largely due to the loss of eel habitat from dams. To account for this potential decline in eel productivity through time, a double-M stanza model was run whereby M was increased and FMSY / M was decreased (thus assuming total mortality that produces MSY (ZMSY) is constant). The sensitivity of the timing and magnitude of this increase in M was explored.

The double M-stanza approach was deemed the preferred approach. Allowing for changes in productivity through time is a novel modification to DB-SRA. The Panel agreed this modification has the potential to be very useful, allowing for the application of DB-SRA to a wider range of species believed to have historical changes in productivity. The Panel discussed additional ways of characterizing a loss of productivity in the model. For example, given the loss of eel habitat through the damming of waterways, one might expect that the carrying capacity (K) of the population has been greatly reduced. Therefore, decreasing K through time could also account for the loss in productivity in the DB-SRA model, and doing so avoids using the assumption that as M increases F_{MSY} decreases (see also ToR 2). The Panel recommended that a sensitivity run with a lower K in recent years be explored. This run was conducted and showed promise, but the limited time for this analysis prevented full consideration of the analysis.

In DB-SRA, a single parameter, K, is estimated, and important management quantities (MSY, B_{MSY}, $B_{CURRENT}$ / B_{MSY}, $F_{CURRENT}$ / F_{MSY}) are determined using this estimate along with the input parameters (note that F_{MSY} is determined solely by the input parameters). In general, model estimates of K and MSY were robust across a wide range of parameter values and across sensitivity runs. However, at higher levels of B_{MSY} / K, M, and F_{MSY} / M, unreasonably high estimates of K resulted, suggesting such values might be not be plausible for eels. In addition, estimates of K were similar across runs that started at different years (1880, 1925, and 1970). The Panel noted, however, that regardless of the starting year for model runs, biomass initially declined very rapidly. For example, in the preferred model run, biomass declined by about 90% in the first 10 years; the Panel wondered whether such a rapid decline was possible. Runs that explored earlier start years with a gradual increase in catches (up to 1880) showed a more gradual decline, but again, biomass still reached near historical lows by 1890. The Panel wondered if there was any evidence (anecdotal or otherwise) to support such a low population in the 1890s.

Estimates of current biomass from the DB-SRA model are dependent upon the input value of $B_{CURRENT}$ / K. In addition, current biomass estimates combined with catch levels determine the current exploitation rate. Thus, although some of the derived management quantities were robust across runs, the estimates of current stock status relative to management reference points are extremely sensitive to the range of $B_{CURRENT}$/ K. The assumed range of $B_{CURRENT}$ / K in the preferred model run was between 0.05 and 0.15. The motivation for use of this range is that a $B_{CURRENT}$ / K value of 0.1 tended to match recent trends in biomass in the 20-30 year coast wide indices. Between 1991 and 2010, both the 20 and 30 year indices showed a roughly 10% increase in biomass, and a $B_{CURRENT}$ / K of 0.1 in a number of model runs resulted in a similar increase across a range of other parameters. Use of trends in the available indices is a potentially productive way to help parameterize the model, particularly the values for $B_{CURRENT}$ / K. However, the Panel was concerned about the model estimates of current stock status and harvest rates being entirely dependent upon the average increase of 10% based on two coast wide indices of abundance (see ToR 4 for uncertainty about the strength of the trend in these indices). The Panel agreed a wider range of $B_{CURRENT}$ / K should be explored, perhaps between 0.05 and 0.3, with the distribution being centered at different values within this range.

Along with additional explorations of the range of $B_{CURRENT}$ / K, there was consensus amongst the Panel that later ages at maturity should be explored. In addition, the DB-SRA exploration would benefit from incorporating uncertainty into the catch series, either based on empirical estimates of uncertainty or on some ad-hoc approach.

Overall, the Panel was impressed with the development of the DB-SRA model for American eels. The SASC explored a wide range of possible models, and used a few novel approaches to overcome some of the model assumptions (i.e. 2 productivity stanzas) and to better inform model parameterization (i.e. using an index trend to select $B_{CURRENT}$ / K). *However, the Panel was not comfortable relying entirely on the trend in this index to center the $B_{CURRENT}$ / K distribution at 0.1, as doing so automatically resulted in the estimated eel population being overfished in the final year.*

4. Evaluate the assessment's best estimates of stock biomass, abundance, and exploitation for use in management, if possible, or specify alternative estimation methods.

There is uncertainty in the magnitude of biomass and fishing mortality estimates from the DB-SRA model, particularly in recent years. However, general patterns in the estimates can be discerned from the model runs. Estimated biomass declined rapidly between 1880 and 1890, and reached historical lows in the early 1900s. Biomass increased gradually starting around 1910, reaching a peak in the early 1970s (but below the biomass in the beginning of the time series). The very high catches in the late 1970s and early 1980s resulted in a rapid decline in biomass until the mid 1990s. It is unclear what the biomass trend in recent years is because this trend depends on the assumed level of $B_{CURRENT}$ / K.

Exploitation rates fluctuated greatly over time, but there were 3 periods of high to very high exploitation rates (approximately between 1890-1910, 1930-1936, and 1978-1995). The highest exploitation rates occurred in the early 1900s, and these values may have been over 5 times higher than the estimated FMSY. Trends in recent exploitation rates are uncertain as they are sensitive to BCURRENT / K.

Abundance indices were not available for most of the time period and thus could not be used to support the DB-SRA trends in biomass in the early years. Trend analyses of abundance indices for more recent years suggested declining or stable abundance of eels in recent decades. The 30-year GLM-estimated index of coast wide abundance showed a decline in biomass in the early to mid-1980s until about 1990. The DB-SRA estimates of biomass showed a decline a few years earlier, but the general trends were in agreement. However, the 40-year GLM index only used indices of abundance from the Chesapeake, and did not match the trends in estimated biomass since the 1970s.

In summary, the estimates of biomass showed two periods of high biomass: during the early 1880s, and from about 1965-1980. As referenced in ToR 3, the Panel questioned whether such a rapid decline in biomass could have occurred in the 1880s, and if eel biomass could have been at such low levels starting around 1890. In addition, current estimates of coast wide biomass could not be determined due to the sensitivity of this estimate to the assumed distribution for $B_{CURRENT}$ / K.

5. Evaluate the choice of reference points and the methods used to estimate reference points. Evaluate the stock status determination from the assessment. If appropriate, specify alternative methods/measures.

The SASC determined three sets of reference points (RPs) – the first based on an ARIMA analysis of the 20 year (or more) coast wide yellow eel survey index, the second using the Traffic Light Approach (TLA) and the third based on the results of the DB-SRA model. The ARIMA derived RPs were proposed as the lower 25th percentile of the fitted abundance index. It was further suggested that a high probability (i.e. 80%) of the current year's index being below this level would provide strong evidence that the stock biomass is below the RP. The Panel considers the utility of this RP as limited. It is not clear what management action should be taken if and when an RP is met or exceeded as the RP is not derived from stock dynamics which could be used to inform a desired management response.

The TLA was applied to all individual, regional, and coast wide indices of relative abundance by the SASC. After scaling, each annual index was assigned to one of three color categories - white (good), gray (intermediate), or black (bad) - based on the 25th and 75th percentiles of each index series (see also ToR 1). The results were complex and difficult to interpret. Nonetheless, empirically-based RPs of this nature have been used in stocks (e.g. Hardie et al., 2011) for which population models are not available. As part of a TLA, they are one metric in a suite of many to inform managers of stock status. Pre-agreed upon decisions on management actions are made if and when RPs are met. The TLA is not without its problems but can allow management actions to ensure stock sustainability in data-poor situations (Halliday et al., 2001). Further, the TLA allows consideration of a wider suite of information than can normally be incorporated into a model (e.g. environmental indicators), thus allowing interpretation of model results in a broader context. *The Panel suggests that a TLA be explored which would incorporate a wide array of data related to American eel stock dynamics. This may be used to assist in coast wide and regional management decision-making while modeling efforts continue.*

The two *M* stanza DB-SRA provided American eel stock RPs which were relatively robust to input assumptions. The carrying capacity (K) ranged from 16,274 - 23,595 t (median of 18,274t). B_{MSY} ranged from 5,085 - 8,912t (median of 6,823t) while MSY ranged from 827 - 1510t (median of 1,060t). The associated F_{MSY} ranged 0.14 - 0.26 (median of 0.19). The Panel considered, however, that while these RPs were generally representative of optimal stock dynamics, the uncertainties in the DB-SRA model did not permit statements on current stock status in relation to these RPs.

In summary, the Panel is very encouraged by the modeling efforts of the SASC and finds they are a significant advance since the 2006 assessment (see also ToR 3). Notwithstanding this, while it is highly likely that the American Eel stock is depleted, the overfishing and overfished status in relation to the biomass and fishing mortality reference points cannot be stated with confidence.

6. Review the research, data collection, and assessment methodology recommendations provided by the Technical Committee and make any additional recommendations warranted. Clearly prioritize the activities needed to inform and maintain the current assessment, and provide additional recommendations that may improve the reliability of future assessments.

The recommendations provided by the SASC were fairly comprehensive and the Panel feels these covered the primary areas needed to improve future assessments. The Review Panel has incorporated these recommendations into Table 1, with prioritization and comments explaining the priority provided.

7. Recommend timing of the next benchmark assessment and updates relative to the life history and current management of the species.

The Panel recommends timing the next benchmark to permit the collection of additional data and assess progress with regard to the Panel's recommendations. This would be at a minimum 5 years from the current benchmark. This is also in keeping with the long generation time for eels (3-5 years in south, 10-20 years in north).

The Panel also concurs with the SASC's suggestion that the next benchmark assessment be conducted together with the corresponding Canadian team. To this end, it was suggested that a planning meeting be convened at the 2014 AFS meeting, which will be held in Quebec City.

Advisory Report

A. Status of stocks: Current and projected, where applicable

The Panel review concluded the American eel population is *depleted* in U.S. waters. The stock is at or near historically low levels. This is likely due to a combination of historical overfishing, habitat loss due to damming mainstems and tributaries of rivers, mortality from passing through hydroelectric turbines, pollution, possibly parasites and disease, and unexplained factors at sea.

A depletion-based stock reduction analysis (DB-SRA) was conducted by the Stock Assessment Subcommittee (SASC); results suggested overfishing has been occurring since the 1980s. However, while it is highly likely the American eel stock is depleted, the overfishing and overfished status in relation to the biomass and fishing mortality reference points cannot be stated with confidence (see ToRs 2, 3 and 5).

B. Stock Identification and Distribution

The American eel is a panmictic species, that is, a single, genetically homogeneous population. This is due to having a single spawning region in the Sargasso Sea. After hatch, American eel leptocephali (the larvae) drift with currents in a generally westward direction, but encounter both the North and South American continents. Consequently, the distribution of American eel ranges from northern South America, into the Gulf of Mexico, and along the North American east coast as far as Labrador and Greenland. As a partially catadromous species (Daverat et al. 2006), American eel colonized a wide range of inland waters, penetrating as far inland as Lake Ontario and its drainages, and the Mississippi River as far as Iowa (Tesch 2003). There is overlap on the spawning grounds with the European eel, *Anguilla anguilla*, and a hybrid zone is found in Iceland (Albert et al. 2006).

Although panmictic, there are distinct, habitat-related trends in size and sex ratio in anguillid eels (e.g., Oliveira 1999, Davey and Jellyman 2006). Sex determination is at least to some extent environmentally determined and appears to be a function of density and growth rate, with males arising at higher local population densities. These differences appear to produce females that are larger and therefore more fecund (but take longer to mature) and males that mature as quickly as possible (Davey and Jellyman 2006). Therefore, loss of larger, older females in the female-dominated Laurentian Great Lakes drainage, and possibly other areas where females are produced, is cause for concern.

C. Management Unit

From the draft stock assessment Executive Summary, p. iv:

“The management unit for American eel under the jurisdiction of ASMFC includes that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida. The goal of the American Eel Fishery Management Plan (approved November 1999) is to conserve and protect the American eel resource to ensure ecological stability while providing for sustainable fisheries."

As noted in the last stock assessment peer review (ASMFC 2006), because of the wide range (over 50 degrees of latitude) and geographic biological differences in this panmictic species (see above), management of eels in U.S. waters must also consider status of eels beyond U.S. territory. This would at a minimum include coordination with Canada and Caribbean countries.

D. Landings

Earliest Federal records of eel fishing date from the late 19th century, but eel fishing has been documented back to the 17th century. Gear ranges from traditional spears to pots, pound nets, and weirs. During the 20th century, heaviest fishing pressure occurred in response to demand from Europe beginning in the 1960s, and decline began to occur in the early 1980s (Figure 2). Harvests have been more or less constant since the previous stock assessment.

From the current stock assessment Executive Summary, p. iv:

"During 1950 to 2010, American eel landings from the U.S. Atlantic Coast ranged between approximately 664,000 pounds (301.2 MT) in 1962 and 3.67 million pounds (1664.7 MT) in 1979. After a decline in the 1950s, landings increased to a peak in the 1970s and 1980s before declining again in the 2000s. The value of U.S. commercial American eel landings as estimated by NMFS has varied between a few hundred thousand dollars (prior to the 1980s) and a peak of $6.4 million in 1997. Total landings value increased through the 1980s and 1990s, dropped in the late 1990s, and increased again in the 2000s.

"Since 1950, the majority (>76%) of American eel commercial landings were caught in pots and traps. Fixed nets (e.g., weirs, pound nets) accounted for about 8% of the landings. Approximately 4% of landings were caught using other gears (non-pot/trap or fixed net). About 12% of landings are reported with unknown gear type. Over the last two decades, pots and traps have become the dominant gear reported for most eel landings."

A glass eel fishery arose in the 1970s in response to demand from Japan. High prices for glass eels periodically drove up effort in this fishery; currently demand is at a record high, due to a shortage of Japanese eels in the wake of the 2011 tsunami and its impacts. Prices currently top $2000/pound (NYT 2012). The glass eel fishery is legal only in the states of Maine and South Carolina, but the high market prices are an encouragement to poaching in many states.

E. Data and Assessment

Data sets were canvassed from as many sources as possible and trends were examined. Fishery-dependent data were examined, but not used in the actual assessment. Fishery-independent data sets were standardized with generalized linear models (GLMs), then analyzed for the ability to detect trends (power analysis), monotonic trends (Mann-Kendall tests), coherence of trends over space (via meta-analysis), and general temporal and geographic trends (geographically based time series (ARIMA) modeling, traffic light analysis). The results indicated variable responses, but most of the data sets indicated decline. See ToR 1 for further elaboration, as well as discussion of data sets.

F. Biological Reference Points

Three approaches were used to create biological reference points. The first was to use ARIMA models with standardized abundance index data sets of at least 20 years' length, to estimate the probability that the abundance in any given year (particularly later years) was less than the 25th percentile of the data in the time series. The ARIMA analysis yielded low probabilities of decline, except for the Hudson River, western Long Island, and the North Carolina estuarine trawl survey. The Panel noted some difficulties with undue weight given to the first datum of the time series (see ToR 1), and interpreting the utility of this as a reference point (see ToR 5).

The second approach was to undertake a 'Traffic Light Approach' by grouping different assessments within geographic regions and years, coding them as indicating 'good', 'intermediate', and 'bad' in terms of percentiles of ranges. The results were complex and difficult to interpret. Nevertheless, the Panel felt the TLA approach could be refined to include more indices – including environmental and habitat indices – related to eel population dynamics.

The third approach was to use depletion-based stock reduction analysis (DB-SRA; Dick and MacCall 2011). Details of the model are found in the stock assessment report and are further discussed in ToRs 2-4 above. As noted in ToR 5, the analysis that assumed two different temporal stanzas of natural mortality ("two M stanza DB-SRA"), where M increased after 1970 to reflect the increased impacts of dams on eel mortality, was robust to different input assumptions, and produced a range of estimates of carrying capacity (K), biomass at MSY (B_{MSY}), and fishing mortality at MSY (F_{MSY}). However, due to uncertainties discussed above, the Panel felt it was not possible to determine current stock status in relation to these reference points.

G. Fishing Mortality

The SASC has made progress in assessing fishing mortality (F) through development of the DB-SRA. While trends in F can be discerned from the model, estimates from recent years are uncertain, as they depend on the assumed level of current depletion. Therefore, the results are tentative, and more analysis is needed.

H. Recruitment

As noted in ToR 1, the young-of-year (YOY) indices that began in 2000 or later show few trends; a longer Hudson River YOY index showed a declining trend; and the ichthyoplankton indices may show a recent, sharp decline (see ToR 1 for discussion). The 2005 stock assessment review noted the value of long term trawl data sets, such as that from VIMS, but trends were difficult to discern because age and size data were not available. The SASC attempted to obtain size data for the VIMS survey, but there were issues in the data that require further exploration.

The Panel strongly supports the recommendation of the SASC to continue the YOY monitoring programs, to encourage all states to participate with comparable, standardized data collection and reporting protocols, and to obtain size- or age-based trend data from VIMS and other long term sources, if possible.

I. Spawning Stock Biomass

The magnitude of spawning stock biomass (SSB) is difficult to assess due to uncertainties in abundance estimates, growth rates (which are variable in eels) and population productivity. And, an unknown fraction of the spawning stock is outside U.S waters. The DB-SRA calculated SSB values that would produce the observed abundance trends, but these are as yet unvalidated.

J. Bycatch

Eel bycatch is not considered to be a major problem. Eels are caught incidentally by recreational fishers, and the Marine Recreational Information Program (MRIP) does list American eel as a bycatch species. The stock assessment report notes that bycatch reported by MRFSS has declined from an average of ca. 22 MT/year in the 1980s to 4 MT/year in the 2000s, but even steeper declines have occurred in the North Atlantic region (see the American Eel Stock Assessment Report, pp. 47-48).

Some eel bycatch information (e.g., from rainbow smelt fisheries in Massachusetts) may be of value as indices of abundance or catch per unit effort (e.g. Figure 5.37 of Stock Assessment report). However, eel capture efficiencies in these fisheries are unknown and would need to be determined.

K. Other Comments

In general, the Panel was satisfied with the progress made by the SASC and encourages them to continue working on the new approaches developed for this stock assessment. The Panel also agreed with the research recommendations of the SASC for further improvements to the stock assessment (Table 1).

Given the unique life history and biology of anguillid eels, which defy national boundaries, it is important to devise means to manage the American eel to account for the contributions of and threats to the portion of the population outside the U.S. Ideally, there would be an ‘International Northwest Atlantic Eel Council’; the American Eel Technical Committee has approached their counterparts in Canada, which is a good start.

As data accumulate and models improve, the SASC is encouraged to further integrate the data and models. In addition, models that explore the stochastic variability of eel growth and its implications for fisheries could integrate such environmental variables as climate, dams, turbines, pollution, and habitat alterations. These or other models would ideally explore the marine phases for recruitment and reproduction, both of which are critical but largely unknown.

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M. Tables

Table 1. Review Panel evaluation and prioritization of American eel research recommendations. Red text indicates recommendations the Technical Committee and SASC presented as improvements needed for the next benchmark assessment.

Research Recommendation	Time Period	Priority	Review Panel Comments
Data Collection			
Fisheries Catch and Effort			
Improve accuracy of commercial catch and effort data			
Compare buyer reports to reported state landings.	Short term	Moderate to high	The Panel agrees these measures could provide a more reliable measure of 'actual' landings.
Improve compliance with landings and effort reporting requirements as outlined in the ASMFC FMP for American eel (see ASMFC 2000a for specific requirements).	Short term		
Require standardized reporting of trip-level landings and effort data for all states in inland waters; data should be collected using the ACCSP standards for collection of catch and effort data (ACCSP 2004).	Short term		
Estimate catch and effort in personal-use and bait fisheries			
Monitor catch and effort in personal-use fisheries that are not currently covered by MRIP or commercial fisheries monitoring programs.	Short term	High	The recommendations would provide for a better understanding of this apparent major source of eel exploitation in U.S. waters.
Implement a special-use permit for use of commercial fixed gear (e.g., pots and traps) to harvest American eels for personal use; special-use permit holders should be subject to the same reporting requirements for landings and effort as the commercial fishery.	Long term	High	
Improve monitoring of catch and effort in bait fisheries (commercial and personal-use).	Short term	High	
Estimated non-directed fishery losses			
Recommend monitoring of discards in targeted and non-targeted fisheries.	Short term	Low to Moderate	Bycatch of American eel is considered minor and MRIP data show it declining since the 1980s.

Table 1 (Cont'd).

Research Recommendation	Time period	Priority	Review Panel Comments
Data Collection (cont'd)			
Fisheries Catch and Effort			
Continue to require states to report non-harvest losses in their annual compliance reports.	Short term	Moderate	If sources of non-harvest losses can be distinguished from passage issues (hydropower; below)
Characterize the length, weight, age, and sex structure of commercially harvested American eels along the Atlantic Coast over time			
Require that states collect biological information by life stage (potentially through collaborative monitoring and research programs with dealers) including length, weight, age, and sex through fishery-dependent sampling programs; biological samples should be collected from gear types that target each life stage; at a minimum, length samples should be routinely collected from commercial fisheries.	Short term	High	Data on age and sex (< 400mm) require sacrificing the eel; not be a feasible undertaking without the collaboration of fishers and dealers. Length and weight should be more readily available.
Finish protocol for sampling fisheries; SASC has draft protocol in development.	Short term	High	See above.
Improve estimates of recreational catch and effort			
Collect site-specific information on the recreational harvest of American eels in inland waters; this could be addressed by expanding the MRIP to riverine/inland areas.	Long term	Low-moderate	The recreational fishery appears to a great extent to be coupled with the bait fishery. The recommendations above should fulfill this need.
Improve knowledge of fisheries occurring south of the U.S. and within the species' range that may affect the U.S. portion of the stock (i.e., West Indies, Mexico, Central America, and South America).	Long term	Moderate-High	This region is an unknown contributor to the American eel spawning population. Its proximity to the spawning area makes this a worthwhile undertaking.

Table 1 (Cont'd).

Research Recommendation	**Time period**	**Priority**	**Review Panel Comments**
Data Collection (cont'd)			
Socioeconomic Considerations			
Perform economic studies to determine the value of the fishery and the impact of regulatory management.	Long term	Moderate	The extent of eel-specific fishers to the proportion of supplemental fishers is needed.
Improve knowledge regarding subsistence fisheries			
Review the historical participation level of subsistence fishers and relevant issues brought forth with respect to those subsistence fishers involved with American eel.	Long term	Low to moderate	The Panel agrees these recommendations may provide some insight into changing exploitation of the species.
Investigate American eel harvest and resource by subsistence harvesters (e.g., Native American tribes, Asian and European ethnic groups).	Long term	Low to moderate	
Distribution, Abundance, & Growth			
Improve understanding of the distribution and frequency of occurrence of American eels along the Atlantic coast over time			
Maintain and update the list of fisheries-independent surveys that have caught American eels and note the appropriate contact person for each survey.	Short term	High	A potentially valuable source of information; however, differing methodologies (i.e sampling gear and ageing) may complicate interpretation.
Request that states record the number of eels caught by fishery-independent surveys; recommend states collect biological information by life stage including length, weight, age, and sex of eels caught in fishery-independent sampling programs; at a minimum, length samples should be routinely collected from fishery-independent surveys.	Short term	High	Length data can be obtained fairly easily. See preceding caution.

Table 1 (Cont'd).

Research Recommendation	Time period	Priority	Review Panel Comments
Data Collection (cont'd)			
Encourage states to implement surveys that directly target and measure abundance of yellow- and silver-stage American eels, especially in states where few targeted eel surveys are conducted.	Long term	High	State implemented surveys may be the best way to control sampling bias and coordinate methods for the collection of all relevant biological data.
A coast wide sampling program for yellow and silver American eels should be developed using standardized and statistically robust methodologies.	Long term	High	See comments from previous three recommendations.
Improve understanding of coast wide recruitment trends			
Continue the ASMFC-mandated YOY surveys; these surveys could be particularly valuable as an early warning signal of recruitment failure.	Short term	High	The Panel agrees the YOY surveys present a valuable warning system for recruitment success or failure. However, a standardized sampling regime would enhance the value of these data.
Develop proceedings document for the 2006 ASMFC YOY Survey Workshop; follow-up on decisions and recommendations made at the workshop.			
Examine age at entry of glass eel into estuaries and freshwater.	Long Term	Moderate	Allows for better estimation of the time lag between spawner escapement and glass eel recruitment. Currently eel ages are only based on years in freshwater (or near freshwater).
Develop monitoring framework to provide information for future modeling on the influence of environmental factors and climate change on recruitment.	Long term	Moderate	A systematic method of gathering environmental and climate change data that can be linked to recruitment could provide the foundation for a working coast wide model.
Improve knowledge and understanding of the portion of the American eel population occurring south of the U.S. (i.e., West Indies, Mexico, Central America, and South America).	Long term	Moderate to high	As previously noted, the proximity of these regions to the spawning area may make their contribution of spawning valuable.

Table 1 (Cont'd).

Research Recommendation	Time period	Priority	Review Panel Comments
Future Research			
Biology			
Improve understanding of the leptocephalus stage of American eel			
Examine the mechanisms for exit from the Sargasso Sea and transport across the continental shelf.	Long term	Moderate	The understanding of larval migration and energetics could provide insight to declines in recruitment due to oceanic causes.
Examine the mode of nutrition for leptocephalus in the ocean.	Long term	Moderate	
Improve understanding of impact of contaminants as sources of mortality and non-lethal population stressors.	Short term	Moderate-high	Unfortunately, the biology of American eels (long lived, lipid rich, benthic) make them ideal for bioaccumulation of contaminants. USFWS currently has a project examining maternal transfer of contaminants in American eel.
Investigate the effects of environmental contaminants on fecundity, natural mortality, and overall health.	Long term	Moderate-high	
Research the effects of bioaccumulation with respect to impacts on survival and growth (by age) and effect on maturation and reproductive success.	Short term	Moderate-high	
Improve understanding of impact of *Anguillicoloides crassus* on American eel			
Investigate the prevalence and incidence of infection by the nematode parasite *A. crassus* across the species range.	Short term	Low-moderate	The parasite has already been documented throughout most of US Coast and the Canadian Maritime provinces. In 2011, it was found in Lake Ontario. However, the effect of the parasite on American eels and especially the spawning migration has yet to be established.
Research the effects of the swim bladder parasite *A. crassus* on American eel growth and maturation, migration to the Sargasso Sea, and spawning potential.	Short term	Moderate to high	
Investigate the impact of the introduction of *A. crassus* into areas that are presently free of the parasite.	Long term	Low to moderate	

Table 1 (Cont'd).

Research Recommendation	Time period	Priority	Review Panel Comments
Future Research (cont'd)			
Biology			
Improve understanding of spawning and maturation			
Investigate relation between fecundity and length and fecundity and weight for females throughout their range.	Long term	Low to moderate	Eel size-fecundity relationships have already been established. Effort would be better spent understanding the size variation in females.
Identify triggering mechanism for metamorphosis to mature adult, silver eel life stage, with specific emphasis on the size and age of the onset of maturity, by sex; a maturity schedule (proportion mature by size or age) would be extremely useful in combination with migration rates.	Long term	Moderate to high	As indicated above, these are valuable data. Important to conduct on a latitudinal and habitat level to allow for use in management.
Research mechanisms of recognition of the spawning area by silver eel, mate location in the Sargasso Sea, spawning behavior, and gonadal development in maturation.	Long term	Moderate	As previously noted for larval stages, an understanding of oceanic conditions (Gulf Stream shifts, etc.) may explain non-anthropogenic declines in recruitment.
Examine migratory routes and guidance mechanisms for silver eel in the ocean.	Long term	Moderate	
Improve understanding of predator-prey relationships.	Long term	Moderate	The Panel agrees. Smaller eels are readily preyed upon in all habitats. Larger females may have a size refuge during the freshwater phase.
Investigating the mechanisms driving sexual determination and the potential management implications.	Long term	High	Eels have sex specific life history strategies. The causes of sex determination would be of major importance to management.

Table 1 (Cont'd.)

Research recommendation	Time period	Priority	Review Panel Comments
Future Research (cont'd)			
Passage & Habitat			
Improve upstream and downstream passage for all life stages of American eels			
Develop design standards for upstream passage devices for eels; this will be a product (at least partial design guidelines) from the ASMFC 2011 Eel Passage Workshop; i.e., the research need may be partially met in the near term.	Short term	High	These are all a high priority recommendations but the Panel would like to emphasize the need to separate upstream and downstream passage. Upstream passage contributes primarily to habitat availability of yellow stage eels while downstream has a more direct and readily measured mortality effect on migrating silver stage eels.
Investigate, develop, and improve technologies for American eel passage upstream and downstream at various barriers for each life stage; in particular, investigate low-cost alternatives to traditional fishway designs for passage of eel.	Long term	High	
Improve understanding of the impact of barriers on upstream and downstream movement			
Evaluate the impact, both upstream and downstream, of barriers to eel movement with respect to population and distribution effects; determine relative contribution of historic loss of habitat to potential eel population and reproductive capacity.	Long term	High	As noted above, it may be more effective to focus on upstream passage and the effects on movement and habitat losses of yellow phase eels. Silver eel downstream access is not significantly reduced but rather impacted by factors such as turbine mortality.
Recommend monitoring of upstream and downstream movement at migratory barriers that are efficient at passing eels (e.g., fish ladder/lift counts); data that should be collected include presence/absence, abundance, and biological information; provide standardized protocols for monitoring eels at passage facilities; coordinate compilation of these data; provide guidance on the need and purpose of site-specific monitoring.	Long term	Moderate	

Table 1 (Cont'd.)

Research recommendation	Time period	Priority	Review Panel Comments
Future Research (cont'd)			
Passage & Habitat			
Improve understanding of habitat needs and availability			
Assess characteristics and distribution of American eel habitat and value of habitat with respect to growth and sex determination; develop GIS of American eel habitat in U.S.	Long term	Moderate	This will have to be a habitat-specific analysis. Past studies show high habitat-specific variability in sex ratios within a drainage system.
Improve understanding of habitat needs and availability			
Assess available drainage area over time to account for temporal changes in carrying capacity; develop GIS of major passage barriers.	Long term	Low-moderate	Following possible changes (GIS) in carrying capacity could also provide an understanding of sex ratio changes.
Improve understanding of within-drainage behavior and movement and the exchange between freshwater and estuarine systems.	Long term	Moderate	Allows for better understanding of habitat use and movement between habitats. May also provide needed data for regions where fisheries are either estuarine or freshwater based.
Improve estimates of mortality associated with upstream and downstream passage			
Monitor non-harvest losses such as impingement, entrainment, spill, and hydropower turbine mortality.	Short term	High	In river systems with hydropower, it is essential to have these data; a substantial source of mortality that must be accounted for.
Evaluate eel impingement and entrainment at facilities with NPDES authorization for large water withdrawals; quantify regional mortality and determine if indices of abundance could be established as specific facilities.	Long term	Moderate	See above.

Table 1 (Cont'd.)

Research Recommendation	Time period	Priority	Review Panel Comments
Future Research (cont'd)			
Assessment Methodology & Management Support			
Coordinate monitoring, assessment, and management among agencies that have jurisdiction within the species' range (e.g., ASMFC, GLFC, Canada DFO).	Short term	High	The Panel gives this recommendation a very high priority. The Panmictic nature of eel and the amount of the species' range within Canadian waters makes any solely U.S. based assessment incomplete.
Perform a joint U.S.-Canadian stock assessment.	Short term	High	
Perform periodic stock assessments (every 5-7 years) and establish sustainable reference points for American eel; required to develop a sustainable harvest rate in addition to determining whether the population is stable, decreasing, or increasing.	Short term	Moderate to high	Periodic assessment is needed but a longer time interval (8-10 years) may better estimate population trends. This longer time period may better reflect the eels generation time.
Develop new assessment models (e.g., delay-difference model) specific to eel life history and fit to available indices.	Long term	Moderate to High	Alternate models that do not rely on tracking ages may prove useful but the complex life history makes this difficult.
Conduct intensive age and growth studies at regional index sites to support development of reference points and estimates of exploitation.	Short term	Moderate to high	In order for these data to be of use, standardization of sampling gear, habitat, and ageing methods must first be completed.

Table 1 (Cont'd.)

Research Recommendation	Time Period	Priority	Review Panel Comments
Future Research (cont'd)			
Develop GIS-type model incorporating habitat type, abundance, contamination, and other environmental factors.	Long term	Low to Moderate	The models would be useful if all factors influencing abundance are included (i.e dams and all fisheries).
Develop population targets based on habitat availability at the regional and local level.	Long term	Low to Moderate	Population targets would be most useful if developed at the local (habitat) level. Regional variation is typically very large.
Implement large-scale (coast-wide or regional) tagging studies of eels at different life stages; tagging studies could address a number of issues including: – Growth – Passage mortality – Movement, migration, and residency – Validation of ageing methods – Reporting rates – Tag shedding or tag attrition rates	Long term	Moderate to-high	A far-reaching recommendation that the Panel feels has good potential. Current long term tagging studies in the St. Lawrence River System have begun to provide data on several of these questions. Some regions would require a long time lag (10 plus years) to address questions.

N. Figures

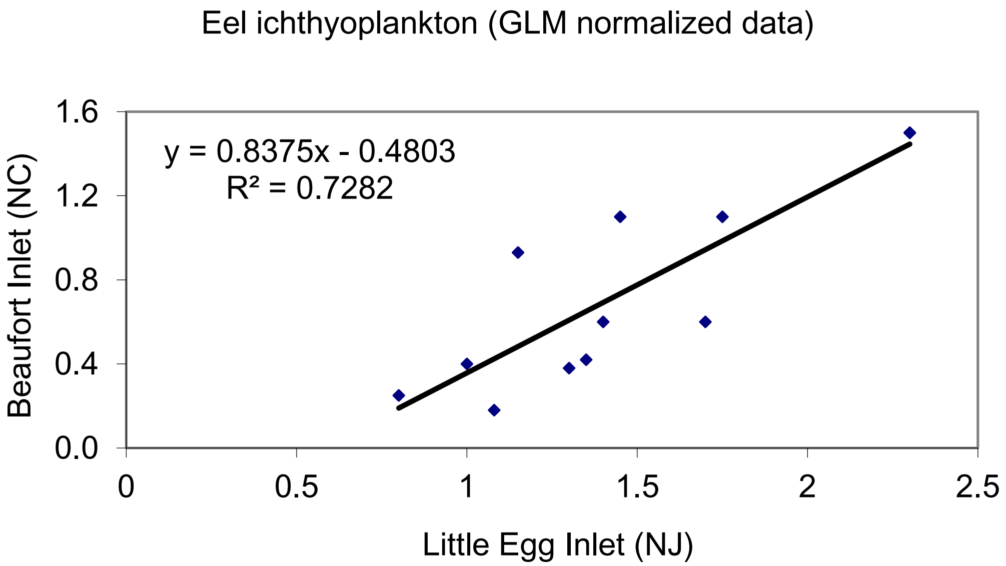


Figure 1. Regression of eel leptocephali indices from Beaufort Inlet, NC on Little Egg Inlet, NJ. The high leverage point consists of two superimposed points.

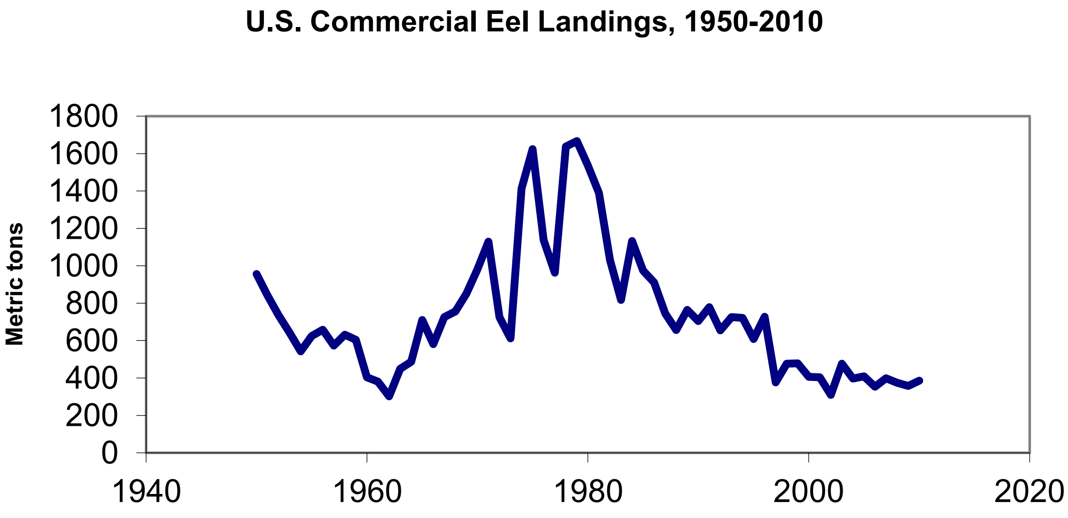
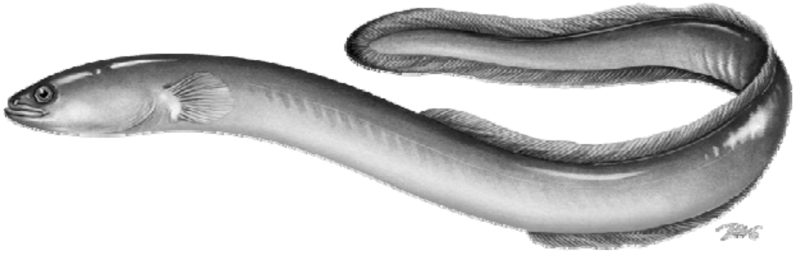


Figure 2. Commercial landings of American eel. Data source: NOAA Fisheries.

American Eel Stock Assessment for Peer Review



Prepared by the
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Atlantic States Marine Fisheries Commission

Working towards healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015

Atlantic States Marine Fisheries Commission

American Eel Stock Assessment for Peer Review

Preface

An External Peer Review Panel of independent experts met in March 2012 to review the American Eel Stock Assessment and concluded, based on the data and analyses performed in the assessment that the American eel stock was depleted. However, the Panel recommended that the Depletion-Based Stock Reduction Analysis (DB-SRA), initially recommended by the ASMFC American Eel Technical Committee (TC) for use in setting overfished and overfishing stock status determinations, undergo additional testing and development before it is used to generate reference points for management. Following the Peer Review Workshop, the TC and American Eel Stock Assessment Subcommittee (SAS) reviewed the Peer Review Panel's Terms of Reference and Advisory Report and agreed that further development of the DB-SRA is needed.

The Peer Review Panel also suggested the term 'depleted' is more appropriate for describing American eel stock status given the combination of causes for decline, including significant levels of harvest in the 1970s, habitat loss, passage impediments and mortality, disease, and potentially shifting oceanographic conditions. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant downward trends in numerous indices over the time period examined. Also, the DB-SRA indicated that the stock is at low biomass compared to previously high levels observed in the 1970s. The TC and SAS agreed with the Peer Review Panel that the stock assessment indicated ***the stock is depleted***. ***No overfishing determination can be made at this time*** based solely on the trend analyses performed (i.e., without finalized DB-SRA results). However, the TC and SAS caution that although commercial fishery landings and effort in recent times have declined in most regions (with the possible exception of the glass eel fishery), ***current levels of fishing effort may still be too high given the additional anthropogenic and environmental stressors affecting the stock.*** Fishing on all life stages of eels, particularly YOY and out-migrating silver eels, could be particularly detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled. ***Management efforts to reduce mortality on American eels in the U.S. are warranted.***

Note that statements highlighted in yellow below have been modified by the TC following the Peer Review Workshop and are accompanied by a footnote explaining the wording change made by the TC with regards to stock status.

DEDICATION

To the eel from the River Neuse….

ACKNOWLEDGEMENTS

The ASMFC American Eel Technical Committee: Keith Whiteford (MD, Chair), Brad Chase (MA, Vice Chair), Jennifer Pyle (NJ), Michael Kaufmann (PA), Jessica Fischer (NH), Tim Wildman (CT), John Whitehead (Appalachian State University), Gail Wippelhauser (ME), Eric Thadey (DC), Richard Maney (NOAA Fisheries), Katy West (NC), Carl Hoffman (NY), Alex Haro (USGS), Allan Hazel (SC), Patrick Geer (GA), Phil Edwards (RI), Shelia Eyler (USFWS), John Clark (DE), Ellen Cosby (PRFC), Theodore Bestor (Harvard University), and Kimberly Bonvechio (FL).

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EXECUTIVE SUMMARY

The management unit for American eel under the jurisdiction of ASMFC includes that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida. The goal of the American Eel Fishery Management Plan (approved November 1999) is to conserve and protect the American eel resource to ensure ecological stability while providing for sustainable fisheries.

In the U.S., all life stages are subject to fishing pressure, and the degree of fishing also varies through time and space. Glass eel fisheries are permitted in Maine and South Carolina. Yellow and silver eel fisheries exist in all Atlantic Coast states with the exception of Pennsylvania. Eels are harvested for food, bait, and export markets.

During 1950 to 2010, American eel landings from the U.S. Atlantic Coast ranged between approximately 664,000 pounds in 1962 and 3.67 million pounds in 1979. After a decline in the 1950s, landings increased to a peak in the 1970s and 1980s before declining again in the 2000s. The value of U.S. commercial American eel landings as estimated by NMFS has varied between a few hundred thousand dollars (prior to the 1980s) and a peak of $6.4 million in 1997. Total landings value increased through the 1980s and 1990s, dropped in the late 1990s, and increased again in the 2000s.

Since 1950, the majority (>76%) of American eel commercial landings were caught in pots and traps. Fixed nets (e.g., weirs, pound nets) accounted for about 8% of the landings. Approximately 4% of landings were caught using other gears (non-pot/trap or fixed net). About 12% of landings are reported with unknown gear type. Over the last two decades, pots and traps have become the dominant gear reported for most eel landings.

A new set of watershed-based geographic regions were created for this assessment— the Gulf of Maine, Southern New England, Hudson River, Delaware Bay/Mid-Atlantic Coastal Bays, Chesapeake Bay, and the South Atlantic. The South Atlantic and Chesapeake Bay regions showed distinct large peaks in commercial landings in the early 1980s. Landings in all regions declined throughout the 1990s. Most regions remained stable throughout the 2000s except for Southern New England and Delaware Bay/Mid-Atlantic Coastal Bays where landings declined.

For this assessment, the committee evaluated nearly 100 fishery-dependent and independent U.S. data sources representing several life stages and geographical and temporal scales. Fifty-two fishery-dependent and independent data sources were selected for use in this assessment because they were considered adequate for describing life history characteristics and abundance trends of eels on either a coast-wide or regional basis. Trends in fishery-dependent CPUE were used to describe the fisheries but were not included in analyses because they were not thought to represent trends in eel abundance over time due to either poor participation in the fishery (i.e., few fishers), major unquantified changes in the fishery over time, or insufficient time series. Reasons for exclusion of a fishery-independent survey or sampling program included:

- Lacked sufficient time series to identify trends (<10 years)
- Reported inconsistent sampling methodology (i.e., frequent changes in survey methodology) that could not be accounted for via standardization techniques
- Intermittent or rare catches of eels

- Operated during a time of the year when or in an area where eel are not typically available to sampling gear
- Used survey gear with rare, uncertain, or biased catchability for eel

Very few fishery-independent surveys target American eels (with the exception of the state-mandated young-of-year surveys and a few surveys in Maryland). All fishery-independent surveys used in this assessment were evaluated using a standard set of criteria that resulted in data-based decisions to inform the analytical framework (primary assumptions regarding the error structure) for each survey independently. Application of these criteria resulted in nearly all surveys being standardized (unless otherwise noted) using a generalized linear model to account for changes in catchability of eels.

Trend analyses of abundance indices provided evidence of declining or, at least, neutral abundance of American eels in the U.S in recent decades. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant downward trends in numerous indices over the time period examined. The Mann-Kendall test detected a significant trend in the 30-year index of coast-wide yellow-phase abundance. The Manly meta-analysis showed a decline in at least one of the indices for both yellow and YOY life stages. Also, there was consensus for a decline for both life stages through time. Both the ARIMA and Mann-Kendall analyses indicate decreasing trends in the Hudson River and South Atlantic regions. In contrast, survey indices from the Chesapeake Bay and Delaware Bay/Mid-Atlantic Coastal Bays regions showed no consistent increasing or decreasing trends. Overall, however, the prevalence of significant downward trends in multiple surveys across the coast is cause for concern.

In addition to trend analyses, historical and recent commercial landings data were used to perform a Depletion-Based Stock Reduction Analysis (DB-SRA). The DB-SRA showed a coast-wide decline in stock biomass since the 1980s. ~~Based on DB-SRA results, the American eel resource in the U.S. is below the overfished threshold and above the overfishing threshold.~~ ~~***Therefore, the stock is overfished and overfishing is occurring relative to MSY-based reference points, given the assumptions made (particularly the depletion level and*** **B_{MSY}/K*****).***~~ ~~The Technical Committee agrees with the DB-SRA model conclusion that overfishing is occurring and that current biomass is below the estimated biomass threshold; however, while the term "overfished" is used to define this condition in terms of the model,~~[1] it is important to recognize that multiple sources of mortality have been contributing to the reduced biomass levels. Significant levels of harvest in the 1970s, loss of habitat, and predation are some of the major contributing factors to the overfished status in the DB-SRA base model results.

Although commercial fishery landings and effort in recent times have declined in most regions (with the possible exception of the glass eel fishery), current levels of fishing effort may still be too high given the additional stressors affecting the stock such as habitat loss, passage mortality, and disease as well as climate change leading to shifting oceanographic conditions. Fishing on all life stages of eels, particularly YOY and out-migrating silver eels, could be particularly detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing

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[1] In accordance with the findings of the Peer Review Panel regarding the need for further development of the DB-SRA, the Technical Committee does not recommend using DB-SRA-derived reference points at this time. Based on the results of the trend analyses and the biomass trends predicted by the DB-SRA, ***the stock is declared depleted***. No overfishing declaration can be made at this time.

oceanographic conditions) cannot be readily controlled. Management efforts to reduce mortality on American eels in the U.S. are warranted. Collaboration with Canada to cooperatively monitor, assess, and manage American eels should provide a more complete and accurate picture of the resource.

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TERMS OF REFERENCE

1. Evaluate precision and accuracy of U.S. Atlantic Coast fishery-dependent and fishery-independent data used in the assessment, including the following but not limited to:
    - Discuss the effects of data strengths and weaknesses (e.g., temporal and spatial scale, gear selectivities, ageing accuracy, sample size, standardization of indices) on model inputs and outputs.
    - Report standard errors of inputs and use them to inform the model if possible.
    - Justify weighting or elimination of available data sources.
2. Evaluate adequacy, appropriateness, application, and uncertainty of models or other analytical methods for use in the assessment of the species and estimating U.S. Atlantic Coast population benchmarks.
    - Did the model have difficulty finding a stable solution? Were sensitivity analyses for starting parameter values, priors, etc. and other model diagnostics performed?
    - Have the model strengths and limitations been clearly and thoroughly explained?
    - If using a new model, has it been tested using simulated data?
    - Has the model theory and framework been demonstrated and documented in the stock assessment literature?
3. State and evaluate assumptions made for all models and explain the likely effects of assumption violations on synthesis of input data and model outputs.
4. Recommend U.S. Atlantic Coast stock status as related to reference points (if available). For example:
    - Is the stock below the biomass threshold?
    - Is F above the threshold?
5. Develop detailed short and long-term prioritized lists of recommendations for future research, data collection, and assessment methodology. Highlight improvements to be made by next benchmark review.

1 INTRODUCTION

1.1 Fisheries Management

1.1.1 Management Unit Definition

The American eel (*Anguilla rostrata*) is one of two catadromous species in North America and historically occurred in all major rivers from Canada through the Brazil. The management unit for American eels under the jurisdiction of ASMFC includes that portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

1.1.2 Regulations & Management

1.1.2.1 Commercial Fishery Management

The ASMFC American Eel Management Board first convened in November 1995 and finalized the Fishery Management Plan (FMP) for American Eel in November 1999 (ASMFC 2000a). The major goal of the FMP is to conserve and protect the American eel resource to ensure ecological stability while providing for sustainable fisheries. Each state is responsible for implementing management measures within its jurisdiction to ensure the sustainability of the American eel population that resides within state boundaries. The FMP requires that all states and jurisdictions implement an annual young-of-year (YOY) abundance survey by 2001 in order to monitor annual recruitment of each year's cohort. In addition, the FMP requires all states and jurisdictions to establish a minimum recreational size limit of six inches and a recreational possession limit of no more than 50 eels per person, including crew members involved in party or charter (for-hire) employment for bait purposes during fishing. Recreational fishermen are not allowed to sell eels without a state license. Commercial fisheries management measures stipulate that states and jurisdictions shall maintain existing or more conservative American eel commercial fishery regulations for all life stages. States with minimum size limits for commercial eel fisheries must retain those minimum size limits, unless otherwise approved by the American Eel Management Board. Current commercial fisheries regulations can be found in Table 1. In addition, the ACCSP will require a comprehensive permit/license system for all commercial dealers and fishermen.

1.1.2.1.1 Glass Eel / Elver Fishery

Glass eel and elver fisheries along the Atlantic Coast are prohibited in all states except Maine and South Carolina. In recent years, Maine is the only state reporting significant glass eel and elver harvest. Maine implemented regulatory changes that increased elver and large eel license fees in 1996. In addition to generating revenue for enforcement and eel research, these changes set both a harvest season and closures during the harvest season. The number, type, and methods of operation of gear units available to each fisher were limited to control fishing effort, as were the allowable fishing areas, and fishing within 46 m of a dam was prohibited (CAEMM 1996). South Carolina could not determine participation in the elver and glass eel fishery in coastal waters until a limited entry permit system was instituted in 1996 (B. McCord, South Carolina Department of Natural Resources, pers. comm.). Ten permits are available to both instate and out-of-state residents. Permit holders abide by monthly effort controls and must report their harvest. There was interest in developing commercial glass eel fisheries in Connecticut, New

Jersey, Virginia, and Florida. Connecticut regulations were minimal until 1996 when the state defined the glass eel as less than 10 cm in length, instituted a glass eel fishing season with a weekly closed period, limited traps, and required monthly catch reporting by logbook. Connecticut prohibited the take or attempted take of glass eels, elvers, and silver eels in 2002. The glass eel and elver fishery in New Jersey was unregulated prior to 1997 when it was restricted to dip nets only and a fishery season was implemented with a Sunday closure. The glass eel and elver fishery was closed in 1998. In Virginia, a six-inch minimum size was passed in 1977. Florida passed regulations in 1998 such that the eel fisheries operate under gear restrictions that do not allow the landings of eels under six inches.

Prior to the implementation of the FMP, Maine was the only state compiling glass eel and elver fishery catch statistics. Under the FMP, all states are now required to submit fishery-dependent information. Poaching of glass eels and elvers is believed to be a serious problem in many states, but enforcement of the regulations is poor due to the nature of the fishery (very mobile, nighttime operation) and low administrative priority.

1.1.2.1.2 Yellow / Silver Eel

The economically important yellow/silver American eel fishery in Maine occurs in both inland and tidal waters. Large eel fisheries in southern Maine are primarily coastal pot fisheries managed under a license requirement, minimum size limit, or gear and mesh size restrictions. New Hampshire has monitored its yellow eel fishery since 1980; effort reporting in the form of trap haul set-over days for pots or hours for other gears has been mandatory since 1990. Small-scale, commercial eel fisheries occur in Massachusetts and Rhode Island and are mainly conducted in coastal rivers and embayments with pots during May through November. Connecticut has a similar small-scale, seasonal pot fishery for yellow eels in the tidal portions of the Connecticut and Housatonic rivers (S. Gephard, Connecticut Department of Energy and Environmental Protection, pers. comm.). All New England states presently require commercial eel fishing licenses and maintain trip level reporting.

Licensed eel fishing in New York occurred primarily in Lake Ontario (prior to the 1982 closure), the Hudson River, the upper Delaware River (Blake 1982), and in the coastal marine district. A slot limit (greater than 6 inches and less than 14 inches to limit PCB concentration) exists for eels fished in the tidal Hudson River (from the Battery to Troy and all tributaries upstream to the first barrier), Lake Ontario, and St. Lawrence strictly for use as bait or for sale as bait only. Due to PCB contamination of the main stem, commercial fisheries have been closed on the freshwater portions of the Hudson River and its tributaries since 1976. In 1995, New York approved a size limit in marine waters. New Jersey fishery regulations require a commercial license, a minimum mesh, and a minimum size limit. A minimum size limit was set in Delaware in 1995. Delaware mandated catch reporting in 1999 and more detailed effort reporting in 2007.

Maryland and Virginia have primarily pot fisheries for American eels in Chesapeake Bay. Large eels are exported whereas small eels are used for bait in the crab trotline fishery. Catch reports were not required in Virginia prior to 1973 and Maryland did not require licenses until 1981. Effort reporting was not required in Maryland until 1990.

North Carolina has a small, primarily coastal pot fishery. A trip ticket system began in 1994 and a commercial logbook system began in 2007. The majority of landings come from the Albemarle Sound area and additional landings reported from the Pamlico Sound and “other areas”. No catch records are maintained for freshwater inland waters, although landings for inland areas may be included under “other areas” reported by the state if brokered by a NCDMF-licensed dealer. South Carolina instituted a permitting system over ten years ago to document total eel gear and commercial harvest. Traps, pots, fyke nets, and dip nets are permitted in coastal waters. Fishing for eels in coastal waters is often conducted under the guise of fishing for crabs.

American eel fishing in Georgia was restricted to coastal waters prior to 1980 when inland fishing was permitted (Helfman 1984). Catch, but not effort, data are available because no specific license is required to fish eels. The Florida pot fishery has a minimum mesh size requirement in the fishery and it is operated under a permit system.

1.1.2.2 Recreational Fishery

Few recreational anglers directly target American eels and most landings are incidental when anglers are fishing for other species. Eels are often purchased by recreational fishermen for use as bait for larger sport fish such as striped bass, and some recreational fishermen may catch their own eels to utilize as bait. Current recreational management regulations can be found in Table 1.2. South Carolina is currently in the process of changing their recreational regulations to include a six-inch minimum size and a fifty-fish creel limit.

1.2 Stock Assessment History

In 2005, a stock assessment for American eel was conducted by the ASMFC and reviewed by a panel of independent experts (ASMFC 2005). The peer review panel recognized sufficient shortcomings with the assessment to warrant additional action prior to its use for future technical and management purposes (ASMFC 2006a). The 2005 stock assessment was not accepted by the Board; therefore, the stock status of American eel is still deemed unknown by the ASMFC.

At the February 22, 2006 meeting of the ASMFC American Eel Management Board, the American Eel Stock Assessment Subcommittee and Technical Committee were tasked with reviewing the recommendations from the peer review advisory report and recommending a follow-up plan. Subsequently, a report was issued in October of 2006 containing updated datasets and the short-term analyses suggested by the review panel (ASMFC 2006b). This stock assessment represents the most recent work performed by the ASMFC to ascertain stock status since 2006.

1.3 Petitions for ESA Listing

In response to the extreme declines in American eel abundance in the Saint Lawrence River-Lake Ontario portion of the species’ range, the ASMFC requested that the U.S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS) conducted a status review of American eels in 2004. The ASMFC also requested an evaluation of a Distinct Population Segment (DPS) listing under the Endangered Species Act (ESA) for the Saint Lawrence River/Lake Ontario and Lake Champlain/Richelieu River portion of the species range, as well as an evaluation of the entire Atlantic coast American eel population. A preliminary status review conducted by USFWS determined that American eel was not likely to meet the requirements of DPS determinations. However, the USFWS initiated a coast-wide status review of the American eel in coordination with the NMFS and ASMFC. At this same time, two private citizens submitted a petition to the USFWS and NMFS to list American eel under the ESA.

In February 2007, the USFWS announced the completion of a Status Review for American eel (50 CFR Part 17; USFWS 2007). The report concluded that protecting eels as an endangered or threatened species was not warranted. The USFWS did note that while the species' overall population was not in danger of extinction or likely to become so in the foreseeable future, the eel population has "been extirpated from some portions of its historical freshwater habitat over the last 100 years…[and the species abundance has declined] likely as a result of harvest or turbine mortality, or a combination of factors".

In 2010, the Center for Endangered Species Act Reliability filed a petition to the USFWS to consider listing the American eel on the endangered species list. The proposal is based on new information that has become available since the last status review. In September 2011, the USFWS published a positive 90-Day Finding, which stated that the petition contained enough information to warrant conducting a status review (USFWS 2011). The proposed rule is expected to be published in 2012 after USFWS completes the status review.

2 LIFE HISTORY

American eels are found from the southern tip of Greenland, Labrador and the northern Gulf of St. Lawrence in the north, south along the Atlantic and Gulf coasts of North America and eastern Central America to the northeast coast of South America, and into the inland areas of the Mississippi and Great Lakes drainages (Tesch 1977). The American eel is regarded as a single, panmictic breeding population. American eels are found in a variety of habitats throughout their life cycle, including the open ocean, large coastal tributaries, small freshwater streams, and lakes and ponds. They are opportunistic feeders that will eat, depending on their life stage, phytoplankton, zooplankton, insects, crustaceans, and fish. Individuals grow in freshwater or estuarine environments for anywhere from 3 to 30 or more years before maturing and returning to the ocean as adults to spawn once and die.

American eels are confronted with many environmental and human-induced stressors which affect all life stages and may reduce survival. Since all eel mortality is pre-spawning, reproduction can be reduced by these cumulative pressures. Commercial harvest occurs at all American eel life stages (glass, elver, yellow, and silver). Blockages and obstructions that limit upstream migration of American eels have reduced habitat availability and limited the range of the species. Dams may also limit or delay downstream movements of spawning adults. Additionally, downstream mortality may be caused by hydroelectric facilities by impingement or turbine passage. Freshwater habitat degradation resulting in reduced food productivity increases mortality of the freshwater life stages. Predation by fish, birds, and mammals can impact eel populations during all life stages. The non-native swim bladder parasite, *Anguillicoloides crassus*, can decrease swimming ability and reduce the silver eel's ability to reach the spawning grounds. Contaminants also may reduce the reproductive success of American eels because they have a high contaminant bioaccumulation rate (Couillard et al. 1997). Oceanographic changes influencing larval drift and migration may reduce year-class success. American eel, as a panmictic species, could be particularly vulnerable to drastic oceanic variations. An understanding of the requirements of the American eel's different life stages is needed to protect and manage this species.

2.1 Stock Definitions

The American eel is a panmictic species, with a single spawning stock that reproduces in the Sargasso Sea. Eel larvae (leptocephali) are randomly dispersed by ocean currents along the Atlantic coasts of northern South, Central, and North America. Genetic research indicates that there is no reproductive isolation of American eels migrating from the Atlantic Coast. Further, any genetic differentiation is a result of natural selection upon a particular cohort within a geographic area rather than actual genetic differences within the species (Avise et al. 1986; Wirth and Bernatchez 2003; Cote et al. 2009).

2.2 Migration Patterns

American eels may travel thousands of miles in their lifetime. They are a catadromous fish that spawn in the Sargasso Sea, and the larvae drift on ocean currents until they reach the eastern seaboard of North America. Young eels actively swim upstream to reach estuarine and freshwater habitats, sometimes hundreds of miles upriver. The young eels spend between 3 and 30 or more years in estuarine or freshwater habitats before maturing and migrating back downstream and to the Sargasso Sea to spawn.

Spawning in the Sargasso Sea occurs over a large area from about 19.5°N to about 29°N and 52°W to 79°W (McCleave et al. 1987). Although spawning or mature American eels have never been observed at sea, spawning is thought to occur in the frontal zone and to the south within this region (Kleckner et al. 1983; McCleave et al. 1987; Munk et al. 2010). Based on collections of leptocephali, spawning is assumed to occur from mid-February through April (McCleave et al. 1987).

Once the eggs hatch, the leptocephali use passive transport in the upper 350 m of the currents to begin their migration to the coasts of the western Atlantic (Kleckner and McCleave 1982, 1985; Munk et al. 2010). Most American eel leptocephali are transported west by the Florida Current from the Sargasso Sea and then north on the Gulf Stream Current (Kleckner and McCleave 1982; McCleave 1993) to reach the coast of North America. Leptocephali spend up to 15 months in the ocean before they reach the Atlantic Coast of the U.S. (Kleckner and McCleave 1985). Because of ocean currents, leptocephali are deposited to the Continental Shelf of North America at higher densities from Cape Hatteras north to Quebec (Kleckner and McCleave 1985).

American eels reach the eastern coast of North America in the glass eel stage and begin their upstream migrations. Glass eels actively swim from the Gulf Stream, and it takes 60 to 110 days to reach the coasts of New Jersey and North Carolina, respectively (Powles and Wharlen 2002; Wuenschel and Able 2008). Timing of inshore migration occurs later in the year with increasing latitude. In the southeast U.S., glass eel migrations occur during the late winter and in the Canadian provinces, migration occurs as late as August (Table 2.1). Glass eels and elvers use selective tidal stream transport for migrating upriver (Sheldon and McCleave 1985; McCleave and Wippelhauser 1987). In the St. Lawrence Estuary, eels are able to travel upstream at the rate of 10 to 15 km/day (Dutil et al. 2009), but the speed is reduced to an average of 1 to 2 km/day further up the St. Lawrence River (Verndon and Desrochers 2003). Migration typically occurs at night and is related to reaching a minimum threshold temperature in rivers (usually 10 to 12 degrees Celsius), and the occurrence of a full or new moon and freshets (Haro and Krueger 1988; Martin 1995; Sorensen and Bianchini 1986; Jessop 2003; Schmidt et al. 2009; Sullivan et al. 2009).

Upstream migration typically occurs in the glass eel and elver stage, but yellow American eels sometimes continue upstream migrations (Jessop et al. 2008). Eels settle in a diversity of habitats, ranging from estuaries to freshwater habitats hundreds of miles from the ocean. When upstream migration is complete, eels are usually in the yellow phase and typically set up relatively small home ranges with some exhibiting local seasonal migrations (Oliveira 1997; Jessop et al. 2008; Hammond and Welsh 2009).

Yellow-phase American eels spend 3 to 30 or more years inland before becoming mature, entering the silver phase. Once silver, eels migrate downstream toward the Sargasso Sea. The timing of silver eel downstream migration occurs on a latitudinal cline, with eels leaving the Canadian Provinces in summer through fall and from winter through early spring in the southern U.S. (Table 2.2). During downstream migration, silver eels typically move at night during the darker moon phases, high water flows, and decreasing water temperatures (Hain 1975; Winn et al. 1975; Euston et al. 1998; Haro et al. 2003; Barber 2004; Brown et al. 2009; Welsh et al. 2009). Downstream migrants use tidal transport and travel near the surface but do make vertical migrations (Parker and McCleave 1997). Ocean migrations of silver eels to the Sargasso Sea are thought to take place in the upper few hundred meters of the water column where differences in water masses are most distinct (McCleave et al. 1987).

2.3 Life Cycle

American eels undergo six distinct life stages. The life cycle begins when the eggs hatch and leptocephali (larvae) are carried by ocean currents from the spawning grounds in the Sargasso Sea. The prevailing currents along coastal areas disperse the leptocephali, which metamorphose into glass eels on the continental shelf. Glass eels move toward inland areas and become pigmented elvers before or during their entry into coastal estuaries. Elvers and yellow eels settle in habitats ranging from estuaries to far upstream freshwater reaches. Eels reach the silver stage at maturity and return to the Sargasso Sea, then spawn once and die.

2.4 Life Stages

2.4.1 Egg

American eels spawn in the winter and early spring in the Sargasso Sea, which is a large portion of the western Atlantic Ocean east of the Bahamas and south of Bermuda. Although no eggs have ever been collected in the Sargasso Sea, it is likely they hatch in the vicinity of the spawning area. Hatching probably occurs within a week of spawning, based on egg incubation times for the Japanese eel, *Anguilla japonica* (Kagawa et al. 2005). Spawning is thought to occur between the months of February and April (McCleave et al. 1987; McCleave 2008), based on collections of leptocephali. There is no information available on the required environmental conditions for the eggs.

2.4.2 Leptocephali

After hatching and a brief pre-larval stage, American eels enter a larval or leptocephali stage. The leptocephali are shaped like a willow leaf, laterally compressed, and transparent. Sampled leptocephali have been less than 5 mm total length and up to 70 mm total length and remain in the ocean for 8 to 15 months (Kleckner and McCleave 1985). They are passively transported within ocean currents, and the spatial and temporal distribution of larvae is a result of oceanic circulation patterns. Leptocephali are positively buoyant allowing them to stay in surface waters where food is more abundant (Tsukamoto et al. 2009). They undergo vertical migrations while in the ocean, being concentrated in the upper 140 m at night and upper 350 m during the day (Kleckner and McCleave 1982; McCleave et al. 1987). Leptocephali grow rapidly from February to October and then growth slows or stops after October. Total lengths of leptocephali increase in the Gulf Stream Current moving north from Florida to North Carolina along the Atlantic Coast (Kleckner and McCleave 1982). At sea, probably at the edge of the continental shelf, the leptocephali undergo a metamorphosis into the glass eel stage.

2.4.3 Glass Eel

The glass eel life stage of American eels begins when the leptocephali metamorphose at sea, on or near the continental shelf (Kleckner and McCleave 1985). Metamorphosis from leptocephali occurs from 6 months (Wang and Tzeng 1998) to 12 months post-hatch (Kleckner and McCleave 1985), usually between the months of October and March. Estimates from otolith ageing indicate metamorphosis from leptocephali to glass eel occurs between 132 and 214 days post-hatch, with duration of metamorphosis ranging from 18 to 80 days (Wang and Tzeng 1998; Arai et al. 2000). Glass eels reach the eastern shores of North America 30 to 80 days after metamorphosis (at age 220 to 284 days; Wang and Tzeng 1998).

The determination of spawning and metamorphosis dates from glass-stage American eel otoliths is somewhat problematic. When estimating hatching dates from back-calculation of otoliths, the spawning season appears to be early August to early October, not corresponding with estimated spawning periods (February to April) based on ocean collection of leptocephali (Kleckner and McCleave 1982). This discrepancy, possibly due to some resorption of the otolith during metamorphosis, indicates that using otolith ageing to back calculate hatching dates of eels may not be accurate (McCleave 2008).

When American eel leptocephali transform into glass eels, they experience a decrease in body length and weight due to loss in water concentration and increase in body thickness (Fahay 1978). Glass eels are transparent with elongated, cylindrical bodies and usually range in length from 48 to 65 mm (Hardy 1978; Kleckner and McCleave 1985). They actively migrate toward land and enter rivers between late winter and summer, with timing related to latitudinal distribution (Table 2.1). Glass eel migration occurs earlier in the southern portion of the range and later in the northern portion. Glass eels are also smaller in southern areas (mean lengths 47.8 mm to 49.0 mm) than in northern areas (mean lengths 58.5 mm to 60.0 mm; Wang and Tzeng 1998, 2000).

Glass-stage American eels arrive into estuaries at 220 to 284 days old, with the youngest glass eels arriving in estuaries in the middle of their range and older glass eels arriving in estuaries at the northern and southern ends of their range (Wang and Tzeng 1998). Glass eels ascend estuaries by drifting on flood tides and holding their position near the bottom on ebb tides, but they also swim upstream along the shore in both tidal and non-tidal waters (Barbin and Krueger 1994). Upstream migration with the glass eel is likely influenced by the detection of the odor of freshwater (Facey and Van Den Avyle 1987; Sullivan et al. 2006).

2.4.4 Elver

The elver life stage of American eels occurs when the glass eels ascend into brackish or freshwater and become pigmented. Elvers are brown in color and are usually fully pigmented at

65 mm to 90 mm in length (Hardy 1978), although pigmented American eel less than 65 mm have been observed in Florida (J. Crumpton, Florida Game and Fresh Water Fish Commission, pers. comm.). Pigmentation is not well correlated with elver size (Haro and Krueger 1988; Wang and Tzeng 2000). Elvers are generally larger in northern locations (Haro and Krueger 1988), and this may be due to additional growth during the extended period in the glass eel phase (62 to 80 days) in the northern part of the range compared to the southern part of the range (32 to 34 days; Wang and Tzeng 1998). Higher condition elvers arrive earlier and colonize upstream habitats, and lower condition elvers arrive later in the season and stay in estuaries (Jessop 1998; Sullivan et al. 2009).

Elvers are active at night and burrow during the day. They move into the water column on flood tides and return to the bottom during ebb tides (McCleave and Kleckner 1982). They swim upstream, drawn by changes in water chemistry and river current velocities (Facey and Van Den Avyle 1987). Upstream migration of glass eels and elvers can occur over a broad period of time from May (during peak migration) through October (Richkus and Whalen 1999). The migration occurs earlier in the southern portion of its range and later in the northern portion (Table 2.1; Helfman et al. 1984a; McCleave and Kleckner 1982).

2.4.5 Yellow Eel

The yellow eel phase is the last developmental stage of the American eel prior to reaching maturity. By the age of two years, most eels are in the yellow phase. They resemble elvers in body shape and typically have skin coloration with various hues of yellow, brown, and green. They inhabit bays, estuaries, rivers, streams, lakes, and ponds. Depending on where they cease their upstream migration, some yellow eels reach the extreme upper portions of the rivers while others stay behind in the brackish areas. Catadromy is not a requirement for completing the life cycle of the eel as many eels live their entire yellow phase in estuarine or oceanic water (Tsukamoto et al. 1998; Morrison et al. 2003; Lamson et al. 2006). The timing and duration of yellow eel upstream migration is watershed specific and can occur over a broad period of time. Most eels migrate upstream during their first years of life and then establish a home range where they live and grow until maturity. However, a portion of yellow eels continue migrating upstream until they reach sexual maturity (Richkus and Whalen 1999), and other yellow eels migrate repeatedly between fresh and brackish water throughout the yellow stage (Morrison et al. 2003; Jessop et al. 2006; Thibault et al. 2007). Yellow eels typically establish relatively small home ranges, indicated by recaptures frequently occurring within 1 km of the original capture location (Gunning and Shoop 1962; Bozeman et al. 1985; Ford and Mercer 1986; Dutil et al. 1988; Morrison and Secor 2003; Thibault et al. 2007; Cairns 2009). Yellow eels will also return to their original capture location after being displaced (Parker 1995; Lamothe et al. 2000).

American eels become sexually differentiated in the yellow phase by the time they reach 270 mm (Oliveira and McCleave 2000). In the northern portion of their range, female eels mature at greater ages and sizes than in the southern portion (Table 2.2; Helfman et al. 1987). Female eel size and age also increases with increased distance from the ocean within river systems (Table 2.3; Smogor et al. 1995; Goodwin and Angermeier 2003; Morrison and Secor 2003; Owens and Gear 2003). Male eels do not exhibit latitudinal differences in size, with most males mature at less than 400 mm. However, male eels from the northern part of the range take longer to mature than in the southern part of the range (Jessop 2010).

2.4.6 Silver Eel

The silver stage of American eels occurs when yellow eels undergo several physiological changes as they become sexually mature, including: (1) changing color from yellow/green to metallic, bronze/black, (2) fattening of the body, (3) thickening of the skin, (4) enlargement of the eye and change in visual pigment, (5) increased length of the capillaries in the rete of the swim bladder, (6) change in gill structure for osmoregulation in sea water, (7) digestive tract degeneration, (8) enlarging of the pectoral fins, and (9) high percentage of late stage oocyte development (reviewed by Dutil et al. 1987; Facey and Van Den Avyle 1987; McGrath et al. 2003a). Yellow eels begin the transformation into silver eels in their freshwater and estuarine habitats and finish the transition between estuaries and the open ocean (Wenner 1973; Facey and Van Den Avyle 1987).

Size at maturation is different between male and female American eels. Females, on average, are 1.9 times larger than males at maturity (Jessop 2010). Silver male eels are the same size regardless of where they are collected within their geographic range (Jessop 2010). Average male sizes are typically between 300 and 350 mm (Wenner and Musick 1974; Winn et al. 1975; Foster and Brody 1982; Facey and Helfman 1985; Oliveira 1999; Oliveira and McCleave 2000; Goodwin and Angermeier 2003; Barber 2004; Jessop et al. 2009). Maximum male size has been reported as 503 mm (Dolan and Power 1977), but generally mature males are less than 400 mm.

The size of female American eels increases with distance from coastal waters (Ingraham 1999; Goodwin and Angermeier 2003; Morrison and Secor 2003; Jessop 2010). Females may reach maturity at 350 mm in estuarine areas and usually do not exceed 1,200 mm in inland areas. Large female silver eels (greater than 900 mm) are common in the St. Lawrence River (Fournier and Caron 2001; Verreault 2002; McGrath et al. 2003a; Tremblay 2009), but eels exceeding that size are also likely to be found in inland areas of the U.S. as indicated by collections in the Shenandoah River, Virginia (Euston et al. 1998; Goodwin and Angermeier 2003).

Timing of downstream migration for silver-phase American eels varies with latitude (Table 2.2). Silver eels begin their seaward spawning migration from Canadian and New England tributaries during late summer through fall (Dutil et al. 1987; Ingraham 1999; Haro et al. 2003; McGrath et al. 2003a; Brown et al. 2009). Silver eel emigration from a small river in southern Delaware peaked in September, usually in the days following a heavy rainfall (Barber 2004). In the southeastern U.S., silver eel migrations typically occur in the winter or early spring (Harrell and Loyacano 1982; Helfman et al. 1984a; Facey and Helfman 1985). Silver eel emigration at a particular location is likely based on both sex-specific length (rather than age) and distance from coastal waters (Helfman et al. 1987; McGrath et al. 2003a; Morrison and Secor 2003; Tremblay 2009).

American eels migrate long distances to the spawning grounds in the Sargasso Sea. Lake Ontario silver eels travel more than 4,500 km to spawn. One migrating silver eel swam 150 km in two days (Welsh et al. 2009), showing considerable vertical movements in the water column but no behavioral changes associated with diel or tidal cycles (Stasko and Rommel 1977). Little is known about the oceanic spawning migration or the means by which the spawning grounds are located by the eels (Miles 1968). American eels may use the geo-electrical fields generated by ocean currents for orientation (Rommel and Stasko 1973). The depth at which American eels migrate in the ocean has been hypothesized to vary with light intensity and turbidity (Edel 1976). Migration has been suggested to occur within the upper few hundred meters of the water column (Kleckner et al. 1983; McCleave and Kleckner 1985). However, Robins et al. (1979)

photographed two Anguilla eels, believed to be pre-spawn American eels, at depths of about 2,000 m (on the floor of the Atlantic Ocean) in the Bahamas. No information exists on the spawning requirements, behavior, or the exact location of spawning within the Sargasso Sea. Adult eels are believed to spawn in the winter and early spring and perish after spawning.

The age of American eels tends to increase moving upstream in tributaries away from the ocean. In the Gulf of St. Lawrence, freshwater eels took 2.4 times as long to reach maturity than their brackish water counterparts (Lamson et al. 2009). In the Hudson River, brackish water female silver eels were 5 to 8 years old, while female silver eels from upstream were 17 to 20 years old (Morrison and Secor 2003). In the lower Potomac River, mature female eels ranged from 5 to 11 years old, but the upstream tributary had females ranging from 10 to 19 years (Goodwin and Angermeier 2003).

Male American eels are typically younger than female eels at maturity (Table 2.2). In Georgia, mean age of silver eels was 5.5 years for males and 8.6 years for females (Facey and Helfman 1985). In the Indian River, Delaware mean silver male age was 7.4 years and 12 years for females (Barber 2004). In Rhode Island, mean silver male age was 10.9 years compared to 12.8 years for silver females (Oliveira 1999). In Nova Scotia, silver males averaged 12.7 years and silver females averaged 19.3 years (Jessop 1987).

2.5 Life History Characteristics

2.5.1 Age

The age of American eels can be determined by taking transverse sections of the sagittal otoliths. Two otolith processing techniques (embedding and sectioning or grinding and polishing) are accepted ageing methods by the ASMFC (ASMFC 2001). American eel otolith ageing methods have been described by Liew (1974), Chisnall and Kalish (1993), and Oliveira (1996).

Several studies have attempted to use daily growth rings to estimate American eel age in the first years of life (Arai et al. 2000; Wang and Tzeng 2000). This method does not accurately estimate age (Tesch 1998) because back-calculation does not reflect the assumed spawning season (McCleave 2008). Using daily growth rings to estimate age is problematic because some of the otolith is lost or resorbed during metamorphosis from leptocephali to glass eel (Cieri and McCleave 2000).

American eels are roughly age one when they reach continental waters, they are typically in the elver stage during age two, and then they become yellow eels by age three. American eels remain in the yellow phase for a variable length of time related to size, sex, and geographic location (Jessop 2010; see also section 2.4.5), until they reach sexual maturity.

Maximum ages tend to be younger in the southern portion of the American eel's distribution and older in the northern areas (Jessop 2010). In the Altamaha River, Georgia, female silver eels were 3 to 6 years (Helfman et al. 1984b). Barber (2004) observed silver female eels ranging from 7 to 20 years in an Atlantic Coast tributary in Delaware. In Nova Scotia, mature female eels ranged from 8 to 43 years (Jessop 1987), and they average 20 years in the St. Lawrence River (Tremblay 2009).

Maturation from the yellow to silver phase in American eels occurs as early as age 2 and as late as age 30 or older (Michener and Eversole 1983; Jessop 1987). Timing of sexual maturity in the yellow eel has been correlated with sex and specific size ranges and varies considerably along their geographic range (Jessop 2010). Maturity typically occurs at younger ages in the southern portion of the range, and age at maturation increases with increasing latitude (Table 2.2). Female eels reach maturity on average between 5 and 8 years in South Carolina and Georgia (Michener and Eversole 1983; Facey and Helfman 1985), while the average age in the St. Lawrence River is around 20 years (Verreault 2002; Casselman 2003; Tremblay 2009). Males reach maturity typically in 5 years or less in areas from the Chesapeake Bay and south (Foster and Brody 1982; Harrell and Loyacano 1982; Hansen and Eversole 1984; Facey and Helfman 1985), and in coastal areas of Maine and Canada, mean age at maturity for males is about 12 years (Jessop 1987; Oliveira and McCleave 2000).

2.5.2 Growth

During the first year post-hatch, American eels drift on the ocean currents as leptocephali and have similar growth rates throughout their distribution. Estimates of growth rates for the first year of life in the ocean range from 0.187 mm/day (Tesch 1998) to 0.45 mm/day (Arai et al. 2000). Total length decreases during the metamorphosis from leptocephali to glass eel.

Glass-stage American eels have a higher growth rate in the southern portion of their range compared to the north (Wang and Tzeng 1998). In a study comparing glass eels collected in North Carolina and New Brunswick, growth rates were similar for the first 10 to 15 daily growth rings, but later growth was faster in North Carolina than New Brunswick (Powles and Wharlen 2002). Glass eels and elvers also grow faster in brackish water compared to freshwater (Cote et al. 2009).

Glass-stage American eels decrease in total length during transformation to elver stage. The size of elvers at transformation from the glass eel stage increases with distance from spawning ground (Haro and Krueger 1988; Jessop 2010). Elver growth rates are higher than rates for yellow eels, averaging 57 mm/year during their first two years (one year oceanic and one year freshwater) and reaching about 127 mm after the first year in freshwater (Bigelow and Schroeder 1953; Machut et al. 2007).

Once American eels reach the yellow stage, growth is highly variable and is based on sex, age, latitude, salinity, and season (Tables 2.4 and 2.5). Female eels have higher growth rates than males (Helfman et al. 1984a; Fenske et al. 2010; Jessop 2010). In Maine, female eels grew faster than males and rates were noticeably different based on sex at year 4 (Oliveira and McCleave 2002). In Rhode Island, eels larger than 400 mm (females) had a growth rate of 62 mm/yr, compared to the pooled growth rate of 30 mm/year for smaller eels (Oliveira 1997). In the Chesapeake Bay, female eels had a mean growth rate of 71.4 mm/year compared to a growth rate of 64.2 mm/year for males (Fenske et al. 2010). In Charleston Harbor, male eels were smaller than female eels in each age class (Michener and Eversole 1983).

Because brackish waters are generally more productive than freshwater areas, American eels in estuarine or brackish water grow faster than their freshwater counterparts (Helfman et al. 1984a; Cairns et al. 2004; Jessop et al. 2008; Cairns 2009; Jessop et al. 2009; Lamson et al. 2009; Fenske et al. 2010). In the Hudson River, eels grow two to three times faster in brackish water than in fresh or salt water (Morrison and Secor 2003). Estuarine eels are more likely to have food in their stomachs than their freshwater counterparts, which may result in the lower growth rates of eels from freshwater habitats (Thibault et al. 2007). Although freshwater eels have lower growth rates, they are generally longer as you progress farther inland because of increased residency times (Goodwin and Angermeier 2003). Dams can impact eel growth when progressing inland. Eels above dams grow faster than eels at the base of dams, suggesting growth may be density dependent (Strickland 2002; Machut et al. 2007).

Slower growth occurs in more northern portions of the American eel's distribution compared to the south (Helfman et al. 1984; Richkus and Whalen 1999; Jessop 2010). However, female eels reach a larger maximum size in the northern portion of their range compared to the south (Jessop 2010). Male maximum size is the same throughout their distribution (Jessop 2010). Eel growth is related to seasons, with most growth occurring during spring through fall and very little growth in the winter (Helfman et al. 1984). The shorter growing seasons in the higher latitudes may explain why eels experience slower growth in the northern portions of their range.

Growth rates are highly variable among fish within the same watershed and of the same sex thus total length is not an accurate predictor of age. In the Hudson River, 50-cm long American eels ranged in age from 5 to 29 years (Morrison and Secor 2003). Growth rates decline with age (Jessop et al. 2009) from an average rate of 57 mm/year in the first year of freshwater residence to 25 mm/year for age-20+ eels (Machut et al. 2007). Reaching a predetermined size within a location, regardless of age, may be the most influential factor in inducing sexual maturity (Jessop et al. 2004; Jessop et al. 2009).

Published literature refers to growth rates for American eels derived from measured growth in the field or back-calculated lengths from otolith analysis. Growth rates derived from the same fish using both methods can be very different. Typically, growth measured directly is higher than that derived from otolith back calculation for the same geographic location (Table 2.4; Helfman et al. 1984; Morrison and Secor 2003).

Published length-at-age (Table 2.6) and length-weight (Table 2.7) relationships vary by geographic location.

2.5.3 Reproduction

The sex of American eels can be determined by gross morphological examination (Vladykov 1967; Krueger and Oliveira 1997). Ovaries are frilled ribbon-like organs, and testes are deeply lobed, with lobes broadly overlapping adjacent lobes (Dolan and Power 1977). Chisnall and Kalish (1993) suggest that morphological examination may not be reliable and recommend using an aceto-carmine "squash" method to prepare gonads (Guerrero and Shelton 1974; Columbo et al. 1984; Chisnall and Kalish 1993; Beullens et al. 1997). However, Dolan and Power (1977) argue that gross morphological examination is sufficient because very rarely does a yellow female's gonads slightly resemble testes.

Differentiation between sexes occurs in the yellow eel stage of American eels. Sex can be identified in most eels at a minimum size between 250 mm and 350 mm (Dolan and Power 1977; Oliveira and McCleave 2000). Mature males are generally less than 400 mm (Krueger and Oliveira 1997; Oliveira and McCleave 2000; Morrison and Secor 2003; Weeder and Hammond 2009). Mature females are typically larger than 400 mm and can reach sizes of over 1,200 mm in more northern and inland portions of their range (Goodwin and Angermeier 2003; Tremblay 2009).

Sex ratios are highly variable among locations (Table 2.8), and there are several hypotheses about sex determination in the American eel. The exact role of genetics and the environment in determining sex in American eels is not known. There is strong evidence for phenotypic or environmental sex determination (Degani and Kushinov 1992; Roncarati et al. 1997). High rearing densities common in aquaculture often produce a preponderance of males (Egusa 1979, cited by Oliveira and McCleave 2002). In a lab experiment with European eel, sex was determined by a combination of hormones and grouping (increased eel density versus solitude; Degani and Kushnirov 1992). Density-based effects or habitat type may determine sex, with males found more commonly in downriver sites and females more common in upriver sites (Facey and Helfman 1985; Helfman et al. 1987; Krueger and Oliveira 1999; Oliveira and McCleave 2000; Goodwin and Angermeier 2003; Davey and Jellyman 2005). In Maine, silver eels ranged from 49% to 98% male, and the proportion of males was inversely related to lacustrine (lake) habitat in the drainage (Oliveira et al. 2001).

Sex-linked migration patterns are another possible explanation for why male American eels are typically found in coastal habitats while females tend to be found in more upstream areas (Jessop 2010). Females are found in habitats that are less densely populated with eels so sex may not be a function of density dependence but rather that female eels migrate further upstream than males (Jessop 2010).

Reported estimates of fecundity for American eels range from 0.4 to 22.0 million eggs per female (Table 2.9; Wenner and Musick 1974; Barbin and McCleave 1997; Tremblay 2009). Fecundity estimates are higher in the northern portion of the eel’s range because of the larger sizes of migrating female eels from northern areas (Barbin and McCleave 1997).

American eels are thought to spawn in the Sargasso Sea during late winter through spring, but spawning has never been observed. It is also unknown if they have paired or group spawning. Because no spent eel has ever been documented, it is assumed that American eels are semelparous.

2.5.4 Food Habits

American eel diet varies greatly depending on life stage and habitat. American eel leptocephali and glass eel feeding habits have not been reported. However, the dentition and gape of the mouth suggest that they are capable of feeding on individual zooplankton and phytoplankton. Prey size increases as eels grow, with elvers and small yellow eels consuming mostly benthic macroinvertebrates and larger yellow eels switching primarily to crayfish and fish. Silver eels are thought not to eat during their migration to the Sargasso Sea.

Bigelow and Schroeder (1953) describe the American eel as feeding on whatever prey/food items happen to be found in its habitat. However, eels are selective in that prey ratios in stomach contents are different than in surrounding habitats (Machut 2006). Given their poor eyesight and nocturnal feeding habits (Sorensen et al. 1986), yellow eels probably rely on their keen sense of smell to locate food (Fahay 1978). Yellow eels swallow some types of prey whole but also can tear pieces from large dead fish, crabs, and other items (Facey and Van Den Avyle 1987) by biting and spinning rapidly (Helfman and Clark 1986).

American eels in the elver and yellow stages are carnivores and consume a variety of foods including demersal fishes and benthic invertebrates. The diet of yellow eels is related to the size of the fish, usually with smaller eels eating small soft-bodied prey (Machut 2006). Eels shorter than 300 to 400 mm in inland areas of Maine, New York, New Jersey, Delaware, and South Carolina mainly ate benthic aquatic insect larvae, including chironomids, mayflies, stone flies, dragonflies, megalopterans, and caddisflies. Larger eels fed primarily on crustaceans (crayfish) and smaller benthic fish (Odgen 1970; Scott and Crossman 1973; Facey and LaBar 1981; Smith 1985; Lookabaugh and Angermeier 1992; Denoncourt and Stauffer 1993; Daniels 1999; Machut 2006). Large yellow eels are also known to be cannibalistic, eating elvers when available (Jessop 2000).

In estuarine waters, American eels primarily fed on polychaetes, crustaceans, and bivalves. Fish were not an important component of the diet, even in larger eels. Seasonally, fish did occur in the diet of intermediate-sized yellow eels during the winter and spring, while insects and mollusks were eaten from spring through fall. Yellow eels in the lower Chesapeake Bay fed on crustaceans including blue crab (*Callinectes sapidus*) and bivalves such as soft-shelled clams (*Mya arenaria*; Wenner and Musick 1975).

2.5.5 Natural Mortality

Very little is known about the natural mortality of American eels. Since eels are highly fecund (Wenner and Musick 1974; Barbin and McCleave 1997; Tremblay 2009), natural mortality is likely very high, particularly during the early life stages. Eel survival is likely impacted by changes in oceanographic conditions, predation, and the spread of the non-native swim bladder nematode.

American eel early life stages are likely highly impacted by changes in oceanographic conditions that affect both survival and transportation to the coast of North America (McCleave 1993; Castonguay et al. 1994b; Friedland et al. 2007; Miller et al. 2009). Global warming may change primary production of open ocean areas and alter food availability for leptocephali, which may contribute to the cause of population declines as seen in American, European (*A. anguilla*), and Japanese eel (Bonhommeau et al. 2008; Miller et al. 2009). Longer migration times, due to changes in ocean currents or temperature, may result in late arrival of glass eels and in turn, increase estuary settlement (Sullivan et al. 2009).

Predation on American eels is a source of natural mortality, but only a small number of diet studies have shown eels comprising significant portions of predator's diets. Fish-eating birds, such as osprey, herons, cormorants, and eagles likely prey on eels (Thompson et al. 2005; ICES 2008). One study in a freshwater tidal portion of the Hudson River found that American eels comprised 21% of the diets of bald eagles (*Haliaeetus leucocephalus*; Thompson et al. 2005). European eels are known to be found in the diets of mammalian predators, such as otter (*Lutra lutra*) and mink (*Mustela vison*; Cuthbert 1979; Britton et al. 2006), and those predators may also target American eels in the U.S. Several piscivorous fish species have been documented to prey on American eel, including striped bass (*Morone saxatilis*) and bluefish (*Pomatomus saltatrix*); however, American eels represented less than 5% of overall diets in those studies (Buckel and Conover 1997; Griffin and Margraf 2003; Walter and Austin 2003). Catfish are known to prey on European eels, and catfish abundance is shown to have a negative relationship with eel abundance (Wysujack and Mehner 2005; Bevacqua et al. 2011). Several catfish species occur in east coast rivers and they may also prey on American eels. Finally, predation by any source may also be influenced by density-dependent factors, such as eels being concentrated in select habitats or at the base of dams (Jessop 2000).

The non-native swim bladder nematode, *A. crassus*, may be reducing American eel survival during the yellow and silver eel life stages. The parasite is native to marine and freshwater areas of eastern Asia, from Japan and China to Vietnam. The nematode prefers freshwater but can survive brackish or salt water (Kirk et al. 2000). Its native host is the Japanese eel; however, the Japanese eel does not show the pathology of infection like that observed in the American eel (Sokolowski and Dove 2006).

Parasitic swim bladder infections in the American eel caused mortalities in farmed eels (Kirk 2003) and possibly wild eels. Heavy infections by *A. crassus* can lead to enlarged abdomens, swim bladder hemorrhagic lesions, fibrosis, rupture, or collapse of the swim bladder, skin ulcerations, decreased appetite and reduced growth, reduced swimming performance, and a reduced ability of the swim bladder to function as a hydrostatic organ (Sprengel and Luchtenberg 1991; Thomas and Ollevier 1992; Barse and Secor 1999; Nimeth et al. 2000; Lefebvre et al. 2002; Sokolowski and Dove 2006; Kennedy 2007). The parasite can also increase stress response that may cause secondary bacterial infections and mass mortalities in shallow lakes at warm temperatures (Kennedy 2007; Sjoberg et al. 2009). Swim bladders are irreversibly damaged by the parasite, and infections can result in early migration failure because of reduced swimming performance and inability to regulate depth during migration (Kennedy 2007; Palstra et al. 2007; Sjoberg et al. 2009).

The nematode in the U.S. likely originated from Japan (Wielgoss et al. 2008) and now occurs in most states along the eastern seaboard as well as in the Canadian provinces (Fries et al. 1996; Morrison and Secor 2003; Aieta and Oliveira 2009). In North Carolina, 52% of American eels (26–100% from different rivers) were infected with the swim bladder parasite from 1998 to 1999 (Moser et al. 2001). Chesapeake Bay infection rates were between 10% and 29% in the late 1990s (Barse and Secor 1999) and had increased to between 13% and 82% by 1998 to 1999 (Barse et al. 2001). From 2004 to 2005, there was an over 50% prevalence rate of the parasitic nematode in sampled eels from Maryland's Chesapeake Bay (K. Whiteford, Maryland Department of Natural Resources, pers. comm.). In 2007, infection rates ranged from 17.8% in the James River to 72% in the Sassafrass River, with increasing infection rates in eels in more northern Chesapeake Bay tributaries (Fenske et al. 2010). Prevalence rates in the upper Chesapeake Bay watershed (Shenandoah River) were about 2% in recent years (Zimmerman 2008), but nematodes were only recently discovered in the watershed (mid-2000s), so it is possible that infection rates may increase with time in the upper watershed as well. In the Hudson River, infection rates in the late 1990s were between 0 and 12% (Barse and Secor 1999), but increasing intensity and prevalence of infestation occurred in the Hudson River from 1997 to 2000. In the Hudson, the prevalence was lower in saline locations, with > 60% prevalence of infection in freshwater locations by 2000 (Morrison and Secor 2003). By 2004, Hudson River tributaries had an average of 39% infection rates, and dams and natural waterfalls reduced infections upstream. There were also elevated infection rates in urbanized areas (Machut and Limburg 2008). From Rhode Island to Maine, infection rates ranged from 7% to 76% in 2005, and the provinces of New Brunswick and Nova Scotia had rates from 3% to 30% in 2006 and 2007. No eels sampled from the St. Lawrence River system were infected in 2006 and 2007 (Aieta and Oliveira 2009). Currently, Cape Breton Island, Nova Scotia is the most northern area where the swim bladder parasite infestation in American eels has been documented (Rockwell et al. 2009).

2.5.6 Incidental Mortality

Incidental mortality, caused by anthropogenic activities other than harvest, can be attributed to habitat alterations and restrictions as well as mechanical and chemical injuries. Inland habitat alterations and restrictions come primarily in the form of barriers to upstream migration for American eels. These can either be physical (dams) or chemical (areas of poor water quality) factors that limit habitat use by eels. This compression of range through habitat restrictions may increase the significance of predation mortality. The location of dams may restrict eel distribution by limiting upstream movements (Levesque and Whitworth 1987; Goodwin and Angermeier 2003; Verreault et al. 2004; Machut et al. 2007). Eels live in higher densities below dams which may reduce survival by causing swim bladder parasites to spread more thoroughly by modifying sex ratios, lowering growth rates, and restricting movements between feeding areas and home areas (Krueger and Oliveira 1999; Oliveira and McCleave 2000; Strickland 2002; Cairns et al. 2004, Verreault et al. 2004; Machut et al. 2007). Upstream passage at dams, designed specifically for eels, may alleviate some of the problems associated with habitat restrictions.

Mechanical and chemical injuries can occur through the use of hydroelectric turbines, navigation lock, industrial and municipal water intakes, chemical barriers, and contaminants. Impingement, entrainment, and turbine operation, such as at dams, locks, and power plants, which can cause high rates of mortality. Entrainment of American eel elvers that pass through turbines after they pass up fish ladders can reach up to 50% with resulting turbine-related mortality (McGrath et al. 2009). Downstream migrating silver eels also can suffer high turbine mortality when moving through hydroelectric plants (Richkus and Dixon 2003; Carr and Whoriskey 2008; Brown et al. 2009; Welsh et al. 2009). Eels passing through turbines suffer up to 100% mortality at some hydroelectric sites (Carr and Whoriskey 2008). In rivers where eels must successfully pass through several hydroelectric facilities, cumulative mortality rates can be very high (McCleave 2001; Verreault and Dumont 2003; Welsh et al. 2009). Further, dams can cause delays in both upstream and downstream migration, further impacting population dynamics and potentially preventing silver eels reaching the spawning grounds during the spawning season (Richkus and Dixon 2003; Brown 2005; Welsh et al. 2009).

Behavioral barriers have not proven effective at deterring American eels and reducing turbine mortality. Physical barriers may work (Amaral et al. 2003) but are practical only in smaller systems (Richkus and Dixon 2003). Complete turbine shutdown is effective, but predicting when migration will occur can be difficult. Seasonal shutdowns can substantially reduce eel mortalities and should be based on environmental characteristics such as flow, lunar phase, and temperature as well as time of day (Haro et al. 2003; Welsh et al. 2009).

Poor water quality, such as low dissolved oxygen, drastic salinity changes, chemical spills, point source releases, and non-point source releases can cause incidental mortality of American eels. Migration through heavily contaminated areas caused acute mortality of silver eels in the early 1970s in the St. Lawrence River (Dutil et al. 1987) because the eel's ability to osmoregulate between fresh and salt water was impaired. Accumulated contaminants may reduce individual survival and reduce both egg viability and larval survival (Couillard et al. 1997). An analysis of the contaminants in migrating silver eels in the St. Lawrence River showed that the highest concentrations of chemicals were found in the gonads. Concentrations of PCB and DDT were found to be 17% and 28% higher in the gonads than in the carcasses. The chemical levels in the eggs could exceed the thresholds of toxicity for larvae. Also, since the energy with which the non-feeding migrating females produce eggs is taken from their fat reserves, the chemical levels in the eggs could be even higher at hatching, increasing the likelihood of toxicity to the larvae (Hodson et al. 1994; ICES 2006; Limburg et al. 2008). Bioaccumulation of contaminants for eels is problematic because they live in upstream areas for many years (Lamson et al. 2009). Acute toxicity and bioaccumulation of contaminants (mercury, PCB, pesticides) may reduce health and increase mortality of yellow and silver eels (Castonguay et al. 1994a).

3 HABITAT DESCRIPTION

3.1 Brief Overview

Section 3 provides a short description of American eel habitat use. A detailed review of American eel habitat requirements can be found in the Atlantic Coast Diadromous Fish Habitat document (Greene et al. 2009).

American eels exhibit a highly complex catadromous life cycle and are found in marine, brackish, and freshwater habitats (Adams and Hankinson 1928; Facey and LaBar 1981; Facey and Van Den Avyle 1987; Helfman et al. 1983). Habitat types used by different phases of eels include open ocean, estuaries, rivers, streams, lakes (including land-locked lakes), and ponds (Facey and Van Den Avyle 1987).

Habitat associations and requirements vary by life stage. After hatching in winter and spring in the Sargasso Sea, larval American eels passively migrate to brackish or freshwater along the east coast of North America where they metamorphose into glass eels (Greene et al. 2009). After developing pigment (becoming elvers), some eels start migrating upstream into freshwater while others remain in coastal rivers and estuaries. Upstream migration may continue throughout the yellow phase as well. During maturation, silver eels migrate downstream to the ocean and return to the Sargasso Sea to spawn before dying (Haro and Krueger 1991).

3.2 Habitat Description by Life History Stage

3.2.1 Spawning Habitat

American eels spawn in the Sargasso Sea from February to April; however, spawning has never been observed (Facey and Van Den Avyle 1987). The area where American eels are thought to spawn is a high salinity (~36.6 ppt) region with warm surface temperatures (>18.2°C; Kleckner and McCleave 1985). Morphological and physiological evidence suggests that American eels may spawn in the upper 150–200 meters of the water column (Kleckner et al. 1983; McCleave and Kleckner 1985).

Larval eels (leptocephali) migrate from the spawning grounds to the eastern seaboard of North America by the Antilles Current, the Florida Current, and the Gulf Stream (Facey and Van Den Avyle 1987; Munk et al. 2010). The leptocephali drift and swim in the upper 300 m of the ocean for several months (Kleckner and McCleave 1985). By August, American eel larvae occupy the entire Gulf Stream area as far north as the Gulf of Maine (Greene et al. 2009).

3.2.2 Glass Eel and Elver Habitat

Larval eels metamorphose into glass eels over the continental shelf then enter estuaries and ascend the tidal portion of rivers during winter and spring (Greene et al. 2009). Glass eels drift on flood tides and hold position near bottom on ebb tides (Wippelhauser and McCleave 1987). They also ascend by active swimming along shore in estuaries above tidal influence (Barbin and Krueger 1994). Glass eels eventually metamorphose into pigmented elvers which burrow or rest in deep water during the day. The presence of soft, undisturbed bottom sediments may be important to migrating elvers for shelter (Deelder 1958; Facey and Van Den Avyle 1987).

Elvers begin migrating upstream to freshwater during the late winter and early spring (Greene et al. 2009; Sorensen and Bianchini 1986). Migration may be triggered by temperatures above 10°C (with maximum activity at temperatures above 20°C) and by changes in water chemistry caused by the intrusion of estuarine water during high spring tides (Sorensen and Bianchini 1986; Jessop 2003). Factors that may affect daily abundance of migrating elvers include tidal height, river water temperature, river discharge, and the temperature differential between bay and river (Greene et al. 2009). Elvers have difficulty swimming in river velocities exceeding 25 cm•s-1, which can delay upstream migration (Jessop 2000; Jessop and Harvie 2003).

3.2.3 Yellow and Silver Eel Habitat

Yellow eels are associated with a wide variety of habitat types and exhibit habitat-specific growth, sexual differentiation, and movement patterns (see section 2.4.5). During the day, yellow eels are typically bottom-dwelling; however, habitat preference is not well documented and may vary by size and geographic region (Greene et al. 2009). Eels have been shown to prefer such substrates as weedy bottoms in Lake Champlain (Ford and Mercer 1986), soft sediments in the St. Lawrence River (Chaput et al. 1997), and detritus, hydroid, or shell bottoms in the Chesapeake Bay (Geer 2003). Riparian vegetation and complex substrate may be important to yellow eel in impounded systems (Thomas 2006).

Yellow eels appear to utilize different depth areas depending on time of day, season, and geographic region (Facey and LeBar 1981; Geer 2003; Thomas 2006). Water temperature affects activity and movement of yellow eels with highest activity observed above ~20°C in most settings (Geer 2003; Verdon and Desrochers 2003). Yellow eels are thought to enter torpor at temperatures less than 8°C (Walsh et al. 1983). American eels are typically found in areas with dissolved oxygen concentration in the range of 4–9 mg/L (Geer 2003; Cudney 2004). In general, yellow eels do not have specific water velocity requirements, but stream-dwelling eels have been shown to prefer sites with complex velocity-depth regimes (Wiley et al. 2004).

As yellow eels metamorphose into the silver phase, they migrate seaward in fall and winter months to their spawning grounds in the Sargasso Sea. Temperatures in the range of 9–18°C may trigger downstream migration (Vøllestad et al. 1986; Barbin et al. 1998; Vøllestad 1998). Other factors likely affecting migration include river/stream discharge, odor, light intensity, and moon phase (Greene et al. 2009). Silver eels encounter a wide range of salinities; salinity gradients may help orient eel out of estuaries (Barbin et al. 1998).

Adult oceanic habitat requirements are not known, but they have been shown to inhabit a range of depths throughout the water column from 15 to 400 m (Wenner 1973; Tesch 1978a, 1978b). Although silver eels have been found to migrate at 50–400 meters, the maximum depth recorded for Anguilla was 2,000 meters (Robins et al. 1979).

3.3 Habitat Areas of Particular Concern

Oceanic waters: The Sargasso Sea is an essential area of reproduction for the panmictic population. Climate change could affect oceanographic conditions that impact survival and transportation of larval eels to the coast of North America (see section 2.5.5).

Continental shelf: Glass eel survival in these areas may be impacted by a variety of activities including channel dredging, shoreline filling, contaminant spills and discharges, and overboard spoil disposal. However, the significance of these impacts remains unknown. Changes in salinity in embayments as a result of dredging projects could also alter eel distributions.

Estuaries and freshwater habitats: These areas serve as important juvenile, sub-adult, and adult migration corridors as well as areas where feeding and growth is concentrated for juveniles and sub-adults. Human development in and along estuaries, rivers, and streams may have a negative impact on eel health, growth, and survival. Machut et al. (2007) found that the condition (weight) of American eels in six tributaries of the Hudson River in New York was significantly lowered with increasing riparian urbanization.

Passage: The blockage of upstream and downstream migrations is a major area of concern for American eels. Upstream passage has been improved in some areas by the removal of dams and the installment of fish passage devices. However, Machut et al. (2007) found that eel densities in Hudson River tributaries were reduced 10-fold and condition (mass) was significantly lower upstream of natural and artificial barriers. In addition, downstream passage at hydropower dams may represent a major source of mortality to pre-spawning adults that has received relatively little attention (Ritter et al. 1997). Busch et al. (1996) used an ecosystem health assessment approach to determine that Atlantic coastal streams from Maine to Florida have over 15,000 dams that can hinder or prevent upstream and downstream fish movement. Such development has resulted in an estimated restriction of or loss of access to 84% of historical stream habitat for diadromous fish.

4 FISHERIES DESCRIPTION

Evidence can be found from historical literature that the American eel was a valuable source of food for indigenous populations in North America. These records are mainly brief references from the early years of European settlement that portray the following seasonal fisheries: winter spearing, spring and fall in-stream weirs to capture migrating eels, and baited wood pots set in the warm months (Lane 1978). These three fishing techniques were passed on to the European colonies and seasonal subsistence fisheries became essential food sources in the 17th and 18th centuries. With relatively little change in the basic fishing methods, subsistence fishing in many locations evolved into commercial activities as coastal populations and commerce grew.

4.1 Commercial Fisheries

Similar to earlier subsistence fisheries, commercial eel fisheries in the United States were poorly documented in the 18th century, but the accounts indicate these fisheries were widespread with local importance. Small-scale eel fisheries were common on the U.S. east coast by the 19th century, mainly supplying local food markets although commerce occurred between major cities. Regulations to preserve commercial eel fisheries in Massachusetts first appeared in statutes for Cape Cod towns starting in 1797. The earliest detailed account of U.S. eel fisheries was provided by Goode (1884) for the period of 1877 to 1880. Eel fisheries during this period were common from Chesapeake Bay to Maine with the trade market centered in New York and Boston. Fishing methods continued to include small-scale baited pots, winter spearing, and in-stream weirs or traps. Pots set from skiffs or small sailboats appeared to account for a majority of landings. The recorded eel landings in New York alone exceeded a million pounds in 1880. It appears likely that total U.S. landings were in the 1–2 million-pound range for the period reported by Goode (1884). It is presumed that the spring weirs, winter spearing, and summer pot fishing targeted yellow eels and the fall weir fishing targeted silver eels. The marine conger eel (*Conger oceanicus*) may have comprised an unknown proportion of these early records.

U.S. American eel fisheries continued without dramatic changes in the early 20th century, with market centers in the Chesapeake Bay region for blue crab fishing bait and New York and Boston for food markets. Declining U.S. landings in the period leading up to World War II may have been influenced by changing public demand for eels as a food source (Bigelow and Schroeder 1953). Overall, the U.S. American eel fishery in the 20th century experienced declining landings with some stability in meeting local market demands until the onset of the European export market in the 1960s (Lane 1978). This scenario is similar to what occurred in Quebec eel fisheries with the exception of a documented catch peak during the Great Depression as the local subsistence demand soared (Robitaille et al. 2003). In the U.S., the relative stability linked to local demand was disrupted as the export market increased in the 1960s and 1970s. Rising prices for yellow and silver eels for the European export market increased fishing effort and led to harvest peaks from the mid-1970s to early 1980s in the U.S. (Lane 1978; Jessop 1997) and eastern Canada (Jessop 1997; Robitaille et al. 2003).

Increasing demand for glass eels from Asian aquaculture operations occurred at a similar time as the European food market, dramatically increasing prices and fishing effort for glass eels (Jessop 1997; Haro et al. 2000). The fisheries for glass eels were primarily in the Canadian Maritime Provinces and Maine. The catch peaks that resulted from the European and Asian export market demand were followed with declining harvest for most regions since the mid-1980s (Peterson 1997; Jessop 1997; Haro et al. 2000). These conditions prompted management concerns over the status of the American eel population in North America.

4.1.1 Glass Eel Fishery

Fishing for glass eels (also called elvers) began relatively recently in North America and has been limited to commercial operations that target the spring runs of eels as they enter coastal rivers following their ocean migration from spawning grounds. Glass eel fisheries use in-stream fykes and traps to intercept glass eels on this spring migration. Interest in fishing glass eels developed in the early 1970s in the U.S. as demand increased from Asian aquaculture operations for “seed” stock (Fahay 1978; Keefe 1982). Glass eel fisheries in Canada came later beginning in 1989 with the issuance of experimental licenses in Nova Scotia and New Brunswick (Jessop 1997; Peterson 1997).

The states of Florida, North and South Carolina, Virginia, Massachusetts, and Maine initiated glass eel fisheries in the early 1970s (Fahay 1978; Keefe 1982). The glass eel fisheries failed to develop in Florida, ceased in 1977 in North Carolina, and were prohibited in 1977 by a six-inch minimum size limit in Virginia and a four-inch minimum size limit in Massachusetts (CBP 1991). The Potomac River Fisheries Commission imposed a six-inch minimum size in 1992 for both commercial and recreational fisheries, eliminating glass eel fisheries within their jurisdiction. The Maine glass eel fishery collapsed after 1978 due to market conditions but continued at a low level until growing substantially in 1994. During the late 1980s or early 1990s, glass eel fisheries were developed or reestablished in Connecticut, Rhode Island, New York, New Jersey, Delaware, and South Carolina, but no catch data are available. Glass eel fisheries do not occur in any Gulf of Mexico states. With the implementation of the ASMFC Interstate Fishery Management Plan for American Eel in 2001 (ASMFC 2000), all Atlantic coast states and jurisdictions except Maine, South Carolina, and Florida implemented a six-inch minimum size limit for American eels. Florida eel fisheries operate under gear restrictions that do not allow the landings of eels under six inches.

Prior to 2010, only the Maine glass eel fishery was consistently active. The fishery operates with relatively few permits and limited entry. Glass eel landings in Maine have been recorded separately from other eel catches since 1994. The peak landings since 1994 occurred in 1995 at 16,599 pounds of glass eels. Landings have been less than 5,000 pounds in the last decade, and the fishery is distinguished by high prices often in the range of $200–300 per pound. In 2011, anecdotal reports were received of glass eel prices exceeding $1,000/pound and renewed fishing activity in South Carolina. The increased demand may also have contributed to an increase in illegal poaching in jurisdictions where glass eel fisheries are prohibited.

4.1.2 Yellow Eel Fishery

The yellow eel life stage is readily captured with baited pots in coastal rivers and freshwater habitats and provides a size range suitable for food and bait markets. This life stage of American eel has been the primary target of U.S. eel fisheries in both historical and modern periods. The U.S. fishery for yellow eels extends from the Gulf of Mexico to Maine. Different geographic regions (Gulf of Mexico, and the North, Mid-, and South Atlantic) have exhibited differing trends and magnitudes in their eel fisheries, which reflect differences in the fisheries and stock abundance among the regions (Fahay 1978). Section 5.0 reviews the fisheries in each region in greater detail.

The dominant gear for targeting yellow eels in U.S. eel fisheries has been baited pots. The practice of using hand spears for winter eel harvest in northeastern coastal rivers was common until fading to an incidental practice in the last two decades. The use of in-river weirs and fykes to capture spring movements of yellow eels has not been a widespread practice but has provided important local fisheries in some regions. The contributions of both spear and other non-pot fisheries have been minor relatively to overall U.S. eel harvests and are incidental in contemporary fisheries.

Harvest patterns in yellow eel fisheries have followed market influences as described in section 4.0. Nineteenth century U.S. harvests are poorly documented; however, the available references (Goode 1884; Bigelow and Schroeder 1953; Lane 1978) portray important local markets driving much higher effort and catch than occurring in the 20th century. After an apparent period of declining demand in the first half of the 20th century, there were catch peaks occurring from approximately 1955–1985 that coincided with increasing market demands from the Chesapeake Bay region crab fisheries and the European food market (Fahay 1978; Lane 1978). The timing of harvest peaks varied among states. By the 1990s, most states experienced declining harvests influenced to an uncertain degree by both the weakening export market and local abundance. The declining U..S and export food market demand has been partially offset by increasing demand for yellow eels as striped bass bait. This relatively new market feature appears to be driving many local fisheries in the last decade, although U.S. catch levels are at historic lows, with few regional exceptions.

4.1.3 Silver Eel Fishery

American eels enter the silver-phase of their life history as they begin their reproductive migration from fresh to marine waters. Silver eels become vulnerable to passive nets and traps set in rivers as they migrate downstream during the fall. Silver eels were targeted by Native Americans and later for commercial food markets because of the high fat content of mature eels, which produced an excellent smoked product. Harvest data and information on silver eel fisheries are poorly documented. Silver eel fisheries have not been nearly as common as yellow eel fisheries in the U.S. past and present. It is likely that niche fisheries occurred at specific rivers along the east coast in historical times leading up to the mid-20th century. A traditional silver eel fishery using fyke nets operated in the Albemare Sound region of North Carolina in late summer and early fall during the mid-1970s with as many as 50 active fishermen (Fahay 1978). Much larger scale silver eel fisheries occurred with fixed traps for much of the 20th century in the St. Lawrence watershed with sharply declining catch in recent decades (Robitaille et al. 2003; Verreault et al. 2003). Under the present ASMFC management plan silver eel fisheries are only allowed in New York and Maine and occur with low levels of catch and effort (ASMFC 2000a).

4.1.4 Bait Fishery

The use of harvested American eels for bait in other fisheries is not well-described, although it does not appear to have been common before the 20th century nor had the relative importance of food markets. Eel harvesting in the South Atlantic Bight prior to the 1970s was focused primarily on harvesting eels for live bait in sportfisheries and secondarily as bait for blue crab pots (Van Den Avyle 1984). Harvesting eels for crab trotline bait was important in the Maryland eel fishery in the 20th century (Foster and Brody 1982). The proportion of the eel harvest sold for bait declined with the advent of the overseas food market in the 1960s, and this disposition declined further as the increased use of crab pots reduced the need for baited trotlines (Lane 1978).

A more recent development in the marketing of American eels in U.S. fisheries is the use of eels for striped bass, cobia, and catfish bait. Several references that summarize U.S. eel fisheries prior to the 1990s (Fahay 1978; Lane 1978; Van Den Avyle 1984) do not mention this harvest disposition, and more recent references mention the practice with no details (Haro et al. 2000; Collette and Klein-MacPhee 2002). It is likely that the practice of rigging eels for striped bass angling originated early in the 20th century but did not become widespread until recently. Presently, the use of eels as striped bass bait is probably the dominant use of harvested eels in New England and comprises a larger proportion of the Chesapeake Bay eel fishery than any time previous. U.S. eel fishery data does not have the resolution to separate striped bass bait from other dispositions. Commercial eel fishery reporting since the implementation of the ASMFC eel management plan in 2001 has improved and could provide information on this recent development.

4.1.5 Exports

The weight and value of U.S. domestic exports of American eels from selected districts along the Atlantic coast for 1981–2010 were provided by the NMFS (1981–1988; Fisheries Statistics Division, Silver Spring, MD, pers. comm.) and the United States International Trade

Commission (USITC) DataWeb (1989–2010; pers. comm.). Export values were converted to 2010 dollar values using conversion factors based on the annual average consumer price index (CPI) values, which were obtained from the U.S. Bureau of Labor Statistics (pers. comm.).

Prior to 1989, exports were classified as either fresh/frozen or live. Since 1989, the fresh/frozen group has been separated into two categories—fresh (or fresh or chilled) and frozen. Live export weight data for American eels are not available for the 1989–1992 time period, likely due to differences in reporting requirements during those years (A. Lowther, NOAA Fisheries, pers. comm.; M. Savage, USITC, pers. comm.).

Domestic exports of American eels from the Atlantic coast ranged from 229 thousand to over 6.07 million pounds per year from 1981 through 2010 (Figure 4.1). Live eels comprised the majority (>50%) of exports in 1983–1988, 1993, 1999, and 2003–2005. In more recent years, exports of fresh and frozen eels have dominated, accounting for an average of 76% of the total eel exports per year during 2006 through 2010. The reason that the magnitude of domestic exports exceeds commercial landings in some years may be that export landings records include significant quantities of hagfish misreported as American eel.

The value of American eel exports ranged from 1.83 to 23.5 million dollars per year over the time series (Figure 4.1). Export values decreased during the earliest years in the time series and then generally increased to the peak observed in 1997. The value of exports substantially dropped following the 1997 peak but has shown a generally increasing trend from 1999 through 2010.

The value per pound of exported American eels classified as live has exceeded the value per pound of fresh and frozen eels (combined) throughout the time series (Figure 4.2). The value per pound of fresh and frozen eels ranged from 0.819 to 4.97 dollars per pound per year from 1981 to 2010. The value per pound of fresh and frozen eels has exhibited a general decline over the time series. The value per pound of live exports has varied over the available time series, ranging from 2.53 to 21.8 dollars per pound per year.

4.2 Commercial Catch-Per-Unit-Effort

Fishery-dependent catch-per-unit-effort (CPUE) was available in some states, but following review of these data they were not considered indicative of trends in the stock as a whole (see section 5 for more details on data inclusion/exclusion decisions). Note that fishery-dependent CPUE is almost exclusively composed of positive trips only; trip reports with zero eels caught are rare because most agencies don't require reports of zero catches.

Maine—Glass Eel

Estimates of number of licenses sold by gear type are available from 1996 forward as an estimate of effort (Figure 4.3). An average of nearly 2,000 harvesters participated in the glass eel fishery annually during 1996 to 1998. In 1999, the Maine DMR implemented effort restrictions, capping the fishery at 827 participants. Since then, effort has averaged approximately 490 participants, with a range of 267 to 743.

Glass eel dealer reporting has been required since 1999, although voluntary data are available back to 1996. Catch per effort in the fyke net fishery has fluctuated without trend since 1999, averaging 7.85 pounds per fyke net licensed, with a range of 3.2 to 19.2 pounds per fyke (Figure 4.4). CPUE for the dip net fishery was generally less than 1 pound per unit of gear from 1999 to

2004 but increased dramatically in 2005 to 16.7 pounds per net (Figure 4.4). Since then, it has fluctuated without trend between 3.2 and 9.6 pounds per net. Harvest per licensed fisherman has followed a similar trend as the fyke net fishery (Figure 4.4).

Attempts were made to identify major factors influencing the Maine glass eel fishery, such as price per pound, YOY abundance, and participation. Unfortunately, changes in management over time (voluntary/mandatory reporting, effort restriction) and other factors made this difficult because there was no consistent time series of all three datasets.

Maine—Yellow Eel

Mandatory harvester reporting for the yellow eel fishery began in 2007 but is considered less reliable than the dealer data (i.e., harvesters report estimated harvest weights at the trip level while dealers report actual weigh-out for individual transactions; G. Wippelhauser, ME DMR, pers. comm.) and therefore will not be considered further in this assessment. Harvester reporting (monthly summaries) has been required in Maine's coastal and inland yellow eel pot fisheries since 2001, with voluntary data back to 1999.

Two measures of effort are available for the yellow eel pot fishery—records of the number of licenses sold by year are available beginning in 1985, while estimates of total gear days fished are available beginning in 2001 (Figure 4.5). Participation in the coastal and inland fisheries generally increased between 1985 and 1995 but has since declined to between 10 and 15 participants per fishery since 2001 for both fisheries and to less than 10 in the inland fishery since 2007. The coastal fishery exerts approximately 85% of the total pot fishery effort (days) despite participation in the two fisheries being roughly equal over much of the last decade. Since license sales have been relatively static, the decline in pot days for the coastal fishery also suggests a general decline in pot days fished per license since 2001.

Trends in catch per license sold are similar to those of harvest as a result of license sales being relatively constant over the last decade. CPUE generally increased during the early 2000s, peaked in the mid-2000s, and has since returned to previous levels (Figure 4.6). CPUE evaluated against pot effort shows more variability with no distinct trends.

Trends in weir fishery effort provide some insight into the observed harvest patterns. Prior to 1996, effort in the weir fishery was unregulated. In 1996, effort was limited to a maximum of 26 harvesters at 42 sites (P. Bourke, ME DMR, pers comm.). Effort declined from 50 licensed sites in 1995 to just 2 sites in 2002 and has remained below six in all years since 2002 (Figure 4.7). Catch per licensed site appears relatively stable with the exception of one high and one low outlier in 2004 and 2006, respectively.

New Hampshire

The New Hampshire Department of Fish and Game has recorded commercial catch and effort for American eel since 1990. Annual CPUE indices were estimated from annual summaries of trips that reported valid total catch, pot number, and soak duration. Trip level reporting was well documented during this period. The total landings reported were low; therefore, the CPUE statistics are generated from landings and effort that may not represent a commercial fishery but rather a small-scale fishery to catch striped bass bait for personal use. Despite the low levels of catch and effort, the CPUE was routinely higher than observed in nearby states such as Massachusetts. Permit holders appeared to be setting few pots and having catches that approximated 1 pound/pot/day on average. The general trend for this time series was > 1.0 pounds/pot/day during the 1990s and < 1.0 pounds/pot/day for most of the 2000s (Figure 4.8).

Massachusetts

Catch-per-unit-effort data were summarized by major coastal drainage areas (Merrimack River, Plum Island Sound, North Coastal Basin, Boston Harbor, South Coastal Basin, Cape Cod, and Buzzards Bay). Annual CPUE indices were computed from annual summaries of trips that reported valid total catch, pot number, and soak duration. Most effort and landings during 2001–2009 occurred in Cape Cod and Buzzards Bay watersheds. Because of the low catches, landings were pooled into the regions of Southern New England (Cape Cod and Buzzards Bay) and Southern Gulf of Maine (all basins north of Cape Cod). The development of indices of abundance from the Massachusetts pot data may be limited because few permit holders are contributing trip-level data and because of apparent changes in fishing practices. In recent years, few participants are targeting larger catches for commercial sales to food or bait markets and most are catching small amounts to supply their own needs bait fishing in the commercial striped bass market. The CPUE for Southern New England during 2001–2009 shows some stability in catch rates with the highest CPUE at the start of the series and in 2009 (Figure 4.9).

Rhode Island

The Rhode Island Division of Fish and Wildlife began trip-level reporting of commercial catches for American eel in 2007. The time series was considered too short for calculating CPUE but will be revisited in the next stock assessment. The Rhode Island eel potting fishery is similar in scope to those described in Connecticut, Massachusetts, and New Hampshire. The relatively low number of participants and total landings reflect a small-scale, part-time, seasonal fishery. Additional quality assurance review is needed to resolve questions on potential misreporting of conger eel catches and reporting of eel landings under trips with lobster gear codes.

Connecticut

Connecticut has recorded catch and effort data for their commercial eel pot fishery since 2000. Annual CPUE indices were calculated from annual summaries of trips that reported valid total catch, pot number, and soak duration. An alternative CPUE estimate was also generated using the sum of annual total catch divided by the sum of annual total pots fished. The trends of the two indices for 2000–2008 were essentially identical. Annual trip level CPUE shows a general increasing trend with CPUE < 1.0 pounds/pot/day in the first half of the series and several years exceeding 1.0 pounds/pot/day in the latter half. The Connecticut CPUE values are within the range recorded in New Hampshire and Massachusetts during this time period, and the Connecticut eel pot fishery displays similar characteristics of low participation and small-scale, seasonal operations.

New Jersey

New Jersey has maintained records of the number of eel licenses sold on an annual basis since 1999. The number of licenses sold has been relatively constant over time, with a minimum of 142 licenses sold in 2001 and a maximum of 202 in 2007 (Figure 4.10). Although not every license sold was active in a given year, these records allow investigation into trends in CPUE (catch per license sold) since 1999. Because effort has been relatively stable, the trend in CPUE has mirrored the trend in harvest. CPUE increased from the time series low of 300 pounds per license in 2000 to a peak of 900 pounds per license in 2006. CPUE has since fallen by approximately 30%.

Delaware

Delaware mandated catch and effort reporting from the American eel fishery in 1999. Delaware considers its American eel catch and effort records since 1999 fairly accurate and has calculated an annual commercial CPUE index from 1999 to the present. The annual index value for CPUE is expressed as catch per pot-day fished and is the ratio of all eel pounds harvested by eel pots divided by the total number of eel pot-days fished (1 pot-day = 1 eel pot fished for 1 day). Annual CPUE ranged from 0.99 pounds/pot-day in 2009 to 2.71 pounds/pot-day in 2005 (Figure 4.11). Pot-days fished has varied and CPUE has usually been higher during years in which pot-days fished was below the time series mean.

Maryland and Delaware Coastal Bays

A commercial CPUE index was calculated for the pot fishery in Maryland (1992–2010) and Delaware (2000–2009) Coastal Bays (Figure 4.12). The annual index value for CPUE is the ratio of the sum of all eel pounds harvested by eel pots and the sum of all eel pots fished. Maryland Coastal Bay eel pot effort in 2001 was reported as 25 pots with 120 pounds of eel harvested. This CPUE, computed as 4.80 pounds/pot, was nearly five times the average for all other years and was considered a severe outlier so the data point was removed. CPUE in Delaware coastal eel pots was 1.53 pounds/pot compared to 0.57 pounds/pot in Maryland's coastal bays. However, pots used in Maryland are typically the smaller cylindrical pots rather than larger square pots commonly used in Delaware. Independently, no trend was apparent in either series. Differences in pot catchability would make it difficult to develop a combined Delaware and Maryland coastal bays CPUE index.

Maryland

From1992, when mandatory catch and effort reporting was fully adopted by commercial eelers a commercial CPUE index was calculated for the pot fishery. The annual index value for CPUE is the ratio of the summation of all eel pounds harvested by eel pots and the summation of all eel pots fished. Average annual CPUE has ranged from a low of 0.31 lbs/pot in 1992 to a high of 1.01 lbs/pot in 2006. The CPUE index was relatively flat from 1992–2002, significantly increased from 2003–2006, and slightly declined and moderated from 2007–2010 (Figure 4.13). Total effort measured as the number of eel pots fished steadily declined from 1999 to 2006, leveled off from 2007–2009, and had an approximate 50% increase in 2010. Effort declined 60% from a high of 889,000 pots fished in 1997 to a low of 320,000 pots fished in 2006 and has averaged approximately 417,000 pots fished from 2007–2010. A significant negative correlation was detected between the pot CPUE and pot effort (Pearson product-moment correlation: $r = -0.78$, $P < 0.01$,).

Potomac River

Monthly catch and effort was required of commercial eelers beginning in 1988. In 1990, monthly reporting was changed to mandatory weekly reporting and then mandatory daily reporting began in 1999. The annual index value for CPUE is the ratio of the summation of all eel pounds harvested by eel pots and the summation of all eel pots fished. The same pattern of increasing CPUE with decreasing effort was noted for the PRFC commercial pot index as well Maryland's over the same time frame (Figure 4.14). Average annual CPUE has ranged from 1.11 lbs/pot in

1999 to a high of 2.19 lbs/pot in 2007. CPUE was relatively flat from 1988–2001 but increased from 2002–2007 before moderating back at approximately 1.5 pounds/pot from 2008–2010. Commercial effort in total eel pots fished declined by over 40% in 2002 from the previous year and has continued to gradually decline through 2010. Effort has decreased approximately 85% from a time series high of 225,000 pots in 1994 to a low in 2010 of 34,500 pots.

Virginia

Catch rates were calculated for Virginia's commercial eel pot fishery by dividing the amount of harvest of American eels landed in Virginia (pounds) by the number of eel pots. The catch rates were calculated for the James, York, and Rappahannock rivers using data specific to each river. Only data associated with positive effort are included in the calculations as commercial harvesters only report positive catches to the VMRC. Records where harvest or effort were missing or zero were excluded from the calculations.

Annual catch rates were variable within and among rivers over the time series (Figure 4.15). Catch rates for the James and York rivers demonstrated a decline during the mid- to late 1990s. The peak catch rate occurred in 2002 for both the James and York rivers. The York River catch rates show evidence of a general decline from 2005 through 2009. Catch rates for the Rappahannock River have shown no obvious trends over the time series.

North Carolina

CPUE from the North Carolina trip ticket data are not a reasonable index of abundance for eel because effort has not been recorded consistently in trip or haul units. Many fishermen keep eels caught from multiple trips in pens, then combine and sell the entire batch to a dealer under one trip ticket. In the future, logbook data (which began in 2007) may be useful for computing fishery-dependent index of abundance; logbooks include exact number of trips, eels caught per trip, pots fished, and soak time.

Florida

Commercial catch and effort data collection began in 2006. The time series was considered too short for calculating CPUE but will be revisited in the next stock assessment.

4.3 Recreational Fisheries

Studies and reports that summarize U.S. eel fisheries provide little information on targeted recreational eel fisheries (Bigelow and Schroeder 1953; Fahay 1978; Lane 1978; and Van Den Avyle 1984). The practice of spearing or gigging eels buried in the mud during winter is an eel fishing method that was developed for subsistence fishing but came to have both commercial and sportfishing appeal in the 19th century until recently. Eels are encountered over much of their U.S. range by recreational anglers as bycatch. Van Den Avyle (1984) reported that no major sport fishery for American eels occurred in coastal rivers of the South Atlantic Bight, but incidental catches were made by anglers in estuaries and rivers. Despite the incidental nature of eel hook-and-line catches, the Marine Recreational Fisheries Statistical Survey (MRFSS) does encounter enough observations to generate catch estimates that indicate widespread and common presence as a bycatch species. Starting with 1981 estimates, the MRFSS survey for all major eastern U.S. regions show much higher catch estimates in the 1980s than in the 2000s (NMFS, pers. comm.). For example, the mid-Atlantic region annual estimates averaged over 49 thousand pounds in the 1980s and about 9 thousand pounds in the 2000s. For the North Atlantic, the decline is sharper: after averaging over 20 thousand pounds annually in the 1980s, no catches have been reported since 1999.

4.4 Subsistence Fisheries

The harvest of American eels as a food source for subsistence has been portrayed as having importance for Native Americans and European settlers in North America with declining importance after the 19th century. Most accounts are anecdotal and entail brief references in popular literature. The journals of William Bradford and Edward Winslow of the pilgrim settlement in Plymouth, Massachusetts make several references to the abundance and use of eels by Native Americans and the Pilgrims in the 1620s for subsistence (Young 1841). Thoreau recorded his travels to Cape Cod, Massachusetts in the mid-19th century and included several references to being served eels with meals prepared with locally gathered food (Thoreau 1951). Robitaille et al. (2003) considered the subsistence catch of eels in Canada to have been important for indigenous tribes and European settlers with declining importance in the 19th century and minor value in the 20th century with the exception of the Great Depression when the highest recorded Canadian catch was made in the 1930s. These accounts portray fried and smoked eel as a common food gathered for subsistence in coastal regions until recent generations. It is likely that changes in eel abundance and demand have diminished this practice in the 20th century resulting in declining cultural importance of eels in coastal communities.

4.5 Gulf of Mexico

A small portion of U.S. landings are attributed to the Gulf of Mexico. Landings records in this region were historically collected by the NMFS but have been administered by the Gulf States Marine Fisheries Commission since 1985 (D. Bellais, GSMFC, pers. comm.). Between 1950 and 1999, landings in the Gulf of Mexico ranged between approximately 200 pounds in 1994 and 28,000 pounds in 1985 (Figure 4.16). Landings reported since 1999 have been negligible and are thus confidential. Fahay (1978) reported total U.S. landings of American eels during 1955–1973 with minor landings registered from the U.S. Gulf of Mexico region during about half of those years but never exceeded 1% of total U.S. landings. Note that the Gulf States (including western Florida) are under the jurisdiction of the Gulf States Marine Fisheries Commission and are not subject to ASMFC-led interstate fisheries management.

4.6 Fisheries Outside the United States

Because of the panmictic status of American eel, fisheries outside the jurisdiction of the United States are relevant to ASMFC management efforts, although they are not subject to management regulations implemented though the ASMFC. Brief descriptions of Canadian eel fisheries and fisheries at locations south of the United States are provided below for perspective on activity at the northern and southern ends of American eel's range. Information on commercial eel landings in Canada and other western Atlantic countries was obtained from the Department of Fisheries and Oceans (DFO) Canada (DFO, pers. comm.) and the Fisheries Department of the Food and Agriculture Organization (FAO) of the United Nations (FAO, pers. comm.), respectively.

4.6.1 Commercial Fisheries in Canada

American eels are present in Canada from Labrador southward and are considered abundant in the St. Lawrence River watershed, southern Newfoundland, and the Maritimes Provinces (Scott and Scott 1988). General regional differences are found in Canadian eel fisheries. Quebec fisheries mainly use weirs set in rivers for silver eels, baited setlines and fyke nets are mainly used in Ontario, and the Maritimes utilize a wider variety of baited pots, weirs, traps and spearing. The eel fisheries in the St. Lawrence River main stem, tributaries, and watershed have traditionally had the highest landings among Canadian regions (Lane 1978; and Scott and Scott 1988). Similar to the eastern U.S., eel fisheries occurred with periods of local importance for several centuries in all the Canadian Atlantic provinces.

Robitaille et al. (2003) describe two harvest peaks in Ontario and Quebec that occurred in the 20th century. The first occurred in the 1930s and was driven by economic influences of the Great Depression. The highest total Canadian eel catch recorded came in 1933 at approximately 1,224 tons (Lane 1978) and was probably underreported (Robitaille et al. 2003). Eel catches declined following this peak with likely but undocumented influences of reduced abundance and market demand due to improved economic conditions. The 1950s and 1960s was a period of relative stability with eel fisheries meeting the demand of local markets. The stability was disrupted by the onset of the export markets, first for food markets in Europe and followed by culture markets for juvenile eels in the Far East (Robitaille et al. 2003). This resulted in the second catch peak during 1975–1980 with Ontario and Quebec landings near 800 tons. Lane (1978) reported that total Canadian eel catch ranged from 800 to 1,200 tons from 1965 to 1973—a period of rising harvest to meet export demands. The total harvest weights in Canada were very similar to the U.S. totals during this period. The eel fisheries in the St. Lawrence River main stem, tributaries, and watershed have traditionally had the highest landings among Canadian regions (Lane 1978; and Scott and Scott 1988).

Eel harvest in the Ontario and Quebec Provinces declined quickly from the late 1970s peak to the 1990s (Peterson 1997). Sharp declines occurred in the St. Lawrence estuary weir fishery that targets female silver eels in the 1990s (Verreault et al. 2003). Management concerns from these regions were a significant impetus for the contemporary review of American eel stock status. At the same time that concerns were growing in Quebec and Ontario, landings increased in Nova Scotia and New Brunswick as export markets in the 1990s attracted effort for food eels and glass eels for culture. Declining market demand, implementation of regulations, and likely abundance reductions have reduced effort and catch in most Canadian regions in the last decade. Canadian Provincial and Federal fishery management agencies are now actively engaged in American eel population assessment and restoration.

The DFO Statistical Services Unit maintains fisheries data for Canada and these data were available for 1972–present. Data from Canada's marine and freshwater commercial fisheries are available via online tables that are summarized by species, province, and region (e.g., Scotia-Fundy vs. Gulf). Trends in seafisheries records from 1972 to 2009 indicate a steady decline in commercial eel landings since the early 1990s (Figure 4.17). Available freshwater fisheries records cover a shorter time span (1990–2006) during which time a small decline in freshwater landings is apparent. However, freshwater landings records may be less reliable than seafisheries records (Figure 4.18; note exact repeated numbers between 1998–2000 and 2004–2006), and it is unclear whether overlap in reporting between freshwater fisheries and seafisheries occurs.

4.6.2 Commercial Fisheries in Central and South America

Studies and reports that summarize U.S. eel fisheries provide no information on commercial eel fisheries in Mexico or the Caribbean Islands other than mentioning that the American eel’s range does extend to these regions (Bigelow and Schroeder 1953; Fahay 1978; Lane 1978; and Van Den Avyle 1984).Annual landings between 1950 and 2008 are available by country and major fishing area from the FAO Fishery Global Statistics Program of the Fisheries Data, Information, and Statistics Unit (FIDI) via online tables. Mexico, the Dominican Republic, and Cuba have reported a small amount of landings (primarily from in-river fisheries) since 1975 (Figure 4.19). It is unknown whether these reports are comprehensive.

5 DATA SOURCES

For this assessment, the committee evaluated nearly 100 fishery-dependent and independent U.S. data sources representing several life stages and geographical and temporal scales. Canadian data sources were examined but not included in this assessment because a Canadian stock assessment was being conducted by DFO concurrently with the U.S. assessment. Hopefully, the two sets of analyses will be considered together and combined to form a West Atlantic assessment in the near future.

Fifty-two fishery-dependent and independent data sources were selected for use in this assessment because they were considered adequate for describing life history characteristics and abundance trends of eels on either a coast-wide or regional basis. After close consideration by the committee, trends in fishery-dependent CPUE were described in section 4 to describe the fisheries themselves but were not included in analyses because they were not thought to represent trend in eel abundance over time due to either poor participation in the fishery (i.e., few fishers) major, unquantified changes in the fishery over time, or insufficient time series.

In addition, some fishery-independent data sources were removed from consideration for one or more of the following reasons:

1. Lacked sufficient time series to identify trends (<10 years)
2. Reported inconsistent sampling methodology (i.e., frequent changes in survey methodology) that could not be accounted for via standardization techniques
3. Resulted in intermittent or rare catches of eel
4. Operated during a time of the year or in an area where when eel are not typically available to sampling gear
5. Used survey gear with rare, uncertain or biased catchability for eel

A summary of all available data sources and a brief description of the reasons any dataset was excluded can be found in Appendix 1. Note that the ASMFC-mandated annual YOY surveys sources are not included in Appendix 1 but are treated separately in section 5.2.1.1.

5.1 Fishery-Dependent

5.1.1 Commercial Fisheries

The FMP for American eel requires states to report commercial harvest by life stage, gear type, month, and region as defined by the states (ASMFC 2000a). At this time, however, not all states are able to provide this level of information.

5.1.1.1 Atlantic Coast

Historical commercial landings data from 1888 to 1940 were transcribed from online U.S. Fish and Fisheries Commission Annual reports (NOAA Central Library Data Imaging Project, pers. comm.).

Commercial landings data collected since the 1900s were obtained from the Atlantic Coastal Cooperative Statistics Program (ACCSP) or from state-specific databases in situations where data flow issues between the states and ACCSP were identified during the 2009 American Eel Data Workshop (see state-specific data collection details below). Since 1950, most landings information on the East Coast has been collected by NMFS through dealer and/or fisherman reporting under a state-federal cooperative program. All historical NMFS data are now housed at ACCSP. Prior to the 1990s, information was summarized annually or monthly; more detailed information became available as states individually began adopting fisherman reports (e.g., trip ticket systems or logbooks).

During 1950 to 2010, Atlantic coast-wide U.S. American eel landings ranged between approximately 664,000 pounds in 1962 and 3.67 million pounds in 1979 (Figure 5.1). After a decline in the 1950s, landings increased to a peak in the 1970s and 1980s before declining again in the 2000s.

Geographic regions used in the 2005 assessment (North, Mid-, and South Atlantic) exhibited differing trends and magnitudes in their eel fisheries (Figure 5.2). The majority of landings were reported in the Mid-Atlantic (New Jersey to Virginia), followed by the South Atlantic (North Carolina to Florida) and North Atlantic (Maine to New York). Since the coast-wide landings peak in the 1970s and 80s North and South Atlantic landings have been minimal compared with Mid Atlantic region landings.

A new set of watershed-based geographic regions were created for this assessment: Gulf of Maine, Southern New England, Hudson River, Delaware Bay/Mid-Atlantic Coast Bays, Chesapeake Bay, and the South Atlantic (Figure 5.3). The temporal extent to which landings could be assigned by region (i.e., divide landings within a state like Massachusetts or Maryland) varied by region (Figure 5.4). The South Atlantic and Chesapeake Bay regions showed distinct large peaks in landings in the early 1980s. Landings in all regions declined throughout the 1990s. Most regions remained stable throughout the 2000s except for Southern New England and Delaware Bay/Mid-Atlantic Coast Bays where landings declined.

The value of U.S. commercial American eel landings as estimated by NMFS has varied between a few hundred thousand dollars (prior to the 1980s) and a peak of $6.4 million in 1997 (Figure 5.5). Total landings value increased through the 1980s and 1990s, dropped in the late 1990s, and increased again in the 2000s.

Since 1950, the majority (>76%) of American eel landings were caught in pots and traps (Figure 5.6). Fixed nets (e.g., weirs, pound nets) accounted for about 8% of the landings. Approximately 4% of landings were caught using other gears (non-pot/trap or fixed net). About 12% of landings are reported with unknown gear type. Over the last two decades, pots and traps have become the dominant gear reported for most eel landings (Figure 5.7).

<u>Potential Biases</u>

NMFS data collection is focused on species that are managed exclusively or jointly at the federal level, although information is also collected on species that are managed at the state level. Other caveats associated with these data are discussed at the following web site: http://www.st.nmfs.noaa.gov/st1/commercial/landings/caveat.html. Because eel is managed by the states and is not a target species for the NMFS, landings information for states that rely on the NMFS estimates may be underreported. In addition, at least a portion of commercial eel landings typically come from non-marine water bodies. Even in states with mandatory reporting, these requirements may not extend outside the marine district, resulting in a potential underestimate of total landings. Despite concern about the level of under reporting, the committee felt that reported landings were indicative of the trend in total landings over time.

In both federal and state landings reports there may be misreporting of other eel species (e.g., conger eel) as American eel either due to data entry mistakes or lack of species-specific reporting requirements (e.g., historical Florida). The committee has vetted the data where possible to eliminate known cases of misreporting by species (e.g., sand eels in Massachusetts); however, an unknown amount of American eel landings used in this assessment may actually be other species of eel; therefore marine landings of American eels in some areas and years may be over reported.

Purchase records made available by the Delaware Valley Fish Company, one of the largest eel dealers in the United States (M. Feigenbaum, DVFC, pers. comm.) were reviewed during the 2005 assessment. In several instances, purchase records from DVFC indicated a larger harvest of eels for a given state and year than were reported as landings by the NMFS. This emphasized the concerns of the Technical Committee that landings may be underreported to the NMFS. However, despite the discrepancies, the trend in total landings reported by NMFS was generally consistent with the trend in landings reported by the DVFC.

5.1.1.2 State-specific data collection

5.1.1.2.1 Maine

Fishery-dependent data collection in Maine consists of dealer reporting for the glass eel fishery, and harvester reporting for glass eel, yellow eel (eel pot), and silver eel (weir) fisheries.

Dealer Reporting

Glass eel dealer reporting has been required since 1999, although voluntary data are available back to 1996 (Figure 5.8). The primary gear used to harvest glass eels is the fyke net, which has accounted for approximately 78% of the landings since 1999. Dip nets are often used as a test gear to evaluate new sites (G. Wippelhauser, ME DMR, pers. comm.), but landings reported from dip net gear has increased since 1999.

Harvester Reporting

Mandatory harvester reporting for the glass eel fishery began in 2007, but it is considered less reliable than the dealer data (i.e., harvesters report estimated harvest weights at the trip level while dealers report actual weigh-out for individual transactions; G. Wippelhauser, ME DMR, pers. comm.), and will not be used in the assessment. Harvester reporting (monthly summaries) has been required in Maine’s coastal and inland yellow eel pot fisheries since 2001, with voluntary data back to 1999. The yellow eel fishery is dominated by the coastal pot fishery, which averaged more than 95% of the harvest between 1999 and 2008, but the trends in landings are similar between the two fisheries.

5.1.1.2.2 New Hampshire

Dealer Reporting

For the years 1955–1990, landings estimates were obtained from the NMFS dealer reporting system (as provided by ACCSP).

Harvester Reporting

Beginning in 1990, the New Hampshire Fish and Game Department provided landings and effort estimates from their coastal harvest logbook program.

5.1.1.2.3 Massachusetts

Dealer Reporting

American eel landings estimates from 1950 to 1993 were obtained from the NMFS dealer reporting system (as provided by ACCSP). The ACCSP and NMFS harvest records were limited for 1994 to 1999, a period when the European export market declined sharply. Historical notes and memos indicate this was a transition period with declining market demand and local eel abundance influencing reduced landings. Local demand for eels as striped bass bait created effort that may not have been picked up by Federal monitoring focused on larger food markets. For the period of 1994–1996, data found in archived files of phone interviews of coastal towns with eel fisheries were used. For 1995 and 1996, the small amount of additional poundage reported by ACCSP was added to the coastal survey totals. No data were found for 1997–1999. Unfortunately, 1994–2003 was a transition period of declining landings in Massachusetts that was poorly documented and underreported. It is likely that the slope of the decline was more gradual from 1993 to 2004 than seen in the available data. In the absence of actual catch reports, landings for 1997–1999 were estimated as the average of the three years before and after this period (3,456 pounds). The small amount of additional poundage in these years reported by ACCSP was added to the average.

The Massachusetts Division of Marine Fisheries monitoring under the ASMFC American Eel Management Plan began in 2000 with a dedicated effort to improve reporting of commercial eel harvest. The landings reported to ASMFC for 2000 to 2009 are the most accurate among all data sources for Masachusetts' commercial eel harvest. Underreporting of eel harvest for striped bass bait is a negative influence on catch data from this period, and this was likely a larger problem in the first few years of this series when the reporting process was being developed. For this reason, landings from the time period 2000–2003 are highly uncertain due to expected underreporting.

Harvester Reporting

Trip-level reporting began in 2004. In general, data quality improved during 2004–2009 (i.e., reports were cross-checked with dealer records and confirmed with phone calls to permitted fishermen). Trip level reporting was requested during 2001–2007, but catch report submittals occurred annually with variable results in the quality of trip level documentation. Since 2008, trip level reporting has been mandatory with monthly reports required. Relatively few fishermen reported landings in the Massachusetts commercial eel fishery during 2001–2009. The number of permit holders has been near 100 in most years, but the number reporting catches has been typically 10–15. The minor commercial landings of this period appear to be historical lows for Massachusetts.

5.1.1.2.4 Rhode Island

Dealer Reporting

American eel landings estimates from 1950 to 2010 for Rhode Island were obtained from ACCSP.

Harvester Reporting

The Rhode Island Division of Fish and Wildlife began trip level reporting of commercial catches of American eel in 2007. The fishery has higher catches during spring and fall. The Rhode Island eel pot fishery is similar in approach and landings to those found in Massachusetts and New Hampshire. The relatively low number of participants and total landings reflect a small-scale, part-time, seasonal fishery. However, the data have outstanding questions on the entry of conger eel catches as American eel and catches reported for the lobster pot gear code.

5.1.1.2.5 Connecticut

Dealer Reporting

American eel landings estimates from 1950 to 2010 for Connecticut were obtained from ACCSP.

5.1.1.2.6 New York

Dealer Reporting

American eel landings estimates from 1950 to 2010 for most of New York waters were obtained from ACCSP.

Harvester Reporting

New York landings from Lake Ontario were obtained from NMFS and the Hudson River Fisheries Unit of the New York Department of Environmental Conservation and added to landings from other regions of New York. Prior to 1976, eel caught in this fishery primarily were exported to Canada and likely on to Europe. In September 1976, sale of eel from Lake Ontario was banned due to concerns with contaminants. In 1978, the Lake Ontario fishery was reopened to foreign markets only. In 1982, the fishery was closed again due to concerns with contaminants and has remained closed since that time. Landings recorded by NMFS from 1983–1996 are either illegal or misreported. Monthly fisherman reporting has been mandatory since the 1970s; however, underreporting is suspected to be as high as 50% or more (Steve LaPan, NYS DEC, pers. comm.).

5.1.1.2.7 New Jersey

Dealer Reporting

Commercial harvest records for American eel are available from the NMFS beginning in 1950.

Harvester Reporting

New Jersey implemented mandatory harvester reporting in 2007 for all licensed eel pot fishermen. It is likely that some landings from less important gears are missed, but data from NMFS indicate that eel pots account for greater than 98% of total harvest. Harvester reported landings estimates have concurred very well with data collected by NMFS. Pots were the primary capture gear throughout the time series, accounting for at least 63% of annual landings in all years except 1955 (Figure 5.9). Pots have accounted for 98% of landings in nearly all years since 1977. Between 1950 and 1975, several other gears contributed significantly to total landings. Spears accounted for between 5 and 15% of annual landings between 1950 and 1963, but dropped down to generally less than 1% thereafter and have been absent from catch records since 1977. Weir landings made up between 10 and 20% of landings from 1959 to 1974 before tapering off to no landings in all but two years since 1977. Fyke nets and pound nets also posted occasional high landings, each accounting for 15% or more of annual landings in three years during the 1950s. All other gears combined have generally contributed less than 1% of total landings except for a few notable occasions. In 1997, more than 10% of annual harvest was collected with dip nets during the height of the glass eel fishery. In 2006, a total of 26,500 pounds (16.7%) were reported as hand-line harvest, although this could be a coding error since the next largest harvest by hand-line was 270 pounds.

Biological Sampling

New Jersey began collecting biological samples from the commercial fishery in 2006, including lengths, weights, and hard parts. Ageing work has been delayed due to staff and funding shortages, but length and weight data have been analyzed to characterize the fishery and the resource. Average length of eels harvested has ranged from 416 mm in 2008 to 500 mm in 2006, with a range across all years of 100 mm to 1037 mm (Table 5.1; Figure 5.10). Weight of eels has ranged from 2 g to 1970 g, with annual averages between 170 g and 270 g. The largest averages for both length and weight were observed in 2006 when a single fyke net fisherman provided a number of large, presumably silver eels. The remaining samples were obtained from eel pot fishermen.

Length-weight parameters were fitted using SAS software (Table 5.2). Predicted weight at length shows slight variation between years at sizes larger than 600 mm (Figure 5.11), although this might be due to small samples sizes of large fish. Regional analysis indicates that eels from the Hudson Bay region are smaller than fish from Delaware Bay or New Jersey coastal regions. Eels greater than 600 mm from Delaware Bay are heavier than their counterparts from the other two regions.

5.1.1.2.8 Upper Delaware River

The Delaware River is one of the longest undammed rivers on the Atlantic coast, providing unhindered access to upstream areas in northern New Jersey, Pennsylvania and New York. Because the main stem has no barriers to fish passage, the upper reaches of the Delaware River are accessible to migratory species such as eel. As such, eels have been the target of commercial and subsistence fisheries throughout history. Eels are captured primarily in fish weirs built midstream during the summer to catch the downstream migrating silver eels in late summer and fall. Records are not available prior to 1998, but recent harvest records indicate fisheries on the Delaware River and the Neversink River, a tributary near Port Jervis, NY. Conversations with a long-time weir harvester indicated that 30 weirs or more were operated in the region historically (commercial weir fisherman, pers. comm.), but effort has declined dramatically, with only two primary harvesters remaining, one on each of the Delaware and Neversink Rivers (M.B. DeLucia, The Nature Conservancy, pers. comm.).

Several sources of data were pooled in an effort to characterize the eel weir fishery in the New York section of the Delaware River and its tributaries. Weir licenses are issued by the NY Special Permitting Unit, which requires annual reporting of the previous year's catch before a new license is issued (C. Schiralli, NYS DEC, pers. comm.). Individual harvester records were made available for the years 1998 to 2007. Records prior to 1998 were not available. In addition, landings data are recorded in a database maintained by the NY Hudson River Fisheries Unit (K. McShane, NYS DEC, pers. comm.). Again, data were only available since 1998. Third, The Nature Conservancy has maintained a database of annual harvest on the Neversink and Delaware Rivers back to 1990 (M.B. DeLucia, The Nature Conservancy, pers comm.). Finally, a 30 year history of harvest (1977 to 2007) was made available by an NPS employee who has been receiving harvest estimates from a single fisherman on the Delaware River (D. Hamilton, National Park Service, pers. comm.).

Unfortunately, there are considerable inconsistencies in the values reported in the different datasets. Attempts were made to match up the reports from the SPU and HRFU because harvester level data were available from both datasets. Where inconsistencies were found, information garnered from discussions with SPU and HRFU staff was used to determine the more appropriate value. Data from TNC matched well with data from the SPU for the years 1998 forward. Since data were not available prior to 1998 from either SPU or HRFU, data reported by TNC were used as the sole source of landings. However, it appears that the TNC dataset is incomplete since the landings estimates for the single harvester obtained from NPS often exceeds the harvest estimates of multiple harvesters obtained from TNC for years prior to 1997. Further, since the TNC dataset includes harvests estimates from the harvester reporting to NPS, this provides an indication of under reporting to the data collection entity in these years.

Harvest estimates were variously provided in pounds and numbers. Numbers were converted to pounds using a conversion of one eel = 0.875 pounds (D. Hamilton, National Park Service, pers. comm.). This matches well with information provided by a long time commercial harvester who indicated eels often weigh about 5/8 pounds early in the season, increasing to approximately one pound by the end of the season (commercial weir fisherman, pers. comm.) Pounds harvested by individual harvesters were summed across all harvesters on both the Delaware and Neversink Rivers to produce annual harvest estimates for the upper Delaware system. Because of a small number of active licenses in some years, landings estimates were standardized to maintain confidentiality. Both multi-harvester and single-harvester estimates were standardized using *Z*-scores (Zar 1998).

The contributions to overall harvest by the single fisherman reporting to NPS are evident in the correlation between the single and multiple harvester trends from 1998 to present. Since 1998 (NYS DEC data), harvest has varied greatly with little observable trend. Landings are often greatly influenced by weather and timing (commercial weir fisherman, pers. comm.). For example, years of high water during the summer can delay building of weirs, resulting in a shortened (or perhaps entirely lost) season. During several years in the early 2000s, hurricanes and tropical storms produced heavy rainfall and flooding that made the weirs unfishable. Overall effort may also contribute to total landings, but records of number of licenses sold are not available for analysis.

Prior to 1997, the trend in harvest depends greatly on the number and avidity of the harvesters, as well as eel abundance and market conditions. However, some insight into the fishery, and possibly population, might be gained using data from the single harvester. Landings in recent years are similar to landings more than three decades ago, but there appear to have been several minor cycles within that time period. Anecdotal accounts of population size structure indicate that fish as large as 5 pounds were once common, but now two pounds is considered large. In addition, “shoestring” eels (possibly males?) once made up 25–50% of the eels captured in the weirs, but recently the proportion has declined to around 5% (commercial weir fisherman, pers. comm.).

5.1.1.2.9 Delaware

Dealer Reporting

American eel landings estimates prior to 1999 were obtained from the NMFS dealer reporting system (as provided by ACCSP). Total landings estimates for 1996 were provided by Delaware Division of Fish and Wildlife. The 1997 NMFS estimate of landings was far lower than expected and was rejected with Delaware and NMFS agreeing that there would be no official eel landings for Delaware in that year.

Harvester Reporting

Delaware mandated catch and effort reporting from the American eel fishery in 1999. NMFS estimates were used when the monthly breakdown concurred with Delaware records; otherwise, NMFS estimates were replaced with state data. From 2000 through 2008, Delaware supplemental landings (reported after the March upload to NMFS) were added to NMFS totals. Estimates from 2009 were provided by Delaware DFW.

Biological Sampling

American eels were sampled from the commercial eel pot fishery in Delaware several times annually during 2000 through 2010. The American eels were taken during onboard sampling; typically the contents of one to three randomly-chosen eel pots were kept for analysis. Sampled American eels were measured to the nearest mm, weighed to the nearest 0.1 gram, and, since 2006, dissected for detection of the swim bladder parasite *A. crassus.* Otoliths were extracted from most of the sampled eels and used for ageing. All of the approximately 3,800 American eels sampled have been measured and weighed, and almost 90% of those sampled were aged. The combined American eel samples since 2000 had a mean length of 378 mm, a mean weight of 122.3 g, and a mean age of 4.

5.1.1.2.10 Maryland and Delaware Coastal Bays

Harvester Reporting

Since mandatory catch and effort reporting was initiated in 1992, American eel harvest from Maryland coastal bays averaged 9,954 lbs accounting for less than 4% of Maryland's total harvest. Harvest in Maryland coastal bays was sporadic and at relatively low levels. Average landings in Delaware coastal bays (18,923 lbs) were nearly twice that of Maryland's throughout their respective time series.

Biological Sampling

A total of 77 commercial biosamples were collected from Delaware coastal bay eel potters from 2000–2008. Length and weight were collected on all eels and 74 were aged. Approximately 700 biosamples were collected from Maryland coastal bay eelers in 2000 and 2001. Length and weight were collected on all eels and 179 eels were aged. All Maryland eels collected were unculled and randomly sampled. Mean length (mm) and age from Delaware and Maryland coastal bay eel were 531 mm and 4.5 years and 471 mm and 3.4 years, respectively.

5.1.1.2.11 Maryland

Harvester Reporting

Commercial eelers in Maryland were first required to be licensed and report harvest of American eel in 1981. Prior eel landings were obtained from the NMFS dealer reporting system. Mandatory monthly catch and effort began in 1990, but was not fully adopted until 1992. Trip level catch and effort reporting was adopted in 2004.

American eel landings estimates for Maryland were obtained from ACCSP with the following caveats. Maryland provided corrected landings estimates in the years 1994, 2004, and 2006. Duplicate records found for the year 1997 were removed. The eel pot fishery in Maryland accounts for over 98% of total eel harvest.

Biological Sampling

Since 1997, American eels have been sub sampled from the commercial eel pot fishery in Maryland's portion of the Chesapeake Bay and tributaries and coastal bays. Twelve tributaries have been sampled over the time period. A minimum of 2 selected tributaries are sampled each year. Approximately 100 pounds of yellow eels purchased in 2 separate batches (usually 2–3 weeks between purchases) from commercial eel potters unculled from live boxes. The live boxes contain catches over multiple days. Since a standard weight is purchased from commercial eelers, depending on size 400–1000 commercially harvested yellow eels are sub sampled per selected river per year. Measurements to the nearest mm and weight to the nearest gram were taken from each eel with approximately 100 of those eels subsampled for aging. Since 2004, approximately 150 eels per year were noted for prevalence of swim bladder parasite *A. crassus*. Since 2006, approximately 90 eels per year were subsampled for sex determination. The mean length of an American eel sampled from the eel pot fishery since 1997 has been 358 mm (N=15,600) with the average freshwater age of 4.0 years (n = 2,790). The prevalence rate of 692 subsampled eels for *A.crassus* was 43%. Since 2006, females have outnumbered males by an approximate 2:1 ratio.

5.1.1.2.12 Potomac River

The Potomac River has jurisdiction for the main stem of the Potomac below the Woodrow Wilson Bridge. The eel pot fishery accounts for over 99% of total eel harvest reported to PRFC.

The Potomac River Fisheries Commission provided records of their landings which were later assigned to either Maryland or Virginia as appropriate. Mandatory harvest reporting for Potomac River Fisheries Commission (PRFC) began in 1964. Monthly catch and effort was required of commercial eelers in 1988. In 1990, the catch and effort was changed to mandatory weekly reporting and then mandatory daily reporting began in 1999.

5.1.1.2.13 Virginia

Dealer Reporting

American eel landings estimates for VirginiaA were obtained from the NMFS dealer reporting system (as provided by ACCSP) with the understanding that supplemental landings (corrections) were added annually by port agents on the arbitrary date of December 31 in 1996. The Virginia Marine Resources Commission (VMRC) also likely sent NMFS supplemental landings updates in 1980 and 1988; therefore the NMFS landings estimates were used in place of VMRC records.

In all other years, NMFS/ACCSP and VMRC records aligned well. A portion of the Potomac River Fisheries Commission landings were assigned to Virginia as appropriate.

Harvester Reporting

The Virginia Marine Resources Commission (VMRC) began collecting voluntary reports of commercial landings from seafood buyers in 1973. A mandatory harvester reporting system was initiated in 1993 and collects trip-level data on harvest and landings in Virginia. Data collected from the mandatory reporting program are considered reliable starting in 1994, the year after the pilot year of program.

Biological Sampling

The VMRC Biological Sampling Program was initiated in 1989 to collect fishery-dependent biological information to support assessment and management activity within the state and coast-wide. There are currently twenty-one species targeted for sampling in the program, although other species, such as American eel, are sampled based on availability and staff time. Limited numbers of American eels have been available to the Biological Sampling Program over the years (Table 5.3). A total of 818 lengths and 787 weights have been collected from American eels to date. No American eels were available for sampling in 2009 or 2010. The majority of American eels sampled have been collected from eel pots and pound nets. American eels collected from eel pots ranged from 244 mm to 768 mm in length (Figure 5.12). The average length of American eels sampled from eel pots (pooled over years) was 428 mm.

5.1.1.2.14 North Carolina

Dealer Reporting

American eel landings estimates for North Carolina were obtained from the NMFS dealer reporting system (as provided by ACCSP) prior to 1972.

Harvester Reporting

The NC Department of Environment and Natural Resources provided landings estimates for 1972–2009. A trip ticket system began in 1994 and a commercial logbook system began in 2007. However, logbooks were found to be consistently lower than trip tickets and are believed to be limited by underreporting. Reconciliation and verification will be required before this data can be used in future assessments.

Biological Sampling

In 1996, length information data on eel caught in commercial pots was collected during a study commissioned by NCDENR (Hutchinson, 1997). This study was designed to determine the loss of legal-sized eel from pots with and without escape panels. Weekly sampling was conducted between May and October 1996 in the Pamlico River. A total of 176 trips were made to sample 4,057 pots fished. Over 6,500 eel across both types of pots were individually measured. Typically, eels between 280 and 360 mm were retained (Figure 5.13). Weight of individual eel was not recorded.

5.1.1.2.15 South Carolina

Dealer Reporting

American eel landings estimates for South Carolina were obtained from the NMFS dealer reporting system (as provided by ACCSP) for years prior to 2008. The South Carolina Department of Natural Resources provided landings for 2008–2010. A commercial glass eel fishery operates in South Carolina, but landings since 2000 have been minimal and are thus confidential. However, in 2011 anecdotal reports were received of glass eel prices exceeding $1,000/lb and renewed fishing activity in South Carolina. NMFS records do not indicate life stage, so a breakdown of glass versus yellow eel landings was not available.

5.1.1.2.16 Georgia

Dealer Reporting

American eel landings estimates for Georgia were obtained from the NMFS dealer reporting system (as provided by ACCSP) for years prior to 1989. The Georgia Department of Natural Resources provided landings between 1989 and 2003. No commercial landings have been reported to the state of Georgia since 2003, but anecdotal reports suggest harvesting may be ongoing.

5.1.1.2.17 Florida

Dealer Reporting

American eel landings estimates for Florida were obtained from the NMFS dealer reporting system (as provided by ACCSP) for years prior to 1980. From 1980 forward, Florida Fish and Wildlife Research Institute (FWRI) data were used in the assessment. Prior to 2003, FL sent annual landings totals to NMFS.

Harvester Reporting

Monthly harvester reporting began in 2003 and a trip ticket system was instituted in July 2006. No species specific “American eel” code was used in Florida data collection until 2006; therefore, “eel” landings prior to 2006 may include other eel landings (e.g., conger eel). Recent NMFS marine data (not used in this assessment) also likely includes other marine eel. Note that eel landings are typically concentrated in a small area; prior to 1997, most commercial eel landings were reported in the St. Johns River, including the areas of Lake Crescent, Lake George, Lake Jesup, Lake Monroe, and the main stem of the St. Johns River. Data recorded include the water body of harvest, the number of pots set, and the weight of American eels harvested. This fishery is primarily a yellow eel fishery with very few reports of silver eels.

Biological Sampling

Biological samples were obtained by purchase from eel harvesters from 2002–2006, but no biological samples exist beyond 2006. Data collected includes total length, weight, and sex. Sex data are not reliable because only a portion of the fish was examined histologically and of those that were examined histologically, nearly all were female. Thus, summaries of the biological data presented here combine both sexes (Figure 5.14). A length-weight relationship was estimated from eels sampled in the St. Johns River system, FL (2002–2010; Figure 5.15).

5.1.2 Recreational Fisheries

5.1.2.1 Data Collection

The primary source of recreational fishery statistics for the Atlantic coast is the Marine Recreational Fishery Statistics Survey (MRFSS) program. The MRFSS collects data on marine recreational fishing to estimate statistics characterizing the catch and effort in marine recreational fisheries. Recreational fisheries statistics for American eels were obtained from the MRFSS online data query (NMFS, pers. comm.). Information on sample sizes was retrieved from the MRFSS raw intercept files.

Survey Methods

Data collection consists primarily of two complementary surveys: a telephone household survey and an angler-intercept survey. In 2005, the MRFFS began at-sea sampling of headboat (party boat) fishing trips. Data derived from the telephone survey are used to estimate the number of recreational fishing trips (effort) for each stratum (see Sampling Intensity, below). The intercept and at-sea headboat data are used to estimate catch-per-trip for each species encountered. The estimated number of angler trips is multiplied by the estimated average catch-per-trip to calculate an estimate of total catch for each survey stratum. A more detailed description of the MRFSS sampling methods is provided in the MRFSS User's Manual (ASMFC 1994).

The MRFSS estimates are divided into three catch types depending on availability for sampling. The MRFSS classifies those fish brought to the dock in whole form, which are identified and measured by trained interviewers, as landings (Type A). Fish that are not in whole form (bait, filleted, released dead) when brought to the dock are classified as discards (Type B1), which are reported to the interviewer but identified by the angler. Fish that are released dead during at-sea headboat sampling are also classified as Type B1 discards. The sum of Types A and B1 provides an estimate of total harvest for the recreational fishery. Anglers also report fish that are released live (Type B2) to the interviewer. Those fish that are released alive during the at-sea headboat survey are also considered Type B2 catch. Total recreational catch is considered the sum of the three catch types (A+B1+B2). The numbers of American eels of each catch type that were sampled by the MRFSS are presented in Table 5.4. Numbers of American eel samples reported by the MRFSS angler-intercept survey and at-sea headboat survey, by catch type, 1981–2010..

Sampling Intensity

The number of telephone interviews conducted during each wave varies based on the amount of fishing activity expected for the season (NMFS, pers. comm.). Telephone sampling effort is allocated among coastal counties in proportion to household populations. Specifically, the allocation is based on the ratio of the square root of the population within each county to the sum of the square roots of all county populations within the state.

Intercept sampling is random and stratified by year, state, wave (two-month sampling period), and mode (type of fishing). A minimum of 30 intercepts are performed per stratum, though samples are allocated beyond the minimum in proportion to the average fishing pressure of the previous three years.

Biological Sampling

The MRFSS interviewers routinely sample fish of Type A catch that are encountered during the angler-intercept survey. Fish discarded during the at-sea headboat survey are also sampled—the headboat survey is the only source of biological data characterizing discarded catch that are collected by the MRFSS. The sampled fish are weighed to the nearest five one-hundredth (0.05) of a kilogram or the nearest tenth (0.10) of a kilogram (depending on scale used) and measured to the nearest millimeter for the length type appropriate to the morphology of the fish. The numbers of American eel biological samples taken by the MRFSS are summarized in Tables 5.4 and 5.5.

Biases

Few anglers fishing in the area covered by the MRFSS target or catch American eels. The MRFSS does not cover inland (freshwater) areas, where the majority of recreational fishing for eels is assumed to occur. In addition, the MRFSS intercept component does not capture information from recreational fishermen who use gears other than hook and line, and therefore does not capture the personal-use sector that may use commercial gear types on a limited basis to harvest eels for personal consumption.

The MRFSS estimates are based on a stratified random sampling design and so are designed to be unbiased. There have been a few instances when the random telephone survey was found to be unrepresentative and an average estimate of trips was substituted. Most recently, the 2002 telephone survey data were discarded for waves 2 and 3 and effort estimates were instead based on a three-year average (1999–2001) for those waves. The MRFSS advises that the weight estimates are minimum values and so may not accurately reflect the actual total weight of fish harvested. There have also been differences in sampling coverage over the duration of the survey. Other caveats associated with these data are discussed at the following web site:
http://www.st.nmfs.gov/st1/recreational/queries/caveat.html.

Recent concerns regarding the timeliness and accuracy of the MRFSS program prompted the NMFS to request a thorough review of the methods used to collect and analyze marine recreational fisheries data. The National Research Council (NRC) convened a committee to perform the review, which was completed in 2006 (NRC 2006). The review resulted in a number of recommendations for improving the effectiveness and utility of sampling and estimation methods. In response to the recommendations, the NMFS initiated the Marine Recreational Information Program (MRIP), a program designed to improve the quality and accuracy of marine recreational fisheries data. The MRIP program is being phased in gradually and will eventually replace the MRFSS. The objective of the MRIP program is to provide timely and accurate estimates of marine recreational fisheries catch and effort and provide reliable data to support stock assessment and fisheries management decisions. The program will be reviewed periodically and undergo modifications as needed to address changing management needs.

5.1.2.2 Development of Estimates

Estimates of harvest in terms of numbers are available for all three catch types (Type A, B1, and B2). Weight estimates are only available for recreational harvest (Type A+B1). Details describing how the MRFSS uses data collected from the telephone interviews and angler intercept survey to develop catch and effort estimates can be found in the MRFSS User's Manual (ASMFC 1994). Finalized recreational fishery statistics were available for 1981 through 2010.

The MRFSS is in the process of applying a new methodology for estimating recreational catch (NMFS, pers. comm.). The new estimation method addresses one of the major concerns identified by the NRC review—there is a mismatch between the MRFSS estimation method and the program's sampling design. The MRFSS plans to apply the new methodology beginning with the 2010 catch estimates. The new method will also be applied to recalculate catch estimates for 2003 through 2009.

Annual length-frequency distributions of American eels sampled by the MRFSS were calculated using the Type A biological sampling data. These data were available for 1981 through 2010.

5.1.2.3 Estimates

Recreational harvest (Type A + B1) of American eels along the Atlantic coast ranged from 3,485 to 161,077 eels per year during 1981 through 2009. In terms of weight, recreational eel harvest ranged from 353 to 157,155 pounds per year during the same time period (Table 5.6). American eel recreational harvest demonstrated an overall decline over the available time series. The number of American eels released alive by recreational anglers ranged from a low of 21,464 eels in 1997 to a high of 126,330 eels in 2003. Live releases of American eels generally declined from the late 1980s through the late 1990s to early 2000s. Numbers of live releases have since increased to levels similar to those observed in the early to mid-1980s.

The precision of the estimated harvest numbers, measured as proportional standard error (PSE), exceeded 20% in all years and exceeded 30% in nineteen of the twenty-nine years for which estimates were available (Table 5.6). The precision of harvest weight estimates exceeded 20% in most years. In some years, the sampling data were insufficient to allow calculation of precision of harvest weight. Estimates of the number of American eels released alive were also associated with low precision, with PSE values exceeding 20% in the majority of years.

The low precision associated with the recreational fishery statistics is due to the limited numbers of American eels that have been encountered during surveys of recreational anglers along the Atlantic Coast (Tables 5.4 and 5.5). These limited numbers are partly due to the design of the MRFSS survey, which does not include the areas and gears assumed to be responsible for the majority of recreational fishing for American eels (see Biases within section 5.1.2.1, this report). As such, the recreational fishery statistics for American eels provided by MRFSS should be interpreted with caution.

Note that recreational fishery statistics and associated precision for 2003 through 2009 are subject to change as the MRFSS plans to recalculate those estimates using the new estimation methodology (see section 5.1.2.2, this report).

The lengths reported for American eels sampled in the MRFSS angler-intercept survey (Type A catch) ranged from 24 mm to 1,104 mm during 1981 to 2009 (Figure 5.16). Smaller recorded lengths are likely recording errors or species misidentifications.

5.2 Fishery-Independent Surveys and Studies

This section summarizes survey data and studies used to inform the stock assessment. Very few fishery-independent surveys target American eels (with the exception of the state-mandated young-of-year surveys and a few surveys in Maryland). All fishery-independent surveys used in this assessment were evaluated using a standard set of criteria (see Appendix 2) that resulted in data-based decisions to inform the analytical framework (primary assumptions regarding the error structure) for each survey independently. Application of these criteria resulted in nearly all surveys being standardized (unless otherwise noted) using a generalized linear model to account for changes in catchability of eel.

5.2.1 Annual Young-of-Year Abundance Surveys

5.2.1.1 Coast-wide Mandatory State Surveys

5.2.1.1.1 Data Collection

Survey Methods

The FMP for American eel requires all states and jurisdictions, except those exempted by the Management Board, to conduct an annual young-of-year (YOY) abundance survey (ASMFC 2000a). The glass eel (YOY) life stage provides the most unique opportunity to assess the annual recruitment of each year's cohort since YOY result from the previous winter's spawning activity and represent individuals of the same age.

The FMP for American eel provides general guidance to the various states and jurisdictions for setting up the mandated YOY survey (ASMFC 2000a). The ASMFC American Eel Technical Committee updated the approved standard protocol for carrying out the mandatory YOY survey shortly after the approval of the FMP (ASMFC 2000b). A number of gear types are permitted, depending on the habitat and geography of the sampling locale. The timing and placement of the young-of-year sampling gear should coincide with periods of peak onshore migration of YOY within the survey region. Sampling locations should be selected based on historical observations of YOY American eel and attempt to provide as wide a geographic distribution as possible.

States are required to weigh and enumerate the catch of eel and to report the catch-per-unit-effort for each sampling day. Standard statistical techniques (sub-sampling) can be applied in instances where the catch of YOY is too large (i.e., several hundred individuals or more) to warrant a complete census. Data collected during the YOY survey are submitted annually as part of each state's annual compliance report for American eel.

A list of the currently active sites is provided in Table 5.7. A map of survey site locations can be found in Figure 5.17.

Sampling Intensity

States and jurisdictions must conduct the required YOY survey at a minimum of one location over a six-week period. The sampling gear should be set during periods of rising or flood tides occurring at nighttime hours. The gear should be inspected as often as possible within the designated sampling period.

Biological Sampling

Biological sampling is not required, but recommendations were given to provide a standardized format for collection and reporting of samples (ASMFC 2000b). The ASMFC American Eel Technical Committee recommends a minimum of 60 elvers be sub-sampled twice a week during the sampling period. Each individual should be measured for total length and weighed to the nearest 0.01 g. Pigmentation stage should also be noted using Haro and Krueger (1988) as a guide in assigning stage.

Biases

In December 2006, the ASMFC held a workshop for those involved in the state annual YOY surveys. At this workshop, one of the issues discussed was timing of the survey. Participants pointed out that the onset of sampling in a given year is occasionally delayed and so the survey may miss part of the peak onshore migration. Criteria for ending sampling vary among states and years. These criteria range from formal stopping rules to the need for staff to attend to other responsibilities.

Differences related to the location of the survey sites and placement of the sampling gear may affect the comparability of data among sites. The YOY survey sites have varying distances to the ocean, river mouth, and tidal influence. The salinity of sampling locations ranges from marine to freshwater. Some sites are located near obstructions, such as dams, and the distance to obstructions, if present, is variable. Other differences include differences in the placement of traps relative to attraction flow and differences in the percentage of the channel width fished.

5.2.1.1.2 Development of Estimates

Annual indices of relative YOY abundance were calculated for sites that have been sampled for at least ten years as of 2010 (Table 5.7). Indices were calculated using the protocol outlined in Appendix 2.

The availability of potential covariates varied among sites and years. Though the ASMFC YOY survey protocol requires that states record effort, water temperature, water level, and discharge (ASMFC 2000b), effort and water temperature were the only auxiliary variables consistently available for all sites. Additional variables were considered as covariates in the GLM analysis if the data were available in all years for a particular site.

Spearman's rank correlation coefficient, ρ, and the associated probability were calculated for all pairs of YOY indices to assess the degree of association among the indices. Indices were considered significantly correlated at $\alpha = 0.10$.

5.2.1.1.3 Estimates

Annual recruitment indices were computed for sixteen sites sampled as part of the ASMFC-mandate (Table 5.8). Water temperature was found to be a significant covariate affecting catchability for most survey sites. Note that effort was not determined to be a significant covariate in the models for any of the survey sites. Most of the survey data were best characterized using a model that had negative binomial errors. For three sites, a stable generalized linear model could not be developed, so arithmetic mean catch per tow was used as an index of abundance.

Trends in the YOY indices were variable within and among survey sites (Figures 5.18–5.33). The degree of correlation between survey sites ranged from significant and negative to significant and positive (Table 5.9).

There were no strong correlations among the YOY indices in the Gulf of Maine region (Table 5.9). In the Southern New England region, only two YOY indices were available—Gilbert Stuart Dam (Rhode Island) and Carman's River (New York), and they were significantly and positively correlated ($\rho = 0.591$, $P = 0.0556$; Table 5.9). Both these indices show an initial decline from 2000 to 2001, a time series peak in 2002, and relatively low levels with limited variability from 2003 to the end of the time series (Figures 5.21—5.22).

In the Delaware Bay and Mid-Atlantic Coastal Bays region, the Patcong Creek (New Jersey; Figure 5.23) YOY index was negatively correlated with both the Millsboro Dam (Delaware; Figure 5.24) and Turville Creek (Maryland; Figure 5.25) YOY indices (Table 5.9). The negative correlation between the Patcong Creek (New Jersey) and Turville Creek (Maryland) indices was statistically significant (ρ = -0.636, P = 0.0353).

Correlations among the YOY indices in the Chesapeake Bay region were generally weak (Table 5.9). One exception was the correlation between the Clark's Millpond (PRFC; Figure 5.26) and Gardy's Millpond (PRFC; Figure 5.27) indices, which was significant and negative (ρ = -0.664, P = 0.0260; Table 5.9).

One significant correlation was detected among the YOY indices in the South Atlantic region. The YOY indices for Goose Creek (South Carolina; Figure 5.31) and Guana River Dam (Florida; Figure 5.33) were significantly and positively correlated (ρ = 0.552, P = 0.0984; Table 5.9). Both of these indices show a peak in recruitment in 2005.

5.2.1.2 Little Egg Inlet Ichthyoplankton Survey

5.2.1.2.1 Data Collection

Survey Methods

The Rutgers University Marine Field Stations (RUMFS) has been conducting ichthyoplankton sampling near the field station beginning in the late 1980s, and with established protocols since 1991 (Hagan et al. 2003). Data from this sampling program include timing and intensity of glass eel ingress to estuarine habitat (Sullivan et al. 2006, 2009). Raw survey data were provided to the ASMFC for use in this stock assessment.

Survey protocol is described in detail by Hagan et al. (2003); Sullivan et al. (2006) characterize the sampling area. Briefly, sampling is conducted in Little Sheepshead Creek, a small "thoroughfare" across a peninsula within the Great Bay-Little Egg Harbor estuarine system along the southern New Jersey coast (Figure 5.34). Maximum depth in the creek is approximately 3 meters, with a tidal range of approximately 1 meter. A 1-meter plankton net with 1 mm bar mesh is used to sample larval and juvenile fishes recruiting to the estuary. The net is deployed weekly, throughout the year, during night time flood tide from a bridge over Little Sheepshead Creek. During initial years of the survey, a number of sampling strategies were implemented, but methods have been standardized since August 1991. Since then, three 30-minute sets are made in succession once per week at mid-water. Catch from each set is preserved for later identification and enumeration in the lab, and the net is re-set. Flow rate for each set is measured with a flow meter attached to the net. Environmental parameters include surface water temperature and salinity.

5.2.1.2.2 Development of Estimates

Over the entire time series, eels were observed in all months except September; however, between June and December, less than 20% of all tows were positive for eels. Data were therefore subset to include only data from the months January to May. In addition, data from 1989 to 1991 were removed due to inconsistent sampling methodology. The resulting dataset was evaluated relative to the standardized criteria, and a generalized linear model was developed consistent with those results. The appropriate error structure was applied to the full model which included year, month, tidal flow rate, mean river discharge (USGS station #01409400 Mullica River near Batsto, NJ), and surface salinity. Non-significant factors were removed to produce the final model. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable.

5.2.1.2.3 Estimates

A negative binomial error structure provided the best fit to the raw data, and this was applied to the full model. All factors considered in the full model were found to be significant (Table 5.10). The overdispersion factor, phi, was estimated at 1.05, indicating the negative binomial error structure was appropriate. The predicted Little Egg Inlet index varied without trend from 1992 to 2008, with relative peaks in 1994–1995 and 2007–2008 (Figure 5.35). The long term average for 1992 to 2008 was 1.52 eels per tow, with a range of 0.81 to 2.31. Since 2008, the index has dropped sharply from more than 1.8 eels per tow in 2008 to just 0.33 eels per tow in 2010.

5.2.1.3 Beaufort Inlet Ichthyoplankton Survey

5.2.1.3.1 Data Collection

Survey Methods

The NOAA National Ocean Service laboratory in Beaufort, NC has been conducting bridge-based plankton sampling near Beaufort, NC since 1985. Ingressing glass eels are often captured in the survey, providing an index of glass eel recruitment to the estuary. The sampling location and methodology are described in Sullivan et al. (2006). Beaufort Inlet is a principal connection between the back bays of North Carolina's Outer Banks and the Atlantic Ocean in the region of Beaufort, NC (Figure5.34). The major systems near Beaufort Inlet include Bogue Sound, Core Sound, Newport River, and North River. Tidal range within the estuary is approximately 1 meter. Approximately 10% of the water entering Beaufort Inlet passes through the Radio Island—Pivers Island channel where sampling occurs.

Sampling is conducted using a 2 m2 rectangular plankton net with 1 mm mesh. A flow meter is attached to the net to measure flow rates. Four replicate sets have been made at the surface (0–1 m) during night time flood tides at weekly (1985 to 2001) or bi-weekly (2001 to present) intervals. Sampling is conducted from November to April in every year, with occasional sampling in May and October. Tow duration was approximately 5 minutes per tow during 1985 to 1997; since 1998 tows have been standardized to volume sampled (approximately 100 m3) rather than tow duration.

5.2.1.3.2 Development of Estimates

The survey has occurred every year since the survey began in December 1985, but data were only available through the 2003 sampling season due to a backlog in processing the samples. Over the entire time series, eels were observed in all months sampled except October. In addition, the proportion of positive tows in May and November were considered too low to include these months in the analysis (less than 5% positive for eels). Data from December to April were included in the analysis, with data from December being lagged to the following calendar year. The resulting dataset was evaluated relative to the standardized criteria, and a generalized linear model was developed consistent with those results. The appropriate error structure was applied to the full model which included year, month, tidal flow rate, and mean river discharge (USGS station #02089500 NEUSE RIVER AT KINSTON, NC). Non-significant factors were removed to produce the final model. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable.

5.2.1.3.3 Estimates

A negative binomial error structure provided the best fit to the raw data, and this was applied to the full model. All factors considered in the full model except tidal flow were found to be significant (Table 5.10). The overdispersion factor, phi, was estimated at 1.14, indicating the negative binomial error structure was appropriate. The predicted Beaufort Inlet index varied without trend from 1987 to 2003, ranging from a low of 0.17 eels per tow in 1999 to a high of 1.54 in 1994 and 1995 (Figure 5.36).

5.2.1.4 HRE Monitoring Program

One additional YOY index was generated from the HRE Monitoring Program in the Hudson River. For more information, see section 5.2.3.4.

5.2.2 Southern New England

5.2.2.1 Rainbow Smelt Fyke Net Survey

5.2.2.1.1 Data Collection

The Massachusetts Division of Marine Fisheries began monitoring anadromous rainbow smelt (*Osmerus mordax*) populations in 2004 using fyke nets at four coastal rivers and four additional rivers have been added since 2005. The spring fyke net monitoring occurs when resident yellow eels become active and are susceptible to capture as non-target bycatch. The eel bycatch data were evaluated because the eel assessment presently has no fisheries-independent indices of abundance on yellow eel abundance for New England states or the Gulf of Maine region.

Survey Methods

The fyke nets are set at mid-channel three nights a week from early March to the third week of May. The fyke net opening is a 4’ x 4’ box frame with 4’ x 4’ wings on both sides and the net mesh is ¼ inch delta. Diadromous fish are counted, measured and released. Water chemistry is measured at each site and discharge is available for most sites.

Biases

Catch efficiency for American eel in fyke nets is unknown. Some stations have low and intermittent eel catches.

5.2.2.1.2 Development of Estimates

Eel bycatch data for 2004–2010 were evaluated for potential utility as catch-per-unit-effort and size composition indices. Mean catch per haul with 95% CI was calculated annually for April and May hauls at each station.

5.2.2.1.3 Estimates

The total catch of eels at the four original stations has ranged from about 100–200 per year. Eel catches peak in May and few eels are seen in March or before water temperatures reach 10°C. The Fore River station (Boston Harbor region) had the highest eel bycatch in most years and peaked in 2010 with 121 eels during 24 hauls (Figure 5.37). Other stations have documented similar size composition and seasonality as the Fore River; however, with lower catch rates and in some cases relatively few occurrences (Figure 5.37). The eel length range for the Fore River in 2010 was 20–90 cm (Figure 5.37) which approximates the length range for all stations during

2004–2010. Overall, the time series is too brief to contribute to the present eel assessment. The smelt fyke net project is an ongoing, annual monitoring series. Therefore, eel bycatch data will be available for consideration in future assessments.

5.2.2.2 CTDEP Electrofishing Survey

5.2.2.2.1 Data Collection

Survey Methods

Connecticut DEP began sampling a 126 m-long section of the Farmill River in 2001. The sample site substrate is coarse sand and cobble. The Farmill River, a tributary of the Housatonic River with a 26 mile^2 watershed, is tidal freshwater at the sampling site in Shelton. There are no barriers to American eel migration between the sampling site and the ocean.

Sampling Intensity

The sample section is electrofished annually using the removal method.

Biological Sampling

All eels captured are anesthetized, counted, and measured to the nearest mm, then released back into the sample site.

5.2.2.2.2 Development of Estimates

A population estimate is derived using maximum weighted likelihood.

5.2.2.2.3 Estimates

Since 2001, American eel density in the Farmill River has increased (Figure 5.38).

5.2.2.3 Western Long Island Study

5.2.2.3.1 Data Collection

Survey Methods

Since 1984, New York DEC has conducted a seine survey targeting yearling striped bass in several western Long Island Sound bays.

A 200-foot (61-m) seine is deployed at fixed stations during May to October. Prior to 2000, sampling was conducted twice per month in May and June and once per month July to October. Since 2000, however, stations are sampled twice per month in all months.

5.2.2.3.2 Development of Estimates

Environmental data are collected for this survey, but data were not provided until late in the assessment process. As a result, two indices were developed for this survey. An index based only on sample design variables (including year, month, and system) was developed before environmental data were available. The results from this index were used in the ARIMA, Mann-Kendall, power analysis and other trend based methods (See Section 6 for descriptions of these methods). When environmental data were provided, a second index was developed to include these additional predictor variables, but these results were not incorporated into other methods.

The datasets were subset to the months May through August, since other months had low occurrence of eels over the time series. In addition, station selection was not always consistent, so the 41 stations sampled were subset to stations (n = 9) that had been sampled at least 123 times over the years (max = 161; all other stations had fewer than 100 observations). A number of environmental parameters have been collected over time, but only two (water temperature and salinity) were retained. Others were dropped due to their unlikely influence on eel catch (e.g., air temperature, wind speed and direction) or not being collected in all years (e.g., dissolved oxygen was not collected until 1987).

The resulting datasets were evaluated relative to the standardized criteria, and a generalized linear model was developed consistent with those results. The appropriate error structure was applied to the full model which included year, month, and system for the dataset without environmental parameters, and year, month, system, water temperature, and salinity for the dataset that included environmental data. Non-significant factors were removed to produce the respective final models.

5.2.2.3.3 Estimates

A negative binomial error structure provided the best fit to both datasets, and this was applied to the full models. All factors considered in the full model that did not include environmental variables were found to be significant (year, month, system; Table 5.10). For the model that included environmental variables, year, month, system, and water temperature were significant while salinity was not. The two indices were highly correlated, with a Pearson correlation coefficient of 0.98. Both indices peaked in 1985, but dropped by approximately 50% to 65% by 1987 (Figure 5.39). Abundance was relatively stable until 1989, but decreased sharply again in 1990. Both indices have been consistently below 5% of their respective peak value since 1990.

5.2.3 Hudson River

5.2.3.1 Morrison & Secor Studies

5.2.3.1.1 Data Collection

Study Methods

Mark recapture experiments were conducted at the six sites during summers 1997–1999. Sites were distributed through the entire length of the estuary but also were chosen to represent similar depths and bottom characteristics (shoal habitats ~2.10 m deep). Eels were tagged using liquid nitrogen brands and insertion of PIT (passive integrated transponder) tags. Branding was used to identify batches of eels according to site and day of capture, whereas PIT tags identified individual eels for growth measurements.

Sampling Intensity

Standard 100-cm long × 25-cm diameter double funnel eel pots were baited with menhaden and soaked overnight. A grid of 36 pots was deployed at each site with pots approximately 200 m apart at all sites. The pots efficiently captured eels between 300 and 750 mm long.

Biological Sampling

During June or July, 100 eels were collected for laboratory analysis from three sites (Haverstraw, Newburgh, and Athens) in 1997 and from all six sites in 1998 and 1999. Length measurements were recorded and gender was identified through gross visual inspection. Paired sagittal otoliths were removed from the eel, and left and right otoliths were randomly selected. Annular rings were counted at least four times for each section, with estimated age calculated as the mean of the counts and error in counts estimated as the difference between the minimum and maximum counts for each otolith.

5.2.3.1.2 Development of Estimates

Methods for development of life history parameters were described in Morrison and Secor (2003) and Morrison and Secor (2004).

5.2.3.1.3 Estimates

Morrison and Secor (2003) found that eel length was similar among sites (total length = 457 ± 3 mm; Figure 5.40). Eel age was substantially lower at brackish-water sites (8 ± 4 years) than at freshwater sites (17 ± 4 years) and growth was higher at brackish-water sites than freshwater sites (80 mm·year-1 and 34 mm·year-1, respectively). Almost all (97%) examined eels (1999 samples; n = 543) were female.

5.2.3.2 Machut et al. Study

Machut et al. (2007) studied anthropogenic impacts on American eel demographics in six tributaries of the Hudson River, New York. Six tributaries of the Hudson River in New York State were studied: Wynants Kill, Hannacroix Creek, Saw Kill, Black Creek, Peekskill Hollow Brook, and Minisceongo Creek.

5.2.3.2.1 Data Collection

Study Methods

For details on data collection methods, consult Machut et al. (2007). Sampling sites were isolated with 5-mm-diameter nylon-mesh block nets and electrofished with a variable-voltage backpack shocker from June to August 2003 and 2004 to collect yellow-phase American eels. Reduction sampling was performed at each site (three to five passes depending upon catch). All barriers, either natural waterfalls or man-made structures, of at least 0.5 m in height were catalogued by type and measured for height.

Sampling Intensity

Of 1,938 American eels captured, 232 eels (a size-stratified random subsample at each sampling site) were then collected. The number of eels collected for analysis at each sampling site depended on the total number of eels collected at site, ranging from 1 (if only 1 eel was collected at that site) to 16 (if numerous eels were available).

Biological Sampling

American eels were sedated with clove oil, counted, measured for total length and weight, and any obvious swellings, lesions, and ulcers were noted. Paired sagittal otoliths were removed from the eel, and left and right otoliths were then randomly selected. Age determinations were made on at least three separate occasions for each eel; if differences in estimates between readers could not be resolved, the otolith was discounted from examination (four otoliths or 2% of all the otoliths read). Male gonads were typified by spermatogonium b cells, while females were identified by presence of oocytes.

5.2.3.2.2 Development of Estimates

Methods for development of life history parameters were described in Machut et al. (2007).

5.2.3.2.3 Estimates

American eels in these six tributaries were generally smaller (Figure 5.41) than reported in the main stem of the Hudson River by Morrison and Secor (2004; Figure 5.40). Eel ranged in total length from 50 to 718 mm (mean = 185 mm; median = 152 mm). Approximately 82% of eels were caught below the first barrier and 94% were caught below the second barrier. Growth rates for tributary American eels ranged from 13 to 114 mm/year (mean = 35 mm; median = 30 mm). Whether an eel was located above or below the first tributary barrier significantly affected growth rates (df = 1, $P = 0.01$), eel growth being higher beyond the first barrier (39.3 mm/year) than below the first barrier (30.5 mm/year). The parameters of the von Bertalanffy growth equation were $L_\infty = 929.1 \pm 210.1$ mm (mean ± SE), $K = 0.0404 \pm 0.014$ mm/year, and $t_0 = -1.431 \pm 0.48$. Additional analyses and information on length, age, and growth estimates can be found in Machut et al. (2007).

5.2.3.3 NYDEC Alosine and Striped Bass Beach Seine Surveys

5.2.3.3.1 Data Collection

Survey Methods

The NYDEC has conducted two beach seine surveys annually on the Hudson River between 1980 and the present. The Alosine Survey targets juvenile alosines and the Striped Bass Survey targets, not surprisingly, juvenile striped bass. The Alosine Survey samples from Newburgh to Albany (river miles 55–140) during the months of June to November using a 100 foot center-bag beach seine. The Striped Bass Survey samples the Haverstraw and Tappan Zee Bays and farther north up the Hudson River (river miles 22–140) during the months of June to November using a 200 foot offset-bag beach seine.

Biases

These surveys were not designed to target American eel. Standardization of the survey data may provide an index of abundance if all important factors have been accounted for properly in the analysis. Also, catchability of eel has not been quantified with this gear and study design.

Biological Sampling

Lengths of individual eel are collected in the Striped Bass Survey; however, the reliability of those measurements was deemed inadequate for use in this stock assessment by NYSDEC personnel because measurements were pooled in 5 mm bins and animals above 400 mm were pooled into one bin. More detailed length information should be available for future stock assessments.

5.2.3.3.2 Development of Estimates

A standardized index of abundance based on the NYDEC Alosine Beach Seine Survey was computed following the steps given in Appendix 2. The initial candidates for covariates included year, month, river mile, latitude, longitude, and water temperature. Data for month were re-coded such that June observations were combined with July and November observations were combined with October because there were too few observations in June and November. Latitude and river mile were highly and positively correlated (Spearman's rank: $\rho = 0.997$, $P < 0.001$) so latitude was removed as a potential covariate.

A standardized index of abundance based on the NYDEC Striped Bass Beach Seine Survey was computed following the steps given in Appendix 2. The initial candidates for covariates included year, month, river mile, latitude, longitude, and water temperature. Data for month were re-coded such that June observations were combined with July and November observations were combined with October because there were too few observations in June and November. Latitude and river mile were highly and positively correlated (Spearman's rank: $\rho = 0.980$, $P < 0.001$) so latitude was removed as a potential covariate.

5.2.3.3.3 Estimates

Data from the NYDEC Alosine Beach Seine Survey were modeled assuming a negative binomial error structure (Table 5.10). Year, month, river mile, and water temperature were included as covariates in the final model. The survey index is variable and demonstrates an overall declining trend over the time series (Figure 5.42). Peaks in relative abundance occurred in 1981, 1986, and 2002.

Data from the NYDEC Striped Bass Beach Seine Survey were modeled assuming a negative binomial error structure (Table 5.10). Year, month, river mile, and water temperature were included as covariates in the final model. The survey index is variable and demonstrates an overall declining trend over the time series (Figure 5.43). There is some evidence of an upward trend in the last three years of the time series (2007 through 2009).

5.2.3.4 HRE Monitoring Program

5.2.3.4.1 Data Collection

Survey Methods

The Hudson River Estuary (HRE) Monitoring Program has been run on behalf of several utility companies with power stations in the Hudson River Estuary since 1974. The Program consists of three different surveys. Data from the HRE Icthyopankton Survey was available in time for this assessment.

The HRE Icthyopankton survey was designed to sample for YOY striped bass and follows a random sampling design that consists of paired Tucker trawl (targeting surface and channel) and epibenthic sled (targeting bottom) tows. The Hudson River is split into 13 sampling areas of equal volume and each area is divided into 3 strata (shoal, channel, bottom).

Sampling Intensity

The HRE survey is conducted primarily between March and October and collects approximately 100–200 samples per week depending on season.

Biological Sampling

All eels are measured; however, life stage (YOY vs. yearling or older) was the only data available for this assessment.

Biases

Multiple sampling design changes have occurred over the time series, including timing, frequency, and volume sampled. This survey was not designed to target eel and generate an index of abundance for stock assessments. Standardization of the survey data may provide an index of YOY and “yearling and older” abundance if all important factors have been accounted for properly in the analysis.

5.2.3.4.2 Development of Estimates

Following the methods outlined in Appendix 2, an index of YOY eel was created using a delta model with a gamma distribution; jackknifed standard error estimates were also calculated. A “yearling plus” index (eel classified as being yearling or older) was generated using a negative binomial generalized linear model with a log link.

5.2.3.4.3 Estimates

Both indices of abundance included the factors year, month, gear (tucker trawl vs. sled), strata (bottom, channel, shoal), river mile, and volume sampled (Table 5.10). The YOY index was highly variable throughout the 1970s and 1980s, increased to a peak in 1993, then declined steadily through to the present (Figure 5.44). The “yearling plus” index showed a clear, steady decline across the time series (Figure 5.45).

5.2.4 Delaware Bay Bay/Mid-Atlantic Coastal Bays

5.2.4.1 NJDFW Striped Bass Seine Survey

5.2.4.1.1 Data Collection

The New Jersey Bureau of Marine Fisheries has conducted a young of year striped bass seine survey in the Delaware River since 1980. Yellow stage eels are occasionally captured and enumerated. Although the number of sets that are positive for eels is very limited (~225 sets out of 3,200 total), preliminary analysis indicated moderate correlation with landings (when lagged appropriately) and other regional indices.

Survey Methods

The Delaware River seine survey is conducted between river miles 53.5 and 126 (Salem Nuclear Plant to Trenton). The survey area is divided into three regions based on salinity (Figure 5.46). Stations are sampled twice per month using a 100-foot bagged seine with 0.25” mesh. Survey methodology has changed considerable since the survey began in 1980. Modifications include changes to station selection, distribution of stations among regions, single/replicate tows, and months sampled. Standardized methodology employed since 1998 includes sampling 32 fixed stations twice per month from June to November. Data collected for eels includes number and min/max length per tow. Other information collected includes tide, water temperature, salinity, and dissolved oxygen.

5.2.4.1.2 Development of Estimates

Since the survey began in 1980, the months sampled in a year have expanded and contracted a number of times, but in all years except 1980 sampling has occurred in the core months of August through October (August 1980 was not sampled). In addition, the number, location, and sampling strategy (fixed/random) of stations have changed over time. To minimize effects from changing sampling design, the index is based only on the months August to October and for stations (n = 12) that have been sampled at least 170 times since the beginning of the survey (max 188 observations; the next most frequently sampled stations have ~100 observations). The resulting dataset was evaluated relative to the standardized criteria, and a generalized linear model was developed consistent with those results. The appropriate error structure was applied to the full model which included year, month, water temperature, and salinity. Non-significant factors were removed to produce the final model. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable.

5.2.4.1.3 Estimates

A negative binomial error structure provided the best fit to the raw data, and this was applied to the full model. All factors considered in the full model were found to be significant (Table 5.10). The overdispersion factor, phi, was estimated at 0.958, indicating the negative binomial error structure was appropriate. The predicted NJ striped bass seine survey index (Figure 5.47) was generally higher early in the time series, with three of the first six years having greater than 0.3 eels per tow, and a maximum of 1.01 eels in 1982. Since 1987, the index has been relatively stable between values of 0.05 and 0.10 eels per tow, with the exception of one high value in 2004.

5.2.4.2 University of Delaware Silver Eel Study

5.2.4.2.1 Data Collection

Study Methods

University of Delaware's silver eel study was designed to catch emigrating silver eels in Indian River, Delaware to determine their numbers, sex ratio, age and other biological characteristics. Silver eels were captured in two small, freshwater tributaries of Indian River with fyke nets during 2002 and 2003. The fyke nets were fished during August through November, but most silver eels were caught at the start of emigration, typically after the first major rainfall in September. Some yellow eels captured in Indian River were also kept to compare their ages and growth rates to yellow eels captured in brackish water.

Sampling Intensity

The fyke nets were checked daily during August through November.

Biases

Only eels longer than 250 mm were kept for analysis.

Biological Sampling

All eels were measured, weighed, and assessed for pigmentation. The sampled eels were then sexed histologically and had their otoliths removed for ageing.

Sex ratios were determined for American eels from both tidal tributaries. Length, weight, and age at maturity were calculated for males and females. Growth rates of the freshwater yellow and silver eels were compared to those of brackish water yellow eels.

5.2.4.2.2 Estimates

The male:female ratio was inversely correlated with the percentage of lacustrine habitat in each watershed. The number of silver eels emigrating was positively correlated with water flow. Mature males were significantly smaller than mature females (Figure 5.48). Growth rate was significantly higher in brackish water than in freshwater.

5.2.4.3 Area 6 Electrofishing

5.2.4.3.1 Data Collection

Survey Methods

Pennsylvania Fish and Boat Commission (PFBC) conducts electrofishing surveys at four fixed sites spread over 72 km of the Delaware River. Sites are located at Yardley (RKM 258), Point Pleasant (RKM 291), Upper Black Eddy (RKM 318), and Raubsville (RKM 330).

Sampling Intensity

Sites have been sampled annually from 1999–2010; however, the Upper Black Eddy and Raubsville sites were not sampled in 2000. At each site, six 50-meter sections of shoreline are electrofished for a total of 300 m of shoreline. The number of “pencil eels” is counted within each 50 meter section.

Biological Sampling

No other biological sampling is conducted.

5.2.4.3.2 Development of Estimates

A negative binomial generalized linear model with a log link was used to derive a standard index of abundance for American eel at all four sites following the methods outlined in Appendix 2. The overdispersion factor, phi, for the generalized linear model estimated at 1.05, indicating the negative binomial error structure was appropriate.

5.2.4.3.3 Estimates

Indices of abundance showed a slight decline in 2000, but have remained stable throughout most of the time series (Figure 5.49).

5.2.4.4 Delaware Finfish Trawl Survey

The Delaware Division of Fish and Wildlife (DEDFW) operates two finfish trawl surveys—one for juvenile finfish and one for adult finfish.

5.2.4.4.1 Data Collection

Survey Methods

Juvenile Survey

The DEDFW’s Juvenile Finfish Trawl Survey has been monitoring juvenile fish and crab abundance in Delaware’s inshore waters since 1980. At each site, the 19-m R/V *First State* tows a 4.8-m semi-balloon trawl with a 1.3-cm cod endliner. Tows are made against the current for ten minutes.

Adult Survey

The DEDFW's Adult Finfish Trawl Survey was implemented in 1966 as a long-term fisheries-independent monitoring program. The survey is primarily used to monitor the abundance of sub-adult and adult fish. There are several gaps in sampling in the survey's history, but sampling has been consistently performed every year since 1990.There are nine fixed sampling sites, which are all located off shore in the Delaware Bay and lower Delaware River. Tows are made using the 19-m R/V *First State*, which tows 9.1-m otter trawl with 5.1cm cod end liner. Tow duration is twenty minutes, and tows are made against the current.

Sampling Intensity

Juvenile Survey

Sampling for the Juvenile Finfish Trawl Survey is conducted monthly from April through October at 23 fixed sites in Delaware Bay, seventeen fixed sites in the Delaware River, and 12 fixed sites in Indian River, Indian River Bay, and Rehoboth Bay.

Adult Survey

Adult Finfish Trawl Survey sampling is conducted monthly from March through December.

Biases

Juvenile Survey

The juvenile component of the survey is a fixed site design. The net used rarely retained eels shorter than 120 mm.

Adult Survey

The adult component of the survey is a fixed site design. The net used rarely caught eels.

Biological Sampling

Juvenile Survey

For the Juvenile Finfish Trawl Survey, the catch from each tow is sorted by species, and individuals are measured and weighed. Ageing of eels captured at the Delaware River sites was begun in 2007 and will be continued. The length of American eels caught during the index period ranged from 55 to 605 mm, with a mean of 248 mm.

Adult Survey

For the Adult Finfish Trawl Survey, the catch from each tow is sorted by species, and individuals are measured and weighed. The 23 American eels caught since the survey was standardized in 1990 ranged in length from 250 to 675 mm. Most of these eels were caught in either early spring or late fall at salinities ranging from oligohaline to polyhaline.

5.2.4.4.2 Development of Estimates

Juvenile Survey

A negative binomial generalized linear model with a log link and factors for year, month, salinity, and water temperature (Table 5.10) was used to derive a standardized index of abundance for American eel following the methods outlined in Appendix 2. The overdispersion factor, phi, for the generalized linear model estimated at 1.02, indicating the negative binomial error structure was appropriate.

Adult Survey

Catches of American eels in the Adult Finfish Trawl Survey were extremely rare and so the data were considered inadequate for deriving an index of relative abundance.

5.2.4.4.3 Estimates

Juvenile Survey

The index declined from a peak in 1982 through the late 1980s, increased through the early 1990s, and remained stable with inter-annual variation throughout the rest of the time series (Figure 5.50).

Adult Survey

No index of abundance was developed based on the Adult Finfish Trawl Survey.

5.2.4.5 Delaware Tidal Tributary Survey

5.2.4.5.1 Data Collection

Survey Methods

The DEDFW's Tidal Tributary Fish Habitat Survey was begun in 1996 to evaluate the fish communities and associated habitat in Delaware's tidal tributaries. Two Delaware River tidal tributaries, four Delaware Bay tidal tributaries, and six Inland Bays (Indian River, Indian River Bay and Rehoboth Bay) tidal tributaries were sampled during 1996 through 2005, the final year of the survey. The sampled tidal tributaries were divided into three sections (upper, middle, and lower) based on salinity and fixed trawl sites were established in each section. Each section was sampled with a 3-m semi-balloon trawl that had a 9.5-mm stretch mesh knotless netting liner and was capable of retaining small eels. Tow duration was ten minutes and tows were made with the current.

Sampling Intensity

Sampling was conducted twice monthly during May through October.

Biases

The survey is a fixed site design. Some of the tidal tributaries were sampled during all ten sampling years, but others were sampled for five years.

Biological Sampling

The catch from each tow is sorted by species, and individuals are measured to the nearest mm. Eels were kept for ageing after the American eel FMP was passed in 2000.

Estimates

The American eel caught in this survey ranged in length from 48 mm to 710 mm and had a mean length of 208 mm. Abundance varied widely within and between tidal tributaries. American eel abundance was highest in the oligohaline and mesohaline sections of the sampled tidal tributaries.

5.2.4.6 Neversink River Electrofishing

5.2.4.6.1 Data Collection

Survey Methods

The USGS and The Nature Conservancy have been monitoring eels in several tributaries of the upper Delaware River since 2006 in an effort to quantify local population densities and biomass, document life history strategies, assess their interrelations with other fish species, and to define the effects of selected factors (including the Neversink Reservoir) on resident eel populations (M.B. Delucia, The Nature Conservancy, pers. comm.; http://ny.cf.er.usgs.gov/nyprojectsearch/projects/2457-EF700.html). The time series of abundance/densities were considered too short for inclusion in the stock assessment, but the biological data collected from these surveys were used to characterize the size structure of eel populations in the Upper Delaware River system.

Sampling has occurred in a number of tributaries, but most of the effort has been focused on the Neversink River. The Neversink forms on the highest peak of the Catskill Mountains, running 55 miles before it empties into the main stem Delaware River at Port Jervis, NY. Other sampling locations include the Beaverkill (a tributary of the Delaware River east branch), Basha Kill (a tributary of the Neversink), and Paradise Pool.

Sampling Intensity

Sampling is conducted during summer months using a backpack electroshocker. As such, access to deeper areas is limited. Between 2006 and 2008, a total of 578 eels were sampled for length and weight information.

5.2.4.6.2 Estimates

Average length of all eels was 363 mm, with a range of 147 to 750 mm (Table 5.11; Figure 5.51). Individual weight ranged from 3 to 639 g, with an overall average of 108 g. Length-weight relationships were consistent across the years (Figure 5.52).

5.2.4.7 PSEG Impingement Monitoring

5.2.4.7.1 Data Collection

Survey Methods

The Public Service Enterprise Group (PSEG) Nuclear, LLC of New Jersey operates several ecological monitoring programs in the Delaware Estuary. The objective of the PSEG impingement monitoring program is to estimate the seasonal frequency, abundance, and the initial survival of fish species impinged at Units 1 and 2 at the Salem Generating Station. In addition to the biological data, other data recorded for all samples includes the number of pumps and screens in operation, screen speed, tidal stage and elevation, air temperature, sky condition, wind direction, wave height, water temperature, and salinity. Any detritus collected with the sample is weighed to the nearest 0.1 kilogram.

Sampling Intensity

Impingement sampling is performed three days per week during January through December. The sampling days are selected randomly within each seven-day weekly sampling time frame. During each 24-hr sampling period, ten samples are collected at approximately 2.5-hr intervals, which allows for monitoring over a complete diel period and two full tidal cycles.

Biases

The PSEG Impingement Monitoring Survey is a fixed-design survey, and sampling occurs at one site. The potential bias of the survey data could not be evaluated in terms of persistence since there is only one site. The use of a single site also complicated the possibility of applying geostatistical methods to derive model-based estimators using the survey data.

Biological Sampling

Biological sampling is conducted in the following manner; however, these data were not available for the assessment. Impinged finfish and blue crab are removed from debris for processing. The condition (live, dead, or damaged) of collected individuals is determined, and organisms are then sorted by species. Aggregate counts and weights are recorded for each species observed in each condition category. All individuals of each species in each condition category are measured for length to the nearest millimeter. Subsamples of at least 100 individuals are taken when catches are too large to process in entirety. Individuals are also weighed to the nearest 0.1 gram.

5.2.4.7.2 Development of Estimates

American eel densities were given as number of eels caught per million m^3 of water for each year of impingement sampling.

5.2.4.7.3 Estimates

American eel impingement densities ranged from 0.75 in 1993 to 14.41 in 1986. Impingement samples have not shown an overall trend in American eel density during the 26 year time series (Figure 5.53).

5.2.4.8 PSEG Trawl Survey

5.2.4.8.1 Data Collection

Survey Methods

The objective of the PSEG Bottom Trawl Monitoring Program is to develop indices of abundance for target species; American eel is not a target species. Sampling is performed in the Delaware River Estuary from the mouth of the Delaware Bay to just north of the Delaware Memorial Bridge.

The survey uses a stratified random design. Sites are randomly selected from each of eight zones in the Delaware Bay: Zones 1, 2, and 3 (lower bay); Zones 4, 5, and 6 (“middle” bay); and Zones 7 and 8 (upper bay / lower Delaware River). The number of sampling sites within each zone was determined using a Neyman allocation procedure based on the proportional area of each zone and historical fisheries data.

All sampling is performed during the daytime using a 4.9-m semi-balloon otter trawl with 17-ftheadrope and 21-ft footrope. The trawl body is nylon net made of #9 thread with 1.5-in stretch (0.75-in square) mesh. The cod end is constructed of #15 thread with 1.25-in stretch (0.625-insquare) mesh and fully-rigged with four 2-in I.D. net rings at the top and bottom for lazy line and purse rope. An inner liner of 0.50-in stretch (0.25-in square) mesh #63 knotless nylon netting is inserted and hogtied in the cod end. The trawl doors are 24 inches in length and 12 inches wide and are made of 0.75-in marine ply board, 1.25-in × 1.25-in straps and braces, and a 0.50-in × 2-

in bottom shoe runner. Tow duration is 10 minutes at 6 ft/sec against the direction of the tide. Information on water quality, water clarity, weather, and tidal stage are also recorded at each sampling site.

Sampling Intensity

A total of 40 sites are sampled once a month from April through November.

Biases

The net used rarely retained eels shorter than 120 mm.

Biological Sampling

After each tow, all finfish and invertebrates are identified to the lowest practicable taxonomic level and counted. The lengths of American eels were measured to the nearest millimeter from 1970 through 2001.

5.2.4.8.2 Development of Estimates

An index of eel abundance was calculated from 1970 to 2010. The abundance index was based on catch per tow of American eels at the Delaware River trawl sites during April through June. A negative binomial generalized linear model with log link was created following the methods outlined in Appendix 2.

5.2.4.8.3 Estimates

Model factors included year, month, and bottom salinity (Table 5.10). The index of American eel abundance spiked in the mid-1980s and again around 2005 but was otherwise relatively stable (Figure 5.54).

5.2.4.9 Turville Creek Pot Survey

Maryland DNR Fisheries performed a fishery-independent eel pot survey in 2008 and 2009 in Turville Creek, a tributary to the Isle of Wight Bay in Maryland’s coastal bay. The objective of this survey was to collect demographic information on the yellow eel population in the same system in which the young-of-year Maryland’s survey had occurred since 2000.

5.2.4.9.1 Data Collection

Survey Methods

Approximately 25 cylindrical pots with galvanized wire mesh of 1.27 x 1.27cm (1/2” x 1/2”) were set in fixed locations on individual lines at depths ranging from 3–14 feet. The pots were baited with razor clams (*Tagellus plebius*) and soaked for 48–72 hours. Sample area totaled 2.5 river miles (4 km).

Sampling Intensity

Pots were typically fished twice a week for a six-week period from early April to middle of May.

Biases

In the second year of the study fixed pot locations were altered as a result of commercial crab pot interference. The section of the river sampled remained relatively the same.

Biological Sampling

All captured eels were retained, euthanized, measured to the nearest mm and weighed to the nearest gram. Subsamples were taken for age, gonad, and swim bladder analysis. The majority of eels measured were between 300 and 600 mm in length (Figure 5.55).

5.2.4.9.2 Estimates

The 308 eels measured and weighed had a mean length and weight of 429 mm and 155 grams. Ages ranged from 2 to 8 years for the 196 eels aged and the mean freshwater age was 4.0 years. Females comprised 95% of the subsampled eels and approximately 35% of all eels displayed swim bladder parasite infestation.

5.2.5 Chesapeake Bay

5.2.5.1 Shenandoah River Study

Welsh et al. (2009) initiated a project in 2007 to evaluate the upstream and downstream movements of American eels near dams on the Shenandoah River. Length and weight data collected from downstream migrants in 2007 and 2008 were available for analysis.

The study has supported several graduate projects, including Zimmerman's (2008) study of swim bladder infection in yellow-phase upstream migrants. Lengths and a limited number of ages were available from this study.

5.2.5.1.1 Data Collection

Study Methods

Welsh et al. 2009

American eels collected as part of the downstream migration study were collected upstream of the Luray hydroelectric dam, located on the South Fork of the Shenandoah River in Virginia (Welsh et al. 2009). These eels were collected using hoop nets and backpack and boat electrofishing.

Hoop nets were set in multiple locations within the headwater areas of the South Fork. The nets were constructed of 3.23-cm stretch delta mesh with five 0.75-m diameter hoops and two funnels. Wings were attached to the net to stretch it across the width of the stream, in order to funnel out-migrating eels into the hoops. The wings were weighted to keep them in place when set in the moving water. Stretched seines were placed upriver of the hoop nets to collect debris that would have otherwise clogged the hoop nets.

Boat electrofishing was performed in impoundments and larger sections of the streams with an 18-foot Smith-Root boat using standard umbrella anodes, and the boat hull acted as the cathode. The boat operated at four amps. Backpack electrofishing was conducted in smaller and shallower areas in the headwaters using Smith-Root backpack electrofishers operating at 200 volts.

Zimmerman 2008

Zimmerman's (2008) American eel samples were collected from an eel ladder on the Millville hydroelectric dam, located in the lower Shenandoah River. The eel ladder is a covered metal sluice that slopes 50° and extends 11 m from the western side of the dam. Three rows of vertical

PVC pipes arranged in a peg board pattern provide substrate for the climbing eels. A pipe at the top of the ladder directs eels into a collection tank that contains a 6.35-mm mesh net.

Sampling Intensity

Welsh et al. 2009

The hoop nets were fished during the five days around the new moons of October and November 2007. Sites in the South River, Middle River, and Christians Creek were sampled in October. In November, sites in the North River, Naked Creek, and Mossy Creek were sampled in addition to the October locations. The nets were set during the late afternoon and early evenings and pulled the following morning. Nets fished approximately 15 hours each night, though periodic clogging prevented the nets from fishing the entire duration of the set.

Backpack and boat electrofishing was conducted in September through November 2007.

Zimmerman 2008

Collection tanks in the Millville Dam eel ladder were checked weekly during the summer to early fall during 2006 to 2008.

Biological Sampling

Welsh et al. 2009

Collected American eels were measured for total length, eye height and width, scanned for passive integrated transponder tags (PIT, 2008 only) and color phase (maturity) was determined. The eels were implanted with coded radio tags and released into the South Fork of the Shenandoah River in Virginia.

Zimmerman 2008

Collected American eels were measured for total length, the presence and intensity of *A. crassus* was determined, and health of the swim bladder was assessed. Otoliths were collected from a subsample of these eels and processed for ageing.

5.2.5.1.2 Development of Estimates

Welsh et al. 2009

Individual lengths and weights were available from a total of 115 American eels. Most of the sampled eels (n = 71) were silver-phase eels. Twenty-one were large yellow-phase eels and the remaining 23 were considered to be in an intermediate phase between yellow and silver. The observed length-frequency distribution for each phase was calculated. The average length and weight of sampled eels was also computed for each of the observed phases.

Zimmerman 2008

The lengths of 242 American eels inspected for the swim bladder nematode were recorded. Otoliths from 42 eels were processed for ageing. The length- and age-frequency distributions for these eels were calculated. Average length and age were also computed.

5.2.5.1.3 Estimates

Welsh et al. 2009

Lengths of downstream migrating American eels collected by Welsh et al. (2009) ranged in length from 720 mm to 1,018 mm total length (Figure 5.56). Individual weights of these sampled eels ranged from 660 g to 2,660 g. The average length of sampled yellow eels was 814 mm, and the average weight was 1,100 g. Silver eels averaged 871 mm in length and 1,499 g in weight. The eels classified as intermediate phase had an average length of 843 mm and an average weight of 843 g.

Zimmerman 2008

Yellow-phase American eels observed in Zimmerman's (2008) study ranged in length from 200 mm to 527 mm, with an average of 351 mm (Figure 5.57). Ages ranged from 4 to 11 years, with an average of 6.74 years (Figure 5.58).

5.2.5.2 Sassafras River Study

The primary objective of this study is to characterize the current population segment of American eels in the Sassafras River through a fishery-independent pot survey. This area was specifically chosen because it was previously sampled through a Maryland DNR fishery-independent eel pot study from 1998–2000. The survey was reinitiated in 2006 and is currently ongoing. This study provides the size and age structure, parasite infestation rates, and sex composition of eels in the Sassafras River, as well as a fishery-independent relative abundance index. The Sassafras River is located on the East Upper Chesapeake Bay near the head of the bay. The river is 22 miles long and the drainage encompasses approximately 97 square miles. Tides are diurnal with approximately 0.55 meters (1.8 feet) normal tide range. Salinities predominantly range from 0 to 3.

5.2.5.2.1 Data Collection

Study Methods

This Sassafras River eel pot study was replicated from 1998 field survey methods with slight modifications. In the current study, approximately 30 cylindrical pots with galvanized wire mesh of either 0.83 x 0.83cm (1/3" x 1/3") or 1.27 x 1.27cm (1/2" x1/2") were set in fixed locations on individual lines at depths ranging from 3–20 feet. Sample area totaled 8.7 river miles and divided equally between an 'upper' and 'lower' pot set.

Sampling Intensity

Sampling from 2006–2010 occurred for 4–6 weeks from the middle of May to early July. 'Upper' and 'lower' pot sets were sampled on alternate weeks. The pots were baited with razor clams (*Tagellus plebius*) and soaked for 48 hours.

Biases

In the 1998–2000 survey only 1/3" x 1/3" mesh pots were used and only a portion of the pots had a 1/2" x 1/2"escape panel installed. All 1/3" x 1/3" mesh pots used in the current study had the escape panel installed. Both menhaden (*Brevoortia tyrannus*) and horseshoe crabs (*Limulus polyphemus*) were used in addition to razor clams in the previous study. Sampling covered approximately 4.5 river miles and consisted primarily of the current study's 'upper' pot set. Sampling in 2000 only occurred on 2 days, both of which were in July.

Biological Sampling

All captured eels were retained, euthanized by an ice slurry, clove oil, or MS 222 and measured to the nearest mm and weighed to the nearest gram. Subsamples were taken for age, gonad, and swim bladder analysis.

5.2.5.2.2 Estimates

The 4,190 eels measured and weighed had a mean length and weight of 322 mm and 70 g. Ages ranged from 1 to 11 years for the 628 eels aged with a mean freshwater age of 4.7 years (n = 628). Over 60% of the eels have displayed swim bladder parasite infestation. The female/male ratio was 3:2.

5.2.5.3 Gravel Run Monitoring

In 2006, Maryland DNR Fisheries Service initiated a silver eel study at Gravel Run, a first order stream to the Corsica River (Chester River Watershed) approximately 170 river miles (275 km) from the mouth of the Chesapeake Bay. Gravel Run is 4.5 km in length with 4.1 km above the dam. The main objective of this study is to collect biological information on the migratory ("silver") phase of the American eel that included length, weight, age, parasite infestation, and sex composition.

5.2.5.3.1 Data Collection

Survey Methods

Biological information collected from the non-tidal freshwater silver eel population included out migration timing, abundance, length, weight, age, sex, and swim bladder parasite infestation.

A 2-foot square trap with 1/2" x 1/2" wire mesh was constructed and attached to an eight-foot section of plastic corrugated drain pipe (2 in diameter) that channeled through an out flow pipe on a 4-foot low head dam. This passive gear operates continuously throughout the sampling period and under most conditions 100% of the water above the blockage as well as out migrating silver eels pass through the pipe.

Sampling Intensity

The sampling period in association to the expected timing of silver eel migration in Maryland begins in early to mid-October and ends in early December. Monitoring occurs three days a week throughout the sampling period although the trap samples continuously for 40–60 days barring extraneous weather conditions.

Biases

Under extremely heavy rain events water is spilled over the dam. This lessens the likelihood of the need for the silver eels to pass through the pipe in the dam and therefore decreased capture probability. Due to the variability and intensity of rain events each year and the inability to predict the number of silver eels spilling during those events, use of abundance estimates would not be recommended.

Biological Sampling

All captured silver eels were retained, euthanized, measured (mm), weighed (g), aged, sexed, and noted for the presence of the swim bladder parasite *A. crassus*.

5.2.5.3.2 Development of Estimates

Due to the low sample size all silver eel captured from 2006–2010 were used to compute mean length, weight, age and parasite infestation by sex.

5.2.5.3.3 Estimates

Males accounted for 68% of the catch (n = 68) and displayed a mean length and age of 335 mm and 6.2 years (range = 3–11 years), respectively (Figure 5.59). Females comprised 32 % (n = 32) of the total catch and displayed a mean length and age of 600 mm and 9.6 years (range = 7–14 years), respectively. Prevalence rate of swim bladder parasite *A. crassus* for combined sexes was 52%.

5.2.5.4 Fenske et al. Study

The Chesapeake Biological Laboratory (CBL) collected demographic information from the commercial eel pot fishery in selected tributaries of the Chesapeake Bay in 2007 (Fenske et al. 2010).

5.2.5.4.1 Data Collection

Study Methods

Approximately 5,000 American eels were collected from a commercial eeler using 1/2” x 1/2” eel pots in Potomac River, located in the southwestern part of the Maryland Chesapeake Bay near the Maryland/Virginia state line. Additionally, approximately 100 eels were sampled from 6 river systems fished commercially in the Chesapeake Bay. In 5 rivers (4 MD, 1 VA), eels donated by Delaware Valley Fish Company (DVFC), were randomly sampled from tanks segregated by river system. Randomly sampled Patuxent River eels were acquired through the donation of a commercial eeler.

Sampling Intensity

Eels from the Potomac River were sampled on 6 separate occasions in the months of June, July, September, and October. Specific dates of harvest were unknown from subsampled eel from 5 rivers that were acquired from the DVFC. Eels were classified as either “fall” or “summer” season. Eels sampled from the Patuxent River were collected on one day in June.

Biases

The eels obtained from the DVFC from the James and Potomac rivers were believed to be size graded before the fish were sold; therefore, length, and age distributions compared to other sampled systems may be biased towards larger and older eels.

Biological Sampling

The 5,000 eels collected from the Potomac River were anesthetized, measured to the nearest mm, and released back to the river. Length to the nearest mm, weight (g), gender (as identified through gross visual inspection), and age were collected from subsampled eels from James, Potomac, Patuxent, Choptank, Chester, and Sassafras rivers.

5.2.5.4.2 Development of Estimates

Mean annual growth rate was estimated by dividing TL by age and assumed linear growth. To account for growth that occurred before entry into the Chesapeake Bay region, 57.1 mm and one year was subtracted from length and age, respectively. Catch curves were calculated for each sub-estuary to obtain loss rate estimates (Ricker1975).

5.2.5.4.3 Estimates

Length and age ranges for American eels from the Chesapeake Bay were 213–647 mm TL (mean =365 mm) and 3–11 years (mean = 5.8 years), respectively; weight ranged from 14.7 to 590.8 g (mean=98.8 g). Females constituted 71.3% of all sampled eels. The overall range and mean of growth rates for American eels (gender categories combined) in the Chesapeake Bay were 26.7–149.3 and 67.5 mm/year, respectively. Estimated instantaneous loss rates (gender categories combined) ranged from 0.52 per year in the Choptank River to 1.01 per year in the Potomac River (mean [all rivers] = 0.72 per year).

5.2.5.5 MDDNR Striped Bass Seine Survey

Maryland DNR Fisheries Service conducted a statewide Striped Bass Juvenile Seine Survey from 1954–2010. The primary objective of this survey is to document annual year-class success of young-of-year striped bass. All fish species, including American eel, are enumerated at each sampling station.

5.2.5.5.1 Data Collection

Survey Methods

Sampling of fishes occurred through the use of a 30.5-m x 1.24-m bagless beach seine of untreated 6.4-mm bar mesh. The survey included inconsistent stations and intensity from 1954–1961. Stations were standardized in 1962 and monthly sampling rounds (excursions) were increased to two per site. A third monthly sampling round was added in 1966. A total of 13 fixed stations were sampled with three sampling rounds since 1966. An additional two fixed sites were added in 1970 totaling 15 fixed sampling stations.

Sampling Intensity

Since 1966 sampling occurred at each fixed station once a month for three consecutive months starting in July.

Biological Sampling

Incidence rate and abundance of American eel during the seine survey was relatively low. At least one eel was captured in 8.0% of fixed stations since 1970. A total of 237 eels were sampled in a total of 1845 sites.

Biases

Despite sufficient geographic coverage of Maryland’s Chesapeake Bay, site selection for fixed stations was not random. Stations were selected based on four major spawning and nursery areas for striped bass, which included the Head of the Chesapeake Bay, Potomac, Nanticoke, and Choptank rivers.

5.2.5.5.2 Development of Estimates

Sixteen stations have been sampled relatively consistently since 1966 (two stations were not sampled until 1970 and one more has not been sampled since 2006). Eight of these stations have captured at least 20 eels over the time series (range 20–67, average 35), while the other eight have caught 11 or fewer eels (average 6.4). Stations with few eel observations were considered to occur in unsuitable habitat and were removed from the analysis.

The remaining data were evaluated relative to the standardized criteria, and a generalized linear model was developed consistent with those results. The appropriate error structure was applied to the full model which included year, month, system, salinity, and water temperature. Non-significant factors were removed to produce the final model. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable.

5.2.5.5.3 Estimates

A negative binomial error structure was found to be most appropriate for the raw data. Year, month, and salinity variables were found to be significant (Table 5.10), while system and water temperature were not. The overdispersion factor, phi, was estimated at 0.97, indicating the negative binomial error structure was appropriate. The predicted Maryland striped bass seine survey index (Figure 5.60) peaked in the first year of the time series, decreasing by more than a factor of 10 between 1966 (1.97) and 1967 (0.13). Eel abundance increased gradually through the late 1970s to approximately 0.5 eels per tow, before declining again to approximately 0.05 eels per tow for the years 1990 to 1996. In 1997, the index increased abruptly to 0.6 eels per tow, and has since varied without trend around a mean of 0.4 eels per tow.

5.2.5.6 VGDIF Electrofishing Survey

The Virginia Department of Game and Inland Fisheries (VDGIF) perform a number of surveys throughout Virginia. Their survey database was queried for all American eel data collected. The majority of American eel observations were collected from the VDGIF’s spring and fall community electrofishing sampling. Biologists in years past have been sampling all of Virginia's water bodies looking at fish populations. These surveys generally target sportfish species (i.e., largemouth bass, smallmouth bass, and sunfish).

5.2.5.6.1 Data Collection

Sampling Intensity

The electrofishing sites, length of runs, and timing vary depending on conditions and specific objectives. Rivers are generally sampled either every year or every other year. Smaller creeks and streams are sampled on a rotational or water availability basis.

Biological Sampling

The lengths and weights of American eels encountered during the VDGIF electrofishing surveys were made available for evaluation and analysis.

5.2.5.6.2 Development of Estimates

Lengths and weights of individual American eels collected by the VDGIF electrofishing surveys were available from 1992 to 2010. These data are briefly summarized below. The raw biological data were included in the growth models discussed later in this report (see section 6.2).

5.2.5.6.3 Estimates

The lengths of American eels sampled by the VDGIF ranged from 34.0 mm to 1,000 mm (Figure 5.61). Weights of American eels ranged from 0.100 g to 850 g.

5.2.5.7 VIMS Juvenile Striped Bass Seine Survey

The Virginia Institute of Marine Science (VIMS) initiated a juvenile striped bass seine survey in 1967, but the survey was not conducted between 1973 and 1979 due to funding cuts. Funding was restored in 1980, and the survey has been conducted in every year since.

5.2.5.7.1 Data Collection

Survey Methods

Sampling strategy has changed multiple times over the duration of the survey, with standardized methods being adopted in 1989. Since then, 40 stations are sampled biweekly from early July through mid September (5 rounds per year) using a 100-foot (30.5 m) seine net. Stations are located in the James, York, and Rappahanock Rivers. Data prior to 1989 are not standardized, and VIMS personnel were hesitant to provide data prior to the standardization. However, data from years prior to the harvest increase observed in the 1970s are limited, making early years of the VIMS seine survey very important in characterizing the population during that time period. VIMS personnel agreed to provide the full time series of data contingent upon adequate mention of the potential for inconsistencies in raw data and the resulting index due to non-standardized sampling methodology (M. Fabrizio, VIMS, pers. comm.). Attempts were made to remove potential biases by subsetting the raw data (described below), but it is unknown if these steps were effective.

5.2.5.7.2 Development of Estimates

Recognizing the potential hazards of combining non-standardized data with standardized data, an index was developed using the entire time series of data from the VIMS seine survey. Since the survey began, 88 separate stations have been sampled at least once. In an attempt to remove some uncertainty due to survey changes, the data were subset to include eight stations that have been sampled at least 152 times over the time series (max = 179) with six of these being sampled 174 times or more. The number of eel observed at these stations over the entire time series ranged from 1 to 28 (total = 96, average = 12).

Because of the low incidence of eels at stations used for the full time series index (above), and to investigate the potential for error due to using non-standardized data, a second index was developed from the VIMS seine survey using only data since methods were standardized in 1989. Eighteen stations were sampled consistently from 1989 to 2010. Eight of these stations captured at least 12 eels (max = 42, average = 27.6), while the remaining 10 stations captured 0 to 9 eels each (average = 4.2). The eight stations with the highest eel incidence were used to develop the short time series index.

The remaining data for both the long (1967+) and short (1989+) time series were evaluated relative to the standardized criteria, and generalized linear models were developed consistent with those results. A negative binomial error structure provided the best fit to both sets of data. , Available predictor variables were the same for both series, and included year, month, system, river, station type (striped bass index station or not), salinity, and water temperature. Non-significant factors were removed to produce the final model.

5.2.5.7.3 Estimates

For the long time series index, only year and system were significant (Table 5.10). The overdispersion factor, phi, was estimated at 0.82, indicating the negative binomial error structure was appropriate. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable (Figure 5.62). The predicted VIMS striped bass seine survey index for 1967+ was highest during the late 1960s, reaching a peak of 0.25 eels per tow in 1968. Abundance declined gradually until 1972. Data are unavailable from 1974 to 1979, but abundance continued or resumed to decline from 1980 to 1988. The predicted index is essentially zero from 1989 to 1993, rose gradually for a number of years, and has been highly variable around a mean of 0.05 (range 0.00 to 0.12) eels per tow for the last decade.

The final model for the short time series index included year, station type, and salinity (Table 5.10). The overdispersion factor, phi, was estimated at 1.07, indicating the negative binomial error structure was appropriate. A predicted index was developed based on the lowest level of each class variable and mean values of each numeric variable (Figure 5.63). The predicted VIMS striped bass seine survey index for 1989+ generally increased during the early 1990s, reaching a peak of 0.29 eels per tow in 1997. This was followed by one of the lowest points of the time series in 1998, recovering to the second high point in the time series in 2001. Since 2002 the index has been relatively stable around 0.07 eels per tow, with the exceptions of the two lowest points of the time series in 2003 and 2010.

Despite only moderate correlation (Pearson correlation coefficient = 0.34), the short and long VIMS indices exhibit similar patterns. Both generally increase during the early 1990s, showing peaks around 1996–1997 and 2000–2001, and low points in 1998–1999, 2003, and 2010. The similarity in these patterns lends credibility to the early years of the long time series.

5.2.5.8 VIMS Juvenile Fish & Blue Crab Trawl Survey

5.2.5.8.1 Data Collection

Survey Methods

The Virginia Institute of Marine Science (VIMS) Juvenile Trawl Survey was implemented in 1955 to monitor the seasonal distribution and abundance of important finfish and invertebrate species occurring in the Chesapeake Bay and its tributaries. The main objective of this survey is to develop indices of relative abundance to track year-class strength of target species.

The survey sites and sampling frequency has not been consistent throughout the history of the survey (Tuckey and Fabrizio 2010). The survey currently employs a mixed design, incorporating both stratified random sites and fixed (historical mid-channel) sites. Prior to 1996, sampling occurred at fixed stations only and these were located generally in deep, mid-channel areas of the rivers. In 1996, random stations were added to the sampling frame in the rivers and account for about 63.3% of the stations sampled in any given year after 1996.

The stratification system is based on depth and latitudinal regions in the bay (random stations), or depth and longitudinal regions in the tributaries (random and fixed stations). Each bay region spans 15 latitudinal minutes and consists of six strata: western and eastern shore shallow (4–12 ft), western and eastern shoal (12–30 ft), central plain (30–42 ft), and deep channel (>42 ft). Each tributary is partitioned into four regions of approximately ten longitudinal minutes, with four depth strata in each (4–12 ft, 12–30 ft, 30–42 ft, and >42 ft). Strata are collapsed in areas where certain depths are limited. In each tributary, fixed stations are spaced at approximately 5-mile intervals from the river mouths up to the freshwater interface. Fixed sites are assigned to strata based on location and depth. The stratified random sites are selected randomly from the National

Ocean Service's Chesapeake Bay bathymetric grid, a database of depth records measured or calculated at 15-cartographic-second intervals.

The trawl gear configuration has been modified a number of times, but was standardized in 1979. The various gear configurations have been compared through extensive sampling in order to standardize the catch rates associated with each gear combination. Currently, a 30-ft semi-balloon otter trawl is towed by the R/V *Fish Hawk* using a 60-ft bridle. The trawl is composed of 1.5-in stretch mesh body, a 0.25-in mesh cod end liner, two 28-in × 19-in steel china-v doors, and an attached tickler chain. Tows are made along the bottom during daylight hours for five minutes. The trawl doors were changed in 1991, but the change did not significantly alter the catch.

Sampling Intensity

Two to four sites are randomly selected for each bay stratum each month, and the number of sites varies seasonally. In shallow water strata, only one station is sampled per month. Bay sampling is not conducted during January and March, when few target species are available. One to two stations are randomly selected for most river strata each month. Fixed stations are sampled monthly.

Biological Sampling

The catch from each tow is sorted by species, and fish are enumerated and measured for length and all are released. Lengths are measured to the nearest millimeter using the length type appropriate for the morphology of each species. Random subsamples are taken when catches of a particular species are too large to process efficiently in the field. Invertebrates are identified and some are measured.

The volume of gelatinous zooplankton caught in the net is also measured for each tow because large catches of these organisms may affect the catch (e.g., changes in gear saturation or efficiency).

Hydrographic and station data such as latitude and longitude, depth, tidal current stage, secchi depth, air temperature, wind direction, wind speed, weather conditions, sea state, water temperature, salinity, and dissolved oxygen are also collected. Data characterizing the habitat or substrate type sampled by the trawl have been recorded since May 1998.

5.2.5.8.2 Development of Estimates

Staff at the VIMS has been revisiting the methods used to analyze the data collected by their various surveys and so the development of estimates based on the VIMS Juvenile Trawl Survey data was performed by VIMS personnel.

The time period spanning from 1980 to 2010 was selected for evaluating observations of American eel in the VIMS Juvenile Trawl Survey. During this time period, the majority of American eels greater than 152 mm (pre-recruits and larger, see below) were encountered in the tributaries (James, York, and Rappahannock rivers) of the Chesapeake Bay. Eels captured in the main stem of the bay accounted for only 0.29% (n = 41 of 14,359 eels) of all eels caught and will not be considered further. A major portion (12,111 out of 14,509 or 83.5%) of the tows contained no eels. Excluding the zero catches, catch per tow ranged between 1 and 363 eels; this large catch occurred in the Rappahannock River in September 1989.

Most of the American eels caught were encountered during April through September. This six-month period encompassed 7,490 tows and 86.4% (n = 12,411) of the 14,359 eels captured. The VIMS Juvenile Trawl Survey did not sample in April 1980, 1981, 1982, 1983, or 1988. The index period for American eel is therefore April through September and includes catches from only the rivers; observations from 7,490 tows were retained for calculation of the index of abundance. Most of the eels captured between April and September (80.9%) were taken from fixed stations in the rivers; this association appears to reflect the higher abundance of eels in the rivers during the 1980s and early 1990s when only fixed stations were sampled. Since 1996, when sampling at random stations commenced, about 58.7% of eels were captured at random stations.

Pooling across years, about half of the catch of American eels was obtained in the Rappahannock River (51.5%); the York River produced the lowest proportional catches overall (17.5%); however, these proportions varied among years, indicating large annual variations in catches among the three tributaries. These differences probably reflect random variation in abundance of eels in these systems and are not the result of annual differences in sampling effort among the tributaries (over all years, total sampling effort—7,490 tows—was allocated as 32.4% in the James, 33.9% in the Rappahannock, and 33.7% in the York).

Indices of American eel abundance were calculated for four size groups using data collected from the rivers during April through September from 1980 to 2010 (Figures 5.64–5.66). The size groups were pre-recruits (less than 300 mm but > 152 mm), recruits (300–400 mm), post-recruits (≥300 mm), and all (>152 mm). The indices were calculated as random stratified means (Cochran 1977) using stratum areas as weighting factors. The means were expressed as the numbers of eels per 5-minute tow. No other standardization could be performed because area swept was not measured prior to 1991; thus, this analysis is based on the assumption that each 5-minute tow sampled a consistent area. Within each stratum, the mean catch was estimated using the delta-lognormal model. Total weights varied annually (especially in the beginning of the time series) because the area sampled by the trawl survey varied. The application of the design-based estimator (random stratified mean) requires the assumption that data were randomly sampled within each river (stratum). Thus, catches from fixed river stations were assumed to represent a random sample from the rivers.

The variance of the stratified mean was estimated from 1,000 bootstrap replicates, which were also used to determine the upper and lower confidence bounds on the mean (α=0.05).

5.2.5.8.3 Estimates

A decline in abundance since the mid-1980s is apparent and index values during the last 7 years are particularly low. Despite the range of lengths sampled, the standardized index values and temporal pattern in abundance were remarkably consistent regardless of the size group considered to construct the four indices (all eels, pre-recruits, recruits, post-recruits; Figure 5.67). These patterns were inconsistent with other recent presentations of the same data (Figure 4b from Fenske et al. 2011, Figure 28 from ASMFC 2005). These inconsistencies could not be resolved without analysis of the raw data, so the VIMS Juvenile Trawl Survey was not included in this stock assessment.

5.2.5.9 North Anna Electrofishing Survey

5.2.5.9.1 Data Collection

Survey Methods

In 1972, the North Anna River was impounded to create Lake Anna, a 3,885 hectare (9,600 acres) reservoir (lake) that provides condenser cooling water for the North Anna Power Station. Adjacent to Lake Anna is a 1,376 hectare (3,400 acre) Waste Heat Treatment Facility that receives the cooling water and transfers excess heat from the water to the atmosphere before discharging into the lake.

Abundance and species composition data for the North Anna River fish assemblage were collected via backpack and seine electrofishing surveys since 1981. An approximately 70-m reach of riffle/run type habitat was sampled at each station with an electric seine. Prior to sampling, each reach was blocked at the downstream end with a 6.5-mm mesh net. Sampling was conducted by working the electric seine from bank to bank in a zigzag pattern from the upstream to the downstream end of the section. Nearby pool type habitats were then sampled for 10 minutes of effort with a via backpack electrofishing. Data for both sampling gear were combined prior through 1989, so only 1990–2009 data were used in this analysis. Water temperatures (°C) were recorded hourly at Station NAR-1 in the lower North Anna River approximately 1 km below the Lake Anna dam.

Sampling Intensity

Sample frequency for electrofishing is typically once per month each year in May, July, and September. Consequently, this provides for a total of 24 river electrofishing collections for a typical sample year (12 electric seine and 12 backpack). Some sampling events over the time series were delayed or canceled due to rain and high flows that made sampling unsafe. For analysis, samples were grouped into three time periods: May–June, July–August, and September–November. No sampling occurred in 2003.

Biases

Sampling was inconsistent across years, so some years (2003, 2006–2007) did not contain enough observations to estimate CPUE during standardization (see below). Likewise, temperature and dissolved oxygen measurements were inconsistently recorded in earlier years of the study. Length and weight records were available for 1990–2006.

Biological Sampling

Most fish collected were preserved in 10% formalin and transported to the laboratory for appropriate processing. Some larger fish were weighed and measured in the field and released. In the laboratory, a maximum of 15 individual specimens of each species were weighed to the nearest 0.1 g and measured to the nearest 1 mm total length. If more than 15 specimens of a species are collected, those in excess of 15 were counted and weighed in bulk.

5.2.5.9.2 Development of Estimates

A negative binomial generalized linear model with a log link was constructed to standardize the electrofishing survey and create an index of abundance for American eel in the North Anna River following the methods outlined in Appendix 2.

5.2.5.9.3 Estimates

The length distribution of eel caught in the electrofishing survey ranged from 36 to 726 mm (mean = 198.6 mm, median = 185 mm). The length distribution exhibits a peak around 200 mm (Figure 5.68).

Year, electrofishing method (seine vs. backpack), time period (May–June, July–August, July–August, or September–October), and station were significant factors in the model (Table 5.10). The standardized abundance index showed a slight decline in the early 1990s followed by a period of steady increase through 2007; a sharp increase was observed in the last two years (Figure 5.69).

5.2.6 South Atlantic

5.2.6.1 NCDMF Estuarine Trawl Survey

In 1971, the DMF initiated a statewide estuarine trawl survey (Program 120). The initial objectives of the survey were to identify the primary nursery areas and produce annual recruitment indices for economically important species such as spot, Atlantic croaker, weakfish, flounders, blue crab, and brown shrimp. Other objectives included monitoring species distribution by season and by area and providing data for evaluation of environmental impact projects.

5.2.6.1.1 Data Collection

Survey Methods & Sampling Intensity

Various gears and methodology have been used in the survey since 1971. In 1978 and 1989 major gear changes and standardization in sampling occurred. In 1978 tow times were set at one minute during the daylight hours. In 1989 an analysis was conducted to determine a more efficient sampling time frame to produce juvenile abundance indices with acceptable precision levels for the target species and the following changes were made: 1) a fixed set of 105 core stations was identified, 2) sampling would be conducted in May and June only, except for July sampling for weakfish (dropped in 1998 because another survey was deemed adequate), and 3) only the 10.5 ft head rope trawl would be used. July sampling for a subset of the cores was reinstituted in 2004 in order to produce a better index for spotted sea trout. Additional habitat fields were added in 2008. A daylight one minute tow is made with an otter trawl covering 75 yards. Environmental data taken include water temperature, salinity, dissolved oxygen, depth, and bottom type.

Biases

This survey and survey gear were not designed to target American eel or to generate an index of abundance for stock assessments. Standardization of the survey data may provide an index of abundance if all important factors have been accounted for properly in the analysis. Also, catchability of eel has not been quantified with this gear and study design.

Biological Sampling

All species taken are identified, sorted and a total number is recorded for each species. For target species, 30–60 individuals are measured.

5.2.6.1.2 Development of Estimates

A negative binomial generalized linear model with a log link was constructed to standardize the estuarine trawl survey and create an index of abundance for American eel in North Carolina waters following the methods outlined in Appendix 2. Unrealistic water temperature measurements were recorded that could not be resolved, so water temperature was not included in the analysis. Dissolved oxygen, salinity dissolved oxygen, and depth were not recorded consistently across the time series, so they were also not included in the analysis. Bottom type records were re-coded into a condensed set of categories (algae, detritus, grass, no grass, and other).

5.2.6.1.3 Estimates

The length distribution of eel caught in the estuarine trawl survey ranged from 26 to 921 mm (mean = 213 mm, median = 205 mm). The length distribution is bimodal with one peak around 75 mm and another peak around 175 mm (Figure 5.70).

The final index of abundance included the following factors: year, latitude, longitude, and bottom type (Table 5.10). A downward trend in the index of abundance was apparent from the peak in the mid 1990s to the present (Figure 5.71).

5.2.6.2 Cudney Study

Cudney (2004) studied an American eel population in North Carolina (northwestern Pamlico Sound, Lake Mattamuskeet, and adjacent canals) between 2002 and 2003 in order to characterize population demographics and critical habitat needs. Lake Mattamuskeet, one of NC's largest coastal lakes, is connected to Pamlico Sound via four major canals. Saltwater intrusion into the lakebed and surrounding areas was managed with water control structures through which eel were able to pass after the installation of flapgates. The area provides excellent eel habitat and is centrally located among coastal eel harvest grounds. No commercial fishery for eel presently exists in the lake; however, an eel processing and distribution plant operated there for a few years in the mid-1970s. Sale of commercial permits to fish on the Mattamuskeet National Wildlife Refuge ceased after the NCDMF enacted a six-inch minimum size limit to protect young eels in North Carolina waters. However, poaching of glass eels and elvers remains a problem.

5.2.6.2.1 Data Collection

Study Methods

For details on data collection, consult Cudney (2004). Eel pots were placed in at least 15 permanent sampling stations through the canals that link Lake Mattamuskeet and Pamlico Sound. Sites were changed during the study based on catch, habitat quality, and the need to supplement eel pots in areas more frequently visited by locals and tourists (pot theft). Sites for eel pots fished in Lake Mattamuskeet were selected using stratified random sampling based on historical vegetation surveys (1989–1997), depth, and distance from shore. Eel were caught using 24-inch and 36-inch eel pots constructed of 0.5-inch square mesh and baited with frozen menhaden.

Sampling Intensity

Cudney sampled from February 2002 through September 2003. Pots were normally allowed to soak overnight in Lake Mattamuskeet except during instances of severe weather (hurricanes). Pots were checked every two or three days, and all catch was removed and enumerated. A total of 768 eel were sampled for length and weight. Sex ratios were calculated based on a sample of 442 eels and age was determined for 566 eels.

Biases

Eel pots were removed by visitors and were sometimes found out of the water with the bait or catch removed, baited with chicken necks or other materials, or moved to a new location. On occasion, sampling locations were changed in an attempt to prevent disturbance. In total, 32 eel pots were stolen from the canals for an estimated loss of 127 fishing days.

Biological Sampling

Cudney weighed and measured a subsample of eel; eels were characterized as either yellow or silver based on coloration, fin shape, eye diameter, and size. Sagittal otoliths were removed and whole mounted otoliths were read by multiple laboratory personnel. Sex was determined through macroscopic observation of gonads and represented a minimum probable sex ratio since histological analysis of gonads was not attempted. Fish were classified as male, female, or undifferentiated/intersexual. Demographic information and physical condition of the local population was comparable to populations in adjacent states.

5.2.6.2.2 Estimates

Lengths of eels sampled varied between 49 and 719 mm with an average of 438 mm and differed between estuarine and freshwater eels (Figure 5.72). Weights varied between 24 and 1027 g with an average of 197 mm. The average age observed was 5 with a minimum of 2 and a maximum of 14.

5.2.6.3 Roanoke Rapids Dam Studies Data Collection

Several studies of American eel at the Roanoke River and Roanoke Rapids Dam have been conducted between 1999 and the present by personnel of Dominion Electric Environmental Services. In 1999–2000, an electrofishing study was conducted to compare size, heath, and relative abundance of eel in the Roanoke River with that of nearby river systems. From 2005 to the present, eel traps have been used to monitor and collect samples of American eel during passage above the Roanoke Rapids Dam.

5.2.6.3.1 Data Collection

Study Methods

During August to September 1999 and July 18–20 in 2000, eel in the Roanoke River were sampled via backpack electrofishing during low flow conditions (to facilitate wading) in each of three different habitat types (riffle, run, and pool). Blocking nets proved infeasible, so field crews made one pass upstream attempting to cover 2-m wide area a distance of 30 m^2. Three people used dip nets (640-mm mesh) to collect stunned eel.

Sampling Intensity

In 1999, four electrofishing stations were sampled; 10 electrofishing stations were sampled during the July 2000 study. During the passage monitoring study (2005–2009 data available), 10 eel traps were used to collect American eels in the Roanoke Rapids bypass on a weekly or biweekly basis. Five of the traps had a 7/16 inch ramp substrate, and four had a 1 inch ramp substrate.

Biases

Given the pilot study nature of the 1999–2000 work and the short time series collected to date for the passage monitoring study, reliable CPUE trends could not be generated at this time. Also, consistent sampling protocols were not maintained across all years of passage monitoring. If consistent protocols can be successfully maintained into the future, the passage monitoring study will have great value for the next assessment as an index of abundance on the Roanoke River.

Biological Sampling

For the electrofishing study, 463 eel were collected between 1999 and 2000. Total length (mm) was reported for all animals and weight (g) was reported for all sampled fish. For the passage monitoring study, 14,692 eel were collected and measured for total length. Weight was reported only for eel caught in 2006 through September 2007.

5.2.6.3.2 Estimates

The average size of eel caught in the passage traps between 2005 and 2009 was 125 mm (range 89–298 mm, median 123 mm; Figure 5.73).

5.2.6.4 South Carolina Electrofishing Survey

5.2.6.4.1 Data Collection

Survey Methods

The SC electrofishing survey began in May 2001, sampling six strata within estuarine systems along the South Carolina coast. These included the lower and upper Edisto Rivers, the Combahee River, the upper Ashley River, the upper Cooper River, and the North Santee River. Winyah Bay replaced the North Santee stratum in November 2003. The Upper Edisto and Combahee River strata are freshwater, whereas the others have salinities of up to ~10 ppt.

At each randomly chosen site, a 15-minute set was made along the shoreline in a Smith-Root electrofishing boat. Sampling was performed with the boat moving in the direction of the current, which allows stunned fish to be easily netted as they float alongside the boat. Straight shorelines were sampled by shocking at idle-speed approximately 1.5 to 3-m from the bank. More complex locations that contained submerged trees, remnants of old docks, mouths of tributaries and sloughs required more maneuvering with the boat to ensure all areas were sampled.

Sampling Intensity

The shorelines of each stratum are partitioned into 926-m (0.5-nautical miles) long intervals, with each one representing a potential sampling site. Prior to each month's sampling, sites are chosen from a table of random numbers without replacement. The number of potential sites in each stratum is: North Santee River = 82; Upper Cooper River = 63; Upper Ashley River = 80; Lower Edisto River = 88; Upper Edisto River = 86; Combahee River = 232; Winyah Bay = 65.

Variability in the number of sites was caused by drought conditions during some years. Since light rainfall reduced freshwater runoff and allowed the penetration of tidal salt water further upriver, additional upstream sites had to be added in some strata, since the effectiveness of the shocking unit declines at salinities of above ~12 ppt.

Biases

This survey was not designed to target eel and generate an index of abundance for stock assessments. Standardization of the survey data may provide an index of abundance if all important factors have been accounted for properly in the analysis. Also, catchability of eel has not been quantified with this gear and study design.

Biological Sampling

Captured fish were placed in a live well until the end of each 15 minute set, at which time they were counted and measured. Standard length measures (nearest mm) were taken from the first 25 randomly selected individuals of each species collected. All fish were released alive.

5.2.6.4.2 Development of Estimates

A negative binomial generalized linear model with a log link was constructed to standardize the electrofishing survey and create an index of abundance for American eel in South Carolina waters following the methods outlined in Appendix 2. The North Santee River stratum was combined with the Winyah Bay stratum (its replacement in the sampling design) for the analysis.

5.2.6.4.3 Estimates

The length distribution of eel caught in the electrofishing survey ranged from 44 to 890 mm (mean = 370 mm, median = 355 mm). The length distribution is bimodal with one peak around 300 mm and another peak around 525 mm (Figure 5.74).

The abundance index included the following factors: year, strata (river system), water temperature, salinity, and tide (Table 5.10). The trend in the index shows an overall decline from a peak in the early 2000s to present (Figure 5.75).

5.2.6.5 FWRI River Electrofishing

5.2.6.5.1 Data Collection

Survey Methods

The FL FWCC has conducted electrofishing surveys in four rivers from 1996–2008. However, only the Suwannee River has been sampled consistently over this time period and this summary focuses on the data from the Suwannee River.

Sampling Intensity

The Suwannee River has been sampled from 1996–2008. The number of sites electrofished each year varies between 1 and 6 sites. No sampling occurred in 2001. The timing of sampling varies from year to year.

Biases

Although the Suwannee River electrofishing survey supplies a time series of relative abundance, the non-standard timing of this sampling within a year brings into question the usefulness of this survey as a relative abundance index.

Biological Sampling

Lengths and weights of captured eel are measured. A weight-length equation was developed (Figure 5.76).

5.2.6.6 FWRI Lake & Marsh Electrofishing

5.2.6.6.1 Data Collection

Survey Methods

The FL FWCC has conducted electrofishing surveys in more than 50 lake/marsh areas from 2006–2010 as part of their long-term monitoring program. Lakes are chosen to represent all chains of lakes within the state and by their importance to freshwater fisheries. The lakes/marshes where eels have been captured include: Crescent Lake, Dead Lakes, Dear Point Lake, Farm 13/Stick Marsh, L-35B, L-67A Canal, Lake Garcia, Lake George, Lake Harris, Lake Jesup, Lake Monroe, Lake Panasoffkee, Lake Poinsett, Lake Sampson, Ocklawaha River, and St. Johns River. Electrofishing surveys are generally conducted in the fall between September and December. A limited number of surveys have been conducted in the spring, but spring surveys are not included in this summary. Standard electrofishing methods are used in each lake. Each lake is divided into 750 m sections of shoreline and 25 of these sections are randomly sampled during each sampling event.

Sampling Intensity

Multiple sites are electrofished during a sampling event within an area with approximately 10 minutes of shock time at a site. Not all areas are sampled each year and the number of sites electrofished within an area varied from 15 to 90.

Biological Sampling

Length and weight data of captured eels are collected.

5.2.6.6.2 Development of Estimates

A weight-length equation was developed from data combined over all areas (Figure 5.77).

5.2.6.6.3 Estimates

Average total length of American eels collected in this survey was 472 ± 136 (±st. dev.) mm and ranged from 110 to 832 mm (Figure 5.78).

6 ASSESSMENT

6.1 Coast-wide Abundance Indices

Indices of coast-wide abundance for YOY and yellow-phase American eel were developed by combining data from multiple surveys along the coast. Detailed information describing the surveys included in the coast-wide indices can be found elsewhere in this report as indicated by the relevant section numbers given below.

6.1.1 Data Collection

Coast-wide Recruitment

Methods of data collection for the ASMFC-mandated YOY abundance surveys, the Little Egg Inlet Ichthyoplankton Survey, and the Beaufort Inlet Ichthyoplankton Survey are described in section 5.2.1. Details describing data collection for the HRE Monitoring Program can be found in section 5.2.3.4.

Coast-wide Yellow-Phase Abundance

The surveys used to develop the coast-wide yellow-phase abundance indices and the report section providing additional details (in parentheses) were: Western Long Island Study (section 5.2.2.3), HRE Monitoring Program (section 5.2.3.4), NYDEC Alosine and Striped Bass Beach Seine Surveys (section 5.2.3.3), New Jersey Striped Bass Seine Survey (section 5.2.4.1), Delaware Juvenile Finfish Trawl Survey (section 5.2.4.4), PSEG Trawl Survey (section 5.2.4.8), Maryland Striped Bass Seine Survey (section 5.2.5.5), North Anna Electrofishing Survey (section 5.2.5.9), VIMS Juvenile Striped Bass Seine Survey (section 5.2.5.7), and the NCDMF Estuarine Trawl Survey (section 5.2.6.1).Although these surveys catch yellow stage eels, it should be noted that some portion of the catch in these surveys may include elvers as well.

6.1.2 Development of Estimates

Coast-wide Recruitment

Two coast-wide indices of American eel recruitment were computed—a short-term index and a long-term index. The short- and long-term indices were developed by combining individual standardized indices into a single, coast-wide index using the generalized linear modeling approach (Appendix 2). The short-term recruitment index was based on the standardized indices developed from the ASMFC-mandated annual YOY surveys. The long-term recruitment index was based on the HRE Monitoring Program, Little Egg Inlet Ichthyoplankton Survey, and Beaufort Inlet Ichthyoplankton Survey standardized indices. The covariates considered for inclusion in the model for the short- and long-term indices were year, region, and survey site. The time period used for generating the long-term coast-wide recruitment index was 1987 to 2009. This time period was selected so that index values from at least two of the long-term YOY surveys were available for every year included in the combined index.

Coast-wide Yellow-Phase Abundance

Three indices of coast-wide, yellow-phase abundance were computed using different time series lengths—twenty, thirty, and forty-plus years. The indices were developed by combining individual standardized indices into coast-wide indices using the generalized linear modeling approach (Appendix 2). The 40-plus-year coast-wide index of yellow-phase abundance was based on the PSEG Trawl Survey, MDDNR Striped Bass Seine Survey, and VIMS Juvenile Striped Bass Seine Survey (long time series) standardized indices. The 1967–2010 time period was used for the 40-plus index because it was the longest time series that could be used for which at least two of the 40-plus-year indices were available for every year included.

The 30-year coast-wide, yellow-phase abundance index included the same survey indices as the 40-plus index as well as the HRE Monitoring Program, NYDEC Alosine Beach Seine Survey, NYDEC Striped Bass Beach Seine, and New Jersey Striped Bass Seine Survey. The 20-year index included the same survey indices as the 30-year index except for the VIMS Juvenile Striped Bass Seine Survey long time series index. Instead, the 20-year yellow-phase abundance index included the short time series index developed from the VIMS Juvenile Striped Bass Seine Survey. In addition, the 20-year index included the Western Long Island Sound Seine Survey, Delaware Trawl Survey, North Anna Electrofishing Survey, and NCDMF Estuarine Trawl Survey standardized indices.

6.1.3 Estimates

Coast-wide Recruitment

The short- and long-term YOY recruitment indices were developed assuming a lognormal error structure. The final model for both indices included year and survey site as covariates. The estimate of overdispersion (phi) for the short-term recruitment index was 1.34 and the estimate for the long-term index was 0.0416.

The short-term, coast-wide recruitment index is variable and exhibits two periods of decline in the time series (Figure 6.1). The first period of decline occurred from 2001 to 2004 when the index declined from the time-series peak in 2001 to the time-series low in 2004. The index increased from 2004 to 2005 and then steadily declined through 2009.

The long-term, coast-wide index is variable and without trend (Figure 6.2). There is little coherence between the short- and long-term recruitment indices for the period of time over which the indices overlap (Spearman's rank: $\rho = 0.212$, $P = 0.556$).

Coast-wide Yellow-Phase Abundance

The coast-wide, yellow-phase abundance indices were developed assuming a lognormal error structure. The final model for all three indices included year and survey site as covariates. Overdispersion estimates for the coast-wide 40-plus, 30-year, and 20-year indices of yellow-phase American eel abundance were 0.145, 0.0945, and 0.0644.

The 40-plus yellow-phase index for the coast demonstrates inter-annual variability, and there is no evidence of an overall trend over the time series (Figure 6.3). The 40-plus index does show peaks in yellow-phase abundance occurring in 1985 and 2005. The peak in 1985 is followed by a decline that continues through 1989. The 30-year coast-wide index of yellow-phase American eel abundance also exhibits a decline from 1985 to 1989 (Figure 6.4). After 1989, the 30-year index show little variability or trend throughout the rest of the time series. The 20-year index of yellow-phase abundance shows limited variability and a slightly increasing trend over the time series (Figure 6.5). The three coast-wide, yellow-phase abundance indices are significantly and positively correlated with each other (Spearman's rank: $P < 0.001$).

6.2 Regional Abundance Indices

Indices of regional abundance for YOY and yellow-stage American eel were developed for each of the regions by combining data from relevant surveys within each region (Table 6.1). Note that the regional indices labeled as yellow-stage indices actually reflect the relative abundance of both yellow-stage eels and elvers, in most cases (see Table 5.10).

6.2.1 Data Collection

Detailed information describing the surveys included in the regional indices can be found in the sub-section for the associated region within section 5.2 of this report.

6.2.2 Development of Estimates

Region-specific indices of YOY and yellow-stage relative abundance were computed for each of the six geographic regions where data were available. Indices of YOY and yellow-stage American eel abundance were developed by combining individual standardized indices (Tables 5.8 and 5.10) using the generalized linear modeling approach (Appendix A). The time period for each regional index was selected so that index values from at least two of the surveys included were available for every year included in the combined index. The surveys used in the development of the regional YOY and yellow-stage indices and the time periods of those indices are listed in Table 6.1.

Spearman's rank correlation coefficient, ρ, and the associated probability were calculated for all pairs of regional YOY indices and all pairs of regional yellow-stage indices to assess the degree of association among the indices. The correlation analysis was also applied to evaluate the degree of association between the yellow-stage indices and the YOY indices within each region. The YOY indices were lagged by 0–4 years for comparison to the yellow-stage indices. Indices were considered significantly correlated at $\alpha = 0.10$.

6.2.3 Estimates

All region-specific YOY and yellow-stage indices of American eel abundance were modeled assuming lognormal error structures and the final models all included year and survey as covariates. The Hudson River region YOY index was based on a single recruitment index because only one such index was available for the region (Table 6.1). No yellow-stage indices of American eel abundance were available for the Gulf of Maine so a yellow-stage index could not be developed for the Gulf of Maine.

The regional YOY and yellow-stage indices of American eel abundance are depicted in Figures 6.6 and 6.7. Both the YOY and yellow-stage regional indices are variable among years. All the YOY indices, except in the Hudson River region, are characterized by relatively large standard errors ($\geq$30% of the index estimates; Figure 6.6). This is partly due to the differences in the magnitudes of the index values among surveys that were combined in developing the region-specific indices.

Among the regional YOY indices for American eel, the South Atlantic index was found to be significantly and positively correlated with Gulf of Maine, Hudson River, and Chesapeake Bay indices ($P < 0.001$; Table 6.2). Significant, positive correlations were also detected between the Gulf of Maine and Hudson River YOY regional indices as well as between the Hudson River and Chesapeake Bay YOY regional indices. There were no statistically significant correlations detected among the region-specific yellow-stage indices (Table 6.3). Few significant correlations were detected between the region-specific yellow-stage and lagged YOY indices (Table 6.4). The Chesapeake Bay yellow-stage index was significantly and negatively correlated with the Chesapeake Bay YOY index that was not lagged ($\rho = -0.627$, $P = 0.0388$). The South Atlantic yellow-stage index was significantly and positively correlated with the South Atlantic YOY index that was lagged three years ($\rho = 0.750$, $P = 0.0522$).

6.3 Analyses of Life History Data

6.3.1 Growth Meta-Analysis

6.3.1.1 Methods

Biological data for American eel were compiled from a number of past and on-going research programs along the Atlantic Coast and classified into one of the six geographic regions used in this assessment (Table 6.5). The biological data were used to model both the length-weight and age-length relationship for American eel.

The relation of length in millimeters to weight in grams was modeled using the allometric length-weight function. Length-weight parameters were estimated by region, sex, and for all data pooled together. The analysis of the residual sum of squares (ARSS) method was performed to compare the length-weight curves among regions and between sexes (Chen et al. 1992; Haddon 2001). The ARSS method provides a procedure for testing whether two or more nonlinear curves are coincident (i.e., not statistically different). Values were considered statistically significant at $\alpha < 0.05$. Note that interpreting the results of this test is partly confounded by the differences in the range of lengths and weights available for the various dataset configurations.

Previous studies that have modeled the age-length relation for American eel have used linear regression (Table 2.6). Linear regression was used here to model the relation of age in years to length in millimeters by region, sex, and for all data pooled together. A test for coincident regressions was applied to test for differences in the regressions among regions and between sexes (Zar 1999). Values were considered statistically significant at $\alpha < 0.05$. As with the ARSS test for coincident curves, the results of the test for coincident regressions will be partly confounded by the differences in the range of ages and lengths available for the various dataset configurations.

Alternative age-length models were fit to the available data to determine what model best characterizes American eel growth. The models considered are described below.

One of the most commonly used models to describe the age-length relationship is the von Bertalanffy model, which is given by:

$$L_t = L_\infty\left[1 - e^{-K(t-t_0)}\right]$$

where L_t is length at age t, L_∞ is the theoretical asymptotic average length (if $K > 0$), K is growth rate at which the asymptote is approached, and t_0 is the hypothetical age at which length is zero.

The Gompertz growth model is a three-parameter sigmoid function and is calculated as:

$$L_t = L_\infty e^{-\frac{1}{K}e^{-K(t-t_0)}}$$

The Richards model is a generalization of the von Bertalanffy model to allow for greater flexibility:

$$L_t = L_\infty\left[1 - \delta e^{-K(t-t_0)}\right]^{\frac{1}{\delta}} \quad \delta \neq 0$$

where δ is an additional parameter estimated by the model.

The logistic age-length model is equivalent to the Richards model when δ = -1 and is given by:

$$L_t = L_\infty\left[1 + e^{-K(t-t_0)}\right]^{-1}$$

Schnute provides a general four-parameter model describing a relative, rather than instantaneous, rate of change in growth that contains most of the preceding models as special cases. The model is given by:

$$L_t = \left[L_1^b + \left(L_2^b - L_1^b\right)\frac{1 - e^{-a(t-t_1)}}{1 - e^{-a(t_2-t_1)}}\right]^{\frac{1}{b}}$$

for case 1 (see Schnute 1981) where t_1 and t_2 were specified as the youngest and oldest ages observed, L_1 is length at age t_1, L_2 is length at age t_2, and the parameters a and b define the shape of the curve and are not equal to zero for case 1.

Model fits were first evaluated based on convergence status; models that did not successfully converge were removed from consideration for the associated dataset. The fits of models that successfully converged were compared using the Akaike Information Criterion (AIC) for use with sum of squares (Hongzhi 1989; Hilborn and Mangel 1997). This method takes into account both the goodness-of-fit and the number of parameters estimated. The model fit associated with the smallest AIC value is considered the most likely to be correct among the models considered, given the data. Akaike weights were also calculated to quantify the relative probability that each model is correct, given the data and set of candidate models (Burnham and Anderson 2002). AIC and Akaike weights apply to comparisons of different models fit to the same dataset.

6.3.1.2 Results

The length-weight model successfully converged when fit to all dataset configurations (Table 6.6). The results of the ARSS indicated that there are statistically significant differences in the length-weight relation among regions ($F_{10,\ 49{,}209} = 295$, $P < 0.001$). The fit of the length-weight function to all pooled data was dominated by data from the Chesapeake Bay region (Figure 6.8), which was the source of the majority of length and weight biological samples (Table 6.5). Sex-specific differences between the length-weight parameters were nearly significant ($F_{2,\ 4{,}993} = 2.89$, $P = 0.0555$; Figure 6.9).

The parameters estimated from the linear regression of length on age for the various dataset configurations are presented in Table 6.7. There are statistically significant differences in the age-length relation among regions based on the results of the test for coincident regressions ($F_{10,\ 13{,}520} = 659$, $P < 0.001$). The final parameter estimates suggest that growth in length with age is fastest in the Delaware Bay/Mid-Atlantic Coastal Bays region and the Chesapeake Bay region (Table 6.7; Figure 6.10). The test for coincident regressions also detected significant differences in the age-length regressions between sexes ($F_{2,\ 4{,}615} = 1{,}102$, $P < 0.001$; Figure 6.11). The results suggest the rate of growth in length with age is faster in females than males (Table 6.7; Figure 6.11).

The various models relating length to age were compared based on ranking of AIC values among candidate models within each dataset configuration. Only models that successfully converged and produced realistic parameter estimates were considered. Estimates from the age-length linear regressions were presented in Table 6.7. The parameter values and associated standard errors estimated by the nonlinear age-length models are shown in Tables 6.8–6.12. None of the nonlinear models considered successfully converged on all dataset configurations. The only dataset configurations that were successfully fit by all models were all data pooled, Hudson River region, and Delaware Bay/Mid-Atlantic Coastal Bays region. Parameter estimates of the Schnute model for the Southern New England region are considered unrealistic because the resulting curve suggests almost no growth as age increases except at the very oldest ages at which growth appears exponential (Table 6.12).

There was no one model that was found to consistently result in the lowest AIC and highest Akaike weight among the all dataset configurations (Table 6.13). This could be attributed to real differences in growth among the different configurations but one must consider the differences in the number of samples (Table 6.5) and range of ages and lengths available among the various dataset configurations. The comparisons of model fits also indicated all models (that converged) were nearly equally likely in predicting growth in length with age for each dataset configuration (very small differences in AIC values and Akaike weights among models within datasets; Table 6.13). This is not surprising given the broad overlap in lengths of adjacent age classes observed in the data (Figures 6.10 and 6.11), which suggests the relationship between age and length for American eel is not well defined and that age is a poor predictor of length for American eel.

6.3.2 SLYME (Sequential Life-table and Yield-per-Recruit Model for the American Eel)

6.3.2.1 Methods

In 2008, the American eel SASC applied a life-table model to available data to examine the effects of a maximum size limit on female spawner escapement and egg production. A copy of the report describing the methods and results is presented in Appendix 3.

6.3.2.2 Results

The SASC feels the SLYME model can be a useful tool for evaluating management options, as long as the assumptions and caveats associated with the model are taken into account.

6.4 Trend Analyses

6.4.1 Power Analysis

Power analysis was performed on all fishery-independent American eel surveys as a means to evaluate the precision of abundance indices.

6.4.1.1 Methods

Power analysis followed methods described in Gerrodette (1987) for both potential linear and exponential trends. A linear trend can be modeled as $A_i = A_1[1+r(i-1)]$ and an exponential trend as $A_i = A_1(1+r)^{i-1}$ where A_i is the abundance index in year i, A_1 is the abundance index in year 1, and r is a constant increment of change as a fraction of the initial abundance index A_1. The overall fractional change in abundance over n years can be expressed as $R = r(n-1)$.

If α and β are the probabilities of type 1 and type 2 errors respectively, the power of a linear trend (1 – β) assuming CV ~ $1/\sqrt{A}$ can be determined by satisfying the equation:

$$r^2 n(n-1)(n+1) \geq 12CV_1^2\left(z_\alpha + z_\beta\right)^2 \left\{1 + \frac{3r}{2}(n-1)\left[1 + \frac{r}{3}(2n-1) + \frac{r^2}{6}n(n-1)\right]\right\}$$

and the power of an exponential trend can be determined by satisfying the equation:

$$[\ln(1+r)]^2 n(n-1)(n+1) \geq 12\left(z_\alpha + z_\beta\right)^2 \left\{\frac{1}{n}\sum \ln\left[CV_1^2(1+r)^{i-1} + 1\right]\right\}$$

where CV_1 is an estimate of the coefficient of variation of the survey. For each of the surveys, the median CV of the survey was calculated over the entire time series of the survey and used as an estimate of CV_1. Power was then calculated for an overall change (R) of ±50% over a 10 year time period ($r = 0.056$) for both a linear and exponential trend.

6.4.1.2 Results

Median CVs of the surveys ranged from 0.04 to 1.02. Resulting estimates of power were a function of CVs with those surveys having low CVs having high power, and those surveys having high CVs having low power. Power values ranged from 0.11 to 1.00 (Table 6.14). For all surveys, there is greater power to detect a decreasing trend compared to an increasing trend which is a property of surveys whose $CV \sim 1/\sqrt{A}$. There was very little difference in power between linear and exponential trends.

The values of power presented in Table 6.14 can be interpreted as the probability of detecting a given linear or exponential trend of ±50% over a ten year period if it actually occurs. These values do not reflect a retrospective power analysis and a survey with low power value may still be capable of detecting a statistically significant trend if given enough years of data.

6.4.2 Mann-Kendall Analysis

6.4.2.1 Methods

The Mann-Kendall trend analysis is a non-parametric test for monotonic trend in time-ordered data (Gilbert 1987). The null hypothesis is that the time series is independent and identically distributed—there is no significant trend across time. The test allows for missing values and can account for tied values if present.

The Mann-Kendall test was applied to all local, regional, and coast-wide indices of relative abundance computed in this assessment. A two-tailed test was used to test for the presence of either an upward or downward trend over the entire time series. Trends were considered statistically significant at $\alpha = 0.05$.

6.4.2.2 Results

Local Indices

No significant temporal trends were detected among the YOY indices developed from the ASMFC-mandated recruitment surveys (Table 6.15). The Mann-Kendall test found statistically significant trends in eleven of the eighteen other individual indices evaluated; three were upward trends and eight were downward trends (Table 6.16). Significant downward trends were detected in all four indices from the Hudson River region. The test found significant downward trends in two of the three indices from the South Atlantic region. In the Southern New England and

Chesapeake Bay regions, both upward and downward significant trends were detected. The YOY index developed from the HRE Monitoring Program was the only YOY index in which a significant trend was detected, and the trend direction was down.

Regional Indices

Significant downward trends were detected in both the YOY and yellow-phase indices for the Hudson River region (Table 6.17). The Mann-Kendall test found a significant upward trend in the Chesapeake Bay region's yellow-phase abundance index. A significant downward trend was found in the yellow-phase index for the South Atlantic region.

Coast-wide Indices

The Mann-Kendall test detected one significant trend among the coast-wide indices (Table 6.17). The 30-year yellow-phase abundance index exhibited a significant downward trend.

6.4.3 Manly Analysis

A meta-analysis was conducted to determine if there was consensus among fishery-independent survey indices for a coast-wide decline in American eel. Meta-analysis is a statistical approach that combines the results from independent datasets to determine if the datasets are showing the same patterns. The meta-analysis techniques employed in this analysis are described by Manly (2001).

6.4.3.1 Methods

American eel surveys were grouped according to life stages (yellow vs. YOY) and one-tailed *p*-values from the Mann-Kendall test for trend were used in the meta-analysis (Manly 2001). Two meta-analysis techniques were used.

Fisher's method tests the hypothesis that at least one of the indices showed a significant decline through time. The test statistic was calculated as $S_1 = -2\Sigma\log_e(p_i)$, where p_i is the one-tailed *p*-value that tests for a negative trend from the *i*th index. The one tailed *p*-value is used because we are interested in whether the index has declined through time. If the null hypothesis is true for a test of significance, then the *p*-value from the test has a uniform distribution between 0 and 1, and if p has a uniform distribution, then $-2\log_e(p)$ has a chi-square distribution with 2 degrees of freedom. The test statistic, S_1, is then compared to a chi-square distribution with $2n$ degrees of freedom, where n equals the number of independent surveys considered.

The Liptak-Stouffer method tests the hypothesis that there is consensus for a decline supported the entire set of indices. The individual one-tailed *p*-values were converted to z-scores. If the null hypothesis is true for all indices, the z-scores are distributed as a normal random variable with mean equal to 0 and variance equal to $1/\sqrt{n}$. This allows for weighting the results from different indices differently. The test statistic is $S_2 = \Sigma w_i z_i/\sqrt{\Sigma w_i^2}$ where w_i is the weight of the *i*th index. In this analysis, the number of years of survey data was used as the weight for the *i*th index. A level of $\alpha = 0.05$ was used in meta-analyses for tests of significance.

6.4.3.2 Results

At least one of the indices for both life stages showed a decline though time (yellow eels: S_1= 174.82, $p < 0.01$; YOY eels: S_1 = 65.80, $p < 0.01$; Table 6.18). Also, there was consensus for a decline for both life stages through time (yellow eels: $S_2 = -6.29$, $p < 0.01$; YOY eels: $S_2 = -15.10$, $p < 0.01$).

6.4.4 ARIMA

Fishery-independent surveys for American eel can be quite variable, making inferences about population trends uncertain. Observed time series of abundance indices represents true changes in abundance, within survey sampling error, and varying catchability over time. One approach to minimize measurement error in the survey estimates is by using autoregressive integrated moving average models (ARIMA, Box and Jenkins 1976). The ARIMA approach derives fitted estimates of abundance over the entire time series whose variance is less than the variance of the observed series (Pennington 1986). This approach is commonly used to gain insight in stock assessments where enough data for size or age-structured assessments (e.g., yield per recruit, catch at age) is not yet available.

Helser and Hayes (1995) extended Pennington's (1986) application of ARIMA models to fisheries survey data to infer population status relative to an index-based reference point. This methodology yields a probability of the fitted index value of a particular year being less than the reference point [$p(\text{index}_t < \text{reference})$]. Helser et al. (2002) suggested using a two-tiered approach when evaluating reference points whereby not only is the probability of being below (or above) the reference point estimated, the statistical level of confidence is also specified. The confidence level can be thought of as a one-tailed α-probability from typical statistical hypothesis testing. For example, if the $p(\text{index}_t < \text{reference}) = 0.90$ at an 80% confidence level, there is strong evidence that the index of the year in question is less than the reference point. This methodology characterizes both the uncertainty in the index of abundance and in the chosen reference point. Helser and Hayes (1995) suggested the lower quartile (25th percentile) of the fitted abundance index as the reference point in an analysis of Atlantic wolfish (*Anarhichas lupus*) data. The use of the lower quartile as a reference point is arbitrary, but does provide a reasonable reference point for comparison for data with relatively high and low abundance over a range of years.

6.4.4.1 Methods

The purpose of this analysis was to fit ARIMA models to time series of eel abundance indices to infer the status of the population(s). The ARIMA model fitting procedure of Pennington (1986) and bootstrapped estimates of the probability of being less than an index-based reference point (25th percentile, Helser and Hayes 1995) were coded in R (R code developed by Gary Nelson, Massachusetts Division of Marine Fisheries). Index values were $\log_e$ transformed ($\log_e$[index + 0.01] in cases where "0" values were observed) prior to ARIMA model fitting. The reported probabilities of being less than the 25th percentile reference point correspond to 80% confidence levels. Only time series with 20 or more years of index values were used in ARIMA modeling because the 25th percentile reference point can be unstable with few observations.

6.4.4.2 Results

Twelve surveys contained 20 or more years of data and were used in ARIMA modeling (Table 6.19). Trends in fitted ARIMA values varied among regions, but were fairly consistent within regions. Surveys from the Chesapeake Bay and Delaware Bay/Mid-Atlantic Coastal Bays regions (Figures 6.12 and 6.13) showed no consistent increasing or decreasing trends. Also, the probability of the terminal year of surveys from the Chesapeake Bay and Delaware Bay/Mid-

Atlantic Coastal Bays regions being less than the 25th percentile benchmark was relatively low, ranging from 0.003 to 0.164. However, surveys from the Hudson River region tended to show consistent declines and probabilities of the terminal year being less than the 25th percentile benchmark ranged from 0.259 to 0.548 (Figure 6.14). There was only one survey from the South Atlantic region (NCDMF Estuarine Trawl Survey) and it showed a consistent decreasing trend and the probability of the terminal year being less than the 25th percentile benchmark was 0.308 (Figure 6.15).

6.4.5 Traffic Light Method

6.4.5.1 Methods

The Traffic Light approach was first introduced as a precautionary approach to fisheries management that can incorporate a variety of qualitative and quantitative information, or indicators, for describing the relative status of the stock and that is easily understood by stakeholders and non-technical personnel (Caddy 1998, 1999). Relevant information may include fishing mortality, biomass, recruitment, length and age at maturity, and spatial distribution (Halliday et al. 2001). The selected indicators are assigned colors in order to normalize the different indicators to a common scale; this process is called scaling. A common approach is to employ a three-color system in which indicator values in each year are assigned a green, yellow, or red 'signal' based on the state of the indicator relative to stock health. Typically, the color green is indicative of a positive stock condition, yellow is indicative of an uncertain or transitioning stock condition, and red is indicative of an undesirable stock condition. The ASMFC has incorporated a grayscale version of the Traffic Light approach into the assessment of American lobster stocks in order to provide a simple characterization of the status of individual stocks (ASMFC 2006c, 2009).

The Traffic Light approach was applied to all individual, regional, and coast-wide indices of relative abundance computed in this assessment. The strict scaling method, one of the simplest scaling methods, was used to assign each annual index value to one of three color categories—white, gray, or black which replace the traditional green, yellow, or red. The 25th and 75th percentiles of each index series were calculated in order to determine color boundaries. Each annual value within an index was compared to the percentiles computed for that series. If an index value was greater than the 75th percentile for the time series, that value was assigned the color white. If an index value was less than the 25th percentile for the time series, that value was assigned the color black. Index values that were less than or equal to the 75th percentile and greater than or equal to the 25th percentile were assigned the color gray. Note that the assignment of color is sensitive to the choice of color boundaries.

6.4.5.2 Results

Local Indices

The Traffic Light representation of the YOY indices demonstrates variability in recruitment trends within and among survey sites (Table 6.20). The Traffic Light analysis suggests that recruitment was relatively high in 2001 at most sites. The year 2009 was characterized by moderate to relatively low recruitment at the majority of the survey sites.

With the exception of the CTDEP Electrofishing Survey and the Beaufort Inlet Ichthyoplankton Survey indices, all indices in the Southern New England, Hudson River, and South Atlantic regions show a progression from white to black signals throughout their time series (Table 6.21). In contrast, the three longest indices from the Delaware Bay/Mid-Atlantic Coastal Bays region exhibit a progression from black to white signals. Indices from the Chesapeake Bay region demonstrate mostly black signals during the 1990s.

The VIMS Juvenile Striped Bass Seine Survey long time series index of yellow-phase abundance exhibited relatively high abundance during 1967 to 1972—the earliest years of that index time series (Table 6.21). All years from 1973 to 1978 were assigned black signals for the PSEG Trawl Survey index of elver and yellow-phase abundance. Mostly white signals are observed for the 1980s for yellow-phase abundance indices derived from the Western Long Island Sound Survey, the HRE Monitoring Program, the NYDEC Alosine Beach Seine Survey, and the NYDEC Striped Bass Beach Seine Survey. The MDDNR Striped Bass Seine Survey index of yellow-phase abundance is characterized by black signals for all years from 1990 to 1996. Abundance of yearling and older American eels appeared relatively low during the late 1990s through the 2000s based on the HRE Monitoring Program. The NYDEC Striped Bass Beach Seine Survey elver and yellow-phase index also suggests abundance was relatively low during most of the 2000s.

Regional Indices

The Hudson River region indices of YOY and yellow-phase American eel abundance exhibit mostly white signals in the early years of their time series (Table 6.22). All but one year from 1974 to 1980 were assigned white signals for the Hudson River region YOY index. All years from 1981 to 1987 were characterized by white signals for the Hudson River region yellow-phase index. The Hudson River region YOY and yellow-phase indices show black signals during most of the 2000s. The Southern New England YOY index progress from gray and red to white signals over its time series while the South Atlantic YOY and yellow-phase indices transition from mostly white to gray and black signals. The Chesapeake Bay index of yellow-phase abundance was assigned mostly black signals during 1990 to 1995 and mostly white signals during 2003 to 2009.

Coast-wide Indices

The coast-wide YOY indices are mostly white during the early years of their respective time series and transition to mostly gray and black signals throughout the rest of the time series (Table 6.22). The 30-year and 40-plus-year indices of coast-wide yellow-phase abundance show white signals during most of the 1980s. All three coast-wide indices of yellow-phase abundance suggest moderate to relatively low abundance of yellow-phase American eels during the early to mid-1990s.

6.5 SEINE (Survival Estimation In Non-Equilibrium Situations)

The Survival Estimates in Non-Equilibrium (SEINE) model was used in exploratory analyses to estimate mortality rates from changes in eel length. The SEINE model is derived from the Beverton and Holt Mortality Estimator that is based on the premise that if a fish population is at equilibrium the mean length will be inversely proportional to the population mortality rate. The Beverton and Holt Mortality Estimator requires equilibrium conditions because changes in length likely will occur gradually after changes in mortality. The assumptions of equilibrium can be difficult to satisfy for many situations involving overfishing when limited fish population data are available. Gedamke and Hoenig (2006) developed the SEINE model from the Beverton and

Holt Mortality Estimator specifically to allow the estimation of instantaneous total mortality from length data in non-equilibrium conditions. The SEINE model requires only von Bertalanffy growth parameters (K and L_∞), length of first capture (L_c, smallest size of capture by fishery or sampling gear), and annual mean length larger than L_c. Regional von Bertalanffy parameters were estimated from age-length data for this assessment (see section 6.3.1, this report). SEINE analyses were made using Fisheries Methods in R (Nelson 2009).

The application of the SEINE model to eel length datasets did not produce mortality estimates that were useful for this assessment. All U.S. fishery-independent surveys with eel length data were reviewed and few had long-term (>10 years) random sampling of eel length. Secondly, the SEINE model requires an input for the years when a fishery or management event would have caused a shift in mortality. None of the data series had both long term length data available and actions expected to cause mortality shifts. Finally, the life history of eel could limit the suitability of the SEINE model given their sexual dimorphism, variable age at length, semelparity, and variable sex ratio among watersheds. The survey with perhaps the most potential is the HRE Monitoring Program ichthyoplankton survey with > 20 year duration and a significant management event (fishery was closed due to tissue contamination); however, the length data were not available for this assessment.

6.6 DB-SRA (Depletion-Based Stock Reduction Analysis)

6.6.1 Methods

Model Description

Depletion Based Stock Reduction Analysis (DB-SRA) is a modification of the Stock Reduction Analysis (SRA) methodology that can be used in data poor situations. SRA was first introduced by Kimura and Tagart (1982) and improved by Kimura et al. (1984). Using catch data and a time series of abundance, the model strives to determine stock size and recruitment rates over time that could have produced the observed population trend given the harvest information. The original model was not widely accepted because it provided only a single, exceedingly unlikely, trajectory of stock size and recruitment (Walters et al. 2006). Walters et al. (2006) improved the method by incorporating stochasticity through Monte Carlo simulation of input parameters to produce a distribution of potential stock sizes over time, providing the ability to describe the statistical probability of biomass and MSY-based reference points.

While Walters et al. (2006) promote stochastic SRA as a useful complement to traditional assessment methodologies, many species do not have sufficient data to run a traditional model or even SRA. In order to provide management advice in these data poor situations, a number of methodologies have recently been developed. One such model is Depletion-Corrected Average Catch (DCAC), an extension of the potential yield formula that can provide useful estimates of long term sustainable yield (MacCall 2009). Input requirements are limited to a time series of observed harvest, an estimated stock depletion level, and biologically based life history parameters (M, F_{MSY}:M [hereafter referred to as the F-ratio], B_{MSY}:K [or B-peak]) and their associated uncertainty values. Monte Carlo distributions of the input parameters are developed and used in conjunction with the harvest data to derive a probability distribution of long term sustainable yield (MacCall 2009).

Depletion-Based Stock Reduction Analysis was first introduced by Dick and MacCall (2011), borrowing aspects of SRA (Kimura and Tagart 1982; Kimura et al. 1984; Walters et al. 2006) and DCAC (MacCall 2009). A full description of the model is provided in Dick and MacCall (2011) but is summarized below.

Implementation of traditional SRA requires a time series of abundance (absolute or relative) which is generally lacking in data poor situations. DB-SRA relaxes that requirement by utilizing a distribution of assumed relative abundance (percent stock depletion) in a recent year (Dick and MacCall 2011). Other data inputs include a time series of harvest, age at maturity, and the same suite of biologically based life history parameters used in DCAC (M, F-ratio, and B-peak). A major assumption of the model is that the stock is at carrying capacity (K) at the beginning of the time series.

Implementation of the model is through a delay difference biomass model:

$$B_t = B_{t-1} + P(B_{t-a}) - C_{t-1}$$

where B is biomass, P is production, a is the median age at maturity, and C is harvest weight. Any production function can be used, but the current model is based on a hybrid of the Pella-Tomlinson-Fletcher and Schaefer models. Dick and MacCall (2011) argue that this parameterization best captures production rates at all levels of biomass, and the hybridization method is fully described in their manuscript. A solver routine is required to iteratively solve for K such that recent biomass relative to K satisfies the input assumed depletion level.

Outputs of the model include a biomass trajectory and estimates of a number of "leading parameters" that are directly useful to management, including K, MSY, B_{MSY}, and F_{MSY}. Statistical distributions of each of these outputs are achieved through Monte Carlo simulation of uncertainty in input parameter values.

Model Development

For the 2011 eel stock assessment, a version of DB-SRA was coded in the R software language, version 2.13.0 for Windows (R Development Core Team 2011), based on the pseudo-code provided in Appendices A and B of Dick and MacCall (2011). The resulting code was ground truthed by replicating (harvest data, input parameter means, uncertainty levels, and error distributions) the NMFS Southwest Fisheries Science Center (SWFSC) DB-SRA model for bank rockfish (*Sebastes rufus*) and comparing results to the SWFSC results used to establish overfishing limits for the species (E.J. Dick, NMFS SWFSC, pers. comm.). Although the results were not exactly the same, biomass trends and production curves followed similar patterns in similar scales.

Input Data

American eel commercial harvest data collected from 1950 to 2010 were compiled as described in section 5.1.1. Prior to 1950, harvest estimates were taken from historical NMFS annual reports (1889–1938; NOAA Central Library Data Imaging Project) and from the NMFS redbook series (1937–1950). Missing data points between 1880 and 1923 were generated using the following process.

1. Calculate the average reported harvest between 1880 and 1923.

2. For each year harvest was reported, calculate the difference between reported harvest and the mean harvest.
3. For years without harvest reports, the average harvest in up to three years of available data prior to and succeeding the missing value (max six years of available data) was calculated and added to a randomly sampled harvest residual.
4. Repeat step 3 one hundred times for each missing value.
5. Estimate harvest as the average of the 100 iterations in a given year.

The resulting harvest trend is shown in Figure 6.16.

Given the lack of knowledge in eel population characteristics, input parameters for preliminary runs were selected based on general knowledge of production theory and proxy information from other species. In addition, because of this lack of knowledge, as well as the potential for latitudinal trends in parameter values, uncertainty in the inputs was modeled using a uniform distribution for the Monte Carlo simulations. Natural mortality, M, was assumed to range from 0.15 to 0.25. This range captures the variability in maximum age reported from northern and southern portions of the U.S. population and is consistent with available data (section 2) and other analyses by the Technical Committee (e.g., SLYME). An F-ratio of 1.0 is used commonly when no other information is available, so this was selected as a median value for the F-ratio. The median F-ratio of 0.80 used by Dick and MacCall (2011) was selected as a lower bound in the eel model, and an upper bound was selected equidistant from the median (F-ratio range 0.80 to 1.20). The range for B-peak of 0.25 to 0.50 was selected because it includes both the default Gompertz and Schaefer values for B_{MSY}:K (~0.37 and 0.50, respectively) and incorporates the median values used by Dick and MacCall to represent two species groups (0.25 for flatfish, 0.40 for rockfish) with different life history strategies that potentially bracket that of eel.

The input range for the ratio of recent biomass to K (referred to as B-ratio) in preliminary runs was developed in a stepwise manner. The oldest available index data for eel are from the late 1960s. In preliminary runs, the DB-SRA biomass from around 1970 was compared to biomass at K to estimate depletion level in 1970. Then, ratios of survey index values in recent years relative to index values around 1970 were developed for a number of surveys (MD seine, VIMS seine, HRE Monitoring). Ratios of B_{1970}:K and I_{Recent}:I_{1970} were multiplied to estimate depletion level in recent years. This method provided estimates of biomass in 2010 that were approximately 3 to 10% of preliminary K values.

The range for B-ratio used in the preferred models was developed slightly differently than for preliminary runs, as a result of more appropriate data being available when final model runs were performed. Rather than using individual indices, the B-ratio range was developed using results of the coast-wide yellow eel GLMs for 20-year and 30-year time series. These indices incorporate data from multiple regions and more likely represent the overall coast-wide trend in abundance than a single index or the 40+ year index which only includes data from a single region (Chesapeake Bay). The fitted 20-year and 30-year indices were each smoothed using a three-year average to reduce variability, and the relative change in index values between the 1991–1993 average and the 2008–2010 average was calculated. For both indices, abundance increased approximately 10% over the specified time period. Results from preliminary runs investigating different B-ratio scenarios (see *Sensitivity Analyses* section below) indicated that a median B-ratio of approximately 10% produced a similar biomass trend in recent years as the 20-year and

30-year indices. To account for uncertainty in the information provided by the indices, a *B*-ratio range of 5–15% was used in the final runs.

Sensitivity Analyses

One major assumption of DB-SRA is that biomass in the first year of the time series is at an unfished level. In addition, Wetzel and Punt (2011) found DB-SRA to be sensitive to incorrectly specified input values, particularly the ratio of recent stock size to *K*. Finally, as described in section 2, life history characteristics of American eel in U.S. waters differ among the sexes and follow latitudinal trends, making selection of input parameters difficult.

To investigate the sensitivity of the model to potentially miss-specified input parameters, a number of sensitivity runs were conducted (Table 6.23). Sensitivity runs took two forms. First, a set of deterministic runs was conducted across a range of values for each input parameter. A total of 108 runs were conducted, one for each combination of four values of *M*, three values of $F_{\mathrm{MSY}}:M$, three values of $B_{\mathrm{MSY}}:K$, and three values of $B_{\mathrm{Recent}}:K$. These runs provided insight into model performance and directional effects of the different input parameters on the results.

The second form of sensitivity consisted of eight runs using input ranges detailed above for *M*, *F*-ratio, and *B*-peak but varying harvest levels, the harvest time series, or *B*-ratio. These runs provided insight into the sensitivity of model results to incorrect input data and violation of the assumption that the stock was at carrying capacity at the beginning of the time series.

Alternate Model Framework

The original DB-SRA model was constructed under the assumption of a single level of *M* for the entire time series. This is likely an invalid assumption given changes in environmental and climatic conditions, predation, parasitism, habitat availability, and other factors. For example, it is well known that dam construction in the U.S. has limited upstream habitat availability to diadromous species such as eel. As such, the assumption of single *M* over time is likely violated. To investigate potential effects of decreasing habitat availability, an alternate version of DB-SRA was coded that incorporated two stanzas of natural mortality, *M*. Dam construction in the U.S. occurred primarily in the years following World War II, peaking in the 1960s (Water Encyclopedia, http://www.waterencyclopedia.com/Da-En/Dams.html). In the two-stanza model, *M* was assumed to be constant at relatively low levels (0.15–0.25) from 1880 through 1969. The second *M* stanza began in 1970, at which time *M* increased in a single step and was assumed constant for the remainder of the time series.

This methodology assumes the eel population can support a certain level of total mortality (e.g., Z_{MSY}) that is constant through time. Dam construction is assumed to result in an increase in natural mortality, which would require a decrease in the fishing mortality level that produces MSY.

Inputs to the two-stanza model were the same as for the one-stanza model for initial *M*, initial *F*-ratio, and *B*-peak. Increased *M* in the second stanza results in a decrease in the *F*-ratio, producing a lower fishing mortality threshold. Sensitivity runs were conducted investigating alternate harvest scenarios, *B*-ratios, the timing of the *M* increase, and the extent of the change in *M* (Table 6.23). Estimates for *B*-ratio in the two-stanza model were chosen as described above for the one-stanza model.

Potential Biases

There are a number of assumptions regarding the model and inputs that, if violated, could affect the output of the model. Many of these were investigated through sensitivity analyses as described above, including incorrect harvest estimates, initial biomass at carrying capacity, the ratio of recent biomass to K, and single M over time assumptions. A number of other potential biases are discussed below.

One of the model input requirements is the median age at maturity. Maturity was assumed to occur at age 8 for eels. This value was selected as a compromise of the differences between sexes and across latitude. No "official" sensitivity runs were conducted regarding this parameter; however, preliminary runs comparing maturity at age 4 and 8 showed K and MSY were approximately 10–20% higher for age 8 than age 4 (results not shown). Incorrect specification of the age at maturity could potentially bias the results.

Another issue concerning the age at maturity is that the DB-SRA models only mature biomass, and therefore assumes harvest is of mature animals only. Given eels' catadromous and semelparous life history, nearly all fishing mortality for the species occurs prior to maturity, and this assumption is clearly violated. For the eel model, biomass and associated parameters are in terms of fishable biomass. In the population biomass equation, production in a given year is based on the stock biomass eight years previous (median age at maturity = 8). Eels generally recruit to the fishery at around age four (K. Whiteford, MD DNR, pers. comm.; J. Clark, DNREC, pers. comm.) and undergo four years of fishing mortality before maturity. The production delay of eight years is still valid, as it takes four years for fish that enter the fishery to become mature and an additional four years until the new cohort recruits to the fishery. Violation of the assumption of maturity does affect the production function. If total mortality between ages 4 and 8 were constant, the result would be simply a shift in the production curve. Because mortality is not constant, the relationship between age-4 biomass and age-8 biomass varies. The directional effect this has on production at a given biomass (over vs underestimate) depends on whether harvest between age 4 and 8 is above or below average, as well as the biomass relative to B_{MSY} (i.e., ascending or descending limb of the production curve).

Another concern of implementing this model for the U.S. eel population is that the U.S. encompasses only a portion of the species range. Trends and reference points developed through this model are therefore only relevant to the U.S. fishery and population. Harvest pressure, habitat availability, and other factors that occur outside the U.S. were not considered in the model, but because of the panmictic nature of the stock, these factors could affect model performance and/or influence the ability of the U.S. to achieve its management goals. Preliminary runs were conducted using combined U.S. and Canada harvest, but are not described here for the reasons given in section 5.

One final concern is that the DB-SRA relies almost entirely on catch data and does not account for the contribution of unfished areas to production. The degree to which fished and unfished areas contribute to the entire population's production is unknown.

Sensitivity runs were conducted to investigate the effect of possible error in early harvest estimates on the model. It is assumed that recent estimates are known without error; however, inaccurate harvest data in the model would likely lead to biased model results, particularly if there is a consistent directionality in the error.

The minimal data requirements for DB-SRA require that all input values be carefully considered, based on biologically sound information, and supported by available data where possible. This is because it is easy to “lead” the model to a given result based on input values. For example, if the B_{MSY}:K ratio is set at 0.40 and the ratio of B_{Recent}:K is input as 0.30, the model will indicate that the stock is not overfished in the terminal year (i.e., assuming $B_{\mathrm{Target}} = B_{\mathrm{MSY}} = 0.4K$ and $B_{\mathrm{Threshold}} = ½B_{\mathrm{Target}} = 0.2K$, current biomass is $0.3K$ and is therefore not overfished). All attempts were made to ensure inputs for the eel model are biologically sound and based on available data. As noted above, a number of sensitivity runs were conducted to investigate model sensitivity to miss-specification of inputs. However, results of this model are conditional on the inputs, and any error in the input parameters could carry through to the results.

6.6.2 Results

Single *M* Stanza Model

The preferred single *M* stanza model produced a median carrying capacity estimate of approximately 18,200 mt (inter-quartile range 17,300-19,200 mt; Figure 6.17). Biomass dropped quickly in the early years of the time series, falling to less than 5,000 mt within the first decade. Between 1890 and 1934, biomass never exceeded 3,500 mt and fell below 1,000 mt during 1902–1905 and 1932–1934. Biomass began a gradual increase in 1935, rising to more than 5,000 mt by 1969 and a peak of 5,400 mt in 1974. Subsequent increases in harvest due to the export market reduced biomass to less than 2,000 mt by the early 1980s and below 1,000 mt once again in 1997. Since 1998, biomass has been increasing gradually, with a median estimate of 1,817 mt (inter-quartile range 1,355–2,276 mt) in the terminal year of 2011.

Median biomass at MSY was estimated at approximately 6,770 mt, with a maximum sustainable yield of 1,057 mt (Table 6.24; Figure 6.18). MSY is attained at a median annual exploitation rate of $u_{\mathrm{MSY}} = 0.158$. Observed annual exploitation rate in recent years averaged approximately $u = 0.221$, based on the median biomass estimates.

Double *M* Stanza Model

Median carrying capacity for the preferred double *M* stanza model of 18,275 mt (inter-quartile range 17,365–19,325 mt) was very similar to that of the single *M* model (Figure 6.19). In addition, the biomass trajectory followed closely that of the single *M* model, but at slightly higher median values and wider inter-quartile range. The differences were most apparent from around 1930 to 2000. Median biomass increased from a low of approximately 1,025 mt in 1933 to a relative peak of 9,520 mt in 1969. This was reduced to a low of 1,305 mt by 1997, but has since recovered to approximately 1,846 mt in 2011 (inter-quartile range 1,380–2,310). As opposed to the single *M* stanza model, the double *M* stanza model displayed a recent peak biomass in the late 1960s/early 1970s which corresponds with peaks observed in fishery-independent surveys from the Chesapeake Bay region during the same time period.

Median biomass at MSY was estimated at approximately 6,820 mt (Table 6.24; Figure 6.20). In the early years (lower *M*) maximum sustainable yield was estimated at 1,060 mt, but this dropped to 810 mt due to increased *M* since 1970. Median annual exploitation rates that achieve MSY were estimated at $u_{\mathrm{MSY}} = 0.159$ in the early period and $u_{\mathrm{MSY}} = 0.123$ in recent years. Average observed annual exploitation rate since 2008 was approximately u = 0.204 based on the median biomass estimates.

Sensitivity

Results of deterministic sensitivity runs were generally consistent with generic production theory. For example, increases in natural mortality resulted in a lower *K* but higher MSY. Similar results were observed for *F*-ratio and *B*-peak within the parameter ranges evaluated. Wetzel and Punt (2011) found that overestimating *B*-ratio led to overestimates of the harvest level in all cases. The deterministic sensitivity runs confirmed that increasing the *B*-ratio value often increased estimated *K* and MSY values, but this phenomenon abated at higher combinations of *M* and *F*-ratio, which included the range of values used in the preferred runs. Deterministic sensitivity runs also indicated that the model became unstable at higher combinations of *M*, *F*-ratio, and *B*-peak, often producing estimates of *K* in the millions of metric tons.

For the single stanza model, stochastic sensitivity runs indicate that increasing harvest early on in the time series or extending the time series prior to 1880 generally led to an increase in estimated carrying capacity and MSY. Runs starting in 1925 and 1970 had lower carrying capacity relative to runs starting in1880, but the 1970 run had higher *K* than the 1925 run. These results are possibly due to the higher harvest levels in the early years of the time series for the 1970 run. Increasing the *B*-ratio level had minimal effect on *K* and MSY at the ranges evaluated. This is contrary to the results of Wetzel and Punt (2011), although the deterministic sensitivity confirmed their findings within a different range of input values.

Stochastic sensitivity results for the double *M* stanza model were similar to those for the single stanza model. Decreasing harvest early on lowered *K* and MSY, and these estimates were not sensitive to the input *B*-ratio over the ranges evaluated. Changing the timing of the change in *M* from 1970 to 1960 had minimal effect on the outputs. The double *M* model had similar estimates of *K* as the single *M* model, and initial estimates of MSY were also similar; however, MSY decreased by approximately 24% after the increase in *M*.

6.7 Age-structured Production Model

The age-structured production model constructed to assess eel in the Potomac River by Fenske et al. (2011) was modified for use in the Delaware Bay and in Maryland waters of the Chesapeake Bay. The Potomac River model estimates fishing mortality and biomass dynamics by incorporating sex- and age-specific maturation mortality and selectivity in a surplus production model framework. Recruitment to the fishery is estimated freely in each year using an index of recruitment to the first age in the model and age-specific catch information. Catchability can be assumed constant or time-varying using a random walk, white noise, effort-dependent, or density-dependent catchability model. The Fenske et al. (2011) model was pursued as a method for obtaining population estimates and biological reference points on a regional basis without assuming an explicit stock-recruitment curve.

6.7.1 Methods

In implementing this model for Delaware Bay, the code was modified to fit multiple years of unsexed age composition data in both the fishery and survey for fish ages 2 to 12. Survey and catch data were available from 1982 to 2009. The Delaware trawl survey was split into an index of age 2s (fish <= 290 mm) and an overall index of abundance (fish > 290 mm). Unsexed age composition data were available from the survey from 1997 to 2009. Unsexed aged catch information was available from 2003 to 2009. Effort in the form of pots per day was used to estimate effort-dependent time-varying catchability between 1999 and 2009. Maturity- and weight-at-age were borrowed from the Potomac River model. Selectivity was calculated using observed proportions caught at age for fully selected ages. For ages that are not fully selected, the difference between observed and back-calculated (predicted) catch at age was used to approximate selectivity.

In implementing the Potomac River model for Maryland waters, the code was modified to fit multiple years of sexed age composition data for fish ages 2 to 12. The Maryland seine survey was used as an index of age-2 fish and annual CPUE from the fishery was used as an index of overall abundance. Survey and catch data spanning 1992 to 2010 were used in the model. A sex-specific catch-at-age matrix was generated using age and length sampling information from 1997 to 2010. Effort in the form of pot days was used to estimate effort- and density-dependent time-varying catchability between 1992 and 2010. Maturity-, selectivity-, and weight-at-age were calculated from Maryland's eel sampling program data.

6.7.2 Results

Despite numerous attempts to reconfigure and tune the Delaware Bay model, the model did not converge on a stable solution. We suspect that the lack of sex-specific information in the catch and survey data and the lack of contrast in available survey trends hindered our ability to achieve convergence. The Maryland model repeatedly converged on a solution that tightly fit the commercial CPUE index and the catch-at-age, but did not fit the recruitment index at all. Depending on the form of time-varying catchability estimated, a much smaller plus class was required in order to achieve convergence. We suspect the Maryland seine survey is not an adequate index of age-2 animals in the population and that a lack of information about the age and maturity structure of the yellow eel population may limit application of this model to the Maryland eel population. The SASC did not feel comfortable recommending this model for management given its reliance on a commercial CPUE index and lack of adequate fit to a recruitment index.

7 STOCK STATUS DETERMINATION

7.1 Status Determination Criteria

Reference points for determining the stock status of American eel in the U.S. were developed using the DB-SRA model[2], a recently developed assessment methodology for use in data poor situations (see section 6.6, this report; Dick and MacCall 2011). Although DB-SRA is not a traditional data-rich assessment methodology, there is substantial support for its use in management. The method received positive feedback during a formal peer review of data-poor assessment methods (SWFSC 2011), and it is the principle method of estimating reference points on the U.S. west coast for data-poor species (E.J. Dick, NMFS SWFSC, pers. comm.).

The DB-SRA was run assuming a single *M* over time and also run assuming a one-time change in *M* over time (the double *M* or two stanza model). Results of the single and double *M* stanza models were very similar; however, the Technical Committee preferred the double *M* model as it

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[2] Note that DB-SRA reference points were not accepted for management use by the Peer Review Panel. The TC now recommends stock status be declared depleted based on trend analyses and biomass trends estimated by the DB-SRA. Refer to the Preface and the Peer Review Report for more information.

takes into account changes in habitat availability that may have possible implications for the stock and fishery. ~~The reference points are therefore based on the results of the double *M* model.~~

~~The U.S. American eel resource will be considered overfished if stock biomass falls below the biomass threshold ($B_{\text{Threshold}}$), which is defined as half of the biomass that produces maximum sustainable yield (B_{MSY}). The double *M* DB-SRA model estimated the biomass target at $B_{\text{Target}} = B_{\text{MSY}}$ = 6,820 mt (inter-quartile range 6,095–7,579 mt; Table 6.24; Figure 6.21), resulting in a median threshold value of $B_{\text{Threshold}}$ = 3,410 mt.~~

~~American eels in the U.S. will be considered to be experiencing overfishing if the exploitation rate exceeds the exploitation level that produces maximum sustainable yield (u_{MSY}). The double *M* DB-SRA model estimated this value at u_{MSY} = 0.159 (inter-quartile range 0.143–0.175; Table 6.24; Figure 6.22) for the early period but, since 1970, the estimate decreased to u_{MSY} = 0.123 (inter-quartile range 0.108–0.138).~~ [3]

Note that Wetzel and Punt (2011) found that DB-SRA often miss-specified harvest limits in their simulation study; however, they found that in most instances the model underestimated true values, suggesting that the method is conservative. The authors also stated that conservative estimates are often preferred in data-poor situations that are associated with a high degree of uncertainty. For these reasons, the Technical Committee is comfortable proposing the above mentioned reference points for the U.S. American eel population.

7.2 Current Stock Status

~~The double *M* DB-SRA model estimated that median biomass for U.S. American eels in 2011 was 1,846 mt (Figure 6.21), which is approximately 54% of the overfished reference point ($B_{\text{Threshold}}$ = 3,410 mt). Exploitation rate in 2010, relative to the median biomass level, was estimated at u_{2010} = 0.215 (Figure 6.21), which exceeds the overfishing reference point by about 75% (recent u_{MSY} = 0.123). Based on these results, the U.S. American eel population is overfished and overfishing is occurring.~~[4]

8 DISCUSSION & CONCLUSIONS

Assessment of the American eel population is complex. Life history traits such as size, age, density, growth rate, sex ratio, and maturity exhibit both spatial and temporal variation throughout the species' range. The GLM analyses performed here indicate that the impact of environmental variables such as water temperature, salinity, and discharge on local abundance is similarly variable. In the U.S., all life stages are subject to fishing pressure, and the degree of fishing also varies through time and space. In addition to fishing, other factors that may negatively affect the eel population include habitat loss and alteration, productivity and food web alterations, predation, turbine mortality, changing climatic and oceanic conditions, toxins and contaminants, and disease (these factors are discussed in detail in the literature; see Haro et al. 2000, GMCME 2007, and DFO 2011c for general information regarding the potential impacts of these factors). As with the fisheries, the impact of these factors at local scales is not well understood, and the impact on the population as a whole, if any, is even less understood.

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[3] Note that DB-SRA reference points were not accepted for management use by the Peer Review Panel. The TC now recommends stock status be declared depleted based on trend analyses and biomass trends estimated by the DB-SRA. Refer to the Preface and the Peer Review Report for more information.

[4] See footnote 3.

The assessment is further complicated by limitations in the available data. Incomplete or underreporting of fisheries landings is a common concern in stock assessments. The FMP for American Eel addressed this issue by providing guidelines for standardized and consistent reporting of commercial fisheries data (ASMFC 2000a); however, the FMP was adopted in 2000 and American eels have been harvested for over a hundred years so a considerable portion of the landings history is questionable. Illegal poaching provides another data limitation. Though glass eel fisheries are limited to a few locations, increases in the value of the glass eels (>$300/lb) often leads to increased poaching in areas where these fisheries are prohibited, resulting in undocumented losses that may be significant. Additionally, there are few reliable long-term fishery-independent data sources available in the U.S. for characterizing trends in American eel abundance. Those that are available likely reflect local trends and were not designed to target eels. Of all the U.S. data sources that are available, the majority originate from the Delaware Bay/Mid-Atlantic Coastal Bays and Chesapeake Bay regions, which presents a spatial bias in the data. Finally, there are currently no standardized programs for monitoring escapement, which makes it difficult to base management on a desired escapement level as is currently done in Europe to facilitate the recovery of European eels (EC 2007).

The data evaluated in this assessment provide evidence of declining or, at least, neutral abundance of American eel in the U.S in recent decades. All three trend analysis methods (Mann-Kendall, Manly, and ARIMA) detected significant downward trends in numerous indices over the time period examined. The Mann-Kendall test detected a significant trend in the 30-year yellow-phase abundance index (Table 6.17). The Manly meta-analysis showed a decline in at least one of the indices for both yellow and YOY life stages (Table 6.18). Also, there was consensus for a decline for both life stages through time. Both the ARIMA and Mann-Kendall analyses indicate decreasing trends in the Hudson River and South Atlantic regions (Tables 6.17 and 6.19). In contrast, survey indices from the Chesapeake Bay and Delaware Bay/Mid-Atlantic Coastal Bays regions showed no consistent increasing or decreasing trends. Overall, however, the prevalence of significant downward trends in multiple surveys across the coast is cause for concern. In addition, historical catch-based results from this assessment's DB-SRA showed a decline in stock biomass coast-wide from the mid- to late 1990s, and there has been evidence of a slight increase since the late 1990s.

The DB-SRA results indicate that the American eel resource in the U.S. is overfished and overfishing is occurring relative to MSY-based reference points given the assumptions made (particularly the depletion level and B_{MSY}/K). The use of the term "overfished" suggests that fishing is the primary reason for the currently reduced levels of biomass;[5] however, it is important to recognize that multiple sources of mortality have been contributing to the reduced biomass levels, and it is difficult, if not impossible, to determine the degree to which different mortality sources have negatively impacted the stock over time. Significant levels of harvest in the 1970s is considered a major factor contributing to the current low biomass levels, but other factors such as habitat loss, predation, and disease have also played a role. Although fishery landings and effort in recent times have declined in most regions (with the possible exception of the glass eel fishery), current levels of fishing effort may still be too high given the additional stressors affecting the stock such as habitat loss, passage mortality, climate change, and disease.

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[5] Note that DB-SRA reference points were not accepted for management use by the Peer Review Panel. The TC now recommends stock status be declared depleted based on trend analyses and biomass trends estimated by the DB-SRA. Refer to the Preface and the Peer Review Report for more information.

Fishing on all life stages of eel, particularly YOY and out-migrating silver eels, could be particularly detrimental to the stock (see Appendix 3), especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled.

In 2000, the ICES Working Group on Eels met to discuss the status and conservation of American eels (ICES 2001). The group concluded "that reductions in habitat, declining or neutral trends in abundance, severe decline in abundance in northern areas, continuous exploitation and unknown oceanographic effects support the adoption of the Precautionary Approach in management." The precautionary approach calls for the assumption that a stock-recruitment relationship exists. For American eels, recruitment to a particular area is independent of the spawners that came from that area. Due to the panmictic nature of the species and because the relative contribution to the spawning stock from different regions is unknown, there is a need for international coordination of management efforts (Petersen 1997; ASMFC 2000a, 2002, 2006a; Haro et al. 2000; ICES 2001; Goodwin and Angermeier 2003; Cairns and Casselman 2004; DFO 2007, 2011a; Casselman and Cairns 2009; Vélez-Espino and Koops 2010; Fenske 2011). Currently, there is no Canada-wide assessment for American eel, but status reviews have been performed for regions within Canada (e.g., Newfoundland and Labrador: Veinott and Clark 2011; Ontario: Mathers and Pratt 2011, Pratt and Mathers 2011; southern Gulf of St. Lawrence: Cairns et al. 2007; also see DFO 2011c). In 2010, a scientific peer review of information on American eel in eastern Canada was held in response to a request from COSEWIC for an updated report and to a request from Canada's DFO Ecosystem and Fisheries Management (DFO 2011c).

Following completion of the Canadian regional and U.S. stock assessments, the American eel resource would benefit from a coast-wide assessment that included both Canadian and U.S. data sources. Recent Canadian efforts to map eel habitat, dam locations, and areas of concentrated fishing pressure along the Atlantic coastline may allow for an assessment that accounts for regional differences in habitat availability and sources of mortality. ~~In conclusion, the status of the American eel resource in the U.S. is overfished with overfishing occurring~~[6] due to a combination of fishing pressure on all life stages, other anthropogenic effects such as habitat loss and passage mortality, disease, and climate changes leading to shifting oceanographic conditions. Evidence of a decline in the American eel population throughout the species' range is further supported by the literature (for example, see Castonguay et al. 1994a; Jessop 1997; Petersen 1997; Richkus and Whalen 1999, 2000; ASMFC 2000a, 2006a; Haro et al. 2000; Beak International 2001; ICES 2001; Anonymous 2003; Casselman 2003; Geer 2003; Wirth and Bernatchez 2003; Cairns and Casselman 2004; Verreault et al. 2004; DFO 2005, 2006, 2007, 2009a, 2009b, 2010, 2011a, 2011b, 2011c; COSEWIC 2006; Casselman and Marcogliese 2007; Casselman and Cairns 2009; Fenske 2011; Mathers and Pratt 2011; Pratt and Mathers 2011; USFWS 2011; Veinott and Clarke 2011). Management efforts to reduce mortality on American eels in the U.S. are warranted. Collaboration with Canada to cooperatively monitor, assess, and manage American eels should provide a more complete and accurate picture of the resource. A formal Memorandum of Understanding between the ASMFC and the Great Lakes Fisheries Commission to coordinate management and science approaches for eel conservation across the

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[6] Note that DB-SRA reference points were not accepted for management use by the Peer Review Panel. The TC now recommends stock status be declared depleted based on trend analyses and biomass trends estimated by the DB-SRA. Refer to the Preface and the Peer Review Report for more information.

North American range is near completion and would be a major step forward for American eel management.

9 INTEGRATED PEER REVIEW RECOMMENDATIONS

The ASMFC's Management and Science Committee requested an integrated peer review process be pursued for the current American eel stock assessment with the goal of contracting an individual with appropriate expertise who could provide the Stock Assessment Subcommittee with initial feedback on the stock assessment during the process (i.e., prior to completion of the Stock Assessment Report and final peer review). Dr. Joseph Hightower attended the second American eel Assessment Workshop held May 23–36, 2011 and wrote a summary report conveying suggestions for improving the stock assessment (ASMFC 2011). A brief summary of the main points from his report and the SASC's response are provided below.

1. Pursue the VIMS trawl survey data and the few other surveys that had consistent methods through time and extend back in time to periods of higher abundance.
    - Completed—see Appendix 1 and section 5. See section 5.2.5.8.3 for discussion of decisions regarding VIMS trawl survey data.
2. Some datasets were initially dropped because of consistently low eel catches. Reexamine as they are long time series (e.g., Maryland striped bass seine survey) that may still be of value.
    - Completed—see Appendix 1 and section 5.
3. Utilize consistent methodology for analyzing the relative abundance data. In the draft assessment, some datasets were analyzed using a negative binomial distribution whereas others were done assuming a lognormal. A consistent approach for model fitting and selection, including how AICs will be used and reported, and in the types of variables included as covariates will insure that year-to-year differences among surveys are not due to variation in the methods used for analysis. There is also the issue of samples with a zero catch when the lognormal distribution is used.
    - Completed—see Appendix 2 and section 5.2.
4. There are clear limits to what is feasible for eels in terms of stock assessment model complexity because fishery-dependent and fishery-independent data are limited. Mann-Kendall tests of CPUE trends and traffic light table methods seem worthwhile to apply as complements to more detailed models that can incorporate the additional biological information contained in most surveys.
    - Completed—see sections 6 and 7.
5. Rather than pursuing a long list of models, a better approach would be to select two or three that appear best suited to the species' biology and the available data, then fully explore those models (see specific comments and recommendations by model type above). Relative abundance data from one or more surveys or a synthesis of multiple surveys would be needed for AIM, surplus production, SRA, or any of the more complex models. Getting a valid coast-wide index or multiple regional indices if that is found to be more appropriate, over a sufficient time frame to show contrast in population size will be the key to a successful assessment.

- Completed—see section 6. Coast-wide and regional GLMs were generated for use in trend analyses and surplus production modeling. One additional method that is independent of indices, DB-SRA, was also presented.

6. Consider the different approaches being taken for American eel compared to that of the European eel. There appears to be a consensus that the dramatic decline in the European eel is due to recruitment overfishing.

   - European eel management concentrates on escapement which we have little to no information on in the U.S. Therefore, quantitative reference points using the DB-SRA and trend-based indicators were pursued.

10 RESEARCH RECOMMENDATIONS

The following research recommendations are based on input from the ASMFC American Eel Technical Committee and Stock Assessment Subcommittee as well as from panel members of the 2006 ASMFC American eel stock assessment. A single asterisk (*) denotes short-term recommendations and two asterisks (**) denote long-term recommendations. Recommendations formatted in **bold** identify improvements needed for the next benchmark assessment.

Data Collection

Fisheries Catch and Effort

- **Improve accuracy of commercial catch and effort data**
  - Compare buyer reports to reported state landings*
  - Improve compliance with landings and effort reporting requirements as outlined in the ASMFC FMP for American eel (see ASMFC 2000a for specific requirements)*
  - Require standardized reporting of trip-level landings and effort data for all states in inland waters; data should be collected using the ACCSP standards for collection of catch and effort data (ACCSP 2004)*

- Estimate catch and effort in personal-use and bait fisheries
  - Monitor catch and effort in personal-use fisheries that are not currently covered by the MRFSS or commercial fisheries monitoring programs*
  - Implement a special-use permit for use of commercial fixed gear (e.g., pots and traps) to harvest American eels for personal use; special-use permit holders should be subject to the same reporting requirements for landings and effort as the commercial fishery**
  - Improve monitoring of catch and effort in bait fisheries (commercial and personal-use)*

- Estimated non-directed fishery losses
  - Recommend monitoring of discards in targeted and non-targeted fisheries*
  - Continue to require states to report non-harvest losses in their annual compliance reports*

- **Characterize the length, weight, age, and sex structure of commercially harvested American eels along the Atlantic Coast over time**
  - Require that states collect biological information by life stage (potentially through collaborative monitoring and research programs with dealers) including length, weight,

  - age, and sex through fishery-dependent sampling programs; biological samples should be collected from gear types that target each life stage; at a minimum, length samples should be routinely collected from commercial fisheries*
  - Finish protocol for sampling fisheries; SASC has draft protocol in development*

- Improve estimates of recreational catch and effort
  - Collect site-specific information on the recreational harvest of American eels in inland waters; this could be addressed by expanding the MRIP into inland areas**

- Improve knowledge of fisheries occurring south of the U.S. and within the species' range that may affect the U.S. portion of the stock (i.e., West Indies, Mexico, Central America, and South America)**

Socioeconomic Considerations

- Perform economics studies to determine the value of the fishery and the impact of regulatory management**

- Improve knowledge regarding subsistence fisheries
  - Review the historic participation level of subsistence fishers and relevant issues brought forth with respect to those subsistence fishers involved with American eel**
  - Investigate American eel harvest and resource by subsistence harvesters (e.g., Native American tribes, Asian and European ethnic groups)**

Distribution, Abundance, & Growth

- **Improve understanding of the distribution and frequency of occurrence of American eels along the Atlantic Coast over time**
  - Maintain and update the list of fisheries-independent surveys that have caught American eels and note the appropriate contact person for each survey*
  - Request that states record the number of eels caught by fishery-independent surveys; recommend states collect biological information by life stage including length, weight, age, and sex of eels caught in fishery-independent sampling programs; at a minimum, length samples should be routinely collected from fishery-independent surveys*
  - Encourage states to implement surveys that directly target and measure abundance of yellow- and silver-stage American eels, especially in states where few targeted eel surveys are conducted**
  - A coast-wide sampling program for yellow and silver American eels should be developed using standardized and statistically robust methodologies**

- Improve understanding of coast-wide recruitment trends
  - Continue the ASMFC-mandated YOY surveys; these surveys could be particularly valuable as an early warning signal of recruitment failure*
  - Develop proceedings document for the 2006 ASMFC YOY Survey Workshop; follow-up on decisions and recommendations made at the workshop*

  - Examine age at entry of glass eel into estuaries and freshwater**
  - Develop monitoring framework to provide information for future modeling on the influence of environmental factors and climate change on recruitment**

- Improve knowledge and understanding of the portion of the American eel population occurring south of the U.S. (i.e., West Indies, Mexico, Central America, and South America)**

Future Research

Biology

- Improve understanding of the leptocephalus stage of American eel
  - Examine the mechanisms for exit from the Sargasso Sea and transport across the continental shelf**
  - Examine the mode of nutrition for leptocephalus in the ocean**

- Improve understanding of impact of contaminants as sources of mortality and non-lethal population stressors
  - Investigate the effects of environmental contaminants on fecundity, natural mortality, and overall health**
  - Research the effects of bioaccumulation with respect to impacts on survival and growth (by age) and effect on maturation and reproductive success**

- **Improve understanding of impact of *Anguillicoloides crassus* on American eel**
  - Investigate the prevalence and incidence of infection by the nematode parasite *A. crassus* across the species range*
  - Research the effects of the swim bladder parasite *A. crassus* on the American eel's growth and maturation, migration to the Sargasso Sea, and the spawning potential*
  - Investigate the impact of the introduction of *A. crassus* into areas that are presently free of the parasite**

- **Improve understanding of spawning and maturation**
  - Investigate relation between fecundity and length and fecundity and weight for females throughout their range**
  - Identify triggering mechanism for metamorphosis to mature adult, silver eel life stage, with specific emphasis on the size and age of the onset of maturity, by sex; a maturity schedule (proportion mature by size or age) would be extremely useful in combination with migration rates**
  - Research mechanisms of recognition of the spawning area by silver eel, mate location in the Sargasso Sea, spawning behavior, and gonadal development in maturation**
  - Examine migratory routes and guidance mechanisms for silver eel in the ocean**

- Improve understanding of predator-prey relationships**

- Investigating the mechanisms driving sexual determination and the potential management implications**

Passage & Habitat

- **Improve upstream and downstream passage for all life stages of American eels**
  - Develop design standards for upstream passage devices for eels; this will be a product (at least partial design guidelines) from the ASMFC 2011 Eel Passage Workshop, so this research need may be partially met in the near term*
  - Investigate, develop, and improve technologies for American eel passage upstream and downstream at various barriers for each life stage; in particular, investigate low-cost alternatives to traditional fishway designs for passage of eel**

- Improve understanding of the impact of barriers on upstream and downstream movement
  - Evaluate the impact, both upstream and downstream, of barriers to eel movement with respect to population and distribution effects; determine relative contribution of historic loss of habitat to potential eel population and reproductive capacity**
  - Recommend monitoring of upstream and downstream movement at migratory barriers that are efficient at passing eels (e.g., fish ladder/lift counts); data that should be collected include presence/absence, abundance, and biological information; provide standardized protocols for monitoring eels at passage facilities; coordinate compilation of these data; provide guidance on the need and purpose of site-specific monitoring**

- **Improve understanding of habitat needs and availability**
  - Assess characteristics and distribution of American eel habitat and value of habitat with respect to growth and sex determination; develop GIS of American eel habitat in U.S.**
  - Assess available drainage area over time to account for temporal changes in carrying capacity; develop GIS of major passage barriers**

- Improve understanding of within-drainage behavior and movement and the exchange between freshwater and estuarine systems**

- Improve estimates of mortality associated with upstream and downstream passage
  - Monitor non-harvest losses such as impingement, entrainment, spill, and hydropower turbine mortality*

- Evaluate eel impingement and entrainment at facilities with NPDES authorization for large water withdrawals; quantify regional mortality and determine if indices of abundance could be established as specific facilities**

- Investigate best methods for reintroducing eels into a watershed; examine approaches for determining optimum density*

Assessment Methodology & Management Support

- Coordinate monitoring, assessment, and management among agencies that have jurisdiction within the species' range (e.g., ASMFC, GLFC, Canada DFO)**
- Perform a joint U.S.-Canadian stock assessment*
- Perform periodic stock assessments (every 5–7 years) and establish sustainable reference points for American eel are required to develop a sustainable harvest rate in addition to determining whether the population is stable, decreasing, or increasing
  - Develop new assessment models (e.g., delay-difference model) specific to eel life history and fit to available indices**
  - **Conduct intensive age and growth studies at regional index sites to support development of reference points and estimates of exploitation***
  - Develop GIS-type model that incorporates habitat type, abundance, contamination, and other environmental factors**
  - Develop population targets based on habitat availability at the regional and local level**
- Implement large-scale (coast-wide or regional) tagging studies of eels at different life stages; tagging studies could address a number of issues including:
  - Natural, fishing, and discard mortality; survival**
  - Growth**
  - Passage mortality**
  - Movement, migration, and residency**
  - Validation of ageing methods**
  - Reporting rates**
  - Tag shedding or tag attrition rates**

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12 TABLES

Table 1.1. Commercial fishery regulations for American eels as of 2012, by state. For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

State	Size Limit	License/Permit	Other
ME		Harvester license; dealer license and reporting	Seasonal closures; gear restrictions
NH	6"	Commercial saltwater license and wholesaler license; monthly reporting	50/day for bait; gear restrictions in freshwater
MA	6"	Commercial permit with annual catch report requirement; registration for dealers with purchase record requirement	Nets, pots, spears, and angling only; mesh restrictions; each of 52 coastal towns has its own regulations
RI	6"	Commercial fishing license	
CT	6"	Commercial license; dealer reporting	Gear restrictions
NY	6"	Commercial harvester license and reporting; dealer license.	Gear restrictions
NJ	6"	License required	Gear restrictions
PA	NO COMMERCIAL FISHERY		
DE	6"	License required	Commercial fishing in tidal waters only; gear restrictions
MD	6"	Licensed required with monthly reporting	Prohibited in non-tidal waters; gear restrictions
DC	NO COMMERCIAL FISHERY		
PRFC	6"	Harvester license and reporting	Gear restrictions
VA	6"	License with two-year delayed entry system; monthly reporting	Mesh size restrictions on eel pots; bait limit of 50 eels/day; seasonal closures
NC	6"	Standard Commercial Fishing License for all commercial fishing	Mesh size restrictions on eel pots; bait limit of 50 eels/day; seasonal closures
SC		License for commercial fishing and sale; permits by gear and area fished; monthly reporting	Gear restrictions
GA	6"	Personal commercial fishing license and commercial fishing boat license; harvester/dealer reporting	Gear restrictions on traps and pots; area restrictions
FL		Permits and licenses	Gear restrictions

Table 1.2. Recreational fishery regulations for American eels as of 2012, by state. For specifics on licenses, gear restrictions, and area restrictions, please contact the individual state.

State	Size Limit	Possession Limit	Other
ME	6"	50 eels/person/day	Gear restrictions; license requirement and seasonal closures (inland waters only)
NH	6"	50 eels/person/day	Coastal harvest permit needed if taking eels other than by angling; gear restrictions in freshwater
MA	6"	50 eels/person/day	Nets, pots, spears, and angling only; mesh restrictions; each of 52 coastal towns has its own regulations
RI	6"	50 eels/person/day	
CT	6"	50 eels/person/day	
NY	6”	50/eels/person/day	Additional length restrictions in specific inland waters
NJ	6"	50 eels/person/day	Two pot limit/person
PA	6"	50 eels/person/day	Gear restrictions
DE	6"	50 eels/person/day	Two pot limit/person
MD	6"	25/person/day limit in non-tidal areas	Gear restrictions.
DC	6"	10 eels/person/day	Five trap limit
PRFC	6"	50 eels/person/day	
VA	6"	50 eels/person/day	Recreational license; two pot limit; mandatory annual catch report; mesh size restrictions on eel pots
NC	6"	50 eels/person/day	Gear restrictions; non-commercial special device license; two eel pots allowed under Recreational Commercial Gear license
SC	None	None	Gear restrictions and gear license fees
GA	None	None	
FL	None	None	Gear restrictions

Table 2.1. Timing and average length reported for glass-stage American eel upstream migrants in various locations.

Location	Peak Timing	Average Length (mm)	Reference
N. Gulf of St. Lawrence	Jun–Aug	62	Dutil et al. 1989
Gulf of St. Lawrence	May–Jul		Dutil et al. 2009
Various locations, Nova Scotia	Apr–Jun	59.5–64.8	Jessop 1998
Nova Scotia	May–Jul	60.3	Jessop 2003
East R., Nova Scotia	May	60	Wang and Tzeng 2000
Musquash R., New Bruns.	Apr	60	Wang and Tzeng 2000
Annaquatucket R., RI	Apr–May	58	Haro and Krueger 1988
Annaquatucket R., RI	Apr	59	Wang and Tzeng 2000
Gilbert Stuart Brook, RI	May	58	Sorenson and Bianchini 1986
Little Egg Inlet, NJ	Jan–Jun	48.7–68.1	Wuenschel and Able 2008
Indian R., DE	Jan–Apr	57	Clark 2009
North Carolina	Mar	48	Wang and Tzeng 2000
Beaufort, NC	Feb–Mar	53.6	Powles and Warlen 2002
Albemarle Sound, NC	Feb–Mar	57.7	Overton and Rulifson 2009
Altamaha R., GA	late winter	52	Helfman et al. 1984b
Florida	Jan–Feb	49	Wang and Tzeng 2000
Haiti	Dec	48	Wang and Tzeng 2000

Table 2.2. Average length, age, and timing reported for migrating silver-phase American eels in various locations, by sex. Length and age ranges are in parentheses.

	Migration	Female		Male		
Location	**Timing**	**Length (mm)**	**Age (yrs)**	**Length (mm)**	**Age (yrs)**	**Reference**
St. Lawrence R. (Upper)	Jun–Oct	915 to 1,000 (890–1,123)	20, 21			Casselman 2003; McGrath et al. 2003a; Tremblay 2009; McGrath et al. 2009
St. Lawrence River	Aug–Nov	853 (475–1,000)	13, 14			Hurley 1972; Dutil et al. 1987; Fournier and Caron 2001; Verreault et al. 2003; Tremblay 2009
St . Lawrence (estuarine)	Aug–Nov	650 to 1,043 (526–1,219)	20 to 23			Dutil et al. 1987; Couillard et al. 1997; Verreault 2002; McGrath et al. 2003a; Verreault et al. 2003; Tremblay 2009
Newfoundland	Aug–Sept	590 to 778 (431–931)	6 to 19 (3–32)	340 (329–361)	(4–15)	Gray and Andrews 1970, 1971; Bouillon and Haedrich 1985; Jessop et al. 2009
New Brunswick	July–Oct	417 to 565 (284–733)		317, 326		Smith and Saunders 1955; Ingraham 1999
Nova Scotia	Aug–Nov	491 to 610 (394–945)	19 (8–43)	392 (346–473)	12.7 (6–18)	Jessop 1987; Carr and Whoriskey 2008
Maine	Aug–Oct	(502–538)	15 to 16 (6–18)	(344–359)	12 to 13	Oliveira and McCleave 2000; Haro et al. 2003
Southeast of Cape Cod	Nov	642		373		Wenner 1973
Rhode Island	Sept–Dec	475 to 537 (410–867)	12.8 (6–20)	(323–335) (228–400)	10.9 (4–15)	Winn et al. 1975; Bianchini et al. 1983, cited by Helfman et al. 1987; Krueger and Oliveira 1997; Oliveira 1999
Connecticut River	Sept–Oct	707				Brown et al. 2009
Indian River, DE	Aug–Nov	571 (367–774)	12 (7–20)	330 (264–412)	7.4 (4–16)	Barber 2004
E of Assateague Is., MD	Dec	636 (609–658)				Wenner 1973
Chesapeake Bay, MD	Oct			306 (275–360)	5.1 (3–10)	Foster and Brody 1982
Chesapeake Bay, VA	Nov	(366–452)		(395–438)		Wenner 1973
Southeast of Ches. Bay	Dec	551 (512–579)				Wenner 1973
Cape Charles, VA	Nov	633 (418–845)		372 (339–438)		Wenner and Musick 1974
Potomac R., VA		(600–800)	(5–11)	350		Goodwin and Angermeier 2003
Shenandoah R., WV	Sep–Dec	869, 872 (560–1,118)	(10–19)			Euston et al. 1998; Goodwin and Angermeier 2003
Cooper R., SC		543, 646 (369–834)	6, 7.6	257, 318 (214–322)	3	Harrell and Loyacano 1982
Charleston Harbor, SC		550	5.8	317	2.7	Michener and Eversole 1983
Altamaha R., GA	Oct–Mar	584, 587 (413–682)	5, 8.6 (4–13)	329 (282–411)	4.1, 5.5 (3–10)	Helfman et al. 1984b; Facey and Helfman 1985

Table 2.3. Average length and age reported for yellow-phase American eels in various locations, by salinity and sex. Length and age ranges are in parentheses.

Location	Salinity	Sex	Length (mm)	Age (years)	Reference
Castors R., Newfoundland	fresh	female	512 (464–576)	18.4 (11–28)	Jessop et al. 2009
Muddy Hole, Newfoundland	brackish	female	440 (335–662)	6.2 (3–10)	Jessop et al. 2009
Lake Champlain, VT	fresh	female	670 (430–900)	15.9 (8–23)	Facey and LaBar 1981
Hudson R., NY	fresh	female	464	(7–30)	Morrison and Secor 2003
Hudson R., NY	brackish	pooled	440	(3–39)	Morrison and Secor 2003
Various locations, NJ	fresh	pooled	350 (145–850)	10 (3–19)	Ogden 1970
Susquehanna R., MD	both	pooled	327 (210–580)	8.5 (5–17)	Foster and Brody 1981
Upper Ches. Bay, MD	both	pooled	377 (226–658)	7 (3–14)	Foster and Brody 1981
Ches. Bay, MD & VA	both	pooled	365 (213–647)	5.8 (3–11)	Fenske et al. 2010
Ches. Bay Tribs., VA	both	pooled	110–560 (60–776)	3–6 (1–18)	Owens and Gear 2003
James R., VA	fresh	unk.	(174–775)		Strickland 2002
Charleston Harbor, SC	brackish	male	317	2.7	Michener and Eversole 1983
Charleston Harbor, SC	brackish	female	437 (213–719)	4.3 (2–6)	Michener and Eversole 1983
Cooper R., SC	fresh	male	257, 318 (214–322)	3	Harrell and Loyacano 1982
Cooper R., SC	fresh	female	397, 425 (280–577)	5	Harrell and Loyacano 1982
Cooper R., SC	brackish	male	(260–406)	2.8 (1–5)	Hansen and Eversole 1984
Cooper R., SC	brackish	female	(287–687)	4.4 (2–12)	Hansen and Eversole 1984
Altamaha R., GA	fresh	pooled	(211–625)	6.2 (3–13)	Helfman et al. 1984b
Altamaha R., GA	brackish	pooled	(249–537)	4.6 (2–7)	Helfman et al. 1984b

Table 2.4. Average growth rate (mm/year) reported for American eels in various locations, by estimation method and salinity.

Location	Method	Salinity	Growth Rate (mm/yr)	Reference
St. Lawrence R., QU	direct measure	fresh	40	Verreault et al. 2009
Gulf of St. Lawrence	back calculated	brackish	94	Lamson et al. 2009
Gulf of St. Lawrence	back calculated	fresh	45	Lamson et al. 2009
Lake Ontario	back calculated	fresh	54.9	Hurley 1972
East River, NS	back calculated	fresh	21.7	Jessop et al. 2006
East River, NS	back calculated	brackish	26.6	Jessop et al. 2006
Medway & LaHave R., NS	back calculated	fresh	41–51	Jessop 1987
Maine rivers (male)	back calculated	fresh	28.9	Oliveira and McCleave 2002
Maine rivers (female)	back calculated	fresh	31.9	Oliveira and McCleave 2002
Annaquatucket R., RI	direct measure	fresh	29.9	Oliveira 1997
Annaquatucket R., RI (male)	back calculated	fresh	31	Oliveira 1999
Annaquatucket R., RI (female)	back calculated	fresh	40	Oliveira 1999
Hudson R., NY	back calculated	both	39	Mattes 1989, cited in Morrison and Secor 2003
Hudson R., NY	back calculated	brackish	55	Morrison and Secor 2003
Hudson R., NY	back calculated	fresh	28	Morrison and Secor 2003
Hudson R., NY	direct measure	brackish	80	Morrison and Secor 2003
Hudson R., NY	direct measure	fresh	34	Morrison and Secor 2003
Hudson R., NY	back calculated	unk.	35	Machut et al. 2007
Indian R., DE	back calculated	brackish	83	Barber 2004
Indian R., DE	back calculated	fresh	47	Barber 2004
Delaware Bay, DE	back calculated	brackish	32	Clark 2009
Ches. Bay, MD & VA	back calculated	both	68	Fenske et al. 2010
Shenandoah R., VA	direct measure	fresh	43	Goodwin 1999
James R., VA	direct measure	fresh	18–43	Strickland 2002
James R., VA	direct measure	fresh	32–43	Roghair et al. 2003
Cooper R., SC	back calculated	fresh	53.5	Harrell 1977
Cooper R., SC	back calculated	brackish	27–69	Hansen and Eversole 1984
Altamaha R., GA	direct measure	both	57	Helfman et al. 1984a
Altamaha R., GA	back calculated	both	44	Helfman et al. 1984a
Altamaha R., GA	back calculated	brackish	53	Helfman et al. 1984b
Altamaha R., GA	back calculated	brackish	50	Helfman et al. 1984b
Louisiana	direct measure	fresh	128 & 325	Gunning and Shoop 1962

Table 2.5. Average length (mm) at age reported for American eels in various locations. Age includes only years spent inland (i.e., does not include first oceanic year).

Location	**Age**												**Reference**
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	
Topsail Pond, NFL				249	277	343	418	493	553	652	706	748	Bouillon and Haedrich 1985
Nanticoke R., MD	258	299	311	365	439	484	501	541	557	561	575	613	Weeder and Hammond 2009
Wye R., MD	300	397	464	524	591	750							Weeder and Hammond 2009
Assawoman Bay, MD	320	381	466	523	557	583	539						Weeder and Hammond 2009
Pocomoke R., MD		485	572	638	643	647	650	680	670				Weeder and Hammond 2009
Ches. Bay, MD		334	382	442	466	480	476	504	527	490	565	578	K. Whiteford (pers. comm.)
Ches. Bay, VA	204	274	346	451	476	493	476	536	624		528		Owens and Gear 2003
Cooper R., SC	224	249	337	403	490	536	596	612	638	509	680	690	Harrell and Loyacano 1982
Cooper R., SC	292	361	411	455	482	511	580	514	611			551	Hansen and Eversole 1984
Altamaha R., GA	242	310	361	403	442	460							Helfman et al. 1984a
Bermuda	226	334	418	472	489								Boetius and Boetius 1967, cited in Harrell and Loyacano 1982

Table 2.6. Parameter estimates for the linear regression of length in millimeters on age in years reported for American eel in previous studies. An asterisk (*) denotes studies for which the biological data were available for inclusion in the current assessment.

Location	**Collection Period**	**n**	***Intercept***	***Slope***	**Reference**
Sheepscot River, ME	Aug–Sep 1996; Jun–Jul 1997	646	77.9	23.7	Oliveira and McCleave 2000 *
Medomak River, ME	Aug–Sep 1996; Jun–Jul 1997	592	119	20.7	Oliveira and McCleave 2000 *
Pleasant River, ME	Aug–Sep 1996; Jun–Jul 1997	378	76.6	23.4	Oliveira and McCleave 2000 *
East Machias River, ME	Aug–Sep 1996; Jun–Jul 1997	709	94.6	24.2	Oliveira and McCleave 2000 *
4 rivers pooled, ME	Aug–Sep 1996; Jun–Jul 1997	2,325	87.8	23.4	Oliveira and McCleave 2000 *
Lake Champlain, VT		426	375	18.8	Facey and LaBar 1981
Lake Mattamuskeet-Pamlico Sound drainage, NC	Feb 2002–Sep 2003	565	379	12.5	Rulifson et al. 2004 *
Altamaha River, GA (estuary)	Fall 1980–Summer 1981 (average)	203	142	49.7	Helfman et al. 1984b
Altamaha River, GA (freshwater)	Fall 1980–Summer 1981 (average)	215	69.3	53.4	Helfman et al. 1984b

Table 2.7. Parameter estimates of the allometric relation of length in millimeters to weight in grams reported for American eel in previous studies. An asterisk (*) denotes studies for which the biological data were available for inclusion in the current assessment.

Location	**Collection Period**	**n**	***a***	***b***	**Reference**
Sheepscot River, ME	Aug–Sep 1996; Jun–Jul 1997	870	7.03E-07	3.15	Oliveira and McCleave 2000 *
Medomak River, ME	Aug–Sep 1996; Jun–Jul 1997	981	1.14E-06	3.07	Oliveira and McCleave 2000 *
Pleasant River, ME	Aug–Sep 1996; Jun–Jul 1997	502	1.18E-06	3.07	Oliveira and McCleave 2000 *
East Machias River, ME	Aug–Sep 1996; Jun–Jul 1997	763	1.13E-06	3.07	Oliveira and McCleave 2000 *
4 rivers pooled, ME	Aug–Sep 1996; Jun–Jul 1997	3,116	9.84E-07	3.09	Oliveira and McCleave 2000 *
Lake Champlain, VT		426	9.33E-04	3.17	Facey and LaBar 1981
New York Bight		5	2.15E-06	2.99	Wilk et al. 1978
James River, VA	1997–2000	174	3.00E-06	2.91	Owens and Geer 2003 *
York River, VA	1997–2000	255	8.03E-07	3.15	Owens and Geer 2003 *
Rappahannock River, VA	1997–2000	187	3.12E-06	2.91	Owens and Geer 2003 *
Lake Mattamuskeet-Pamlico Sound drainage, NC	Feb 2002–Sep 2003	759	5.99E-06	2.81	Rulifson et al. 2004 *
White Oak River, NC	May–Jun 2002	270	2.42E-07	3.41	Hightower and Nesnow 2006
White Oak River, NC	Jul–Aug 2003	218	2.07E-07	3.41	Hightower and Nesnow 2006
Pinopolis Dam, Cooper River, SC	Sep 1975–Sep 1976	258	2.40E-07	3.36	Harrell and Loyacano 1982
Wadboo Creek, Cooper River, SC	Jun–Dec 1975	157	6.03E-07	3.20	Harrell and Loyacano 1982
Cooper River, SC		462	1.41E-06	3.07	Hansen and Eversole 1984
Charlestown Harbor, SC	Jul 1978–Sep 1979	475?	1.92E-06	3.07	Michener and Eversole 1983
Altamaha River, GA (estuary)	Fall 1980	86	2.78E-07	3.32	Helfman et al. 1984b
Altamaha River, GA (freshwater)	Fall 1980	145	3.04E-07	3.31	Helfman et al. 1984b
Altamaha River, GA (estuary)	Winter 1981	305	9.82E-07	3.10	Helfman et al. 1984b
Altamaha River, GA (freshwater)	Winter 1981	265	2.69E-07	3.32	Helfman et al. 1984b
Altamaha River, GA (estuary)	Spring 1981	109	1.58E-06	3.04	Helfman et al. 1984b
Altamaha River, GA (freshwater)	Spring 1981	327	1.19E-06	3.09	Helfman et al. 1984b
Altamaha River, GA (estuary)	Summer 1981	59	4.42E-07	3.25	Helfman et al. 1984b
Altamaha River, GA (freshwater)	Summer 1981	73	7.13E-07	3.15	Helfman et al. 1984b

Table 2.8. Percentage of females reported for American eels in various locations, by salinity.

Location	Salinity	% Female	Reference
Newfoundland	fresh	94	Vladykov 1966
Newfoundland	fresh	99	Gray and Andrews 1970
Newfoundland	brackish	100	Gray and Andrews 1970
New Brunswick	fresh	80	Vladykov 1966
Nova Scotia	fresh	100	Vladykov 1966
Medway R., NS	fresh	97	Jessop 1987
LaHave R., NS	fresh	100	Jessop 1987
Quebec	fresh	99	Vladykov 1966
Matamek R., QU	brackish	95	Dolan and Power 1977
Matamek R., QU	fresh	99	Dolan and Power 1977
Ontario	fresh	100	Vladykov 1966
Maine Rivers	both	24	Oliveira et al. 2001
Lake Champlain, VT	fresh	100	Facey and LaBar 1981
Massachusetts	brackish	91	Vladykov 1966
Rhode Island rivers	fresh	12	Winn et al. 1975
Rhode Island rivers	brackish	45	Winn et al. 1975
Pawcatucket R., RI	fresh	90	Bianchini et al. 1983, cited by Helfman et al. 1987
Coastal rivers, RI	fresh	11	Bianchini et al. 1983, cited by Helfman et al. 1987
Annaquatucket R., RI	fresh	5	Oliveira 1999
New York	brackish	67	Vladykov 1966
Hudson R., NY	both	97	Morrison and Secor 2003
Hudson R., NY	fresh	100	Morrison and Secor 2003
New Jersey	brackish	42	Vladykov 1966
Indian R., DE	fresh	22	Barber 2004
Upper Ches. Bay, MD	both	100	Foster and Brody 1982
Chesapeake Bay, MD[7]	both	40	Weeder and Hammond 2009
Ches. Bay, VA & MD[8]	both	71	Fenske et al. 2010
Potomac R., VA	both	71	Goodwin and Angermeier 2003
Shenandoah R., VA	fresh	100	Goodwin and Angermeier 2003
Cooper R., SC	fresh	98	Harrell and Loyacano 1982
Charleston Harbor, SC	brackish	93	Michener and Eversole 1983
Cooper R., SC	brackish	96	Hansen and Eversole 1984
Altamaha R., GA	brackish	64	Helfman et al. 1984b
Altamaha R., GA	fresh	94	Helfman et al. 1984b
Georgia rivers	brackish	64	Helfman et al. 1987
Georgia rivers	fresh	82	Helfman et al. 1987
Florida	brackish	47	Vladykov 1966
Mississippi	brackish	95	Ross et al. 1984, cited by Helfman et al. 1987
Louisiana	brackish	17	Vladykov 1966
Bermuda	brackish	96	Boetius and Boetius 1967, cited by Harrell and Loyacano 1982
Trinidad	brackish	62	Vladykov 1966

[7] 29% undifferentiated

[8] 23% intersexual, 4% undifferentiated

Table 2.9. Parameters of the allometric fecundity (F)-length (L) and fecundity-weight (W) relationship for American eels estimated by studies in various locations. The length range of individual eels used in the study and estimated fecundity values are also given. These parameter values apply to length measured in millimeters and weight measured in grams. The unit for fecundity is millions of eggs.

Location	Gear	Collection Period	n	Length $F = \alpha L^{\beta}$		Weight $F = \alpha W^{\beta}$		Length Range (mm)	Estimated Fecundity (millions of eggs)	Reference
				α	β	α	β			
St. Lawrence	Various traps	Sep–Oct 2001; Aug–Sep 2002	150	1.57	2.29	35,237	0.762	532–1,159	3.4–22	Tremblay 2009
Various rivers, ME	Weirs & fyke nets	Oct–Nov 1996	63	0.0198	2.96	14,608	0.915	450–1,130	1.84–19.9	Barbin & McCleave 1997
Chesapeake Bay, VA	Commercial pound nets	Nov 1970	21	5.07E-05	3.74	1,694	1.12	420–720	0.4–2.6	Wenner & Musick 1974

Table 5.1. Summary of (A) length (mm) and (B) weight (g) data from New Jersey commercial biosamples.

(A)

Statistic	2006	2007	2008	2009	2010
Average	500.39	479.45	416.37	472.57	443.67
St Dev	106.47	112.09	146.17	99.74	87.76
Min	234	232	100	128	252
Max	1,030	751	768	792	744
n	457	237	547	478	399

(B)

Statistic	2006	2007	2008	2009	2010
Average	278.47	233.92	170.41	216.94	181.84
St Dev	201.55	175.69	157.62	127.25	137.07
Min	20	10	2	2	27
Max	1,970	840	975	910	1,075
n	457	237	547	478	399

Table 5.2. Length-weight parameters from New Jersey commercial biosamples.

Year	Region	*a*	Approx SE[*a*]	*b*	Approx SE[*b*]
2006	All areas	1.08E-06	2.21E-07	3.0951	0.0318
2007	All areas	3.27E-07	1.98E-07	3.2732	0.0947
2008	All areas	8.65E-07	2.13E-07	3.1083	0.0384
2009	All areas	3.48E-06	1.09E-06	2.8957	0.0494
2010	All areas	4.64E-08	1.31E-08	3.5930	0.0447
All years	Coast	5.08E-07	8.80E-08	3.2034	0.027
All years	Delaware Bay	1.56E-07	3.26E-08	3.4036	0.0333
All years	Hudson	1.25E-08	1.45E-08	3.7526	0.1783
All years	All areas	6.84E-07	8.67E-08	3.1576	0.0198

Table 5.3. Numbers of American eels available for sampling in the VMRC's Biological Sampling Program, by gear, 1989–2010. Other gears include fyke net, crab pot, and gill net.

	Eel Pot		**Pound Net**		**Other**	
Year	**Lengths**	**Weights**	**Lengths**	**Weights**	**Lengths**	**Weights**
1989	192	192	2	2	0	0
1990	186	186	0	0	0	0
1991	216	216	0	0	0	0
1992	0	0	0	0	0	0
1993	0	0	2	2	0	0
1994	50	50	0	0	3	1
1995	0	0	0	0	0	0
1996	0	0	1	1	0	0
1997	0	0	5	4	0	0
1998	0	0	6	4	6	0
1999	0	0	0	0	0	0
2000	0	0	0	0	0	0
2001	0	0	1	1	0	0
2002	0	0	1	0	0	0
2003	0	0	3	3	17	17
2004	0	0	24	16	0	0
2005	59	59	7	7	0	0
2006	0	0	10	3	0	0
2007	0	0	19	19	0	0
2008	0	0	8	4	0	0
2009	0	0	0	0	0	0
2010	0	0	0	0	0	0

Table 5.4. Numbers of American eel samples reported by the MRFSS angler-intercept survey and at-sea headboat survey, by catch type, 1981–2010.

	Type A	**Type B1**		**Type B2**	
Year	**Intercept**	**Intercept**	**Headboat**	**Intercept**	**Headboat**
1981	22	75		94	
1982	75	44		43	
1983	28	19		73	
1984	28	12		26	
1985	53	17		91	
1986	62	41		138	
1987	16	34		49	
1988	35	36		74	
1989	57	31		150	
1990	36	16		154	
1991	113	30		123	
1992	13	25		101	
1993	224	40		101	
1994	98	48		89	
1995	23	6		96	
1996	18	29		77	
1997	9	8		50	
1998	7	3		84	
1999	4	7		70	
2000	7	5		43	
2001	1	8		44	
2002	6	10		79	
2003	16	16		155	
2004	13	16		99	
2005	7	3	1	65	1
2006	7	3	0	76	2
2007	39	7	0	73	1
2008	4	5	0	66	8
2009	9	4	0	75	7
2010	14	22	0	117	2

Table 5.5. Numbers of American eels that available for biological sampling in the MRFSS angler-intercept survey and at-sea headboat survey, by survey component, 1981–2010.

	Intercept (Type A)		Headboat (Type B2)
Year	Weighed	Measured	Measured
1981	21	21	
1982	46	49	
1983	16	16	
1984	22	22	
1985	30	27	
1986	25	18	
1987	13	10	
1988	28	27	
1989	47	29	
1990	12	17	
1991	37	35	
1992	3	3	
1993	15	32	
1994	21	13	
1995	2	2	
1996	5	5	
1997	7	7	
1998	3	4	
1999	1	2	
2000	7	7	
2001	0	1	
2002	1	2	
2003	0	2	
2004	11	13	
2005	4	6	1
2006	3	3	1
2007	3	4	0
2008	2	3	6
2009	4	4	6
2010	6	6	1

Table 5.6. Estimates of recreational fishery harvest and released alive for American eels along the Atlantic coast, 1981–2010. The precision of each estimate, measured as proportional standard error (PSE), is also given.

	Harvest (Type A + B1)				**Released Alive (Type B2)**	
Year	**Numbers**	**PSE[Numbers]**	**Weight (pounds)**	**PSE[Weight]**	**Numbers**	**PSE[Numbers]**
1981	85,858	22.6	71,943	34.6	94,136	28.0
1982	144,376	28.3	94,187	32.3	68,314	34.2
1983	88,190	40.2	76,310	50.2	67,258	21.1
1984	59,528	22.9	56,380	36.2	39,603	32.1
1985	161,077	37.3	157,155	10.9	68,338	25.2
1986	101,192	24.5	80,920	26.4	97,240	18.3
1987	37,761	29.0	28,060	41.0	52,729	26.7
1988	62,419	21.8	29,639	15.4	84,050	27.9
1989	50,199	20.7	66,665	19.2	91,119	15.9
1990	24,333	24.1	13,133	34.1	80,366	15.9
1991	77,712	28.5	57,315	29.6	64,312	22.2
1992	31,286	33.2	1,955		44,836	25.3
1993	71,313	35.7	43,715	51.1	70,133	21.3
1994	49,652	28.6	24,782	42.4	56,329	16.6
1995	9,199	54.6	939		51,820	23.8
1996	20,554	31.2	6,312	46.6	45,111	17.8
1997	15,521	56.1	6,565	51.9	21,464	22.7
1998	6,238	38.9	3,331	78.7	46,455	21.8
1999	5,651	42.8	359		45,467	50.6
2000	27,078	74.1	13,247	82.9	38,672	27.9
2001	10,805	76.6			24,704	20.9
2002	5,568	35.5	584		38,538	16.4
2003	31,093	60.4			126,330	17.3
2004	23,129	37.0	13,411	55.9	90,829	24.8
2005	8,362	49.7	2,469	98.3	50,702	21.2
2006	19,717	44.2	11,043	45.2	66,307	24.5
2007	57,986	56.9	49,068	76.8	82,385	26.6
2008	3,485	53.5	353	100.1	45,323	23.0
2009	6,213	46.4	5,600	32.4	56,522	20.0
2010	60,202	67.7	25,922	87.3	75,102	25.3

Table 5.7. Currently active sampling sites for the ASMFC-mandated annual American eel YOY abundance survey. Sites formatted in **bold** font have been sampled for at least 10 years as of 2010.

State	Site	Gear	Start Year
ME	**West Harbor Pond**	**Irish Elver Ramp**	2001
NH	**Lamprey River**	**Irish Elver Trap**	2001
MA	Acushnet River Reservoir	Sheldon Elver Trap	2005
MA	Acushnet River Sawmill	Sheldon Elver Trap	2005
MA	Cold Brook	Irish Elver Ramp	2008
MA	**Jones River**	**Sheldon Elver Trap**	2001
MA	Parker River	Sheldon Elver Trap	2004
MA	Saugus River	Sheldon Elver Trap	2005
MA	Saugus River	Irish Elver Ramp	2007
MA	Wankinco River	Irish Elver Ramp	2009
RI	**Gilbert Stuart Dam (Pettasquamscutt River)**	**Irish Elver Ramp**	2000
RI	Hamilton Fish Ladder (Annaquatucket River)	Irish Elver Ramp	2004
CT	Ingham Hill	Irish Elver Ramp	2007
NY	**Carman's River**	**Fyke Net**	2000
PA	Poquessing Creek	Modified Minnow Trap	2008
PA	Poquessing Creek	Lift Net	2008
PA	Poquessing Creek	Backback Electrofisher	2008
NJ	**Patcong Creek**	**Fyke Net**	2000
DE	**Millsboro Dam (Indian River)**	**Fyke Net**	2000
MD	**Turville Creek**	**Irish Elver Ramp**	2000
DC	Anacostia River, Washington Channel, and Rock Creek	eel pots, boat and backpack efishing, and Irish elver traps	2005
PRFC	**Clark’s Millpond (Coan River)**	**Irish Elver Ramp**	2000
PRFC	**Gardy’s Millpond (Yeocomico River)**	**Irish Elver Ramp**	2000
VA	**Bracken’s Pond (York River)**	**Irish Elver Ramp**	2000
VA	**Kamp’s Millpond (Rappahannock River)**	**Irish Elver Ramp**	2000
VA	Warehams Pond (James River)	Irish Elver Ramp	2003
VA	**Wormley Creek (York River)**	**Irish Elver Ramp**	2001
SC	**Goose Creek (Cooper River)**	**Fyke Net**	2000
GA	**Altamaha Canal**	**Fyke Net**	2001
GA	Hudson Creek	Fyke Net	2003
FL	**Guana River Dam**	**Dip Net**	2001

Table 5.8. Summary of GLM analyses used to standardize YOY indices developed from the ASMFC-mandated recruitment surveys. Phi is the overdispersion parameter.

Region	State	Location	Years	Gear	GLM?	Error Structure	Response	Predictors	Phi
Gulf of Maine	ME	West Harbor Pond	2001–2010	Irish Elver Ramp	Y	NB	Catch	Year+WaterTemp	1.02
	NH	Lamprey River	2001–2010	Irish Elver Trap	Y	LN	Catch	Year+WaterTemp	1.92
	MA	Jones River	2001–2010	Sheldon Elver Trap	Y	NB	Catch	Year+Discharge	0.952
Southern New England	RI	Gilbert Stuart Dam	2000–2010	Irish Elver Ramp	Y	NB	Catch	Year+WaterTemp+Water Level	1.46
	NY	Carman's River	2000–2010	Fyke Net	Y	NB	Catch	Year+WaterTemp	1.90
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Patcong Creek	2000–2009	Fyke Net	Y	NB	Catch	Year+WaterTemp	1.67
	DE	Millsboro Dam	2000–2010	Fyke Net	Y	NB	Catch	Year+Discharge	1.41
	MD	Turville Creek	2000–2010	Irish Elver Ramp	N				
Chesapeake Bay	PRFC	Clark's Millpond	2000–2010	Irish Elver Ramp	Y	NB	Catch	Year+WaterTemp	1.69
	PRFC	Gardy's Millpond	2000–2010	Irish Elver Ramp	Y	Delta-gamma	Catch	Year+WaterTemp	
	VA	Bracken's Pond	2000–2010	Irish Elver Ramp	N				
	VA	Kamp's Millpond	2000–2010	Irish Elver Ramp	Y	NB	Catch	Year+WaterTemp	1.64
	VA	Wormley Creek	2001–2010	Irish Elver Ramp	Y	NB	Catch	Year+WaterTemp	1.50
South Atlantic	SC	Goose Creek	2000–2010	Fyke Net	Y	LN	Catch	Year+WaterTemp+Water Level	1.36
	GA	Altamaha Canal	2001–2010	Fyke Net	Y	LN	Catch	Year+WaterTemp	1.11
	FL	Guana River Dam	2001–2010	Dip Net	N				

Table 5.9. Spearman's rank correlation between YOY indices developed from the ASMFC-mandated recruitment surveys. Values formatted in **bold** font are statistically significant at $\alpha < 0.10$.

	Region	Gulf of Maine			Southern New England		Delaware Bay & Mid-Atlantic Coastal Bays			Chesapeake Bay					South Atlantic	
Region	**Survey Site**	West Harbor Pond (ME)	Lamprey River (NH)	Jones River (MA)	Gilbert Stuart Dam (RI)	Carman's River (NY)	Patcong Creek (NJ)	Millsboro Dam (DE)	Turville Creek (MD)	Clark's Millpond (PRFC)	Gardy's Millpond (PRFC)	Bracken's Pond (VA)	Kamp's Millpond (VA)	Wormley Creek (VA)	Goose Creek (SC)	Altamaha Canal (GA)
Gulf of Maine	Lamprey River (NH)	0.0424														
	Jones River (MA)	0.164	-0.248													
Southern New England	Gilbert Stuart Dam (RI)	0.236	0.164	-0.0424												
	Carman's River (NY)	-0.127	0.0182	-0.297	**0.591**											
Delaware Bay & Mid-Atlantic Coastal Bays	Patcong Creek (NJ)	-0.0182	0.370	0.176	0.436	0.236										
	Millsboro Dam (DE)	-0.139	-0.0545	0.491	-0.191	**-0.655**	-0.0636									
	Turville Creek (MD)	0.00606	-0.261	-0.0909	-0.155	-0.436	**-0.636**	0.418								
Chesapeake Bay	Clark's Millpond (PRFC)	-0.212	0.0424	-0.406	0.0455	0.118	0.209	-0.0818	-0.345							
	Gardy's Millpond (PRFC)	**0.648**	-0.200	0.103	0.500	0.364	-0.0636	-0.282	0.173	**-0.664**						
	Bracken's Pond (VA)	-0.188	-0.224	**0.685**	0.118	-0.236	-0.245	**0.636**	0.236	-0.200	-0.00909					
	Kamp's Millpond (VA)	**0.600**	0.406	-0.0303	0.164	0.0182	-0.0364	-0.00909	-0.273	0.0636	0.173	-0.0545				
	Wormley Creek (VA)	-0.152	0.309	-0.224	0.0788	-0.442	-0.152	**0.685**	0.479	0.248	-0.309	0.382	0.0667			
South Atlantic	Goose Creek (SC)	**0.564**	-0.0667	0.333	0.136	-0.0818	0.355	0.118	-0.255	-0.291	0.436	-0.164	**0.573**	-0.248		
	Altamaha Canal (GA)	-0.0424	0.188	0.309	-0.382	-0.200	0.0182	0.394	-0.127	-0.406	0.0667	0.200	0.0424	0.176	0.297	
	Guana River Dam (FL)	**0.770**	-0.0182	0.418	0.248	-0.236	-0.103	0.164	0.0303	-0.139	0.382	0.273	**0.709**	-0.0788	**0.552**	-0.236

Table 5.10. Summary of GLM analyses used to standardize fisheries-independent indices developed from non-ASMFC-mandated surveys. Phi is the overdispersion parameter.

Region	State	Survey	Location	Years	Gear	Life Stage(s)	GLM?	Error Structure	Response	Predictors	Phi
Southern New England	CT	CTDEP Electrofishing	Farmill River	2001–2010	Electrofishing	Elver & Yellow	N				
	NY	Western Long Island Study[9]	Long Island Sound	1984–2010	Seine	Yellow	Y	NB	Catch	Year+Month+System	0.611
Hudson River	NY	HRE Monitoring[10]	Hudson River	1974–2009	Epibenthic Sled and Tucker Trawl	YOY	Y	Delta-gamma	Catch	Year+Month+Gear+Strata+RiverMile+Volume	
	NY	HRE Monitoring[9,11]	Hudson River	1974–2009	Epibenthic Sled and Tucker Trawl	Yearling and older	Y	NB	Catch	Year+Month+Gear+Strata+RiverMile+Volume	1.66
	NY	NYDEC Alosine Beach Seine[9,11]	Hudson River	1980–2009	Seine	Elver & Yellow	Y	NB	Catch	Year+Month+RiverMile+WaterTemp	1.25
	NY	NYDEC Striped Bass Beach Seine[9,11]	Hudson River	1980–2009	Seine	Elver & Yellow	Y	NB	Catch	Year+Month+RiverMile+WaterTemp	1.28
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Little Egg Inlet Ichthyoplankton[10]	Little Egg Harbor	1992–2010	Plankton Net	YOY	Y	NB	Catch	Year+Month+Tidal Flow+Discharge+Salinity	1.05
	NJ	NJDFW Striped Bass Seine[9,11]	Delaware Bay	1980–2009	Seine	Yellow	Y	NB	Catch	Year+Month+Temp+Salinity	0.958
	DE	Delaware Trawl Survey[9]	Delaware River	1982–2010	Trawl	Elver & Yellow	Y	NB	Catch	Year+Month+Salinity+WaterTemp	1.02
	DE	PSEG Trawl[9,11,12]	Delaware River	1970–2010	Trawl	Elver & Yellow	Y	NB	Catch	Year+Month+BottomSalinity	1.85
	PA	Area 6 Electrofishing	Delaware River	1999–2010	Electrofishing	Elver	Y	NB	Catch	Year+Site	1.05
Chesapeake Bay	MD	MDDNR Striped Bass Seine[9,11,12]	Chesapeake Bay	1966–2010	Seine	Yellow	Y	NB	Catch	Year+Month+Salinity	0.973
	VA	North Anna Electrofishing[9]	North Anna River	1990–2009	Electrofishing	Elver & Yellow	Y	NB	Catch	Year+GearType+TimePeriod+Station	1.20
	VA	VIMS Juvenile Striped Bass Seine--long[11,12]	Lower Ches Bay & Tribs	1967–1973; 1980–2010	Seine	Yellow	Y	NB	Catch	Year+System	0.751
	VA	VIMS Juvenile Striped Bass Seine--short[9]	Lower Ches Bay & Tribs	1989–2010	Seine	Yellow	Y	NB	Catch	Year+Station Type+Salinity	1.07
South Atlantic	NC	Beaufort Inlet Ichthyoplankton[10]	Beaufort Inlet	1987–2003	Plankton Net	YOY	Y	NB	Catch	Year+Month+Discharge	1.14
	NC	NCDMF Estuarine Trawl[9]	NC waters	1989–2010	Trawl	Elver & Yellow	Y	NB	Catch	Year+Lat+Lon+BottomType	1.51
	SC	SC Electrofishing	SC waters	2001–2010	Electrofishing	Elver & Yellow	Y	NB	Catch	Year+Strata+WaterTemp+Salinity+TideCode	1.22

[9] Included in calculation of 20-year coast-wide, yellow-phase abundance index

[10] Included in calculation of long-term coast-wide recruitment index

[11] Included in calculation of 30-year coast-wide, yellow-phase abundance index

[12] Included in calculation of 40-plus coast-wide, yellow-phase abundance index

Table 5.11. Summary (A) length and (B) weight information by year from the Upper Delaware, all locations combined.

(A)

Statistic	2006	2007	2008	All
Average	357.35	362.98	377.79	362.88
St Dev	111.37	93.28	113.84	108.38
Min	147	189	172	147
Max	750	665	685	750
n	331	125	122	578

(B)

Statistic	2006	2007	2008	All
Average	105.39	102.48	121.83	108.23
St Dev	104.18	81.68	117.04	102.79
Min	3	9	7	3
Max	588	490	639	639
n	331	125	122	578

Table 6.1. Summary of surveys used in development of region-specific indices of American eel relative abundance. Asterisks (*) denote the ASMFC-mandated recruitment surveys.

Region	**Life Stage**	**Time Period**	**Survey**
Gulf of Maine	YOY	2001–2010	West Harbor Pond (ME) *
			Lamprey River (NH) *
			Jones River (MA) *
	Yellow		*none available*
Southern New England	YOY	2000–2010	Gilbert Stuart Dam (RI) *
			Carman's River (NY) *
	Yellow	2000–2010	CTDEP Electrofishing Survey (CT)
			Western Long Island Study (NY)
Hudson River	YOY	1974–2009	HRE Monitoring Program (NY)
	Yellow	1980–2009	HRE Monitoring Program (NY)
			NYDEC Alosine Beach Seine Survey (NY)
			NYDEC Striped Bass Beach Seine Survey (NY)
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	2000–2010	Little Egg Inlet Ichthyoplankton Survey (NJ)
			Patcong Creek (NJ) *
			Millsboro Dam (DE) *
			Turville Creek (MD) *
	Yellow	1999–2010	NJDFW Striped Bass Seine (NJ)
			Delaware Trawl Survey (DE)
			PSEG Trawl Survey (DE)
			Area 6 Electrofishing Survey (PA)
Chesapeake Bay	YOY	2000–2010	Clark's Millpond (PRFC) *
			Gardy's Millpond (PRFC) *
			Bracken's Pond (VA) *
			Kamp's Millpond (VA) *
			Wormley Creek (VA) *
	Yellow	1990–2010	MDDNR Striped Bass Seine (MD)
			North Anna Electrofishing Survey (VA)
			VIMS Juvenile Striped Bass Seine Survey—short (VA)
South Atlantic	YOY	2001–2010	Beaufort Inlet Ichthyoplankton Survey (NC)
			Goose Creek (SC) *
			Altamaha Canal (SC) *
			Guana River Dam (FL) *
	Yellow	2001–2010	NCDMF Estuarine Trawl Survey (NC)
			SC Electrofishing Survey (SC)

Table 6.2. Spearman's rank correlation between regional YOY indices for American eel. Values formatted in **bold** font are statistically significant at α < 0.10.

	Gulf of Maine	Southern New England	Hudson River	Delaware Bay/ Mid-Atlantic Coastal Bays	Chesapeake Bay
Southern New England	0.333				
Hudson River	**0.633**	0.261			
Delaware Bay/ Mid-Atlantic Coastal Bays	0.370	-0.191	0.309		
Chesapeake Bay	0.273	0.155	**0.879**	0.364	
South Atlantic	**0.612**	0.212	**0.767**	0.212	**0.612**

Table 6.3. Spearman's rank correlation between regional yellow-phase indices for American eel. Values formatted in **bold** font are statistically significant at α < 0.10.

	Southern New England	Hudson River	Delaware Bay/ Mid-Atlantic Coastal Bays	Chesapeake Bay
Hudson River	0.400			
Delaware Bay/ Mid-Atlantic Coastal Bays	-0.139	0.164		
Chesapeake Bay	0.442	-0.323	0.462	
South Atlantic	-0.442	0.100	0.309	-0.00606

Table 6.4. Spearman's rank correlation coefficients (ρ) and associated *P*-values from correlation of region-specific yellow-phase indices and lagged YOY indices for American eel. Values formatted in **bold** font are statistically significant at $\alpha < 0.10$.

Region	**Yellow vs.**	**Lag (years)**	ρ	$P > \lvert\rho\rvert$
Southern New England	YOY	0	-0.139	0.701
		1	-0.261	0.467
		2	-0.233	0.546
		3	-0.0476	0.911
		4	-0.429	0.337
Hudson River	YOY	0	-0.197	0.357
		1	0.0178	0.936
		2	-0.168	0.456
		3	0.0364	0.876
		4	-0.0677	0.777
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	0	0.0545	0.873
		1	-0.491	0.150
		2	-0.0333	0.932
		3	0.0476	0.911
		4	0.0357	0.939
Chesapeake Bay	YOY	0	**-0.627**	0.0388
		1	-0.176	0.627
		2	-0.0167	0.966
		3	0.310	0.456
		4	0.179	0.702
South Atlantic	YOY	0	0.224	0.533
		1	0.317	0.406
		2	0.381	0.352
		3	**0.750**	0.0522
		4	0.257	0.623

Table 6.5. Summary of the number and types of biological data for American eel compiled from past and current research programs along the Atlantic Coast.

Region	**Type**	**Length**			**Weight**			**Age**		
		Male	**Female**	**Other**	**Male**	**Female**	**Other**	**Male**	**Female**	**Other**
Gulf of Maine	Fish-Dep			56			55			
	Fish-Ind	1,978	2,036	11,581	1,419	1,324	623	873	872	622
Southern New England	Fish-Dep			187			196			
	Fish-Ind	402	73					847	117	
Hudson River	Fish-Dep			56			55			
	Fish-Ind	30	701	2,078	22	70	2,068	28	699	148
Del Bay/Mid-Atlantic Coastal Bays	Fish-Dep			6,718			6,680			3,624
	Fish-Ind	8	54	743	8	54	743	8	54	134
Chesapeake Bay	Fish-Dep	143	813	20,094	143	813	13,939	138	785	2,480
	Fish-Ind	156	240	11,547	156	240	10,009	152	237	1,050
	Mixed[13]			594						594
South Atlantic	Fish-Dep	1	332	4,486	1	332	1,443			
	Fish-Ind	15	404	24,392	15	401	8,563	11	296	264

Table 6.6. Parameter estimates (standard errors in parentheses) of the allometric length (mm)-weight (g) relation fit to available data for American eel by region, sex, and all data pooled. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	***a***	***b***
None	All	49,221	3.87E-07 (6.77E-09)	3.25 (0.00270)
Region	Gulf of Maine	3,420	6.49E-07 (3.54E-08)	3.17 (0.00834)
	Southern New England	143	3.88E-05 (3.30E-05*)	2.56 (0.131)
	Hudson River	2,215	1.27E-06 (1.99E-07)	3.06 (0.0244)
	Del Bay/Mid-Atl Coastal Bays	7,468	2.15E-07 (1.20E-08)	3.35 (0.00877)
	Chesapeake Bay	25,230	3.44E-07 (7.21E-09)	3.27 (0.00322)
	South Atlantic	10,745	1.00E-07 (5.72E-09)	3.48 (0.00902)
Sex	Male	1,764	2.88E-06 (4.66E-07)	2.91 (0.0275)
	Female	3,233	6.97E-07 (4.32E-08)	3.16 (0.00960)

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[13] Data provided by one study included samples from both fisheries-dependent and fisheries-independent sources and these data could not be separated by collection type

Table 6.7. Parameter estimates (standard errors in parentheses) for the linear regression of length (mm) on age (years) fit to available data for American eel by region, sex, and all data pooled. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	***Intercept***	***Slope***
None	All	13,532	329 (1.72)	8.33 (0.235)
Region	Gulf of Maine	2,356	87.5 (2.96)	23.5 (0.271)
	Southern New England	475	192 (18.7)	14.5 (1.57)
	Hudson River	875	238 (7.68)	13.7 (0.556)
	Del Bay/Mid-Atl Coastal Bays	3,820	243 (4.28)	37.0 (1.04)
	Chesapeake Bay	5,436	267 (3.61)	27.5 (0.731)
	South Atlantic	570	375 (13.5)	12.9 (2.68)
Sex	Male	1,604	279 (2.58)	4.76 (0.254)
	Female	3,015	368 (3.31)	6.70 (0.295)

Table 6.8. Parameter estimates (standard errors in parentheses) of the von Bertalanffy age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_∞ represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	L_∞	K	t_0
None	All	13,532	420 (1.81)	0.573 (0.0219)	-0.110 (0.0781*)
Region	Gulf of Maine	2,356	1,397 (191)	0.0220 (0.00392)	-2.15 (0.254)
	Southern New England	475	*failed to converge*		
	Hudson River	875	484 (5.36)	0.230 (0.0133)	0.347 (0.139*)
	Del Bay/Mid-Atl Coastal Bays	3,820	636 (41.6)	0.165 (0.0290)	-1.94 (0.385)
	Chesapeake Bay	5,436	779 (93.3)	0.0751 (0.0188)	-4.92 (0.711)
	South Atlantic	570	504 (42.7)	0.258 (0.176*)	-3.31 (3.20*)
Sex	Male	1,604	*failed to converge*		
	Female	3,015	*failed to converge*		

Table 6.9. Parameter estimates (standard errors in parentheses) of the Gompertz age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_∞ represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	L_∞	K	t_0
None	All	13,532	418 (1.72)	0.687 (0.0265)	-0.0297 (0.136*)
Region	Gulf of Maine	2,356	735 (26.4)	0.0944 (0.00451)	-17.2 (1.33)
	Southern New England	475	*failed to converge*		
	Hudson River	875	473 (4.30)	0.359 (0.0203)	-0.266 (0.382*)
	Del Bay/Mid-Atl Coastal Bays	3,820	588 (25.7)	0.259 (0.0302)	-4.84 (1.05)
	Chesapeake Bay	5,436	675 (46.8)	0.138 (0.0192)	-14.4 (2.70)
	South Atlantic	570	502 (39.4)	0.289 (0.179*)	-6.65 (7.12*)
Sex	Male	1,604	*failed to converge*		
	Female	3,015	1,425 (1,796*)	0.0130 (0.0141*)	-312 (353*)

Table 6.10. Parameter estimates (standard errors in parentheses) of the Richard's age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_∞ represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	L_∞	K	t_0	δ
None	All	13,532	415 (1.76)	1.05 (0.133)	1.69 (0.273)	-3.32 (1.11*)
Region	Gulf of Maine	2,356	*failed to converge*			
	Southern New England	475	*failed to converge*			
	Hudson River	875	478 (5.57)	0.292 (0.0453)	1.66 (0.776*)	0.506 (0.355*)
	Del Bay/Mid-Atl Coastal Bays	3,820	541 (28.1)	0.484 (0.203*)	2.40 (1.03*)	-2.40 (2.12*)
	Chesapeake Bay	5,436	*failed to converge*			
	South Atlantic	570	*failed to converge*			
Sex	Male	1,604	835 (2,810*)	0.252 (2.11*)	61.4 (231*)	-16.8 (140*)
	Female	3,015	514 (16.9)	1.11 (1.32*)	16.6 (2.39)	-69.0 (81.7*)

Table 6.11. Parameter estimates (standard errors in parentheses) of the logistic age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_∞ represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	L_∞	K	t_0
None	All	13,532	417 (1.66)	0.797 (0.0311)	0.974 (0.0557)
Region	Gulf of Maine	2,356	631 (14.8)	0.165 (0.00525)	9.67 (0.320)
	Southern New England	475	*failed to converge*		
	Hudson River	875	468 (3.96)	0.495 (0.0293)	3.74 (0.134)
	Del Bay/Mid-Atl Coastal Bays	3,820	562 (19.2)	0.353 (0.0316)	1.51 (0.156)
	Chesapeake Bay	5,436	629 (31.9)	0.200 (0.0197)	1.94 (0.475)
	South Atlantic	570	500 (36.8)	0.319 (0.183*)	-1.55 (1.76*)
Sex	Male	1,604	*failed to converge*		
	Female	3,015	929 (475*)	0.0291 (0.0140*)	14.4 (35.4*)

Table 6.12. Parameter estimates (standard errors in parentheses) of the Schnute age-length model fit to available data for American eel by region, sex, and all data pooled. Values of L_1 and L_2 represent length in millimeters. Asterisks (*) denote standard errors that are ≥ 30% of the parameter estimate.

Class	**Subset**	**n**	L_1	L_2	a	b
None	All	13,532	223 (6.17)	415 (1.76)	1.05 (0.133)	-3.32 (1.11*)
Region	Gulf of Maine	2,356	97.5 (5.86)	733 (29.8)	0.0371 (0.0191*)	0.794 (0.253*)
	Southern New England[14]	475	338 (6.65)	663 (49.1)	-1.39 (1.04*)	16.6 (14.4*)
	Hudson River	875	72.7 (12.2)	478 (5.56)	0.292 (0.0453)	0.507 (0.355*)
	Del Bay/Mid-Atl Coastal Bays	3,820	261 (8.91)	536 (20.8)	0.484 (0.203*)	-2.40 (2.12*)
	Chesapeake Bay	5,436	266 (8.45)	717 (42.9)	-0.151 (0.101*)	4.49 (1.47*)
	South Atlantic	570	*failed to converge*			
Sex	Male	1,604	*failed to converge*			
	Female	3,015	*failed to converge*			

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[14] Parameter estimates considered unrealistic

Table 6.13. Calculated AIC values (Akaike weights in parentheses) for age-length models fit to available data for American eel by region, sex, and all data pooled. Values in bold indicate the model with the smallest AIC value and largest Akaike weight for the associated dataset.

Class	**Subset**	**Linear**	**von Bertalanffy**	**Gompertz**	**Richards**	**Logistic**	**Schnute**
None	All	18.9806 (0.16235)	18.9187 (0.16745)	18.9180 (0.16752)	**18.9175 (0.16756)**	18.9176 (0.16755)	**18.9175 (0.16756)**
Region	Gulf of Maine	16.014 (0.19927)	**16.000 (0.20065)**	16.004 (0.20030)		16.015 (0.19920)	16.001 (0.20059)
	Southern New England	**14.5 (1.00)**					
	Hudson River	16.0695 (0.14004)	15.6563 (0.17218)	15.6557 (0.17222)	**15.6555 (0.17224)**	15.6691 (0.17108)	**15.6555 (0.17224)**
	Del Bay/Mid-Atl Coastal Bays	17.1333 (0.16585)	17.1221 (0.16678)	17.1215 (0.16683)	17.1215 (0.16683)	**17.1211 (0.16686)**	17.1215 (0.16683)
	Chesapeake Bay	17.968 (0.19980)	17.966 (0.20007)	17.966 (0.20003)		17.966 (0.19998)	**17.965 (0.20012)**
	South Atlantic	15.6313 (0.24964)	15.6276 (0.25011)	15.6275 (0.25012)		**15.6274 (0.25013)**	
Sex	Male	14.549 (0.4999)			**14.548 (0.5001)**		
	Female	16.979 (0.2500)		16.980 (0.2499)	**16.977 (0.2503)**	16.980 (0.2499)	

Table 6.14. Result of power analysis for linear and exponential trends in American eel abundance indices over a ten-year period. Power was calculated according to methods in Gerrodette (1987).

				Median	Linear Trend		Exponential Trend	
Region	**Life Stage**	**Survey**	**State**	**CV**	**+50%**	**-50%**	**+50%**	**-50%**
Gulf of Maine	YOY	YOY Survey—Jones River	MA	0.36	0.30	0.43	0.31	0.45
	YOY	YOY Survey—Lamprey River	NH	0.34	0.34	0.47	0.35	0.50
	YOY	YOY Survey—West Harbor Pond	ME	0.52	0.20	0.27	0.21	0.30
Southern New England	Elver & Yellow	CTDEP Electrofishing	CT	0.043	1.0	1.0	1.0	1.0
	YOY	YOY Survey—Carman's River	NY	0.20	0.65	0.83	0.65	0.84
	YOY	YOY Survey—Gilbert Stuart Dam	RI	0.24	0.53	0.72	0.54	0.73
Hudson River	Elver & Yellow	NYDEC Alosine Beach Seine	NY	0.18	0.75	0.91	0.75	0.91
	Elver & Yellow	NYDEC Striped Bass Beach Seine	NY	0.24	0.52	0.70	0.52	0.72
	Yearling and Older	HRE Monitoring Program	NY	0.078	1.0	1.0	1.0	1.0
	Yellow	Western Long Island Study	NY	1.0	0.11	0.13	0.12	0.17
	YOY	HRE Monitoring Program	NY	0.16	0.84	0.96	0.84	0.96
Delaware Bay/Mid-Atlantic Coastal Bays	Elver	Area 6 Electrofishing	PA	0.18	0.74	0.90	0.74	0.91
	Elver & Yellow	Delaware Trawl Survey	DE	0.35	0.33	0.46	0.34	0.48
	Elver & Yellow	PSEG Trawl Survey	DE	0.45	0.24	0.33	0.25	0.36
	Yellow	NJDFW Striped Bass Seine Survey	NJ	0.60	0.17	0.23	0.18	0.26
	YOY	Little Egg Inlet Ichthyoplankton Survey	NJ	0.19	0.72	0.89	0.72	0.89
	YOY	YOY Survey—Millsboro Dam	DE	0.31	0.38	0.53	0.39	0.55
	YOY	YOY Survey—Patcong Creek	NJ	0.25	0.50	0.68	0.51	0.70
	YOY	YOY Survey—Turville Creek	MD	0.26	0.47	0.64	0.47	0.66

Table 6.14. *Continued*.

Region	**Life Stage**	**Survey**	**State**	**Median**	**Linear Trend**		**Exponential Trend**	
				CV	**+50%**	**-50%**	**+50%**	**-50%**
Chesapeake Bay	Elver & Yellow	North Anna Electrofishing Survey	VA	0.24	0.54	0.72	0.54	0.74
	Yellow	MD Striped Bass Seine Survey	MD	0.66	0.15	0.20	0.17	0.23
	Yellow	VIMS Juvenile SB Seine Survey—long	VA	0.74	0.14	0.18	0.15	0.21
	Yellow	VIMS Juvenile SB Seine Survey—short	VA	0.55	0.19	0.25	0.20	0.28
	YOY	YOY Survey—Bracken's Pond	VA	0.24	0.52	0.70	0.53	0.72
	YOY	YOY Survey—Clark's Millpond	PRFC	0.28	0.43	0.59	0.44	0.61
	YOY	YOY Survey—Gardy's Millpond	PRFC	0.32	0.37	0.52	0.38	0.54
	YOY	YOY Survey—Kamp's Millpond	VA	0.26	0.47	0.65	0.48	0.67
	YOY	YOY Survey—Wormley Creek	VA	0.24	0.54	0.72	0.54	0.74
South Atlantic	Elver & Yellow	NCDMF Estuarine Trawl Survey	NC	0.28	0.44	0.61	0.44	0.62
	Elver & Yellow	SC Electrofishing Survey	SC	0.097	0.99	1.0	0.99	1.0
	YOY	Beaufort Inlet Ichthyoplankton Survey	NC	0.21	0.63	0.82	0.64	0.83
	YOY	YOY Survey—Altamaha Canal	GA	0.32	0.36	0.50	0.37	0.53
	YOY	YOY Survey—Goose Creek	SC	0.78	0.13	0.17	0.15	0.20
	YOY	YOY Survey—Guana River Dam	FL	0.28	0.43	0.60	0.44	0.62

Table 6.15. Results of the Mann-Kendall trend analysis applied to YOY indices developed from the ASMFC-mandated recruitment surveys. S is the Mann-Kendall statistic, Z_S is the test statistic when $n \geq 10$, P-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected (P-value < α; $\alpha = 0.05$). NS = not significant.

Region	**State**	**Location**	**Gear**	**Time Period**	**n ***	***S***	**Z_S**	***P*-value**	**Trend**
Gulf of Maine	ME	West Harbor Pond	Irish Elver Ramp	2001–2010	10	-3	-0.179	0.858	NS
	NH	Lamprey River	Irish Elver Trap	2001–2010	10	-7	-0.537	0.592	NS
	MA	Jones River	Sheldon Elver Trap	2001–2010	10	-15	-1.25	0.211	NS
Southern New England	RI	Gilbert Stuart Dam	Irish Elver Ramp	2000–2010	11	-11	-0.778	0.436	NS
	NY	Carman's River	Fyke Net	2000–2010	11	-15	-1.09	0.276	NS
Delaware Bay/ Mid-Atlantic Coastal Bays	NJ	Patcong Creek	Fyke Net	2000–2009	10	13	1.07	0.283	NS
	DE	Millsboro Dam	Fyke Net	2000–2010	11	-5	-0.311	0.755	NS
	MD	Turville Creek	Irish Elver Ramp	2000–2010	11	9	0.623	0.533	NS
Chesapeake Bay	PRFC	Clark's Millpond	Irish Elver Ramp	2000–2010	11	13	0.934	0.350	NS
	PRFC	Gardy's Millpond	Irish Elver Ramp	2000–2010	11	-13	-0.934	0.350	NS
	VA	Bracken's Pond	Irish Elver Ramp	2000–2010	11	-19	-1.40	0.161	NS
	VA	Kamp's Millpond	Irish Elver Ramp	2000–2010	11	-17	-1.25	0.213	NS
	VA	Wormley Creek	Irish Elver Ramp	2001–2010	10	-1	0	1.00	NS
South Atlantic	SC	Goose Creek	Fyke Net	2000–2010	11	-13	-0.934	0.350	NS
	GA	Altamaha Canal	Fyke Net	2001–2010	10	-15	-1.25	0.211	NS
	FL	Guana River Dam	Dip Net	2001–2010	10	-3	-0.179	0.858	NS

* Years with missing values included in count

Table 6.16. Results of the Mann-Kendall trend analysis applied to indices developed from non-ASMFC-mandated recruitment surveys. *S* is the Mann-Kendall statistic, Z_S is the test statistic when $n \geq 10$, *P*-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected ($P\text{-value} < \alpha$; $\alpha = 0.05$). NS = not significant. The length range of observed American eels is shown in parentheses after the life stage if the information was available.

Region	**Survey**	**Gear**	**Life Stage**	**Time Period**	**n ***	***S***	**Z_S**	***P*-value**	**Trend**
Southern New England	CTDEP Electrofishing Survey	Electrofishing	Elver & Yellow (50–590 mm)	2001–2010	10	23	1.97	0.0491	↑
	Western Long Island Study	Seine	Yellow (35–770 mm)	1984–2010	27	-152	-3.18	0.00148	↓
Hudson River	HRE Monitoring Program	Epibenthic Sled and Tucker Trawl	YOY	1974–2009	36	-167	-2.96	0.00306	↓
	HRE Monitoring Program	Epibenthic Sled and Tucker Trawl	Yearling and Older	1974–2009	36	-394	-5.35	8.65E-08	↓
	NYDEC Alosine Beach Seine	Seine	Elver & Yellow	1980–2009	30	-149	-2.64	0.00828	↓
	NYDEC Striped Bass Beach Seine	Seine	Elver & Yellow	1980–2009	30	-273	-4.85	1.22E-06	↓
Delaware Bay/ Mid-Atlantic Coastal Bays	Little Egg Inlet Ichthyoplankton Survey	Ichthyoplankton Net	YOY	1992–2010	19	-45	-1.54	0.124	NS
	NJDFW Striped Bass Seine Survey	Seine	Yellow (50–750 mm)	1980–2009	30	39	0.678	0.498	NS
	Delaware Trawl Survey	Trawl	Elver & Yellow (55–690 mm)	1982–2010	29	42	0.769	0.442	NS
	PSEG Trawl Survey	Trawl	Elver & Yellow (97–602 mm)	1970–2010	41	163	2.04	0.0417	↑
	Area 6 Electrofishing	Electrofishing	Elver	1999–2010	12	6	0.343	0.732	NS

* Years with missing values included in count

Table 6.16. *Continued.*

Region	**Survey**	**Gear**	**Life Stage**	**Time Period**	**n ***	***S***	**Z_S**	***P*-value**	**Trend**
Chesapeake Bay	MDDNR Striped Bass Seine Survey	Seine	Yellow (77–687 mm)	1966–2010	45	-41	-0.391	0.696	NS
	North Anna Electrofishing Survey	Electrofishing	Elver & Yellow (32–726 mm)	1990–2009	20	107	3.71	0.000209	↑
	VIMS Juvenile Striped Bass Seine Survey—short	Seine	Yellow	1989–2010	22	-25	-0.677	0.499	NS
	VIMS Juvenile Striped Bass Seine Survey—long	Seine	Yellow	1967–2010	44	-159	-7.29	3.03E-13	↓
South Atlantic	Beaufort Inlet Ichthyoplankton Survey	Ichthyoplankton Net	YOY	1987–2003	17	-30	-1.19	0.232	NS
	NCDMF Estuarine Trawl Survey	Trawl	Elver & Yellow (26–921 mm)	1989–2010	22	-93	-2.59	0.00948	↓
	SC Electrofishing Survey	Electrofishing	Elver & Yellow (44–890 mm)	2001–2010	10	-29	-2.50	0.0123	↓

* Years with missing values included in count

Table 6.17. Results of the Mann-Kendall trend analysis applied to regional and coast-wide indices of American eel abundance. S is the Mann-Kendall statistic, Z_S is the test statistic when $n \geq 10$, P-value is the two-tailed probability for the trend test, and trend indicates the direction of the trend if a statistically significant temporal trend was detected (P-value < α; $\alpha = 0.05$). NS = not significant.

Region	**Life Stage**	**Time Period**	**n ***	***S***	**Z_S**	***P*-value**	**Trend**
Gulf of Maine	YOY	2001–2010	10	-15	-1.25	0.211	NS
Southern New England	YOY	2000–2010	11	-15	-1.09	0.276	NS
	Yellow	2001–2010	10	21	1.79	0.0736	NS
Hudson River	YOY	1974–2009	36	-167	-2.96	0.00306	↓
	Yellow	1980–2009	30	-297	-5.28	0	↓
Delaware Bay/ Mid-Atlantic Coastal Bays	YOY	2000–2010	10	5	0.311	0.755	NS
	Yellow	1999–2010	12	-4	-0.206	0.837	NS
Chesapeake Bay	YOY	2000–2010	11	-21	-1.56	0.119	NS
	Yellow	1990–2010	21	108	3.23	0.00123	↑
South Atlantic	YOY	2001–2010	10	-17	-1.43	0.152	NS
	Yellow	2001–2010	10	-25	-2.15	0.0318	↓
Atlantic Coast	YOY (short-term)	2000–2010	11	-21	-1.56	0.119	NS
	YOY (long-term)	1987–2009	23	-39	-1.00	0.316	NS
	Yellow (40+ year)	1967–2010	44	52	0.516	0.606	NS
	Yellow (30-year)	1981–2010	30	-129	-2.28	0.0224	↓
	Yellow (20-year)	1991–2010	20	60	1.91	0.0556	NS

* Years with missing values included in count

Table 6.18. Results of the meta-analysis to synthesize trends for American eel. The meta-analysis techniques are from Manly (2001) where S_1 tests whether at least one of the datasets shows a significant decline through time and S_2 tests whether there is consensus among the datasets for a decline. S_2 incorporates a weight equal to the number of years of the survey, *n*. The value of *p* represents the one-tailed *p*-value from the Mann-Kendall nonparametric test for a decreasing trend through time.

Life stage	**Survey**	***n***	***p***	**Meta-analysis statistics**
Yellow	Area 6 Electrofishing	12	0.63	
	CTDEP Electrofishing Survey	10	0.98	
	NYDEC Alosine Beach Seine	30	0.0041	S_1: 175
	NYDEC Striped Bass Beach Seine	30	6.1E-07	df: 30
	Delaware Trawl Survey	29	0.78	$P(X^2>S_1\|\text{df})$: <0.01
	PSEG Trawl Survey	41	0.98	
	North Anna Electrofishing Survey	20	1.0	S_2: -6.29
	NCDMF Estuarine Trawl Survey	22	0.0047	$P(Z>S_2)$: <0.01
	SC Electrofishing Survey	10	0.0061	
	HRE Monitoring Program	36	4.3E-08	
	Western Long Island Study	27	7.4E-04	
	NJDFW Striped Bass Seine Survey	30	0.75	
	MD Striped Bass Seine Survey	45	0.35	
	VIMS Juvenile Striped Bass Seine Survey—short	22	0.25	
	VIMS Juvenile Striped Bass Seine Survey—long	44	1.5E-13	
YOY	YOY Survey—West Harbor Pond	10	0.43	
	YOY Survey—Lamprey River	10	0.30	
	YOY Survey—Jones River	10	0.11	S_1: 65.8
	YOY Survey—Gilbert Stuart Dam	11	0.22	df: 38
	YOY Survey—Carman's River	11	0.14	$P(X^2>S_1\|\text{df})$: <0.01
	HRE Monitoring Program	36	0.0015	
	Little Egg Inlet Ichthyoplankton Survey	19	0.062	S_2: -15.1
	YOY Survey—Patcong Creek	11	0.86	$P(Z>S_2)$: <0.01
	YOY Survey—Millsboro Dam	11	0.38	
	YOY Survey—Turville Creek	11	0.73	
	YOY Survey—Clarks Millpond	11	0.82	
	YOY Survey—Gardys Millpond	11	0.18	
	YOY Survey—Brackens Pond	11	0.081	
	YOY Survey—Kamps Millpond	11	0.11	
	YOY Survey—Wormley Creek	10	0.50	
	Beaufort Inlet Ichthyoplankton Survey	17	0.12	
	YOY Survey—Goose Creek	11	0.18	
	YOY Survey—Altamaha Canal	10	0.11	
	YOY Survey—Guana River Dam	10	0.43	

Table 6.19. Summary statistics from ARIMA model fits to American eel surveys with 20 or more years of data. Q0.25 is the 25th percentile of the fitted values; *P*(<0.25) is the probability of the final year of the survey being below Q0.25 with 80% confidence; r1–r3 are the first three autocorrelations; θ is the moving average parameter; SE is the standard error of θ; and σ^2_c is the variance of the index.

Region	**Survey**	**Final Year**	**Q0.25**	***P*(<0.25)**	**n**	**r1**	**r2**	**r3**	**θ**	**SE**	**σ^2_c**
Hudson River	Western Long Island Study	2010	-4.24	0.513	27	-0.32	-0.1	-0.02	0.26	0.17	0.42
	HRE Monitoring Program	2009	-3.96	0.548	30	-0.21	-0.18	-0.04	0.07	0.2	1.24
	HRE Monitoring Program	2009	-2.14	0.259	36	-0.07	-0.24	0.24	0.41	0.15	0.29
	NYDEC Alosine Beach Seine	2009	-1.20	0.316	30	-0.37	0.06	-0.15	0.64	0.16	0.29
	NYDEC Striped Bass Beach Seine	2009	-1.30	0.47	30	0.02	-0.15	-0.11	0.44	0.48	0.22
Delaware Bay/Mid-Atlantic Coastal Bays	NJDFW Striped Bass Seine Survey	2009	-2.55	0.003	30	-0.2	-0.42	0.12	1	0.12	1.08
	Delaware Trawl Survey	2010	-0.68	0.141	29	-0.36	-0.1	0.2	0.97	0.3	0.27
	PSEG Trawl Survey	2010	-0.60	0.069	38	-0.13	-0.01	-0.26	0.93	0.18	1.39
Chesapeake Bay	MD Striped Bass Seine Survey	2010	-1.70	0.116	45	-0.27	0.01	-0.12	0.83	0.22	1.49
	VIMS Juvenile SB Seine Survey—short	2010	-2.53	0.164	22	-0.24	-0.39	-0.01	0.9	0.49	0.39
	VIMS Juvenile SB Seine Survey—long	2010	-3.36	0.062	38	-0.26	-0.44	0.15	0.66	0.13	0.87
South Atlantic	NCDMF Estuarine Trawl Survey	2010	-1.99	0.308	22	-0.58	0.13	0.04	0.77	0.12	0.45

Table 6.20. Traffic Light representation of YOY indices developed from the ASMFC-mandated recruitment surveys. The 25th and 75th percentiles used to define the shading for each index series such that positive (white) values are > 75th percentile, neutral (gray) values are between the 25th and 75th percentiles, and negative (black) values are < 25th percentile.

	Region	**Gulf of Maine**			**Southern New England**		**Delaware Bay/ Mid-Atlantic**			**Chesapeake Bay**					**South Atlantic**		
	State	**ME**	**NH**	**MA**	**RI**	**NY**	**NJ**	**DE**	**MD**	**PRFC**		**VA**			**SC**	**GA**	**FL**
	Location	**West Harbor Dam**	**Lamprey River**	**Jones River**	**Gilbert Stuart Dam**	**Carman's River**	**Patcong Creek**	**Millsboro Dam**	**Turville Creek**	**Clark's Millpond**	**Gardy's Millpond**	**Bracken's Pond**	**Kamp's Millpond**	**Wormley Creek**	**Goose Creek**	**Altamaha Canal**	**Guana River Dam**
Year	**2000**	▨	▨	▨	356	43.3	55.7	4,454	5,423	0.334	28.5	1,038	15.4	▨	16.2	▨	▨
	2001	3,861	5.28	543	27.5	7.59	300	11,736	6,162	0.176	23.2	480	136	908	246	9.84	102
	2002	1,187	18.3	93.0	679	345	2,182	3,344	647	2.68	4.49	128	474	481	144	1.27	24.2
	2003	523	1.71	902	3.38	6.34	57.1	8,180	3,489	0.528	1.98	981	61.2	207	105	1.39	47.9
	2004	88.3	3.53	118	6.59	25.2	63.4	5,092	3,422	3.52	0.964	348	8.48	797	4.49	1.55	7.84
	2005	3,719	1.85	809	48.2	16.0	712	5,307	1,263	4.90	2.78	741	91.0	378	101	1.19	150
	2006	138	42.8	492	20.8	7.32	3,502	6,812	1,377	1.44	1.04	520	7.50	877	36.9	3.11	8.55
	2007	105	0.882	449	44.6	11.3	318	12,904	7,362	1.79	4.47	866	3.93	1,430	80.0	1.31	12.4
	2008	1,894	0.997	219	10.1	14.7	291	1,166	3,171	0.646	7.24	21.2	17.3	125	141	1.69	15.9
	2009	1,406	2.408	264	35.7	23.5	356	846	4,260	0.606	6.28	1.64	4.61	113	56.8	0.723	18.5
	2010	1,845	4.97	39.2	16.5	6.04	▨	6,539	8,636	3.28	1.94	412	66.2	2,575	34.8	0.878	30.6
%ile	**25th**	235	1.74	143	13.3	7.45	120	3,899	2,274	0.567	1.96	238	7.99	250	35.8	1.21	13.2
	75th	1,882	5.20	530	46.4	24.4	623	7,496	5,793	2.98	6.76	804	78.6	900	123	1.66	43.6

Table 6.21. Traffic Light representation of indices developed from non-ASMFC-mandated recruitment surveys. The 25th and 75th percentiles used to define the color boundaries for each index series are also shown. The 25th and 75th percentiles used to define the shading for each index series such that positive (white) values are > 75th percentile, neutral (gray) values are between the 25th and 75th percentiles, and negative (black) values are < 25th percentile.

	Region	**Southern New England**		**Hudson River**				**Delaware Bay/ Mid-Atlantic Coastal Bays**				
	Survey	**CTDEP Electrofishing**	**Western Long Island Sound**	**HRE Monitoring**		**NYDEC Alosine Beach Seine**	**NYDEC Striped Bass Beach Seine**	**Little Egg Inlet Ichthyoplankton**	**NJDEP Striped Bass Seine**	**Delaware Trawl**	**PSEG Trawl**	**Area 6 Electrofishing**
	Life Stage	**E/Y**	**Y**	**YOY**	**E/Y**	**E/Y**	**E/Y**	**YOY**	**Y**	**E/Y**	**E/Y**	**E**
Year	**1966**											
	1967											
	1968											
	1969											
	1970										0.437	
	1971										0.641	
	1972										0.487	
	1973										0.268	
	1974			0.197	1.39						0.212	
	1975			0.0799	1.63						0.276	
	1976			0.0843	1.11						0.328	
	1977			0.0328	0.603						0.208	
	1978			0.106	0.382						0.189	
	1979			0.247	0.350						1.25	
	1980			0.0684	0.496	1.13	0.401		7.74E-14		0.480	
	1981			0.00235	0.667	1.18	1.14		0.0901		2.14	
	1982				0.593	0.620	1.28		1.01	1.50		
	1983				0.799	0.665	1.40		0.531	0.600		
	1984		0.367	0.000961	1.24	0.287	1.36		2.59E-14	0.451	5.24	
	1985		0.691		0.702	0.578	0.720		0.174	0.318	8.02	
	1986		0.177		0.618	0.990	0.434		0.349	0.486	2.49	
	1987		0.0719		0.843	0.704	0.729		1.73E-14	0.370	2.38	
	1988		0.0670	0.00546	0.322	0.490	0.990		0.0933	0.379	2.65E-11	
	1989		0.0932	0.0368	0.371	0.357	0.633		0.0669	0.368	0.277	
	1990		0.0102		0.538	0.315	0.673		0.0456	0.240	0.709	
	1991		0.00810	0.0593	0.399	0.267	0.646		0.0257	0.359	0.275	
	1992		0.0168	0.0587	0.182	0.341	0.517	1.43	0.0866	1.009	0.484	
	1993		0.0240	0.218	0.183	0.237	0.228	1.77	0.0315	0.697	0.503	
	1994		0.0114	0.0864	0.257	0.234	0.385	2.32	0.126	0.166		
	1995		2.22E-16	0.0456	0.256	0.260	0.346	2.30	0.0559	0.718	0.549	
	1996		2.22E-16	0.0450	0.403	0.344	0.271	1.55	0.0881	0.563	0.0815	
	1997		2.22E-16	0.0263	0.133	0.0739	0.613	1.36	0.0818	0.626	1.04	
	1998		0.0108	0.0387	0.0857	0.368	0.317	1.76	0.0445	0.650	0.919	
	1999		2.22E-16	0.0160	0.0812	0.494	0.423	1.09	0.0504	0.986	0.887	23.4
	2000		0.0130	0.0365	0.0670	0.423	0.236	0.845	0.0739	0.231	0.490	8.64
	2001	257	0.00630	0.0595	0.167	0.350	0.210	1.37	0.0580	0.793	0.692	30.6
	2002	179	0.00750	0.0277	0.0324	0.782	0.222	1.22	0.0758	0.614	0.625	23.0
	2003	151	0.0174	0.0320	0.107	0.274	0.318	1.03	0.0531	0.419	1.38	18.0
	2004	272	0.0237	0.0133	0.126	0.484	0.389	0.806	0.326	1.30	0.962	25.1
	2005	225	0.0175	0.0557	0.0768	0.166	0.142	1.61	0.132	0.907	3.13	17.1
	2006	227	2.22E-16	0.0143	0.0877	0.256	0.141	1.68	0.131	0.542	1.17	23.3
	2007	241	0.0122	0.0247	0.165	0.405	0.140	1.81	0.0967	0.615	0.604	26.3
	2008	340	2.22E-16	0.00926	0.0922	0.404	0.232	1.82	0.0813	0.360	0.644	23.1
	2009	283	0.00790	0.00994	0.191	0.197	0.385	0.996	0.167	0.742	1.36	32.4
	2010	337	2.22E-16					0.332		0.469	0.547	17.5
	25th	226	0.00315	0.0182	0.132	0.269	0.245	1.06	0.0511	0.370	0.356	17.8
%ile	**75th**	280	0.0239	0.0662	0.607	0.557	0.666	1.77	0.130	0.718	1.14	25.4

Table 6.21. *Continued.*

	Region	**Chesapeake Bay**				**South Atlantic**		
	Survey	**MDDNR Striped Bass Seine**	**North Anna Electrofishing**	**VIMS Juvenile Striped Bass Seine—short**	**VIMS Juvenile Striped Bass Seine—long**	**Beaufort Inlet Ichthyoplankton**	**NCDMF Estuarine Trawl**	**SC Electrofishing**
	Life Stage	**Y**	**E/Y**	**Y**	**Y**	**YOY**	**E/Y**	**E/Y**
Year	**1966**	1.97						
	1967	0.132			0.184			
	1968	0.202			0.247			
	1969	0.109			0.144			
	1970	0.503			0.121			
	1971	0.521			0.190			
	1972	2.22E-16			0.117			
	1973	0.0967			0			
	1974	0.242						
	1975	0.929						
	1976	0.380						
	1977	0.527						
	1978	0.651						
	1979	0.620						
	1980	0.285			0.0801			
	1981	0.407			0.109			
	1982	0.255			0.0701			
	1983	2.58E-16			0			
	1984	0.313			0.0667			
	1985	0.337			0.0256			
	1986	0.195			0.0276			
	1987	0.166			0.0256	0.643		
	1988	0.505			0.0434	1.02		
	1989	0.122		0.0989	0	0.842	0.146	
	1990	0.0529	6.72	0.0441	0	0.624	0.561	
	1991	2.22E-16	6.15	0.0512	0	0.260	0.307	
	1992	0.0604	6.94	0.0735	0	1.13	0.348	
	1993	0.0632	3.34	0.0894	0.0219	0.610	0.199	
	1994	0.0545	3.82	0.114	0	1.54	0.315	
	1995	0.0765	5.26	0.0822	0.0430	1.54	0.192	
	1996	0.0583	9.34	0.209	0.0602	0.609	0.363	
	1997	0.588	7.30	0.287	0.0621	0.330	0.114	
	1998	0.366	5.74	0.0354	0.0420	1.10	0.113	
	1999	0.539	7.45	0.108	0.0219	0.170	0.364	
	2000	0.363	7.57	0.167	0.0861	0.280	0.0351	
	2001	0.198	11.6	0.235	0.0602	0.409	0.131	1.06
	2002	0.211	6.28	0.0894	0.0406	0.959	0.185	0.804
	2003	0.894		0.0299	0	0.406	0.106	1.27
	2004	0.395	18.5	0.0912	0.0420		0.279	0.973
	2005	0.900	10.9	0.0675	0.0892		0.169	0.947
	2006	0.0546	11.6	0.0979	0		0.112	0.852
	2007	0.0661	10.6	0.0665	0.0213		0.0925	0.703
	2008	0.595	23.8	0.0734	0.0694		0.143	0.667
	2009	0.256	33.6	0.0888	0.126		0.0422	0.803
	2010	0.237		0.0317	0.0420		0.161	0.692
	25th	0.0967	6.21	0.0667	0.0215	0.406	0.113	0.728
%ile	**75th**	0.505	11.3	0.106	0.0846	1.02	0.300	0.966

Table 6.22. Traffic Light representation of regional and coast-wide indices of American eel abundance. The 25th and 75th percentiles used to define the shading for each index series such that positive (white) values are > 75th percentile, neutral (gray) values are between the 25th and 75th percentiles, and negative (black) values are < 25th percentile.

	Region	Gulf of Maine	Southern New		Hudson River		Delaware Bay/ Mid-Atlantic Coastal		Chesapeake Bay		South Atlantic		Atlantic Coast				
	Life Stage	**YOY**	**YOY**	**Y**	**YOY**	**Y**	**YOY**	**Y**	**YOY**	**Y**	**YOY**	**Y**	**YOY (short-term)**	**YOY (long-term)**	**Y (40+ year)**	**Y (30-year)**	**Y (20-year)**
Year	**1967**														0.805		
	1968														0.887		
	1969														0.767		
	1970														0.830		
	1971														0.924		
	1972														0.669		
	1973														0.608		
	1974				0.197										0.587		
	1975				0.0799										0.851		
	1976				0.0843										0.690		
	1977				0.0328										0.690		
	1978				0.106										0.720		
	1979				0.247										1.13		
	1980				0.0684	1.03									0.759		
	1981				0.00235	1.34									1.13	1.15	
	1982					1.17									0.804	1.17	
	1983					1.29									0.612	1.05	
	1984				0.000961	1.24									1.37	1.19	
	1985					1.05									1.54	1.21	
	1986					1.05									1.02	1.05	
	1987					1.14								1.14	0.997	0.997	
	1988				0.00546	0.965								1.29	0.645	0.790	
	1989				0.0368	0.856								1.25	0.619	0.732	
	1990					0.903				0.624				1.11	0.690	0.781	
	1991				0.0593	0.837				0.590				0.961	0.575	0.696	1.05
	1992				0.0587	0.756				0.644				1.30	0.647	0.711	1.14
	1993				0.218	0.647				0.523				1.36	0.661	0.663	0.988
	1994				0.0864	0.714				0.548				1.70	0.645	0.731	1.01
	1995				0.0456	0.711				0.600				1.49	0.686	0.705	1.06
	1996				0.0450	0.758				0.758				1.21	0.563	0.671	1.10
	1997				0.0263	0.669				0.908				1.03	0.974	0.804	1.19
	1998				0.0387	0.675				0.687				1.37	0.868	0.761	1.09
	1999				0.0160	0.734		23.7		0.825				0.916	0.904	0.803	1.24
	2000		103		0.0365	0.659	1.58	14.5	632	0.804			4.89	0.897	0.789	0.728	1.05
	2001	1,034	12.4	252	0.0595	0.668	3.44	23.7	803	0.884	2.08	0.680	7.62	1.10	0.770	0.722	1.17
	2002	575	399	211	0.0277	0.715	2.30	21.1	696	0.682	0.924	0.649	7.26	1.17	0.751	0.739	1.10
	2003	457	4.25	196	0.0320	0.658	1.71	21.3	408	0.987	0.913	0.710	3.21	1.01	1.09	0.831	1.19
	2004	157	11.1	264	0.0133	0.740	1.49	28.4	349	1.04	0.169	0.735	2.05	0.955	0.886	0.847	1.40
	2005	863	23.4	239	0.0557	0.568	2.43	28.5	692	1.01	1.13	0.675	6.00	1.21	1.43	0.894	1.31
	2006	646	10.6	236	0.0143	0.595	3.96	23.5	328	0.765	0.410	0.624	3.80	1.17	0.77	0.699	1.11
	2007	181	19.0	246	0.0247	0.659	3.93	21.9	466	0.737	0.473	0.579	3.67	1.20	0.684	0.688	1.09
	2008	384	10.5	288	0.00926	0.662	1.71	19.9	172	1.19	0.660	0.594	2.48	1.24	0.888	0.769	1.22
	2009	462	24.4	265	0.00994	0.681	1.60	27.8	80.0	1.19	0.422	0.577	2.09	1.04	0.953	0.818	1.32
	2010	332	8.70	287			2.74	17.8	744	0.719	0.440	0.609	4.13		0.743	0.720	1.04
	25th	345	10.5	236	0.0182	0.669	1.66	20.8	338	0.644	0.427	0.598	2.84	1.03	0.680	0.721	1.05
%ile	**75th**	629	23.9	265	0.0662	1.01	3.09	24.7	694	0.908	0.922	0.679	5.44	1.27	0.909	0.882	1.20

Table 6.23. Summary of stochastic sensitivity runs conducted for the DB-SRA model.

Run	M*	M regime	Initial F:M*	B.mnpl*	B.ratio*	Harvest
1	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	Reconstructed harvest
2	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	**Lower harvest 1880–1885**
3	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	**High harvest 1870–1879**
4	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	**Ramp up harvest 1870–1879**
5	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	**Start in 1925**
6	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	3–10%	**Start in 1970**
7	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	**18–25%**	Reconstructed harvest
7A	0.15 to 0.25	Constant	0.8 to 1.2	0.25 to 0.5	**40–50%**	Reconstructed harvest
8	0.15 to 0.25	**increase by 20–40% in 1970**	0.8 to 1.2	0.25 to 0.5	3–10%	Reconstructed harvest
9	0.15 to 0.25	**increase by 15–30% in 1970**	0.8 to 1.2	0.25 to 0.5	3–10%	Reconstructed harvest
10	0.15 to 0.25	**increase by 15–30% in 1970**	0.8 to 1.2	0.25 to 0.5	**15–25%**	Reconstructed harvest
11	0.15 to 0.25	**increase by 15–30% in 1970**	0.8 to 1.2	0.25 to 0.5	**40–50%**	Reconstructed harvest
12	0.15 to 0.25	**increase by 15–30% in 1970**	0.8 to 1.2	0.25 to 0.5	**15–25%**	**Run 2 harvest**
13	0.15 to 0.25	**increase by 15–30% in 1960**	0.8 to 1.2	0.25 to 0.5	**15–25%**	Reconstructed harvest

Table 6.24. Summarized results from the DB-SRA (A) single and (B) double *M* models.

(A) Single *M* stanza model

Parameter	**Percentile**								
	0.025	0.05	0.1	0.25	0.5	0.75	0.9	0.95	0.975
K	16,220	16,405	16,686	17,314	18,219	19,180	20,126	20,704	21,253
B_{MSY}	5,080	5,218	5,439	5,991	6,770	7,550	8,134	8,440	8,664
F_{MSY}	0.1344	0.1408	0.1493	0.1687	0.1901	0.2119	0.2304	0.2388	0.2443
u_{MSY}	0.1165	0.1213	0.1280	0.1425	0.1579	0.1729	0.1854	0.1913	0.1948
MSY	755	789	834	926	1,057	1,190	1,292	1,338	1,368

(B) Double *M* stanza model

Parameter	**Percentile**								
	0.025	0.05	0.1	0.25	0.5	0.75	0.9	0.95	0.975
K	16,274	16,445	16,744	17,365	18,274	19,324	21,214	22,496	23,595
B_{MSY}	5,085	5,299	5,561	6,095	6,823	7,579	8,194	8,581	8,912
$F_{\mathrm{MSY\text{-}early}}$	0.1358	0.1420	0.1510	0.1696	0.1922	0.2155	0.2349	0.2441	0.2500
$F_{\mathrm{MSY\text{-}late}}$	0.0976	0.1037	0.1115	0.1281	0.1481	0.1685	0.1869	0.1956	0.2018
$u_{\mathrm{MSY\text{-}early}}$	0.1176	0.1224	0.1293	0.1433	0.1592	0.1751	0.1885	0.1948	0.1986
$u_{\mathrm{MSY\text{-}late}}$	0.0840	0.0890	0.0949	0.1076	0.1225	0.1376	0.1505	0.1568	0.1609
MSY-early	827	850	880	945	1,060	1,197	1,305	1,374	1,510
MSY-late	614	636	660	711	810	930	1,041	1,110	1,178

13 FIGURES

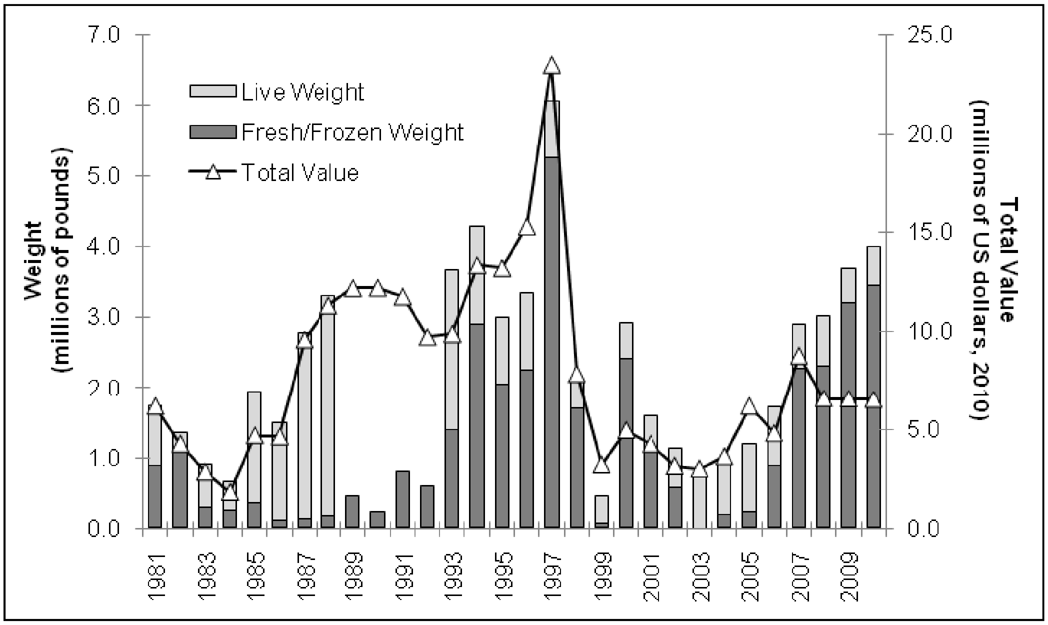

Figure 4.1. Annual U.S. domestic exports of American eels from districts along the Atlantic coast, 1981–2010. Note that the weights of live exports were not available for 1989 to 1992.

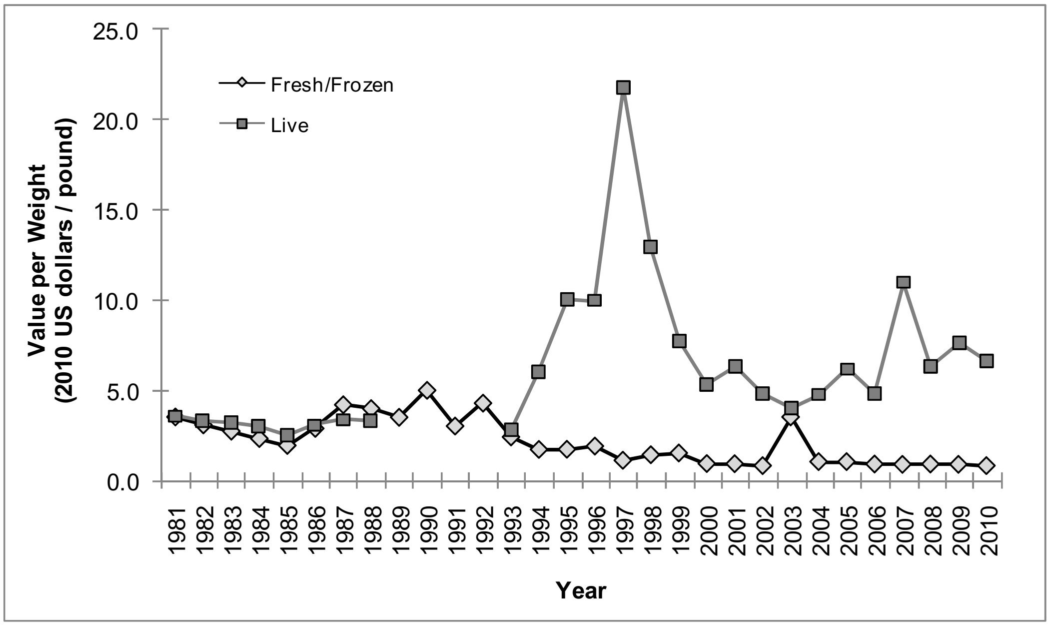

Figure 4.2. Annual U.S. domestic exports of American eels from districts along the Atlantic coast, 1981–2010. Note that the weights of live exports were not available for 1989 to 1992.

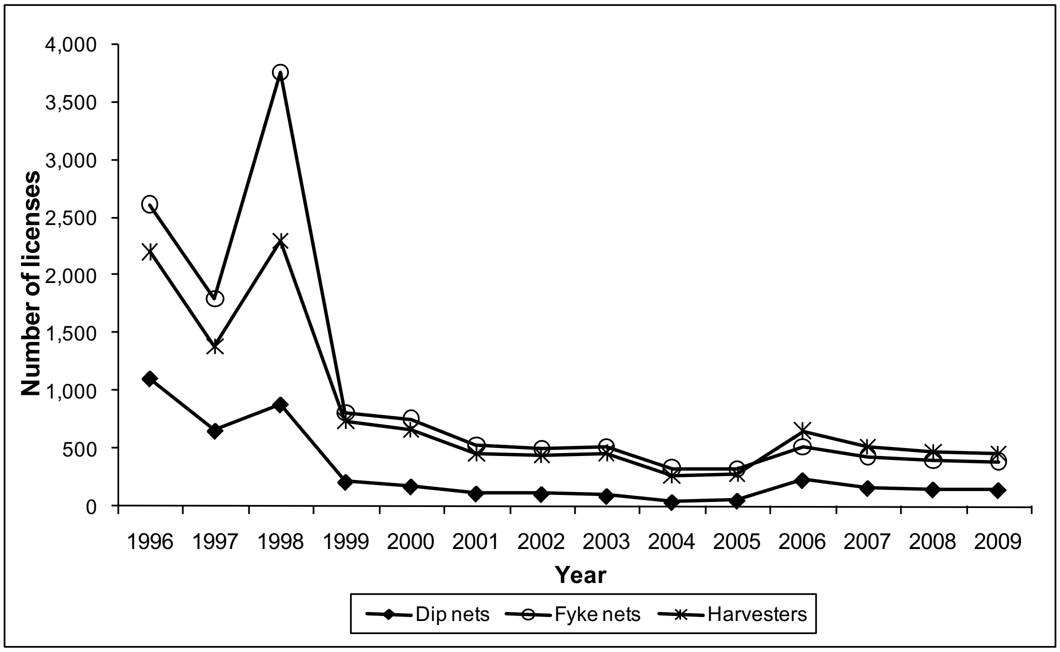

Figure 4.3. Commercial glass eel fishery effort in Maine, 1996–2009. Note: the number of harvesters does not equal the sum of the licensed gears since each harvester may license more than one piece of gear.

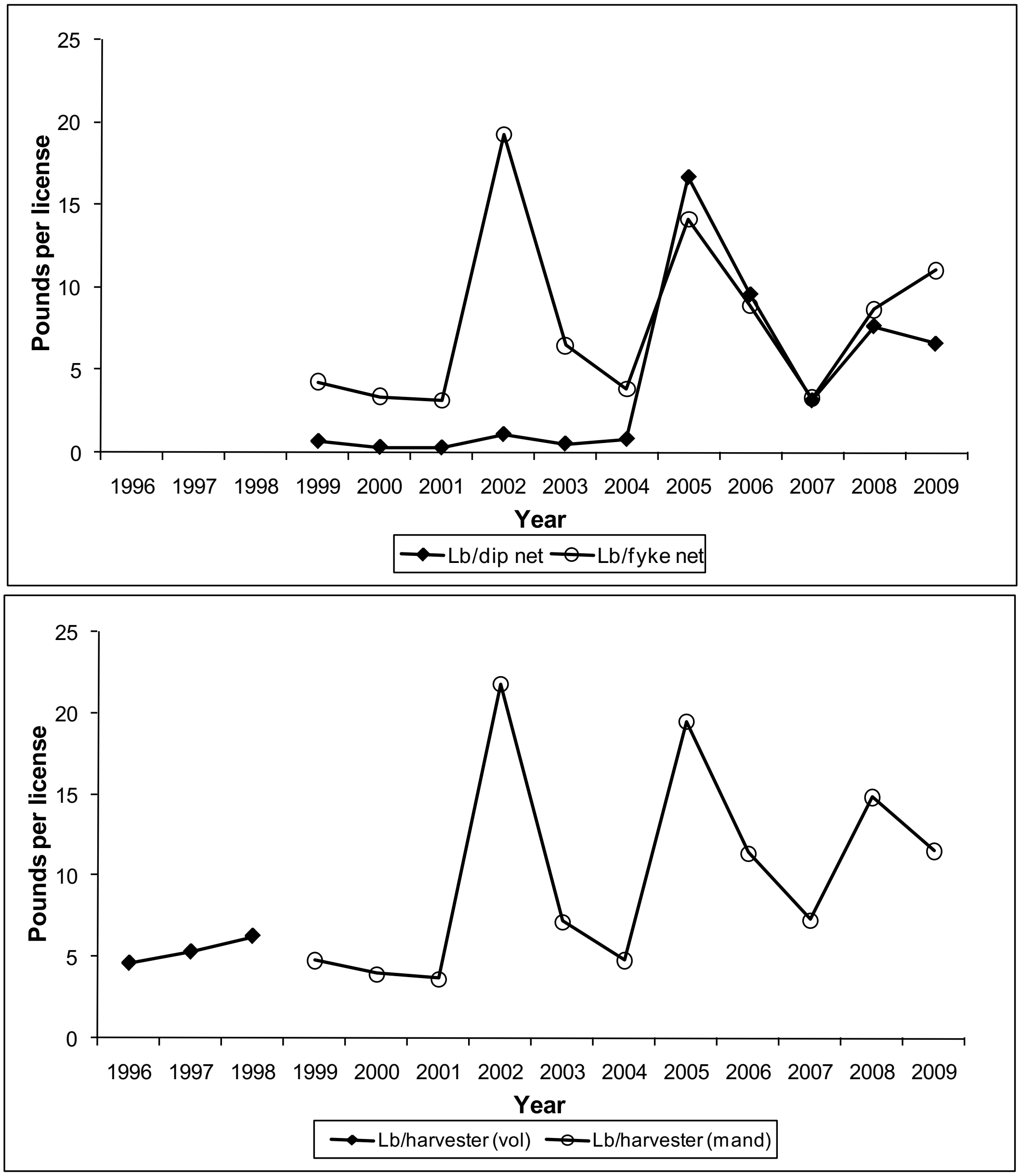

Figure 4.4. Catch per unit effort in the Maine commercial glass eel fishery per licensed gear (upper graph) and per license holder (lower graph).

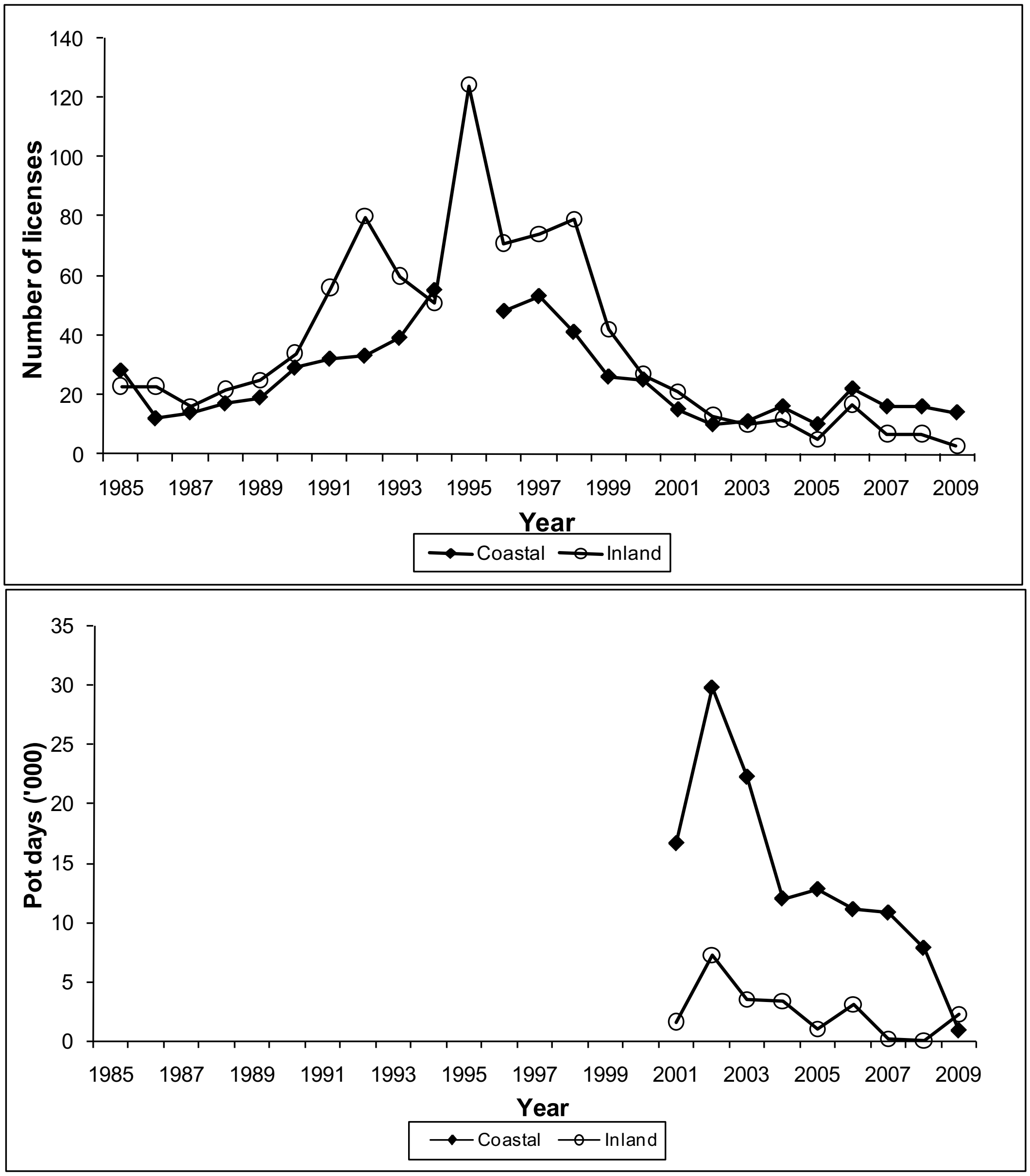

Figure 4.5. Effort in the Maine commercial yellow eel pot fisheries expressed as number of licensees (upper graph) and number of gear days fished (lower graph).

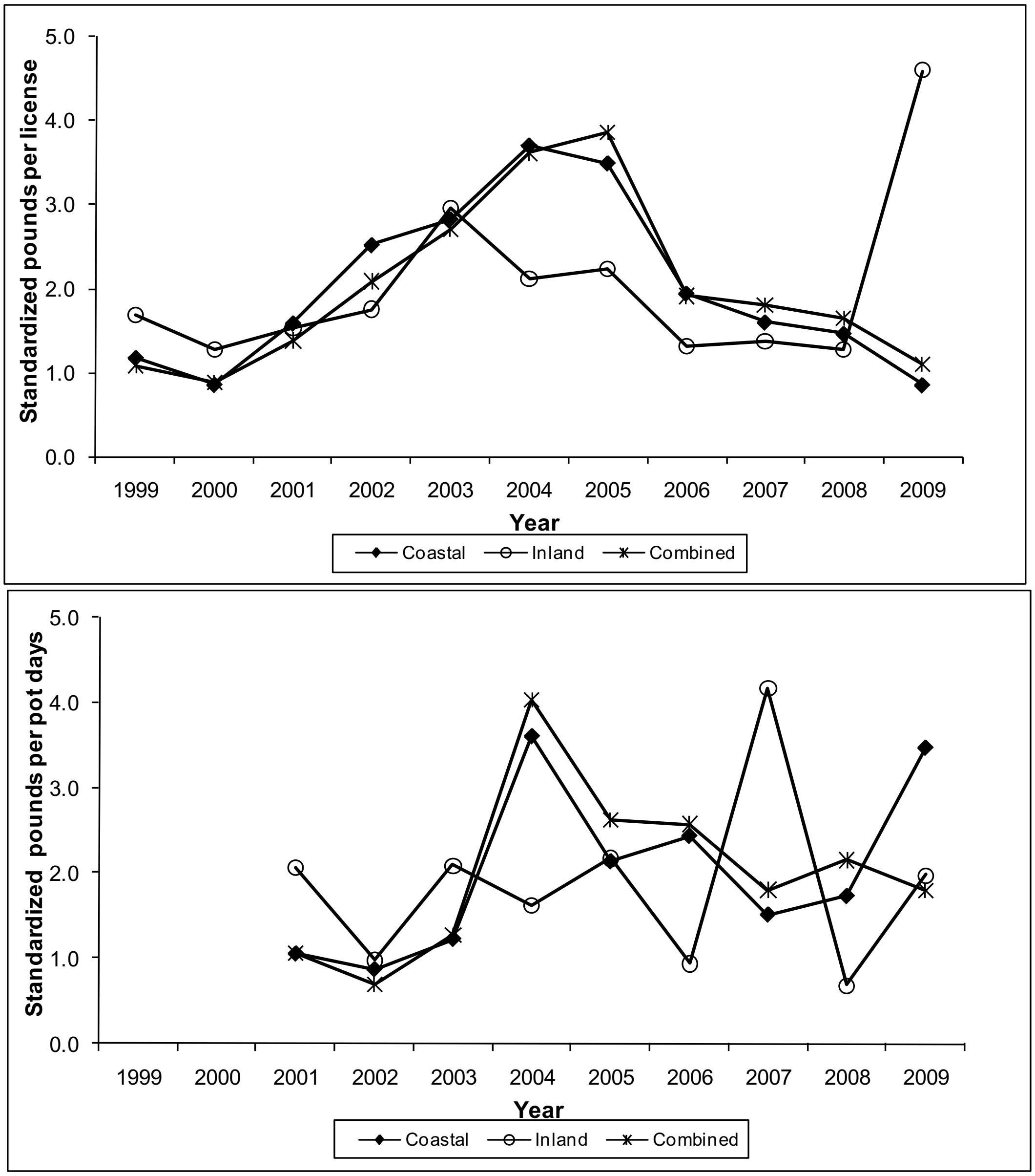

Figure 4.6. Standardized catch per unit effort in the Maine commercial yellow eel pot fisheries expressed as pounds per license holder (upper graph) and pounds per pot days (lower graph).

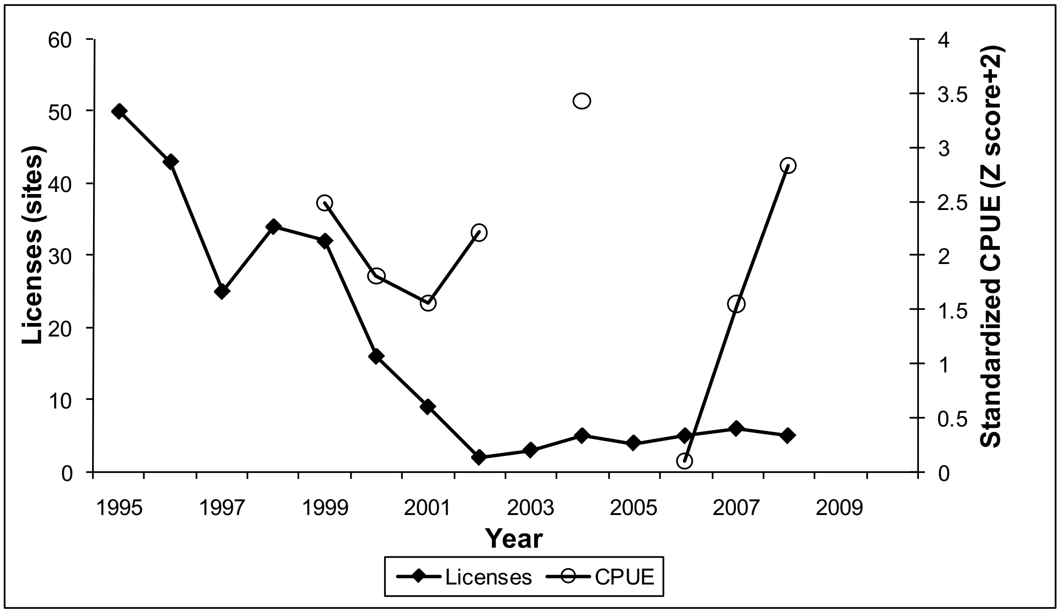

Figure 4.7. Standardized effort and CPUE from the Maine commercial silver eel weir fishery.

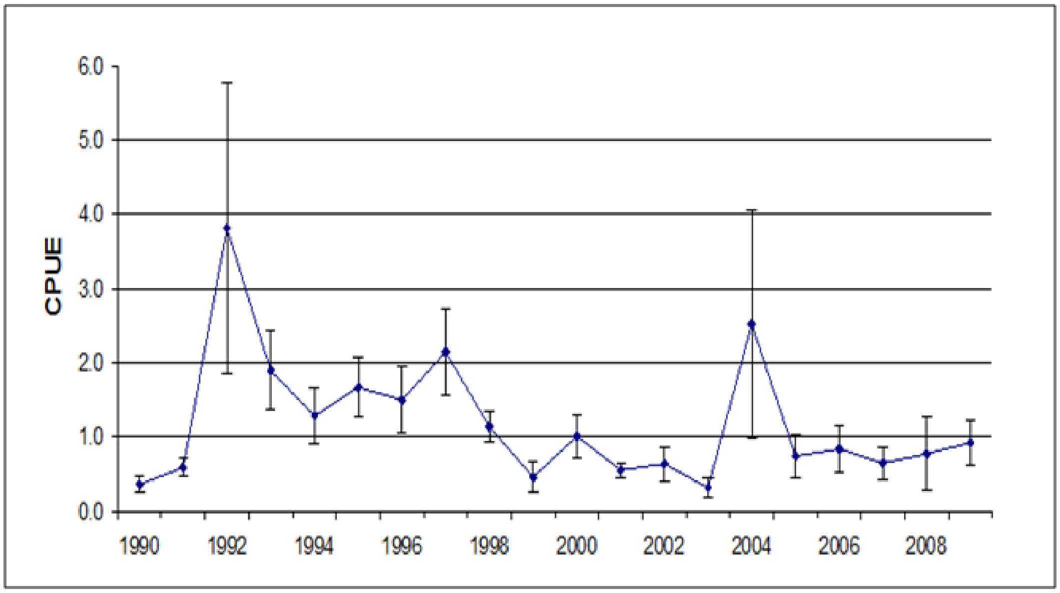

Figure 4.8. Catch-per-unit-effort in New Hampshire commercial eel pot fishery, 1990–2009. Error bars represent ± 2 standard errors.

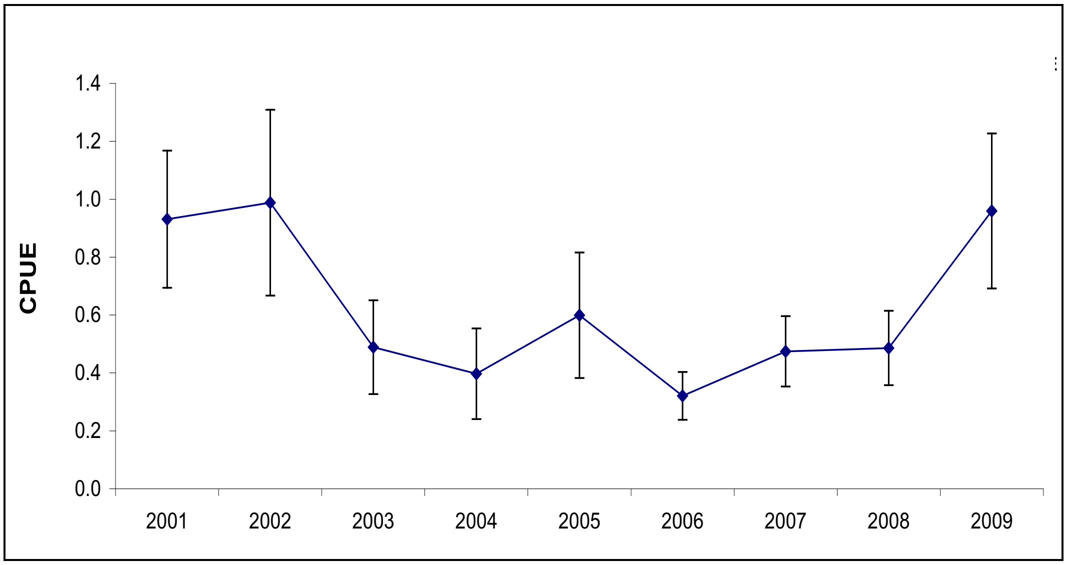

Figure 4.9. Catch-per-unit-effort in Massachusetts commercial eel pot fishery in Southern New England region, 2001–2009. Error bars represent ± 2 standard errors.

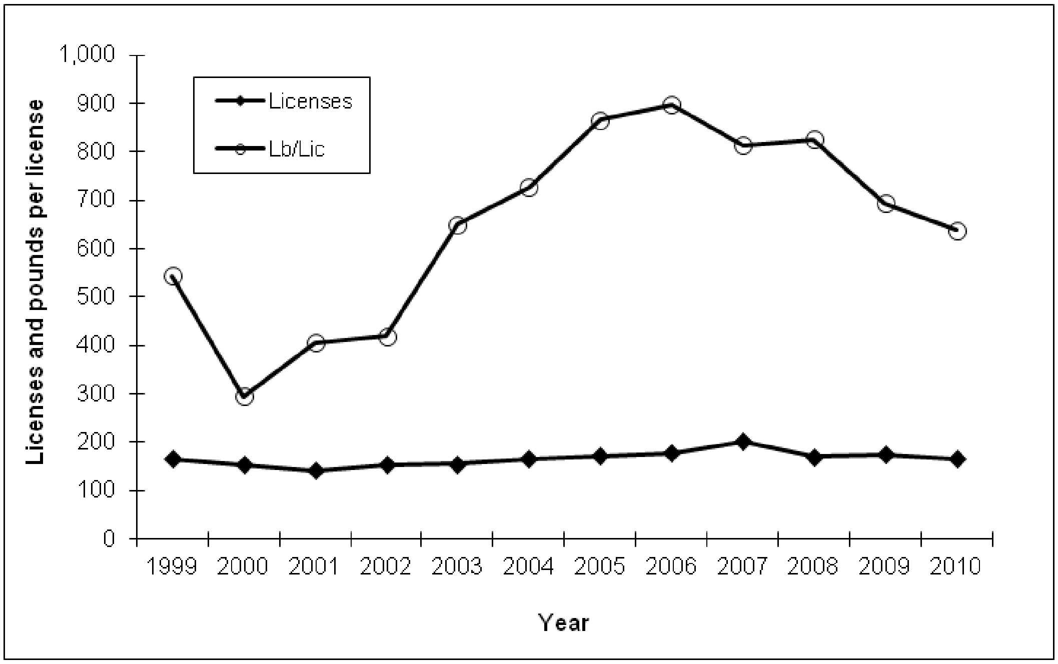

Figure 4.10. Effort and CPUE in New Jersey's commercial eel fishery, 1999–2010.

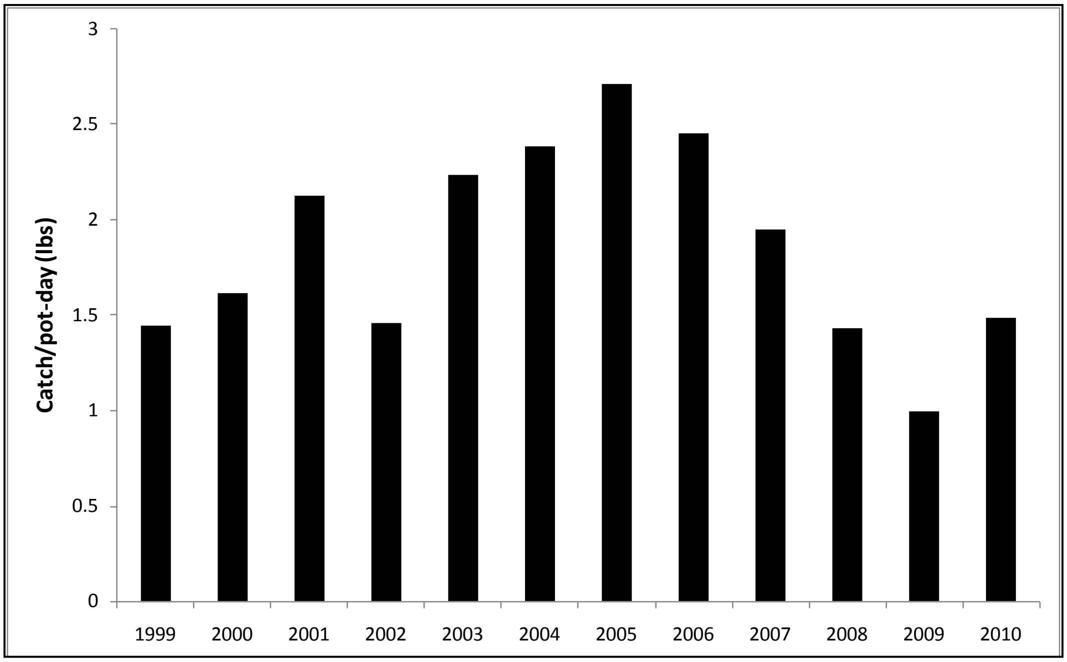

Figure 4.11. Delaware commercial fishery annual mean catch per pot-day fished (lbs), 1999–2010.

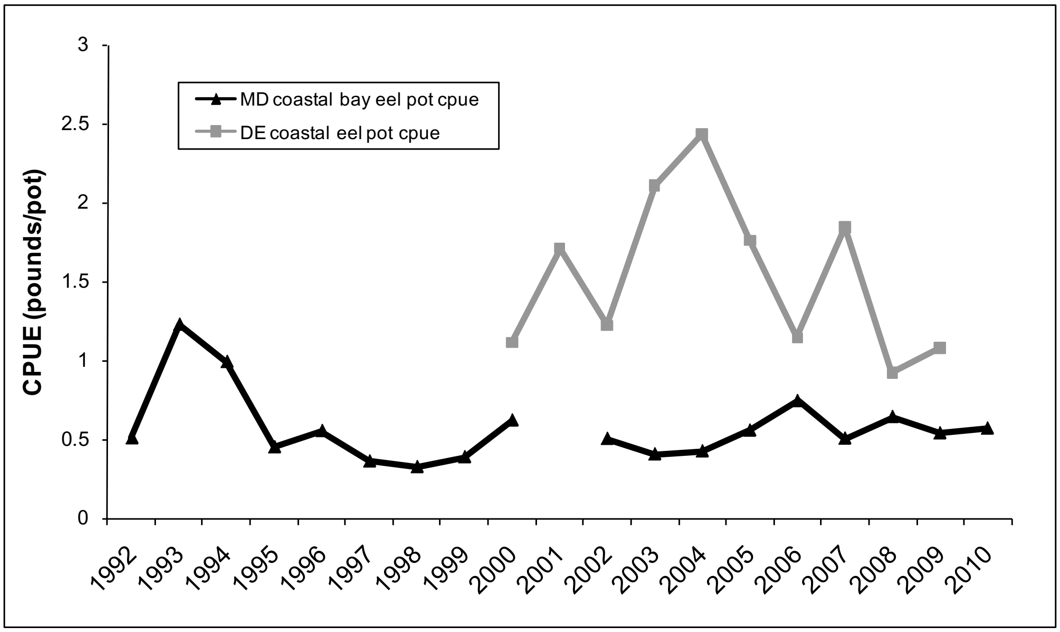

Figure 4.12. Maryland and Delaware commercial fishery eel pot CPUE (pounds/pot) for Coastal Bays, 1992–2010.

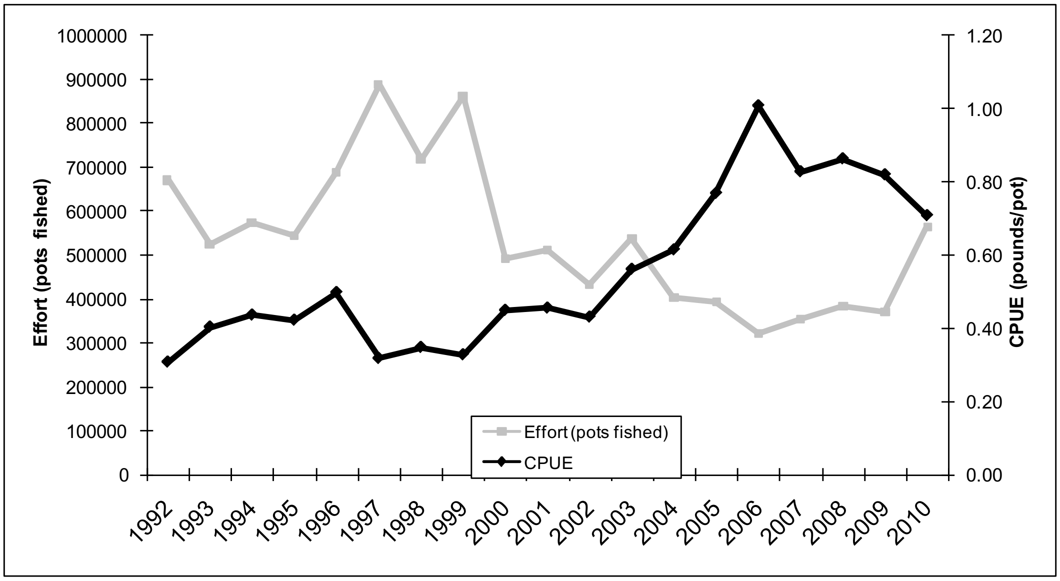

Figure 4.13. Maryland commercial fishery eel pot CPUE (lbs/pot) and effort (total pots fished), 1992–2010.

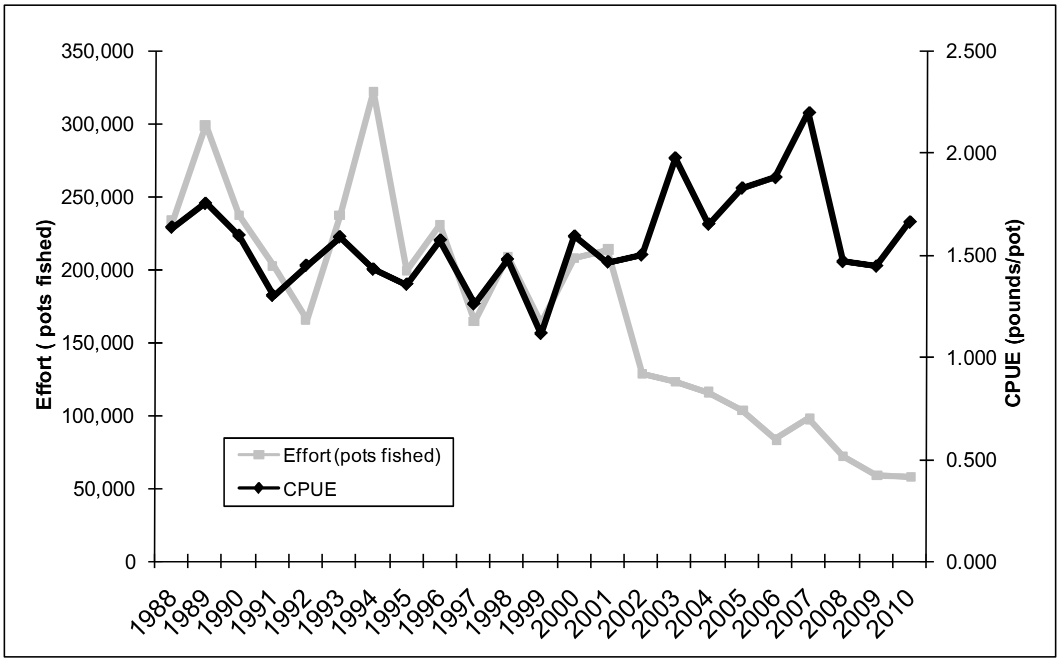

Figure 4.14. PRFC commercial fishery eel pot CPUE (pounds/pot) and effort (total pots fished), 1988–2010.

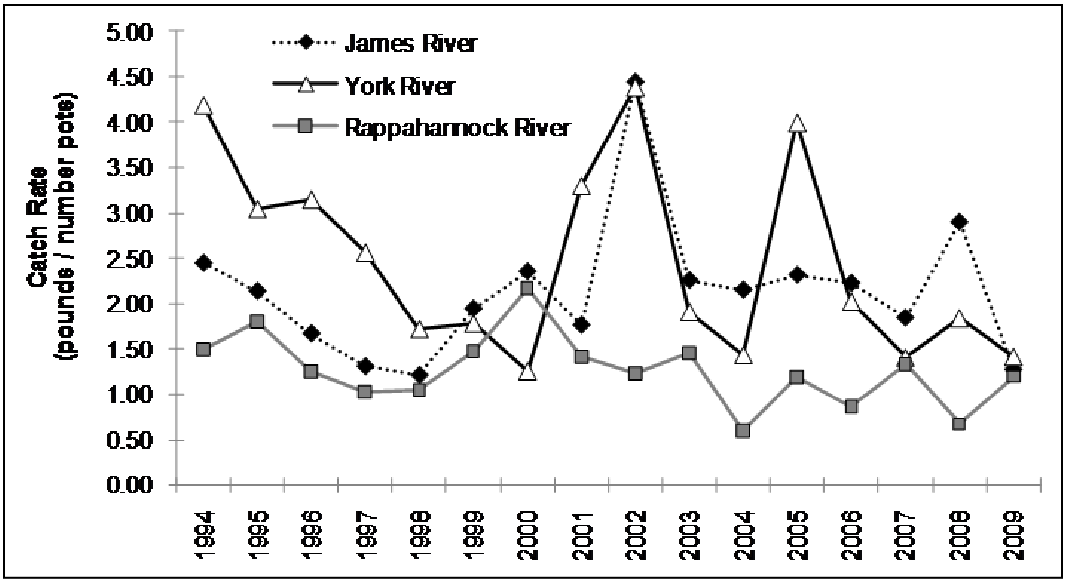

Figure 4.15. Annual commercial fishery catch rates (pounds/number pots) for American eels harvested by eel pots from the primary tributaries of the Chesapeake Bay and landed in Virginia, by tributary, 1994–2009.

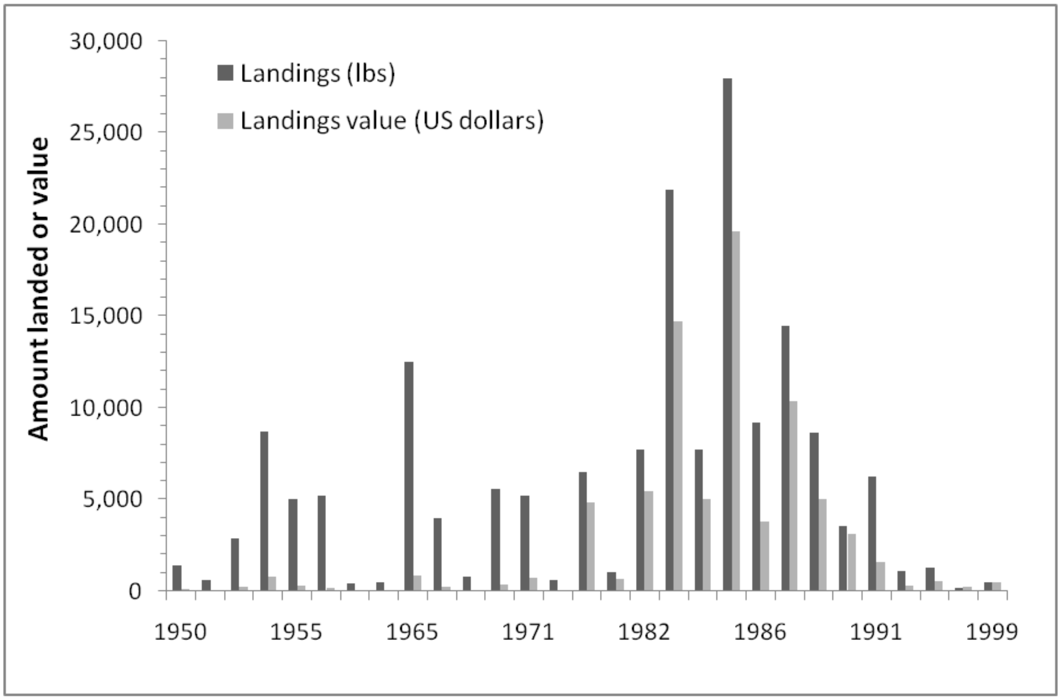

Figure 4.16. Total weight and value of American eel commercial landings in the Gulf of Mexico, 1950–1999. Recent landings are confidential.

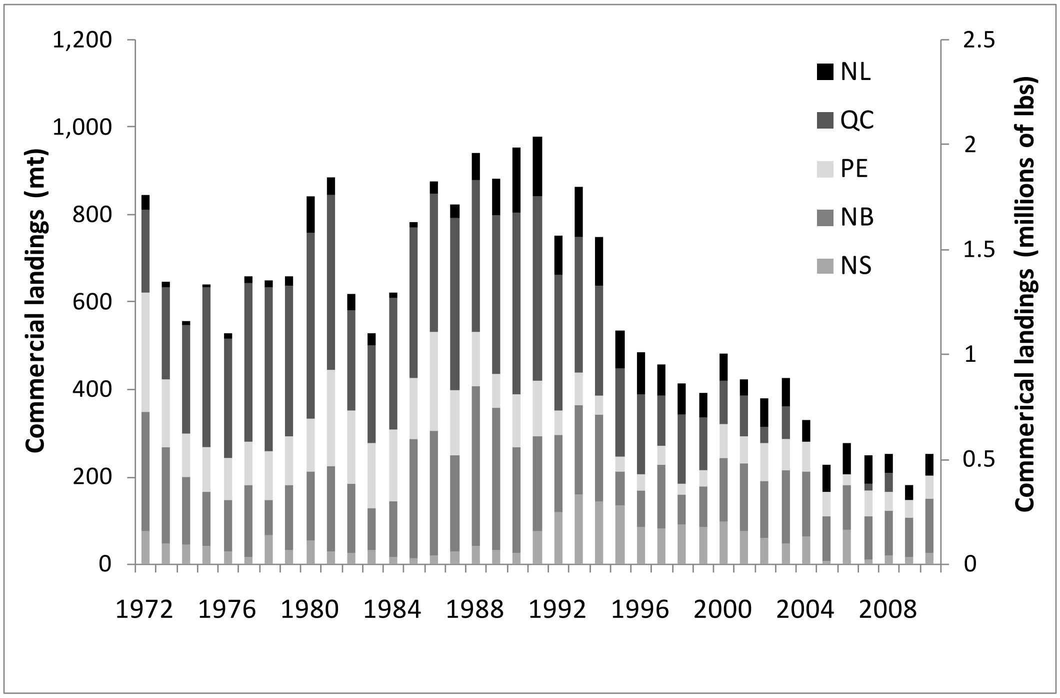

Figure 4.17. Annual commercial seafisheries landings (live weight) of American eel along Canada's Atlantic Coast summarized by province, 1972–2009.

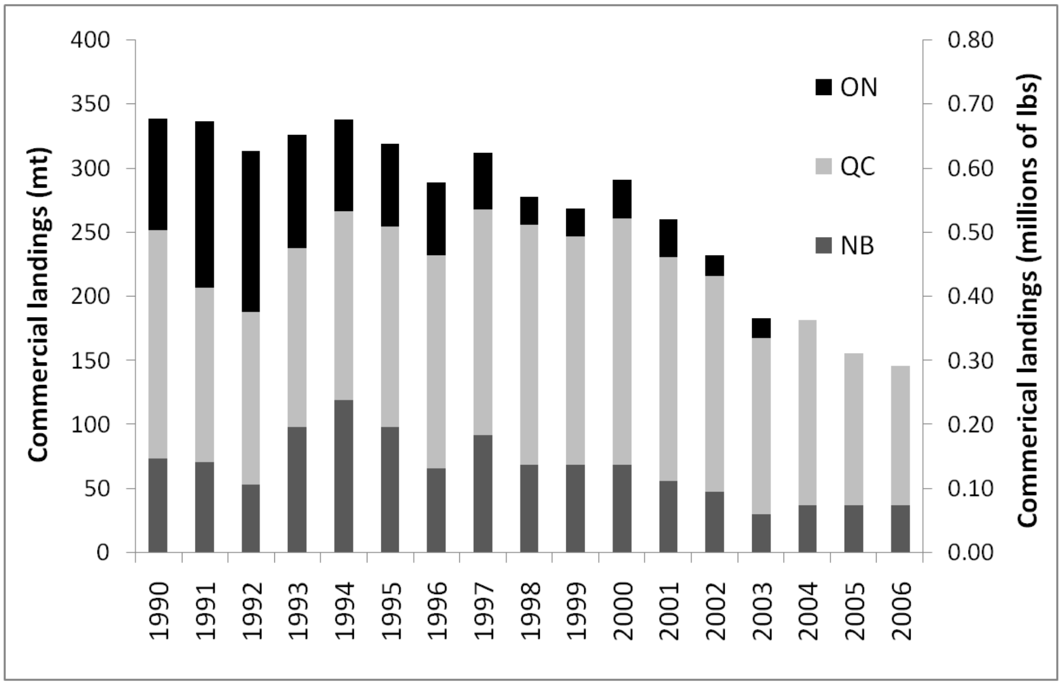

Figure 4.18. Annual commercial freshwater landings (live weight) of American eel along Canada's Atlantic Coast summarized by province, 1990–2006.

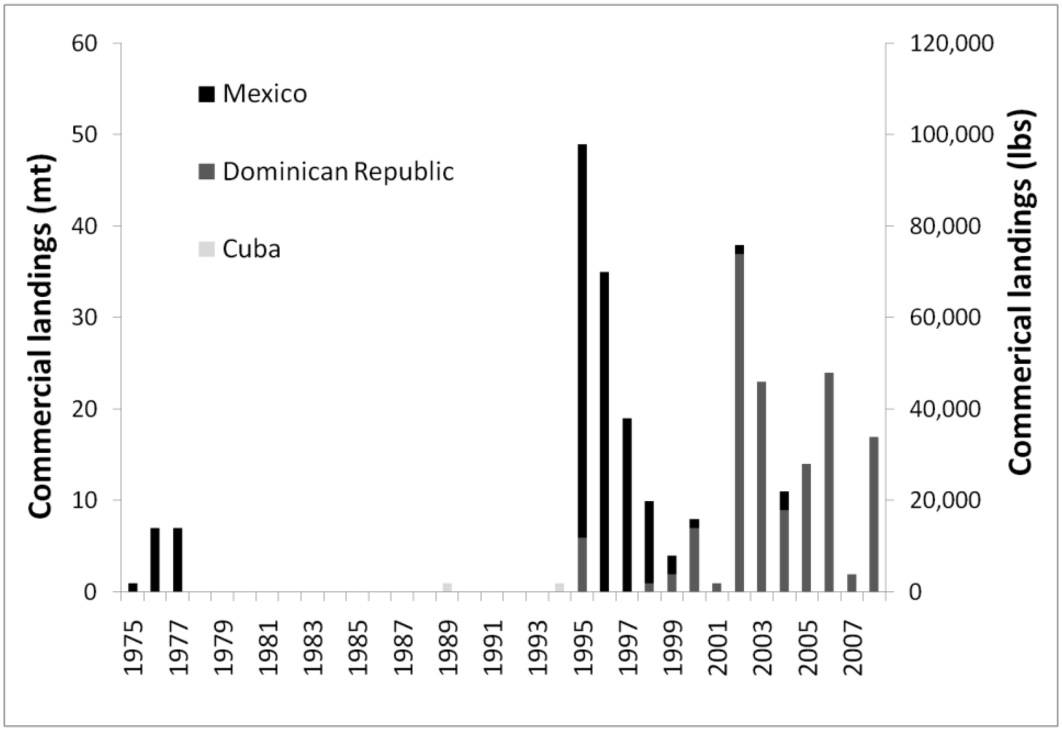

Figure 4.19. Annual commercial landings (live weight) of American eel reported by the FAO from Central and South America, 1975–2008. No landings were reported between 1950 and 1974.

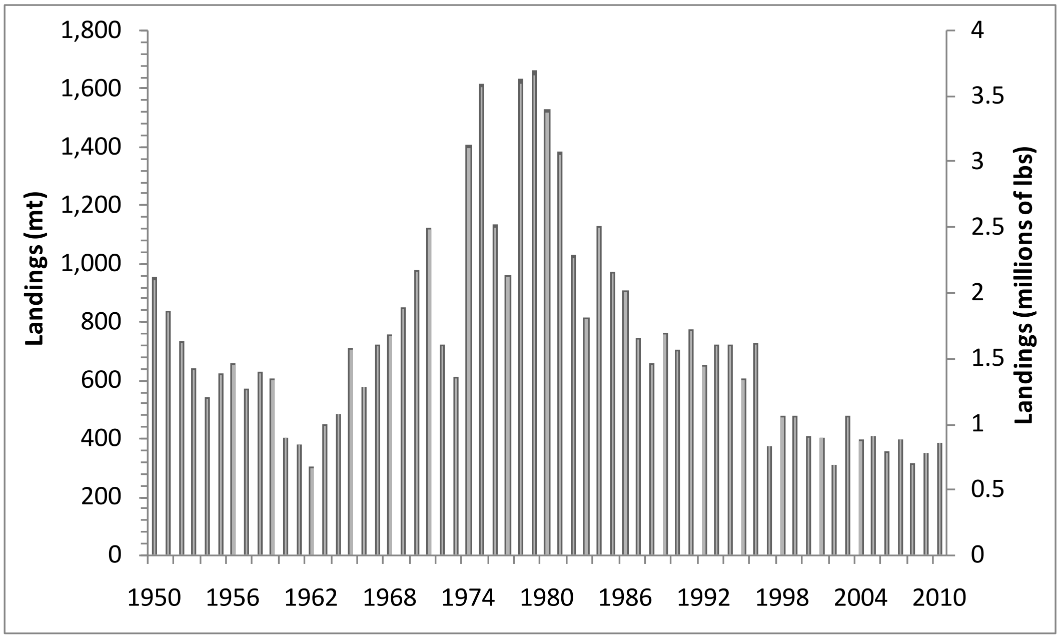

Figure 5.1. Total commercial landings of American eel along the U.S. Atlantic Coast, 1950–2010.

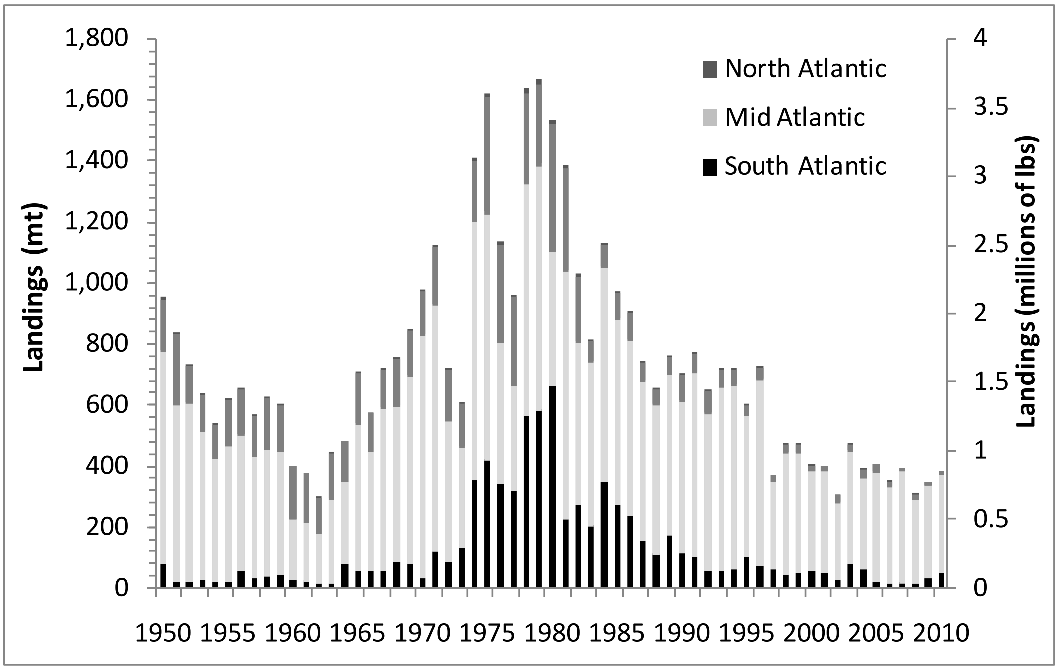

Figure 5.2. Total commercial landings of American eel by old geographic region along the U.S. Atlantic Coast, 1950–2010.

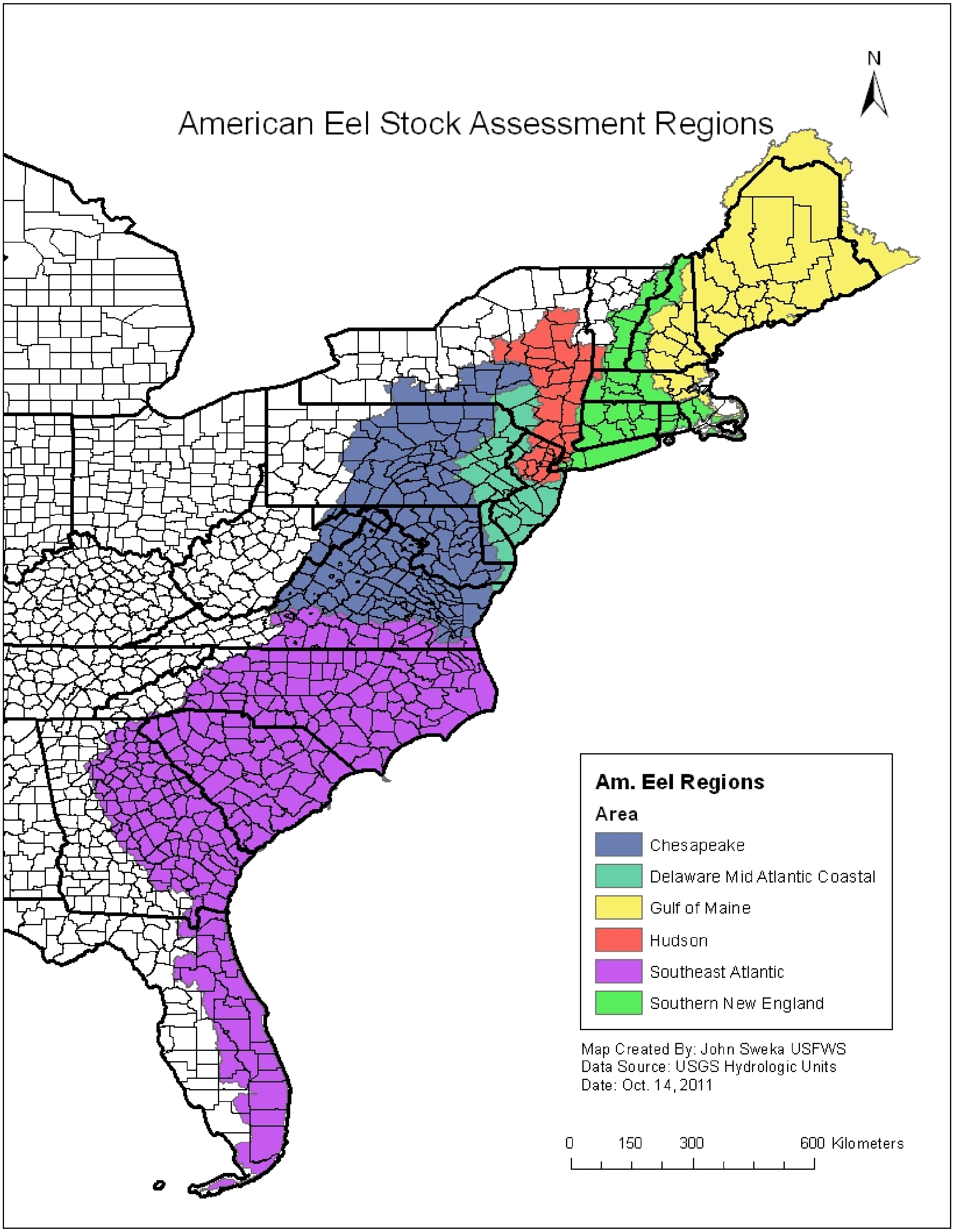

Figure 5.3. Watershed-based geographic regions used in the current assessment.

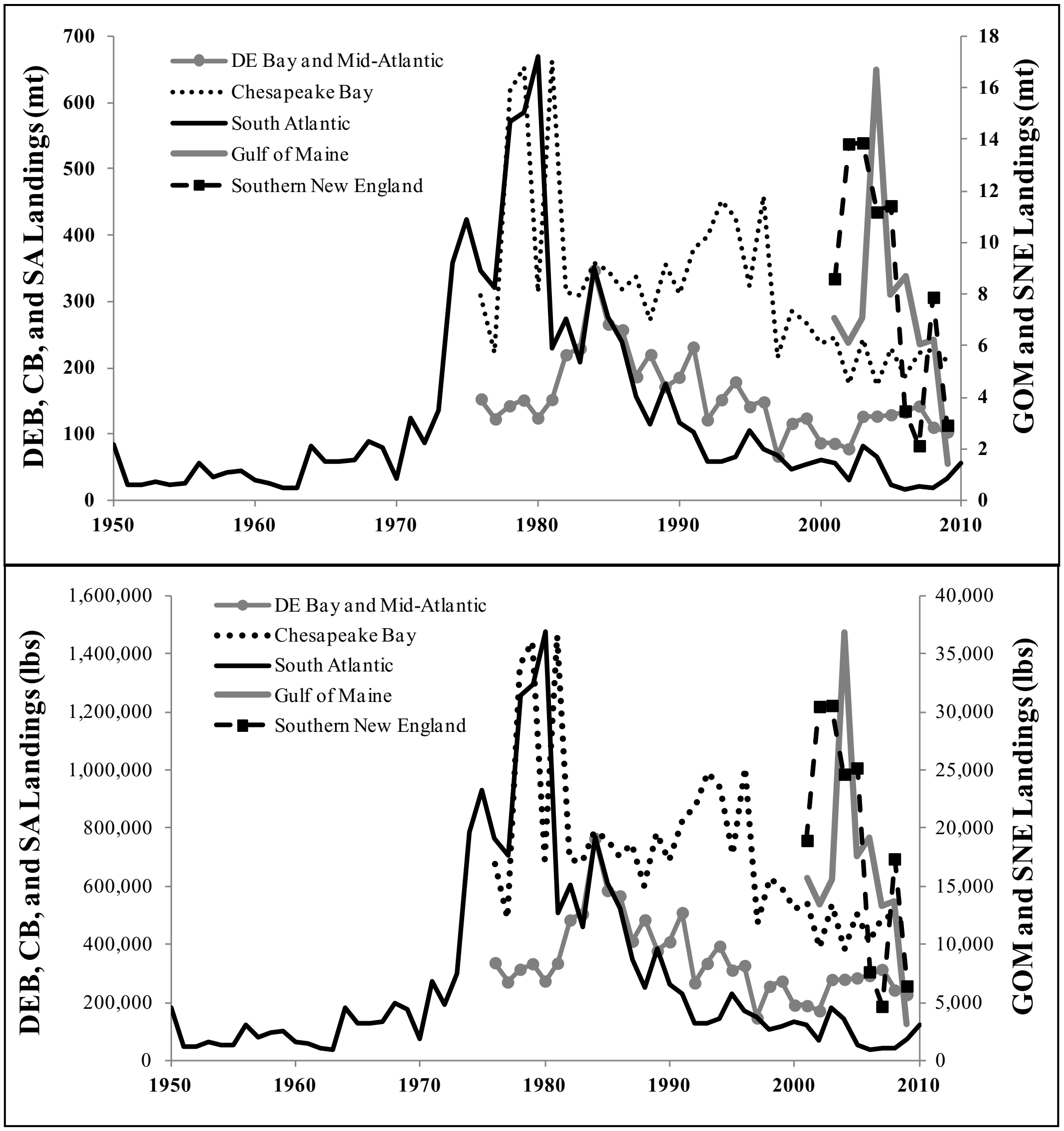

Figure 5.4. Total metric tons (upper graph) and pounds (lower graph) of American eel commercial landings by new geographic region along the U.S. Atlantic Coast, 1950–2010. Note Gulf of Maine and Southern New England are plotted on the secondary axis.

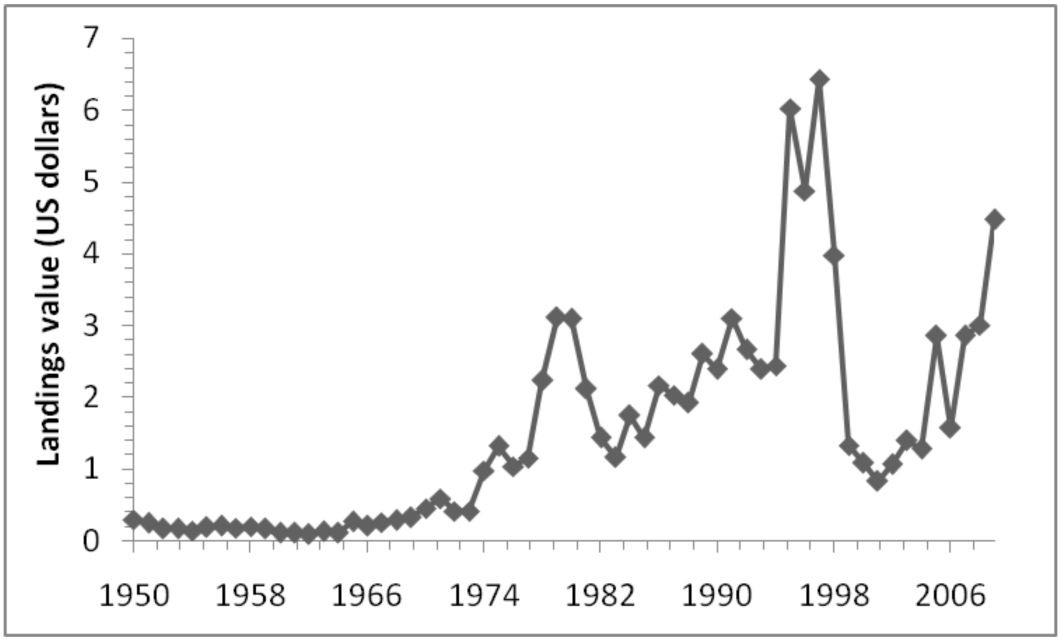

Figure 5.5. Estimated value of U.S. American eel landings, 1950–2009.

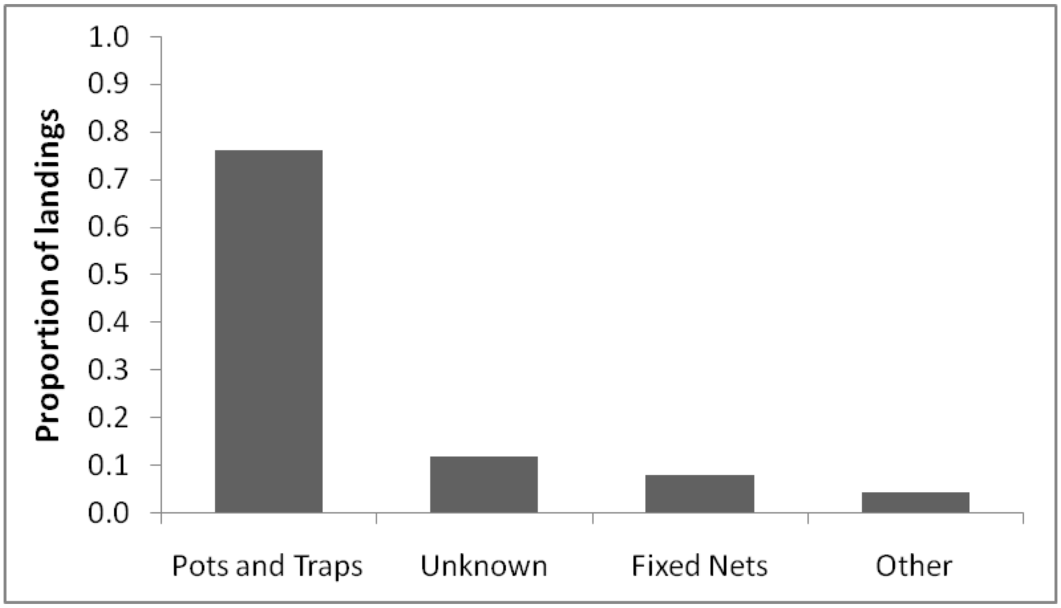

Figure 5.6. Proportion of Atlantic coast commercial landings by general gear type, 1950–2010.

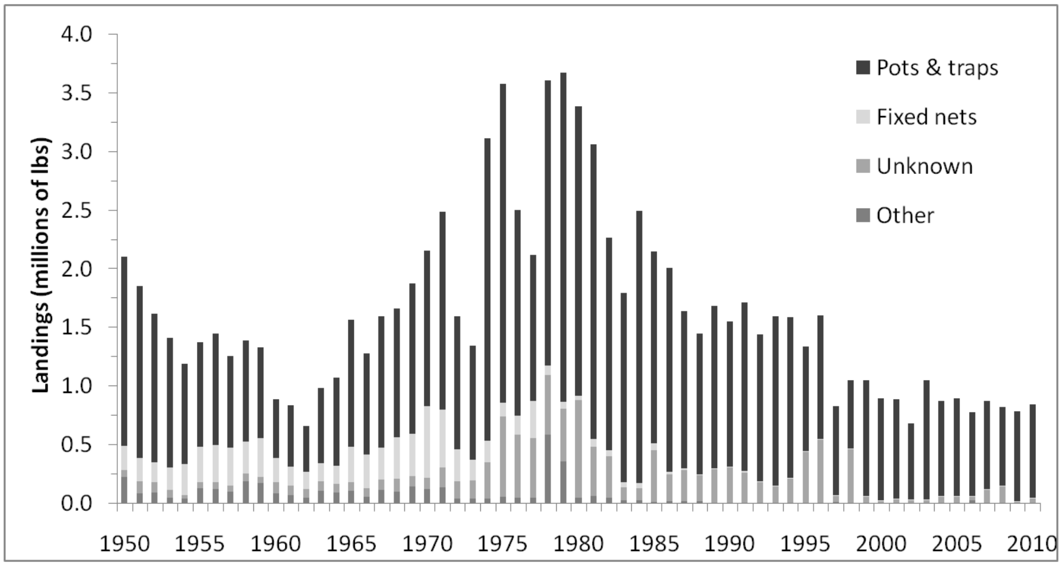

Figure 5.7. Trends in the proportion of Atlantic coast commercial landings by general gear type.

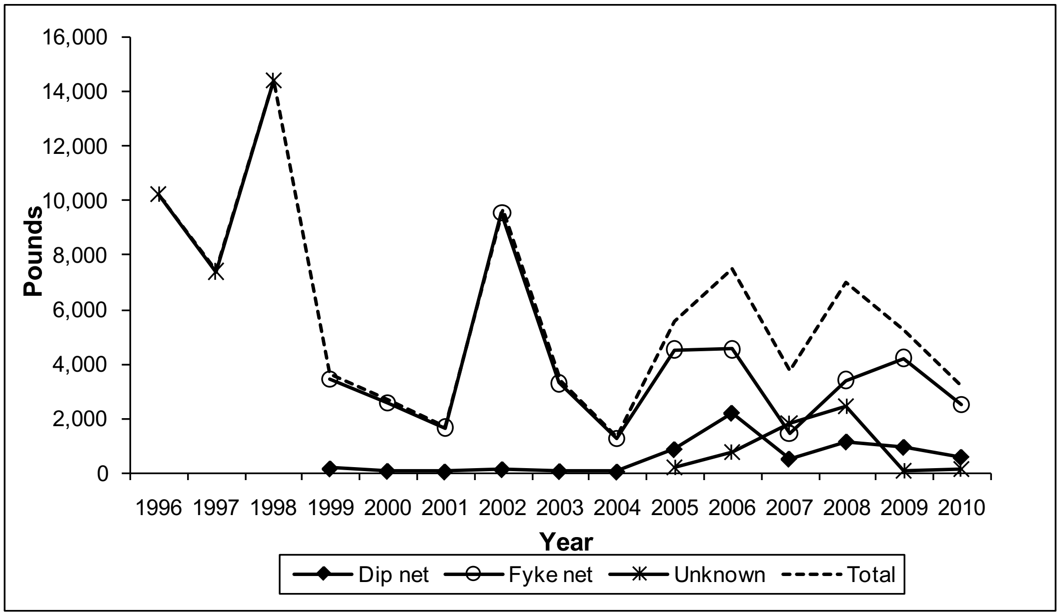

Figure 5.8. Dealer reported commercial glass eel landings in Maine.

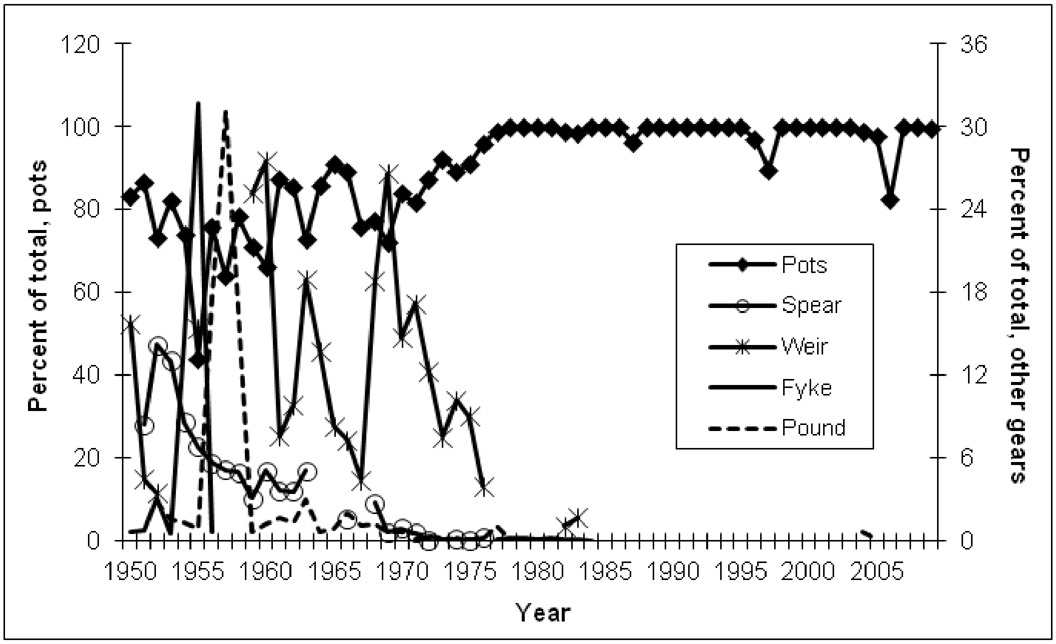

Figure 5.9. Percentage of New Jersey commercial eel landings by gear.

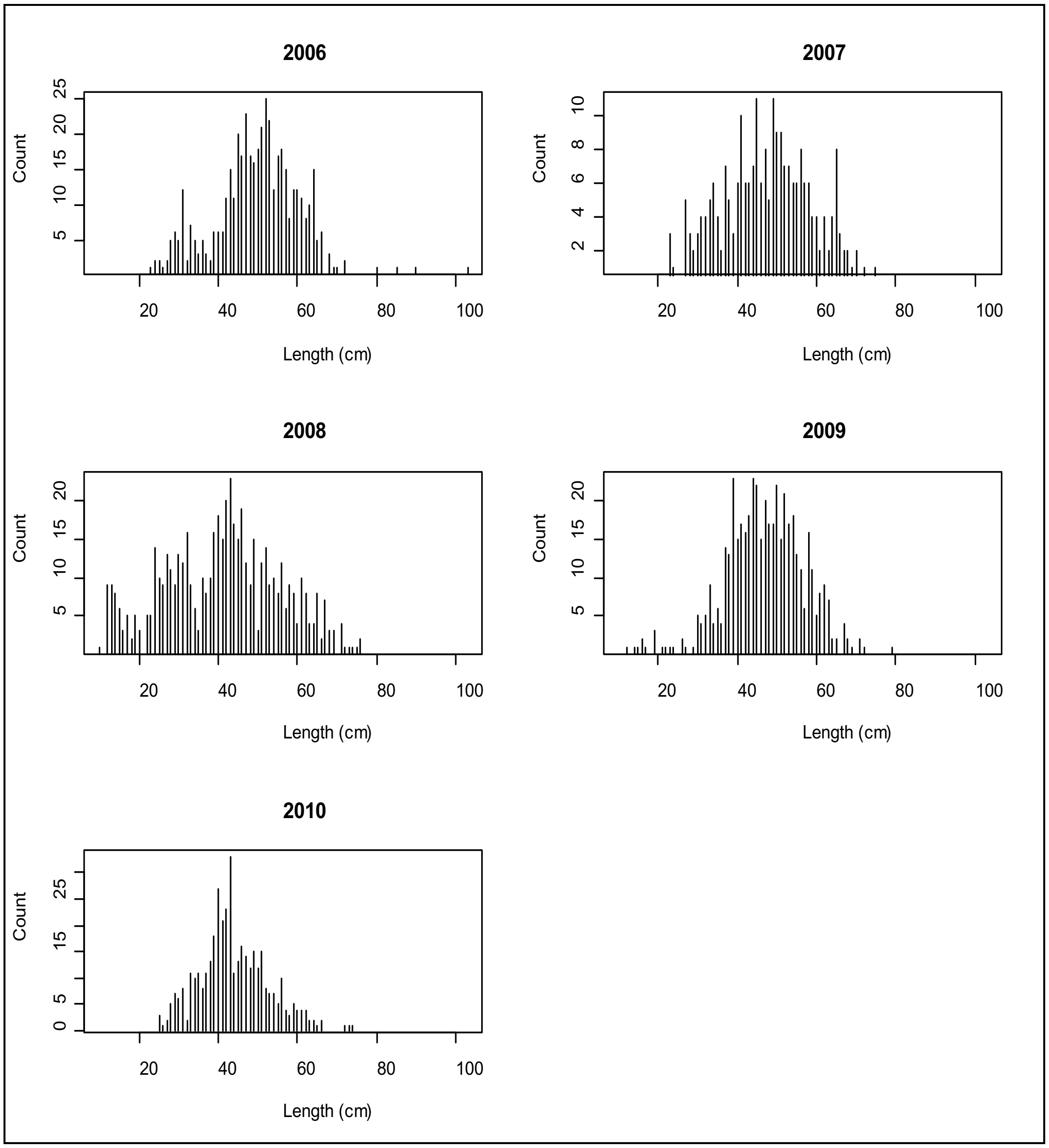

Figure 5.10. Average length (centimeters) of eels sampled from New Jersey's commercial harvest.

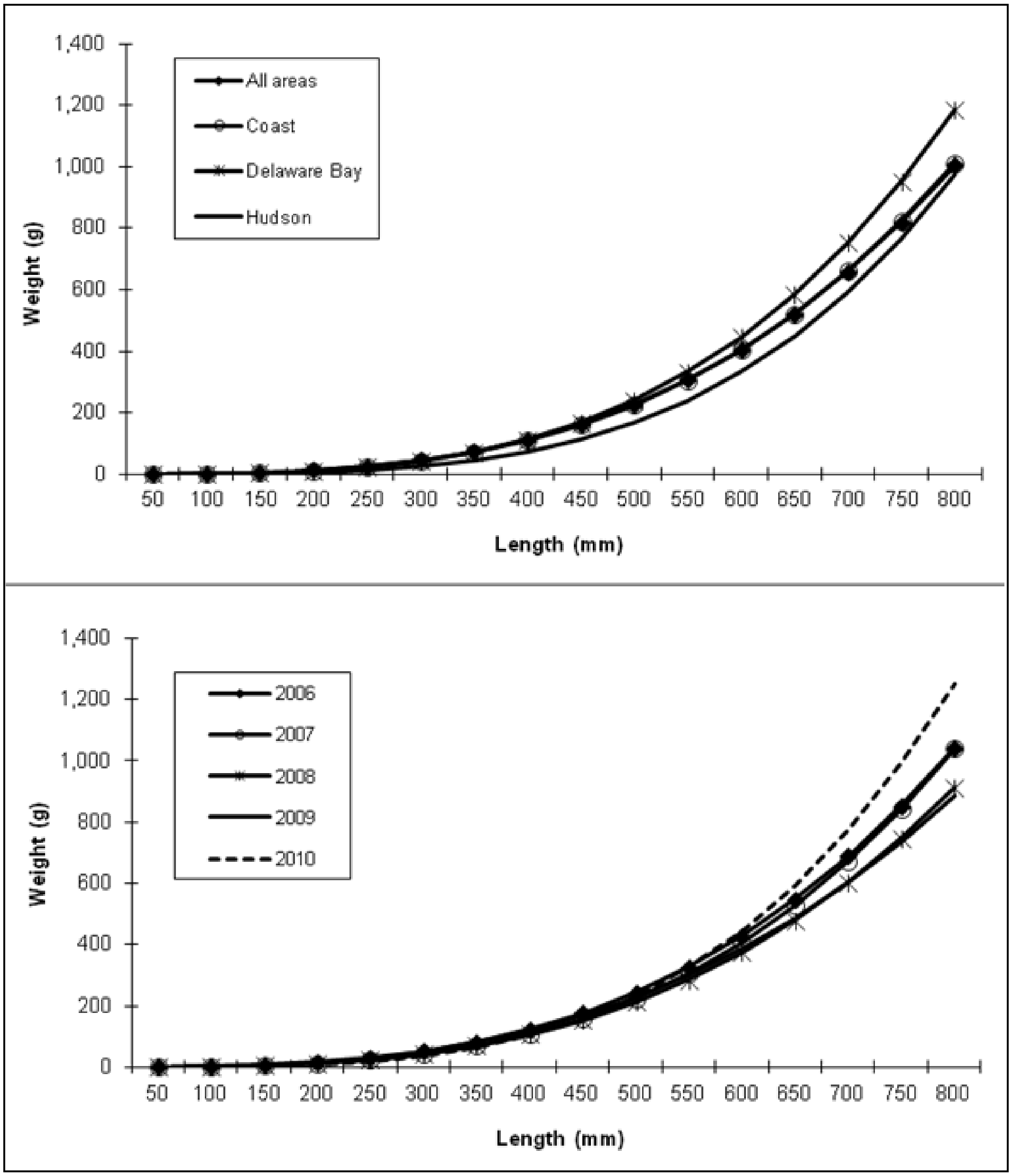

Figure 5.11. Predicted weight at length of American eels sampled from New Jersey's commercial harvest by area for all years combined (upper graph) and by year for all areas combined (lower graph).

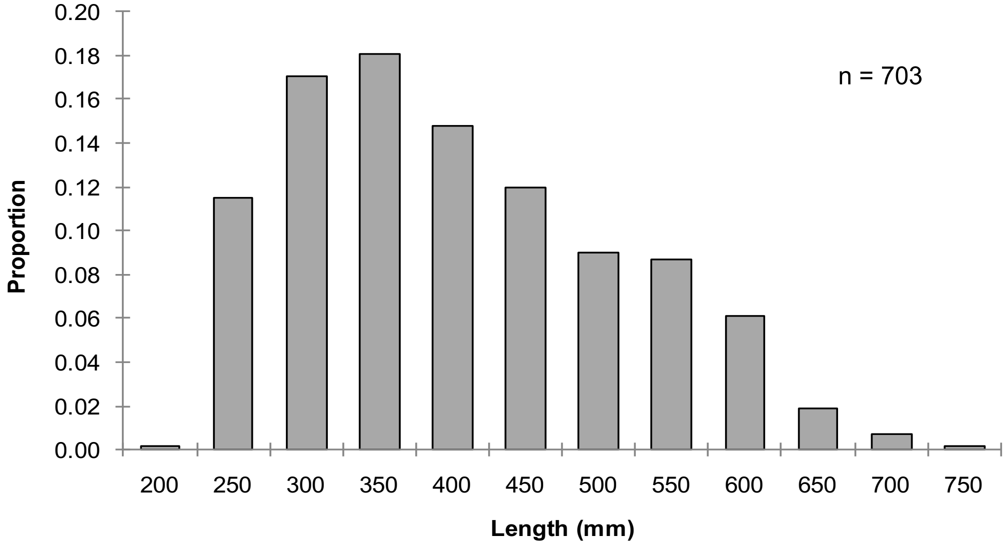

Figure 5.12. Length-frequency distribution of American eels sampled from Virginia's eel pot landings, 1989–2008. No American eels were available for sampling in 2009 or 2010.

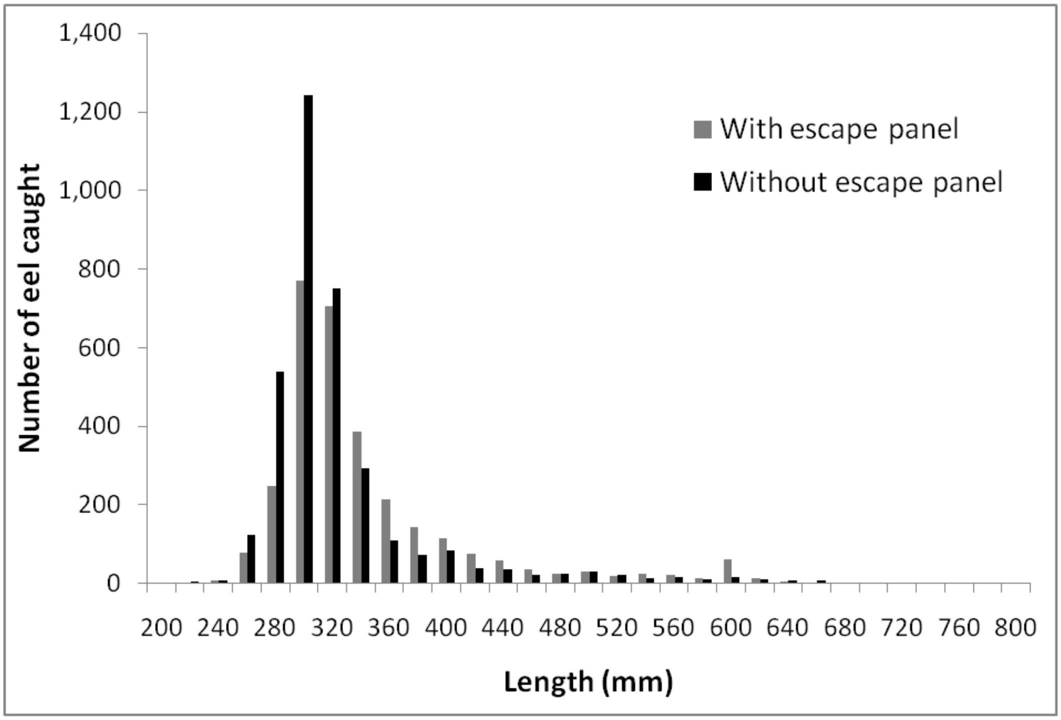

Figure 5.13. Length distribution of American eels sampled from commercial eel pots with and without escape panel, Pamlico River, 1996.

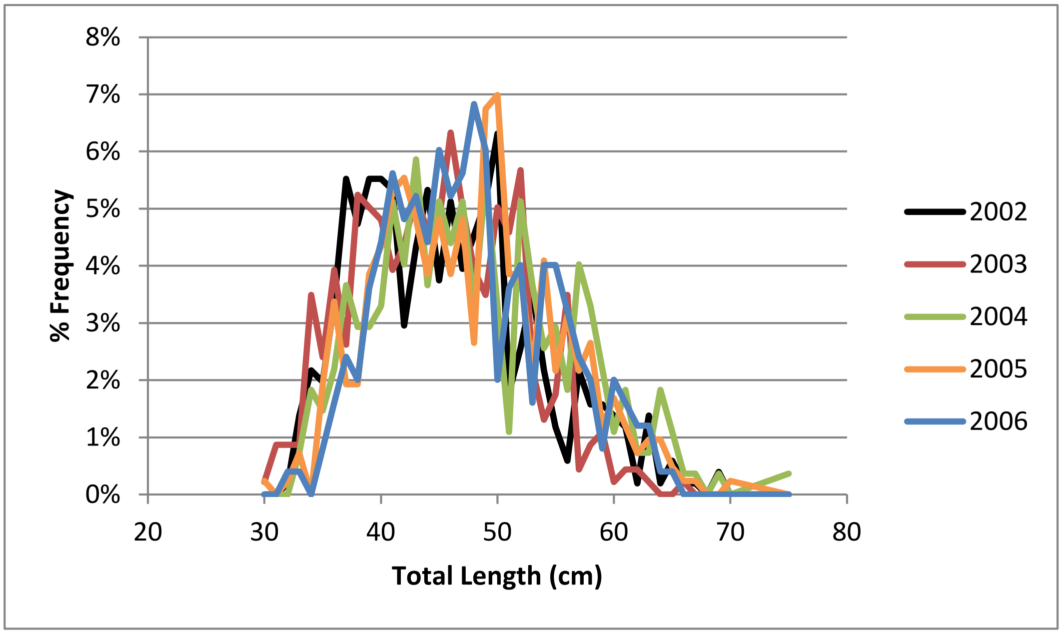

Figure 5.14. Length frequency distribution of American eels from the St. Johns River system, Florida. Biological sampling was discontinued after 2006.

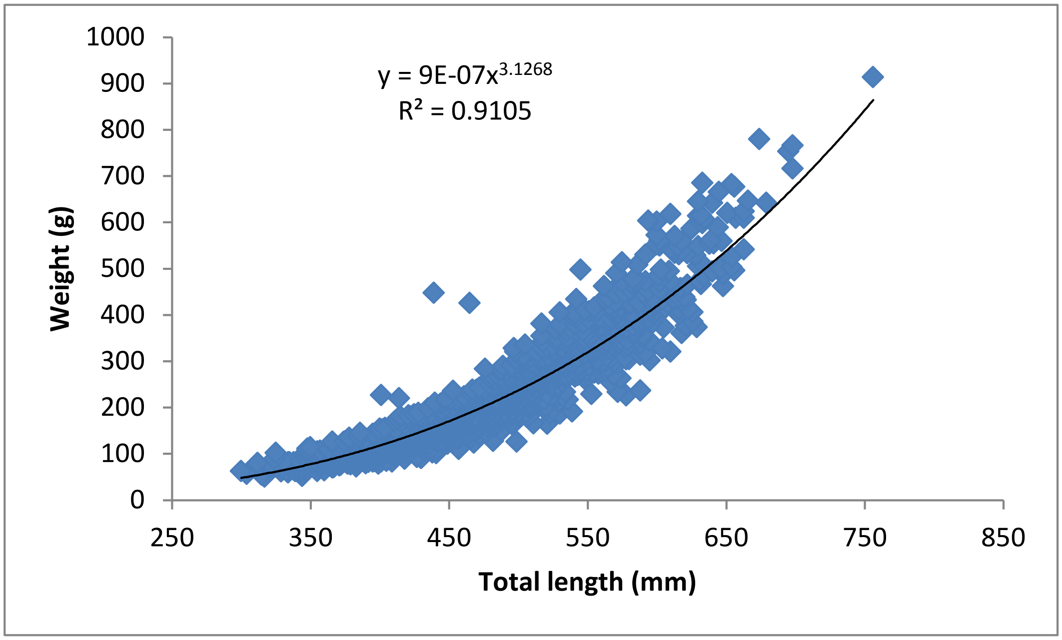

Figure 5.15. Weight-length relationship for American eels in the St. Johns River system, Florida, 2002–2006.

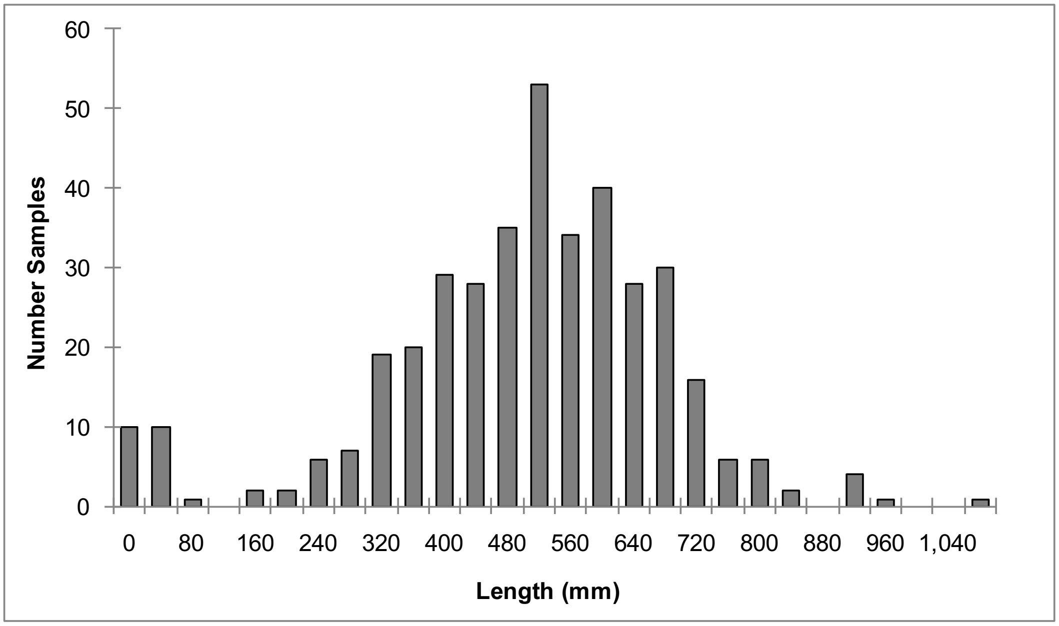

Figure 5.16. Length-frequency of American eels sampled by the MRFSS angler-intercept survey (Type A catch), 1981–2010.

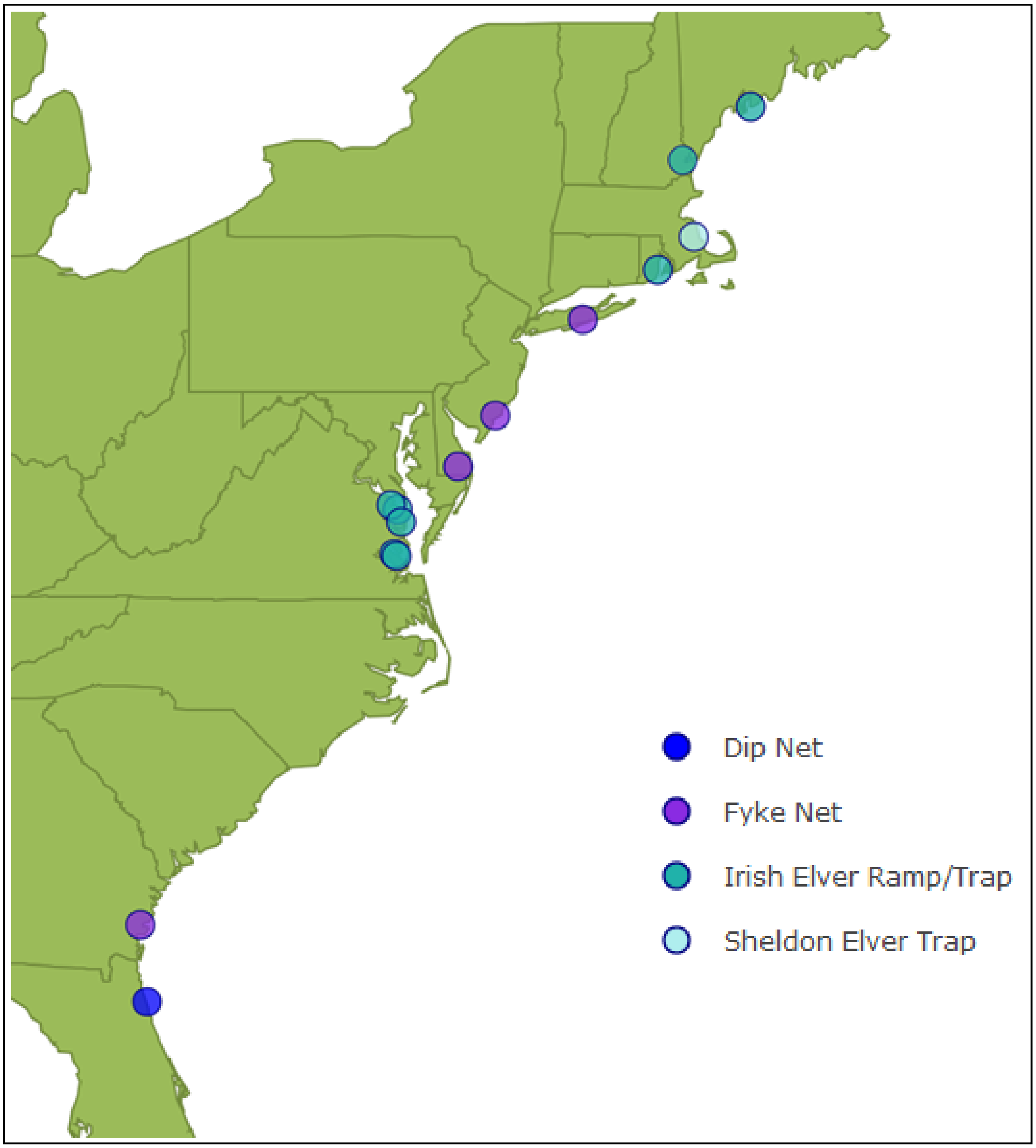

Figure 5.17. Locations of ASMFC-mandated annual American eel YOY abundance survey sites that have been sampled for at least 10 years, as of 2010.

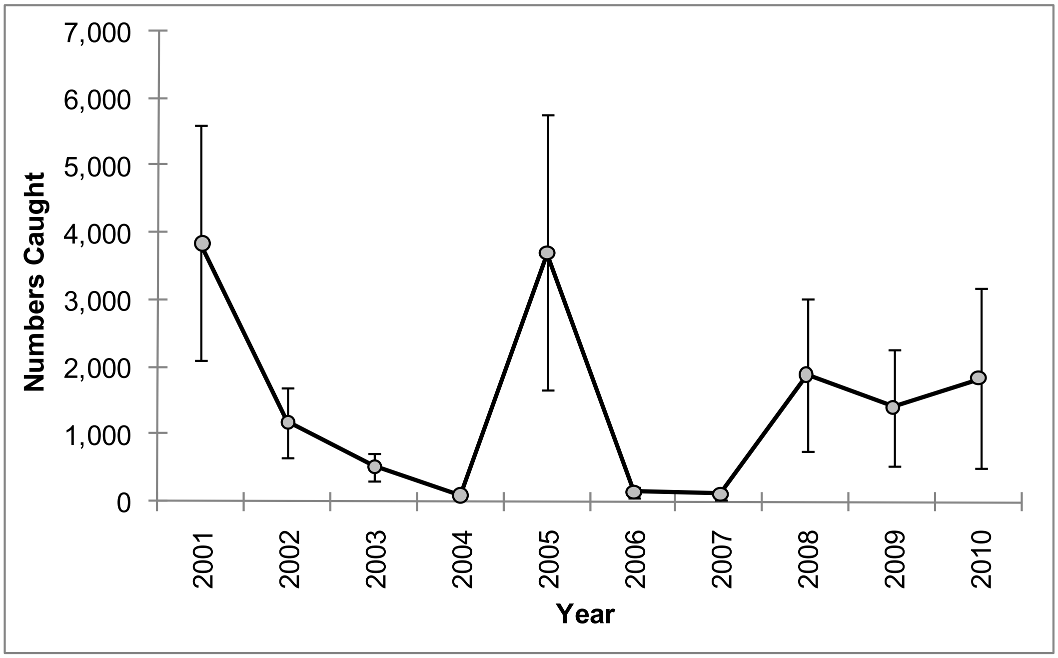

Figure 5.18. GLM-standardized index of abundance for YOY American eels caught by Maine's annual YOY survey in West Harbor Pond, 2001–2010. The error bars represent the standard errors about the estimates.

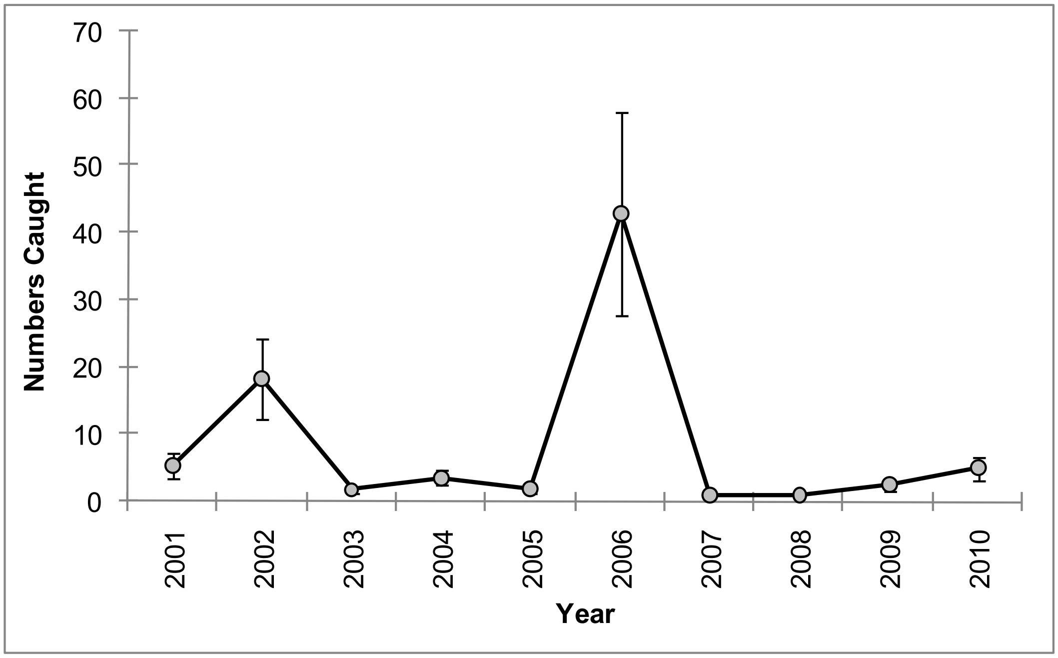

Figure 5.19. GLM-standardized index of abundance for YOY American eels caught by New Hampshire's annual YOY survey in the Lamprey River, 2001–2010. The error bars represent the standard errors about the estimates.

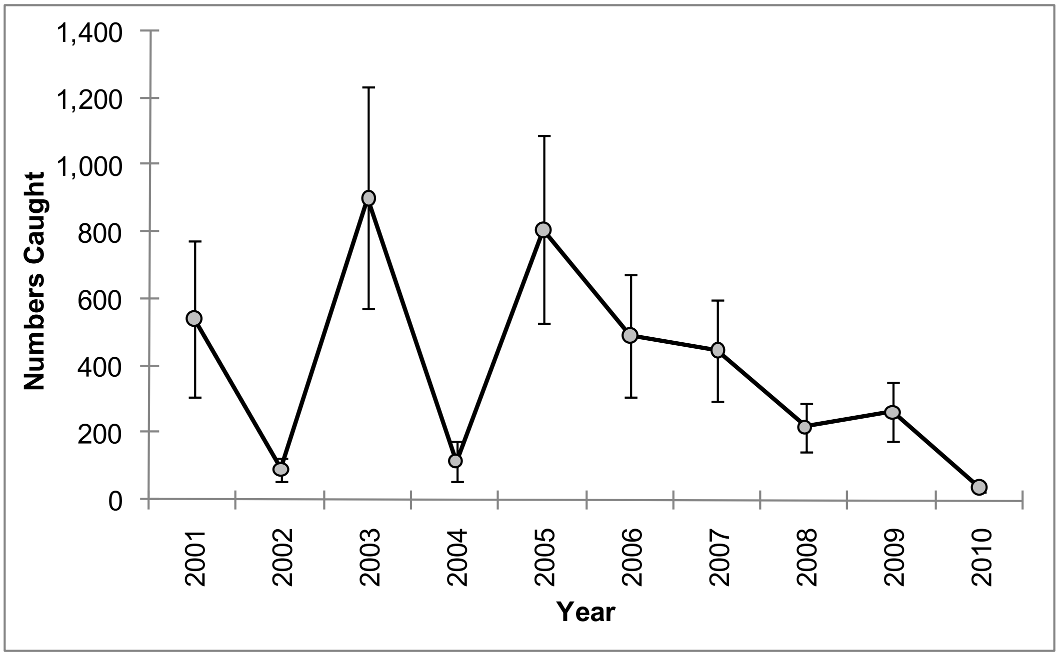

Figure 5.20. GLM-standardized index of abundance for YOY American eels caught by Massachusetts' annual YOY survey in the Jones River, 2001–2010. The error bars represent the standard errors about the estimates.

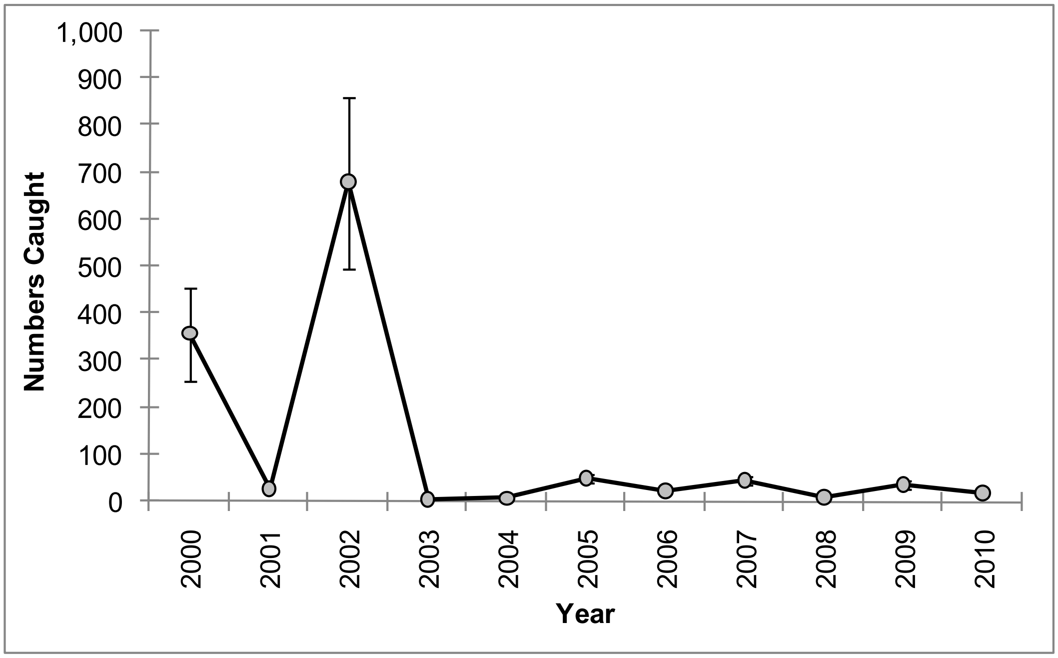

Figure 5.21. GLM-standardized index of abundance for American eels caught by Rhode Island's annual YOY survey near Gilbert Stuart Dam, 2000–2010. The error bars represent the standard errors about the estimates.

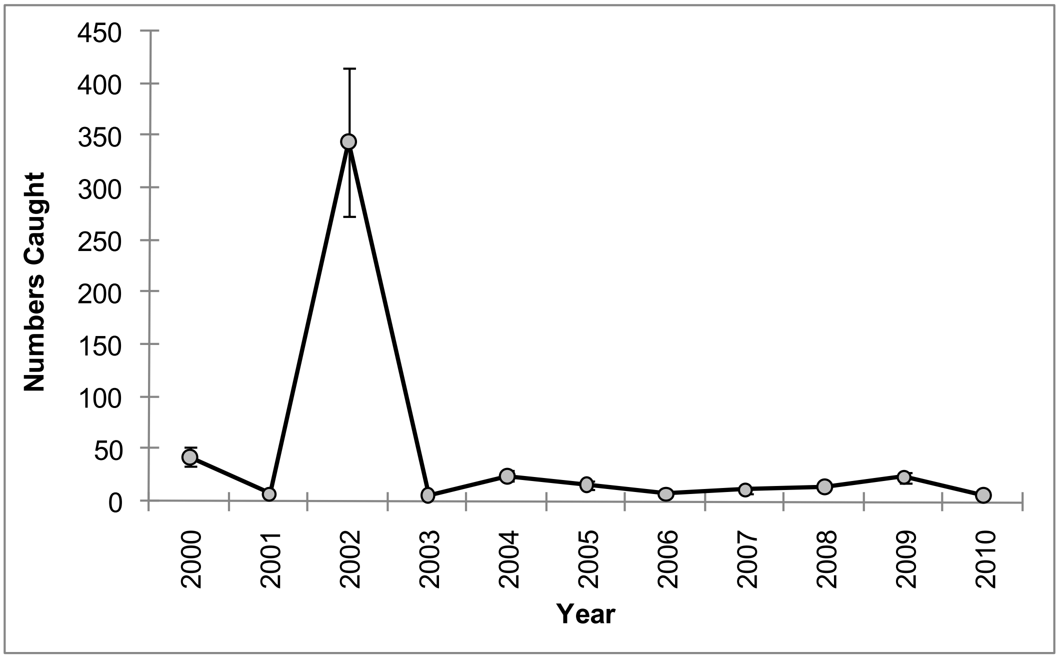

Figure 5.22. GLM-standardized index of abundance for American eels caught by New York's annual YOY survey in Carman's River, 2001–2010. The error bars represent the standard errors about the estimates.

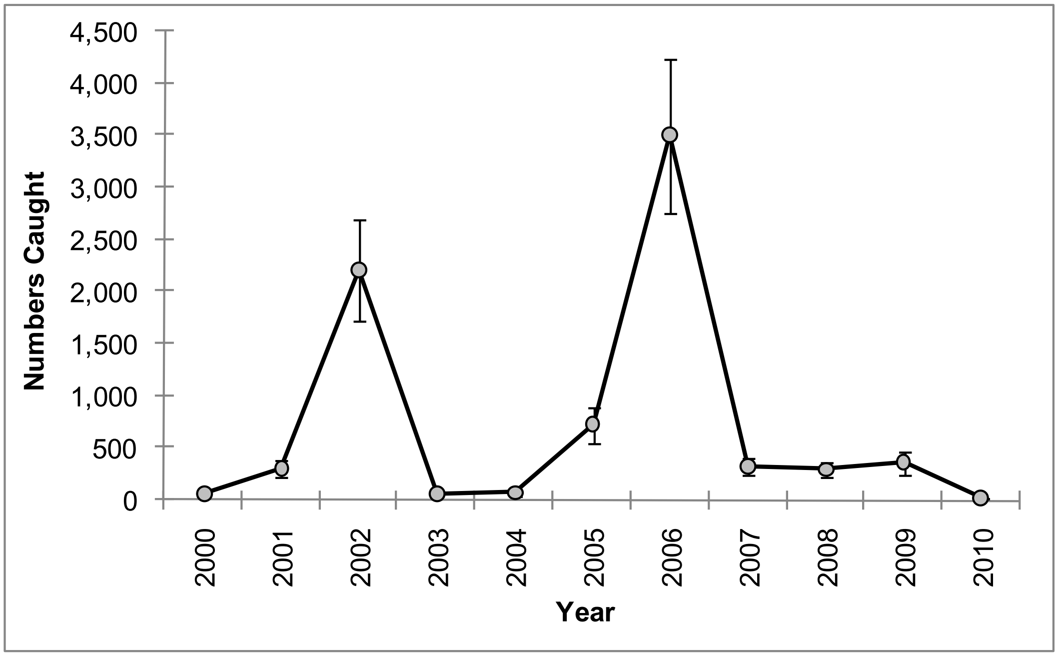

Figure 5.23. GLM-standardized index of abundance for YOY American eels caught by New Jersey's annual YOY survey in Patcong Creek, 2000–2010. The error bars represent the standard errors about the estimates.

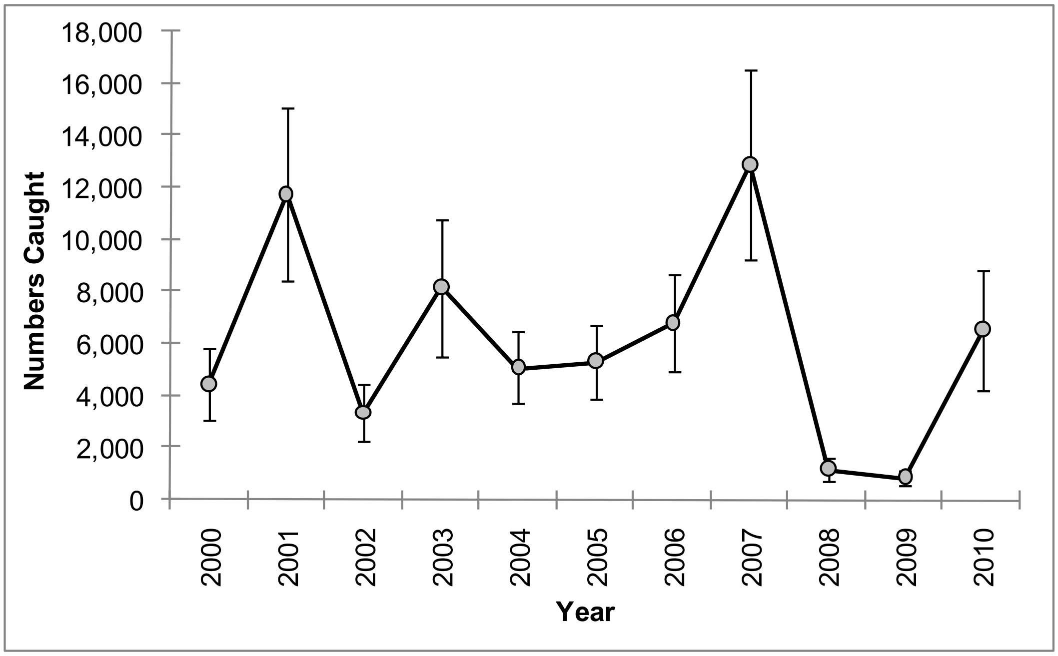

Figure 5.24. GLM-standardized index of abundance for American eels caught by Delaware's annual YOY survey near the Millsboro Dam, 2000–2010. The error bars represent the standard errors about the estimates.

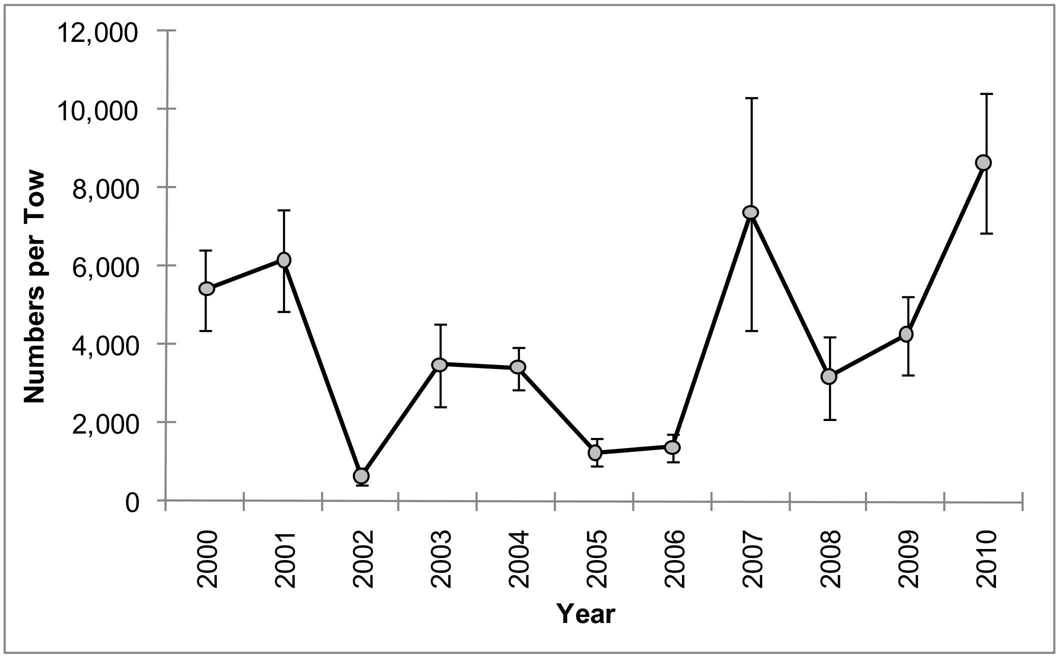

Figure 5.25. Annual index of abundance for American eels caught by Maryland's annual YOY survey in Turville Creek, 2000–2010. The error bars represent the standard errors about the estimates.

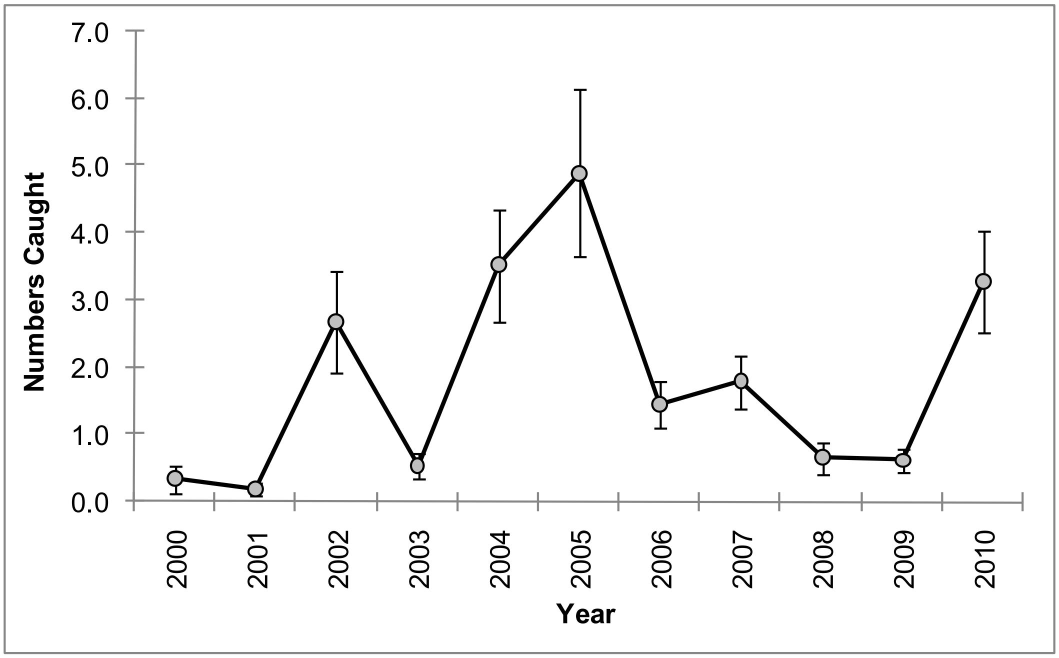

Figure 5.26. GLM-standardized index of abundance for American eels caught by PRFC's annual YOY survey in Clark's Millpond, 2000–2010. The error bars represent the standard errors about the estimates.

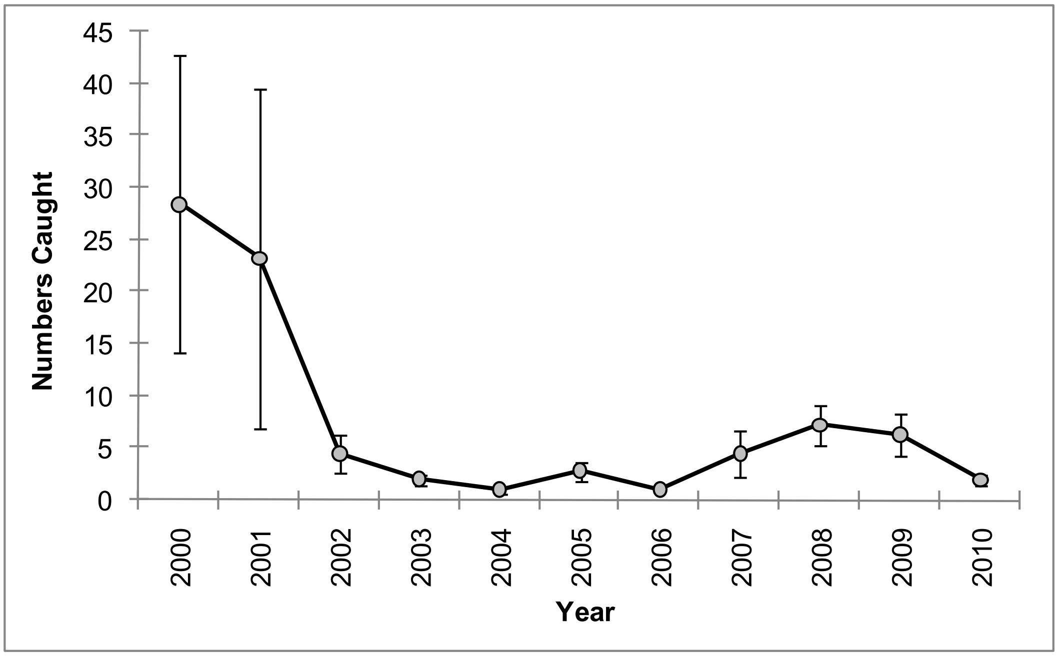

Figure 5.27. GLM-standardized index of abundance for American eels caught by PRFC's annual YOY survey in Gardy's Millpond, 2000–2010. The error bars represent the standard errors about the estimates.

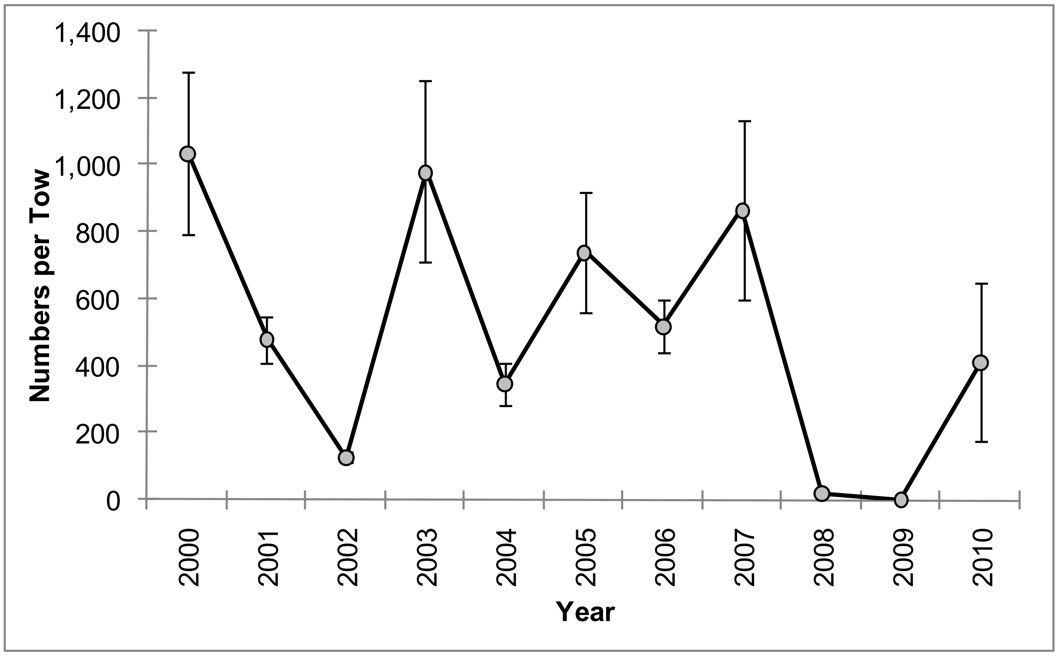

Figure 5.28. Annual index of abundance for American eels caught by Virginia's annual YOY survey in Bracken's Pond, 2000–2010. The error bars represent the standard errors about the estimates.

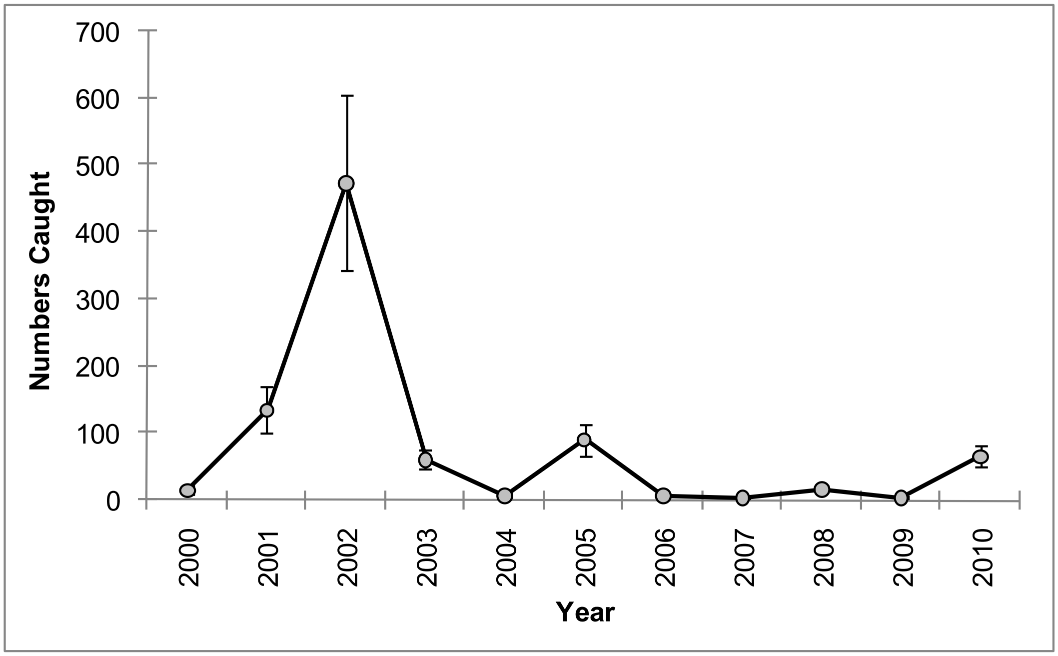

Figure 5.29. GLM-standardized index of abundance for American eels caught by Virginia's annual YOY survey in Kamp's Millpond, 2000–2010. The error bars represent the standard errors about the estimates.

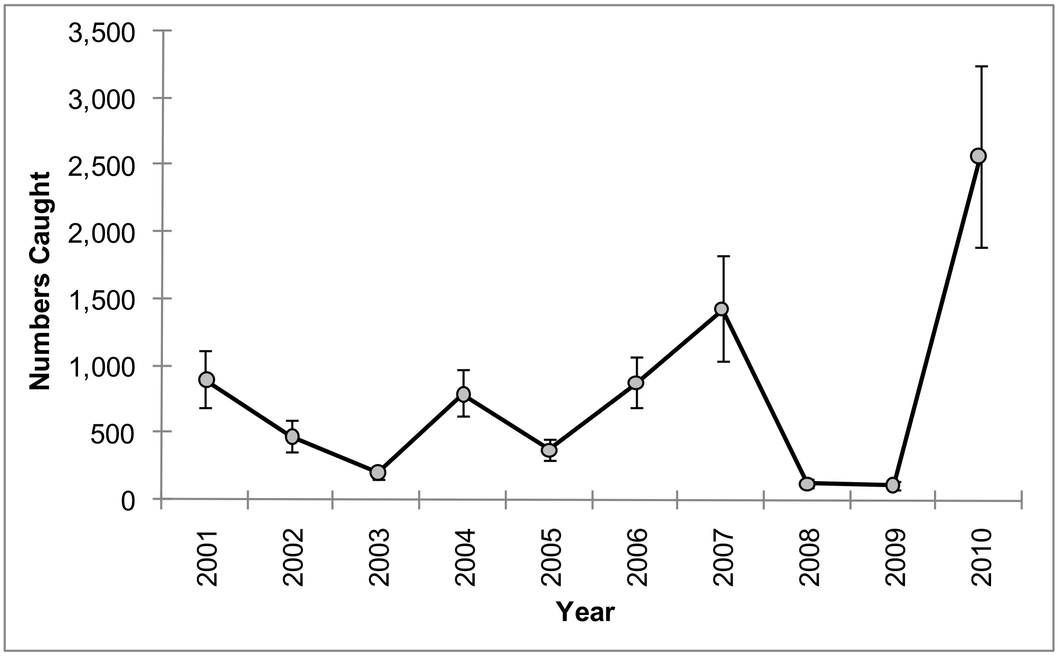

Figure 5.30. GLM-standardized index of abundance for American eels caught by Virginia's annual YOY survey in Wormley Creek, 2001–2010. The error bars represent the standard errors about the estimates.

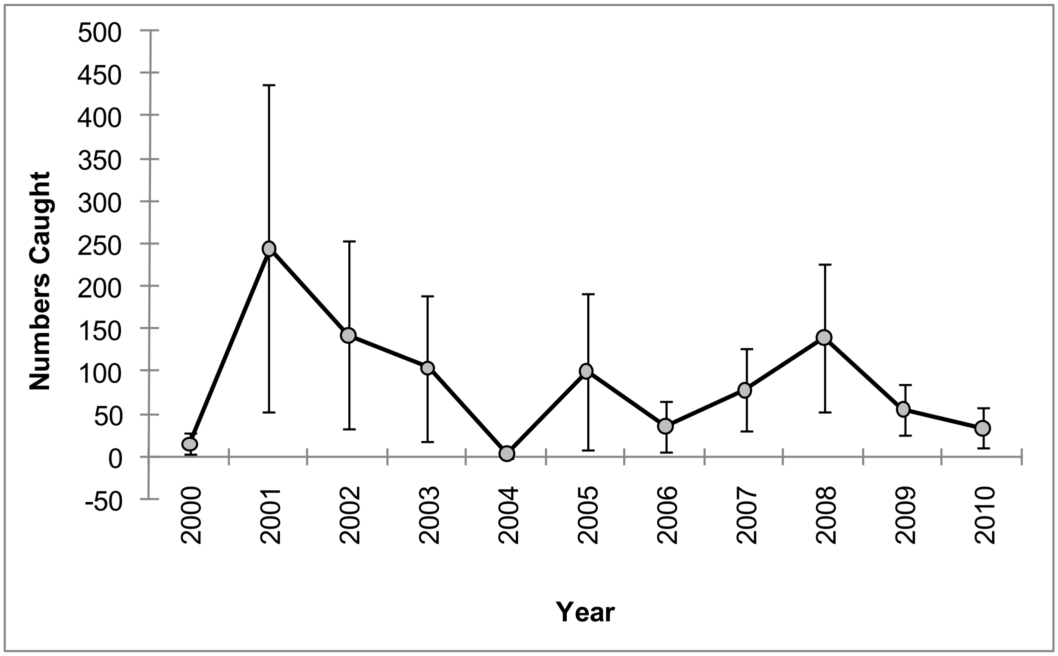

Figure 5.31. GLM-standardized index of abundance for American eels caught by South Carolina's annual YOY survey in Goose Creek, 2000–2010. The error bars represent the standard errors about the estimates.

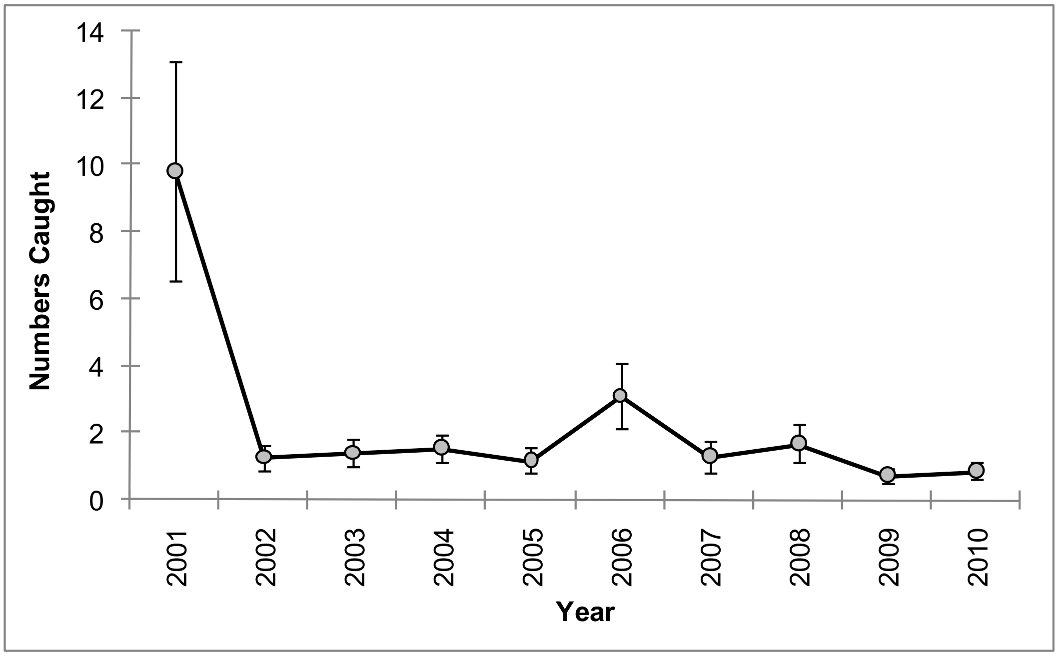

Figure 5.32. GLM-standardized index of abundance for American eels caught by Georgia's annual YOY survey near the Altamaha Canal, 2001–2010. The error bars represent the standard errors about the estimates.

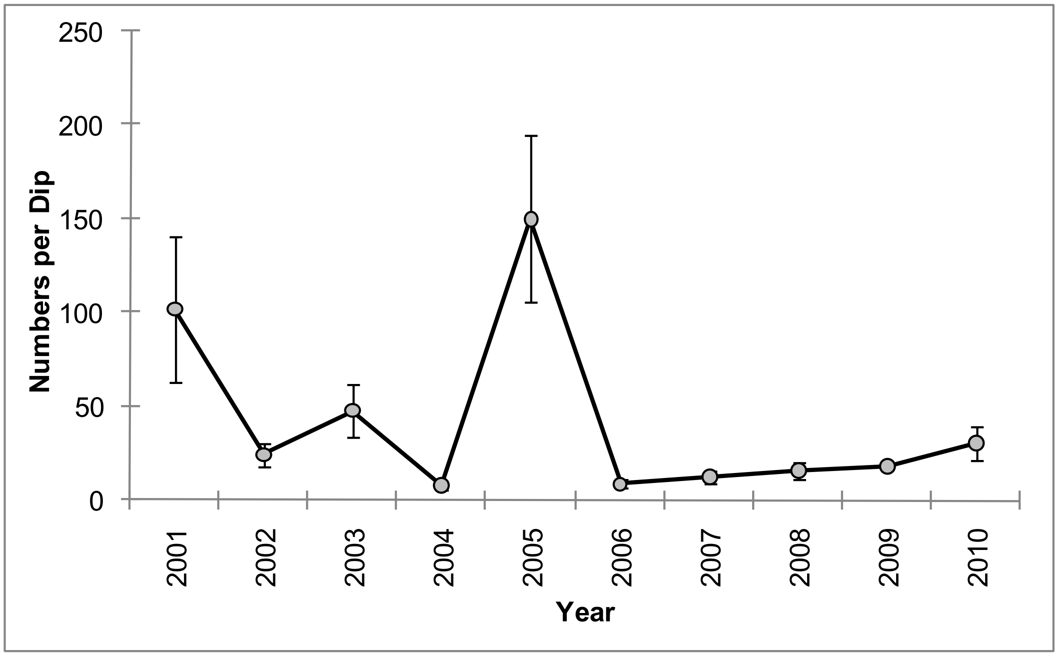

Figure 5.33. Annual index of abundance for American eels caught by Florida's annual YOY survey near Guana River Dam, 2001–2010. The error bars represent the standard errors about the estimates.

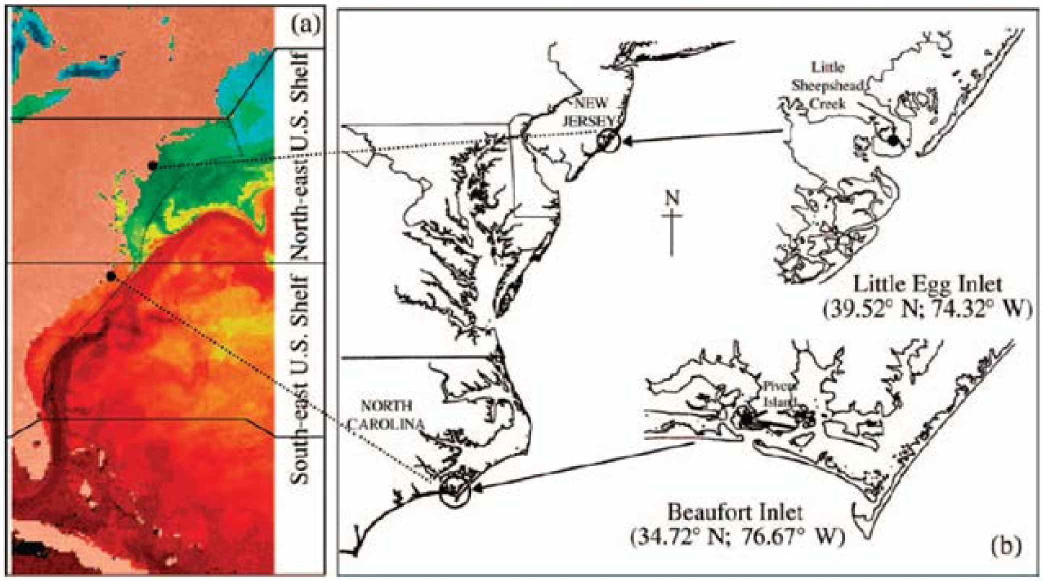

Figure 5.34. Map of Little Egg Inlet Ichthyoplankton and Beaufort Inlet Ichthyoplankton Survey study areas. (Adapted from Sullivan et al. 2006.)

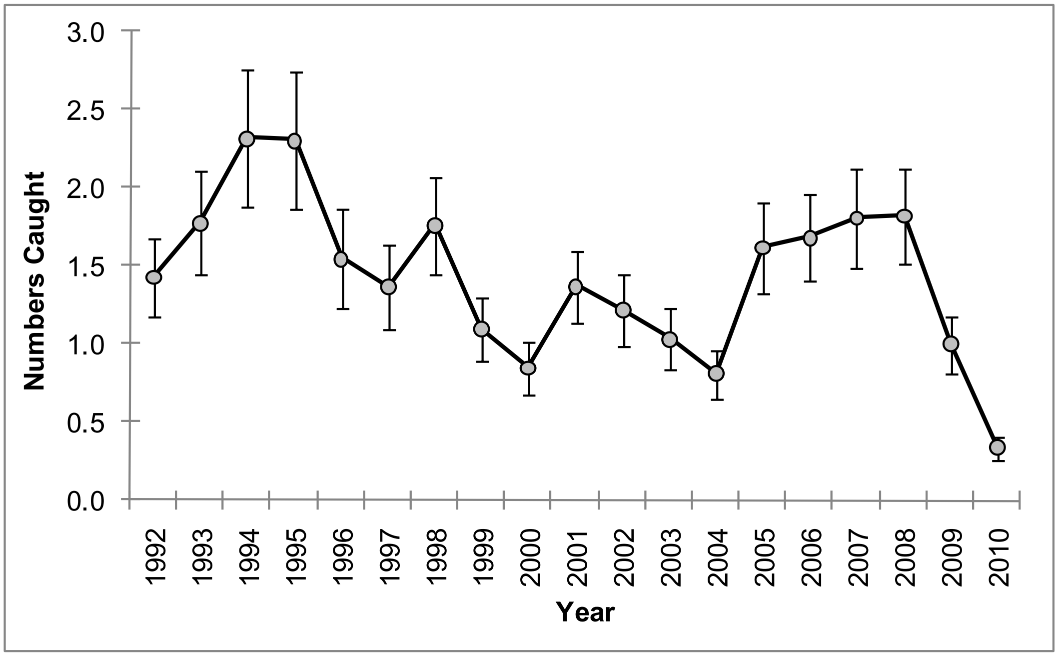

Figure 5.35. GLM-standardized index of abundance for YOY American eels caught by the Little Egg Inlet Ichthyoplankton Survey, 1992–2010. The error bars represent the standard errors about the estimates.

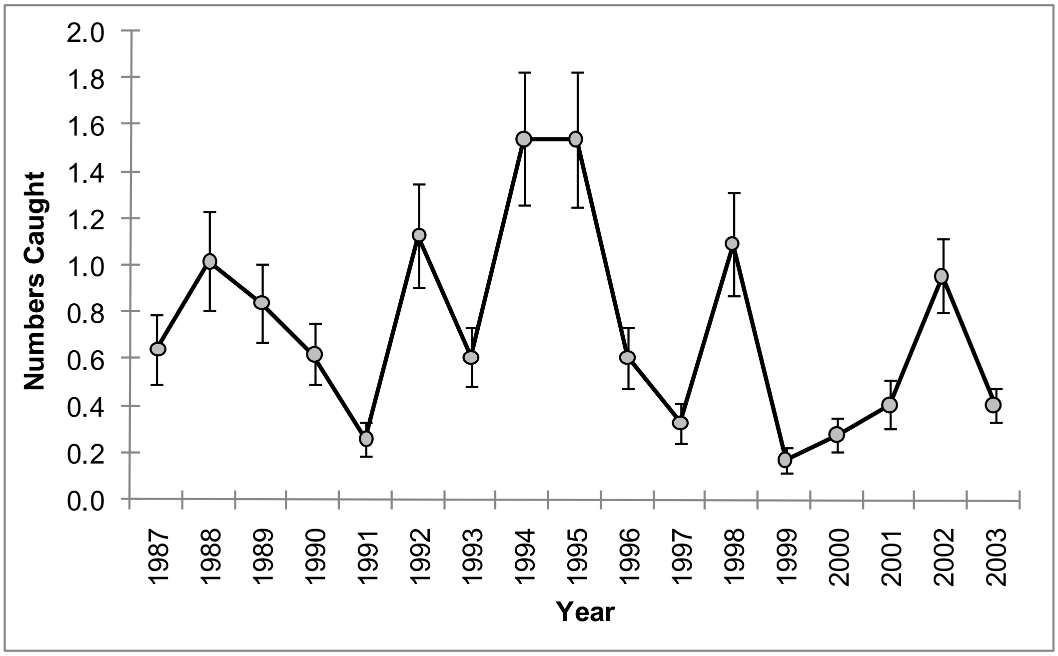

Figure 5.36. GLM-standardized index of abundance for American eels caught by the Beaufort Inlet Ichthyoplankton Survey, 1987–2003. The error bars represent the standard errors about the estimates.

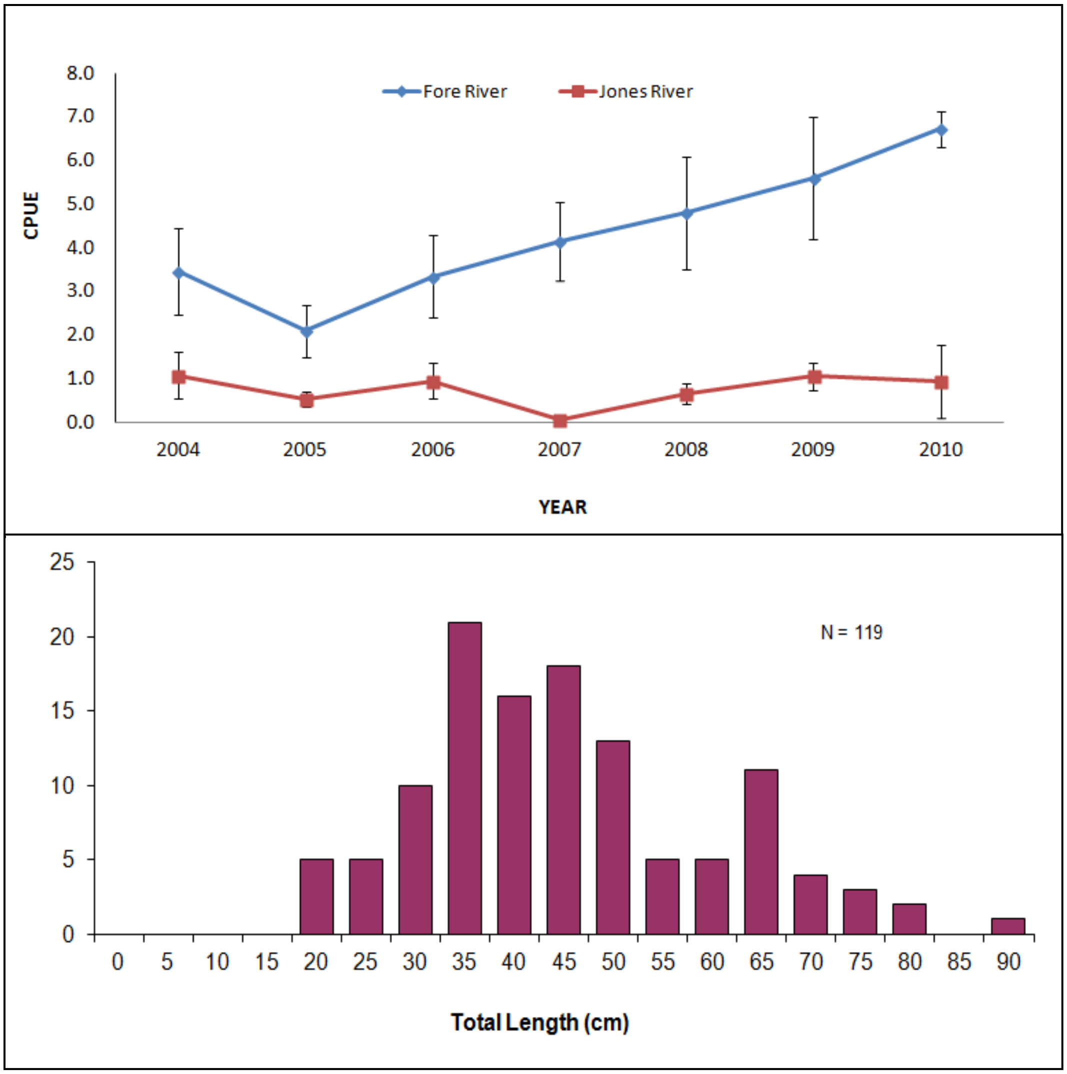

Figure 5.37. CPUE (upper graph) and length frequency (lower graph) of American eels caught as bycatch in the MADMF rainbow smelt survey in the Fore and Jones rivers, 2004–2010.

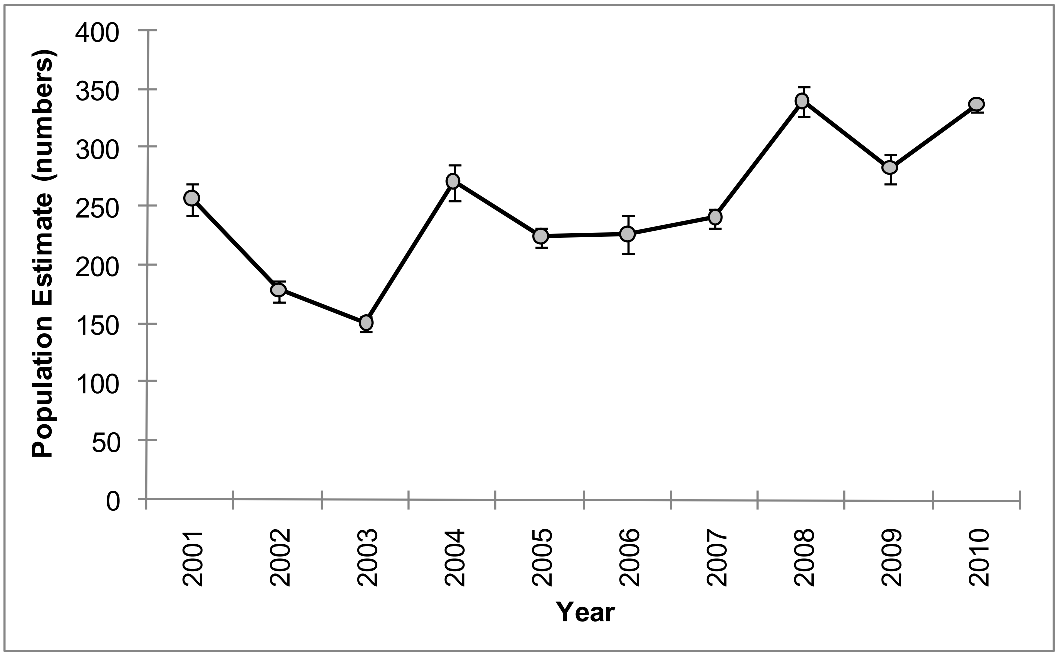

Figure 5.38. Annual index of abundance for American eels caught by the CTDEP Electrofishing Survey in the Farmill River, 2001–2010. The error bars represent the standard errors about the estimates.

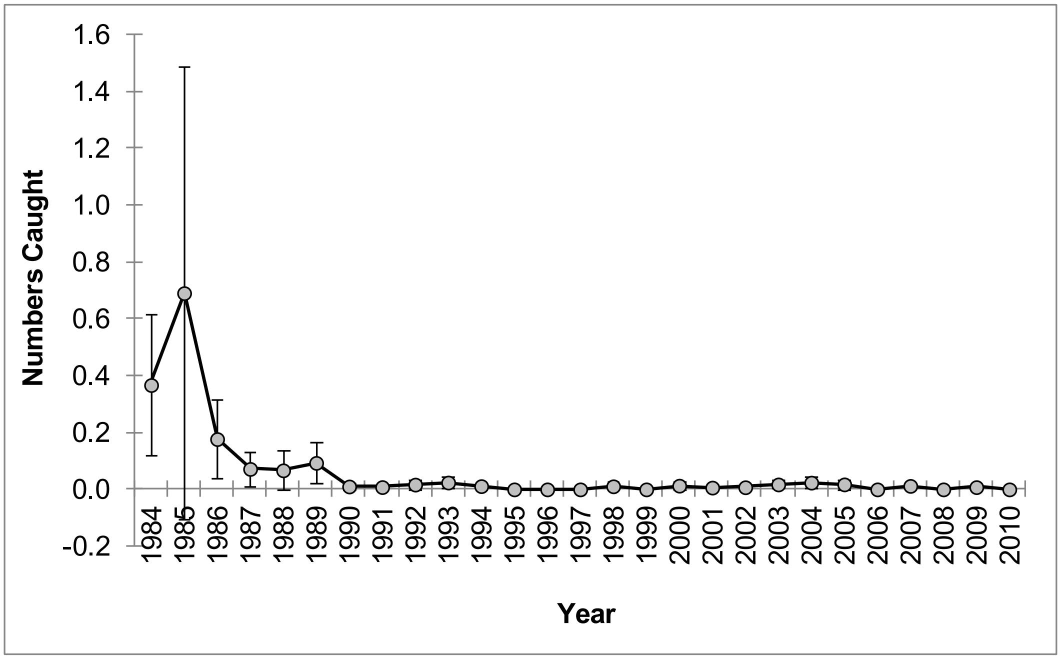

Figure 5.39. GLM-standardized index of abundance for American eels caught by the Western Long Island Study, 1984–2010. The error bars represent the standard errors about the estimates.

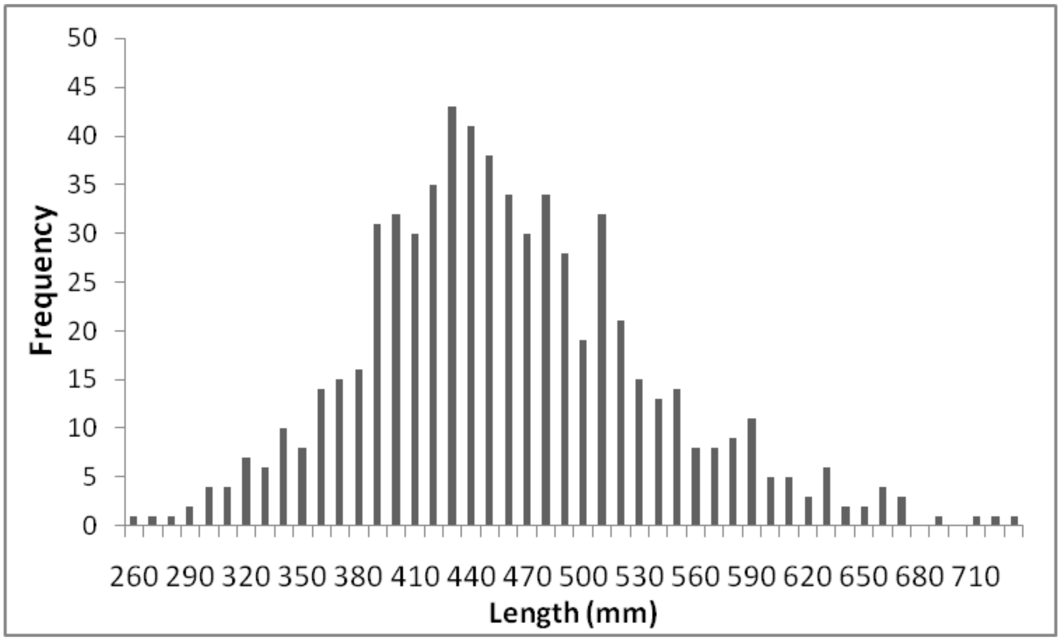

Figure 5.40. Length distribution of eel collected by Morrison and Secor (2003, 2004) from tidal portion of the Hudson River estuary, 1997–1999.

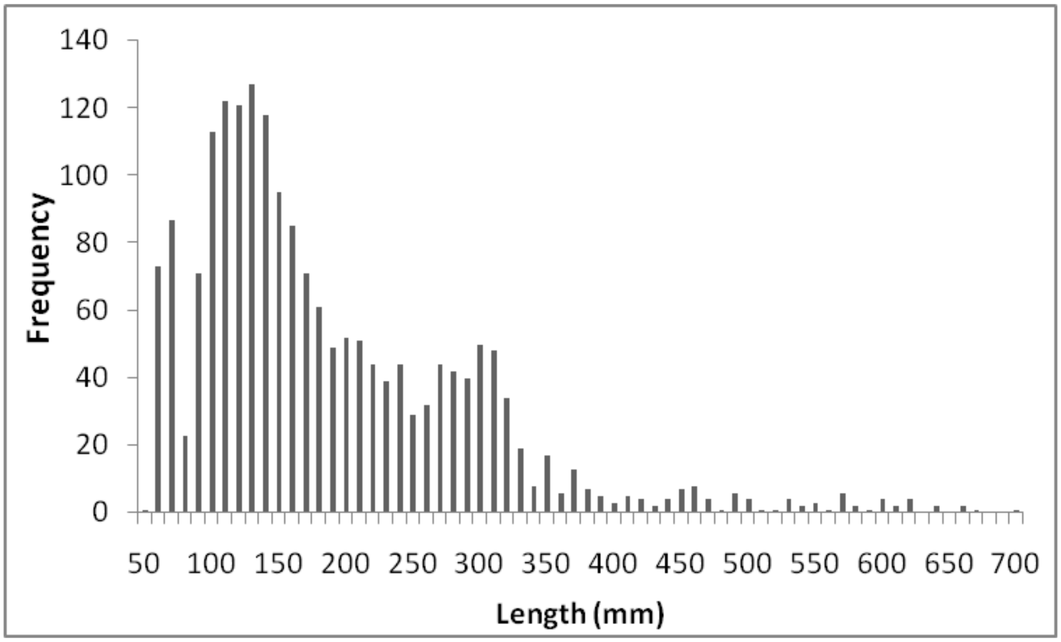

Figure 5.41. Length distribution of eel collected by Machut et al. (2007) from six Hudson River tributaries, 2003–2004.

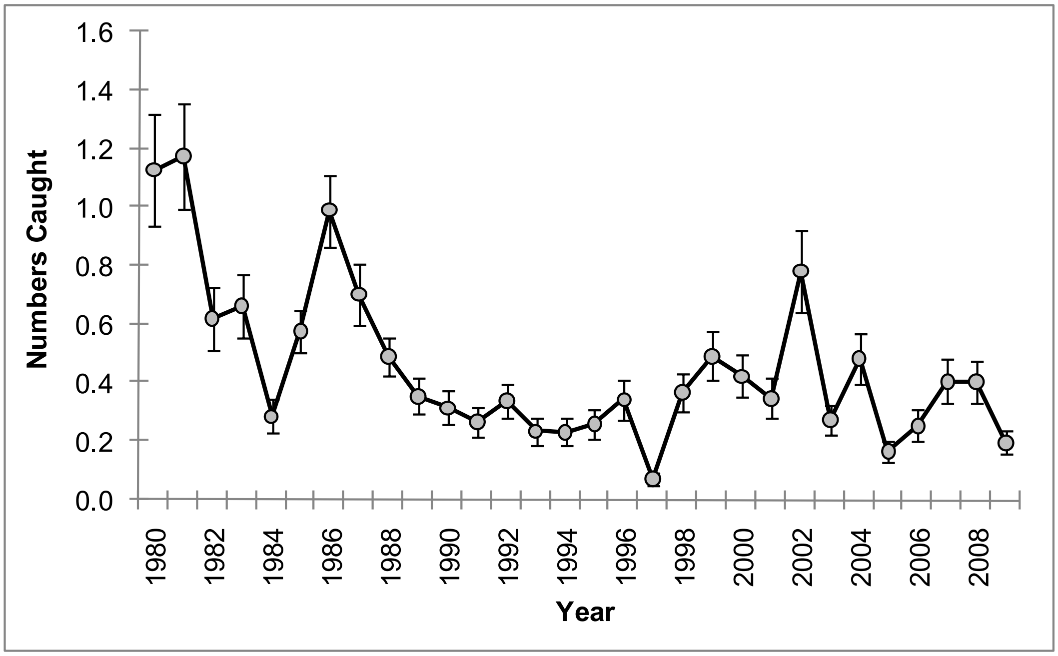

Figure 5.42. Annual index of abundance for American eels caught by the NYDEC Alosine Beach Seine Survey, 1980–2009. The error bars represent the standard errors about the estimates.

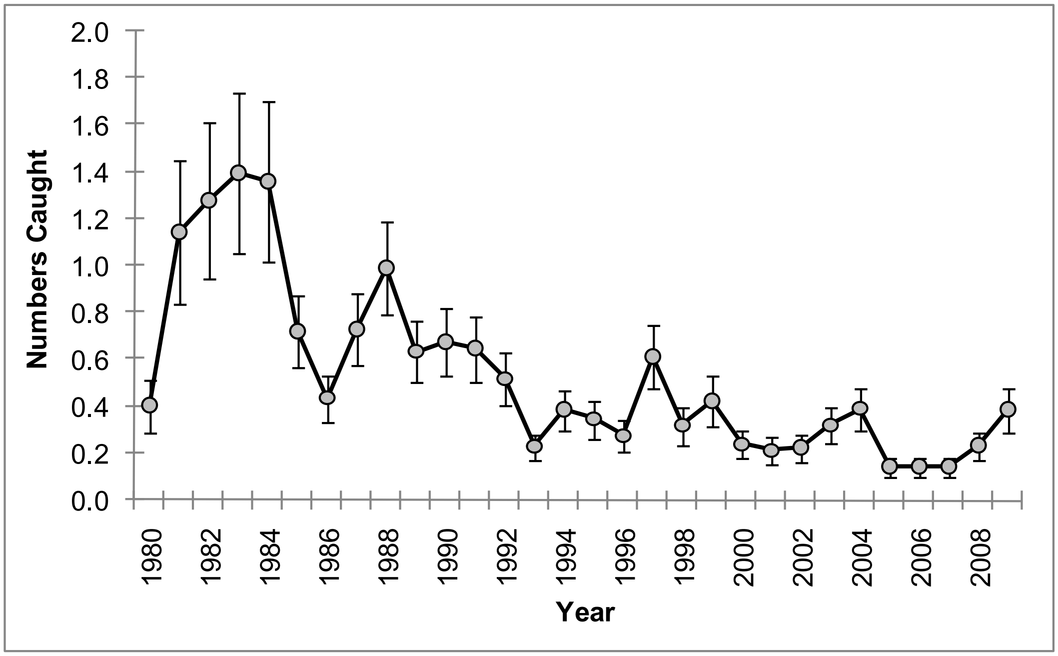

Figure 5.43. Annual index of abundance for American eels caught by the NYDEC Striped Bass Beach Seine Survey, 1980–2009. The error bars represent the standard errors about the estimates.

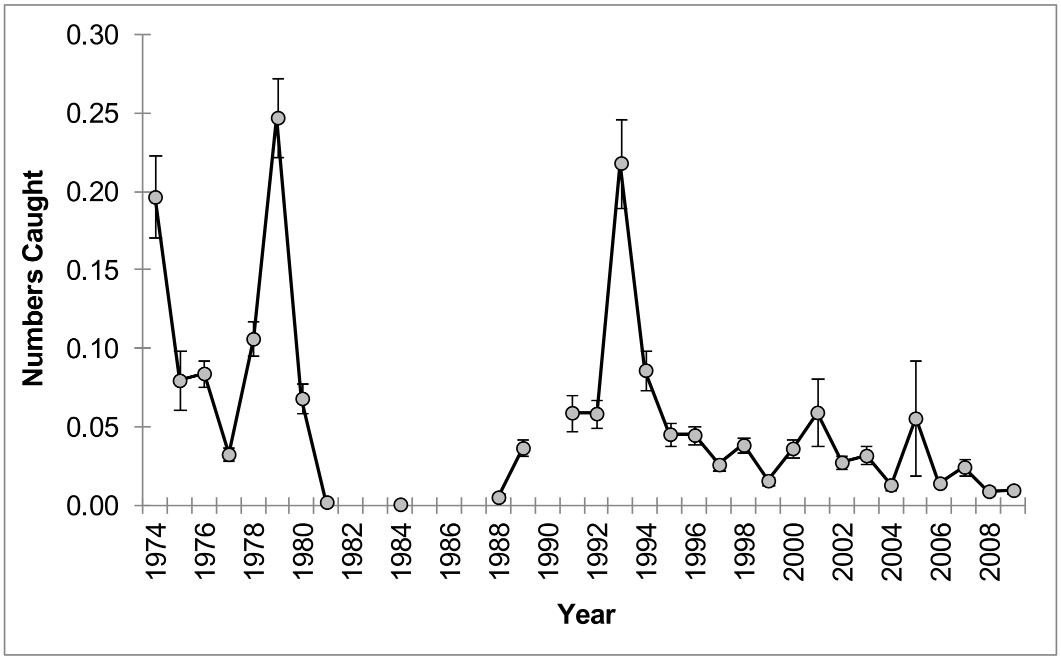

Figure 5.44. GLM-standardized index of abundance for YOY American eels caught by the HRE Monitoring Program, 1974–2009. The error bars represent the standard errors about the estimates.

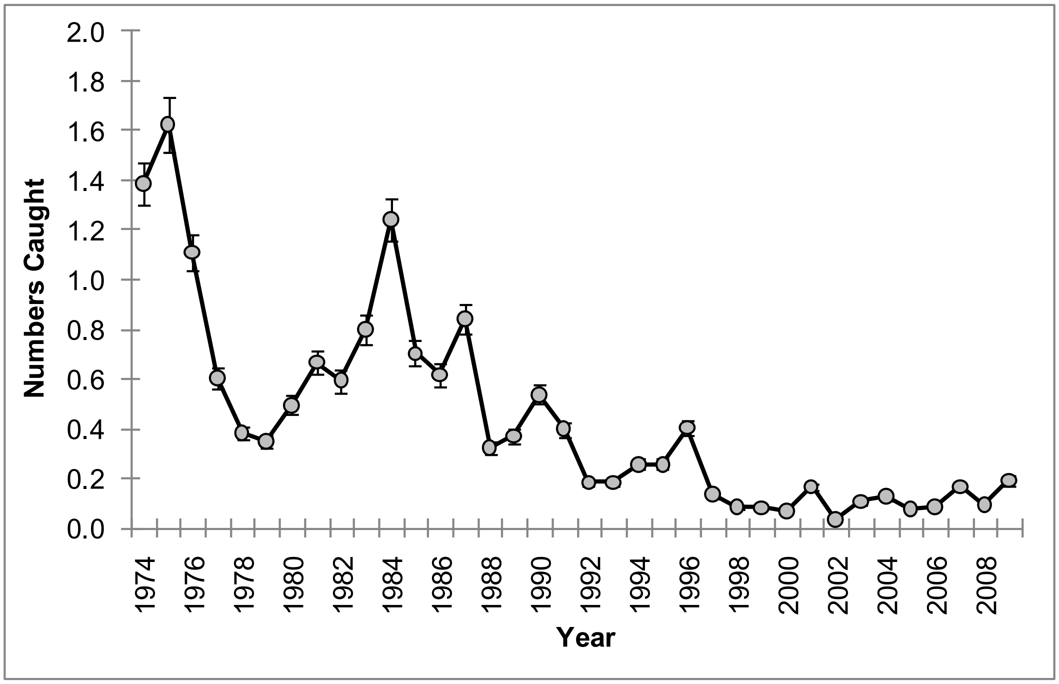

Figure 5.45. GLM-standardized index of abundance for yearling and older American eels caught by the HRE Monitoring Program, 1974–2009. The error bars represent the standard errors about the estimates.

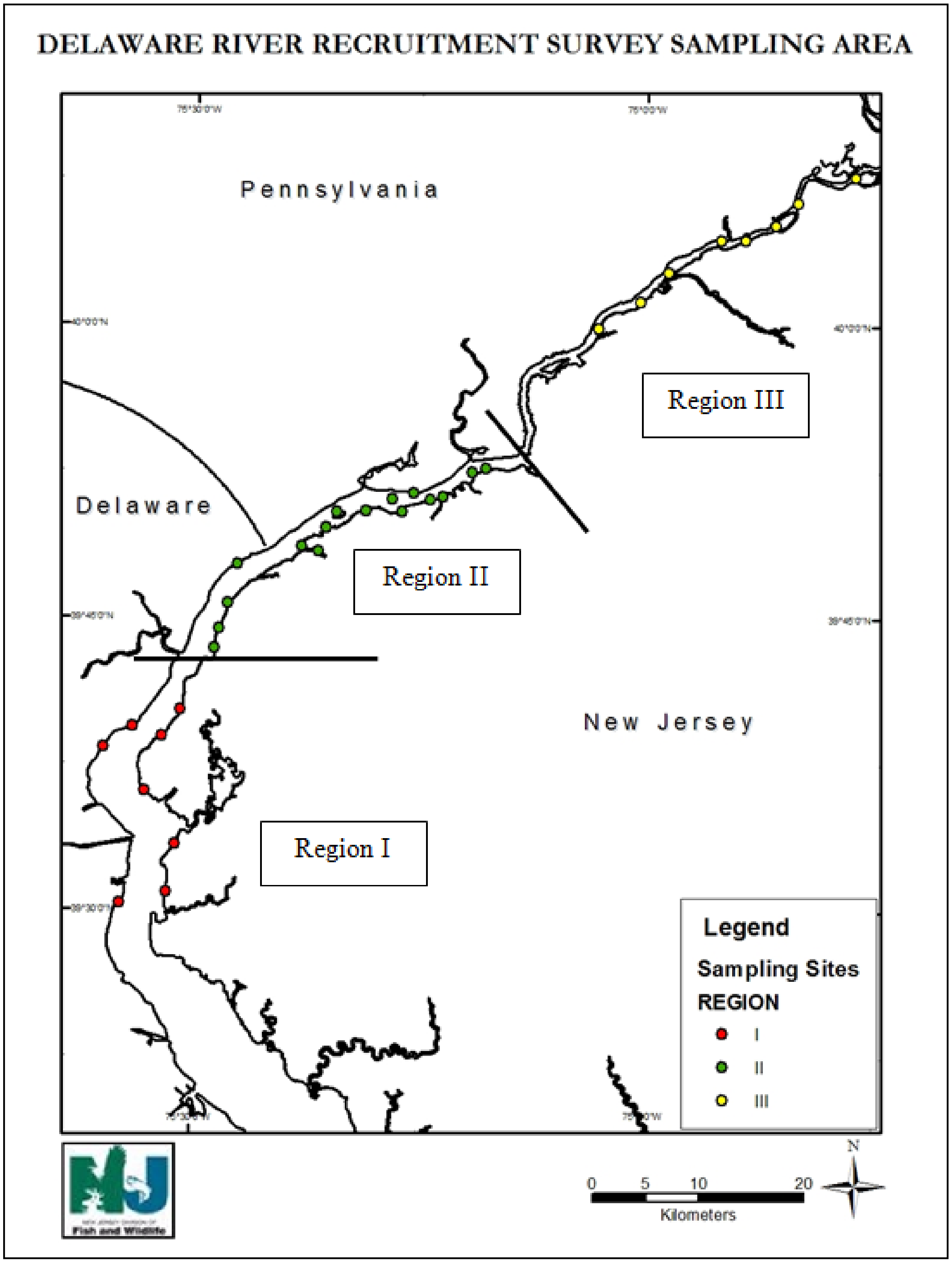

Figure 5.46. Map of Delaware River Recruitment Survey sampling stations (2011).

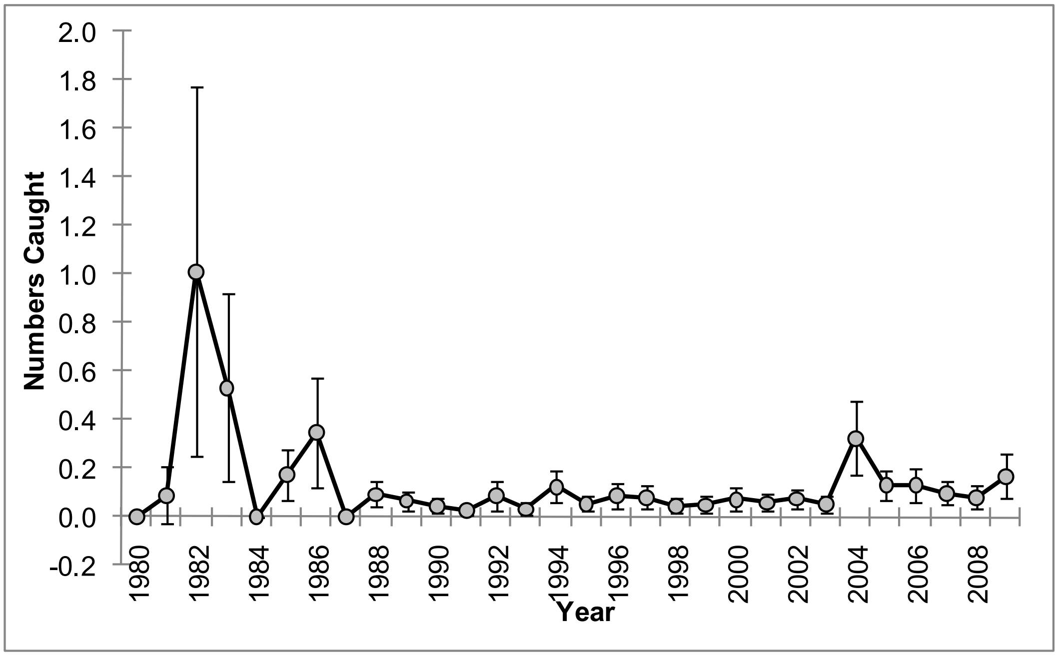

Figure 5.47. GLM-standardized index of abundance for American eels caught by NJDFW's Striped Bass Seine Survey, 1980–2009. The error bars represent the standard errors about the estimates.

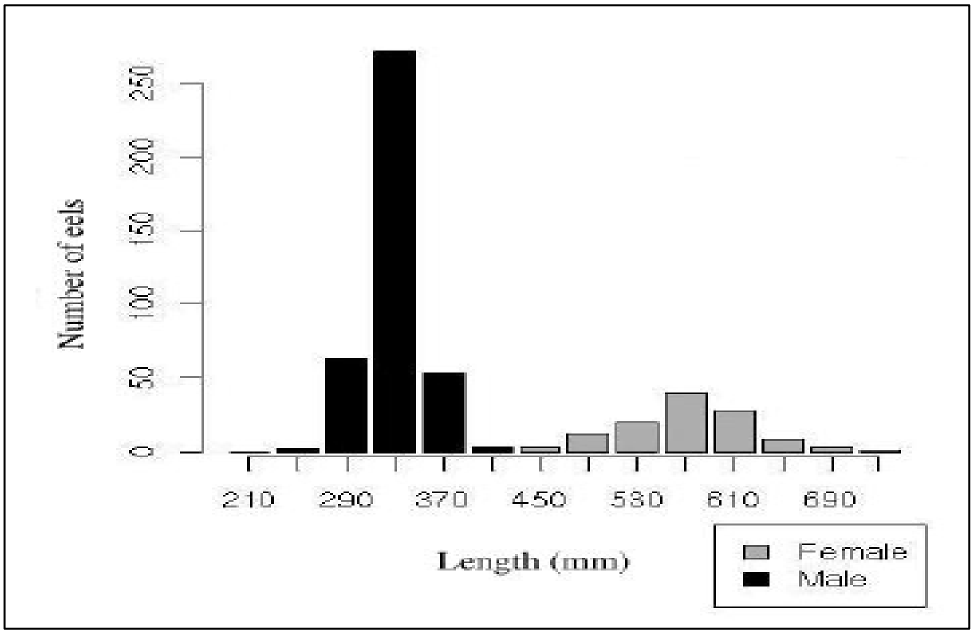

Figure 5.48. Lengths of American eels collected in the University of Delaware Silver Eel Study, by sex.

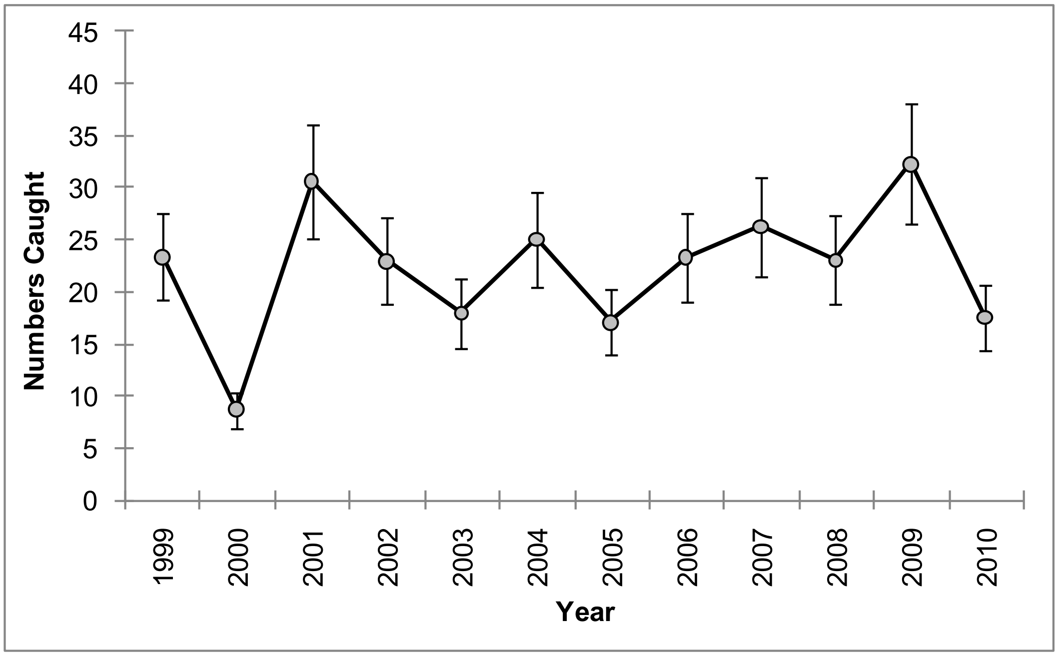

Figure 5.49. GLM-standardized index of abundance for American eels caught by the Area 6 Electrofishing Survey, 1999–2010. The error bars represent the standard errors about the estimates.

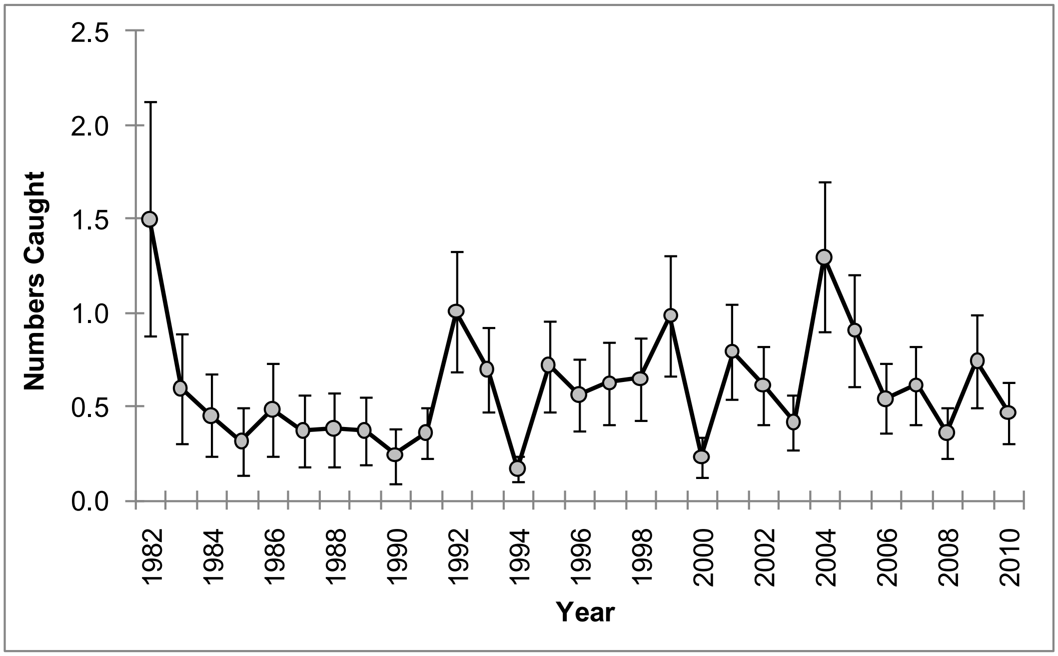

Figure 5.50. GLM-standardized index of abundance for American eels caught by the Delaware Trawl Survey, 1982–2010. The error bars represent the standard errors about the estimates.

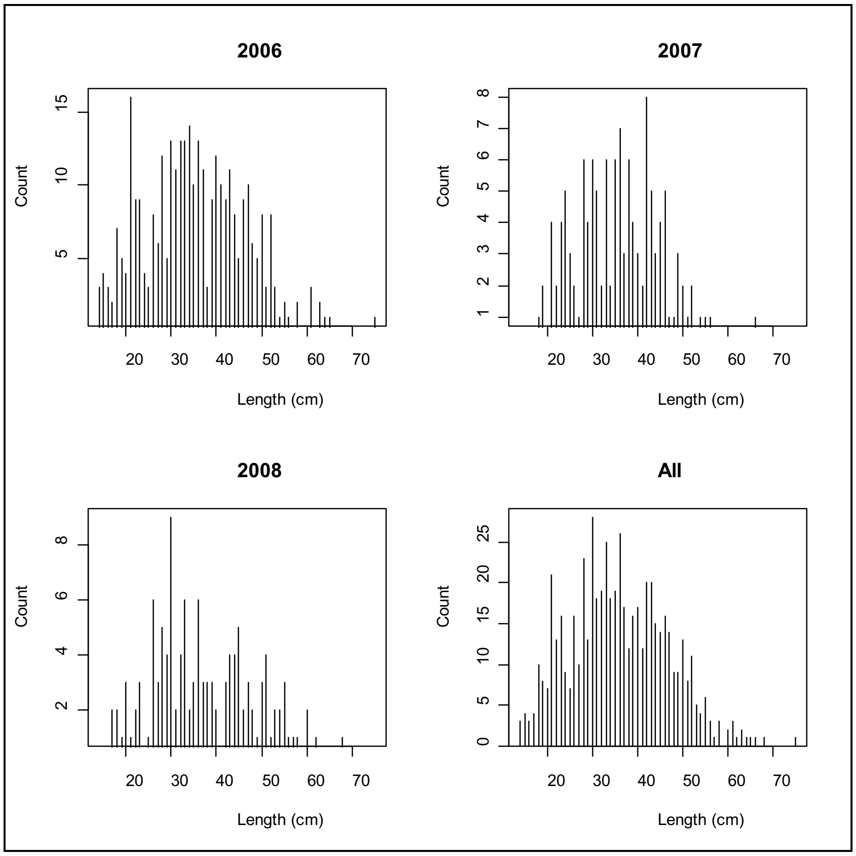

Figure 5.51. Length frequency data from upper Delaware electrofishing samples (Source: The Nature Conservancy).

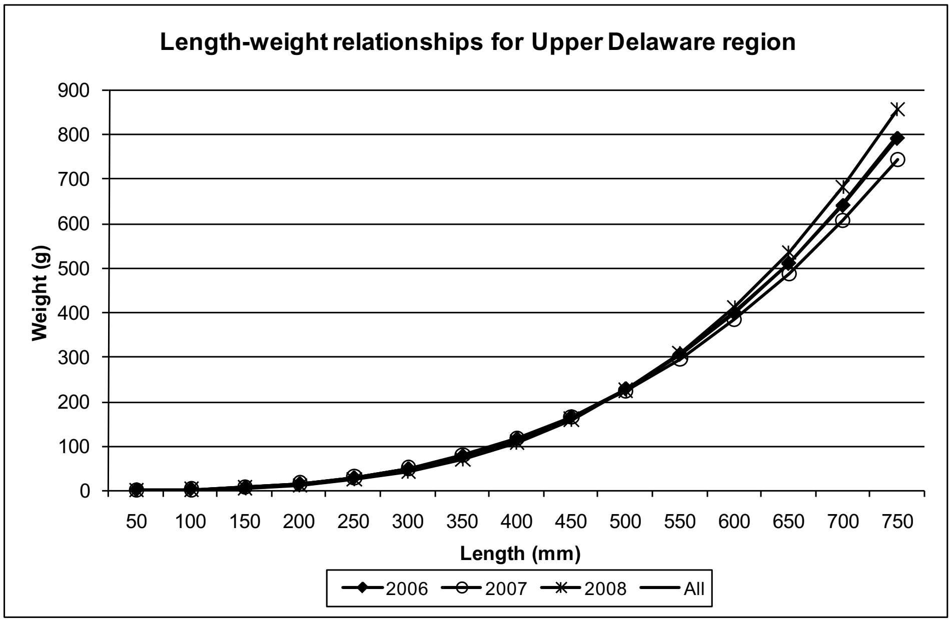

Figure 5.52. Length-weight relationship for Upper Delaware River samples.

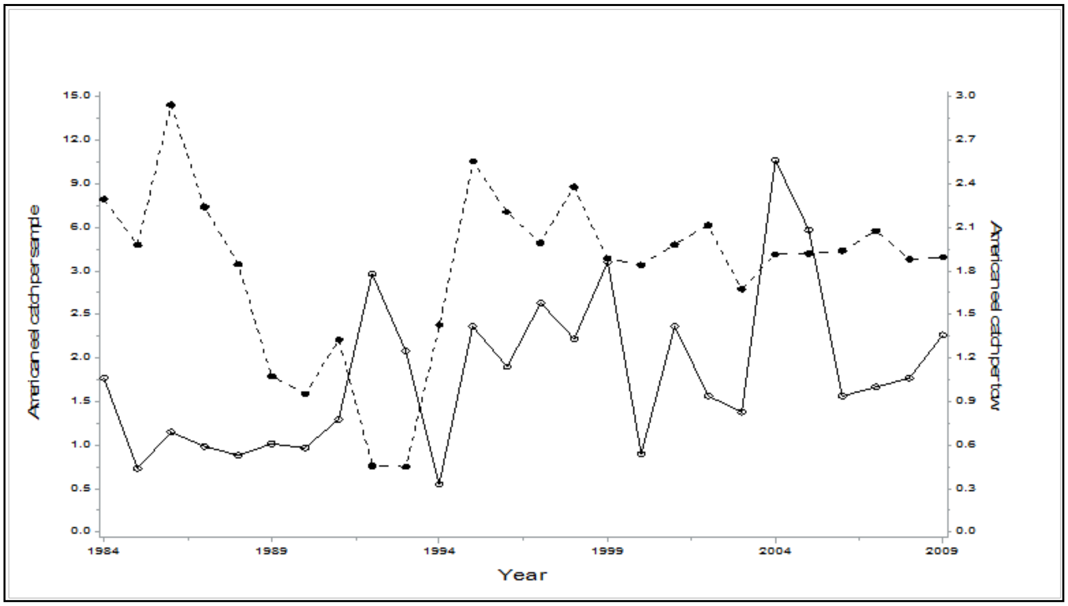

Figure 5.53. American eel abundance trends during 1984 through 2009 from the Delaware Juvenile Finfish Trawl Survey (solid) and PSEG Impingement Monitoring (open).

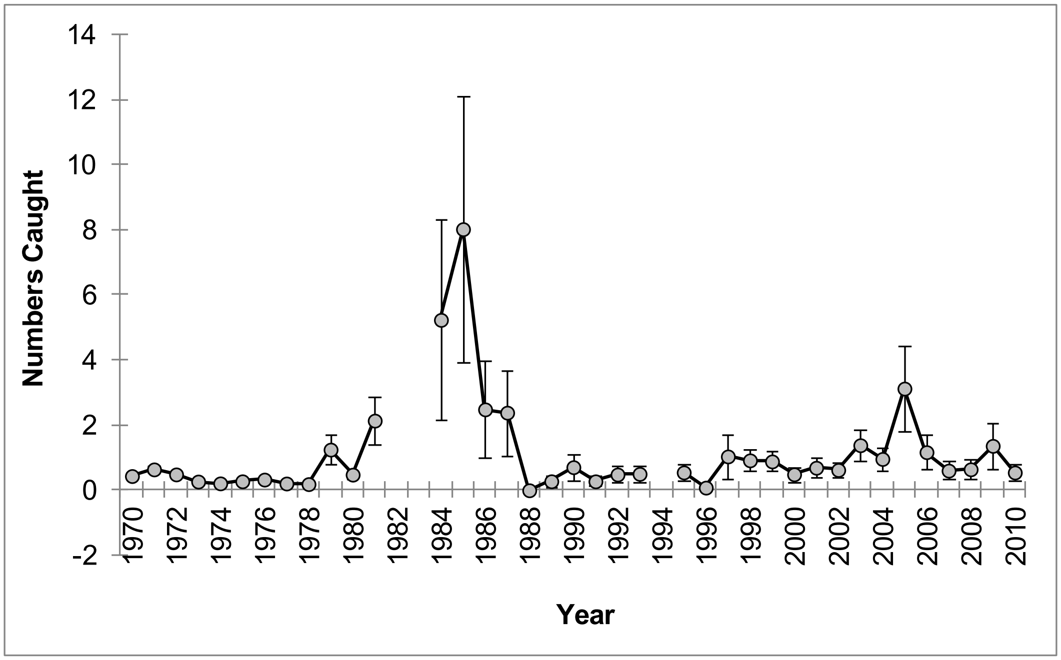

Figure 5.54. GLM-standardized index of abundance for American eels caught by PSEG's Trawl Survey, 1970–2010. The error bars represent the standard errors about the estimates.

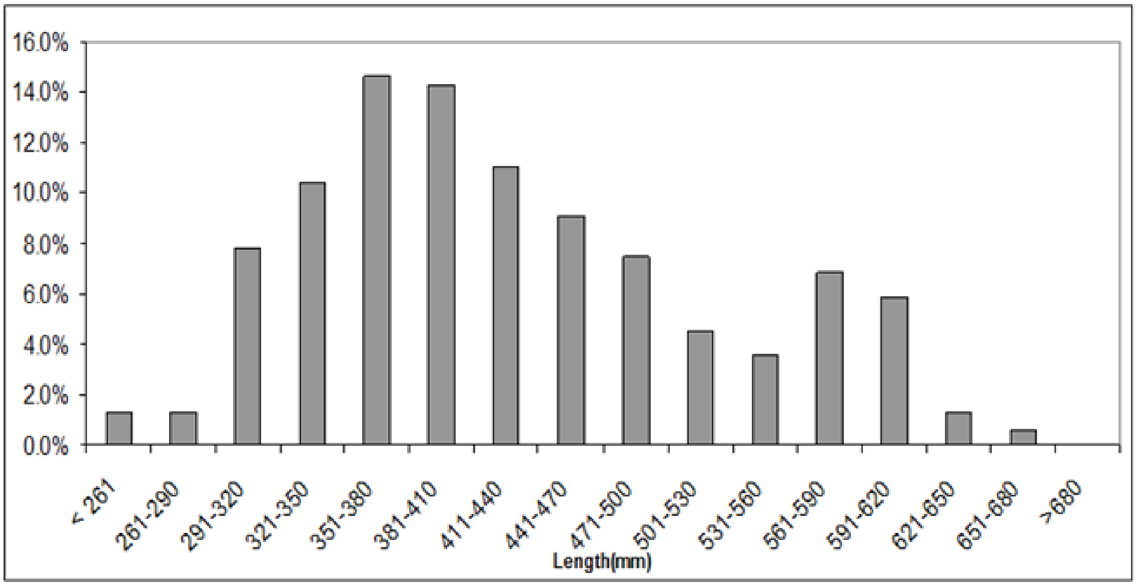

Figure 5.55. Length distribution of American eels collected by the Maryland pot survey in Turville Creek, 2009 and 2010.

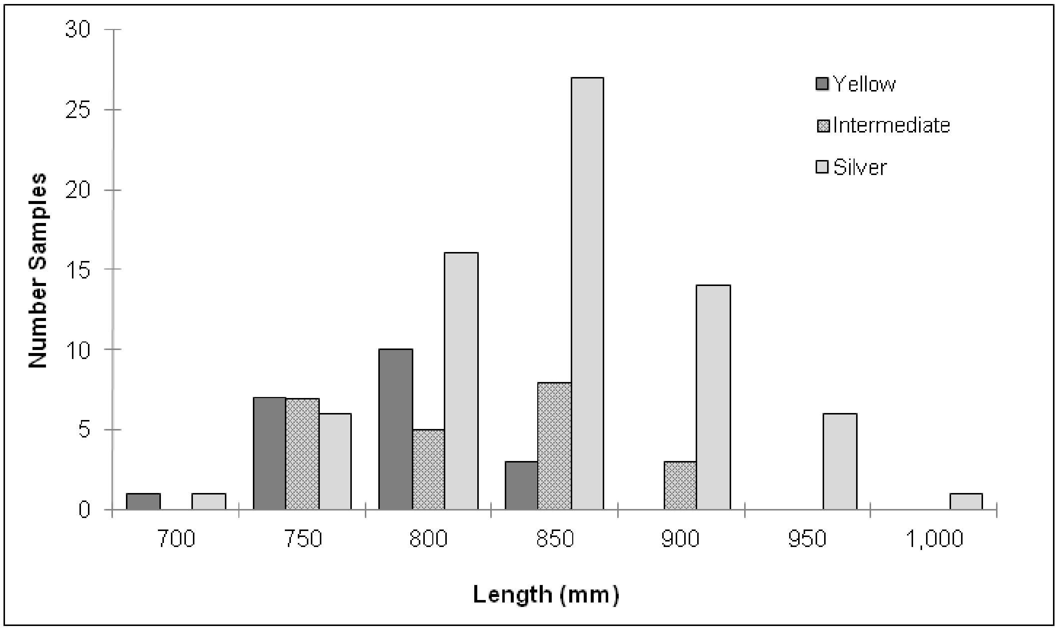

Figure 5.56 Length-frequency of American eel downstream migrants collected from the South Fork of the Shenandoah River in Virginia, 2007–2008. (Data Source: Welsh et al. 2009).

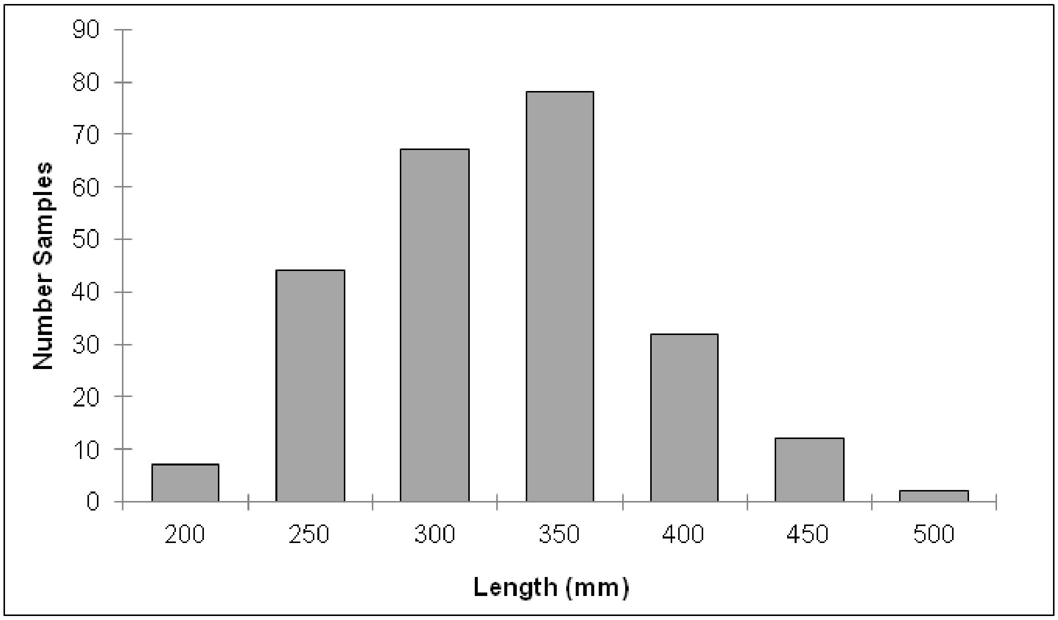

Figure 5.57. Length-frequency of American eel upstream migrants collected from the Millville Dam eel ladder on the lower Shenandoah River, 2006–2008. (Data Source: Zimmerman 2008).

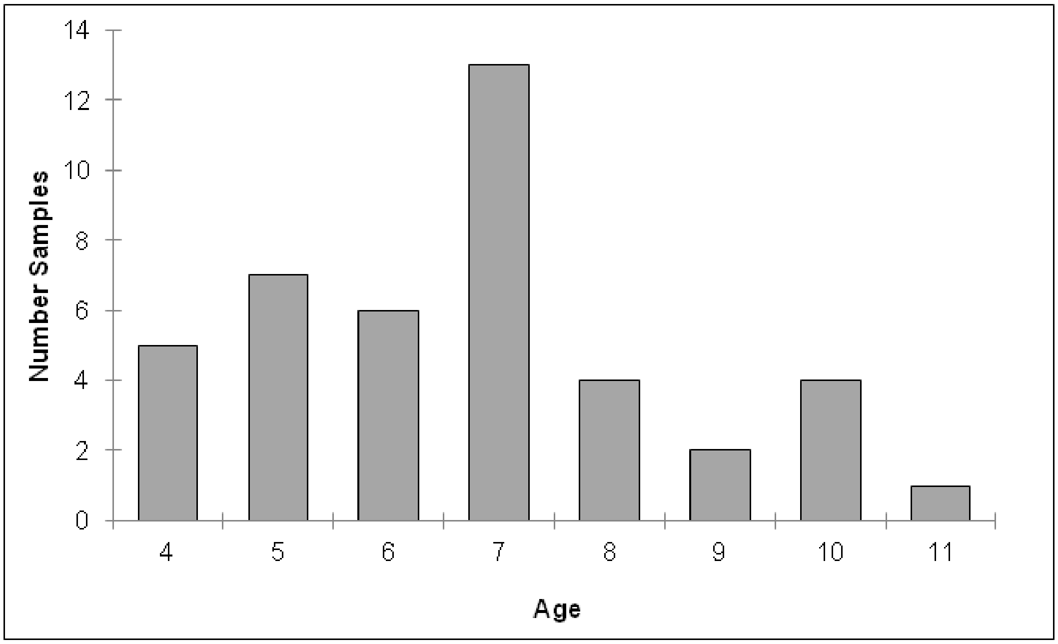

Figure 5.58. Age-frequency of American eel upstream migrants collected from the Millville Dam eel ladder on the lower Shenandoah River, 2006–2008. (Data Source: Zimmerman 2008).

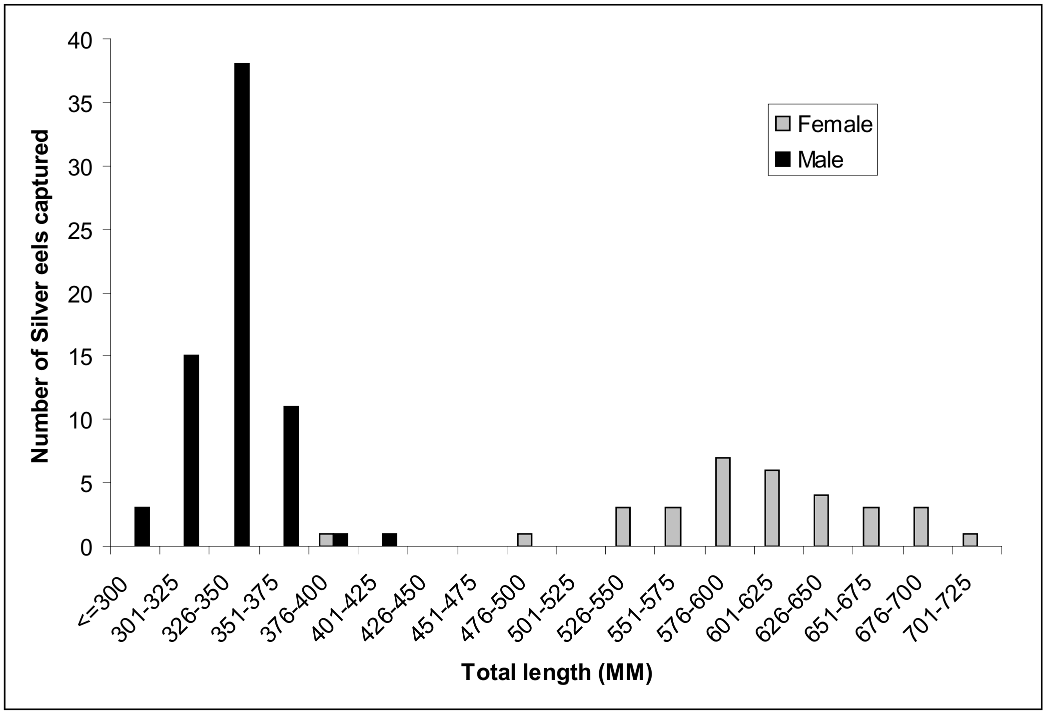

Figure 5.59. Maryland Gravel Run survey silver eel length distribution by sex, 2006–2010.

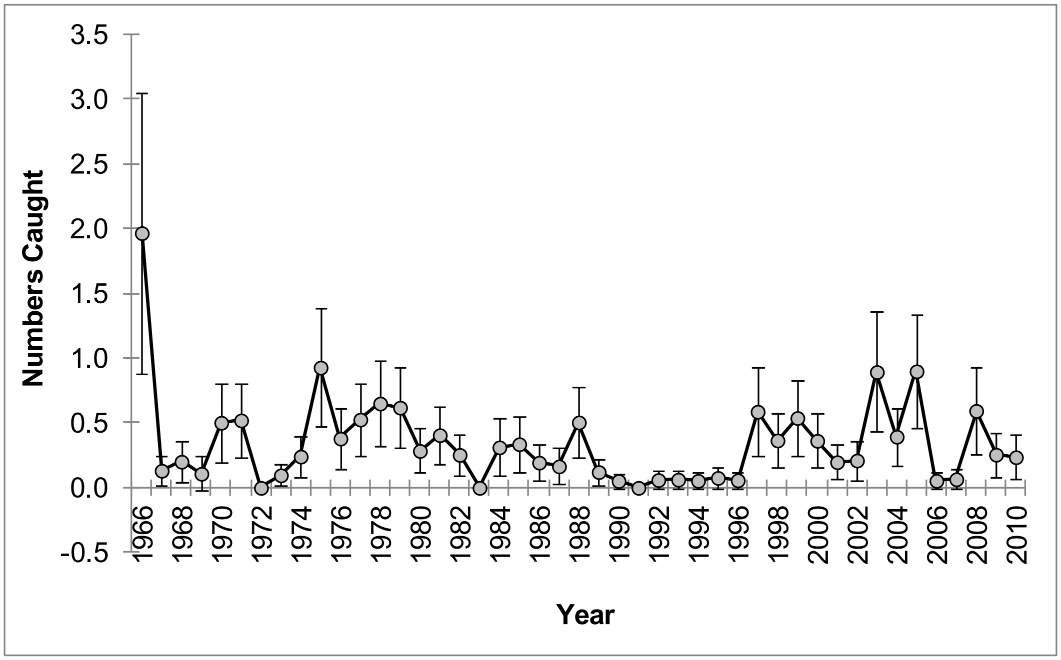

Figure 5.60. GLM-standardized index of abundance for American eels caught by the MDDNR Striped Bass Seine Survey, 1966–2010. The error bars represent the standard errors about the estimates.

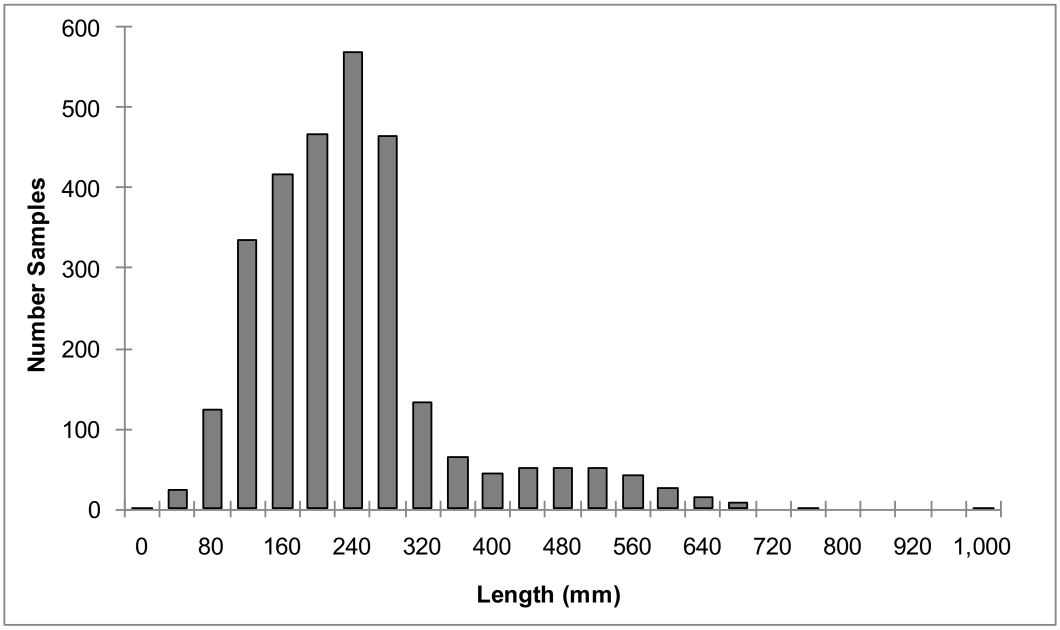

Figure 5.61. Length-frequency of American eels collected by VDGIF fishery-independent surveys of Virginia water bodies, 1992–2010**.**

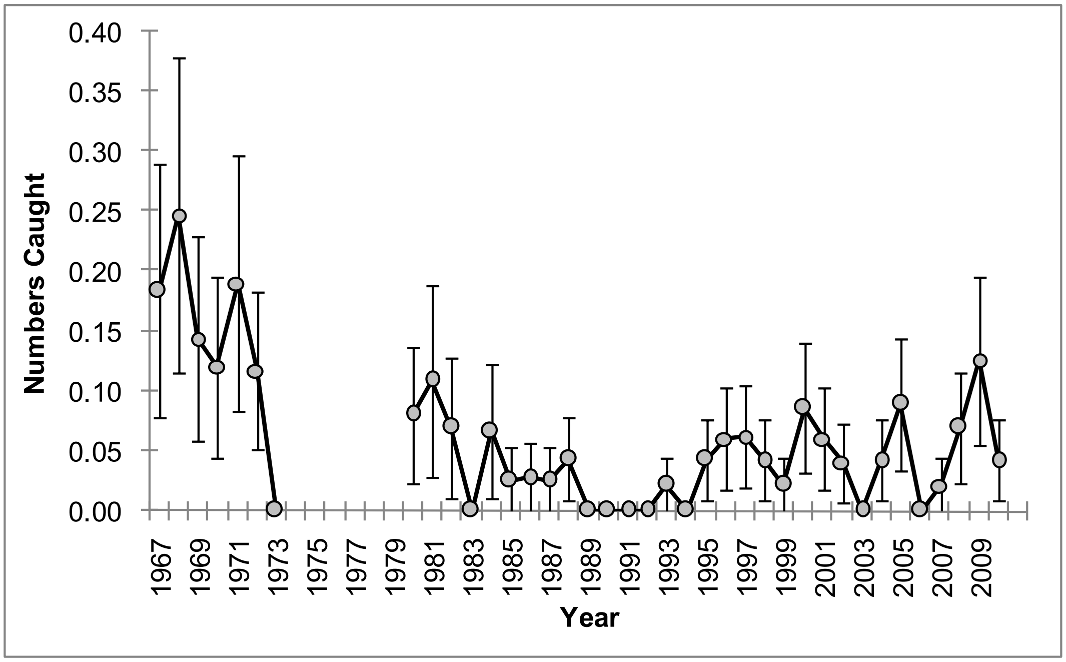

Figure 5.62. GLM-standardized index of abundance for American eels caught by the VIMS Juvenile Striped Bass Seine Survey, 1967–2010. The error bars represent the standard errors about the estimates.

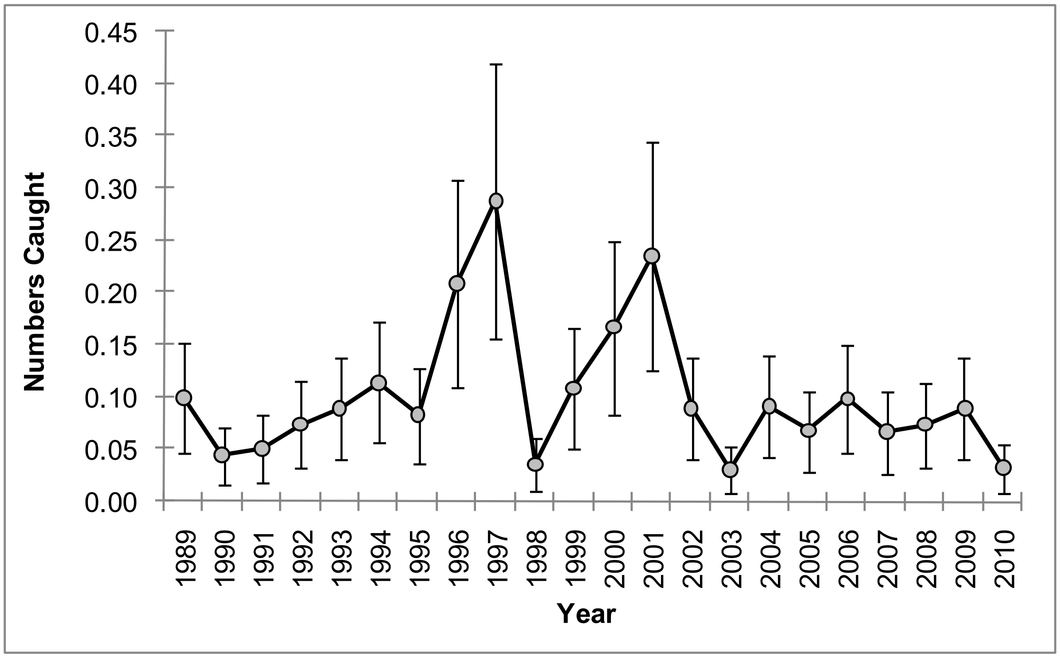

Figure 5.63. GLM-standardized index of abundance for American eels caught by the VIMS Juvenile Striped Bass Seine Survey, 1989–2010. The error bars represent the standard errors about the estimates.

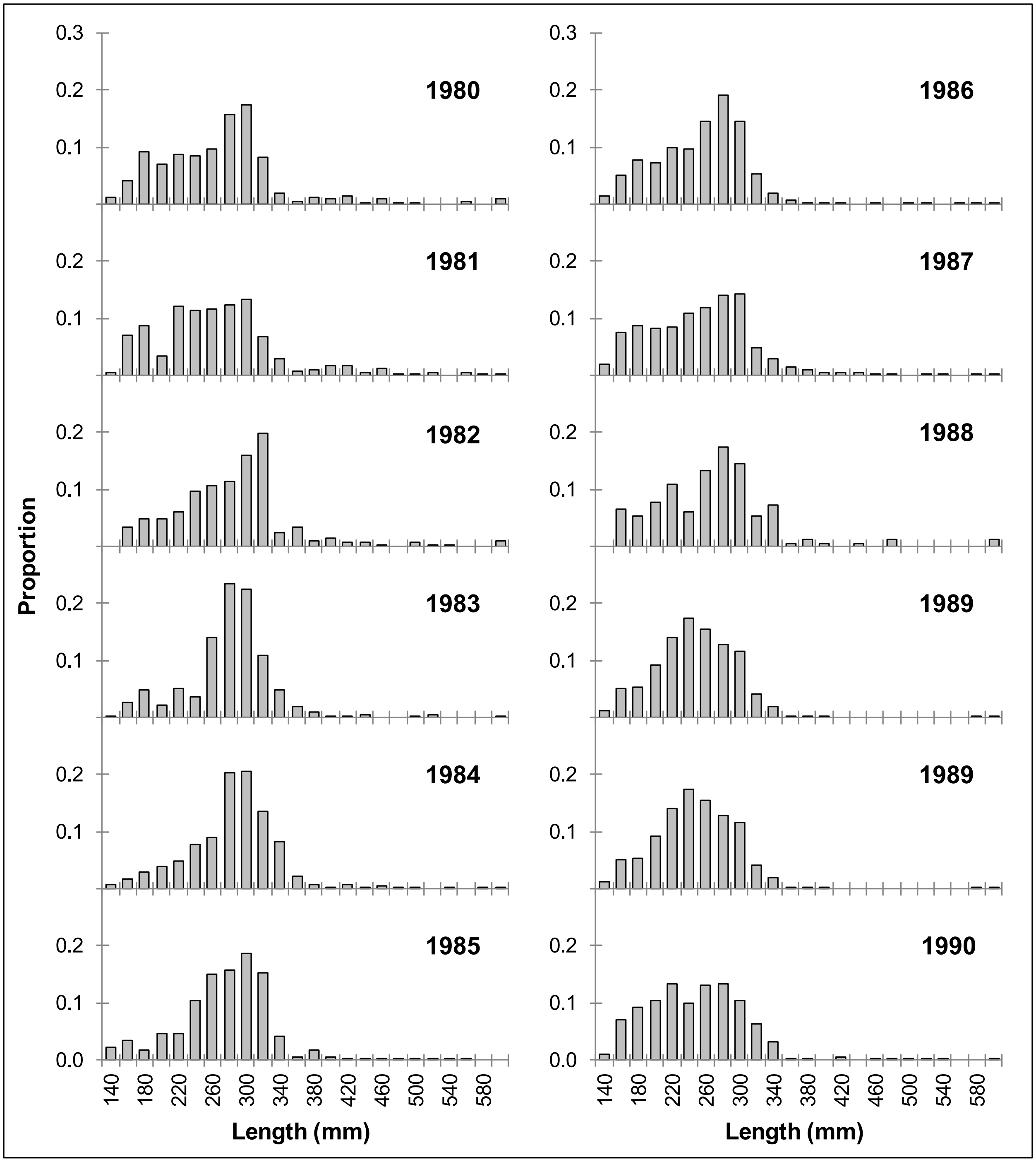

Figure 5.64. Annual length-frequency distributions of American eels collected from tributaries of the Chesapeake Bay during April through September by the VIMS Juvenile Fish and Blue Crab Trawl Survey, 1980–1990.

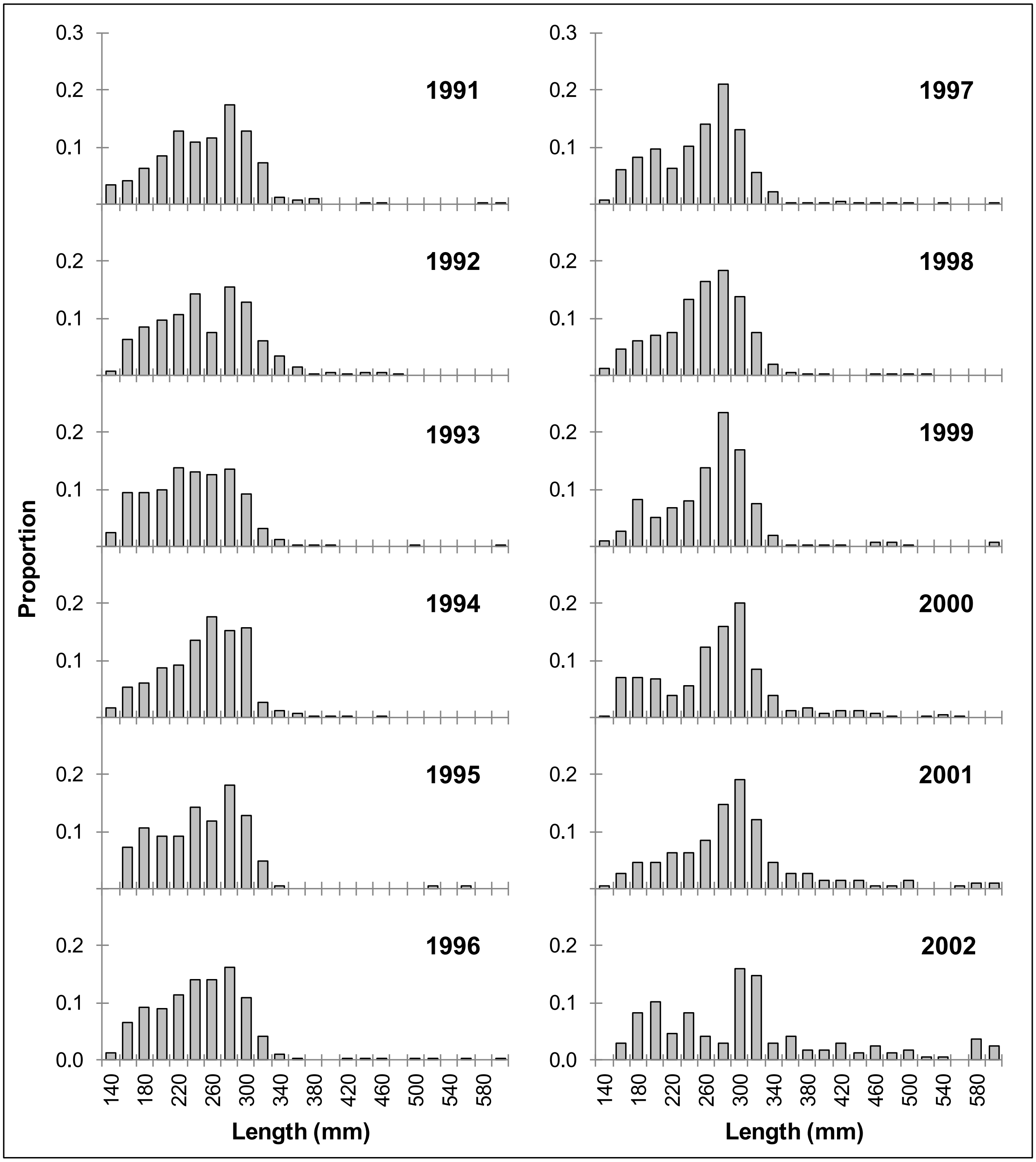

Figure 5.65. Annual length-frequency distributions of American eels collected from tributaries of the Chesapeake Bay during April through September by the VIMS Juvenile Fish and Blue Crab Trawl Survey, 1991–2002.

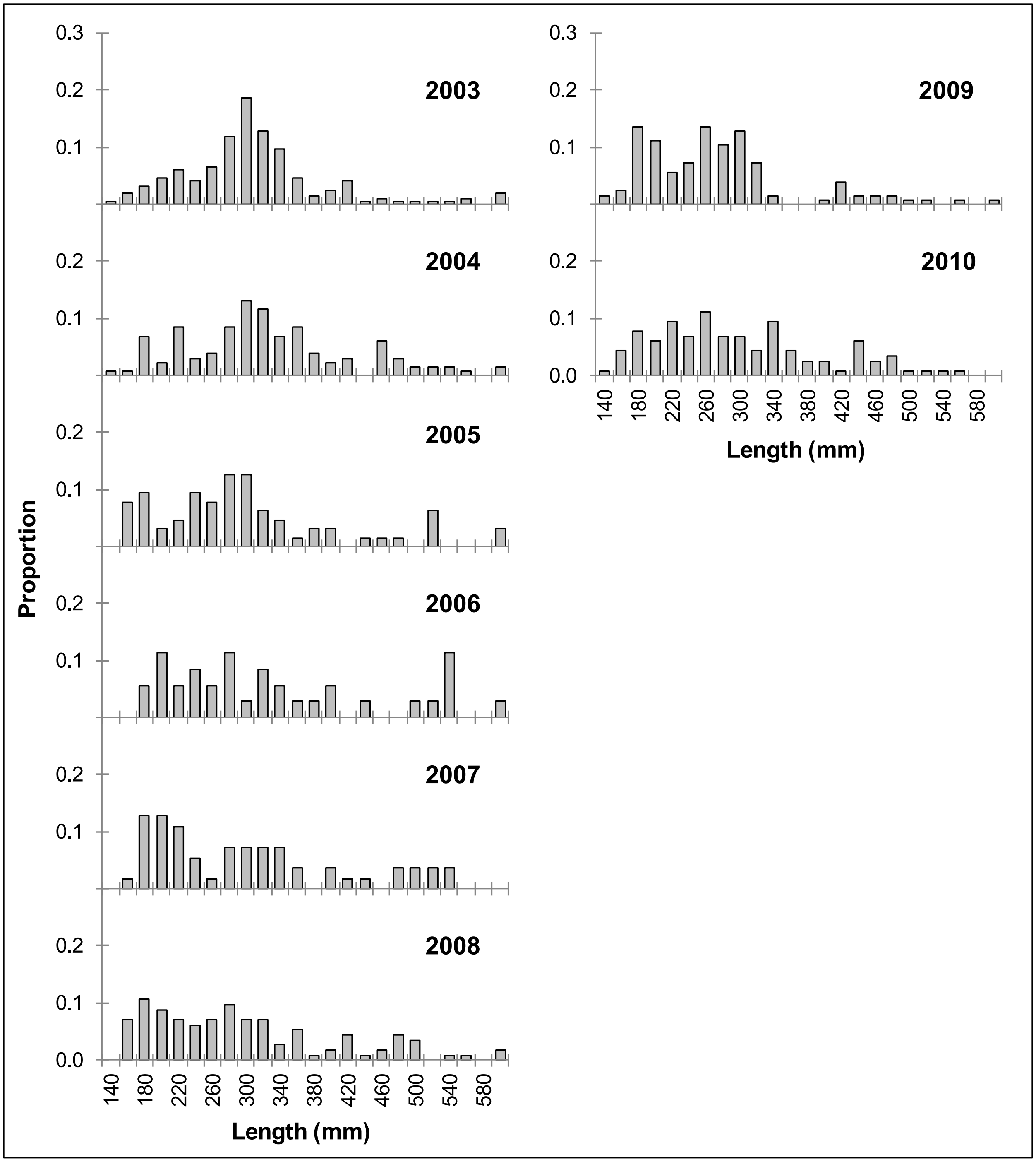

Figure 5.66. Annual length-frequency distributions of American eels collected from tributaries of the Chesapeake Bay during April through September by the VIMS Juvenile Fish and Blue Crab Trawl Survey, 2003–2010.

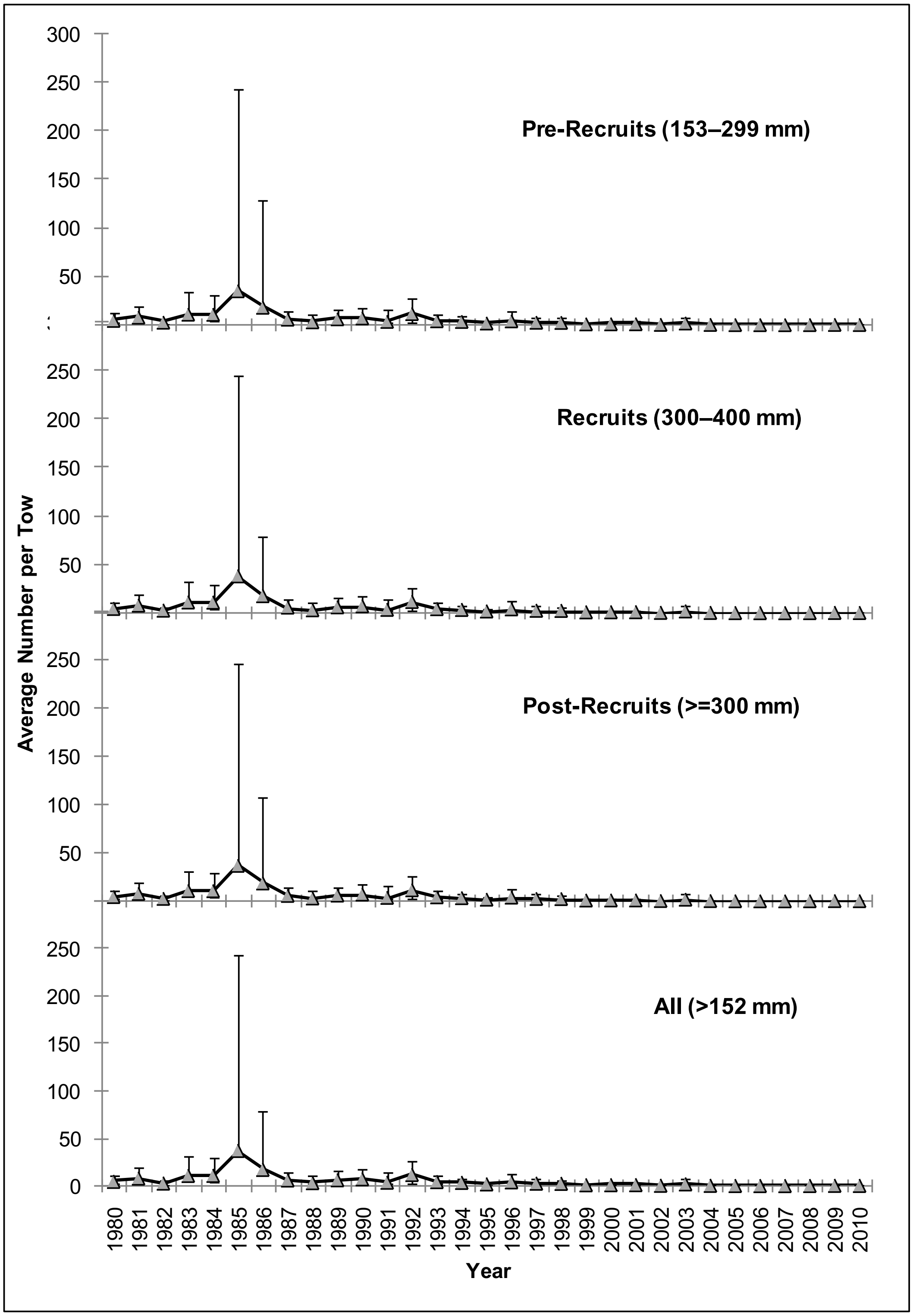

Figure 5.67. Indices of relative abundance for four size groups of American eels based on data collected from tributaries of the Chesapeake Bay during April through September by the VIMS Juvenile Fish and Blue Crab Trawl Survey, 1980–2010. Error bars represent upper and lower 95% confidence limits.

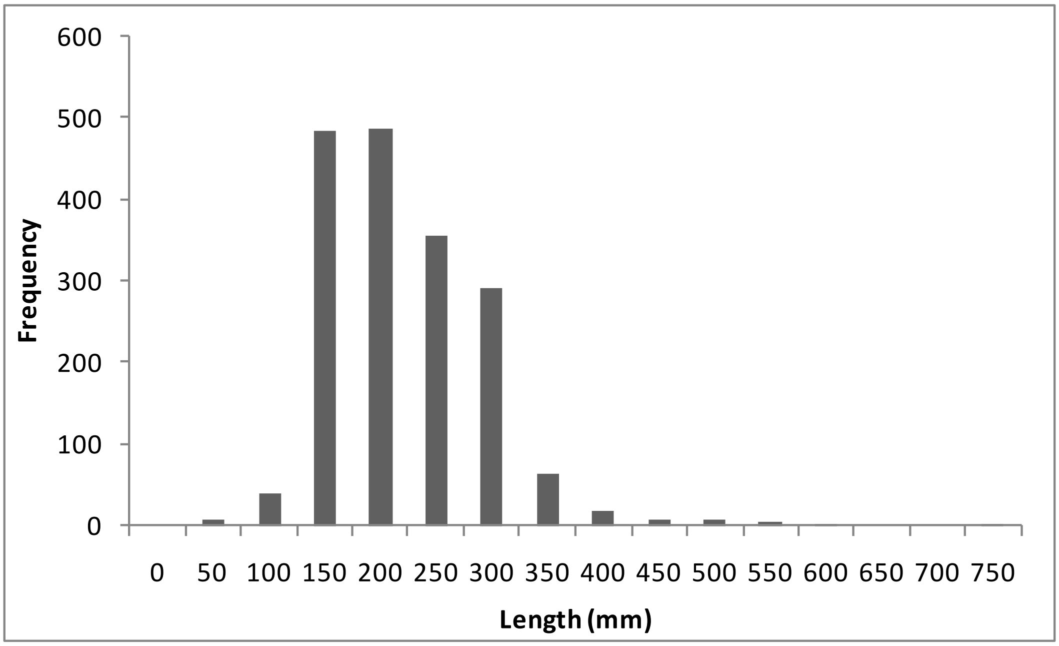

Figure 5.68. Length distribution of American eels sampled from the North Anna River, 1990–2006.

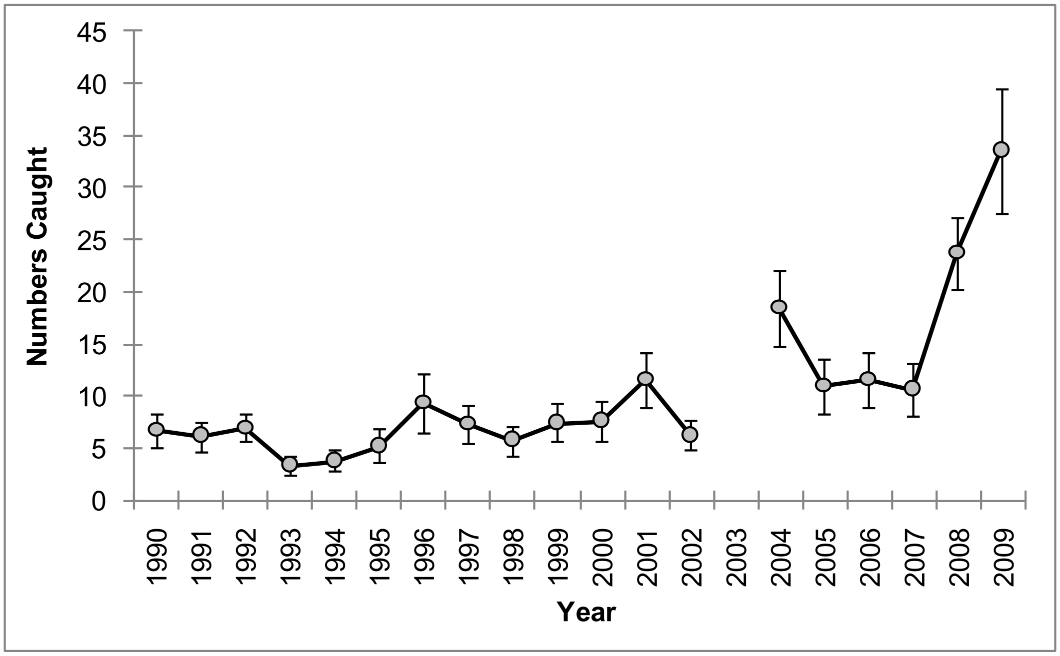

Figure 5.69. GLM-standardized index of abundance for American eels caught by the North Anna Electrofishing Survey, 1990–2009. The error bars represent the standard errors about the estimates.

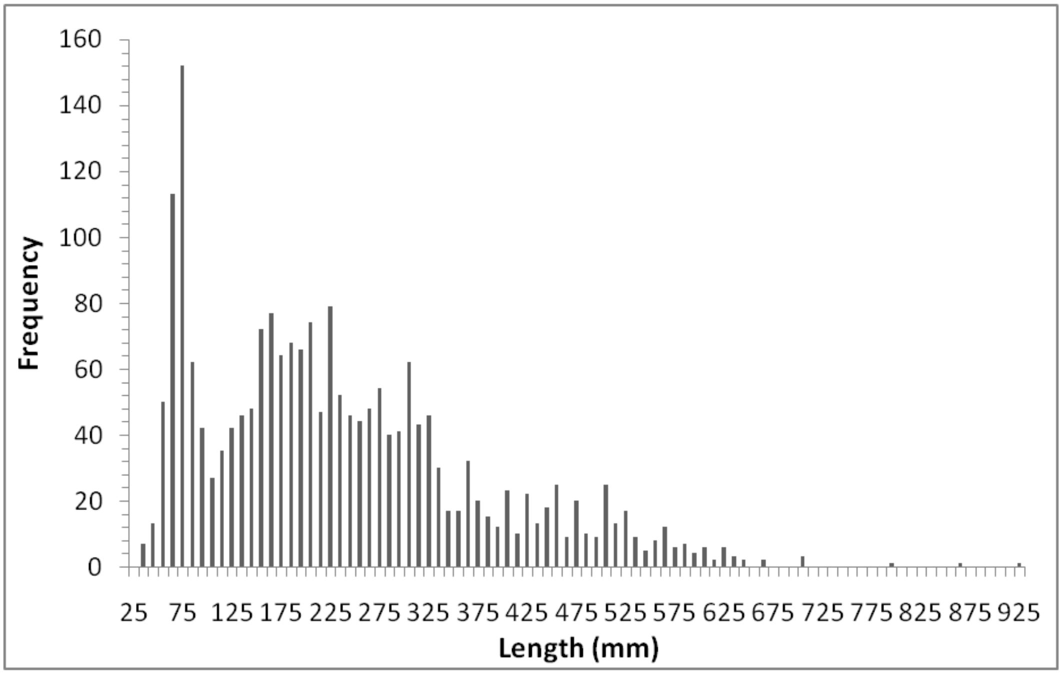

Figure 5.70. Length distribution of eels collected by the estuarine trawl survey in North Carolina waters, 1971–2010**.**

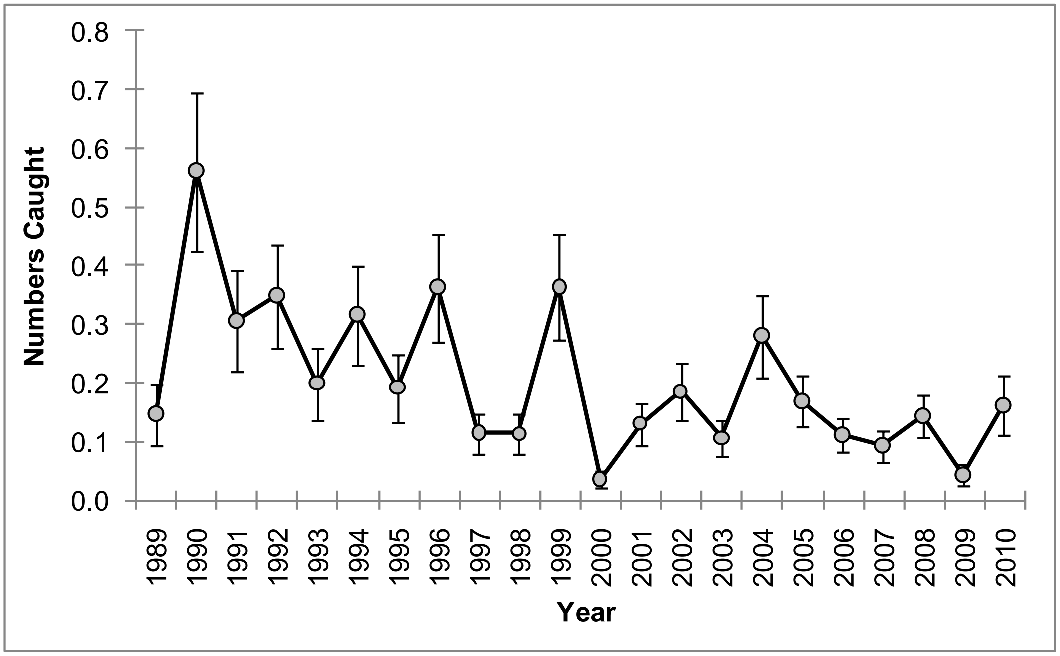

Figure 5.71. GLM-standardized index of abundance for American eels caught by the NCDMF Estuarine Trawl Survey, 1989–2010. The error bars represent the standard errors about the estimates.

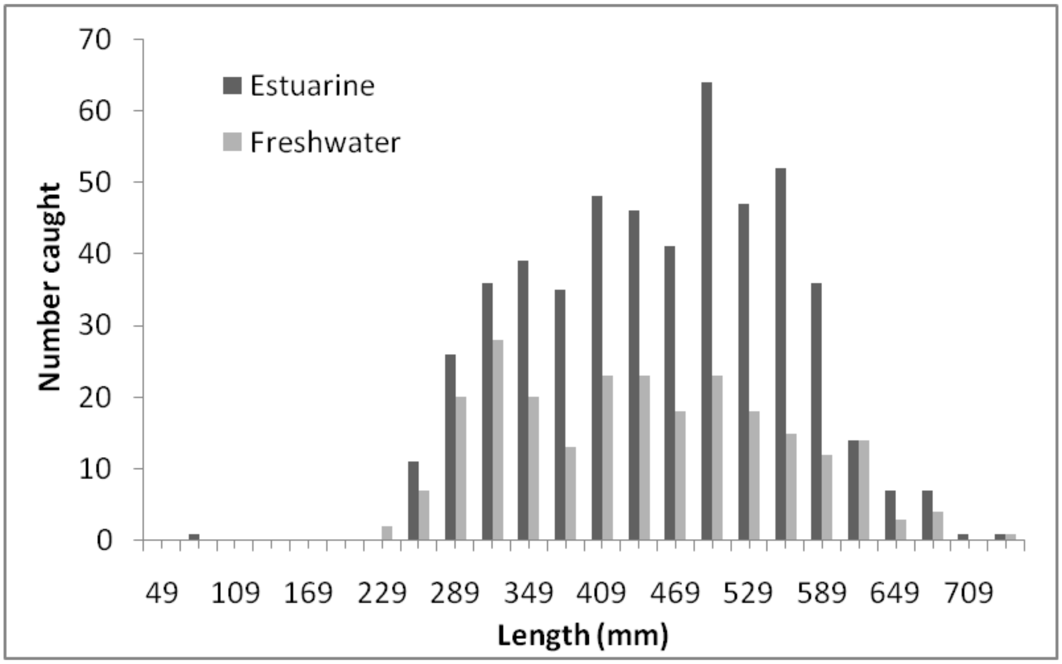

Figure 5.72. Length distribution of eels sampled in estuarine and freshwater habitats of Northwest Pamlico Sound and Lake Mattamuskeet, North Carolina, 2002–2003 (Cudney 2004).

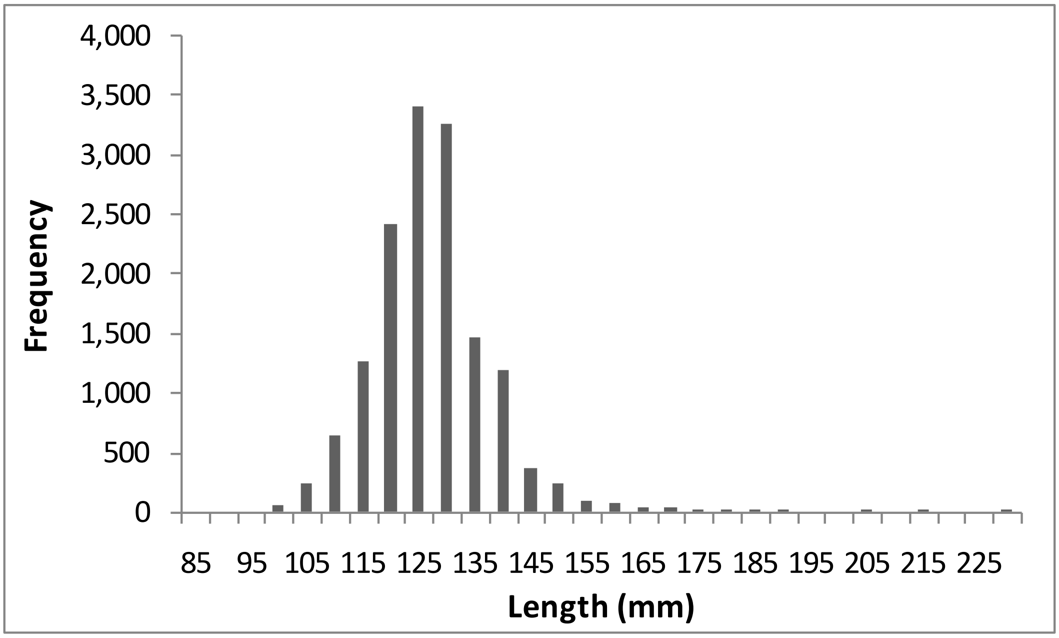

Figure 5.73. Length frequency of American eel caught in eel traps at the Roanoke River Dam, North Carolina, 2005–2009 (Graham, Dominion Power, pers. comm.).

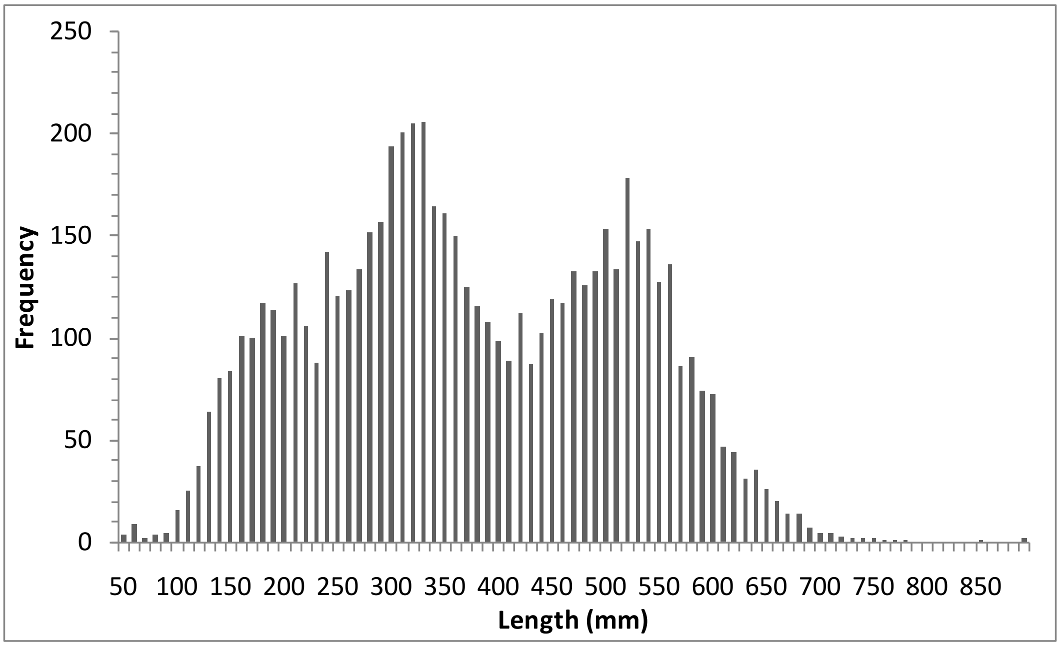

Figure 5.74. Length distribution of eel collected by the SC Electrofishing Survey, 2001–2010.

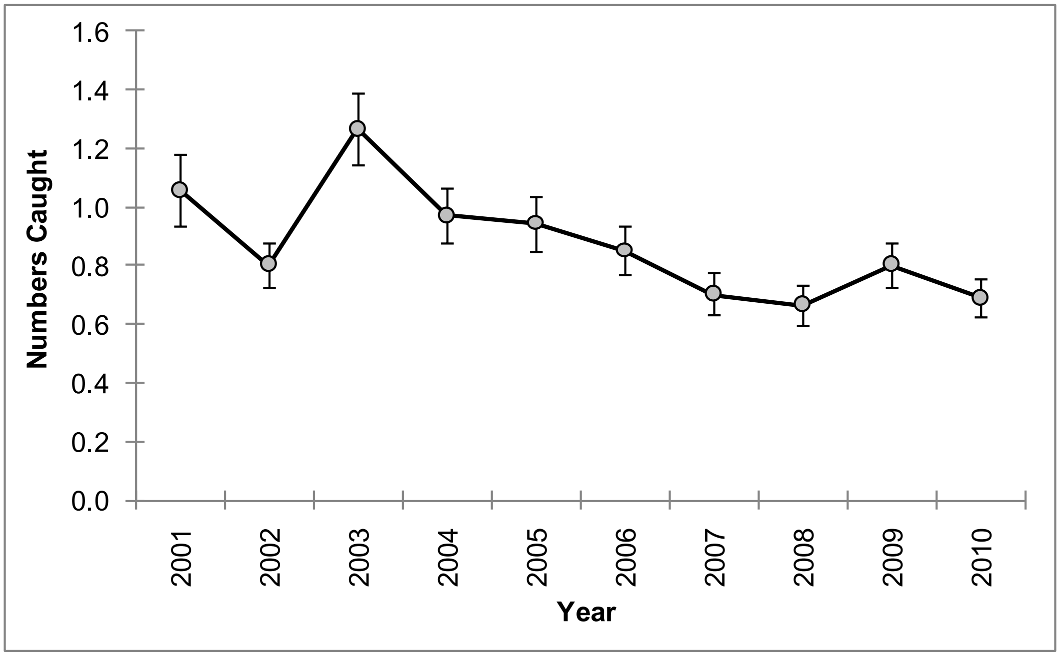

Figure 5.75. GLM-standardized index of abundance for American eels caught by the SC Electrofishing Survey, 2001–2010. The error bars represent the standard errors about the estimates.

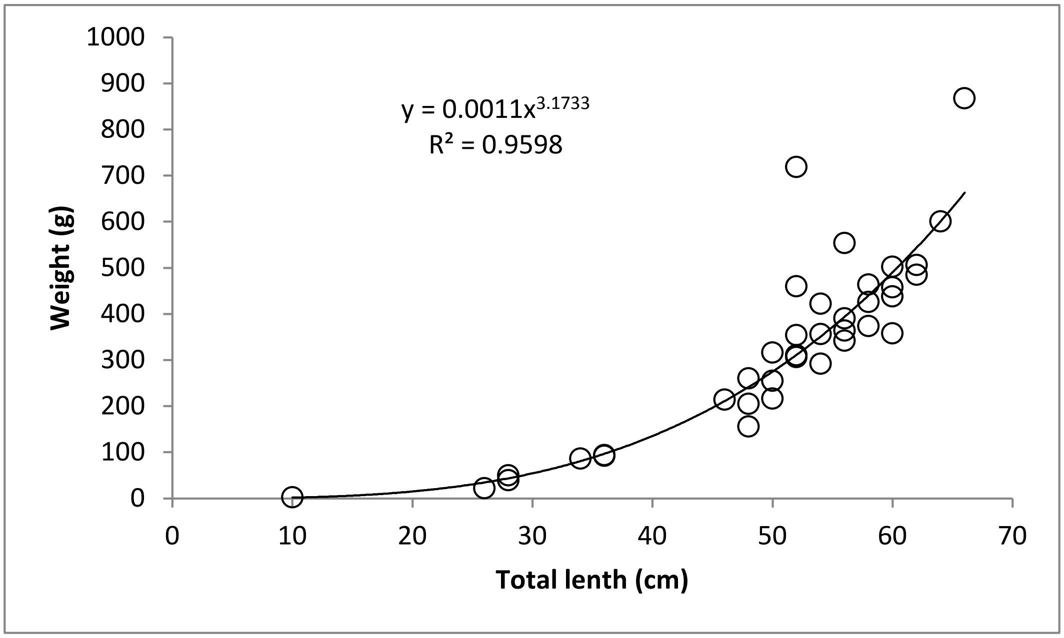

Figure 5.76. American eel weight-length relationship for the Suwannee River, Florida, 1996–2008. Years were combined (n = 38).

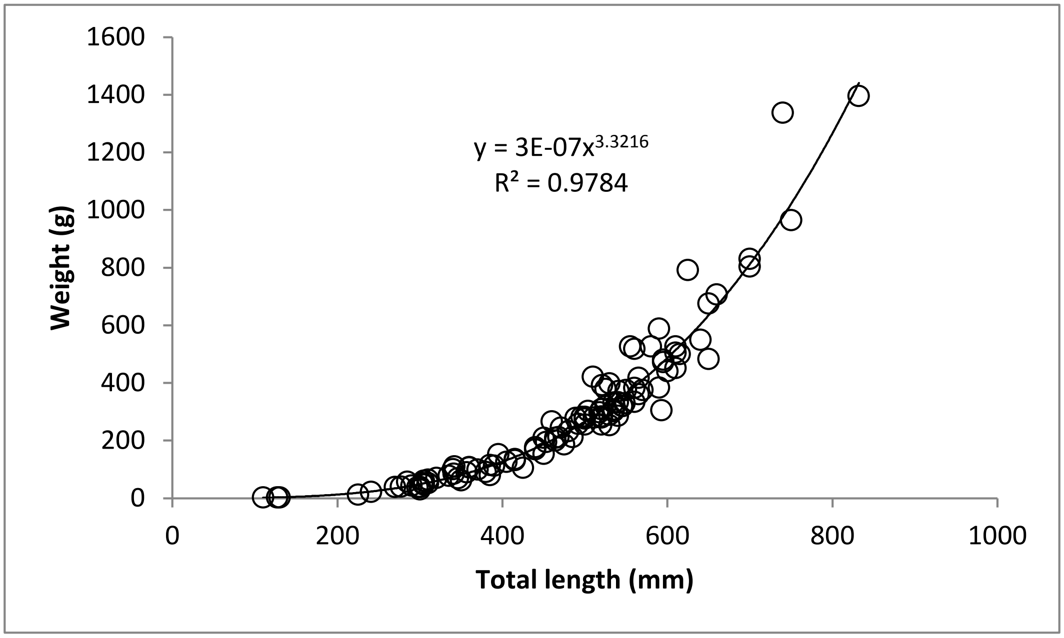

Figure 5.77. Weight-length relationship for American eels in the FL FWCC lake and marsh electrofishing survey.

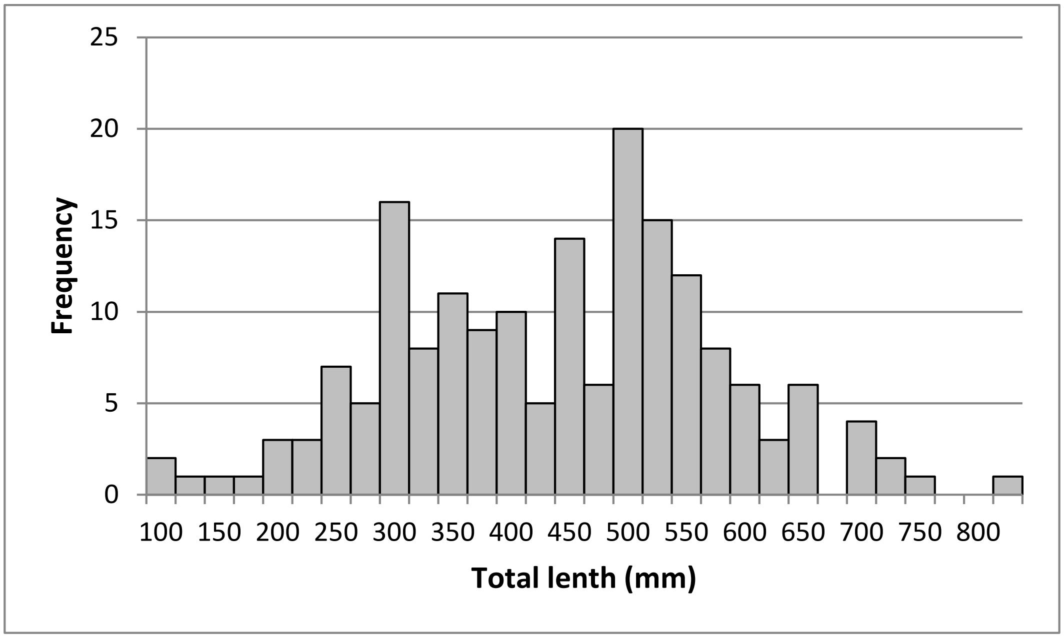

Figure 5.78. Length frequency distribution of American eels in the FL FWCC lake and marsh electrofishing survey. Mean total length was 472 mm.

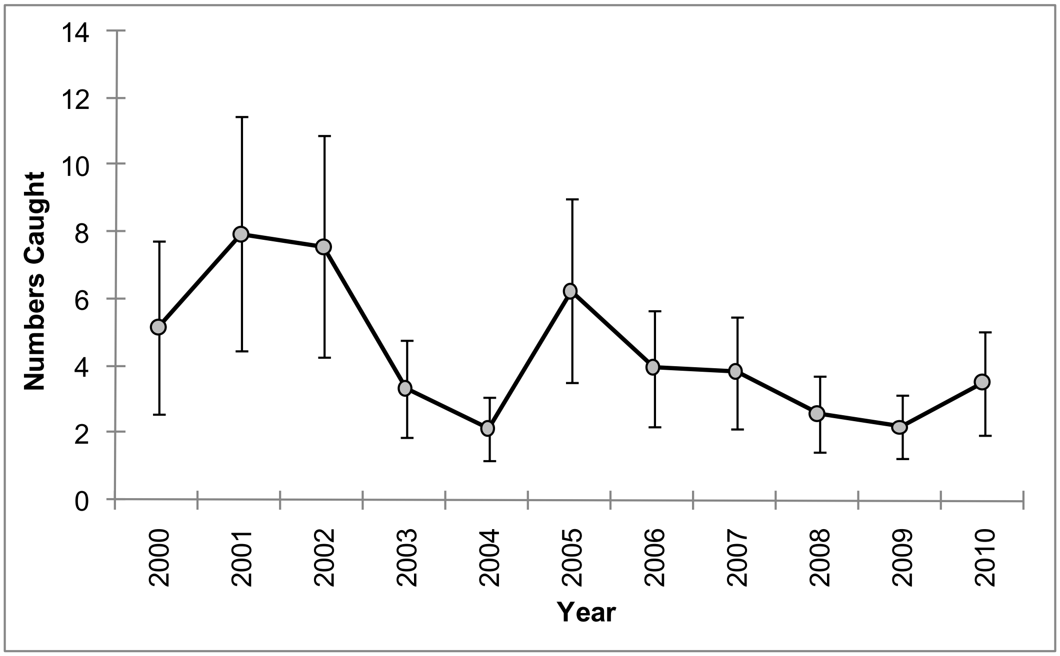

Figure 6.1. GLM-standardized, short-term index of abundance for YOY American eels along the Atlantic Coast, 2000–2010. The error bars represent the standard errors about the estimates.

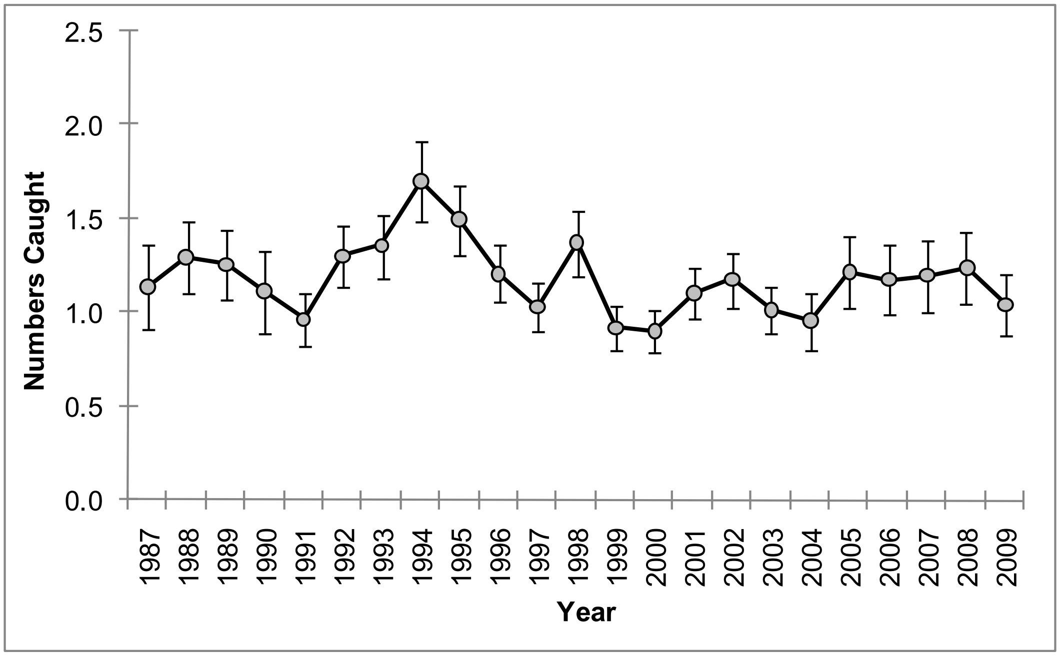

Figure 6.2. GLM-standardized, long-term index of abundance for YOY American eels along the Atlantic Coast, 1987–2009. The error bars represent the standard errors about the estimates.

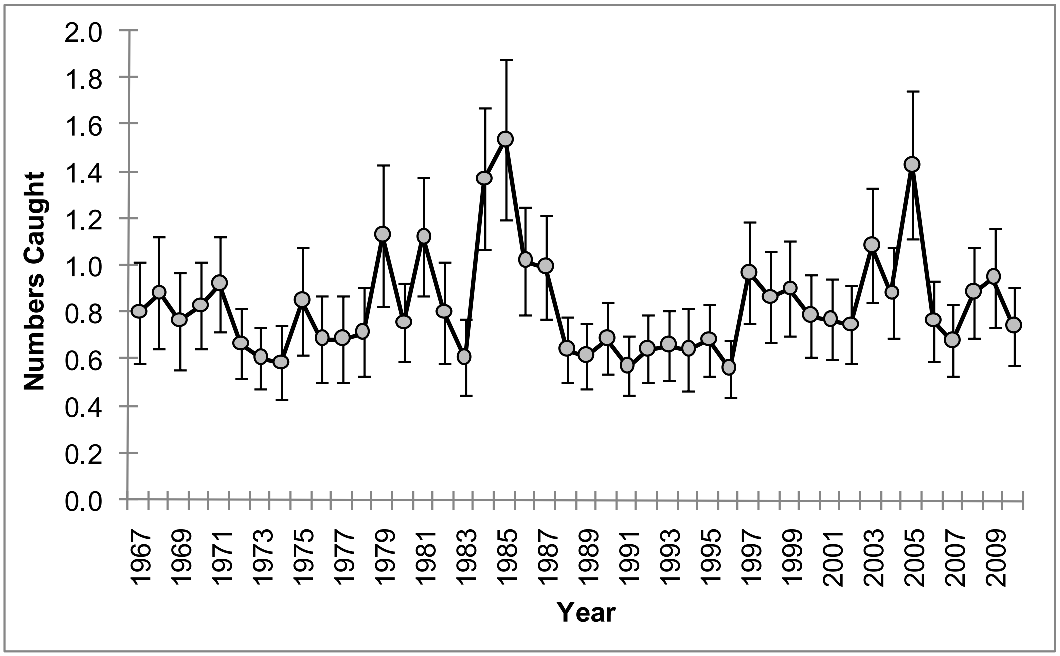

Figure 6.3. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1967–2010 (40-plus-year index). The error bars represent the standard errors about the estimates.

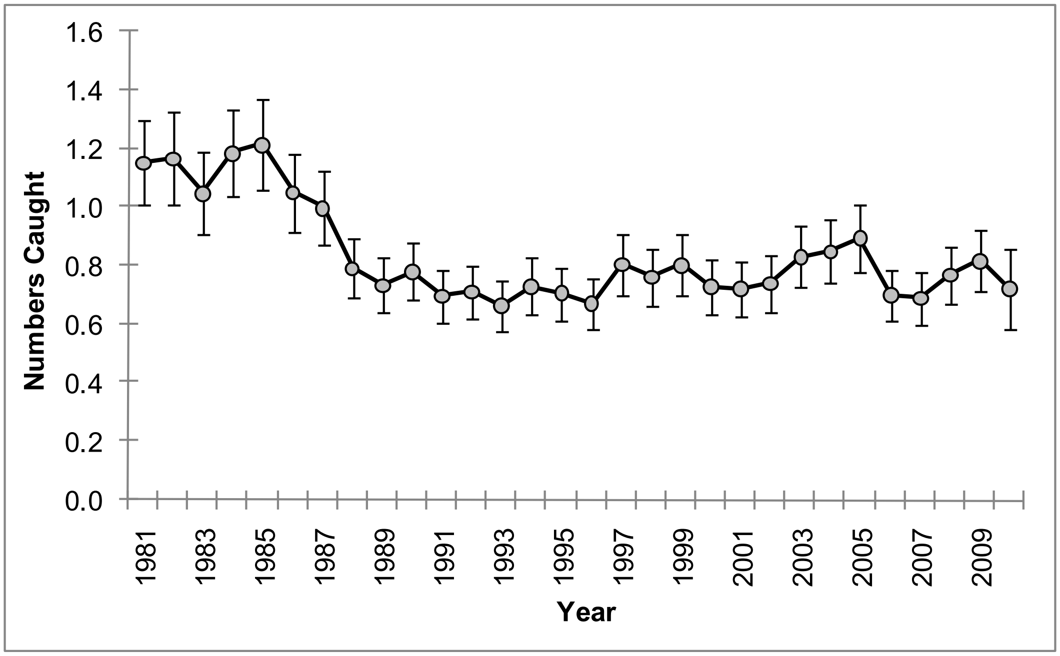

Figure 6.4. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1981–2010 (30-year index). The error bars represent the standard errors about the estimates.

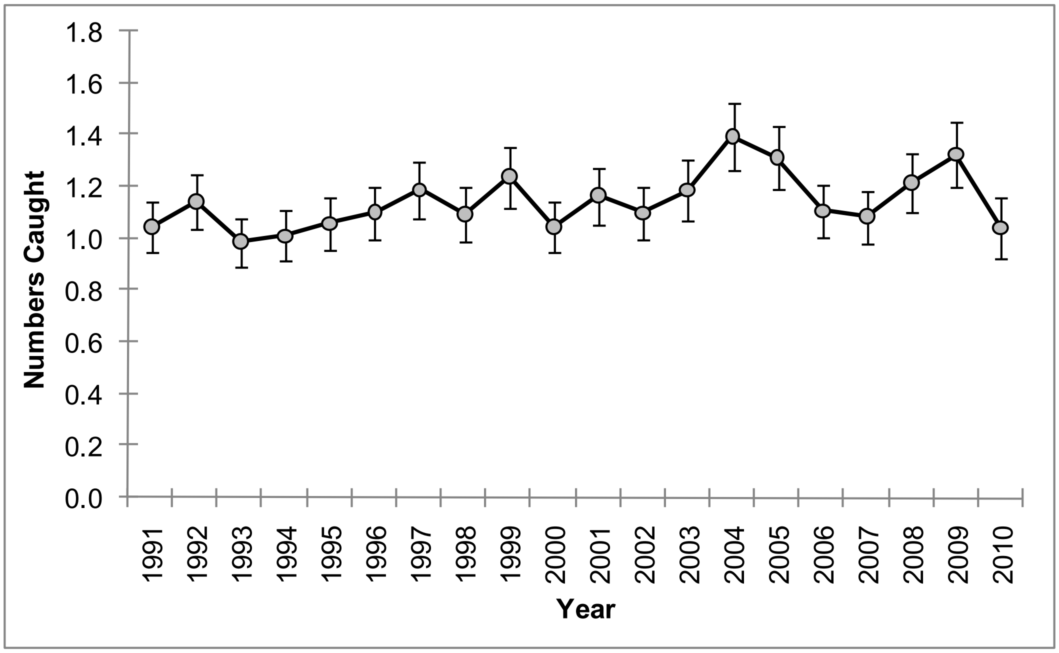

Figure 6.5. GLM-standardized index of abundance for yellow-phase American eels along the Atlantic Coast, 1991–2010 (20-year index). The error bars represent the standard errors about the estimates.

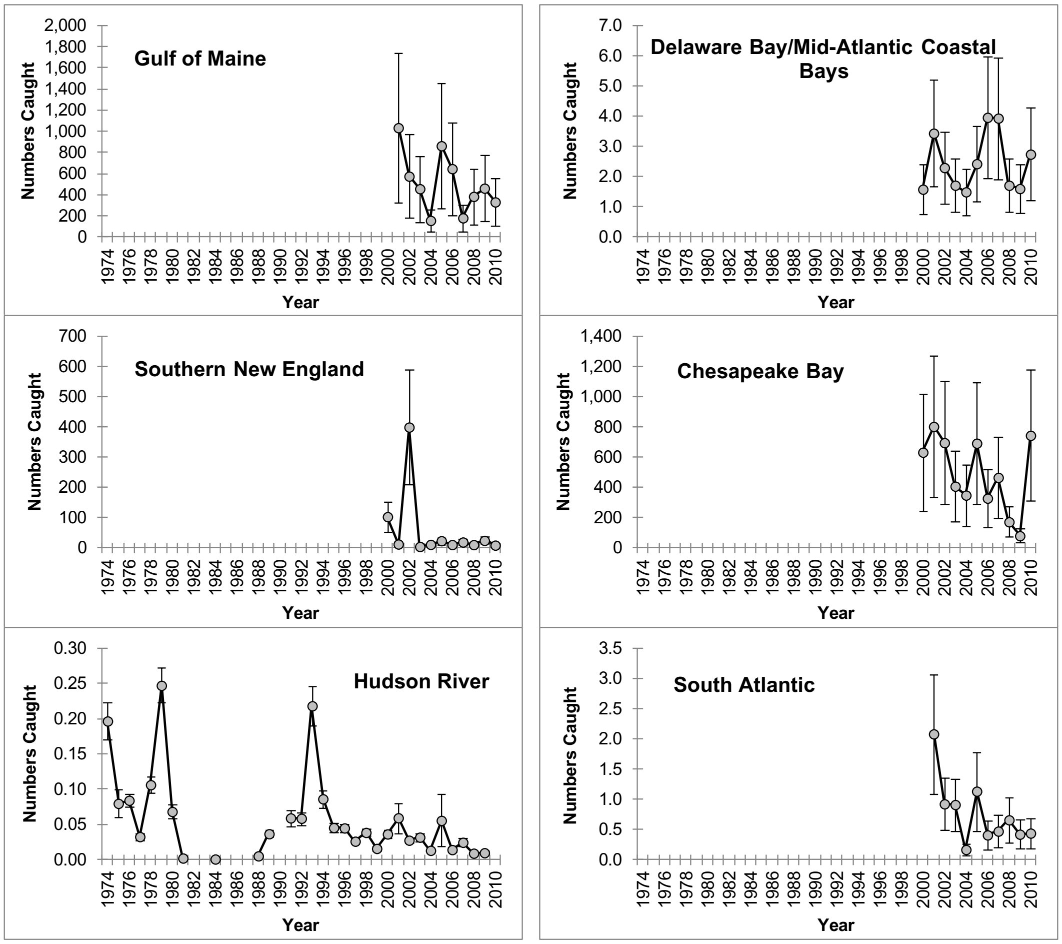

Figure 6.6. Regional indices of YOY abundance for American eels. The error bars represent the standard errors about the estimates.

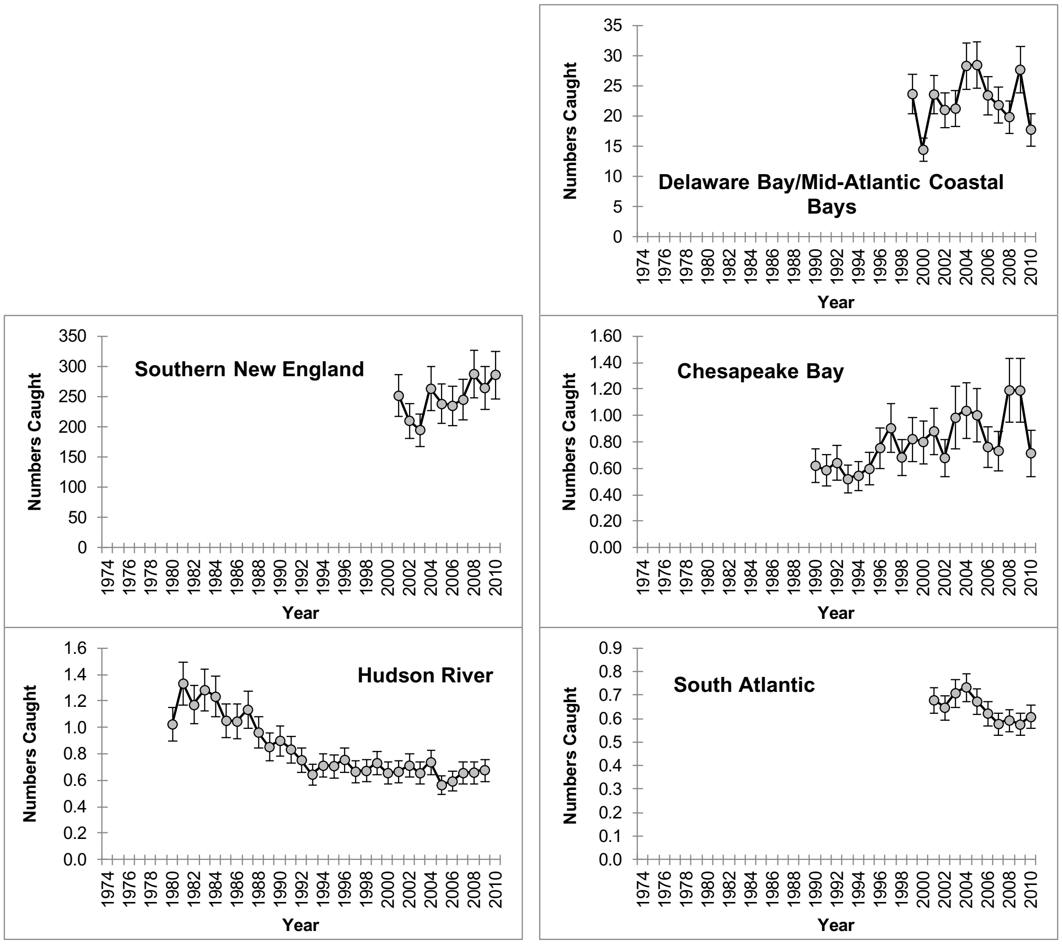

Figure 6.7. Regional indices of yellow-stage abundance for American eels. The error bars represent the standard errors about the estimates.

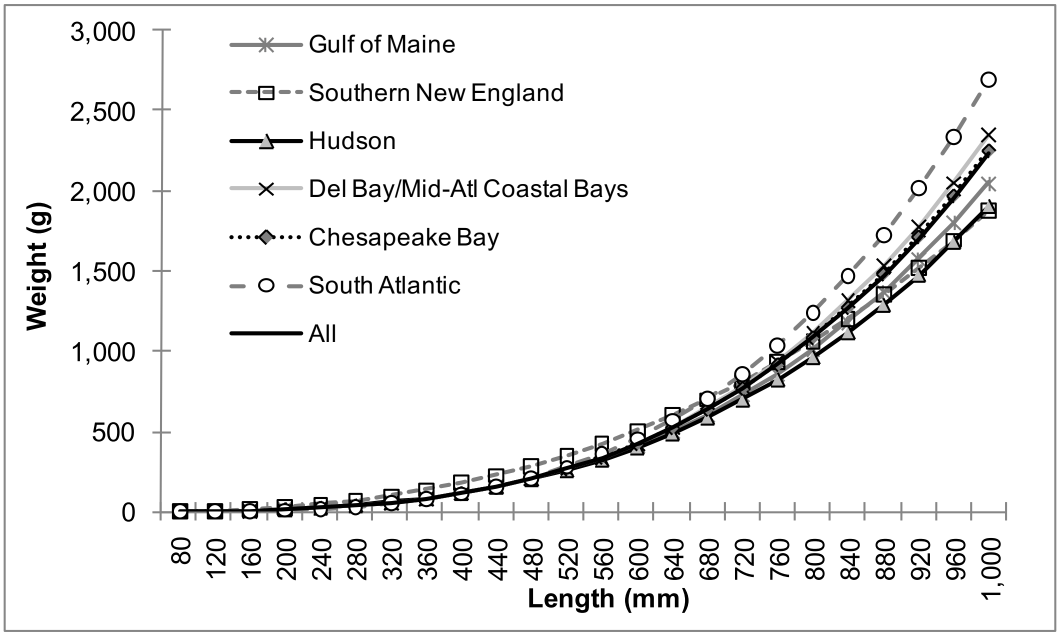

Figure 6.8. Predicted length-weight relation for American eel based on available data, by region.

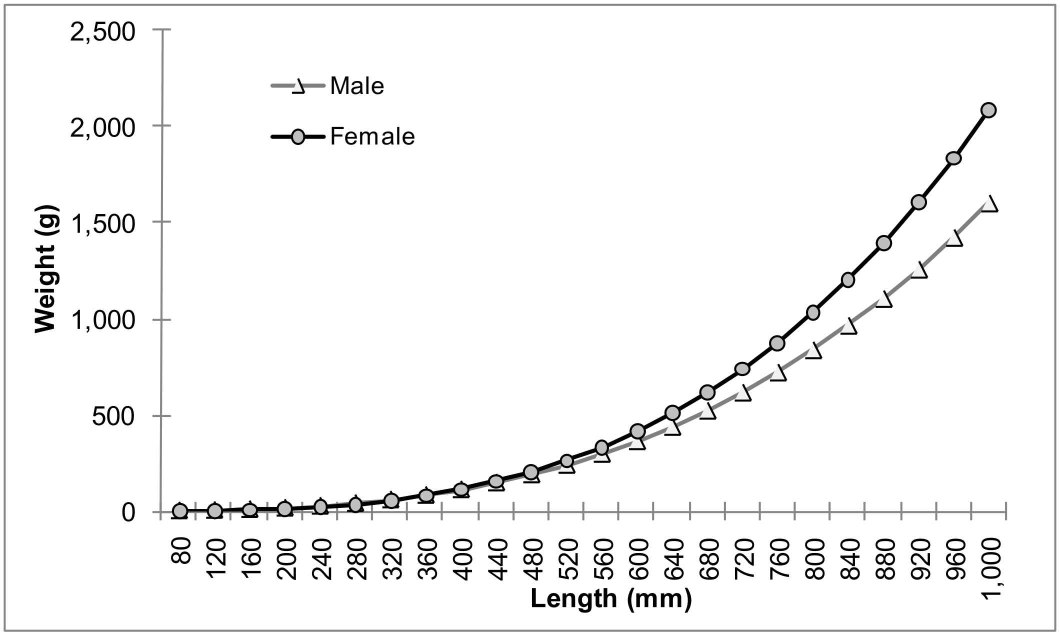

Figure 6.9. Predicted length-weight relation for American eel based on available data, by sex.

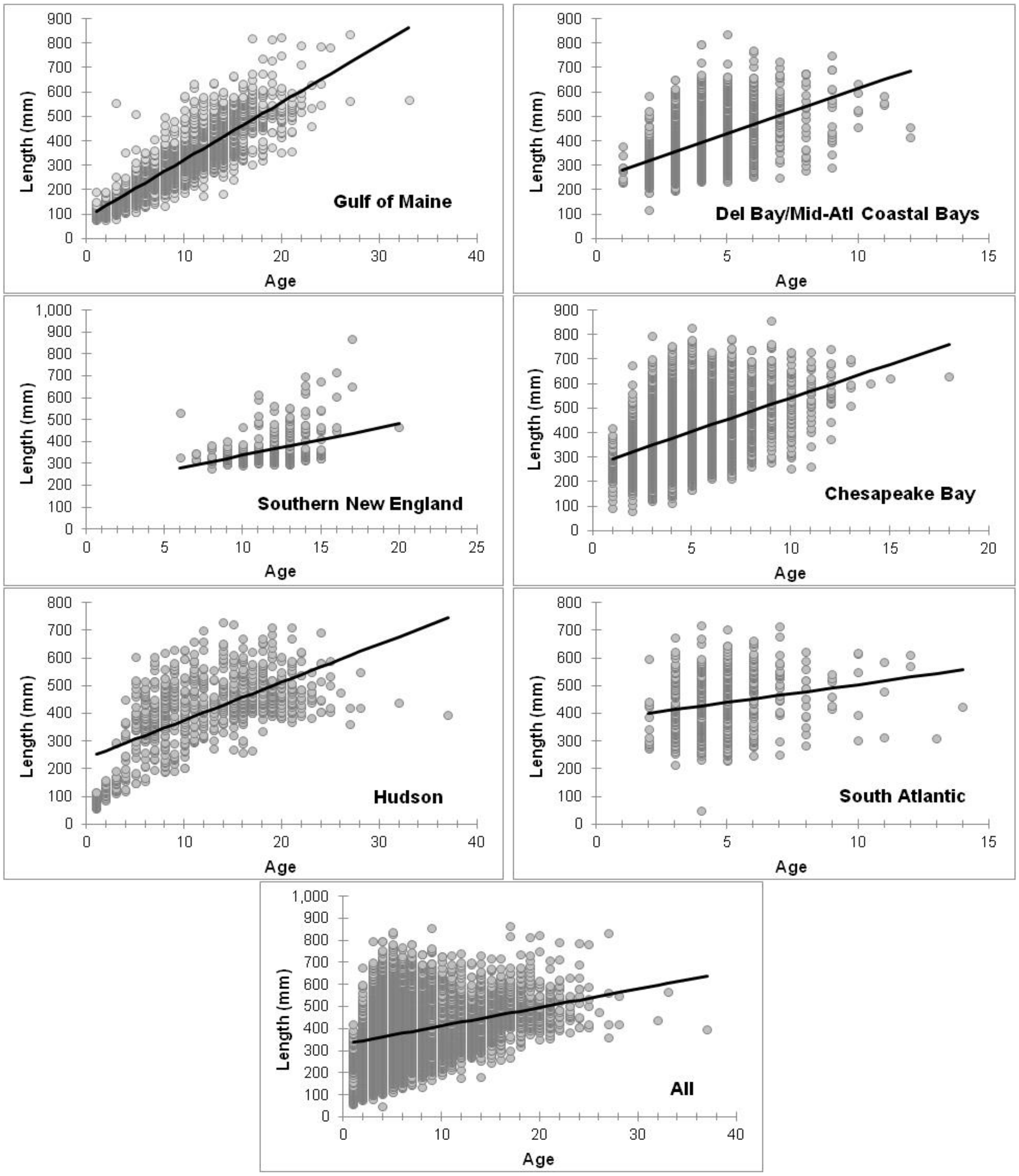

Figure 6.10. Observed age-length data (circles) and predicted linear age-length relation (solid line) for American eel based on available data, by region and for all data pooled.

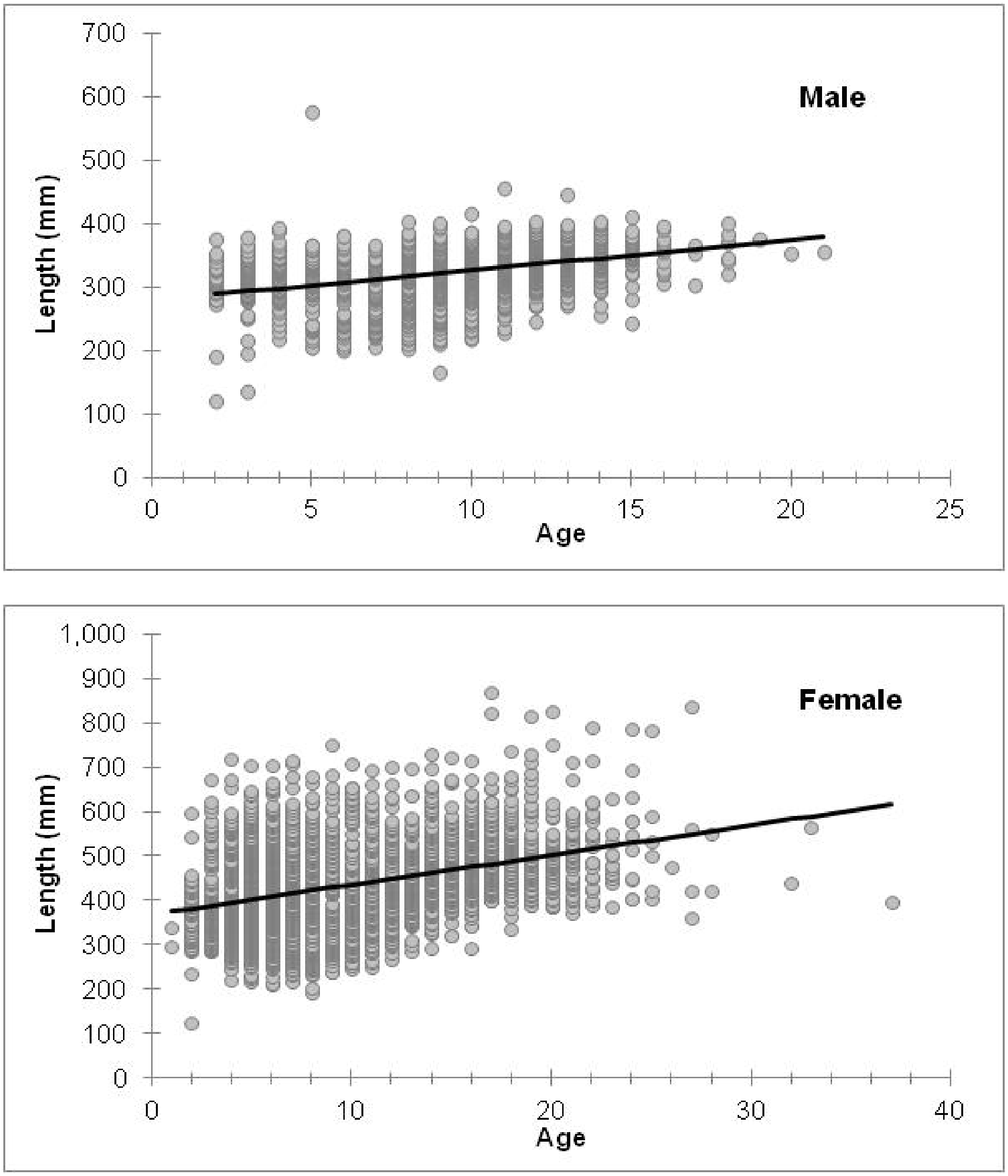

Figure 6.11. Observed age-length data (circles) and predicted linear age-length relation (solid line) for American eel based on available data, by sex.

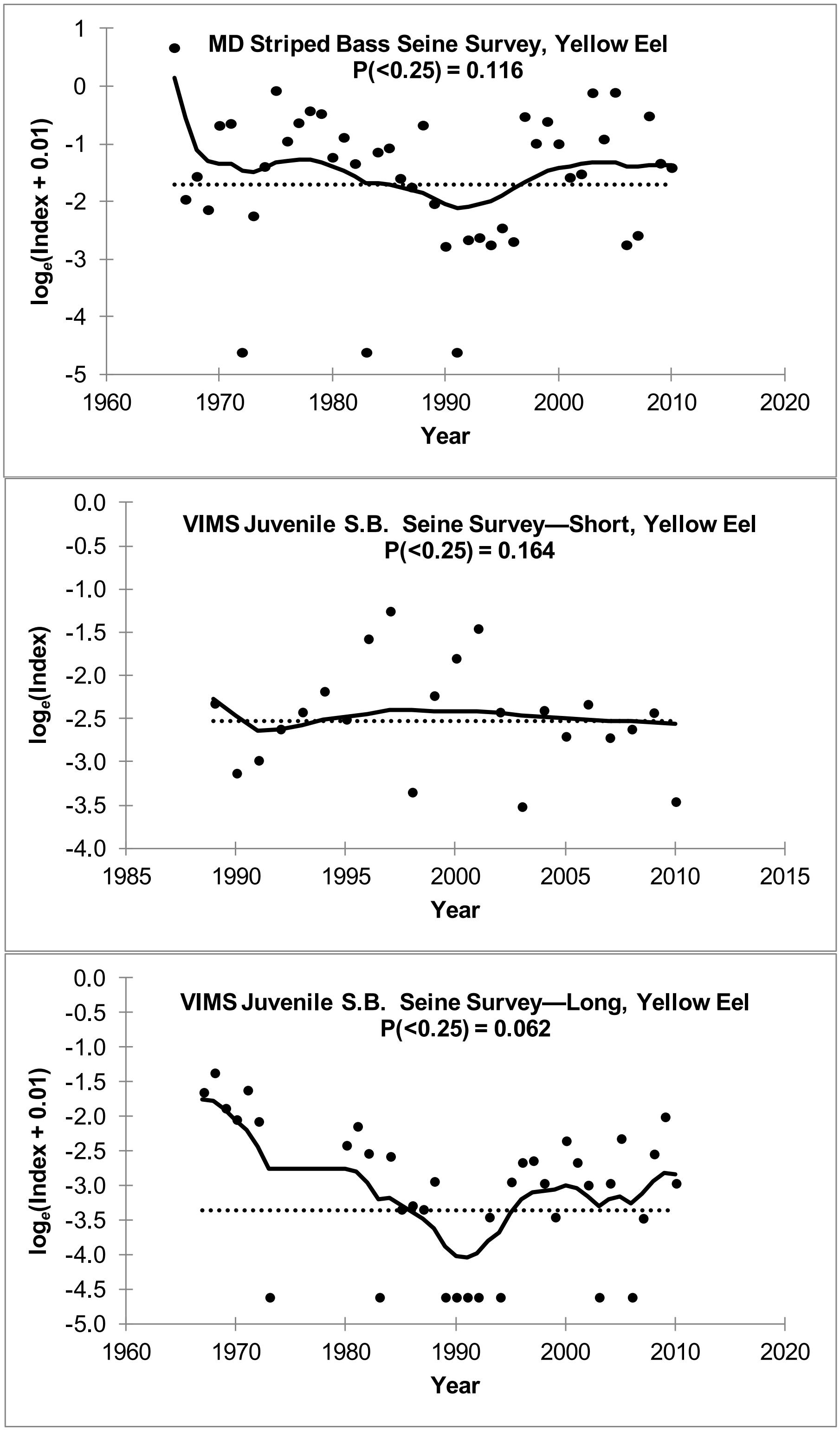

Figure 6.12. ARIMA model fits to American eel surveys from the Chesapeake Bay region. The dotted line represents the 25th percentile of the fitted values.

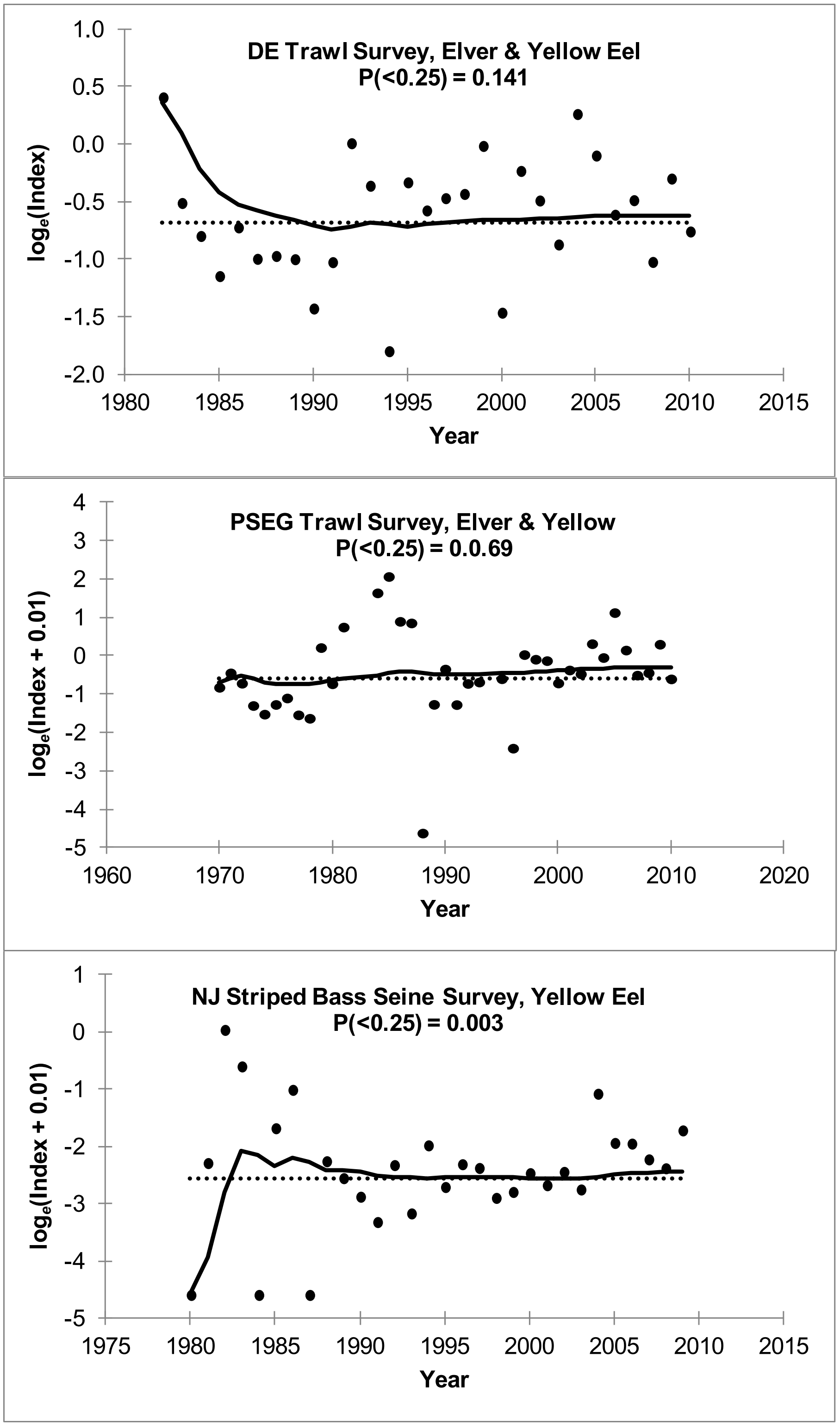

Figure 6.13. ARIMA model fits to American eel surveys from the Delaware Bay/Mid-Atlantic Coastal Bays region. The dotted line represents the 25th percentile of the fitted values.

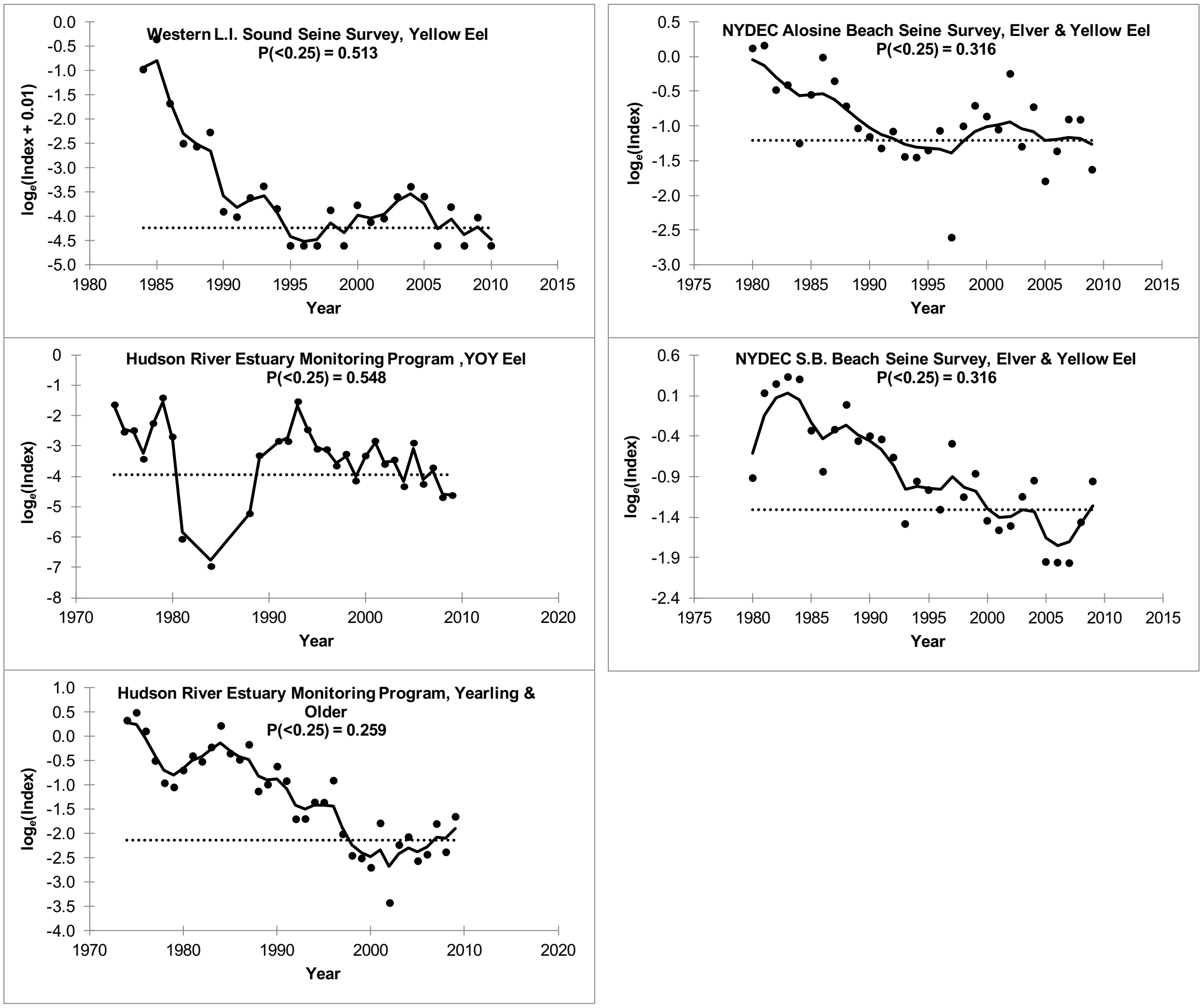

Figure 6.14. ARIMA model fits to American eel surveys from the Hudson River region. The dotted line represents the 25th percentile of the fitted values.

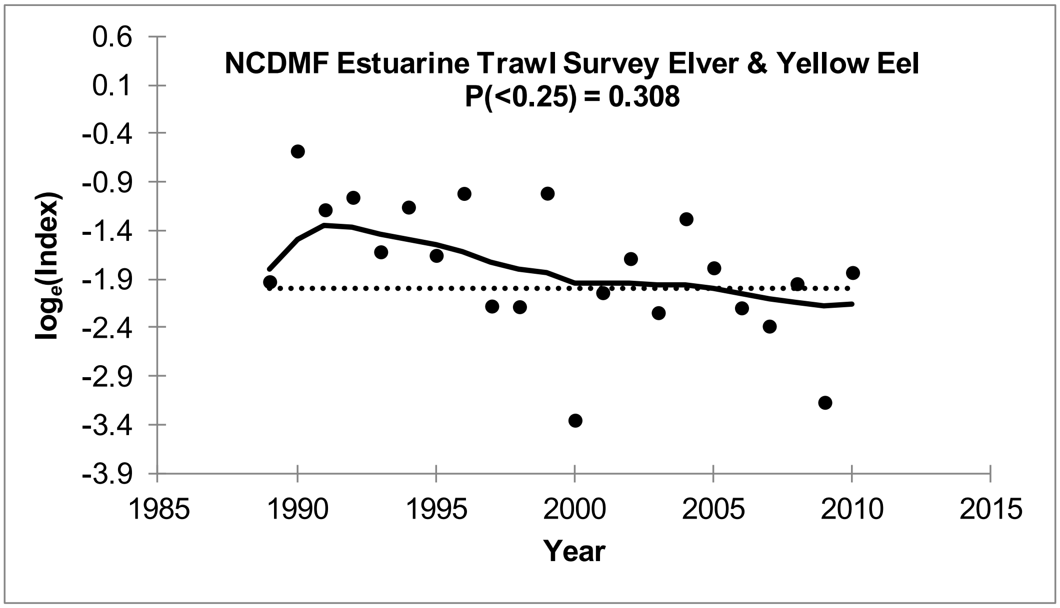

Figure 6.15. ARIMA model fits to American eel survey from the South Atlantic region. The dotted line represents the 25th percentile of the fitted values.

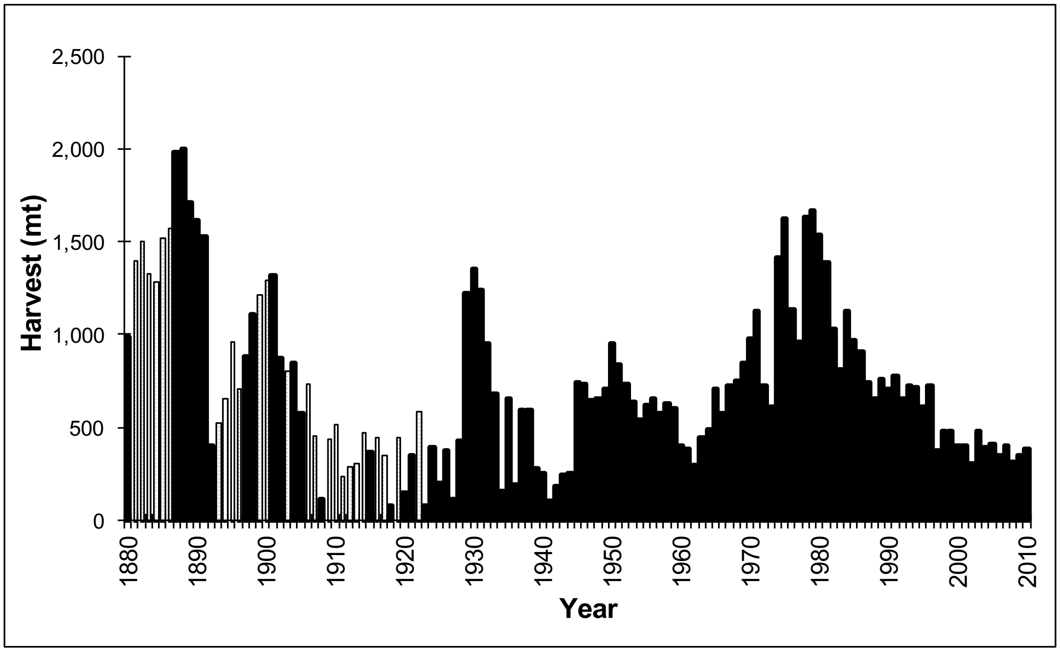

Figure 6.16. U.S. harvest of American eels used in DB-SRA. Light-colored bars indicate years for which harvest was reconstructed.

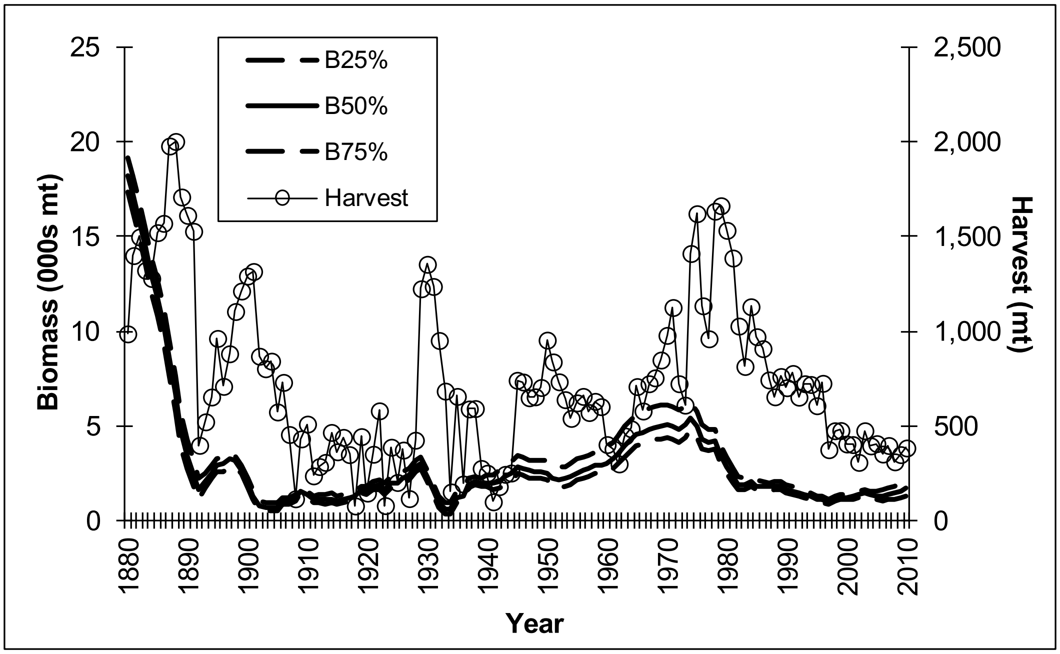

Figure 6.17. Estimated exploitable eel biomass from the DB-SRA single *M* stanza model.

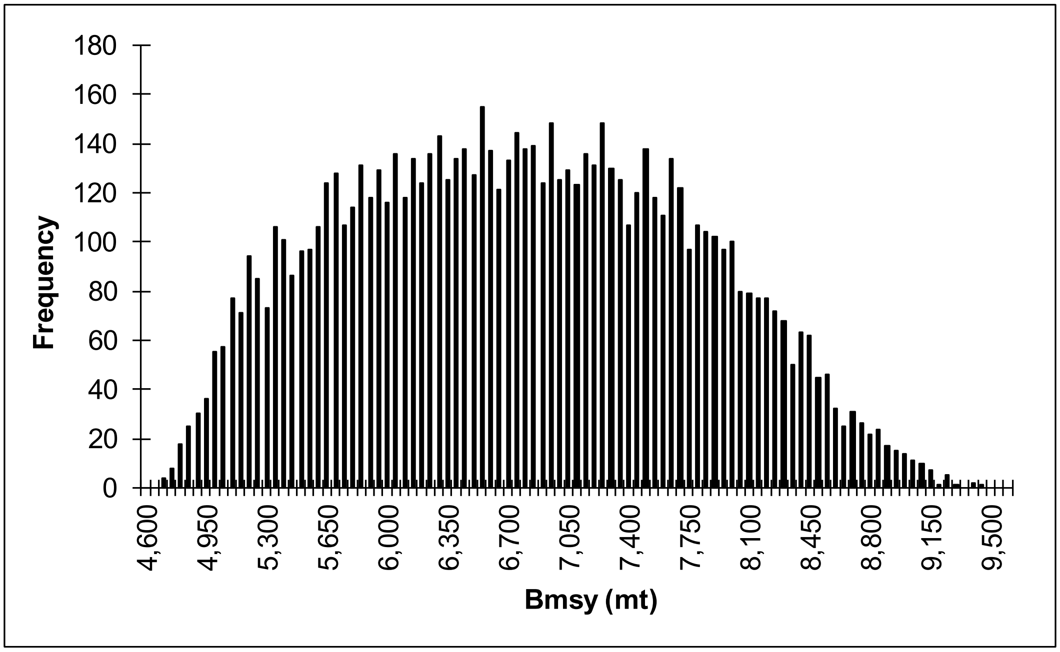

Figure 6.18. Distribution of estimated B_{MSY} from the DB-SRA single *M* stanza model.

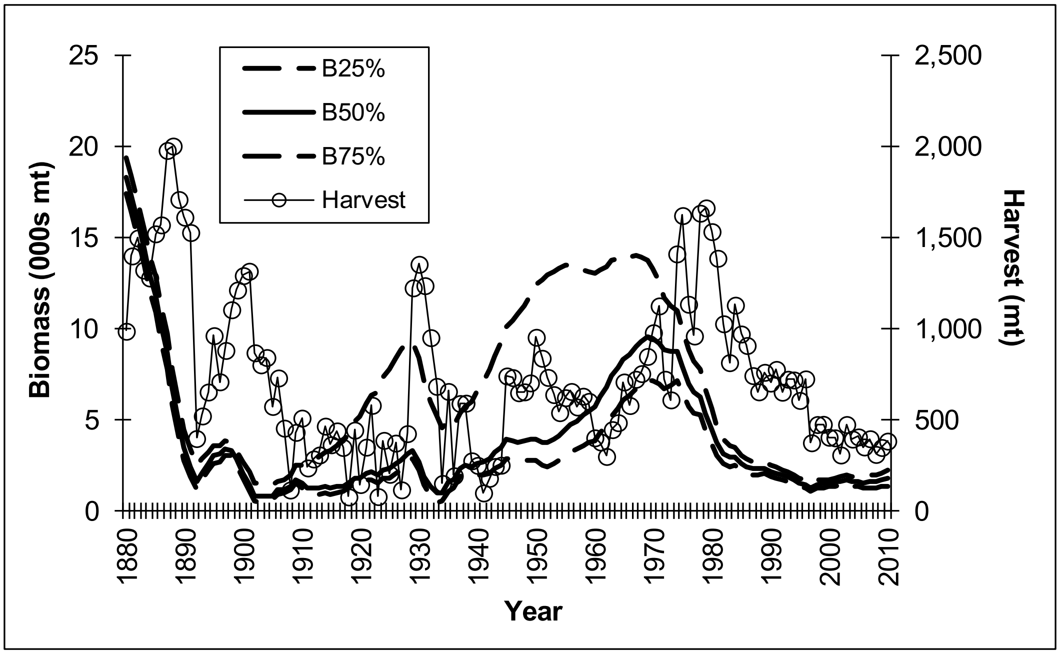

Figure 6.19. Estimated exploitable eel biomass from the DB-SRA double *M* stanza model.

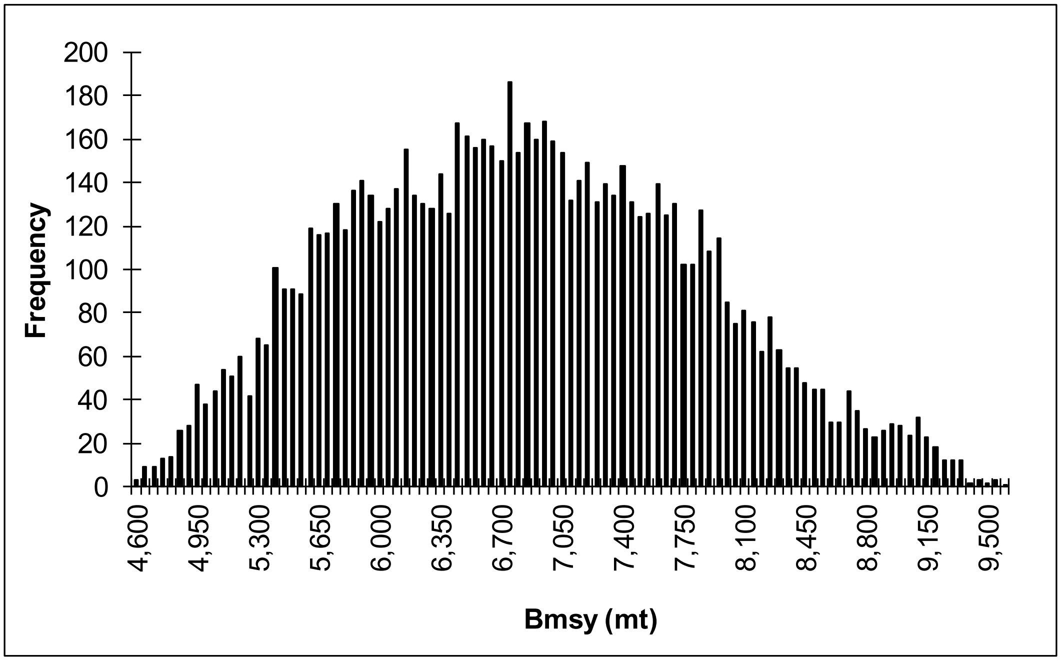

Figure 6.20. Distribution of estimated B_{MSY} from the DB-SRA double *M* stanza model.

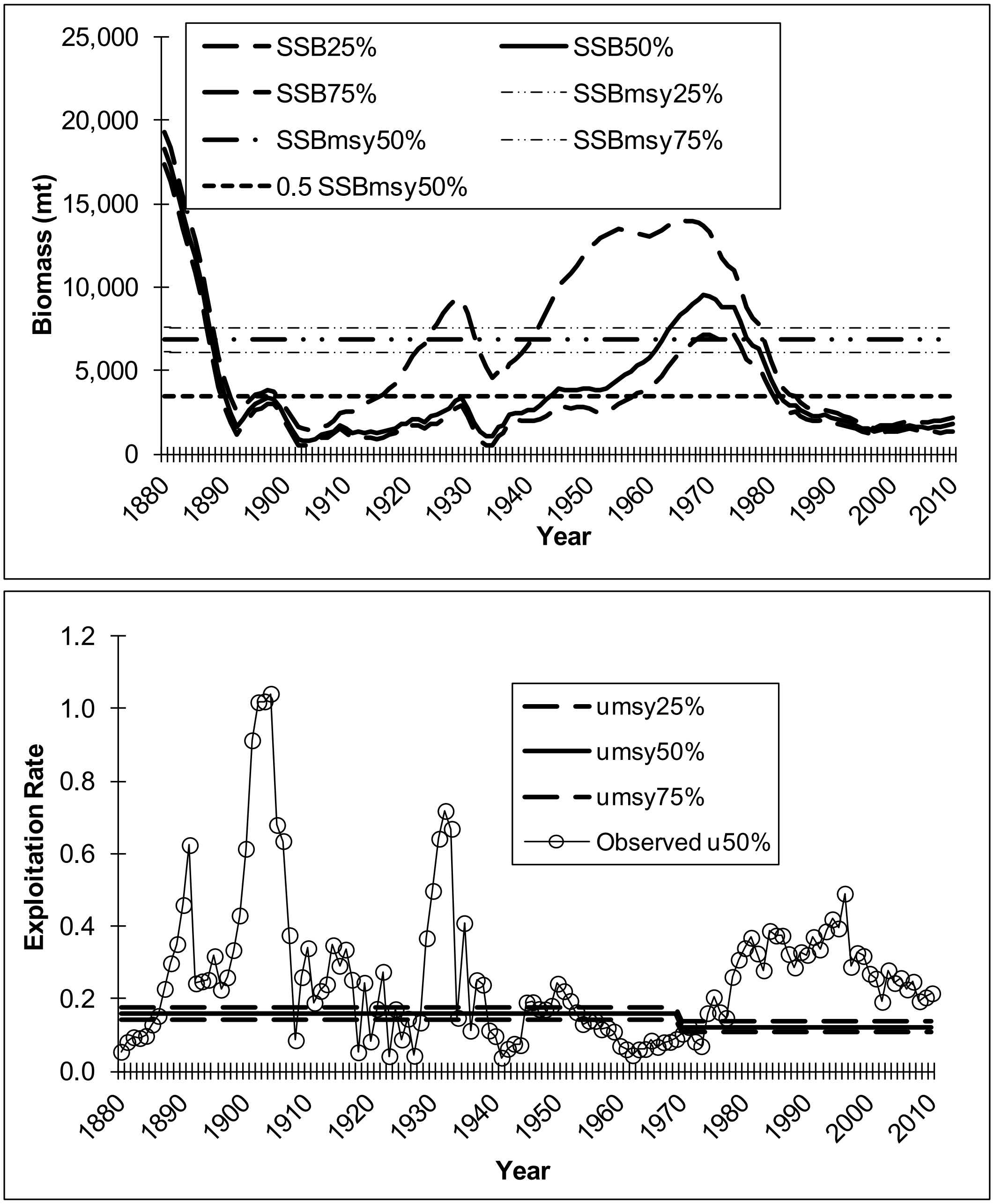

Figure 6.21. Stock status for U.S. American eel population based on the DB-SRA double *M* stanza model. Biomass vs B_{MSY} (upper graph) and annual exploitation (based on median biomass; lower graph) vs u_{MSY}.

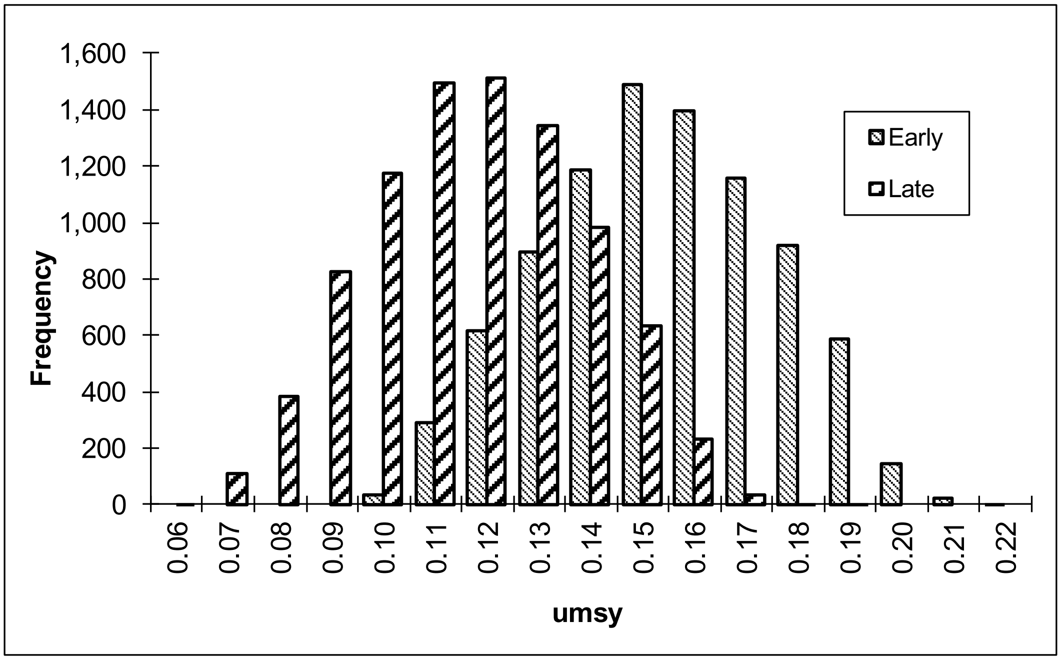

Figure 6.22. Estimated distribution of u_{MSY} from DB-SRA double *M* stanza model.

APPENDIX 1A. Summary of data sources included in assessment[15].

Region	State	Data Source	Data Type	Description	Location	Years	Method	Stage	Index	Bio Data
Gulf of Maine	ME	UMass	FI	Oliveira study	Maine rivers	1996–1998	Fyke, Weir, Electrofishing	S/Y		X
	ME	MEDMR	FI	Fort Halifax Dam	Sebasticook River	1999–2008	Dip net, Ladder	Y		X
	MA	MADMF	FD-comm	MA smelt bycatch	8 coastal rivers	2005–2011	Fyke net	Y		X
Southern New England	RI	UMass	FI	Oliveira study	Annaquatucket River	1990–1991	Fyke net	S		X
	CT	CTDEP	FI	CT electrofishing survey	Farmill River	2001–2009	Electrofishing	Y	X	
	NY	NYDEC	FI	Western Long Island Sound Survey	LIS	1984–2010	Seine	Y	X	
Hudson River	NY	NYDEC	FI	Hudson River Estuary Monitoring Program	Hudson River	1974–2009	Epibenthic sled and Tucker trawl	E/Y	X	
	NY	NYDEC	FI	Alosine survey	Hudson River	1980–2009	Beach seine	E/Y	X	
	NY	NYDEC	FI	Striped bass survey	Hudson River	1980–2009	Beach seine	E/Y	X	
	NJ	NJDEP	FD-comm	Commercial sampling	Hudson River	2008	Pot	Y		X
	NY	UMCES CBL	FI	Morrison study	Hudson River	1997–1999	Pot	Y		X
	NY	SUNY ESF	FI	Machut study	Hudson River	2003–2004	Electrofishing	E/Y		X
Delaware Bay/Mid-Atl Coastal Bays	PA	PAFBC	FI	PA Area 6 electrofishing	Non-tidal Delaware River	1999–2010	Electrofishing	E/Y	X	
	NY	TNC	FD-comm	Neversink tagging study	Neversink River	2008	Tagging	Y		X
	NY	TNC	FI	Neversink Electroshocking	Neversink River & tribs	2006–2008	Electrofishing	Y		X
	NJ	NJDFG	FI	NJ striped bass seine	Tidal DE River	1985–2009	Beach seine	Y	X	
	NJ	NJDFW	FD-comm	Commercial sampling	Statewide	2006–2010	Pot	Y		X
	DE/NJ	PSEG	FI	PSEG impingement	DE Bay	1984–2009	Impingement	Y		X
	DE/NJ	PSEG	FI	PSEG trawl studies	DE Bay	1970–2009	Trawl	Y	X	
	DE	DEDFW	FI	DE juvenile trawl survey	Delaware River	1982–2010	Trawl	Y	X	X
	DE	DEDFW	FI	DE adult trawl survey	Delaware River	1990–2009	Trawl	Y		X
	DE	DEDFW	FI	DE tidal tribs survey	Delaware River	1996–2005	Trawl	E/Y		X
	DE	UDE	FI	Fox silver eel study	Indian River	2002–2003	Fyke net	S		X
	DE	DEDFW	FD-comm	Commercial sampling	Statewide	2000–2010	Pot	Y		X
	MD	MDDNR	FD-comm	Fisheries-Eel Project	Assawoman Bay	2001–2002	Pot	Y		X
	MD	MDDNR	FI	Turville Creek Survey	Turville Creek	2009–2010	Pot	Y		X

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[15] This table does not include the ASMFC-mandated annual YOY recruitment surveys.

APPENDIX 1A. *Continued.*

Region	State	Data Source	Data Type	Description	Location	Years	Method	Stage	Index	Bio Data
Chesapeake Bay	WV	USFWS	FI	Shenandoah River study	Shenandoah River	2003–2009	Ladder count	Y		X
	WV	USFWS	FI	Silver eel turbine mortality	Shenandoah River	2007–2010	Electrofishing/ tag	Y/S		X
	WV	USFWS	FI	Shenandoah River study	Shenandoah River	2003–2005	Ladder	Y		X
	WV	USFWS	FI	Parasite infection rates	Shenandoah River	2006–2008	Ladder	Y		X
	MD	MDDNR	FD-comm	Fisheries-Eel Project	Statewide	1997–2010	mostly pots	Y		X
	MD	MDDNR	FI	Fisheries-Eel Project	Statewide	1997–2010	mostly pots	All		X
	MD	USFWS	FI	Pot study	Susquehanna River	2005	Pot	Y		X
	MD	MDDNR	FI	Sassafrass River survey	Sassafrass River	1998–2010	Pot	Y		X
	MD	MDDNR	FI	Gravel Run	Corsica River	2006–2010	Trap at low head dam	S		
	MD	MDDNR	FI	Juvenile striped bass seine		1966–2010	Beach seine	Y	X	
	MD/VA	UMCES CBL	FI	Fenske study	Potomac River	2007	Pot	Y		X
	VA	VMRC	FD-comm	Sanpling	Statewide	1993–2008	mostly pots	Y		X
	VA	VDGIF	FI	Electrofishing	Statewide	1992–2010	Electrofishing	E/Y		X
	VA	Dominion Power	FI	Utilities study	North Anna River	1990–2009	Electrofishing	Y	X	X
	VA	VIMS	FI	Striped bass seine survey	Chesapeake Bay	1967–1973, 1980–2010	Seine	E/Y	X	
South Atlantic	VA	VDGIF	FI	Electrofishing	Nottoway River	2000–2010	Electrofishing	Y		X
	NC	Dominion Power	FI	Trap study	Roanoke River	2005–2009	Trap	E/Y		X
	NC	Dominion Power	FI	Pot study	Roanoke River	1999	Pot	Y		X
	NC	Dominion Power	FI	Electrofishing	Roanoke River	1999–2000	Electrofishing	Y		X
	NC	NCDMF	FI	NC Program 120	Statewide	1989–2010 (index), 1971–2010 (biodata)	Trawl	E/Y	X	X
	NC	ECU	FI	Cudney study	Lake Mattamuskeet	2002–2003	Pot	Y/S		X
	NC	NCDMF	FD	Hutchinson Study	Pamlico River	1996	Pot	Y		X
	SC	SCDNR	FI	SC red drum electrofishing	Multiple river systems	2001–2010	Electrofishing	Y	X	X
	FL	FMRI	FD-comm	Commercial sampling	St. Johns River	2001–2006	Pot	Y		X
	FL	FWRI	FI	Lake City Regional Office River survey	Suwannee River & others	1996–2008	Electrofishing	Y		X
	FL	FWRI	FI	Long Term Freshwater Fisheries Monitoring	12 lakes & marshes	2007–2008	Electrofishing	Y		X

APPENDIX 1B. Summary of reviewed data sources deemed inadequate for assessment[16].

Region	State	Data Source	Data Type	Description	Location	Years Available	Collection Method	Stage/Length Range	Justification for exclusion
GOM	ME	MEDMF	FI	Fort Halifax Dam	Sebasticook River	2000–2008	Ladder census	Yellow	Short time series (1999 used dip nets and dam removed in 2008)
GOM	MA/NH	USFWS	FI	Dam survey	Merrimack River	2000–2001	Electrofishing	Yellow	Upstream of dam, low eel catch
GOM	MA	MADMF	FD	MA smelt bycatch	8 coastal rivers	2005–2011	Fyke net	Yellow	Short time series - consider in future
GOM/SNE	MA	MADMF	FI	Dam pasage monitoring	~6 river systems	variable (2004+)	Eel ramp, Sheldon trap	YOY and ages 1+	Short time series - consider in future
SNE	MA	UMass	FI	Paskamansett River study	Paskamansett River	2001–present	Fyke net	Silver	Eels not processed aged yet; also all males - consider in future
SNE	CT	CTDEP	FI	Dam pasage monitoring	~6 river systems	variable (2000+)	Passage counts	Elver/Yellow	Short time series - consider in future
SNE	RI	RIDEM	FI	Trawl survey	Narragansett Bay, Rhode Island, & Block Island Sound	1979–2010	Trawl	Elver/Yellow	Eels rarely caught
SNE	RI	RIDEM	FI	Juvenile finfish survey	Narragansett Bay	1986–2010	Beach seine	Elver/Yellow	Eels rarely caught
SNE	CT	USFWS	FI	Pot survey	Connecticut River	2009	Pot survey	Yellow	Short time series - consider in future
HR	NY	NYDEC	FI	Dam impingement monitoring	Hudson River	1973–2009	Dam sampling	Silver/Yellow	Collection inconsistent, incidental catch (not direct passage)
HR	NY	NYDEC	FI	HRE Monitoring Program (Fall Shoals and Beach Seine)	Hudson River	1970s–2008	Beach seine and trawl	Elver/Yellow	Data not available, consider in future
DER/DEBay	DE	DEDFW	FD	Fox study	St. Jones River	2005–2009	Pot/tagging	Yellow	Data not available, consider in future
DER/DEBay	NJ	NJDFG	FD	NJ fisheries sampling	Statewide	2006–2010	Pot, weir, fyke net	Yellow	Ages collected, not yet processed

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[16] This table does not include the ASMFC-mandated annual YOY recruitment surveys.

APPENDIX 1B. *Continued.*

Region	State	Data Source	Data Type	Description	Location	Years Available	Collection Method	Stage/Length Range	Justification for exclusion
DER/DEBay	NJ	NJDFG	FD	NJ fisheries sampling	Statewide	2006–2010	Pot, weir, fyke net	Yellow	Ages collected, not yet processed
DER/DEBay	PA	PAFBC	FI	Small mouth bass survey	Delaware & Lehigh rivers	2005–2009	Electrofishing	Elver	Short time series - consider in future
DER/DEBay	PA	PAFBC	FI	Passage sampling	Lehigh River	2005–2011	Passage counts	Elver/Yellow	Missing years, 2 sites sampled
DER/DEBay	PA	PAFBC	FI	Warmwater stream survey	Tribs of Delaware River	2007–2009	Electrofishing	Elver/Yellow	Missing years, 2 sites sampled
DER/DEBay	PA	Acad NS	FI	Electrofishing study	Tidal tribs of Delaware River	1995–2004	Electrofishing	Glass/Yellow	Locations and sampling methodology changed
DEBay/ ChesBay	DE/MD/ VA	UDE/CBL/ VIMS	FI	Ichtyoplankton survey	Delaware & Chesapeake Bay	2007–2009	Neuston net	Lepto/Glass	Short time series - consider in future
ChesBay	DC	DC Fisheries	FI	Electrofishing survey	Potomac & Anacostia rivers	2005–2009	Boat electrofishing	Elver/Yellow	Short time series - consider in future
ChesBay	DC	DC Fisheries	FI	Pot survey	Potomac & Anacostia rivers	2008–2009	Pot survey	Yellow	Short time series - consider in future
ChesBay	MD	MDDNR	FI	Juvenile fish & blue crab survey	Chesapeake Bay	1991–2010	Trawl	Yellow	Only one river, eels rarely caught
ChesBay	VA	VIMS	FI	Juvenile fish & blue crab survey	Chesapeake Bay	1980–2010	Trawl	Elver/Yellow	Indices of different length eels produced nearly identical trends
SA	NC	NCDMF	FD	NC Fishery	Statewide	1994–2009	Trip ticket	Yellow	Unkown # of trips per trip ticket due to penning
SA	NC	NCDMF	FD	NC Fishery	Statewide	2007–2009	Logbooks	Yellow	Short time series - consider in future
SA	NC	NCWRC	FI	Electrofishing survey	Roanoke, Tar, Neuse, & Cape Fear rivers	1979–2004	Electrofishing	Yellow	Unable to collect eels from vessel until 2010 - consider in future
SA	NC	NCWRC	FI	Yellow perch monitoring	Chowan River & tribs	2005–2010	Plankton net	Elver	Short time series, eel rarely caught
SA	NC	Dominion Power	FI	Pot and electrofishing CPUE	Roanoke River	2005–2009	Pot & electrofishing	Elver/Yellow	Short time series; consider in future

APPENDIX 1B. *Continued.*

Region	State	Data Source	Data Type	Description	Location	Years Available	Collection Method	Stage/Length Range	Justification for exclusion
SA	SC	SCDNR	FI	Juvenile marsh fish monitoring	Tidal marshes	1986–1994	Rotenone	Elver	Eels rarely caught
SA	SC	SCDNR	FD	Trammel net survey	Estuaries	1991–2010	Trammel net	Elver/Yellow	Eels rarely caught
SA	SC	SCDNR	FD	Biological sampling	SC waters	2000–2008	Trip level reports	Elver/Yellow	Very small fishery
SA	GA	GADNR	FI	Stream electrofishing study	Multiple streams statewide	1998–2002	Electrofishing	Unknown	No biological data, missing years, eels rarely caught
SA	FL	FWRI	FD	Biological sampling	4 lakes and St. Johns River	2006–2009	Trip ticket	Yellow	Short time series - consider in future
SA	FL	FWRI	FI	Seine & trawl surveys	5 bays/rivers	1989–2008	Mixed seine, trawl	Yellow	Mixed gears, eel rarely caught
SA	FL	FLGFWFC	FI	Blocknet & electrofishing studies	Multiple water bodies statewide	1973–2000	Block net & electrofishing studies	Yellow	Total weights/lengths, locations changed
SA	FL	FWRI	FI	Electrofishing	South Florida	2002–2004	Electrofishing	Yellow	Short time series, eel rarely caught
SA	FL	FWRI	FI	Electrofishing	South Florida	2000–2003	Electrofishing	Yellow	Short time series, eel rarely caught
SA	FL	FWRI	FI	Electrofishing	Gopher, Santa Fe, Withlacoochee Rivers	1996–2008	Electrofishing	Yellow	Inconsistent sampling locations

APPENDIX 2. Description of index standardization methodology.

1. Identify response variable. If data were collected using a standardized effort unit (e.g., electrofishing catch/15 min sampling event or catch/tow trawl surveys), model numbers caught (not CPUE). If concerned about changes in effort in the dataset, model catch as a function of effort and other covariates. If testing multiple models, make sure the response variables are the same.

2. Identify explanatory variables and associated data type (e.g., categorical, continuous):
   - Year will always be included as a categorical explanatory variable in all models.
   - Include a small subset of other appropriate variables using the literature and expert judgment if necessary. Do not include all potential variables - only ones that might be affecting **catchability (not abundance)** or you may standardize away the factors that actually affect trends in abundance.
   - Scatterplot each potential covariate…
   - If obvious breaks or groupings appear, (e.g., seasons, depth/habitat categories, etc.) make that a categorical variable. Otherwise, make it a continuous variable if no obvious breaks in the data. Always assign year as a categorical variables to estimate year effects. For all categorical variables, check to make sure you have adequate number of samples in each category or your model will blow up. Lump categories if necessary/meaningful. If not, categories with no samples should be eliminated (data points removed from dataset) because the model cannot provide estimates for that factor if there are no observations. If there are only a few observations in that category, try to run the model (if it blows up, you'll have to go back and remove it).
   - If two or more variables are highly (>0.9) or logically correlated, pick the one that makes the most sense biologically if possible; for example, don't include both temperature and dissolved oxygen, or latitude and river system. If desperate, include interaction terms (with anything but year) as an initial test if you're not sure how things will pan out, but don't include interaction terms in the final model (nearly impossible to interpret and calculate final year effect for index).
   - Check if any factor is orders of magnitude different from others and adjust accordingly (turn 1,000,000 into 1 "million" to be on scale with other measurements in model).

3. Plot histogram of number of animals caught. Determine if there is a large gap between # of zeros and next highest bar (e.g., determine if you tend to either catch either no animals or a lot of animals).

   - If so, use delta approach (R code from Erik Williams, NMFS Beaufort) which models pres-abs with binomial model and positive tows with a different distribution (usually lognormal or gamma).

   - Otherwise, proceed to other generalized/general linear models in next step.

4. If delta methods are not appropriate, identify what distributional assumptions might be. Plot catch rate vs. variance in catch rate aggregated by each categorical factor and compare pattern with figures from Punt et al. (2000). A linear relationship supports an overdispersed Poisson error model, and variance in catch rate proportional to the square of the average catch rate suggests the log-normal and gamma error models. The negative binomial error model implies that the variance in catch rate is a function of both the average catch rate and the square of the average catch rate. Choose from below depending on outcome of mean-variance inspection. Avoid transformations of your response variable or covariates.
    - If lognormal or gamma error models are implied, perform the gamma. If you must for some reason use the lognormal, model catch as Gaussian with log link to avoid transforming catch. If you must for some reason model CPUE, use $\log_e$[CPUE+min(value/2) ~.]
    - If Poisson error model is implied, run the basic Poisson model (implying data are probably not overdispersed) and compare with the zero-inflated Poisson using the Vuong test. (Note: you will not be able to compare zero-inflated models with other sub-models in step 6).
    - If the negative binomial error models are implied, run the basic negative binomial model and compare with the zero-inflated negative binomial using the Vuong test. (Note: you will not be able to compare zero-inflated models with other sub-models in step 6).
5. Select the appropriate canonical link function (relates mean of response variable to explanatory variables) for the model you’ve selected. Gamma – inverse. Poisson and negative binomial – log.
6. If all factors in the final model are not significant, run all sub-models and select best model as one with lowest AIC. If too many covariates are included for this to be practical, use stepwise selection of covariates (or better yet, reconsider what covariates you are including). You will not be able to do this for the zero-inflated models.
7. Evaluate goodness-of-fit.
    - Check for overdispersion; if is > 2 suggests overdispersion. NA for Poisson model.
    - Plot standardized residuals against fitted values; presence of pattern may suggest overdispersion, miss-specification of link function, missing covariate, outliers
8. If desired, perform back-transformation and include bias correction. Pull out mean year effects and SEs.

APPENDIX 3.   SLYME model report.

American Eel SLYME Model:

Report to the
ASMFC American Eel Technical Committee

July 2008

Prepared by:

ASMFC American Eel Stock Assessment Subcommittee

Laura M. Lee (VMRC), *Chair*
Jeffrey Brust (NJDEP)
Keith Whiteford (MDDNR)
John Clark (DNREC), *Technical Committee Chair*
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BACKGROUND

The ASMFC American Eel Management Board initiated the development of Draft Addendum II in January 2007 to propose measures that would facilitate escapement of silver eel as a means to improve American eel recruitment and abundance. The Management Board asked the Technical Committee (TC) and Advisory Panel (AP) to consider closed seasons, gear restrictions, size limits, or a combination of these measures to reduce the harvest of emigrating eels. The TC and AP were asked to comment on the draft addendum, though both groups felt more information was needed in order to evaluate the proposed options. The Management Board requested the Stock Assessment Subcommittee (SASC) quantify the potential benefits of a maximum size limit. The SASC proposed a life-table approach to examine the potential impact of a maximum size limit on the population's egg production. The TC supported this approach. In August 2007, the Management Board approved the use of the life-table model, known as SLYME, to aid in the evaluation of implementing a maximum size limit. In May 2008, Management Board asked the SASC to also consider various slot limits in their evaluations.

INTRODUCTION

Although the available data for American eel in the U.S. have not been sufficient to perform a reliable quantitative assessment of the population size or fishing mortality rates (ASMFC 2001, 2006), there has been evidence that the stock has declined and is at or near low levels (ASMFC 2000, 2001, 2006; USFWS 2007). The ASMFC American Eel Management Board initiated Draft Addendum II based on a concern about evidence of declines in abundance of the yellow eel life-stage of American eel. The primary management objective of this Draft Addendum was to propose measures to facilitate escapement of silver eels during or just prior to their spawning migration with the intent of halting any further declines in eel abundance. Given the proposed measures, the ASMFC American Eel TC agreed with the advice from the American Eel SASC that implementing a maximum size limit was a feasible way to increase silver eel escapement.

In the absence of estimates of stock size and exploitation rates, the SASC proposed the use of a per-recruit approach to evaluate the potential impacts of maximum size limits. The SLYME (**S**equential **L**ife-table and **Y**ield-per-recruit **M**odel for the American **E**el) model was initially developed by David Cairns (DFO Canada) for the August 2000 meeting of the ICES Working Group on Eels. The model was used to evaluate the effect of the Prince Edward Island American eel fishery on spawning escapement. The model has since undergone several revisions and was most recently updated in 2003. The SASC used a modified version of the deterministic SLYME model to investigate the effects of different maximum size limits on female spawner escapement and egg production.

Although a stock-recruitment relationship for American eel has not been quantified, it is believed that an increase in the number of silver eels that escape and are allowed to spawn will ultimately increase juvenile recruitment and future production. Imposing a maximum size limit will reduce exploitation of large eels, allowing the opportunity for more eels to mature and undertake their spawning migration. The current model shows the relative impact of varying fishing mortalities on egg production and the relative increases in egg production as a result of changing the maximum harvest size limits.

At the May 2008 Management Board meeting, the use of slot limits was suggested. Participants were interested in whether increasing the current minimum size limit (6.0 inches) would add to the potential benefits gained from a maximum size limit. The SASC evaluated the impact of various slot limits on egg production in addition to the evaluation of maximum size limits alone.

Minimum size regulations have been a key component of Canada's American eel management strategy for the past twenty five years in the Maritime Provinces. Canada's most recent American eel management plan went into effect in 2003. The goal of the plan is a 50% reduction in eel harvest to be achieved through minimum size regulations, seasonal closures, limited entry to the fishery and limits on gear spacing. The minimum size has been increased several times since 2003 and the 2008 minimum size limits range from approximately 12 inches (30 cm) in Newfoundland to approximately 21 inches (53 cm) in the Gulf of St. Lawrence drainages of Nova Scotia, New Brunswick, and Prince Edward Island. Canada's glass eel fishery is exempted from the minimum size regulations. Canada does not have a bait eel fishery, so the large minimum size does not have as negative an economic impact on the Canadian eel fishery as it would have on the U.S. eel fishery.

MODEL STRUCTURE AND DATA

A detailed description of the model equations and notation is available from members of the SASC.

The SLYME model describes effects of growth and mortality on the population by age class from the time glass eels arrive at the coast to the time adult eels deposit eggs during spawning.

Important assumptions of the model include:

- The portion of the population that resides in areas where American eels are exploited make some contribution to the spawning population.
- Under the current management regime, recruitment to the coast has been constant.
- All glass eel recruitment to the coast is instantaneous and occurs March 1.
- All glass eel fishing is instantaneous and occurs one day after glass eel arrival.
- All glass eels surviving the glass eel fishery join the segment of the population residing in continental waters.
- The yellow eel life stage is discrete, without immigration or emigration.
- Fishing for resident eels occurs year round and is concurrent with natural mortality.
- All eels greater than 400 mm (15.75 in) are considered females.
- Silver eel emigration is instantaneous and occurs on October 1.
- The silver eel fishery is instantaneous and occurs one day after emigration.
- The fishery for emigrating silver eels is geographically separate from the resident eel fishery.
- Spawning occurs February 27.
- Growth and mortality processes are density-independent.

Several researchers generously provided raw data collected from studies on American eel (Table 1). Inputs required by the model were primarily derived from these data. Access to the raw data enabled the SASC to combine or subset datasets based on appropriate stratification in order to provide a representative characterization of the stock under ASMFC management. When possible, data collected from systems that are known to be exploited were used. If a required parameter could not be estimated from the available data, a literature review was performed to solicit values for the model. Efforts were made to apply data that could be considered representative of the coast-wide stock, though many of literature studies were limited in geographic scope. The SLYME framework models males and females separately and so sex-specific input data were used when possible. Sampled eels ≥ 400 mm (15.75 in) in length were assumed to be female (Krueger and Oliveira 1997; Oliveira and McCleave 2000).

The SLYME model calculates the number of American eels remaining in each age class following mortality, harvest, and emigration. Assumption regarding the initial number of glass eels recruiting to the coast and the maximum age (T) must be specified by the user. The model is sex-specific so the user must provide a value for the proportion of eels that are destined to become males. The glass eel fishery is prosecuted the day after arrival to the continent. The number of glass eels harvested is based on exploitation rate specific to the fishery that is supplied by the user. Glass eels not harvested by the glass eel fishery join the population residing in continental waters. Biological sampling data collected from the ASMFC-mandated annual young-of-year (YOY) survey were used to compute length and weight of age-0 eels that have not yet joined the continental segment of the stock. These data occasionally include lengths of older eels, so the length distributions from each state were examined by year to identify and exclude these older eels. The data suggested an upper limit of 75 mm (2.95 in) was an appropriate cut-off for age-0 eels. The average length and weight of individual eels ≤ 75 mm (2.95 in) were computed.

Growth in length for continental eels age-0 and older was described as a function of age. Weight was modeled as a function of length. Parameters describing the growth rates of American eels were estimated from the available biological data on individuals age 1 and older.

Natural mortality, M, was described as a function of weight based on a modified version of Lorenzen's (1996) equation:

$$M_t = \gamma 3.00 W_t^{-0.288}$$

where γ is an adjustment factor and W_t is weight at age t. The exponent value (-0.288) is considered fairly stable (McGurk 1996), but the coefficient value (3.00) may vary (D. Cairns, DFO Canada, pers. comm.). Application of the SLYME model to other systems applied an adjustment factor to account for the variability.

The model assumes that fishing for American eels that reside in continental waters occurs year-round and is concurrent with natural mortality. Catch curves were applied to fishery-dependent age samples to estimate total mortality rates for ages considered fully recruited to the gear. Catch curves were calculated within cohorts for cohorts that could be tracked for three or more years. The catch curves were computed assuming both constant and variable age at full recruitment. Estimates of total mortality were used as a starting point for determining an appropriate input estimate for resident fishing mortality. The user inputs an assumed fishing mortality rate for the resident fishery, which is allocated to each age class based on a vector of partial recruitment-at-age. The partial recruitment represents the proportion of each age class caught by the fishing gear. To determine the partial recruitment vector, catch-at-age was combined across years and sources; the ages of full recruitment were estimated by eye and a negative exponential curve was fit to these ages. The relationship was then applied to ages that are not fully recruited in order to estimate how many would be harvested if they were fully recruited. Partial recruitment-at-age was calculated as the observed number recruited-at-age divided by the potential fully recruited harvest at that age.

A fraction of the resident eels that survive the resident fishery were assumed to begin their spawning migration. The maturity of American eels is more dependent on length than age. A logistic function was fit to available data to predict the proportion of female eels that were mature at length. Maturity was modeled as a function of length in the SLYME model based on the logistic parameter estimates. The migrating silver eels were subject to an emigrant fishery the day after migration. The emigrant fishery catch was calculated based on an assumed fishing mortality rate and partial recruitment vector specific to the fishery. Both the resident and emigrant fisheries assumed no mortality on American eels less than the current minimum size limit (6.0 inches).

Fecundity was modeled as an allometric function of length. The number of eggs produced in each age class was calculated by multiplying the estimated fecundity-at-age by the number of female spawners that survived the emigrant fishery and subsequent natural mortality. The number of eggs was summed over all age classes to provide an estimate of total production. Dividing the total production by the initial number of recruits gave the number of eggs-per-recruit, which was the metric used in evaluating potential maximum size limits and slot limits.

The impact of maximum size limits and slot limits on the modeled population was investigated by setting fishing mortality rates equal to zero for eels exceeding the legal size given the maximum size or slot limit under consideration. The sensitivity of the results to assumptions made about the input parameters was also evaluated. Ranges of values were used in different model scenarios to understand how changing assumptions about the input parameters (e.g., proportion of future males, glass eel exploitation rate, maximum age, resident fishing mortality rate, emigrant fishing mortality rate) influenced results.

The amount of yield that would be foregone under a maximum size or slot limit was calculated to estimate the “cost” to the fishery of the size limit options evaluated. The percent of landings that would be considered illegal was calculated based on available data for recent years. The costs were calculated both in terms of landed numbers and landed weights. The costs associated with the various maximum size and slot limits were then compared against the increase in EPR that was predicted for the associate size limit.

RESULTS

Estimation of Input Parameters

The initial number of glass eels recruiting to continental waters was set equal to 1,000,000. Sex ratios were computed from available data where the sex of individuals was recorded. The percentage of males observed was highly variable among life stages and sampling locations, ranging from 1–97% where total sample sizes ≥ 50 individuals. In datasets that also identified life stage, 32–64% of yellow-stage eels were male while 45–97% of silver-stage eels were male (where n ≥ 50). The SASC also reviewed the literature for research on American eel sex ratios in the U.S. and found that published estimates were also variable (Michener 1980; Harell and Loyacano 1982; Hansen and Eversole 1984; Helfman et al. 1984; Oliveira and McCleave 2000; Rulifson et al. 2004). The percent of males in published studies ranged from 3–97% depending on the life stage, habitat, and sampling location. Initial runs of the model assumed 50% of the recruits were destined to become male. Alternate configurations of the model assumed proportions of future males that ranged from 10-90%. The maximum age observed in exploited areas was 15 years (one individual). Approximately 99% of aged samples from those areas were younger than 10 years. The maximum age observed from all areas (fished and unfished) was 33 years.

Based on the annual YOY survey, the average length of age-0 American eels was 56.7 mm (2.23 in) with an average weight of 0.150 g (3.31E-04 lbs). The von Bertalanffy growth curve was fit to available data to estimate the age-length relationship. The best fit parameter estimates were: L_∞ = 28.2 inches, K = 0.22, t_0 = -1.63. The relationship of weight to length was modeled using an allometric function. The length-weight parameter values from the best-fit were: a = 2.70E-05, b = 3.31.

There was limited information available to determine an appropriate adjustment factor for Lorenzen's natural mortality equation. As a starting point, the SASC assumed $\gamma = 0.164$, the value assumed in recent applications of the SLYME model to Canadian data (D. Cairns, pers. comm.).

Few glass eel fisheries are currently active, so a relatively small value was assumed for the exploitation rate in this fishery. An exploitation rate value of 0.01 was assumed for initial runs. Alternate runs considered values ranging from 0–0.75. The age distribution of commercially caught American eels suggested that female American eels are fully recruited to the resident fishery by age 3 or 4 (Figure 1). The catch curves estimated total mortality rates ranging from 0.14–0.77 or 0.19–0.60 depending on whether age at full recruitment is assumed variable or constant. Average total mortality was estimated at 0.50 when age at full recruitment was assumed constant (age-4). Assuming variable age at full recruitment, the average value among cohorts was 0.46. Subtracting the average of the estimated natural mortality rates at age for ages 4 and older ($M_{\text{avg, }4+} = 0.04$) suggested remaining loss could be 0.46 or 0.42 for fully recruited ages, depending on the assumption regarding age at full recruitment. A review of previous research identified only one estimate of yellow eel fishing mortality in the U.S. A study in Maryland estimated instantaneous fishing mortality for selected systems to equal 0.43 (J. Weeder, NOAA Marine Fisheries, pers. comm.). For initial runs, a value of 0.43 was assumed for resident fishing mortality.

A logistic curve was fit to available data to predict female maturity at length (Figure 2). No age or size composition data from silver eel fisheries were available for estimating mortality rates or deriving a partial recruitment vector. No published estimates of emigrant fishing mortality in the U.S. were found. One study on silver eels in the St. Lawrence River Estuary, Canada estimated instantaneous fishing mortality at about 0.26 (Caron and Verreault 1997). The SLYME model assumed a value of 0.26 for instantaneous fishing mortality in the emigrant fishery during initial runs. In the absence of available data, partial recruitment to the emigrant fishery was assumed equal to 1.0 for all ages.

Data for deriving a function for fecundity were not available in the data provided to the SASC. A review of the literature yielded only two studies that estimated fecundity for American eels in U.S. sampling locations (Wenner and Musick 1974; Barbin and McCleave 1997). Though both studies were limited by small sample sizes ($n = 21$ and $n = 63$, respectively) and limited geographic and temporal scope, the SASC decided to use the parameter estimates from the more recent study. A preliminary analysis comparing cumulative fecundity at size as a percent of total fecundity showed only minor differences in the two relationships. This suggests that perceived benefits of a size limit would be comparable using either relationship.

Eggs-per-Recruit

The estimated number of eggs-per-recruit (EPR), based on the initial values assumed for the input parameters, was compared to the EPR calculated under various model scenarios that considered a range of values for select input parameters. EPR was inversely related to the value assumed for the proportion of the stock destined to become males (Figure 3). That is, increasing the number of males resulted in decreasing EPR. The estimated EPR was also sensitive to the value assumed for the Lorenzen adjustment factor, γ (Figure 4). An increase in γ results in an increase in natural mortality-at-age, which results in a decrease in the EPR estimate. Increases in the assumed rate of glass eel fishery exploitation resulted in decreasing EPR; this effect was most noticeable at exploitation rates ≥ 0.50 (Figure 5).

Maximum size limits ranging from 16–28 inches at 1-inch intervals were applied to the simulated population to evaluate the impact on stock productivity. As the assumed maximum size increased, the estimated EPR increased (Figure 6). Maximum size limits ranging from 24–28 inches provided less than a 1.0% increase in EPR (Figure 7). A maximum size of 23 inches resulted in a potential 2.4% increase in EPR. As one would expect, the estimated gain in EPR increased as the maximum size limit considered decreased. The largest predicted gain in EPR expectedly occurred at the smallest maximum size limit evaluated, 16 inches. At this maximum size, the EPR was predicted to increase by 133% relative to the base model.

The effect of a maximum size limit on stock productivity was also evaluated assuming a range of values for the minimum size limit. For this slot limit analysis, the estimated change in EPR was calculated for various combinations of minimum and maximum size limits relative to the base model, which assumed no maximum size limit and a minimum size limit equal to the current minimum size (6.0 inches). The slot limit analysis suggested the gain in EPR achieved from coupling minimum sizes less than 17 inches with a maximum size limit was only marginal compared to the gain in EPR predicted for the maximum size limit alone (generally < 8%; Table 2; Figure 7). Slot limit combinations with fairly narrow (< 5 inches) slots and minimum sizes > 17 inches could provide an estimated 42–123% increase in EPR (Table 2). Combinations with larger slots (≥ 5 inches) at minimum sizes > 17 inches were estimated to provide increases in EPR ranging from 16–66%. Recall that the gain in EPR achieved from these maximum sizes (≥ 22 inches) alone was less than 12% relative to the base model (Figure 7). As such, these larger maximum sizes made only a small contribution to the predicted increase in EPR achieved from those slot combinations.

The sensitivity of EPR estimates at different maximum size limits was evaluated by varying assumptions about the values of selected input parameters (Figures 8–10). The relationship of EPR to changes in the assumed values for proportion of future males, Lorenzen adjustment factor, and glass eel fishery exploitation rates to the maximum size limits considered was similar to trends for the base model (Figures 3–5), with varying magnitude. One of the largest sources of uncertainty with the input parameters was the harvest mortality of resident and emigrating eels. The impact of this uncertainty on productivity was evaluated by calculating EPR over a range of assumed values for the resident and emigrant fisheries (Figure 10). Increasing the fishing mortality rate in either fishery expectedly results in a decreased EPR. The evaluation showed that EPR was more sensitive to changes in the harvest of emigrating eels than the harvest of resident eels.

Costs to the Fishery

The percentage of commercial landings exceeding a range of proposed size limits was calculated for selected states to estimate the amount of landings that would be considered illegal for those size limits. Data for estimating these costs were only available from New Jersey, Delaware, Maryland, and Florida. The three most recent years of available data from each state were used. Data collected during 2005–2007 were provided from Delaware and Maryland. New Jersey data were available from 2006 and 2007. Data provided by Florida were available from 2004–2006.

The percent of landings greater than each of the maximum size limits evaluated was calculated for each state and year. The percentage values were then averaged across years for each state to provide the estimated cost in terms of both landed numbers and landed weight. For all states evaluated the costs in landed numbers and landed weight decreased as the maximum size increased (Figure 11). The percentage of landings in terms of weight that would be considered illegal exceeded the percentage that would be considered illegal in terms of numbers. The cost in weight and numbers of the various maximum size limits considered varied among the states. For example, approximately 94% of New Jersey's landings in weight would be foregone if a maximum size limit of 16 inches were imposed. However, Delaware would lose an estimated 42% of their landings in weight for a 16-inch maximum size limit.

The comparison of costs to the fishery to gains in egg production demonstrated that as predicted EPR increased, so did the expected loss to the fishery. Gains in EPR greater than 50% were predicted to cost a minimum of 25% in commercially landed weight, depending on the maximum size and the state affected (Figures 12–15). At maximum size limits greater than 22 inches, the expected gains in EPR were less than 3%. However, the cost in landings could range from 8% to 41% in weight.

The percentage values of landings in each state that have exceeded the slot limit combinations considered are shown in Tables 3–6. The costs of the various slot limit combinations to each state were variable. In general the costs in terms of landed weight exceeded the costs in landed numbers for slots with smaller minimum and smaller maximum sizes. As both the minimum and maximum sizes of the slot increased, the costs in terms of landed numbers increased relative to the costs in terms of landed weight. The slot combinations predicted to provide larger increases in EPR (Table 2) were those associated with the higher costs to the fishery (Tables 3–6).

DISCUSSION

The results of the per-recruit analyses suggested there could be a potential gain in American eel stock productivity by imposing a maximum size limit. Larger maximum size limits were predicted to result in higher egg production. However, as the predicted gains in EPR increased, so did the estimated cost to the fishery. The cost analysis showed that even nominal gains in EPR could still result in substantial losses to the fishery. The model results also showed that a slot limit could also potentially benefit egg production. Slot limit combinations that included minimum sizes greater than 16 inches were predicted to increase EPR from 16% to 123% relative to the base model. Though, the gain in EPR relative to a maximum size limit alone was estimated to yield an increase of 16% to 70% at slots with minimum sizes greater than 16 inches. Slots with minimum sizes less than 17 inches were predicted to provide less than an 8% increase in EPR relative to maximum sizes alone.

An effective maximum size limit should result in an increase in the number of emigrating female eels, but information on the size at which female eels emigrate is limited. A recent study found that female eels emigrated from Indian River in southern Delaware during September and November in 2002 and 2003 and their length ranged from 14.4–29.3 inches (367–744 mm) with an average length of 22.6 inches (571 mm; Barber 2004). Although this size range for female emigration could not be confirmed for the entire distribution range of the American eel, coast-wide similarities in the length range of commercially caught eels suggested that a maximum size limit based on the mean length of emigrating female eels in Delaware could increase the number of female eels emigrating to spawn.

Sex ratios estimated from available data were variable, as were estimates found in the literature (Michener 1980; Harell and Loyacano 1982; Hansen and Eversole 1984; Helfman et al. 1984; Oliveira and McCleave 2000; Rulifson et al. 2004). Sex ratios may be different among life stages (this report; Oliveira and McCleave 2000). Future work with the SLYME model may want to consider different sex ratios for the yellow and silver stage segments of the population.

American eels residing in waters along the U.S. East Coast are considered a single unit and were treated as such in the model. However, literature studies and analyses performed for this report have demonstrated evidence of spatial and temporal differences in life history, timing of events (e.g., recruitment to the continent, emigration), and exploitation patterns throughout the species range. Biological sampling of American eel has improved in recent years, but is still not comprehensive. Both fishery-dependent and -independent data gaps exist for different geographic regions, gear types, life stages, unexploited systems, and time periods. Data from sampled areas were used to supplement areas lacking information; this required the assumption that available data were representative of unsampled areas. Efforts to improve data collection throughout the American eel's range are needed if the reliability of this and other models is to be increased. In the meantime, regional models or a single model that incorporates regional-weighted data could provide more appropriate results and should be considered for future work.

The estimated gains in EPR and costs to the fishery are relative and the assumptions made in developing the model must be considered when evaluating the results. The reliability of the results is largely dependent on the degree to which these assumptions hold and so should be interpreted with caution. Numerous assumptions were needed because of the complex life history of the American eel and the uncertainty regarding stock size and mortality. For instance, the assumption that glass eel arrival at the coast and silver eel emigration occur during one day one a coast-wide basis is not accurate, but it's considered necessary to simplify the assumption for carrying out model computations. The assumption of constant effort implies that harvest rates will not change. An increase in fishing pressure would reduce the predicted EPR and so could limit the effectiveness of a maximum size or slot limit. Two of the weakest assumptions were those made for the exploitation rate and partial recruitment at age in the emigrant fishery. No data were available to characterize the composition of the catch and only one estimate of exploitation rate—from Canada—could be found. The evaluation of the effect of exploitation rates on resident and emigrating eels demonstrated that EPR was sensitive to fishing mortality in the emigrant fishery.

The assumption that all female eels age-4 and older are fully vulnerable to the resident fishery may not be representative of the entire U.S. stock. Large-size eels (> 27.6 in or 700 mm) that have large girths (> 2.0 in or 50.8 mm) are likely not fully selected by pots, the primary commercial gear that harvests American eels (K. Whiteford, Maryland Department of Natural Resources, pers. comm.). However, the maximum length attained by females in the model was 27.5 inches (698.5 mm) when the maximum age was assumed to equal 15 years. As such, it is assumed that female eels in the simulated population do not reach the length and girth at which their selectivity becomes limited.

The costs to the states were dependent on the length and weight composition of recent landings. The characterization of landings was based upon biological samples collected from commercial landings in each of the states. The estimation of costs is therefore dependent upon how well those biological samples represent the landings as a whole.

The estimated gains in EPR assumed all other factors contributing to pre-spawning mortality remained constant. Many factors besides fishing are known or expected to affect overall mortality, including impediments to upstream migration, turbine mortality during out-migration, loss/alteration of habitat, predation and competition, harvest in areas outside the Atlantic coast, and so on. Increases in mortality due to these factors would reduce the estimated gain in EPR. Conversely, decreases in mortality from these factors could increase the expected gain in EPR. The contribution of the various sources of mortality, including harvest, to the total mortality of American eel is unknown. The impact of reducing fishing mortality will depend on the degree to which harvest mortality contributes to the total mortality. Efforts to reduce or eliminate any source of anthropogenic would benefit the stock and promote the rebuilding.

Incorrect input values or violation of assumptions would result in different model results; however, it is not possible to characterize the directionality of all differences (i.e., would results be higher or lower). In addition, many of the parameters are interrelated, and may work to amplify or dampen the effects of incorrect starting values. The SASC performed several analyses in an effort to evaluate confidence in model estimates. Sensitivity analyses were conducted with input values to determine which parameters had the greatest effect on model results. Also, where multiple input values were available, these were often used to estimate bounds on model results. The SASC has recommended that further sensitivity analyses, including addition of stochastic growth and recruitment, be performed to provide a better understanding of the model's sensitivity to input assumptions.

The relative gains in EPR estimated by the model should be considered upper bounds of potential benefits. The model evaluated the response of the exploited segment of the stock to size limit regulations as eels in exploited areas are the ones directly impacted by fishery restrictions (i.e., size limits will only apply to fished areas). As such, the predicted increases in EPR are relative to the portion of the stock that is subject to exploitation, given the assumption that eels emigrating from exploited areas contribute to the spawning population. The proportion of the stock that is exploited is not known and the relative contribution of spawners from fished and unfished areas is unknown, so the actual observed benefit can not be predicted. In addition, the American eel population is panmictic and extends beyond the Atlantic seaboard. Increases in escapement resulting from U.S. management measures have the potential to benefit the species anywhere within its range (i.e., U.S. management could result in increased recruitment anywhere from Labrador to Brazil).

The SASC believes that the results of the SLYME model provide a reasonable insight into the effects of imposing a maximum size limit or slot limit, as long as consideration for the underlying assumptions is given. The costs associated with the potential management scenarios evaluated should be weighed against the estimated gain in egg production, keeping in mind that the impact on recruitment is unknown. Additionally, issues of enforceability and the ability of the commercial fishery to conform to size limit regulations should be evaluated.

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Table 1. Summary of raw datasets provided to the SASC for evaluation.

Name	Affiliation	Sampling Region	Start	End	Length	Weight	Age	Sex	Stage	Comment
Various	ASMFC States/Juris.	Multiple	2000	2005	X	X				Annual YOY surveys
K. Oliveira	UMass Dartmouth	Maine	1996	1998	X	X	X	X	X	
K. Oliveira	UMass Dartmouth	Massachusetts			X	X		X	X	
K. Oliveira	UMass Dartmouth	Rhode Island	1990	1991	X		X	X	X	Silver eel sampling
W. Morrison	UMaryland CEES	New York	1998	1999	X	X		X		Unexploited system
V. Vecchio*	NY Dept. Env. Cons.	New York	2002	2002	X	X				Electrofishing
V. Vecchio*	NY Dept. Env. Cons.	New York	2002	2006	X	X				Fyke survey
K. Strait	PSEG	New Jersey	1998	2001	X					Trawl survey
J. Brust	NJ Dept. Env. Prot.	New Jersey	2006	2006	X	X				Commercial sampling
C. Cairns	Delaware State Univ.	Delaware	2005	2006	X	X				Tagging study
J. Clark	DE DNREC	Delaware	2000	2006	X	X	X			Commercial sampling
K. Whiteford	MD Dept. Nat. Res.	Maryland	1999	2001	X	X	X			Freshwater sampling
K. Whiteford	MD Dept. Nat. Res.	Maryland	1997	2006	X	X	X	X		Commercial sampling
K. Whiteford	MD Dept. Nat. Res.	Maryland	1997	2006	X	X	X	X		Pot survey
M. Montane	VIMS	Virginia	1997	2005	X	X	X			Trawl survey
J. Cimino	VA Marine Res. Comm.	Virginia	1989	2008	X	X		X		Commercial sampling
H. Hildebrand	Univ. West VA.	West Virginia	2003	2004	X	X	X			Shenandoah River
R. Graham	Dominion Power	North Carolina	2000	2005	X	X				Roanoke Rapids
J. Cudney	ECU	North Carolina	2002	2003	X	X	X	X	X	
K. Bonvechio	FL FWCC	Florida	2002	2006	X	X		X		

* Currently with NOAA Marine Fisheries

Table 2. Estimated percentage (%) increase in eggs-per-recruit for various combinations of potential slot limits relative to the base model (current minimum size limit* = 6.0 inches; no maximum size limit).

		Slot Minimum Size (in)													
		6 *	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	133	133	133	133	133	133	135	137	137					
	17	115	115	115	115	115	115	117	119	119	132				
	18	92	92	92	92	92	92	93	95	95	106	123			
	19	65	65	65	65	65	65	67	68	68	78	92	114		
	20	40	40	40	40	40	40	41	42	42	50	62	80	106	
	21	22	22	23	23	23	23	23	24	24	32	42	58	79	108
	22	12	12	12	12	12	12	12	13	13	20	29	43	63	88
	23	2	2	2	2	2	2	3	4	4	10	19	31	48	70
	24	0.3	0.3	0.3	0.3	0.4	0.4	1	2	2	8	16	29	45	66
	25	0.03	0.03	0.04	0.04	0.06	0.06	1	2	2	8	16	28	45	65
	26	0.0001	0.0001	0.01	0.01	0.03	0.03	1	2	2	8	16	28	45	65
	27	0	0	0.01	0.01	0.03	0.03	1	2	2	8	16	28	45	65
	28	0	0	0.01	0.01	0.03	0.03	1	2	2	8	16	28	45	65
	none	0	0	0.01	0.01	0.03	0.03	1	2	2	8	16	28	45	65

Table 3. Estimated percentage of New Jersey's commercial landings in number (A) and weight (lb; B), exceeding the associated slot limit combination.

A		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	79	79	80	82	85	89	93	96					
	17	72	72	74	75	78	83	86	90	93				
	18	64	64	65	67	70	74	78	81	85	91			
	19	54	54	55	57	60	64	68	71	75	82	90		
	20	45	45	47	48	51	55	59	63	66	73	82	91	
	21	35	35	36	37	41	45	49	52	56	62	71	81	89
	22	27	27	28	30	33	37	41	44	48	55	63	73	82
	23	19	19	21	22	25	30	33	37	41	47	56	65	74
	24	14	14	15	17	20	24	28	31	35	42	50	60	69
	25	9	9	11	12	15	19	23	26	30	37	45	55	64
	26	4	4	5	7	10	14	18	21	25	31	40	50	59
	27	2	2	3	5	8	12	16	19	23	30	38	48	57
	28	1	1	2	4	7	11	15	18	22	29	37	47	56

B		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	94	94	95	95	95	96	97	98					
	17	91	91	91	91	92	93	94	95	97				
	18	85	85	85	86	86	87	88	89	91	94			
	19	78	78	78	79	79	80	81	82	84	87	93		
	20	71	71	71	71	72	73	74	75	77	80	86	93	
	21	60	60	60	60	61	62	63	64	66	69	75	82	89
	22	51	51	51	51	52	53	54	55	57	60	66	73	80
	23	41	41	41	41	42	43	44	45	47	50	56	63	70
	24	32	32	32	33	33	34	35	36	38	41	47	54	61
	25	23	23	23	23	24	25	26	27	29	32	38	45	52
	26	11	11	11	11	12	13	14	15	17	20	26	33	40
	27	7	7	7	7	8	8	10	11	12	16	21	28	36
	28	4	4	5	5	5	6	7	8	10	13	19	26	33

Table 4. Estimated percentage of Delaware's commercial landings in number (A) and weight (lb; B), exceeding the associated slot limit combination.

A		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	17	19	26	41	60	75	85	94					
	17	12	14	22	36	56	71	81	89	96				
	18	9	11	19	33	53	68	78	87	93	97			
	19	8	10	17	31	51	66	76	85	91	95	98		
	20	6	8	15	30	49	64	75	83	89	94	96	98	
	21	4	6	14	28	48	63	73	82	88	92	95	97	99
	22	3	5	12	27	46	61	72	80	86	91	93	95	97
	23	2	4	11	26	45	60	70	79	85	89	92	94	96
	24	1	3	10	25	44	60	70	78	84	89	92	93	95
	25	0.3	2	10	24	44	59	69	77	84	88	91	93	94
	26	0.1	2	10	24	44	59	69	77	83	88	91	93	94
	27	0.1	2	10	24	44	59	69	77	83	88	91	93	94
	28	0.1	2	10	24	44	59	69	77	83	88	91	93	94

B		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	42	43	46	52	64	74	83	92					
	17	35	36	39	46	57	68	76	85	93				
	18	30	31	34	41	52	62	71	80	88	95			
	19	26	27	30	36	47	58	67	76	84	91	96		
	20	22	22	25	32	43	54	63	72	80	86	92	96	
	21	18	18	21	28	39	50	58	67	75	82	87	92	96
	22	13	13	16	23	34	45	54	63	71	77	82	87	91
	23	8	9	12	19	30	41	49	58	66	73	78	82	87
	24	5	5	8	15	26	37	45	55	63	69	74	79	83
	25	1	2	5	12	23	33	42	51	59	66	71	75	79
	26	0.04	1	4	10	21	32	41	50	58	65	70	74	78
	27	0.04	1	4	10	21	32	41	50	58	65	70	74	78
	28	0.04	1	4	10	21	32	41	50	58	65	70	74	78

Table 5. Estimated percentage of Maryland’s commercial landings in number (A) and weight (lb; B), exceeding the associated slot limit combination.

A		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	22	23	26	36	57	78	89	96					
	17	18	19	22	32	52	73	85	91	96				
	18	14	15	18	28	49	69	81	88	92	96			
	19	12	12	15	26	46	67	78	85	89	93	97		
	20	9	10	13	23	43	64	75	82	86	91	94	97	
	21	7	7	10	20	41	62	73	80	84	88	92	95	98
	22	5	5	8	19	39	60	71	78	82	86	90	93	96
	23	3	4	7	17	37	58	69	76	80	85	88	91	94
	24	2	3	6	16	36	57	69	75	79	84	88	90	93
	25	1	2	5	15	35	56	68	74	78	83	86	89	92
	26	0.4	1	4	14	35	55	67	74	78	82	86	89	92
	27	0.2	1	4	14	34	55	67	73	78	82	86	89	91
	28	0.1	1	4	14	34	55	67	73	78	82	86	88	91

B		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	59	59	59	62	71	81	89	95					
	17	52	53	53	56	65	75	83	89	94				
	18	45	46	46	49	58	68	76	82	87	93			
	19	40	40	40	43	52	62	70	76	81	87	94		
	20	33	33	34	37	45	56	64	69	75	81	88	94	
	21	27	28	28	31	39	50	58	64	69	75	82	88	94
	22	21	21	22	25	33	44	52	57	63	69	76	82	88
	23	15	15	16	19	27	38	45	51	56	63	70	75	82
	24	11	11	12	15	23	34	41	47	52	58	65	71	78
	25	6	6	7	9	18	28	36	42	47	53	60	66	73
	26	2	3	3	6	14	25	33	39	44	50	57	63	69
	27	1	1	2	5	13	24	31	37	42	48	56	61	68
	28	0.3	1	1	4	13	23	31	37	42	48	55	61	67

Table 6. Estimated percentage of Florida's commercial landings in number (A) and weight (lb; B), exceeding the associated slot limit combination.

A		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	82	82	82	82	82	82	84	91					
	17	68	68	68	68	68	69	71	78	87				
	18	57	57	57	57	57	57	59	66	75	88			
	19	46	46	46	46	46	46	48	55	64	77	89		
	20	33	33	33	33	33	33	35	42	51	64	76	87	
	21	24	24	24	24	24	25	26	33	42	56	67	78	91
	22	16	16	16	16	16	17	18	25	34	48	59	70	83
	23	9	9	9	9	9	10	12	19	28	41	53	64	77
	24	6	6	6	6	6	6	8	15	24	37	49	60	73
	25	2	2	2	2	2	3	5	11	20	34	45	56	69
	26	0.5	0.5	0.5	0.5	1	1	3	10	19	32	44	55	68
	27	0.2	0.2	0.2	0.2	0.3	1	3	9	19	32	43	55	68
	28	0.1	0.1	0.1	0.1	0.2	1	3	9	18	32	43	54	67

B		Slot Minimum Size (in)												
		8	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Slot Maximum Size (in)	**16**	92	92	92	92	92	92	93	95					
	17	84	84	84	84	84	84	84	87	92				
	18	75	75	75	75	75	75	76	79	83	91			
	19	65	65	65	65	65	66	66	69	74	82	90		
	20	52	52	52	52	52	52	53	56	60	69	77	87	
	21	42	42	42	42	42	42	43	45	50	58	67	76	90
	22	30	30	30	30	30	31	31	34	39	47	55	65	78
	23	20	20	20	20	20	20	20	23	28	36	45	54	68
	24	13	13	13	13	13	13	13	16	21	29	38	47	61
	25	5	5	5	5	5	5	6	9	13	22	30	40	53
	26	2	2	2	2	2	2	2	5	10	18	27	36	49
	27	1	1	1	1	1	1	1	4	9	17	26	35	49
	28	0.3	0.3	0.3	0.3	0.4	1	1	4	9	17	25	35	48

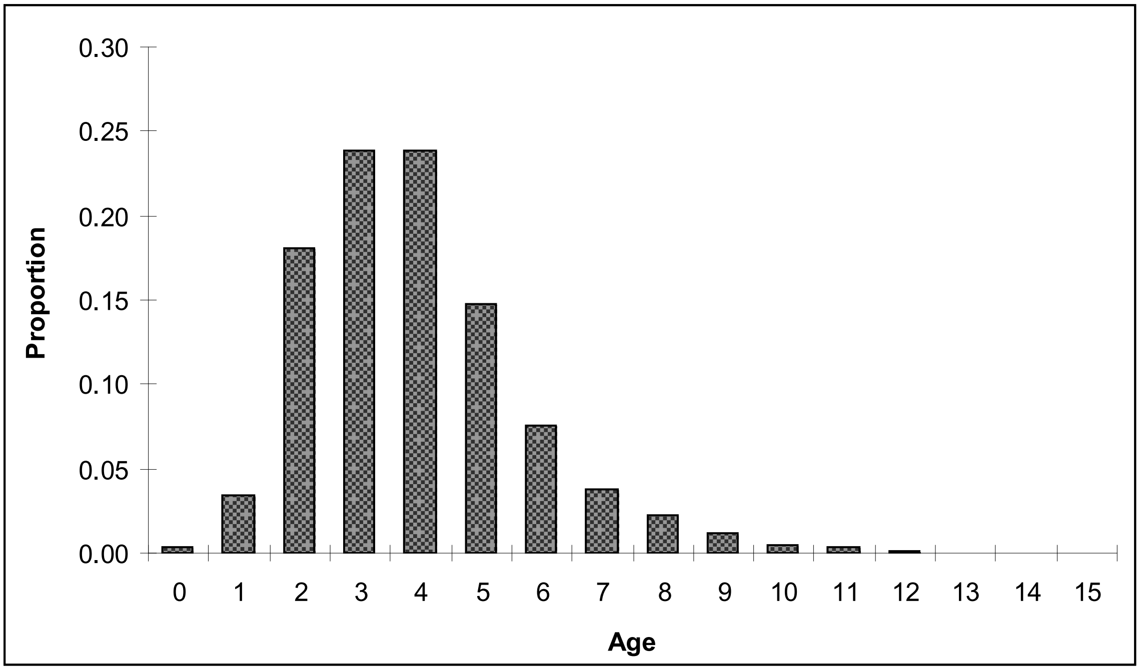

Figure 1. Proportion of American eels at age based on samples from the commercial fishery.

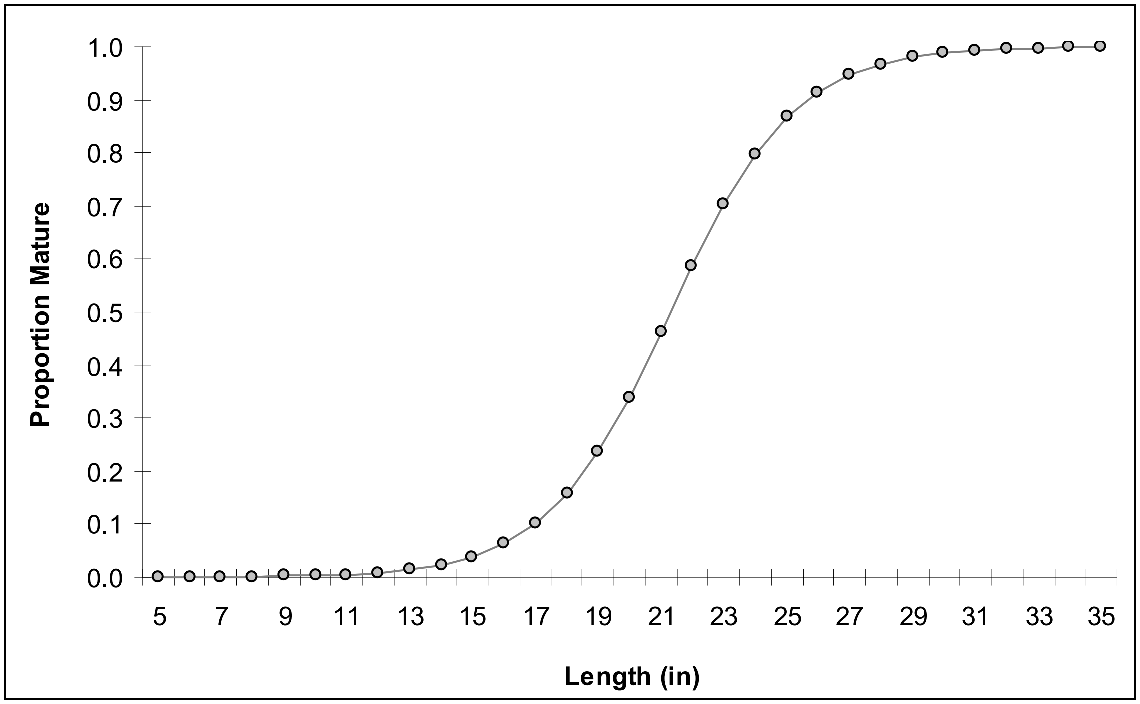

Figure 2. Maturity-at-length for American eel based on best fit of logistic curve to available data.

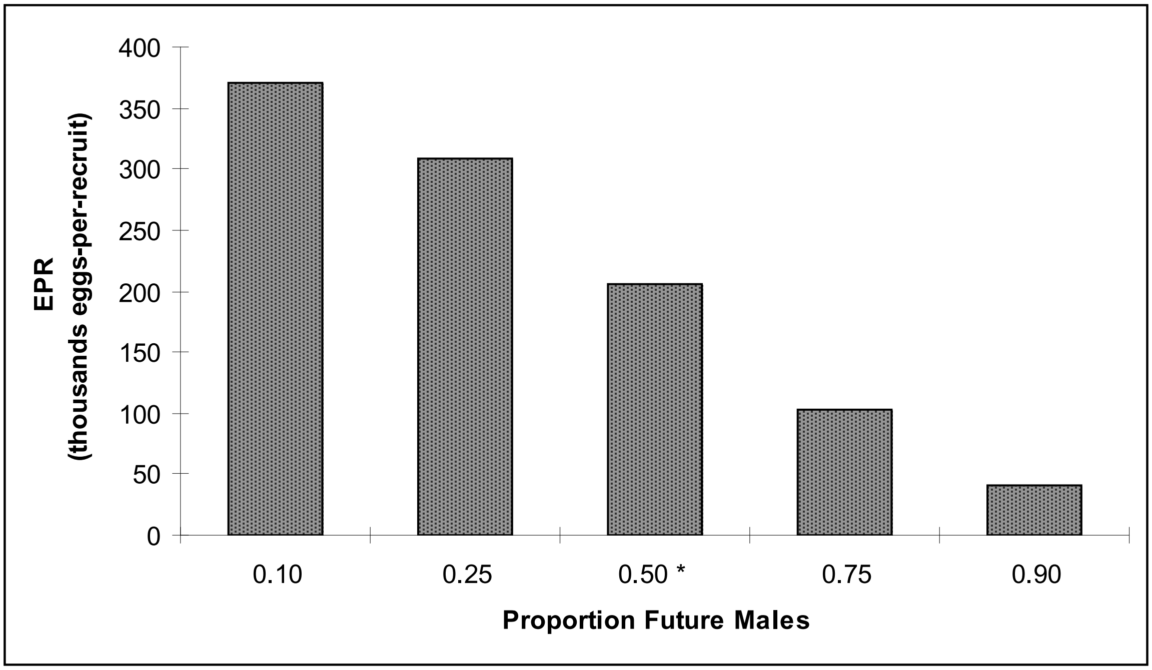

Figure 3. Estimated number of eggs-per-recruit (thousands of eggs/recruit) over a range of assumed values for the proportion of the stock destined to become male. Asterisk indicates value assumed for initial run.

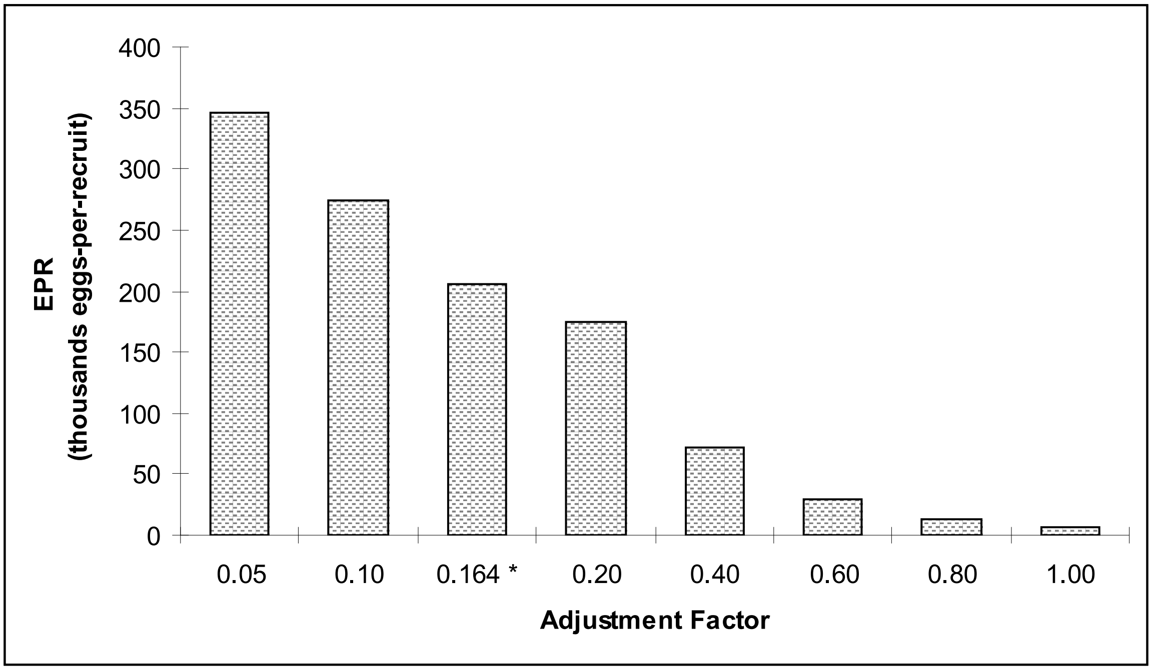

Figure 4. Estimated number of eggs-per-recruit (thousands of eggs/recruit) over a range of assumed values for the adjustment factor (γ) to the Lorenzen equation relating weight and natural mortality. Asterisk indicates value assumed for initial run.

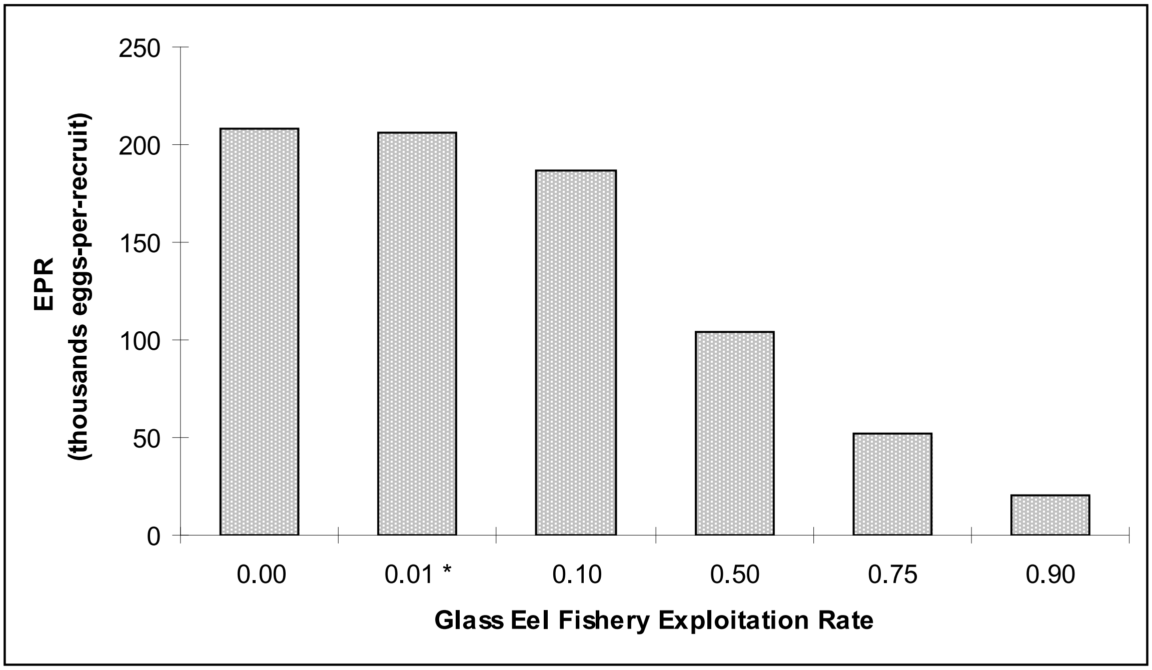

Figure 5. Estimated number of eggs-per-recruit (thousands of eggs/recruit) over a range of assumed exploitation rates for the glass eel fishery. Asterisk indicates value assumed for initial run.

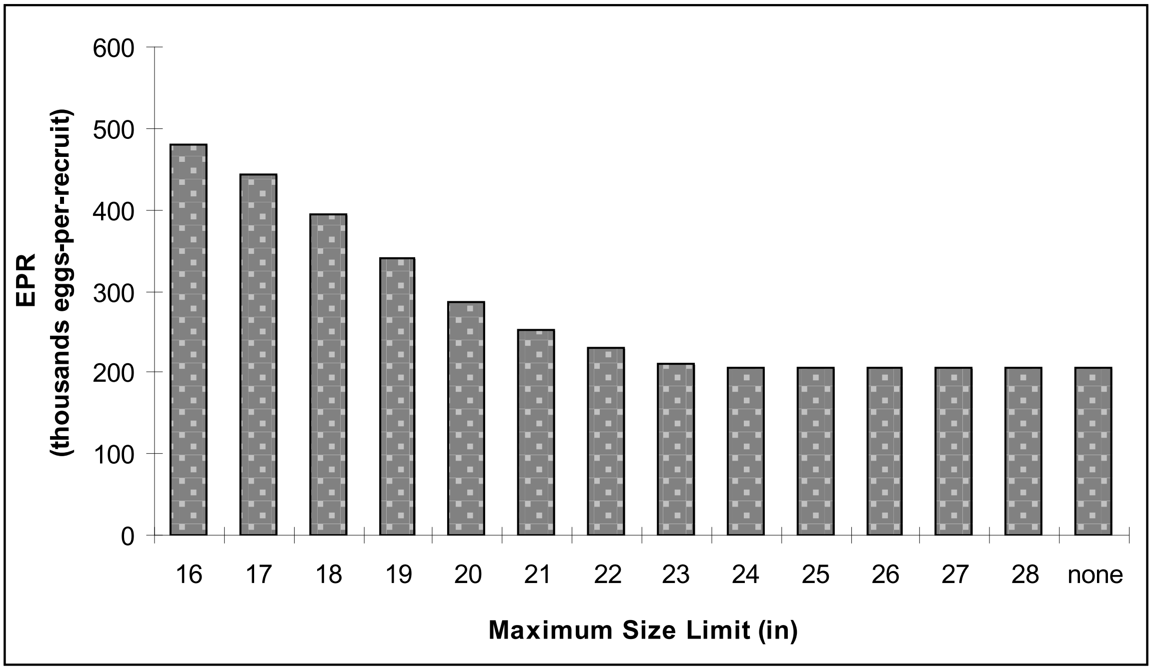

Figure 6. Estimated number of eggs-per-recruit (thousands of eggs/recruit) for various potential maximum size limits.

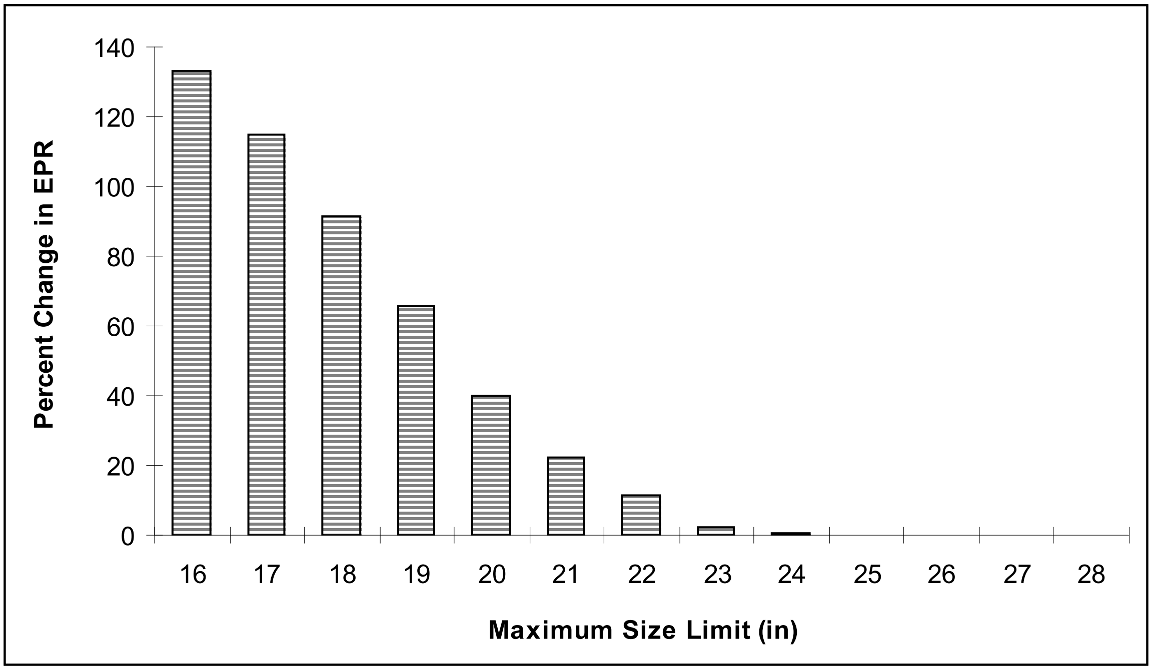

Figure 7. Estimated percentage (%) increase in eggs-per-recruit for various maximum size limits relative to the base model (current minimum size limit = 6.0 inches; no maximum size limit).

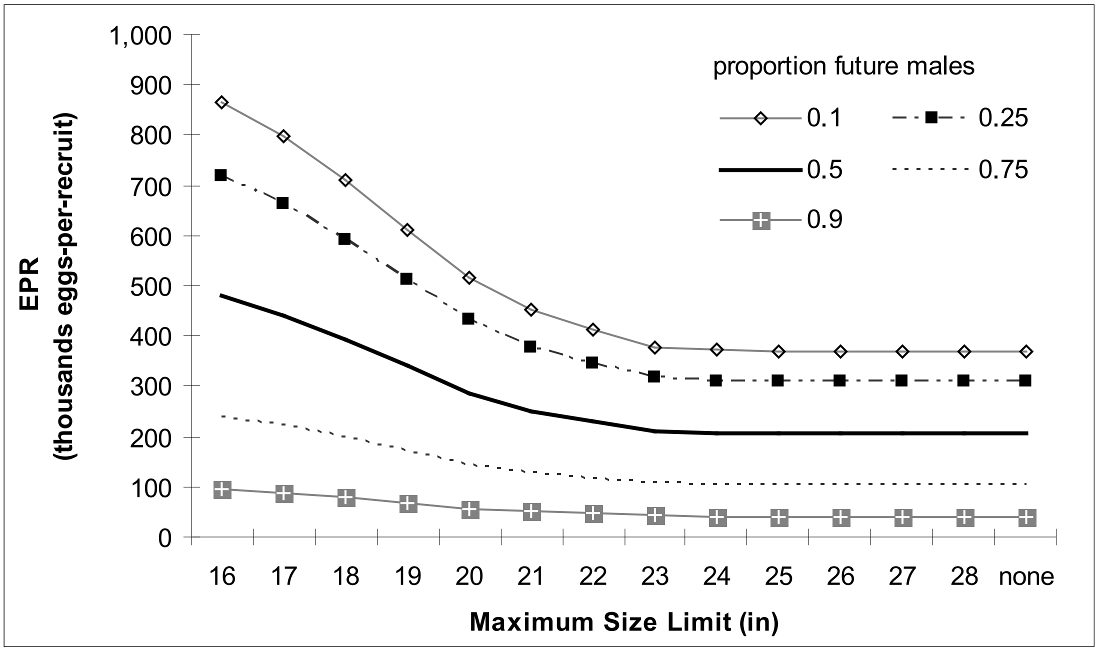

Figure 8. Estimated number of eggs-per-recruit (thousands of eggs/recruit) for various maximum size limits over a range of assumed values for the proportion of the stock destined to become males.

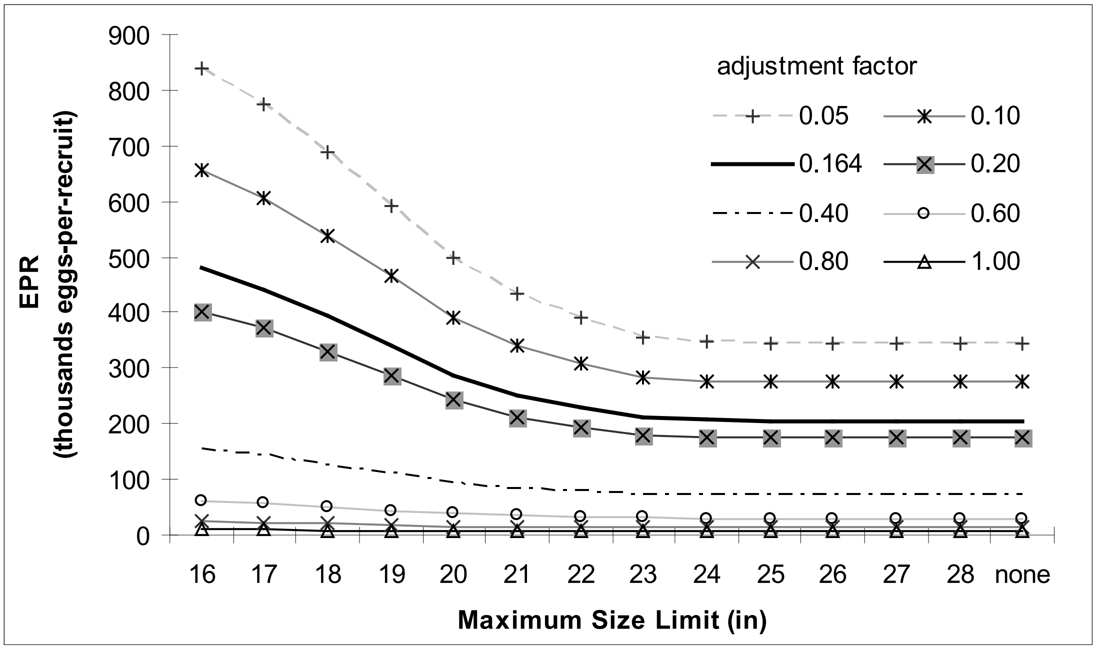

Figure 9. Estimated number of eggs-per-recruit (thousands of eggs/recruit) for various maximum size limits over a range of assumed values for the adjustment factor (γ) to the Lorenzen equation relating weight and natural mortality.

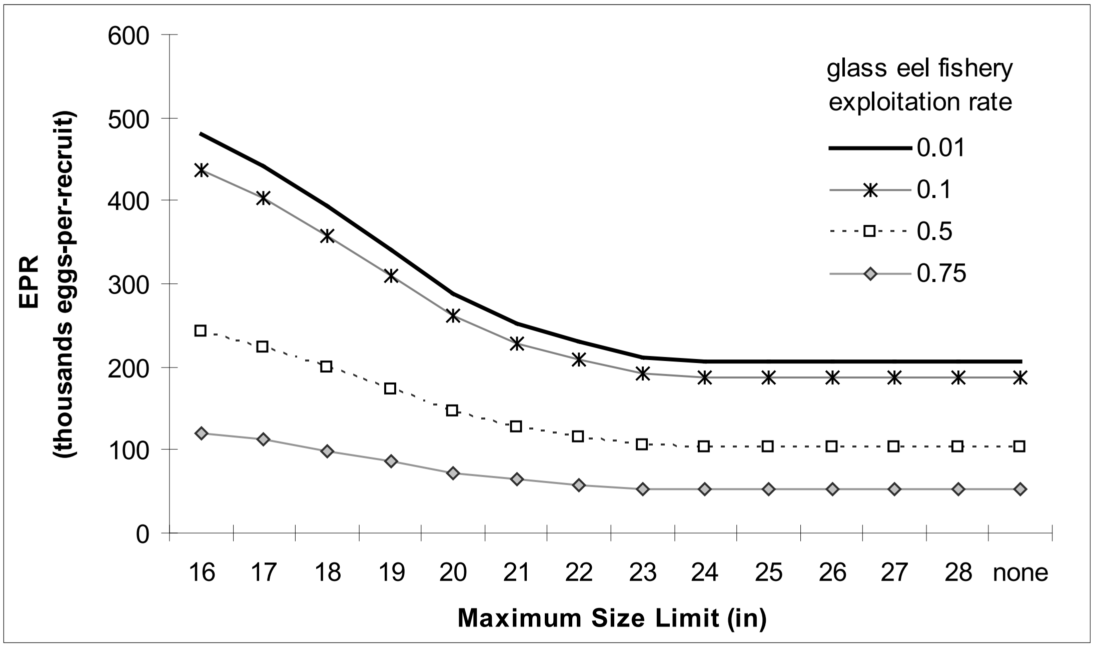

Figure 10. Estimated number of eggs-per-recruit (thousands of eggs/recruit) for various maximum size limits over a range of assumed exploitation rates for the glass eel fishery.

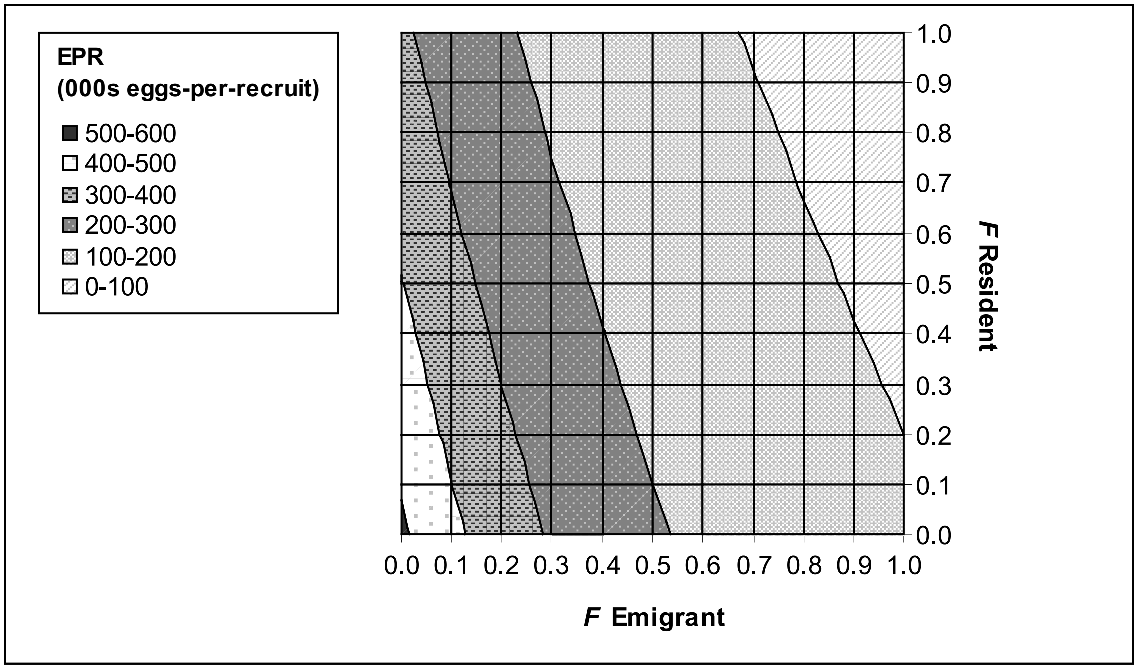

Figure 10. Estimated number of eggs-per-recruit (thousands of eggs/recruit) over a range of assumed fishing mortality rates for the resident ($F_{Resident}$) and emigrant ($F_{Emigrant}$) fisheries.

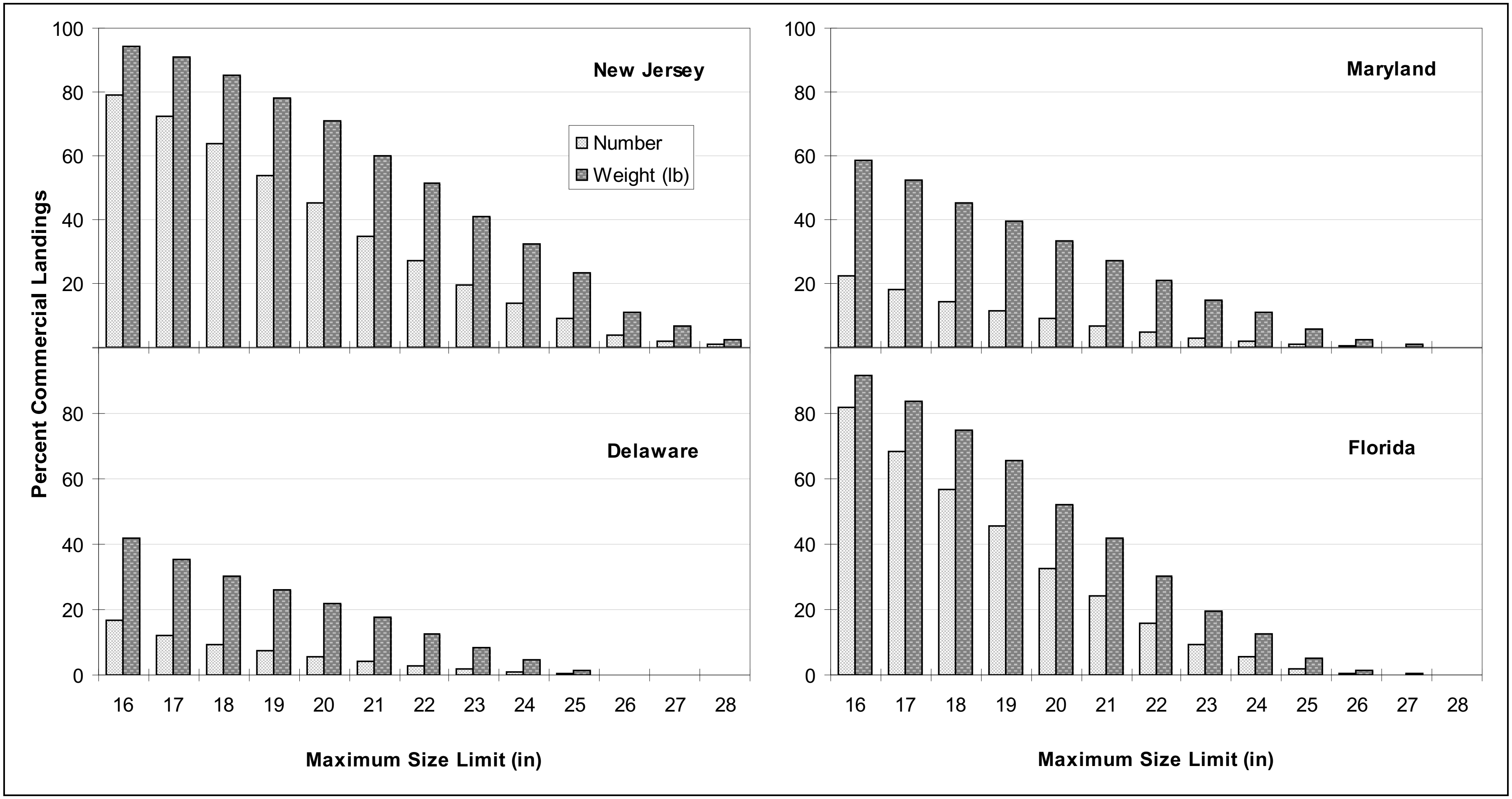

Figure 11. Estimated percentage of commercial landings, in terms of weight (lb) and numbers, greater than the maximum size limits considered, for selected states.

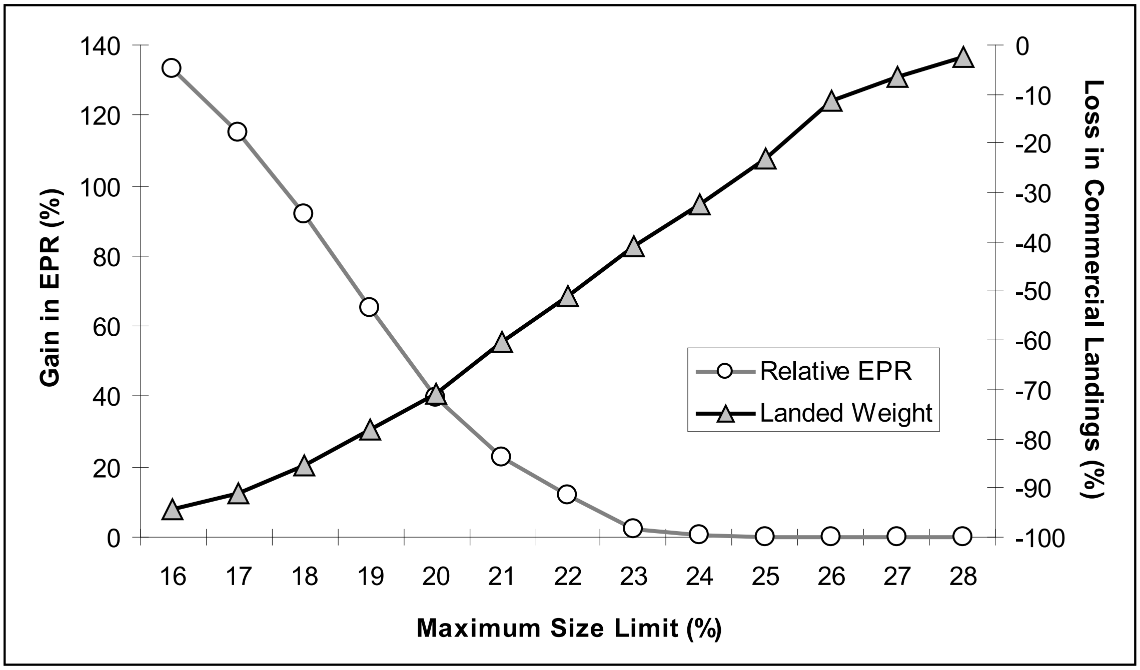

Figure 12. Estimated increase (%) in eggs-per-recruit versus the estimated loss (%) in commercially landed weight for various maximum size limits based on New Jersey's commercial landings.

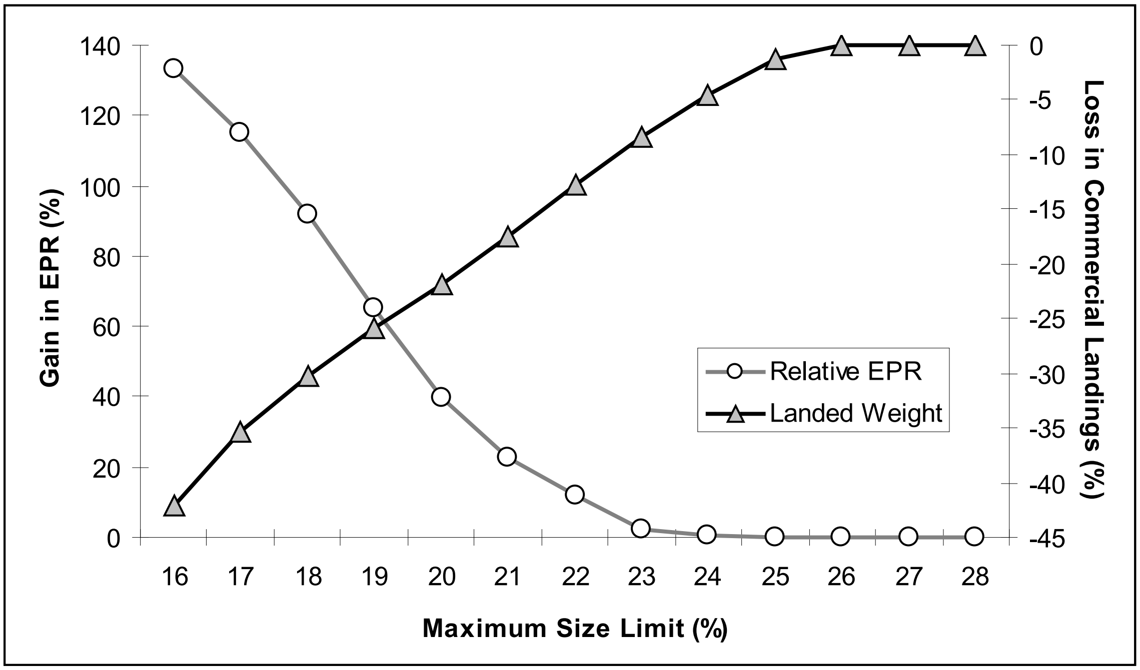

Figure 13. Estimated increase (%) in eggs-per-recruit versus the estimated loss (%) in commercially landed weight for various maximum size limits based on Delaware's commercial landings.

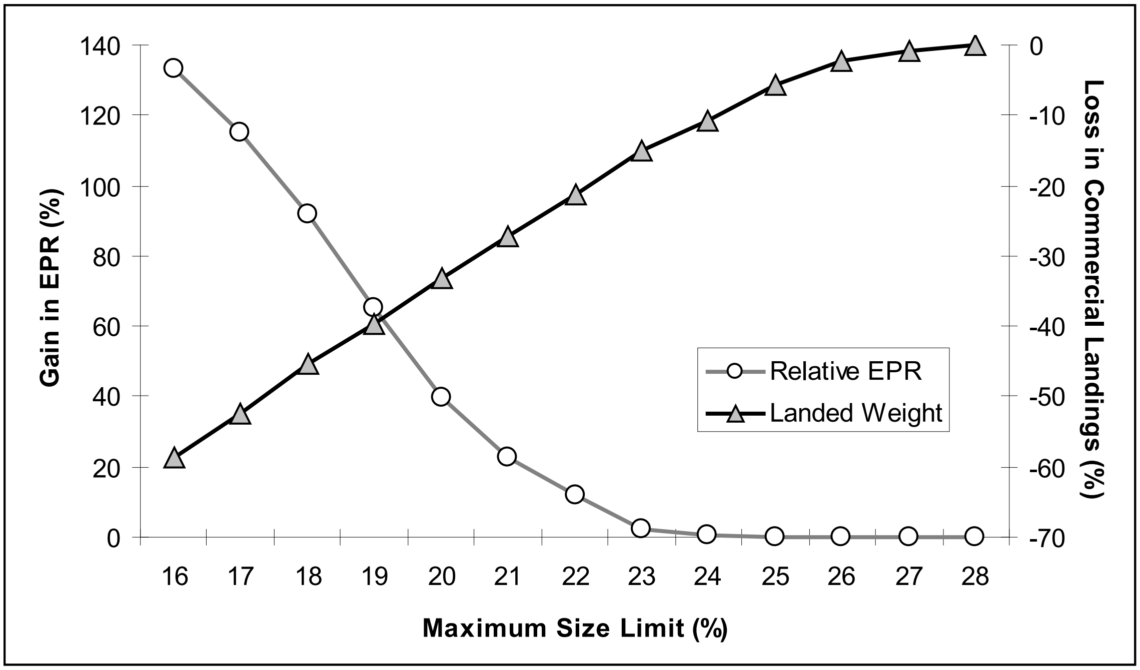

Figure 14. Estimated increase (%) in eggs-per-recruit versus the estimated loss (%) in commercially landed weight for various maximum size limits based on Maryland’s commercial landings.

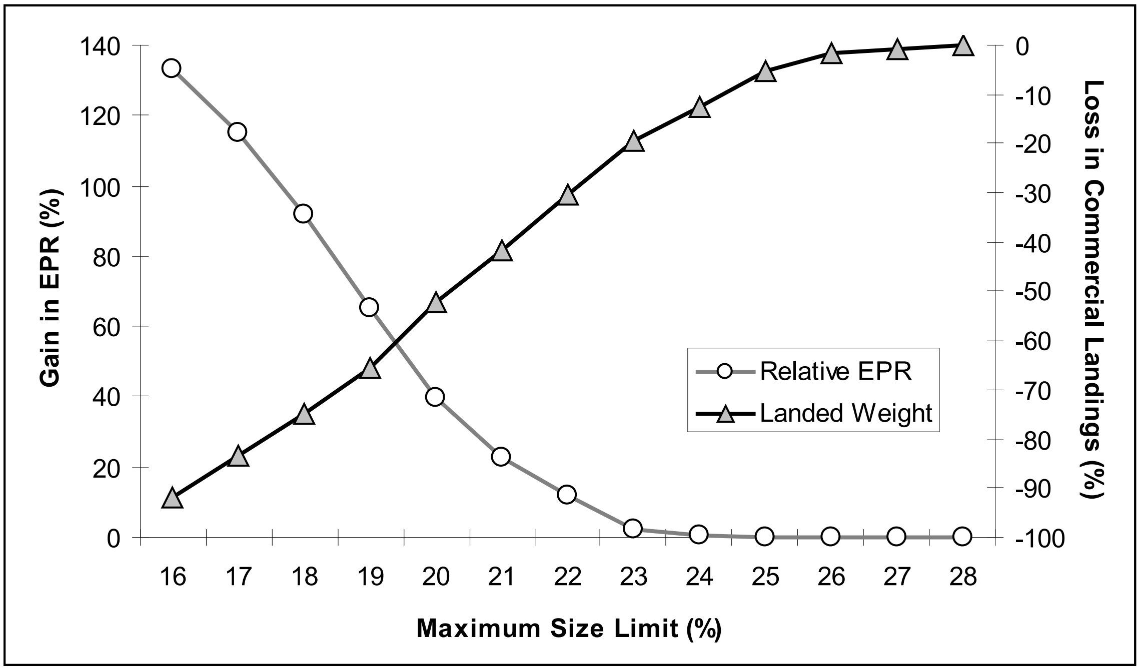

Figure 15. Estimated increase (%) in eggs-per-recruit versus the estimated loss (%) in commercially landed weight for various maximum size limits based on Florida’s commercial landings.

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Atlantic States Marine Fisheries Commission

ADDENDUM III TO THE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL

ASMFC Vision Statement:
Healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015.

Approved August 2013

EXECUTIVE SUMMARY

The Commission's American Eel Management Board initiated the development of Addendum III with the goal of reducing mortality and increasing conservation of American eel stocks across all life stages, in response to the 2012 Benchmark Stock Assessment which found that the American eel population in U.S. waters is depleted. The assessment concluded that the stock is at or near historically low levels due to a combination of historical overfishing, habitat loss and alteration, productivity and food web alterations, predation, turbine mortality, changing climatic and oceanic conditions, toxins and contaminants, and disease.

It is important to emphasize that the 2012 American Eel Stock Assessment was a benchmark or baseline assessment that synthesized all available fishery-dependent and independent data yet was not able to construct eel population targets that could be related to sustainable fishery harvests. This is not an uncommon result of baseline stock assessments. Despite the absence of fishery targets derived from population models, it is clear that high levels of yellow eel fishing occurred in the 1970s and 1980s in response to high prices offered from the export food market. For all coastal regions, peak catches of yellow eels in this period were followed by declining catches in the 1990s and 2000s, with some regions now at historic low levels of harvest. Fishing on all life stages of eels, particularly young-of-the-year and in-river silver eels migrating to the spawning grounds, could be particularly detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled. Given that high catches in the past could have contributed to the current depleted status it is prudent to reduce mortality on all life stages while enhancing and restoring habitat. This approach is further justified in light of the public interest in eel population conservation demonstrated by two recent petitions to list American eel under the Endangered Species Act.

This Addendum establishes new management measures for both the commercial (glass, yellow, and silver) and recreational eel fisheries, as well as implements fishery independent and fishery dependent monitoring requirements. As approved, this Addendum reduces overall mortality of American eel. Given the wide range of public input received during the development of this Addendum, some of the proposed management options originally considered in the public comment draft of Addendum III were transferred to Draft Addendum IV for further development. Draft Addendum IV primarily focuses on management measures for the glass eel fishery and will be considered in Spring 2014.

Management Measures

- *Commercial Glass Eel Fisheries* - Pigmented Eel Tolerance
- *Commercial Yellow Eel Fisheries* – Increase Minimum Size Limit and Gear Restrictions
- *Commercial Silver Eel Fisheries Measures* - Seasonal Closure
- *Recreational Fisheries Measures* – Reduction in Bag Limit with Party/Charter Boat Exemption

TABLE OF CONTENTS

1. INTRODUCTION

The Atlantic States Marine Fisheries Commission (ASMFC) has coordinated interstate management of American eel (*Anguilla rostrata*) from 0-3 miles offshore since 2000. American eel is currently managed under the Interstate Fishery Management Plan (FMP) and Addenda I-III to the FMP. Management authority in the exclusive economic zone (EEZ) from 3-200 miles from shore lies with NOAA Fisheries. The management unit is defined as the portion of the American eel population occurring in the territorial seas and inland waters along the Atlantic coast from Maine to Florida.

1.1. STATEMENT OF THE PROBLEM

The 2012 American Eel Benchmark Stock Assessment found that the coastwide stock has declined in recent decades and the stock was declared depleted. Additionally, the prevalence of significant downward trends in multiple surveys across the coast is a cause for concern. In response the American Eel Management Board (Board) initiated the development of Addendum III with the goal of furthering eel conservation and reducing mortality throughout all life stages. As approved, this addendum reduces overall mortality of American eel. Further conservation measures will be considered in Draft Addendum IV.

1.2. BACKGROUND

American eel inhabit fresh, brackish, and coastal waters along the Atlantic from the southern tip of Greenland to Brazil. American eel eggs are spawned and hatch in the Sargasso Sea. After hatching, leptocephali—the larval stage—are transported by ocean currents to the coasts of North American and the upper portions of South America. After ocean drift, metamorphosis transforms leptocephali into glass eel. In most areas, glass eel enter nearshore waters and begin to migrate up-river, although there have been reports of leptocephali found in freshwater in Florida. Glass eel grow in fresh, brackish, and marine waters, becoming yellow eel. Eel reach the silver eel life stage upon nearing sexual maturity. Silver eel migrate to the Sargasso Sea, completing sexual maturation en route, where they spawn and die.

Yellow eel can metamorphose into a silver eel (termed *silvering*) from three years old and up to twenty-four years old, with the mean age of silvering becoming greater with increasing latitude. Environmental factors (e.g., food availability and temperature) may play a role in the triggering of silvering. Additionally, males and females differ in the size at which they begin to silver. Males begin silvering at a size typically greater than 14 inches and females begin at a size greater than 16-20 inches (Goodwin and Angermeier 2003). Actual metamorphosis is a gradual process occurring in the summer and fall; a drop in temperature appears to trigger the final events of metamorphosis, which lead to migratory movements under the appropriate environmental conditions.

Juvenile eel and silver eel make extensive use of freshwater systems, but they may migrate to and from or remain in brackish and marine waters. Therefore, a comprehensive eel management plan and set of regulations must consider the various unique life stages and the diverse habitats of American eel, in addition to society’s interest and use of this resource.

American eel occupy a significant and unique niche in the Atlantic coastal reaches and tributaries. Historically, American eel were very abundant in East Coast streams, comprising more than 25 percent of the total fish biomass. Eel abundance had declined from historic levels but remained relatively stable until the 1970s. More recently, fishermen, resource managers, and scientists postulated a further decline in abundance based on harvest information and limited assessment data. This resulted in the development of the American Eel FMP.

The goals of the FMP are:

- Protect and enhance the abundance of American eel in inland and territorial waters of the Atlantic states and jurisdictions, and contribute to the viability of the American eel spawning population; and
- Provide for sustainable commercial, subsistence, and recreational fisheries by preventing over-harvest of any eel life stage.

In support of this goal, the following objectives were included in the FMP:

- Improve knowledge of eel utilization at all life stages through mandatory reporting of harvest and effort by commercial fishers and dealers, and enhanced recreational fisheries monitoring.
- Increase understanding of factors affecting eel population dynamics and life history through increased research and monitoring.
- Protect and enhance American eel abundance in all watersheds where eel now occur.
- Where practical, restore American eel to those waters where they had historical abundance but may now be absent by providing access to inland waters for glass eel, elvers, and yellow eel and adequate escapement to the ocean for pre-spawning adult eel.
- Investigate the abundance level of eel at the various life stages necessary to provide adequate forage for natural predators and support ecosystem health and food chain structure.

1.3. STATUS OF THE STOCK

The Benchmark American Eel Stock Assessment was completed and accepted for management use in May 2012. The assessment indicated that the American eel stock has declined in recent decades and the prevalence of significant downward trends in multiple surveys across the coast is cause for concern. The stock is considered depleted, however no overfishing determination can be made at this time based solely on the trend analyses performed. The ASMFC American Eel Technical Committee (TC) and Stock Assessment Subcommittee (SAS) caution that although commercial fishery landings and effort have declined from high levels in the 1970s and 1980s (with the recent exception of the glass eel fishery), current levels of fishing effort may still be too high given the additional stressors affecting the stock such as habitat loss, passage mortality, and disease as well as potentially shifting oceanographic conditions. Fishing on all life stages of eels, particularly young-of-the-year and in-river silver eels migrating to the spawning grounds, could be particularly

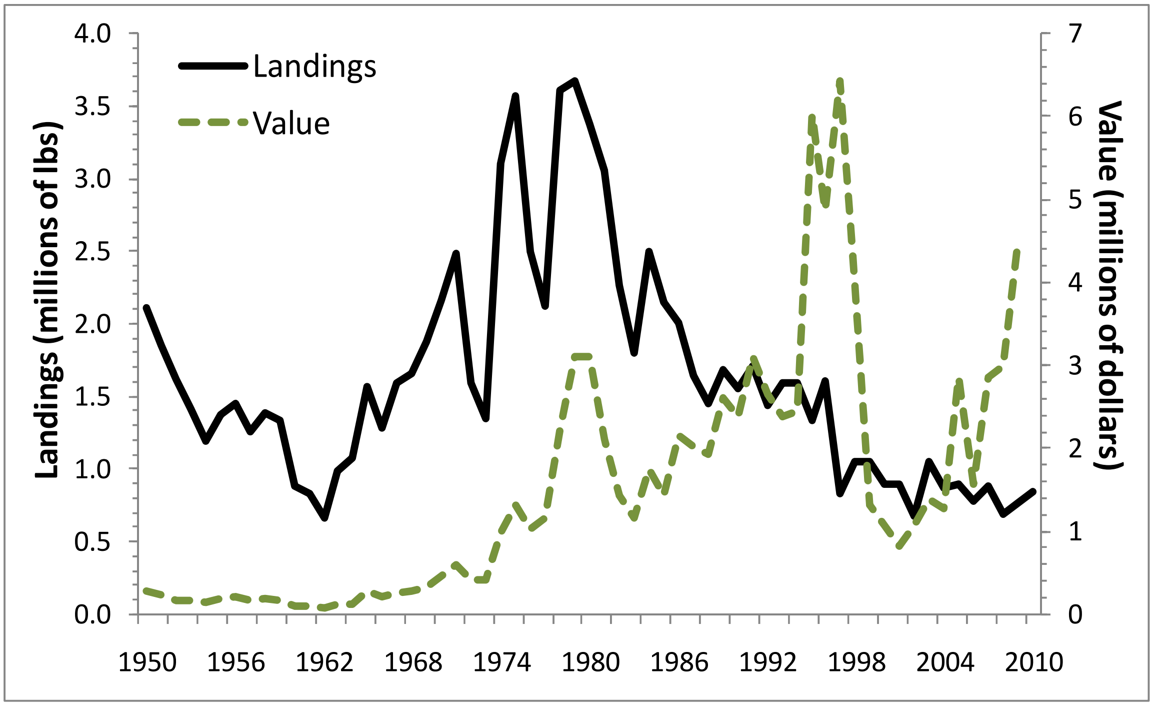

Figure 1. Total commercial landings of American eels and value in 2010 dollars along the U.S. Atlantic Coast, 1950–2010.

detrimental to the stock, especially if other sources of mortality (e.g., turbine mortality, changing oceanographic conditions) cannot be readily controlled.

1.4. STATUS OF THE FISHERY

The American eel fishery primarily targets yellow stage eel. Silver eels are caught during their fall migration as well. Eel pots are the most typical gear used; however, weirs, fyke nets, and other fishing methods are also employed. Glass eel fisheries along the Atlantic coast are prohibited in all states except Maine and South Carolina. In recent years, Maine is the only state reporting significant glass eel and elver harvest. Harvest has increased the last few years as the market price has risen to over $2,000 per pound. Although yellow eels were harvested for food historically, today's fishery sells yellow eels primarily as bait for recreational fisheries. Glass eels are exported to Asia to serve as seed stock for aquaculture facilities.

From 1950 to 2010, U.S. Atlantic coast landings ranged from approximately 664,000 pounds in 1962 to 3.67 million pounds in 1979 (Figure 1). After an initial decline in the 1950s, landings increased to a peak in the 1970s and 1980s in response to higher demand from European food markets. In most regions, landings declined sharply in the 1990s and 2000s following a few years of peak landings. The value of U.S. commercial American eel landings as estimated by NOAA Fisheries has varied from less than a $100,000 (prior to the 1980s) to a peak of $6.4 million in 1997 (Figure 1). Total landings value increased through the 1980s and 1990s, dropped in the late 1990s, and increased again in the 2000s.

2. HABITAT RECOMMENDATIONS

To meet the goal of reducing mortality on all life stages ASMFC should focus efforts on understanding habitat requirements for American eels, engaging the relevant regulatory agencies to increase or improve upstream /downstream eel passage, and encouraging habitat restoration. Specifically the following items are recommended for completion:

1. Development of quantifiable eel habitat enhancement goals through the creation of a coastwide eel habitat GIS database. The goal of the database would be the generation of coastwide, regional, state, and watershed maps that would quantify the amount of available habitat relative to historical habitat and identify major barriers to eel migration. This information would allow the ASMFC to prioritize eel habitat enhancement programs at coastwide, regional, and state scales. Efforts should be coordinated with existing GIS efforts already underway in Canada (see: http://www.dfo-mpo.gc.ca/Library/345546.pdf). Potential funding and coordination with the Atlantic Fish Habitat Partnership should be considered. This project is considered a high priority item and should be completed either prior to the start of the next benchmark stock assessment or in conjunction with the stock assessment.

2. The TC should work with other appropriate ASMFC committees to develop materials to support states or jurisdictions interested in making recommendations to the Federal Energy Regulatory Commission (FERC) for upstream and downstream fish passage provisions for American eels in the hydropower licensing and relicensing process. A list of FERC requirements in coordinating with the states in the hydropower licensing and relicensing process is included in Appendix I.

3. Work with states and jurisdictions to develop a list of non-FERC licensed dams and other impoundments which impact eel movements and migration. The Nature Conservancy recently completed an online, interactive inventory of dams from Maine to Virginia (see: The Northeast Aquatic Connectivity and Assessment of Dams) which could be adapted to meet this goal. An evaluation should be conducted on each general type of impoundment to assess the potential for eel passage without assistance (i.e. no eel passage constructed) or determine what type of eel passage for each type of impoundment would be most beneficial for all, or specific, life stages. The recommendations from the workshop proceedings (in preparation) from the ASMFC American Eel Passage Workshop held in Gloucester, MA, (March 2011) should be a useful document to assist in the completion of this task. Additional recommendations on eel passage are found in Appendix II.

4. Based on #1 – 3, all states and jurisdictions should develop a timeline and target for 1) the amount of habitat to open up through creation of fish passage or dam removal, where feasible and/or 2) the amount of habitat to enhance to increase survival for all, or specific, life stages.

5. The TC should assess and provide recommendations related to other potential impacts caused by water supply and withdrawal operations, water diversions, and agricultural water use.

6. The TC and SAS should increase coordination with the ASMFC Fish Passage, Habitat, and FERC Guidance Committees. The state marine fisheries agencies should also encourage increased communication and collaboration with their inland fisheries agencies counterparts where applicable. The Commission should also continue the development of a Memorandum of Understanding between the Great Lakes Fisheries Commission, U.S. Fish and Wildlife Service, and NOAA Fisheries in order to reduce mortality on eels throughout their range, as well as improving access to suitable habitat.

3. MONITORING PROGRAM

Monitoring programs should be implemented to maximize the collection of the most useful data for monitoring the annual health of the stock, as well as to provide both statistically valid and scientifically rigorous information for stock assessment analysis. Additionally, the design of a new program will need to take into consideration the priorities of state monitoring programs as well as available funding and personnel.

3.1. FISHERIES INDEPENDENT SURVEYS

The 2012 American Eel Benchmark Stock Assessment made the following recommendations with regard to coastwide fisheries independent sampling:

1. Recommend states collect biological information by life stage including length, weight, age, and sex of eels caught in fishery-independent sampling programs; at a minimum, length samples should be routinely collected from fishery-independent or fisheries-dependant surveys.
2. Encourage states to implement surveys that directly target and measure abundance of yellow- and silver-stage American eels, especially in states where few targeted eel surveys are conducted.
3. A coast-wide sampling program for yellow and silver American eels should be developed using standardized and statistically robust methodologies.
4. Continue the ASMFC-mandated young-of-the-year surveys; these surveys could be particularly valuable as an early warning signal of recruitment failure.

3.1.1. Annual Young-of-Year Abundance Survey

The requirements of the annual young-of-the-year survey will remain as specified under Section 3.1.1 of the FMP.

3.1.2. Annual Yellow Eel Survey

States and jurisdictions currently conducting yellow eel surveys, as specified in Table 1, will be required to maintain these surveys. For those surveys that are targeting another species (either as required by separate ASMFC FMP or at the discretion of the state) and collects information on bycaught American eels, if the state discontinues the survey it is recommended that a similar survey be implemented, as possible, to continue data collection. Under this Addendum collection of data on bycaught eels is not a compliance requirement. As funds and/or personnel become available it is recommended that states/jurisdictions consider implementing additional yellow eel monitoring programs.

Table 1. Fisheries Independent Monitoring for American Eel

State	System	Monitoring Program	Targeted Life Stage				Information Collected
			G	E	Y	S	
Maine	West Harbor Pond	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Sebasticook River (Benton Falls)	Irish Elver Ramp^^{A}		X	X		length, weight, count, EV
New Hampshire	Lamprey River	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Squamscott, Oyster, and Winnicut	Fyke net*^			X		length, weight, count, EV
Massachusetts	Acushnet, Parker, and Jones Rivers	Sheldon/Irish Elver Trap*^	X				count, length, weight, pigment stage, EV
	6 Coastal Rivers	Bycatch survey*^			X		length, weight, count, EV
Rhode Island	Gilbert Stuart	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Annaquatucket River	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Narragansett Bay	Trawl Survey^			X		length, weight, count, EV
	Narragansett Bay	Seine Survey^			X		length, weight, count, EV
Connecticut	Ingham Hill	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Farmill River	Electrofishing survey ^^{A}			X		length, weight, count, EV
New York	Carmans River	Fyke net^	X				count, length, weight, pigment stage, EV
	Hudson River	Striped Bass Survey*^^{A}		X	X		length, weight, count, EV
	Hudson River	Alosine Survey*^^{A}		X	X		length, weight, count, EV
	Western Long Island	Seine Survey*^		X	X		length, count, EV
New Jersey	Patcong Creek	Fyke net^	X				count, length, weight, pigment stage, EV
	tributary of Delaware River/Bay	River Herring electrofishing survey*			X		length, weight, count, EV
	Delaware River	Striped Bass Seine Survey*^^{A}			X		length, weight, count, EV
Pennsylvania	non-tidal DE River	Small mouth bass survey^		X	X		count
Delaware	Millsboro	Fyke net^	X				count, length, weight, pigment stage, EV
	Delaware River	Trawl survey ^^{A}		X	X		length, weight, count, EV

Table 1. Fisheries Independent Monitoring for American Eel (continued)

State	System	Monitoring Program	Targeted Life Stage				Information Collected
			G	E	Y	S	
Maryland	Turville Creek	Irish Elver Ramp^^{A}	X				count, length, weight, pigment stage, EV
	Bishopville	Irish Elver Ramp	X				count, length, weight, pigment stage, EV
	Sassafrass River	Pot Survey^^{A}			X		length, weight, count, EV
	Chesapeake Bay	Juvenile Striped Bass Survey*^^{A}			X		length, weight, count, EV
	Corsica River	Trap Survey^^{A}				X	length, weight, count, EV
PRFC	Clarks Millpond (Coan R.)	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Gardys Millpond (Yeocomico R.)	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
DC	Potomac River	Electrofishing survey^			X		length, weight, count, EV
	Potomac River	Pot Survey^			X		length, weight, count, EV
Virginia	James	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	York	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Rappahannock	Irish Elver Ramp^	X				count, length, weight, pigment stage, EV
	Inland Waters	Electrofishing survey**^^{A}			X		length, weight, count, EV
North Carolina	Beaufort Bridge	Net Survey^**	X				count, length, weight, pigment stage, EV
	Estuarine Trawl Survey	Trawl Survey^^{A}			X		length, count, EV
South Carolina	Goose Creek	Fyke net^	X				count, length, weight, pigment stage, EV
	Lower Edisto, Combahee, Ashley, Cooper Rivers and Upper Winyah Bay	Red Drum electrofishing survey*^^{A}			X		length, weight, count, EV
	PeeDee, Edisto, Savannah Rives	Juvenile Am. Shad electrofishing survey*^			X	X	length, weight, count, EV
Georgia	Altamaha	Pot Survey			X		length, weight, count, EV
Florida	Guana River Dam	Dip Net Survey^	X				count, length, weight, pigment stage, EV

*Survey is primarily targeting another species and collects information on American eels caught as bycatch. The survey is conducted either as required by separate ASMFC FMP or at the discretion of the state. Under this addendum collection of data on bycaught eels is not a compliance requirement. However, if the state discontinues the survey it is recommended that a similar survey be implemented, as possible, to continue data collection.

** Survey is currently conducted by the inland or freshwater division in the state. G = Glass Eel E = Elver Eel Y = Yellow Eel S = Silver Eel

^ Survey currently conducted. A = Survey used in 2012 American Eel Stock Assessment. EV = Environmental Variables, as specified under Section 3.1.1 of the FMP

3.1.3. Annual Silver Eel Survey

States and jurisdictions currently conducting silver eel surveys, as specified in Table 1, will be required to maintain these surveys. For those surveys that are targeting another species (either as required by separate ASMFC FMP or at the discretion of the state) and collects information on bycaught American eels, if the state discontinues the survey it is recommended that a similar survey be implemented, as possible, to continue data collection. Under this addendum collection of data on bycaught eels is not a compliance requirement. As funds and/or personnel become available it is recommended that states/jurisdictions consider implementing additional silver eel monitoring programs.

3.1.4. Multiple Life Stages Survey

Where possible, the TC recommends the identification of areas where multiple life stage surveys can be conducted. Ideally the survey would target glass eel immigration and silver/yellow eel emigration in the same system in order to track recruitment, age, growth, survival, and mortality.

3.2. FISHERY DEPENDENT SURVEYS

To increase accuracy of reporting, states and jurisdictions with a commercial yellow eel fishery will be required to implement a trip level reporting system for both dealer and harvester reporting. Dealer and harvester landing catches must submit reports to the state of landing monthly or more frequently, if possible. This includes reporting on directed commercial harvest, by trip, (pounds landed by life stage, gear type, and catch per unit effort (CPUE)). Cross referencing between dealer and fishery trip level reporting should be conducted to ensure accuracy. States with more conservative reporting requirements in place will be required to maintain them.

Additionally, states must continue collect biological data, per Section 3.4.1 of the FMP, from a representative sub-sample of the commercial catch, if available, to evaluate sex and age structure (for yellow/silver eels), length and weight. States must also continue report on the estimated percent of harvest going to food versus bait.

States and jurisdictions may continue to petition the Board for *de minimis* status (met if commercial landings are less than 1% of the coastwide total), which exempts them from additional fishery dependent monitoring requirements, per Section 4.4.2 of the FMP.

The ASMFC American Eel Plan Development Team (PDT) and TC have discussed the need to improve harvest data for eel caught under commercial permits and kept for personal use and not sold. There is concern this practice may be underreported especially in New England where some commercial permit holders save eels as bait for the commercial striped bass fishery. Under this Addendum states and jurisdictions are recommended to implement strategies within their reporting system to recover data on eels harvested for personal use. This could be accomplished by updating current reporting criteria or implementing a special-use permit. A related reporting gap likely exists for recreational eel potting, however the coast-wide magnitude is expected to be lower. Where feasible, states and jurisdiction are encourage to also investigate strategies for improving recreational harvest data on eels kept for personal use.

Additionally, this Addendum recommends that the state marine agencies work with their state inland counterparts, where applicable, to standardize reporting of trip-level landings and effort data that occur in inland waters on diadromous populations of eels.

4. MANAGEMENT PROGRAM

This Addendum establishes new management measures for both the commercial (glass, yellow, and silver) and recreational eel fisheries. Given the wide range of proposed management measures and public input received during the development of this Addendum, some of the proposed management options originally considered in the public comment draft of Addendum III were transferred to Draft Addendum IV in order to be further developed. Draft Addendum IV primarily focuses on management measures for the glass eel fishery and will be considered in Spring 2014.

4.1. COMMERCIAL FISHERY MANAGEMENT MEASURES

These regulations replace Section 4.2.1 of the FMP. States/jurisdictions shall maintain existing or more conservative American eel commercial fishery regulations, unless otherwise approved by the American Eel Management Board. The implemented provisions will be considered a compliance requirement and are effective as specified under Section 5.0. Management measures also include all mandatory monitoring and annual reporting requirements as described in Section 3.0 of this addendum.

4.1.1. Glass Eel Fisheries

The following measures apply to the glass eel fisheries that currently operate in Maine and South Carolina. For all other jurisdictions, states are required to maintain existing or more conservative measures at the time of implementation of the American Eel FMP to control the development glass eel fisheries. The development of any future glass eel fisheries would be subject to the following measures, unless otherwise specified by the Board.

PIGMENTED EEL RESTRICTIONS

An increase in harvest of pigmented eels has been observed in recent years during the glass eel fishery. Glass eels generally become pigmented as the season progresses and water temperatures increase, although there may be other factors that affect this pigmenting process (Haro and Krueger 1988). The pigmentation provides disruptive coloration and countershading for the eels, which presumably reduces predation and increases survivorship. While the glass eel fishery is a traditional fishery, the pigmented eel fishery represents the development of a new fishery. It has been observed that catches are predominately either glass eels or pigmented eels (i.e. the catch is not a mixture of both pigmented and glass eels).

Therefore, under this Addendum, for states with a commercial glass eel fishery, only a small tolerance (maximum of 25 pigmented eels per pound of glass eel catch) of pigmented eels will be allowed. In order to meet this requirement, it is recommended that states implement the use of a 1/8 inch non-stretchable mesh to grade all catch immediately upon harvesting.

States may propose alternative restrictions to meet the goal of minimizing the development of a pigmented eel fishery, which would require review by the TC and approval by the Board. It is also recommended that all catch be graded on the boat or streamside and that any bycatch is immediately returned to the waters where the fish were harvested.

4.1.2. Yellow Eel Fisheries

Yellow eel fisheries currently operate in all states with the exception of Pennsylvania and the District of Columbia. The following measures apply to all current yellow eel fisheries. The development of any future yellow eel fisheries would be subject to the following measures, unless otherwise specified by the Board.

MINIMUM SIZE AND MESH REQUIREMENTS

It is generally accepted that American eel in the northern portion of the species' range are larger than eel in the southern end of the range. However, there is not enough information at this time to develop regional or state specific maximum sizes for the coast. Nonetheless, there is growing concern about the development of fisheries on small yellow eels and an increase in the minimum size is a means to prevent this fishery from developing further. The benefit of effective gear restrictions is smaller eels are not landed, thus eliminating the need for harvesters to handle these fish or enforcement having to measure fish. No gear requirements are sought to exclude larger eels from pots at this time because only a low number of silver eels are caught in pot fisheries. Gear restrictions that are instituted should be monitored for effectiveness.

States and jurisdictions are required to adopt a nine (9) inch minimum size limit for all yellow eel fisheries. Harvesters are required to sort their catch and discard eels smaller than the size limit.

States and jurisdictions are required to implement a ½ by ½ minimum on the mesh size used in commercial yellow eel pots.

States may allow, for up to three years starting January 1, 2014 , the use of a 4 by 4 inch escape panel constructed of a mesh size of at least ½ by ½ inch mesh in order to reduce the financial burden of gear changes on the fishery.

4.1.3. Silver Eel Fisheries

SEASONAL CLOSURE RESTRICTIONS

States and jurisdictions are required to implement no take of eels from September 1st through December 31st from any gear type other than baited traps/pots or spears (e.g. fyke nets, pound nets, and weirs). These gears may still be fished, however retention of eels is prohibited. A state or jurisdiction may request an alternative time frame for the closure if it can demonstrate the proposed closure dates encompass the silver eel outmigration period. Any requests will be reviewed by the TC and submitted to the Board for approval.

The Delaware River and its tributaries within New York are exempt from this requirement. This exemption will sunset one year from the date of implementation (implementation date is January 1, 2014). If alternative management measures are not implemented by January 1, 2015, then the requirements under this section will apply. Alternative management measures for the Delaware River and its tributaries within New York will be considered under Draft Addendum IV.

4.2 RECREATIONAL FISHERY MANAGEMENT MEASURES

These regulations replace Section 4.1 of the FMP. The implemented provisions will be considered a compliance requirement and are effective as specified under Section 5.0.

RECREATIONAL MINIMUM SIZE

In order to minimize the chance of excessive recreational harvest, as well as circumvention of commercial eel regulations, the ASMFC member states/jurisdictions shall establish uniform possession limits for recreational fisheries. States and jurisdictions are required to adopt a nine (9) inch minimum size limit for all recreational fisheries.

RECREATIONAL BAG LIMIT

Given the interest to have all fishery sectors contribute to conservation measures under Addendum III all states and jurisdictions are required to implement a daily recreational bag limit of 25 fish per day per angler.

PARTY/CHARTER (FOR-HIRE) EXEMPTION

Crew and captain involved in party/charter (for-hire) employment on party/charter (for-hire) activities are exempt from recreational bag limit reduction. Crew members involved in for-hire employment are allowed to maintain the current 50 fish per day bag limit for bait purposes during fishing, as specified under the American Eel FMP.

5. IMPLEMENTATION SCHEDULE

The measures contained in Section 4.0 will be effective on January 1, 2014.

6. LITERATURE CITED

Goodwin, K.R., and P.L. Angermeier. 2003. Demographic characteristics of American eel in the Potomac River drainage, Virginia. Transactions of the American Fisheries Society 132(3):524–535.

Haro, A.J., and W.H. Krueger. 1988. Pigmentation, size, and migration of elvers (*Anguilla rostrata* (Lesueur)) in a coastal Rhode Island stream. Canadian Journal of Zoology 66(11):2528–2533.

Hutchinson, 1997. Evaluation under commercial fishing conditions the effectiveness of the mandated eel pot grading panel on the North Carolina American eel. North Carolina Department of Environment, Health, and Natural Resources, Division of Marine Fisheries Grant Program. Grant No. FRG-95-105. Contract No. M-6028.

Appendix IV. Current State Fish Passage Considerations

FERC Guidelines

Under section 401(a)(1) of the Clean Water Act (CWA), the FERC may not issue a license for a hydroelectric project unless the State water quality certifying agency has issued water quality certification for the project or has waived certification. Certification (or waiver) is required in connection with any application for a Federal license or permit to conduct an activity which may result in a discharge into U.S. waters. Any conditions of the certification become conditions of the license.

Section 18 of the Federal Power Act states that the Commission shall require construction, maintenance, and operation by a licensee of such fishways as the Secretaries of Commerce or the Interior may prescribe. The Commission's policy is to reserve such authority in a license upon the request of either designated Secretary.

Pursuant to section 10(j)(1) of the FPA,the Commission, when issuing a license, includes conditions based on the recommendations of Federal and State fish and wildlife agencies submitted pursuant to the Fish and Wildlife Coordination Act, for the protection and enhancement of fish and wildlife and their habitat affected by the project.
The Commission makes a preliminary determination of whether the recommendations are consistent with the FPA or other applicable law. If there is a preliminary inconsistency determination, the agency in question is invited to meet with the Commission staff to try to resolve the matter prior to action on the license application

For example:
On August 31, 1999, Northeast Generation Services Company (NGS)1 filed an application for a single new license, pursuant to sections 4(e) and 15 of the Federal Power Act (FPA),2 for the continued operation and maintenance of the existing 105.9-megawatt (MW) Housatonic Project. The Housatonic River flows southward 149 miles through western Massachusetts and Connecticut before reaching Long Island Sound. The watershed drains some 2,000 square miles consisting of rugged terrain in the north, and rolling hills and flat stretches
of marshland in the south.

FWS made 28 recommendations in this proceeding, of which the Commission staff preliminarily determined that five were not consistent with the FPA or other applicable law. Based on comments filed by Interior and others on the Draft EIS, and
additional staff analysis, it was determined that three of the five recommendations are not within the scope of section 10(j), and the Final EIS recommends that they be included in the license. The two remaining inconsistencies are Interior’s recommendations to operate the Falls Village and Bulls Bridge developments in a run-of-river mode year-round. The EIS found that year-round run-of-river operation would disadvantage recreational users and businesses associated with whitewater boating, and would cost NGS about $108,000 in lost generation. The EIS recommended that these developments be operated in run-of-river mode during the spring, and in peaking mode from July through March to benefit the whitewater-

boating community and reduce economic impacts to NGS. This issue was however mooted by Connecticut DEP's water quality certification, which requires run-of-river operation at these developments year round.

The Licensee shall, in a manner approved by the U.S. Fish and Wildlife Service (Service) and the Department, design, construct, operate, maintain and monitor the effectiveness of upstream and downstream American eel passage facilities. The Licensee shall implement the American eel passage effectiveness monitoring plan when the facilities are place in operation. The Licensee shall, in a manner approved by the Service and the Department, design, construct, operate, maintain and monitor the effectiveness of upstream and downstream anadromous fish passage facilities that are capable of excluding the passage of sea lamprey. The Licensee shall implement the anadromous fish passage effectiveness-monitoring plan when the facilities are placed in operation.

The Licensee shall, in a manner approved by the Service and the Department, develop a plan to assess the impact on the littoral-zone community due to impoundment fluctuations associated with normal operations (excluding emergency or maintenance draw downs). The assessment will analyze impacts on aquatic resources such as fish, mussels, wetlands and wildlife that inhabit the littoral-zone of Lake Lillinonah. The results of the assessment will be presented in a report and submitted to the Department and the Service. If the Department and the Service determine that significant adverse impacts occur during normal operations, the Licensee will implement corrective actions to mitigate the impacts.

Maine

Permitting Agency: Maine Dept of Environmental Protection (http://www.mainelegislature.org/legis/statutes/38/title38ch5sec0.html)

*Initial Approval: (*38 §636. Approval criteria)

The department shall make a written finding of fact with respect to the nature and magnitude of the impact of the project on each of the considerations under this subsection, and a written explanation of their use of these findings in reaching their decision.

B. Whether the project will result in significant benefit or harm to fish and wildlife resources. In making its determination, the department shall consider other existing uses of the watershed and fisheries management plans adopted by the Department of Inland Fisheries and Wildlife and the Department of Marine Resources

D. Whether the project will result in significant benefit or harm to the public rights of access to and use of the surface waters of the State for navigation, fishing, fowling, recreation and other lawful public uses

Minimum Flow Requirements if Hearing is Sought: (38 §840. Establishment of water levels)

4. Evidence. At the hearing, the commissioner shall solicit and receive testimony, as provided by Title 5, section 9057, for the purpose of establishing a water level regime and, if applicable, minimum flow requirements for the body of water. The testimony is limited to:
   - A. The water levels necessary to maintain the public rights of access to and use of the water for navigation, fishing, fowling, recreation and other lawful public uses;
   - C. The water levels and minimum flow requirements necessary for the maintenance of fish and wildlife habitat and water quality

New Hampshire
Permitting Agency: NH Dept of Env. Services
http://des.nh.gov/organization/divisions/water/dam/permit_dam.htm
No guidelines for fish passageways: See
http://www.gencourt.state.nh.us/rsa/html/NHTOC/NHTOC-L-482.htm
Statute regarding inspection and erection of dams: See
http://www.gencourt.state.nh.us/rsa/html/L/482/482-9.htm

Massachusetts
Massachusetts
Permitting Agency: Massachusetts Division of Marine Fisheries
Authorization and management of fish passage for sea-run fish: M.G.L Chapter 130, Sections 1 and 19.
Fishway Construction Permit: 322 CMR Sections 7.01 (4(f)) and (14(m)).

Rhode Island
Permitting Agency: Dept. of Env. Management
http://www.dem.ri.gov/
Impact Minimization: Rhode Island's Freshwater Wetlands Act (R.I. Gen. Laws Section 2-1-18 et seq.) and Water Pollution Act (R.I. Gen. Laws Section 46-12-1 et seq.) require the Director to protect freshwater wetland values and water quality, respectively. It is important for the dam owner to recognize the Director's responsibilities under these laws and to plan his/her repair projects to minimize any negative impacts to freshwater wetlands and water quality values. In particular, the dam owner must:

> (A) Minimize the impacts from lowering the water elevation in a reservoir during a repair project, such as by installing a temporary cofferdam. This is necessary to reduce detrimental impacts to fish and wildlife associated with the wetland environment and to reduce loss of aquatic vegetation that serves as wildlife habitat. In the event that a dam owner is unable to install controls to maintain water in the reservoir to assist in protecting fish and wildlife habitat, the dam owner must specifically inform the Director of this situation and document in writing why water is not proposed to be maintained upstream of the dam during the repair activity. Efforts must be made to avoid drawdowns between April 15 to July 1, and to avoid significant drawdowns between October 15 and March 15.
> (http://www.dem.ri.gov/pubs/regs/regs//compinsp/dams07.pdf)

Connecticut
Permitting Agency: Dept. of Energy and Env. Protection
www.ct.gov/deep
Permits for Construction: (b) The commissioner or his representative, engineer or consultant shall determine the impact of the construction work on the environment, on the safety of persons and property and on the inland wetlands and watercourses of the state in accordance with the provisions of sections 22a-36 to 22a-45, inclusive, and shall further determine the need for a fishway in accordance with the provisions of section 26-136, and shall examine the documents and inspect the site, and, upon approval thereof, the commissioner shall issue

a permit authorizing the proposed construction work under such conditions as the commissioner may direct.

New York
Permitting Agency: Dept of Env. Conservation
www.dec.**ny**.gov/
§608.8 Standards
The basis for the issuance or modification of a permit will be a determination that the proposal is in the public interest, in that:

(c) the proposal will not cause unreasonable, uncontrolled or unnecessary damage to the natural resources of the state, including soil, forests, water, fish, shellfish, crustaceans and aquatic and land-related environment. (http://www.dec.ny.gov/regs/4438.html)
For existing dams, when they are inspected: Conditions causing or requiring temporary or permanent adjustment of the pool level include: Requirements for recreation, hydropower, or water fowl and fish management (p. 27, http://www.dec.ny.gov/docs/water_pdf/damguideman.pdf)

Pennsylvania
Permitting Agency: Dept. of Env. Protection, Bureau of Waterways and Engineering
http://www.portal.state.pa.us/portal/server.pt/community/waterways_engineering/10499
Requirements for Permit: (d) An application for a permit shall be accompanied by information, maps, plans, specifications, design analyses, test reports and other data specifically required under this chapter and additional information as required by the Department to determine compliance with this chapter.

(x) *Impacts analysis.* A detailed analysis of the potential impacts, to the extent applicable, of the proposed project on water quality, stream flow, fish and wildlife, aquatic habitat, Federal and State forests, parks, recreation, instream and downstream water uses, prime farmlands, areas or structures of historic significance, streams which are identified candidates for or are included within the Federal or State wild and scenic river systems and other relevant significant environmental factors. If a project will affect wetlands the project description shall also include:

(http://www.pacode.com/secure/data/025/chapter105/chap105toc.html)
Reviewing Permit: (b) In reviewing a permit application under this chapter, the Department will use the following factors to make a determination of impact:

(4) The effect of the dam, water obstruction or encroachment on regimen and ecology of the watercourse or other body of water, water quality, stream flow, fish and wildlife, aquatic habitat, instream and downstream uses and other significant environmental factors.

(5) The impacts of the dam, water obstruction or encroachment on nearby natural areas, wildlife sanctuaries, public water supplies, other geographical or physical features including cultural, archaeological and historical landmarks, National wildlife refuges, National natural landmarks, National, State or local parks or recreation areas or National, State or local historical sites
§ 105.121. Fishways.

Upon the request of the Fish and Boat Commission, the permittee shall install and maintain chutes, slopes, fishways, gates or other devices that the Fish and Boat Commission may require under 30 Pa.C.S. § § 3501—3505.

§ 105.244. Protection of fish life.

A low flow channel and habitat improvement device will be required when, in the opinion of the Fish Commission, it is necessary to provide satisfactory channel for maintenance of fish.

New Jersey

Permitting Agency: Dept. of Env. Protection

http://www.state.nj.us/dep/

For new dams: (d) No person may construct a dam in any waterway of this state which is a runway for migratory fish, without installing a fish ladder or other approved structure to permit the fish to pass the dam in either direction (see N.J.S.A. 23:5-29.1).

1. This provision is applicable to dams of any size.
2. The Department will determine whether a stream is currently a runway for migratory fish, during the review of the dam permit application. Applicants should consult the Division of Fish and Wildlife in this matter prior to finalizing the application.

(http://www.nj.gov/dep/damsafety/docs/standard.pdf)

Delaware

Permitting Agency: Dept. of Natural Resources and Environmental Control

http://www.dnrec.delaware.gov

No guidelines for new dams or fish passageways

Maryland

Permitting Agency: Dept of the Environment

http://www.mde.state.md.us

For existing dams: 5. Pool levels are sometimes adjusted for recreation, hydropower, or waterfowl and fish management. (p. 47, http://www.mde.state.md.us/programs/Water/DamSafety/GuidelinesandPolicies/Documents/www.mde.state.md.us/assets/document/damsafety/MD%20Dam%20Safety%20Manual%201996.pdf)

Dam in a Recreational Park: The Lake Waterford Dam was repaired in 1993. A new principal pipe spillway along with a concrete ogee spillway were installed to safely pass the 100-year storm. In addition a cement bentonite slurry wall was installed and a fish passage was constructed to access the upstream spawning areas.

No guidelines for new dams or fish passageways

Virginia

Permitting Agency: Dept. of Conservation and Recreation, Virginia Soil and Water Conservation Board

http://www.dcr.virginia.gov/stormwater_management/index.shtml

No guidelines for new dams or fish passageways: See http://www.dcr.virginia.gov/dam_safety_and_floodplains/documents/dsregs.pdf

North Carolina
Permitting Agency: Dept. of Env.and Natural Resources
http://portal.ncdenr.org
For existing dams: 5. Pool levels are sometimes adjusted for recreation, hydropower, or waterfowl and fish management.
(http://portal.ncdenr.org/c/document_library/get_file?uuid=6968a202-c971-40ef-9efb-40883a9f9bd8&groupId=38334)
No other guidelines for new dams or specifically concerning fish passageway.

South Carolina
Permitting Agency: Dept. of Health and Env. Control, http://www.scdhec.gov/
No guidelines for new dams or fish passageways.

Georgia
Permitting Agency: Dept of Natural Resources, http://www.gadnr.org/
No guidelines for new dams or fish passageways.

Florida
Permitting Agency: Dept. of Env. Protection -
http://www.dep.state.fl.us/water/mines/damsafe.htm
No guidelines for new dams or fish passageways.

Appendix II – Fish Passage Recommendations for American eel

The fragmentation of habitat and blockage of upstream and downstream migrations is a major area of concern for American eels. Traditional fish passage is not effective for upstream migration of juvenile American eels, presumably due to velocity barriers. While low-head weir and pool fishways may allow juvenile eel passage, it is likely that most Denil and Alaskan Steeppass ladders are not passable. Eel Passage structures often vary in design via substrate type, slope and length. However, eel passage is relatively new practice in the US, and additional investigation is needed on standard design criteria and quantitative metrics of passage success. Eel passage structures should only be deployed after evaluating the potential for eels to pass the present impediment and the possibility of removing the impediment. If an eel passageway is necessary, the design should initially focus on the size range of eels below the impediment and the specific location where an eel pass can suitably attract eels. With this information, designs can progress towards selecting water supply for the eel pass, the choice of having a monitoring tank, and structural dimensions for the eel pass and associated hardware. Recently some strides have been made in upstream eel passage structures (see ASMFC 2011 American Eel Passage Workshop Proceedings, in prep.). With these considerations, the PDT recommends that each jurisdiction actively seeks opportunities to improve upstream eel passage through obstruction removal and deployment of eel passage structures.

Downstream passage of out migrating eels is seen as more difficult than upstream migrations issues, as the results of passage through a hydroelectric project can often be mortality of mature, fecund individuals. Downstream mortality rate is often highly variable and is depended on dam configuration, turbine type, and operational conditions. Generally turbine strikes positively relate to eel length, putting larger female silver eels at particular high risk. Light barriers, louver screens, high flow bypass and generation shut downs during predicted migration windows have all shown promise but there are few quantitative studies showing the level of effectiveness. Important gains in eel survival and recruitment could be realized through widespread reductions in downstream passage mortality of silver eels. The PDT recommends that each jurisdiction identify opportunities to work within the FERC review process and with non-FERC dam owners to improve downstream eel passage.

Three-Dimensional Movement of Silver-Phase American Eels in the Forebay of a Small Hydroelectric Facility

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American Fisheries Society Symposium 58:277–291, 2009

Three-Dimensional Movement of Silver-Phase American Eels in the Forebay of a Small Hydroelectric Facility

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Abstract.—Declines in the population of the American eel, *Anguilla rostrata*, along the northwestern Atlantic have stimulated resource managers to consider the impact of hydroelectric facilities on silver-phase eels as they migrate downstream to the sea. During the fall of 2002, we investigated the movement of migrant eels passing downstream of a small hydroelectric facility on the Connecticut River (Massachusetts). We used three-dimensional acoustic telemetry to monitor fine-scale movement of telemetered silver eels in the forebay (the first 100 m of area directly upstream of the dam). Eel movements were tracked approximately every three seconds, and individual swimming pathways were reconstructed to compare the three-dimensional results with biotelemetry methods previously used at this site; conventional telemetry systems included radio, PIT, and acoustic telemetry. We found that three-dimensional acoustic telemetry provided the necessary fine-scale resolution to characterize dominant movement patterns and locations of passage. Eels were detected at all depths throughout the forebay; however, they spent the greatest proportion of their time near the bottom, with occasional vertical movements to the surface. Eels exhibited a range of movements interpreted to be downstream searching behavior, including altered vertical and horizontal positions at or near the trash racks and various looping movements directly upstream of the trash racks and throughout the entire forebay. A substantial number of these eels (28%) were detected re-entering the acoustic array on multiple dates before passing the station. The majority (89%) were detected passing downstream of the dam through the turbines.

Introduction

Downstream migration of freshwater eels can be restricted by hydroelectric facilities (Haro et al. 2000b; Richkus and Whalen 2000; EPRI 2001; Dixon 2003). As migrant eels travel downstream and encounter hydroelectric facilities, they may experience migration delays within the impoundments created by dams, be impinged on intake screens or trash racks, or be exposed to direct turbine mortality or turbine-induced injuries (Berg 1986; Adams and Schwevers 1997; Haro et al. 2000a; EPRI 2001; Haro et al. 2003; Richkus and Dixon 2003). Turbine mortality at each hydroelectric facility can be extremely variable, depending on runner type, size, speed, number of blades, blade spacing and thickness, and size of the fish (Berg 1986; Boubée et al. 2001; EPRI 2001; Larinier and Travade 2002).

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Turbine mortality of downstream migrant eels has frequently been estimated to be more than 25% and turbine-induced injuries may be even higher, yet few studies have investigated the behavior of reproductively mature silver-phase eels as they approach, encounter, and pass downstream of hydroelectric facilities (Haro et al. 2000b; Boubée et al. 2001; EPRI 2001; McCleave 2001; Behrmann-Godel and Eckmann 2003; Dixon 2003; Durif et al. 2003; Watene et al. 2003).

Initial biotelemetry studies of silver-phase eels have focused on general migratory behavior and patterns of downstream movement in freshwater and estuarine habitats. These studies have shown that initial downstream movements occur at night and are typically associated with heavy precipitation and high-flow events (summarized by Tesch 1977 and Haro 2003). More recent studies investigated movements and passage of silver-phase eels at or near hydroelectric facilities where eels frequently displayed movements interpreted to be searching behaviors within a project forebay, the impounded area directly upstream of the dam (Haro et al. 2000a; Behrmann-Godel and Eckmann 2003; Durif et al. 2003; Watene et al. 2003).

In 1996 and 1997, conventional biotelemetry studies (radio, PIT, and acoustic) were conducted at Cabot Station, a small hydroelectric facility on the main stem of the Connecticut River (Haro et al. 2000a). Downstream movement of silver-phase eels occurred primarily at night, and some of the eels appeared to spend varying amounts of time in search of a downstream-passage route in the forebay rather than passing directly through trash racks. Eels traveling downstream were observed at a variety of depths, including the surface, and were detected quickly altering their swimming depth. Migrant eels were recorded entering the forebay up to 15 times before passing, and the majority of migrant eels were believed to have passed downstream through the turbines. However, the limited radio and acoustic (primarily one-dimensional) telemetry methodologies could not adequately describe the behavior of eels with high spatial resolution or provide exact locations and depths at which eels passed through the trash racks.

To improve downstream passage, a better understanding of the behavior of migrant eels as they encounter hydroelectric dams and their movement around such obstacles is required. The primary objective of this study was to use three-dimensional (3D) acoustic telemetry to build on the foundation of telemetry data collected by Haro et al. (2000a) at Cabot Station and to further characterize downstream movements of adult silver-phase eels with higher spatial and temporal resolution. Secondary objectives included establishing the number of occurrences detected within the forebay for each telemetered eel, as well as a more specific location of passage through the trash racks; determining the portion of detections by depth of migrant eels throughout the entire forebay and directly upstream from the turbine intakes and trash rack structures; and reviewing operating conditions at the time of passage.

Methods

Study Site

The experiment was conducted in the forebay of Cabot Station from 4 October to 21 November 2002 (Massachusetts, Connecticut River, river kilometer (rkm) 198; Figure 1). Cabot Station is outfitted with six vertical Francis turbine runners; total generation capacity during the study (when units 1, 2, 5, and 6 were operating at maximum generation capacity) was 38.2 megawatts per hour (MW), and average flow was 262 $m^3 \cdot s^{-1}$. Two recently replaced runners, units 1 and 2, which are located at the south end of the powerhouse, were operated up to 10.3 MW per unit throughout the study (Figure 2). Units 3 and 4, located in

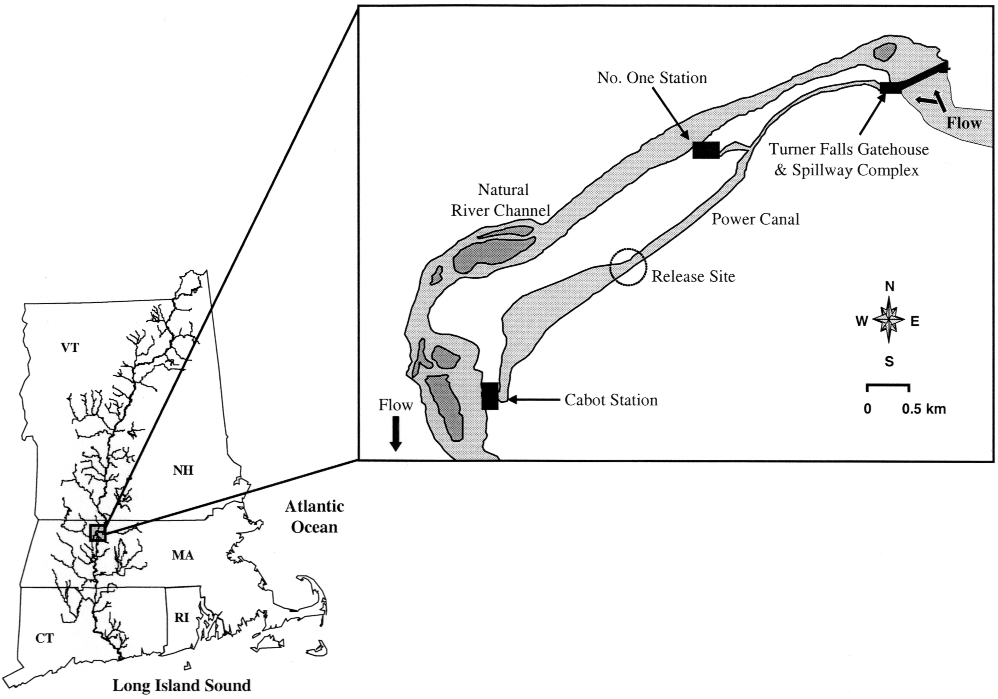

Figure 1. Study site showing the location of Cabot Station hydroelectric facility and power canal (between the Turners Falls Gatehouse and Cabot Station), Connecticut River, Massachusetts (rkm 198).

the middle of the powerhouse, were not operating during the study period because the runners were being replaced, identical to those at units 1 and 2 in 2001. The remaining two units, 5 and 6, which are located on the north end of the powerhouse, were operated up to 8.8 MW per unit throughout the study. The turbines were not operated in a specified pattern; unit generation (on/off) was highly variable.

The forebay is approximately 10 m deep. At the powerhouse, water flows through the trash racks, a series of bar racks spaced 3.2 cm apart from the surface to 3.5 m deep. At depths more than 3.5 m, the bar spacing of the trash racks is 10.2 cm. Approach velocities at the trash racks ranged from 0.3 (minimum generation capacity) to 1.2 $m^3 \cdot s^{-1}$ (maximum generation capacity; Haro et al. 2000a).

Although Cabot Station does not have downstream-passage structures built specifically to reduce turbine entrainment of silver-phase eels, a surface bypass is located in the forebay for passage of juvenile Atlantic salmon *Salmo salar* in the spring and juvenile American shad *Alosa sapidissima* in the fall. The surface bypass is located adjacent to turbine unit 1 at the south end of the powerhouse and is positioned to attract downstream migrants primarily within a meter of the surface (Figure 2). A 1,000-W mercury-vapor light used to enhance passage of juvenile shad illuminates the bypass entrance and a considerable area of the forebay intake area; walkways are also illuminated at night. During the fall of 2002, the bypass was operated from 1 September to 15 November at 2–3% of the facility's maximum flow and was typically between 6 and 8 $m^3 \cdot s^{-1}$. Historically, only a few eels have been collected at this surface bypass sampler each season (Haro et al. 2000a).

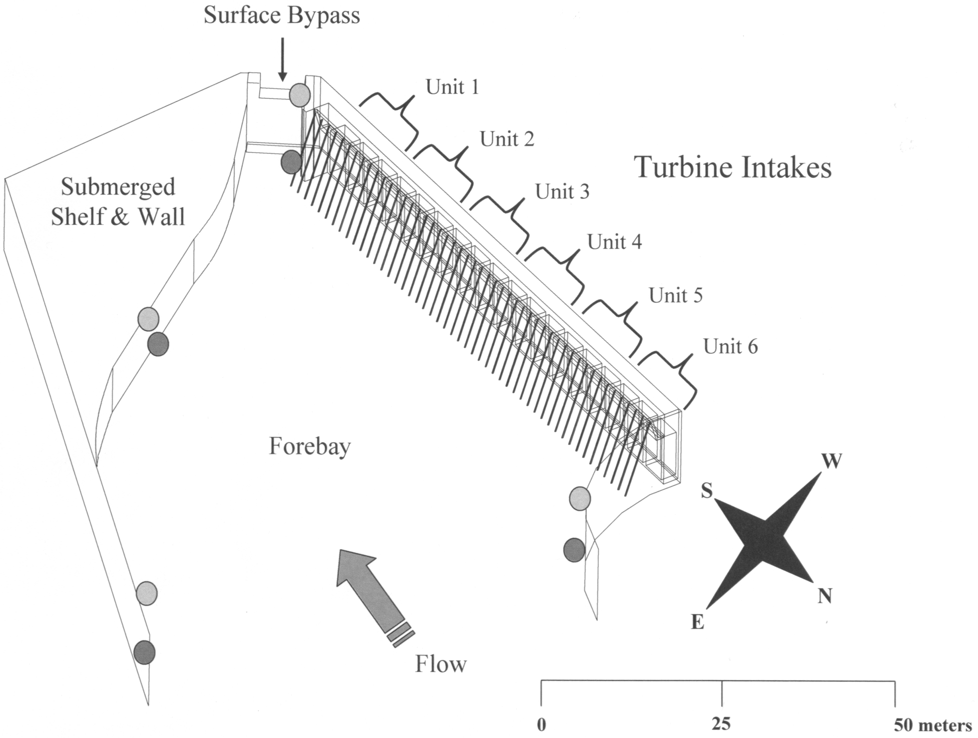

Figure 2. Cabot Station forebay with hydrophone locations (light gray circles = surface hydrophones, dark gray circles = bottom hydrophones), the trash rack structure positioned at 17-degree slope (vertical lines), and turbine unit intakes. Total depth is 10 m, with a level bottom. Units 3 and 4 were not operational during the time of the study.

Telemetry

The primary spatial telemetry system used at Cabot Station during the fall of 2002 was a Hydroacoustic Technology, Inc. (HTI) model 290 Acoustic Telemetry Receiver. This system is designed to calculate the 3D position of a tagged fish based on the difference in arrival times of tag signals at multiple hydrophones strategically positioned in a forebay. The hydrophone array was positioned to detect movements of telemetered eels within 100 m upstream of the powerhouse. Eight 300 kHz hydrophones were deployed throughout the forebay: four were mounted approximately 1 m below the surface and four approximately 1 m from the bottom of the canal (Figure 2). We used HTI model 795F tags (8 mm diameter by 18 mm length, weight 2.1 g, 300 kHz, 2.9–3.1 s ping rate); each tag was programmed to emit a unique frequency. To verify system precision and accuracy, test tags were deployed at known positions within the forebay.

In addition to acoustic telemetry, we tagged fish with conventional coded radio tags (Lotek model MCFT-3D; 10 mm in diameter by 29 mm in length, weight 3.7 g, 148.5 MHz, 5 s burst rate). Radio-tagged eels were monitored in the forebay and tailrace of Cabot Station with 4- and 9-element yagi antennas and Lotek SRX-400 data-logging receivers. We also implanted a passive integrated transponder (PIT) tag into each eel (Texas Instruments TIRFID system; 3 mm in diameter by 32 mm in length, weight 0.8 g, 134.2 kHz). To detect migrant eels passing at the surface bypass, a PIT detection antenna and a data-logging receiver were installed in the bypass entrance (Castro-Santos et al. 1996). All telemetry systems logged data continuously for the duration of the study; data were downloaded every several days, and receiver clocks were synchronized (nearest s) to Eastern Standard Time. We terminated data logging nearly two weeks after the last detection (13 November) and after water temperature fell below 5°C. Water temperature was recorded hourly with a LI-COR LI-1000 data logger and a thermocouple sensor placed in the power canal.

Fish Capture, Tagging, Release, and Monitoring

Twenty silver-phase American eels were collected at the Hadley Station downstream bypass sampler in Holyoke, Massachusetts (rkm 140). These downstream migrants were large, more than 500 mm total length, and presumably mostly female, based on size (Krueger and Oliveira 1997). Collections were made between 18:30 and 23:00 h, and fish were transported to the U.S. Geological Survey, S.O. Conte Anadromous Fish Research Center throughout October 2002. Eels were typically held 24 h for observation before tagging; transmitters were surgically implanted using methods similar to those of Baras and Jeandrain (1998).

Each eel was anesthetized in a 10% clove oil and ethanol stock solution added to a 10-l ambient river-water bath. Eels were typically held in the anesthesia bath for 10–15 min before tagging. Once the eels were heavily sedated, total length (TL) and eye diameter (horizontal and vertical) were measured to the nearest 0.1 mm. Eye indices, a metric for estimating yellow-phase metamorphosis to the migratory silver phase, were calculated from horizontal and vertical eye-diameter measurements using methodology from (Pankhurst 1982). The eye index (I) was calculated using the equation:

$$I = ([(\mathrm{v} + \mathrm{h})^2/4]\pi * 100)/\mathrm{TL}$$

where v is vertical diameter of the eye, h is horizontal diameter of the eye, and TL is total body length of the individual eel. Silver-phase American eels typically have an eye index between 6.0 and 13.5, with a bronze coloration along the lateral line that separates the dark, silver back from the white belly (Pankhurst 1982). Eels were tagged if this criterion was met.

After characteristics were recorded, eels were placed in a surgical trough, an additional supply of anesthetic solution was circulated through the gills, and transmitters were surgically implanted in the abdomen. The incision was closed with two to three sutures, and tissue adhesive (Vetbond by 3M) was administered to aid in closing of the incision. The duration of surgical implantation of transmitters did not exceed five minutes. Upon completion of the surgery, each eel spent a minimum of 30 min in an initial recovery tank before being transferred to a large (1,000 l) acclimation tank supplied with ambient river water for an additional 48-h observation period to allow eels to recover from surgery and to verify tag functionality.

Eels were released into the power canal approximately 1.5 km upstream of Cabot Station. Movements of eels in the canal and downstream of Cabot Station were monitored with an additional portable radio-telemetry receiver and yagi antenna. Eels were determined to have passed downstream of Cabot Station when the final 3D detection was positioned at the trash racks or the bypass, confirmed when radio-telemetry detections ceased throughout the stretch of the power canal and within the forebay. Downstream passage was also confirmed by the initial radio-telemetry detections logged in the tailrace downstream of Cabot Station.

Data Analysis

The 3D acoustic telemetry data were compiled and organized into a database. Records were filtered with HTI software (*AcousticTag* and *MarkTags*) to remove erroneous signals received during data collection (HTI 2000). Poorly recorded signals are typically caused by either noise interference (commonly associated with operation of hydroelectric dams and high-voltage environments) or a secondary, reflection of the acoustic tag signal (multipath echo). Additionally, we removed any invalid depth records from the dataset that were a result of poor signal selection (i.e., multipath echo included in the calculation of 3D positions created detections at depths >10 m; maximum forebay depth). All detections were plotted in 3D, and eel tracks were reviewed for trends in downstream movements as they encountered the facility (Figure 3 and 4).

We classified each eel occurrence in the forebay as the time from when the eel first entered the acoustic array until the time of passage or movement back upstream outside the acoustic array. Detections separated by less than 15 min were considered a single occurrence. For each of the telemetered eels, median duration of occurrence in the forebay was calculated to the nearest minute, and total residence time within the power canal (referred to as canal residence time) was calculated to the nearest h, beginning with the time of release to the time of exit at Cabot Station. Acoustic, radio, and PIT tag telemetry data were compared to determine final locations of downstream passage (i.e., upstream of forebay, through the surface bypass, or through the turbines).

All detections were combined and analyzed using the chi-square test to determine the proportion of time spent in the upper (0–3.3 m), middle (3.4–6.6 m), and bottom (6.7–10.0 m) portions of the water column in the forebay. To illustrate the distribution of detections by depth, the total number of detections were combined and analyzed at 1-m depth intervals. Because of annual sediment accumulation along the floor of the forebay and the unevenness of depth during monitoring, we combined the two meters closest to the bottom of the power canal. Forebay detections were further analyzed by grouping the proportion of detections within 10 m of the trash racks into three horizontal zones (similar to the classifications defined by Haro et al. 2000a); units 1 and 2 (south), units 3 and 4 (center), and units 5 and 6 (north). Proportion of detections within each zone at each of three depth categories (lower, middle, and upper) were arcsine transformed. To allow for comparisons with Haro et al. (2000a), we conducted separate chi-square analyses of horizontal and vertical distributions.

Results

Calibration tests indicated that 95% of all detections were within 0.26 m horizontally and 0.93 m vertically of the true positions. Our confidence level decreased slightly when the tag was suspended in the water column for prolonged periods; standard error increased in the third dimension (depth) to 1.07 m. We were unable to determine a single source of the increased error but believe it was caused by several factors, including the innate error in the system due to the geometry of the hydrophone array, conservative 3D parameters used during the generation of position(s), and unknown hydraulic conditions that may have altered the test tag position during calibration tests. Acoustic noise did not seem to influence the quality of received signals, but ambient electrical noise intermittently decreased signal detection quality. Recorded tracks of eels were generally continuous, although pings from transmitters were occasionally not detected when tags were in the margins of the forebay.

Eels were collected from the Connecticut River (Hadley Station Bypass Sampler, 140-

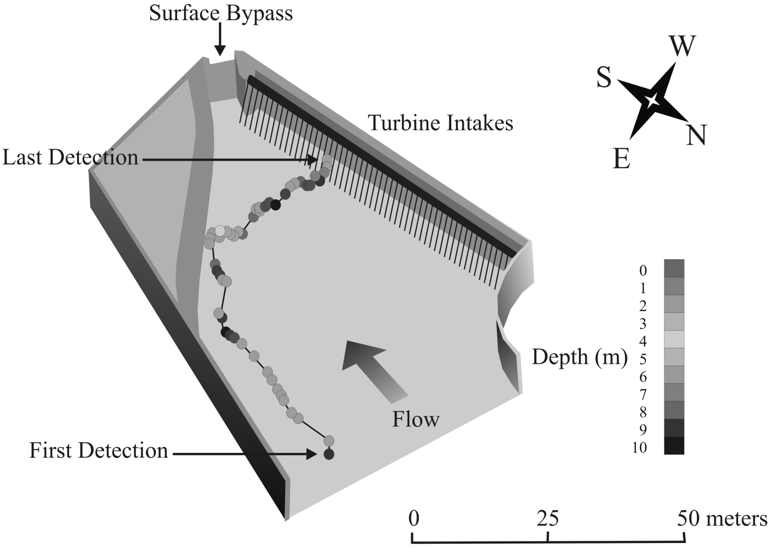

Figure 3. Three-dimensional track of Eel ID 9 in the forebay of Cabot Station (26 October 2002). The depth of each detected position is displayed in gray scale (see legend). This eel spent 8.0 min in the forebay during this occurrence before it passed at 20:02 through unit 2. At the time of passage, Cabot Station was generating at 6.3 MW, which is less than 17% total station capacity, and only unit 2 was operating. The total flow was 112.8 $m^3 \cdot s^{-1}$ at the time of passage.

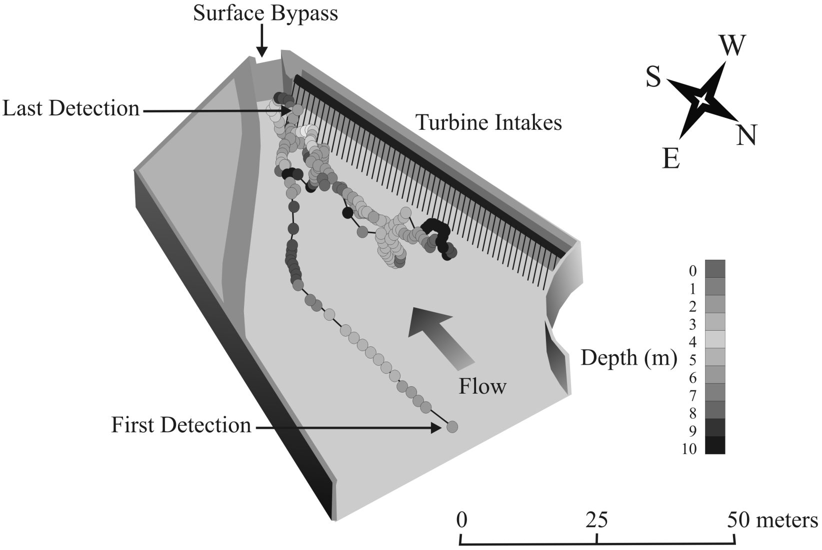

Figure 4. Three dimensional track of Eel ID 10 in the forebay of Cabot Station (23 October 2002). The depth of each detected position is displayed in gray scale (see legend). This eel spent 18.1 min in the forebay during this occurrence before it passed at 18:02 through unit 1. At the time of passage, Cabot Station was generating at 37.5 MW, which is approximately 98% full generation capacity, and turbine units 1, 2, 5, and 6 were operating. The total flow was 315.2 $m^3 \cdot s^{-1}$ at the time of passage.

Table 1. Summary of forebay occurrence data of telemetered eels detected in the forebay at Cabot Station. Median duration of occurrence(s) is measured in min. Depth distributions were calculated based on per cent of total time each eel spent in each of the three depth intervals (0–3.3 m, 3.4–6.6 m, 6.7–10.0 m), summed over all occurrences. Horizontal location distributions were calculated as a per cent of the total time each eel spent in one of the three horizontal locations (south, units 1 and 2; center, units 3 and 4; north, units 5 and 6) to a distance of 10 m upstream of the trash racks. Downstream passage routes were assigned based on last detection and are indicated by B for surface bypass, N for northern turbine units (units 5 and 6), and S for southern turbine units (units 1 and 2). Eels 6 and 11 were not detected in the forebay.

Eel ID	Number of Occurrences	Median (Range) Duration of Occurrence(s)	Passage Route	Distribution of Detections by Depth (%)			Distribution of Detections by Location (%)		
				0 - 3.3 m	3.4 - 6.6 m	6.7 - 10 m	North	Center	South
1	3	14.1 (2.9 - 15.8)	B	9.1	31.5	59.4	11.1	32.0	57.0
2	1	6.4	B	39.7	46.6	13.8	0.0	0.0	100.0
3	3	9.6 (4.1 - 14.3)	N	12.6	25.5	61.9	28.0	31.8	40.2
4	3	10.6 (3.0 - 14.7)	N	3.3	41.9	54.7	21.8	16.5	61.7
5	4	17.1 (5.9 - 33.7)	S	10.6	26.2	63.2	6.0	11.1	82.9
6	0	---	---	---	---	---	---	---	---
7	1	3.6	S	15.2	39.4	45.5	0.0	0.0	100.0
8	1	1.4	S	27.3	45.5	27.3	0.0	28.6	71.4
9	2	19.9 (10.9 - 28.8)	S	18.6	30.9	50.5	22.3	8.2	69.5
10	1	8.0	S	1.8	38.2	60.0	0.0	0.0	100.0
11	0	---	---	---	---	---	---	---	---
12	1	17.5	S	10.9	36.8	52.2	22.9	12.4	64.8
13	1	8.6	S	11.7	25.5	62.8	2.4	10.7	86.9
14	1	8.8	S	19.2	35.8	45.0	0.0	0.0	100.0
15	1	1.9	S	26.3	47.4	26.3	0.0	11.1	88.9
16	1	1.7	N	11.4	60.0	28.6	100.0	0.0	0.0
17	1	10.9	N	18.6	18.6	62.8	100.0	0.0	0.0
18	2	75.6 (35.0 - 116.2)	N	6.2	14.9	78.9	41.0	30.1	28.9
19	1	2.4	N	3.3	41.9	54.7	100.0	0.0	0.0
20	1	1.8	S	4.2	29.2	66.7	0.0	0.0	100.0
Median	1	8.6		11.6	36.3	54.7	8.6	9.5	70.5

rkm) between 27 September and 13 October 2002. A total of 20 silver eels were collected, tagged, released, and monitored in the forebay of Cabot Station. Mean TL was 707.5 mm and mean eye index was 7.74 mm. Eels were released between 13:00 and 16:00 h on 4 October (Eel ID 1–4), 13 October (Eel ID 5), 18 October (Eel ID 6–10), 23 October (Eel ID 11–15), and 1 November (Eel ID 16–20). Water temperatures ranged from 19.7°C (4 October) to 7.0°C (13 November) between the first day of release and the last detected downstream movement in the forebay.

Of the 20 telemetered eels released into the canal, 18 (90%) entered the forebay acoustic array at least once (Table 1). All eels detected in the forebay eventually passed downstream of Cabot Station by using either the turbines or surface bypass as a final passage route. Most of the detections occurred at twilight or at night; 15 out of the 18 eels (83%) that entered the forebay and passed downstream of the station did so between 18:00 and 22:00 h. Eight of the 18 (44%) eels moved downstream into the forebay within the first 24 h following release. Most eels were detected entering the forebay where the dominant flow existed, primarily in the center of the power canal or slightly to the east of the true center. Mean depth of eels entering the forebay was 6.06 m, but eels were detected entering the forebay throughout the entire water column (0.38 m to 9.85 m). Twenty-eight per cent of eels (5 out of 18) were detected re-entering the acoustic array on multiple dates, from one to four times, before passing the station.

Duration of each occurrence in the forebay was variable. Median duration was 11 min, ranging from 1.4 min to 2.8 h. Eel transit times, or total time from release to first forebay detection, were also variable; median transit time was 4.7 h but ranged from 1.0 h to 294.1 h. Median canal residence time was 49 h but ranged from 1.1 h to 294.1 h (12.3 d). Nearly all eels (16 out of 18) used the turbines as a final route of passage; four were detected at or near the entrance of the surface bypass, but only two were confirmed to have passed at that location. The 3D acoustic telemetry indicated that the remaining two eels that were recorded at the entrance of the surface bypass continued to search for a downstream passage route and ultimately passed through the turbines. All passage events were confirmed with the use of radio and PIT telemetry. Of the eels that passed downstream through the turbines, 12 out of 16 (75%) were detected using the southern turbine intakes of units 1 and 2 as a final route of passage. Example tracks of telemetered eels that passed downstream via turbines are illustrated in Figure 3 and 4.

Throughout the forebay, the distributions of detection by depth were highly variable; however, eels spent significantly more time near the bottom (chi-square; $P < 0.001$). While some eels were detected at or near the surface, more than two-thirds of the detections were within the deepest third of the forebay (6.7–10.0 m). The highest percentage of detections occurred near the bottom (Figure 5). Eels also spent significantly more time within the first 10 m directly upstream of units 1 and 2 compared with the other units (chi-square; $P < 0.001$; Figure 6).

The majority of downstream passage events occurred through the turbines; turbine passage was identified under two broad behavioral tendencies. First, eels were detected passing directly through the trash racks and into one of the four turbine units upon their first encounter. Seven eels out of the 16 that passed downstream via the trash racks passed the station through the turbines immediately after contact with the trash racks. Second, the remaining nine eels were recorded passing through the trash racks after one or more combinations of searching behaviors and then passing downstream of Cabot Station through the turbines. At least 50% of these fish undertook vertical searching movements when they encountered the trash racks, swimming up

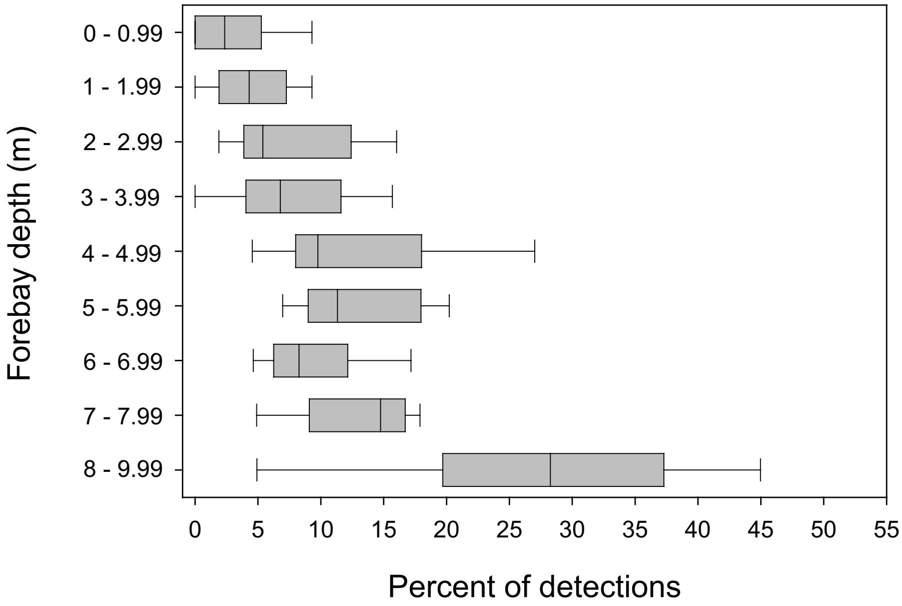

Figure 5. Proportion of time spent by telemetered eels within the entire forebay by depth (1-m intervals). Data were generated from individually standardized track data of 18 eels. Central vertical bar = median; shaded bar = 75th percentile; whisker = 90th percentile.

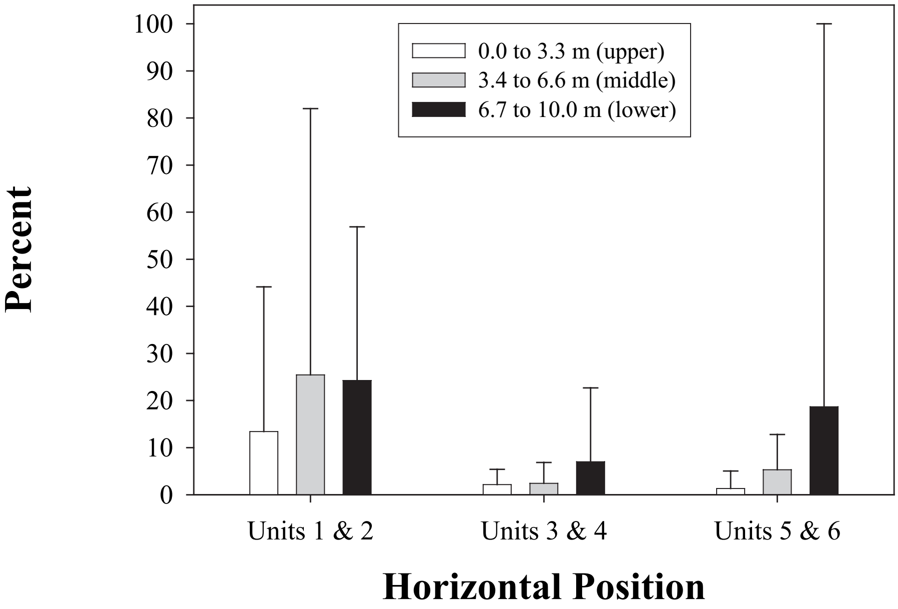

Figure 6. Proportion of time spent by telemetered eels directly upstream of the trash racks (within 10 m) by depth (3 depth intervals) and horizontal position (unit numbers). Data were generated from individually standardized track data of 18 eels. Bar = median; whisker = 90th percentile.

and down in the water column of the forebay. After approaching the trash racks, 10 out of 16 eels also swam horizontally along the trash racks. In addition, 10 out of 16 (63%) exhibited circling movements directly in front of the trash racks and larger circling movements encompassing the entire forebay. Circling movements were often associated with either vertical and/or horizontal searching behavior along the trash racks. Both patterns of prepassage movement at the trash racks (direct passage versus searching before passage) occurred under both low (<256 $m^3 \cdot s^{-1}$) and high (>256 $m^3 \cdot s^{-1}$) flow conditions.

Discussion

Protection of downstream-migrant fishes at hydroelectric intakes requires that fish be attracted, guided, or physically diverted to safe passage routes. Three-dimensional acoustic telemetry provided a more comprehensive measure of position and eel movements than has been previously possible with conventional telemetry methods. Error in positioning of fish was relatively low; thus, we are confident that our 3D eel movements represent true behaviors in the forebay. The behaviors that were detected as telemetered eels approached and encountered Cabot Station were variable, yet the results of this study supported the findings of Haro et al. (2000a).

Eels moved very little during the day; downstream movements to the forebay occurred primarily within several hours after sunset. Although eels spent a significant amount of time at or near the bottom of the forebay, they were detected swimming near the surface occasionally and were observed rapidly altering their depth of swimming. Some telemetered fish were reluctant to pass either through the racks or into the bypass on their first forebay encounter and did not pass the station for several days, even after several occurrences within the forebay. This trend was observed by Haro et al. (2000a), who reported that some eels attempted to pass Cabot Station up to 15 times.

Similar delays have been noted for other anguillids, particularly European eels *Anguilla anguilla* at a hydroelectric dam in Germany (Behrmann-Godel and Eckmann 2003) and New Zealand longfin *A. dieffenbachia* and shortfin *A. australis* eels at hydroelectric dams in New Zealand (Boubée et al. 2001; Watene et al. 2003). Behrmann-Godel and Eckmann (2003) observed similar circling of European eels within the forebay of a hydroelectric facility. While our sample size is limited ($n = 18$), our results support previous biotelemetry studies conducted on the downstream migration of four silver-phase anguillid species and suggest that behavioral movements at or near hydroelectric facilities may be relatively consistent across taxa.

Only two eels used the surface bypass as a route of downstream passage. Although the surface bypass at Cabot Station is equipped with a uniform acceleration weir-entrance structure, it passes under 3% of the total maximum flow of the project. The extensive lighting at the powerhouse at night, in particular the supplemental illumination in the surface bypass, may have prevented eels from using this route for downstream passage. Although eels have been observed passing through the surface bypass, they have also been video-recorded entering the bypass, reversing direction, and swimming upstream to re-enter the forebay (Haro et al. 2000a; video recording by Haro, unpublished data). The combination of low relative flow, surface orientation, and high illumination may contribute to the ineffectiveness of the bypass for eels.

Eels either passed immediately upon entry into the forebay or were delayed; delay was characterized by searching behaviors and, in some cases, repeated occurrences. The presence of trash racks and the hydraulic conditions at or near the trash racks may be one of the primary reasons silver eels, even those that are large enough, do not always pass quickly though the bar racks and ultimately may slightly alter their downstream-passage behavior. Both patterns of prepassage movement, direct passage during initial encounter of trash racks versus searching before passage, occurred under both low and high flow conditions, which suggests that the behaviors are not driven exclusively by hydraulics.

When close to the trash racks, eels were observed rapidly diving to the deepest portions of the forebay. This is consistent with sounding behavior speculated to occur when eels first come into contact with trash racks or other novel structures (Haro et al. 2000a). In the case of downstream migrant anguillids, responses to hydraulics and topography of forebay environments are complex, but our findings provide some insights into generalized behaviors. As silver eels approach an intake from a head pond or impoundment, they may initially follow dominant flow and drift or swim downstream at mid or upper depths, as they have been shown to do in natural, open-river systems (Tesch 1994; Parker and McCleave 1997). We found that the presence of Cabot Station, and perhaps the illumination associated with the station, may have altered the swimming depth of some migrant eels to deeper locations than those previously reported of migrant eels in natural, unobstructed waterways.

Upon encountering trash racks or other similar structures that obstruct or interrupt downstream migration or flow characteristics, the initial response of eels is to either pass directly through the turbines or sound, reverse direction, and swim laterally or upstream, or a combination of these behaviors. The use of 3D telemetry aided in detecting and characterizing these fine-scale movements and behavioral trends. We found that some eels swam upstream rapidly after initial contact with the trash racks. Within the area 10 m upstream of the trash racks, eels also spent the greatest proportion of time near units 1 and 2, where two-thirds of the eels passed downstream of Cabot Station. This behavior was frequently observed when greater flow passed through these units, which were the primary units operating during the study period and the units where the highest flows and approach velocities occurred. These results suggest that silver eels are attracted to dominant flow fields and support the limited literature of downstream passage of silver eels (Vøllestad et al. 1986; Boubée et al. 2001, 2003; Haro et al. 2003).

The majority of eels passed through the trash racks, likely because of the dominant flow at the trash racks versus the surface bypass. The final locations of downstream passage were detected throughout the entire water column; however, the majority (two-thirds) were detected in the lower portion of the trash racks (3.5–10.0 m deep) probably because of the increase in spacing (10.2 cm) in this region. Nearly half of the eels (7 out of 16) that were last detected passing through the trash racks were subsequently detected moving back upstream through the racks. While it is possible that high approach velocities may have ultimately made it more difficult for some eels to swim back upstream or avoid being entrained through the trash racks, it is apparent that over half the eels that passed downstream of Cabot Station were not passively entrained into the turbine units. Similar behavior of American and European silver-phase eels has been observed with video in response to angled bar racks in laboratory flumes (Adams and Schwevers 1997; Amaral et al. 2003). Downstream migrant fishes and various larval fishes can be easily entrained into turbine units at small-scale hydroelectric facilities (Nestler et al. 1992; Travnichek et al. 1993; Mathur et al. 2000); this does not appear to be typical for all silver-phase eels. More than half of the eels (9 out of 16) did not pass through the trash racks during their initial encounter but first searched for a downstream passage route and then voluntarily passed through the turbines at Cabot Station.

Management Implications

Behaviors of migrant eels in hydroelectric forebays are unlike those of traditional downstream migrant species such as juvenile salmonids and anadromous clupeids, which remain near the surface and can be attracted and guided by directional flow fields and lighting at a bypass entrance (Haro et al. 2000a). While the passage of downstream migrant eels appears to be influenced heavily by directional flow fields, eels are not surface oriented and often are repelled by light. We found that the majority of eels at Cabot Station were delayed during their downstream migration; many of these eels made multiple attempts to pass downstream of this station. Our results also indicate that while eels spend a significant amount of time near the bottom of the power canal, they can alter their position frequently, particularly at or near the trash racks.

Establishing safer downstream passage alternatives for eels is challenging because they appear to be attracted to the dominant flow field, which usually is associated with the turbines unless the turbine units are not generating and a modified downstream passage facility exists or can be built that handles a large enough volume of water to attract downstream migrants before they encounter the turbine intakes. Our results indicate that eels exhibiting searching behaviors within a forebay may have a higher probability of encountering a submerged or bottom bypass entrance than a conventional surface bypass entrance. Recent studies conducted by Durif et al. (2003) illustrate that significantly more European eels will pass a hydroelectric project by means of a submerged bypass than through a traditional surface-oriented bypass.

Although eels appear able to detect and avoid trash rack structures and other obstructions, they can also pass voluntarily through bar racks, limiting opportunities to guide eels to a bypass under these conditions. Development of structures or altered project operations to protect downstream migrant anguillids from entrainment and subsequent turbine mortality are at a very preliminary stage. Behavioral barriers and guidance devices have also been shown to have limited effectiveness for eels (i.e., angled bar racks and louvers, light, sound, water jets and air bubbles, and electrical fields: see Dixon 2003; EPRI 2001; Richkus and Dixon 2003). As an alternative, operational shutdowns may be effective in decreasing overall mortality of silver eels at some small hydroelectric facilities in Maine (Haro et al. 2003). Shutting down hydroelectric facilities in combination with peak environmental conditions during downstream migration of eels (heavy rain, increased flow, perhaps four to six hours after sunset) could reduce the risk of turbine-related injuries and mortality. However, under most circumstances, electricity demands may make this alternative unfeasible. Future efforts to develop effective bypasses or guidance structures should consider these behaviors to improve safer downstream passage of migrant silver eels.

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FEATURE

Understanding Barotrauma in Fish Passing Hydro Structures: A Global Strategy for Sustainable Development of Water Resources

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ABSTRACT: *Freshwater fishes are one of the most imperiled groups of vertebrates, and population declines are alarming in terms of biodiversity and to communities that rely on fisheries for their livelihood and nutrition. One activity associated with declines in freshwater fish populations is water resource development, including dams, weirs, and hydropower facilities. Fish passing through irrigation and hydro infrastructures during downstream migration experience a rapid decrease in pressure, which can lead to injuries (barotrauma) that contribute to mortality. There is renewed initiative to expand hydropower and irrigation infrastructure to improve water security and increase low-carbon energy generation. The impact of barotrauma on fish must be understood and mitigated to ensure that development is sustainable for fisheries. This will involve taking steps to expand the knowledge of barotrauma-related injury from its current focus, mainly on seaward-migrating juvenile salmonids of the Pacific Northwest, to incorporate a greater diversity of fish species and life stages from many parts of the world. This article summarizes research that has examined barotrauma during fish passage and articulates a research framework to promote a standardized, global approach. The suggested approach provides clearly defined links to adaptive development of fish friendly technologies, aimed at mitigating the threats faced by global freshwater fisheries from the rapid expansion of water infrastructure.*

Sobre el barotrauma en peces durante su tránsito por hidro-estructuras: una estrategia global para el desarrollo sustentable de los recursos hídricos

RESUMEN: *los peces de agua dulce constituyen uno de los grupos más amenazados entre los vertebrados y las disminuciones poblacionales se consideran como alarmantes en términos de biodiversidad y suceden en perjuicio de las comunidades humanas cuyo bienestar y nutrición dependen de las pesquerías basadas en estos recursos. Una actividad que se asocia a la declinación de las poblaciones de peces de agua dulce es la construcción de infraestructura para el desarrollo de recursos hídricos, como presas, weirs e instalaciones hidroenergéticas. Los peces que transitan a través de la infraestructura hidráulica y de irrigación durante su migración hacia el mar, experimentan disminuciones de presión que producen lesiones (barotrauma), las cuales pueden contribuir a la mortalidad. Existe una nueva iniciativa para expandir la infraestructura para la hidroenergía e irrigación y aumentar así la seguridad de agua y la generación de energía de bajo costo en términos de producción de carbono. El efecto del barotrauma en los peces debe ser estudiado y mitigado para asegurar que el progreso sea sustentable para las pesquerías. Esto implicará expandir el conocimiento acerca de las lesiones relacionadas al barotrauma con respecto a como se encuentra ahora; sobre todo el conocimiento de la migración hacia el mar que realizan los juveniles de especies de salmón en el Pacífico noroeste, con el fin de incorporar una mayor diversidad de estadios de vida y especies de diferentes partes del mundo. En este artículo se resume la investigación concerniente al barotrauma en los peces durante su tránsito por hidro-estructuras y se plantea un marco investigativo para promover un enfoque estándarizado y global. El enfoque que se ofrece provee relaciones precisas para el desarrollo adaptativo de tecnologías amigables para los peces, diseñadas con la finalidad de mitigar las amenazas que enfrentan las pesquerías de agua dulce ante la rápida expansión de la infraestructura hídrica.*

INTRODUCTION

Freshwater fish are the second most endangered vertebrate group (Saunders et al. 2002), and many species currently face extinction (Ricciardi et al. 1999). Species declines are not abating, and in many parts of the world such declines have significant social and economic implications. Many of the world's developing nations rely heavily on freshwater fish for their livelihood, as both a source of income and food. For example, the Lower Mekong River basin (i.e., Cambodia, Laos, Thailand, Vietnam) supports the world's largest inland fishery, worth between US$4.3 and $7.8 billion annually (Hortle 2009). Fish and other aquatic organisms are essential for the livelihood, nutrition, and food security of citizens of the Lower Mekong River basin, accounting for 47%–80% of total animal protein consumed (Hortle 2007).

Many activities have had a role in freshwater fish declines throughout the world, including development of water infrastructure (Dudgeon et al. 2006). Water infrastructure, including dams, weirs, and hydropower facilities, can change natural flow regimes, degrade habitat and water quality, and interrupt or otherwise negatively impact important upstream and downstream fish migrations (Kingsford 2000; Agostinho et al. 2008). Though water infrastructure can create a complete barrier to fish movements, structures can also selectively injure or kill fish as they pass (Williams et al. 2001; Godinho and Kynard 2009). In such cases, barotrauma (trauma due to changes in barometric pressure) is of particular concern where hydropower facilities and irrigation structures create adverse hydraulic conditions that can injure and kill passing fish (Cada 1990; Baumgartner et al. 2006; Brown et al. 2012a).

Globally, the infrastructure associated with hydropower and other water resource development are extensive and expanding rapidly, especially in areas such as China, Brazil, and Africa (Geoscience Australia and ABARE 2010). Brazil is one example where hydropower generation is projected to increase 38% by 2020 (Ministério de Minas e Energia/Empresa de Pesquisa Energética [MME/EPE] 2011) through large hydropower projects, such as the Belo Monte Dam on the Xingu River of the Amazon Basin (the third largest [11,233 MW] hydropower production facility in the world; MME/EPE 2011; Castro et al. 2012) and the Santo Antônio (3,150 MW power potential) and Jirau (3,300 MW power potential) dams on the Madeira River. Worldwide, opportunities are being explored to install small-scale (typically less than 10 MW) hydroelectric facilities at water infrastructures built for other purposes, such as existing irrigation weirs (Bartle 2002; Paish 2002; Baumgartner et al. 2012).

The expansion of hydropower generation is in response to increasing demand for power in developing regions and a global desire for increased use of renewable energy in response to climate change. However, to maintain fish diversity and curb social and economic impacts in light of this development, research is needed to guide the design and management of hydropower facilities and other water infrastructure. In particular, minimizing barotrauma associated with passage through water infrastructure is a complex issue and of particular concern. In this article we review the science related to barotrauma with the objective of highlighting what is known and the knowledge gaps that exist in adaptively managing the threats faced by freshwater fisheries from the rapid expansion of water infrastructure. Though information covered may provide insight for barotrauma induced by angling, commercial fishery bycatch operations, or scientific sampling involving quickly bringing fish to the surface of a water body, the main focus of this article is furthering the understanding of barotrauma among fish passing downstream through dams, weirs, and hydropower facilities. In addition, this article does not provide an exhaustive review of all such water infrastructure passage related barotrauma (for further background information see Cada 1990) but focuses on the state of the science, provides insight for interpreting past research, and provides modeling and research frameworks for future endeavors in barotrauma research.

BAROTRAUMA DURING WATER INFRASTRUCTURE PASSAGE

It has long been acknowledged that fish can be killed or injured when passing through hydroturbines at hydroelectric facilities (Cramer and Oligher 1964). Similarly, it has been shown that fish can be harmed during passage through bypass systems or spillways at hydroelectric facilities (Muir et al. 2001). But the impact is not confined to structures specifically designed for the generation of hydropower, and considerable injury and mortality rates have also been reported for fish passing weirs primarily built to capture and divert river flows for irrigation (Baumgartner et al. 2006). This aside, research carried out to understand the mechanisms for injury during water infrastructure passage has been predominately focused around hydroelectric turbine passage (Coutant and Whitney 2000).

When fish pass through hydrostructures, such as hydroturbines, shear forces, blade strike, and pressure changes can lead to injury and death (Deng et al. 2005, 2007a, 2010; Cada et al. 2006; Brown et al. 2009, 2012b). Although one of the most apparent sources of injuries to fish may be strike from turbine blades, the likelihood of strike is low for small fish (Franke et al. 1997). Not all fish passing through hydroturbines are exposed to damaging levels of shear force or blade strike (Deng et al. 2007b), because this depends on the route taken by fish through the system and blade strike can vary to a large degree with fish size (Franke et al. 1997). All fish, however, are exposed to pressure changes, and the magnitude of change depends largely on turbine design, the path of the fish through the turbine, the operation of the turbine, the total operating head, the submergence of the turbine, and the rate of flow through the turbine (Carlson et al. 2008; Deng et al. 2010; Brown et al. 2012b).

As fish pass between turbine blades, they are typically exposed to a sudden (occurring in <1 s) decompression before returning to near surface pressure as they enter the downstream channel (Deng et al. 2007b, 2010). In hydroturbines, this can commonly involve decreases in pressure to levels between surface pressure (101 kPa) and half of surface pressure of approximately 50 kPa (Carlson et al. 2008). Fish passing through other types of hydrostructures are also exposed to rapid pressure changes (see Carlson et al. [2005] for an example of pressure fluctuations at a pump storage facility). Although little research has been done to quantify pressure changes outside of the hydroturbine realm, initial hydraulic investigations of irrigation weirs, where water is discharged under a gate (referred to as “undershot weirs”), show that passing fish would experience rapid decompression (in <1 s) to slightly below surface pressure as they are taken from depth in the upstream pool and discharged into surface waters downstream of a structure (C. A. Boys [New South Wales Department of Primary Industries] and Z. D. Deng [Pacific Northwest National Laboratory], personal communication).

The rapid decompression associated with infrastructure passage can lead to barotrauma arising from one of two major pathways. The first is governed by Boyle’s law, where damage occurs due to the expansion of a preexisting gas phase within the body of the fish, such as contained in the swim bladder (Keniry et al. 1996; Brown et al. 2012e; Pflugrath et al. 2012). Boyle’s law ($P_1V_1 = P_2V_2$ [where P_1 and V_1 are the initial pressure and gas volume and P_2 and V_2 are the resultant pressure and gas volume]) states that within a closed system (at constant temperature), the volume of a gas is inversely proportional to the pressure acting on the volume (Van Heuvelen 1982). For a fish passing through infrastructure, if the surrounding pressure is decreased by half, the volume of the preexisting gas in the body doubles. Injuries arising from this pathway typically include ruptured swim bladders and exopthalmia (Figure 1), everted stomach or intestine (Figures 2A and 2B), internal rupture of vasculature (hemorrhaging), and gas bubbles (emboli) in the vasculature, organs, gills, and fins (Tsvetkov et al. 1972; Rummer and Bennett 2005; Gravel and Cooke 2008; Brown et al. 2009, 2012b).

The second pathway is governed by Henry’s law, where gas may come out of solution due to decompression-induced reduction in solubility, resulting in bubble formation (Brown et al. 2012e). Henry’s law states that the amount of gas that can be dissolved in a fluid, such as blood plasma, is directly proportional to the partial pressure to which it is equilibrated. Thus, when the surrounding pressure is reduced, the dissolved gas may come out of solution, resulting in gas bubble formation, the basis for the bends in scuba divers who return to the surface too quickly. As fish pass through areas of low pressure, such as through hydroturbines, and experience decompression, their blood and other bodily fluids may become temporarily supersaturated and gas bubbles may form in the blood, organs, gills, or fins (emboli). As the gas bubbles grow, they can also lead to internal rupture of vasculature (hemorrhaging; Brown et al. 2012b; Colotelo et al. 2012).

Henry’s and Boyle’s laws may not be equally important in governing injury to fish during water infrastructure passage. Brown et al. (2012e) determined that, among juvenile Chinook Salmon (*Oncorhynchus tshawytscha*), injury and mortalities observed due to rapid decompression (simulating turbine passage) were largely caused by swim bladder expansion and rupture (as governed by Boyle’s law), and the likelihood of mortality due to gases coming out of solution in the blood and tissue (as governed by Henry’s law) was relatively low. They found that if juvenile Chinook Salmon were slowly decompressed to very low pressures (13.8 kPa; with 101 kPa representing surface pressure) over 2.9–3.6 min (median = 3.3 min), the fish could expel gas from their swim bladder via the pneumatic duct (a connection between the swim bladder and esophagus; Figure 3), preventing its rupture and subsequent barotraumas (e.g., emboli in the fins, gills, and blood vessels; exopthalmia; hemorrhaging). If fish were maintained at these low pressures, it took several

Figure 1. Exopthalmia (eyes popped outward) observed in (A) the Brazilian species Corvina captured downstream of a hydropower facility and (B) in juvenile Steelhead exposed to rapid decompression from depth (510.1 kPa, the equivalent to 40.7 m) to near surface pressure (117.2 kPa; Brown et al. 2012e). Photo credit: Carlos Bernardo M. Alves, Bio-Ambiental Consultancy.

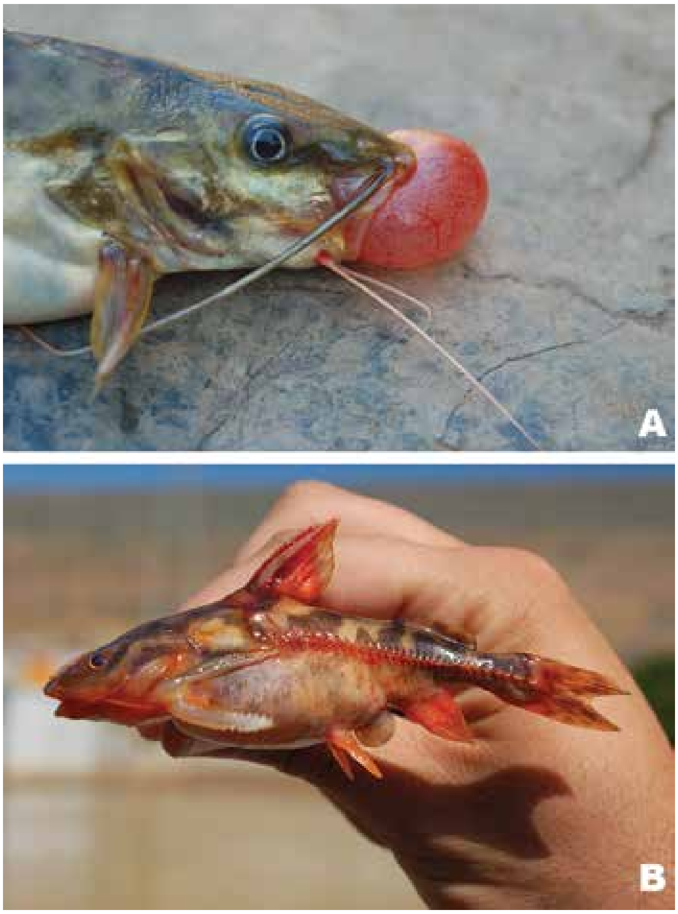

Figure 2. Images of an (A) everted stomach in the Brazilian species Mandi-amarelo and (B) an everted intestine in Serrudo. Photo credit: Carlos Bernardo M. Alves, Bio-Ambiental Consultancy.

minutes (mean = 3.0; range 2.2–7.0) before emboli and mortality were observed, presumably associated with Henry's law. In comparison, however, if juvenile Chinook Salmon were rapidly decompressed, the swim bladder often ruptured, expelling gas into the tissue and vasculature leading to hemorrhaging, emboli, and exopthalmia.

Though it appears that barotraumas governed by Henry's law are slow to develop relative to those linked to Boyle's law in juvenile Chinook Salmon, there are species-specific differences in damages that occur when fish are exposed to decompression. For instance, where Brown et al. (2012e) saw mortality due to Henry's law in juvenile Chinook Salmon exposed to 2.2–7.0 min of low pressure (13.8 kPa), Colotelo et al. (2012) found that juvenile Brook (*Lampetra richardonii*) and Pacific Lamprey (*Entosphenus tridentatus*) were uninjured when exposed to these same low pressures for over 17 min. Thus, the likelihood of emboli formation (and associated injuries such as hemorrhaging) may vary substantially among species. Though only a few species have been examined to date, it appears unlikely that gas coming out of suspension and forming emboli is the major cause of injury and mortality among fish passing hydrostructures because they are seldom if ever exposed to pressures below surface pressure for more than even a single second.

However, it should be kept in mind that supersaturation of gas is a large problem associated with dams. High levels of total dissolved gas (TDG) are associated with water routed over spillways. Water falling over spillways and into deep plunge basins of dams can cause gas to be entrained into the water (Ebel 1969). Prolonged exposure to elevated TDG can cause gas bubble disease (GBD) in fish. The difference between GBD and bubbles forming in the blood associated with barotrauma is that GBD involves gas moving from the surrounding supersaturated water into the tissues of the fish, leading to the formation of emboli (Beyer et al. 1976). Alternatively, when fish are decompressed during passage of a hydrostructure, the temporary supersaturation of the blood can cause bubbles to come out of suspension in the blood and tissues (Beyer et al. 1976). Thus, the source of the supersaturated gas is from within the fish instead of from the surrounding supersaturated water. Although a review of GBD is not within the scope of this article, it is possible that elevated TDG could lead to an increase of barotrauma. If fish with emboli present in their body due to GBD are decompressed during passage of hydrostructures, a higher amount of barotrauma may occur due to the expansion of those bubbles than may occur when the river water does not have elevated levels of TDG.

This leads to another factor that should be kept in mind when interpreting the barotrauma literature. Some researchers have had issues with confusing barotrauma with GBD when conducting decompression studies on fish. If the water the fish are held in while under pressure in test chambers is aerated or otherwise saturated with gas (similar to experiments by Bishai [1960] and D'Aoust and Smith [1974]), fish could experience GBD when decompressed, essentially the same condition as the bends in humans. This would lead to an extended period where the blood and tissues of the fish would be supersaturated instead of the very short period of supersaturation that fish would be exposed to during hydrostructure passage.

IMPLICATION OF SWIM BLADDER MORPHOLOGY

Barotrauma damage is frequently attributed to swim bladder expansion and rupture and, as such, the diversity in swim bladder form and function among fish may have significant implications for the relative susceptibility to injury. There are two broad groups, physoclists and physostomes. Physostomes, which are evolutionarily more basal fishes (e.g., lungfishes, sturgeons, and euteleosts), have a swim bladder that is connected to the esophagus via a pneumatic duct (often referred to as an open swim bladder). These fish gulp air at the surface and force it into their swim bladder. The second group is called physoclists, which are evolutionarily more derived fishes (neoteleosts), which have a swim bladder that is not connected to the esophagus (often referred to as a closed swim bladder; Figures 3 and 4) and the presence of a gas gland and countercurrent vasculature (called "retia") is used to regulate swim bladder volume and thus buoyancy (Pelster and Randall 1998). Physoclists may be much more likely to be injured during passage of hydrostructures than physostomes because they cannot quickly

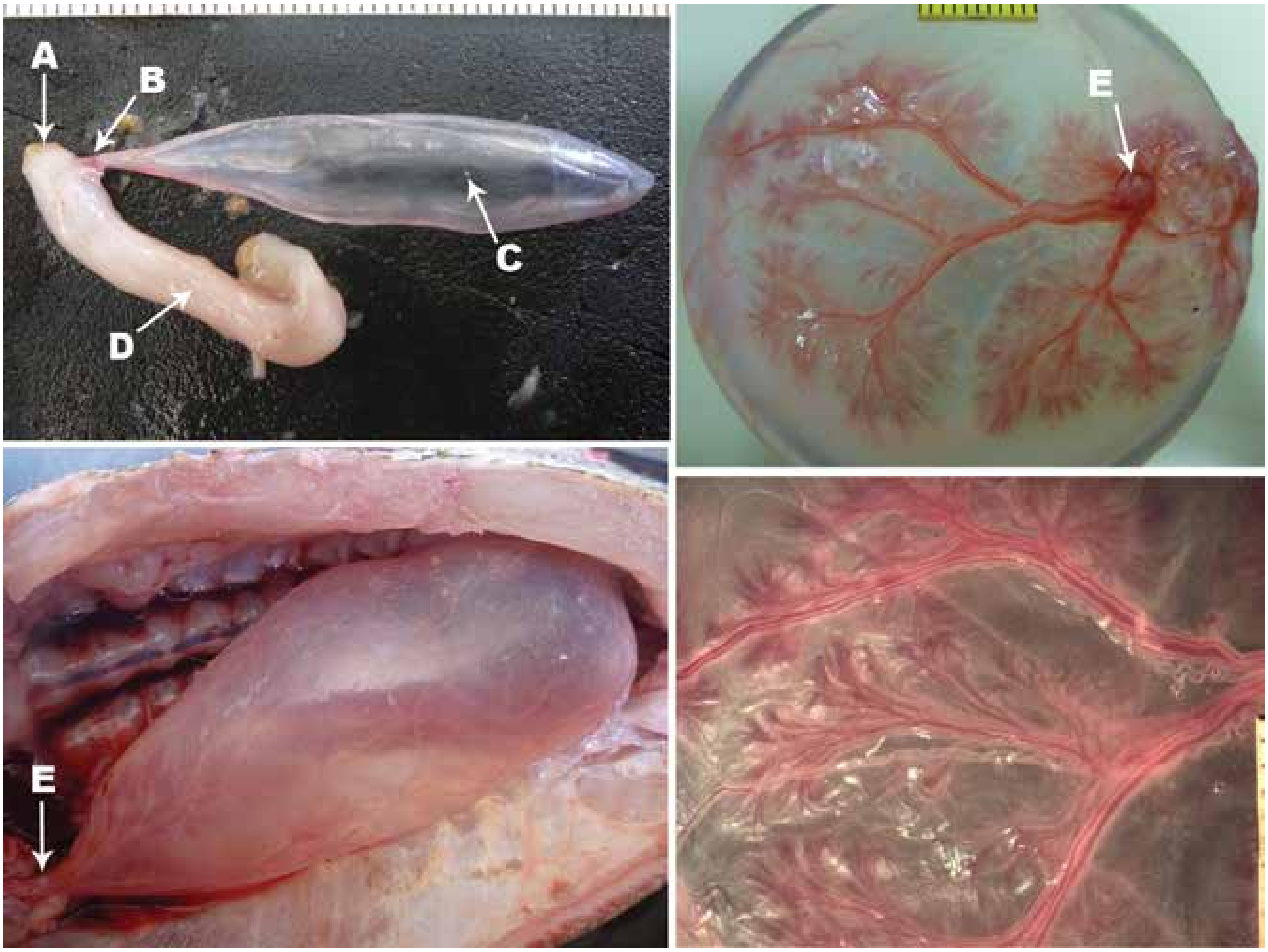

Figure 3. (A) Esophagus, (B) pneumatic duct, (C) physostomous swim bladder, and (D) stomach of a juvenile Chinook Salmon are shown in the upper left panel. The other three panels are photos of a physoclistous Smallmouth Bass swim bladder. The inflated swim bladder is shown in the lower left panel with the incoming vasculature source shown (E). The upper and lower right panels show a deflated swim bladder and the vascular rete (E also shows the incoming source of the vasculature). Photo credit: Ricardo W. Walker.

release gas as the swim bladder expands during rapid decompression (Brown et al. 2012e). To add to the complexity, most fish that are physoclistous as adults are physostomous as larvae, which enables initial swim bladder inflation by gulping air (e.g., Bailey and Doroshov 1995; Rieger and Summerfelt 1998; Trotter et al. 2003). Thus, the vulnerability to barotrauma may vary greatly within a species depending on its life stage (Tsvetkov et al. 1972). Another noteworthy variation in swim bladder morphology is found in the most diverse family of freshwater fishes, the cyprinids, which form a major component of the migratory fauna of Asian rivers. They have a physostomous swim bladder, but it has two chambers with an anterior projection closer to the Weberian apparatus to enhance hearing (Alexander 1962; Figure 5). The chambers are connected by an additional duct under autonomic muscular control (Dumbarton et al. 2010). Thus, during rapid decompression, excess gas would need to be voided through both chambers and two ducts simultaneously in order to prevent barotrauma due to swim bladder damage.

In order to predict the extent of inter- and intraspecific barotraumas that may be induced by hydrostructures within a given river system, it is crucial to understand how pressure changes affect fish with different types of swim bladders at different life stages. Physostomes are able to quickly expel gas via the pneumatic duct, using the *gass-puckreflex* (gas spitting reflex; Franz 1937), which is under autonomic control. The rate of this reflex is likely critical in reducing injury due to rapid decompression but appears to vary between—and even within—species (Harvey et al. 1968; Shrimpton et al. 1990). Shrimpton et al. (1990) determined that smaller Rainbow Trout had a higher gas pressure release threshold than larger fish (when examining fish in a range from less than 10 to ~250 g). Additionally, there have been observations of siluriform Catfish with everted stomachs (Figures 2A and 2B) downstream of hydroelectric facilities, which indicates that gas was not released fast enough from their physostomous swim bladder to avoid barotrauma during rapid decompression.

Unlike physostomes, physoclists can only regulate buoyancy through a relatively slow process of gas diffusion into and out of the swim bladder (see Figure 3). The physoclistous swim bladder is filled predominantly by oxygen that is released from a pH-sensitive hemoglobin as it is acidified within the retia of the swim bladder (Pelster and Randall 1998). The rate of swim bladder filling and the partial pressures that can be ultimately generated varies widely among physoclists, with some species able to attain neutral buoyancy at much deeper depths than others (Fange 1983). Some species, like Tench (*Tinca tinca*), can

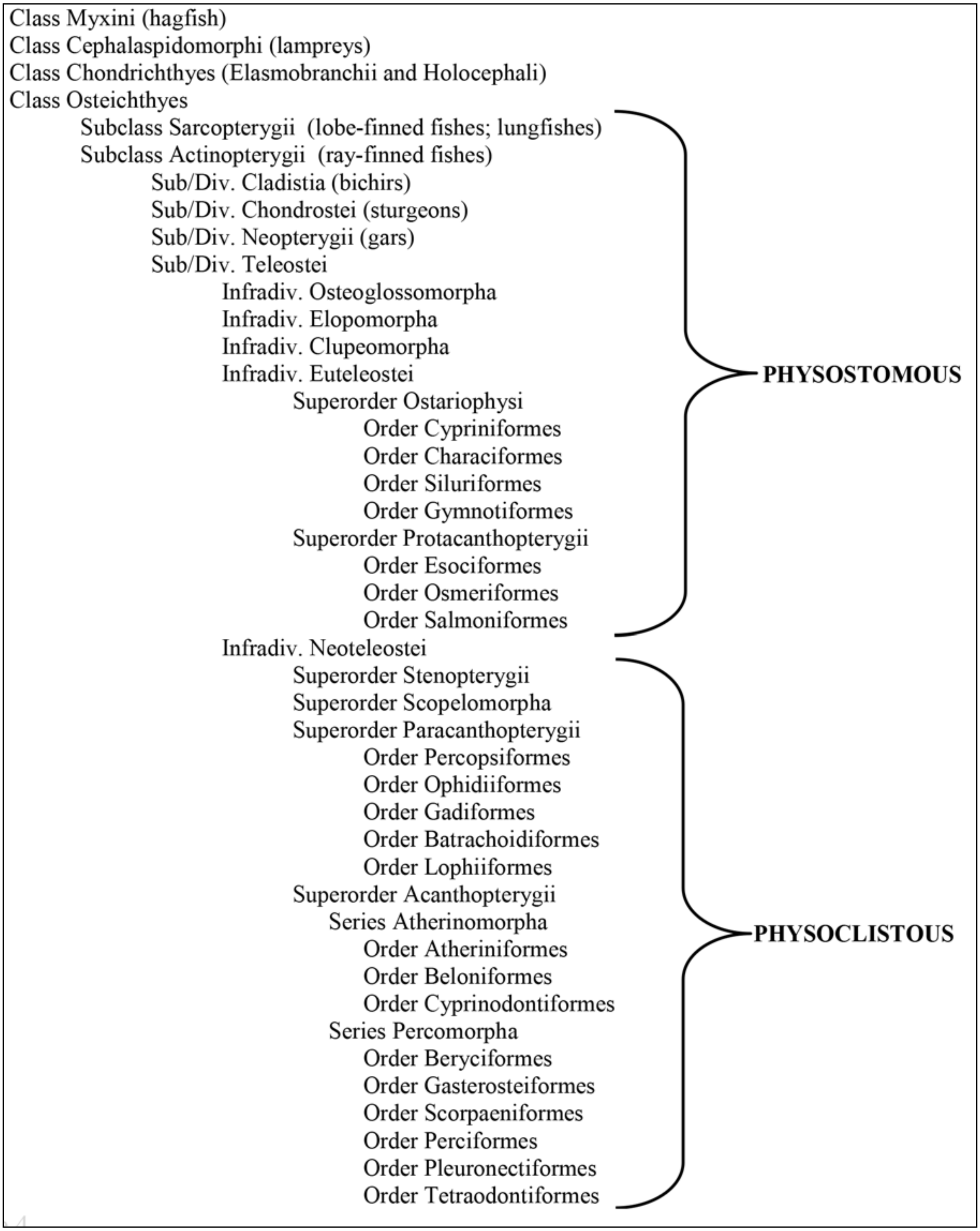

Figure 4. The type of swim bladder present in different taxa of fish. Fish with an opening between the swim bladder and the esophagus (physostomes) and without this opening (physoclists) are shown, as well as fish without a swim bladder (the upper most three classes).

take weeks to fill their swim bladder (Jacobs 1934), whereas Bluefish (*Pomotomus saltatrix*) may be able to do so relatively rapidly (less than 4 h after puncture; Wittenberg et al. 1964) but still require hours to days. Presumably, these rates of swim bladder filling are indicative of rates of emptying, which are much too slow to prevent barotrauma due to the rapid (occurring in a fraction of a second) pressure changes that occur during water infrastructure passage. Thus, physoclistous species are likely very susceptible to barotraumas and likely much more sensitive than physostomous species; however, this remains to be investigated.

In addition to physoclists and physostomes, there is a third group of freshwater fishes that do not have a swim bladder and are therefore likely to have low susceptibility to barotrauma arising from Boyle's law. Juvenile Brook and Pacific Lamprey

Figure 5. Two-chambered swim bladder of the *Hypsibarbus lagleri*, a species endemic to the Mekong basin of South East Asia.

are two such species and were uninjured when rapidly decompressed in simulations of hydroturbine passage including exposure to pressures much lower (13.8 kPa) than commonly seen during turbine passage (Colotelo et al. 2012). Additionally, both species were held at this low pressure for an extended period of time (>17 min) without either immediate or delayed (>120 h) mortality (Colotelo et al. 2012). Together, these results suggest limited susceptibility to barotrauma via either the Boyle's or Henry's law pathways. In comparison to the Pacific Lamprey, when juvenile Chinook Salmon were rapidly decompressed to these same low pressures, more than 95% suffered mortal injuries (Brown et al. 2012b). Migratory fish species that reside in freshwater at least part of their lives and do not have swim bladders are not common but include Bull Shark (*Carcharhinus leucas*), freshwater Sawfish (*Pristis microdon*; a threatened species), and lampreys.

Other researchers have noted that fish without swim bladders had low susceptibility to barotrauma. For example, Bishai (1961) found larval Plaice (*Pleuronectes platessa* L.; 3.5–5.0 cm long) held at 202 kPa for 2–8 days were uninjured when decompressed over 5–10 min back to surface pressure (101 kPa). Similarly, Tsvetkov et al. (1972) found no damage to larval Atlantic Salmon (*Salmo salar*; 2–2.5 cm long; without a developed swim bladder) after being held at 101–606 kPa for 40 h or more and brought to surface pressure in less than 3 s. However, neither of these experiments involved reducing fish to pressures below surface pressure where barotrauma due to Henry's law (gas coming out of suspension in their blood and tissues) would have been anticipated.

IMPLICATION OF LIFE HISTORY AND BEHAVIOR

In addition to the physiological traits of fish, barotrauma research on freshwater species needs to be based on a template of ecology and behavior (Table 1). Understanding what life stages will be exposed to water infrastructure passage is critical to understanding the susceptibility of wild populations to barotrauma. The majority of research related to hydroturbine passage has been focused on seaward-migrating juvenile salmonids. Most salmonid species are semelparous (having a single reproductive episode before death) and, as such, the only life stage that may be affected by downstream passage is juveniles. There are, however, iteroparous (having multiple reproductive cycles over a lifetime) species that may pass through turbines as they migrate back to the ocean after spawning (e.g., Steelhead Trout [*Oncorhynchus mykiss*], Brown Trout [*Salmo trutta*], Atlantic Salmon, and Dolly Varden [*Salvelinus malma malma*]). Iteroparous species are also common in other bioregions such as South America, Asia, and Australia, where both adult and juvenile life stages may have to migrate downstream through hydropower and irrigation structures. In large floodplain rivers such as in South East Asia, South America, and Australia, egg and larval drift is a common life history trait (Baran et al. 2001; Humphries et al. 2002; Koehn and Harrington 2005; Godinho and Kynard 2009), and this mode of migration will increase the likelihood of encountering water infrastructure. Within North America, there are also many species (such as Paddlefish [*Polyodon spathula*], Walleye [*Sander vitreus*], and sturgeon [*Scaphirhynchus* spp.]) where eggs, larvae, or small juveniles can drift for long distances (Purkett 1961; Corbett and Powles 1986; Braaten et al. 2008). Early life stages are fragile and may be more susceptible to barotrauma than larger individuals because their bodies (swim bladder and other internal organs) are less robust (Tsvetkov et al. 1972), and the expansion of gas in the swim bladder may be more likely to cause damage relative to their body size. Understanding the ecology and timing of larval drift, as well as the time of first inflation of the swim bladder, will be critical in understanding their susceptibility to barotrauma. Additionally, more information is needed about physiological changes in larval physoclistous fish. They commonly have larvae with an open swim bladder but lose the connection between their swim bladder and esophagus as they develop. Identifying when this occurs may aid in understanding their increased susceptibility to barotrauma, important information for managing systems where these types of fish are present.

Larval drifting fish may also be susceptible to barotrauma due to expansion of metabolically produced gas. Brown et al. (2013a) noted barotrauma in the form of erratic swimming, death, and herniation-like abnormalities on the abdomen of larval White Sturgeon (*Acipenser transmontanus*) at the point when they first started feeding (8 days after hatching) but did not have an inflated swim bladder. They also noted gas in the intestines about 7 months after hatching that could also lead to barotrauma upon decompression.

Susceptibility to barotrauma is also likely to be influenced by the position fish occupy in the water column. Neutral buoyancy in fish is achieved by maintaining swim bladder volume constant, which is accomplished at deeper depths by having a higher gas pressure according to Boyle's law (see above). The depth and water pressure a fish has occupied prior to infrastructure passage (commonly referred to as "acclimation pressure") likely dictates the amount of gas a fish must have in its swim bladder to maintain neutral buoyancy because gases are compressible. If fish are benthic oriented, such as catfish, which are abundant riverine species in Asia and North and South America,

Table 1. Various traits that can influence the susceptibility of fish to barotrauma, along with example species.

Physiological, behavioral, or life history trait affecting susceptibility to barotrauma	Presence or absence	Susceptibility to barotrauma	Example species or project	References
The amount of free (undissolved) gas in the body				
Presence of a swim bladder	Yes	High	Chinook Salmon	Colotelo et al. (2012)
	No	Low	Pacific Lamprey	
Type of swim bladder	Open (physostomous)	Low	Chinook Salmon	Abernethy et al. (2001)
	Closed (physoclistous)	High	Bluegill	
Ability to expel gas out of the swim bladder through pneumatic duct	Better	Low	Large Rainbow Trout	Shrimpton et al. (1990)
	Poorer	High	Small Rainbow Trout	
Ability to fill the swim bladder with vasculature (rete)	Better	High	Bluegill	Harvey (1963); Fange (1983)
	Poorer	Low	Chinook Salmon	
Acclimation depth ability	Better	High	Burbot, Rainbow Trout	Fange (1983)
	Poorer	Low	Chinook Salmon	
Pressure exposure				
Acclimation depth	Deeper	High	Burbot	Stephenson et al. (2010); Fange (1983)
	Shallower	Low	Chinook Salmon	
Exposure pressure	Higher	Low	Irrigation weirs/spillways	Brown et al. (2012b)
	Lower	High	High-head dams	
Ratio of pressure change (acclimation pressure/ exposure pressure)	Higher	High	Hydroturbine	Brown et al. (2012a)
	Lower	Low	Bypass system	
Rate of ratio pressure change	Higher	High	Hydroturbine	Brown et al. (2012e)
	Lower	Low	Angling	
Life history				
Migrational patterns	More migratory	High	Murray Cod, Salmonids	
	More sedentary	Low	Trout Perch (*Percopsis omiscomaycus*)	
Larval or juvenile drift stage	Yes	High	Sturgeon, Murray Cod	Brown et al. (2013); Baumgartner et al. (2009)
	No	Low	Salmonids	
Structural integrity				
	High	Low	Adult fish	Baumgartner et al. (2009); Tsvetkov et al. (1972)
	Low	High	Larval or juvenile fish or eggs	

their initial acclimation pressure may be high and the lowest pressure (often referred to as “nadir”) experienced during hydroturbine passage will likely have a greater impact on swim bladder expansion. The ratio of pressure change (acclimation pressure/hydroturbine nadir pressure) experienced by the fish during passage is therefore likely a major factor dictating the level of injury a fish may experience. In contrast, fish that typically occupy shallower depths (including those species with buoyant drifting larval stages) require less gas to achieve the same swim bladder volume needed for neutral buoyancy and therefore may be less susceptible to barotrauma due to rapid decompression. However, research is needed to determine whether benthic-oriented fish are neutrally or negatively buoyant, because this will have implications for the impact of the pressure change on barotrauma.

IMPLICATION OF THE RATIO OF PRESSURE CHANGE ON SWIM BLADDER INJURY

Fish injury following rapid pressure change is predominantly associated with expansion of preexisting gases, which often leads to rupture of the swim bladder (Brown et al. 2012e). Thus, prediction of barotraumas in fish passing through hydrostructures requires a firm understanding of the degree to which gas expands within fish when they are decompressed. Based upon Boyle’s law (see above), one of the primary determinants of swim bladder volume change (and therefore likelihood of injury) will be the ratio of pressure change experienced by the fish during passage. This ratio may be as simple as dividing the pressure associated with the depth to which fish are acclimated and neutrally buoyant prior to passage with the nadir (lowest pressure) experienced during infrastructure passage. The following analogy acts to illustrate the importance of the ratio of pressure change rather than absolute pressure change to swim bladder volume and thus the potential for barotrauma. If a fish is brought to the surface (101 kPa) from an acclimation depth of 10 m (202 kPa) at which it is neutrally buoyant, it will experience a pressure change ratio of 2 (202 kPa/101 kPa), which implies that swim bladder volume would double (in the absence of body wall constraints). In this scenario, the absolute pressure change is 101 kPa (202 − 101 kPa; see Figure 6 for an example). The same doubling of swim bladder volume would also occur in a fish acclimated to surface water (101 kPa) that passes through a hydroturbine with a nadir pressure of 50.5 kPa because the ratio of pressure change is 2, even though the absolute pressure change is only 50.5 kPa, half the value of the example above. Understanding the significance of Boyle’s law and its potential impacts on fish can inform the hydraulic design of hydroturbines and other water control structures to control the nadir pressure and minimize the ratio of pressure change. This approach is currently being used by the U.S. Army Corps of Engineers to design new turbines to replace aging turbines at Columbia and Snake River dams (Brown et al. 2012a; Trumbo

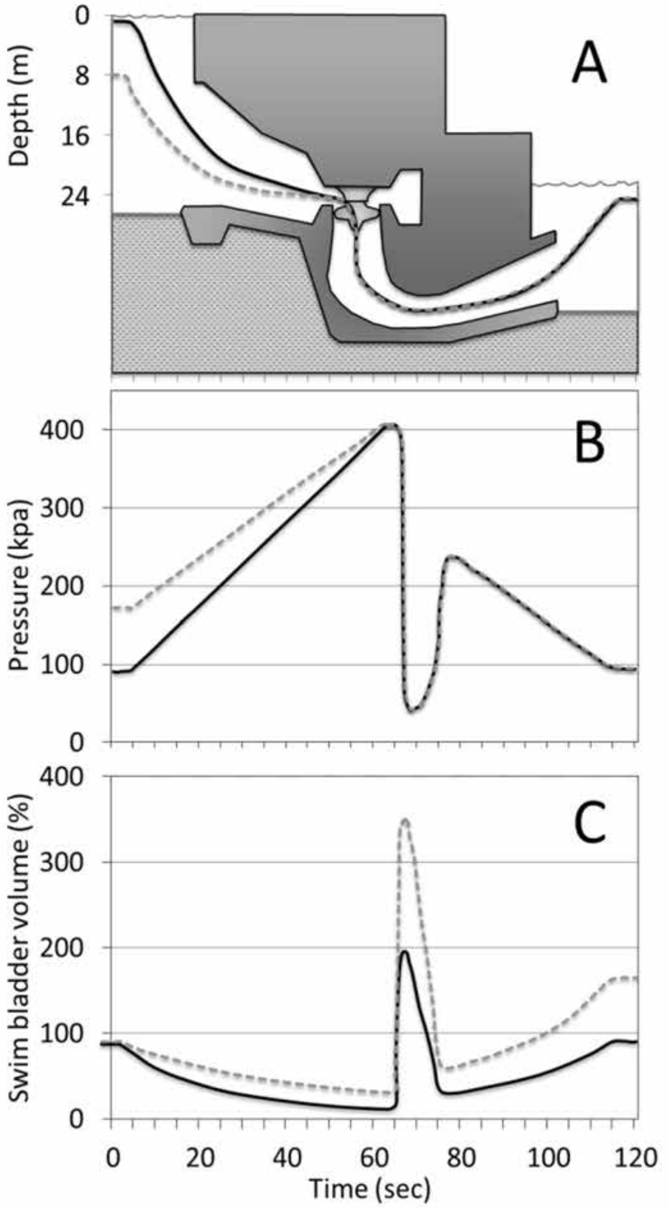

Figure 6. (A) Path through a hydroturbine, (B) an example of a pressure scenario that could be experienced, and (C) the swim bladder volume change (%) for fish neutrally buoyant at two different depths. The solid line represents a fish acclimated to near surface pressure, and the dotted line represents a fish acclimated to a depth of approximately 8 m (181.7 kPa).

et al. 2013). They recently contracted with industry to design and supply two new turbine runners for installation into Ice Harbor Lock and Dam.

DETERMINING ACCLIMATION PRESSURES AND CAPACITY FOR SWIM BLADDER INFLATION

Due to the importance of the ratio of pressure change in predicting the likelihood of barotrauma, it is necessary to determine the acclimation depth of fish as they approach hydrostructures and then determine the extent of the low pressures the fish will be exposed to during passage. Consideration must also be given to the swim bladder volume immediately prior to nadir exposure because some fish may expel gas from the bladder when exposed to pressure reductions associated with hydrostructure passage (Brown et al. 2012e) but before the nadir pressure exposure. Some different approaches can be used when trying to determine the acclimation depths of approaching fish, based upon the physiology of that particular species.

As a first approach, the depth from which fish are approaching structures should be known. Fish could be captured or monitored just upstream of dams or weirs under the assumption that this is the depth occupied during downstream migration. Identifying these migration depths could be facilitated by stratified sampling at different depths in the water column. Fish could then be captured and placed into a simple field hyperbaric chamber, where the pressure could be controlled and modified to determine the pressure or depth where the fish is neutrally buoyant. A neutrally buoyant fish appears level in the water column, instead of head down (positively buoyant) or head up (negatively buoyant; see Pflugrath et al. 2012). Another approach would be to move fish up and down in a water column (thus varying pressure) to determine at which depth they are neutrally buoyant. It may be necessary, depending on behavior, for some fish species to be sedated in order to determine buoyancy (Brown et al. 2005). Though these types of approaches have been used in laboratory research (Brown et al. 2005), field research into this area is needed.

The above methods may be fairly straightforward in fish with physoclistous swim bladders but more complicated in physostomes where gases can be expelled through a pneumatic duct. The latter may be minimized by sedating fish in a way to minimize stress such as slowly adding anesthetic to the water (similar to Brown et al. 2005); however, specific methods need to be developed.

Determining the maximum depth at which a fish species or life stage can attain neutral buoyancy is also very important information. This information can be used to predict susceptibility to barotraumas because it will influence the maximal ratio of pressure change that a fish may experience when passing through a specific hydroturbine or weir structure. Pflugrath et al. (2012) determined the maximum depth at which juvenile Chinook Salmon could maintain neutral buoyancy by attaching weights to the outside of the fish. As more mass was added, fish would gulp air at the water surface and fill their swim bladder until they were again neutrally buoyant. As more mass was added, the point at which fish could no longer attain neutral buoyancy was determined. Calculations of swim bladder volume and Boyle's law were then used to estimate the depth at which the determined maximum swim bladder volume resulted in neutral buoyancy (Pflugrath et al. 2012). This method is only useful for physostomous fish that only fill their swim bladder through gulping air at the water surface and forcing it through the pneumatic duct (such as Chinook and Sockeye Salmon; Harvey 1963; unlike fish like American Eels [*Anguilla rostrata*], which have an open swim bladder and an active retia).

Determining the maximum depth of neutral buoyancy in physoclistous fish or physostomous fish with a functioning rete

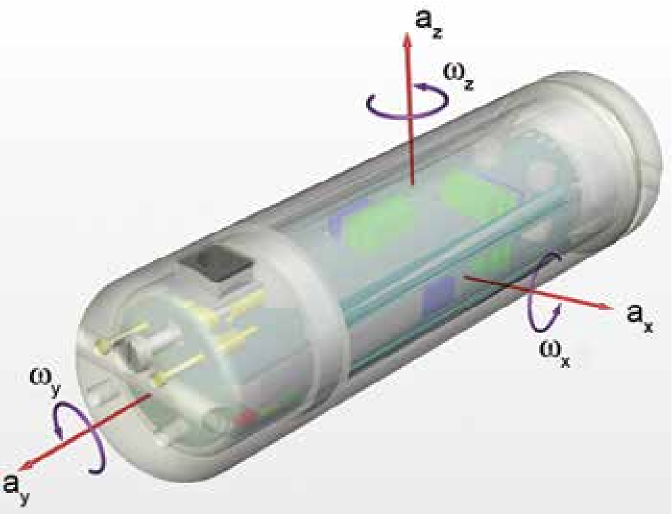

Figure 7. The mulitsensor fish surrogate showing the location of the measurement axes for the three rate gyros (that measure angular velocity, ω), three linear accelerometers (that measure the acceleration, a), and pressure transducers (Deng et al. 2007b).

could be conducted by slowly increasing the pressure in a hyperbaric chamber until neutral buoyancy can no longer be attained. The rate of swim bladder inflation in these fish is slow and variable among species and life stages (Fange 1983). This will have to be taken into account in experimental designs to assess maximum acclimation depths because some species may need to be held under pressure for long periods to determine the bounds of their buoyancy regulatory abilities. In addition, if pressures are increased too quickly, fish may not be able to attain neutral buoyancy at depths as great as those treated with slower increases in pressure. For physostomous fish, it may be necessary to remove all gas bubbles from the chamber to ensure that the swim bladder is inflated solely through the rete and not by gulping compressed gas bubbles inside the chamber, which could otherwise overestimate acclimation depths.

DETERMINING EXPOSURE PRESSURES DURING FISH PASSAGE

The nadir pressure is critical in determining the ratio of pressure change and is an essential parameter in predicting barotraumas as fish pass through hydro or irrigation structures. This pressure can be estimated using computational fluid dynamics models or can be determined in situ using a multiple sensor fish surrogate (Deng et al. 2007b). The latest generation 6-degree-of-freedom version of this device is an autonomous sensor package, consisting of three rate gyros, three acceleration sensors, a pressure sensor, and a temperature sensor (Deng et al. 2007b; Figure 7). It was developed at Pacific Northwest National Laboratory for the U.S. Department of Energy and U.S. Army Corps of Engineers to characterize the physical conditions and physical stressors to which fish are exposed as they pass through complex hydraulic environments. This device is currently 24.5 mm in diameter and 90 mm in length, weighs 42 g, and is nearly neutrally buoyant in freshwater. Although this makes it similar to the size and density of a migrating yearling Salmon smolt, this does not preclude its usefulness in systems where juvenile salmonids are not present. The multiple sensor fish surrogate provides actual measurements of pressure, the three components of linear acceleration (up–down, forward–back, and side-to-side), and the three components of rotational velocities (pitch, roll, and yaw) and internal temperature at a sampling frequency of 2,000 Hz, extending from its release location to the end of the particular passage.

For barotrauma research, the most important parameter to measure from a multiple sensor fish surrogate is pressure, which can be used to determine pressure profiles, estimate the depth of the fish during passage, and determine passage rates through different regions of a hydropower or weir structure. For example, the pressure profile of a typical turbine passage is characterized by an increase in pressure as fish pass downward toward and through the turbine intake, a rapid decompression (typically significantly below surface pressure in a fraction of a second) as the fish pass the turbine blade and a slow return to surface pressure through the draft tube (examples are provided in Brown et al. [2009] and Stephenson et al. [2010]). For passage through an undershot irrigation weir (where bypass water flows underneath the weir), the pressure profile reveals a slow increase in pressure upstream of the gate and a rapid decompression (<1 s) to slightly below surface pressure under the gate and a return to surface pressure in the tailwater.

The rate of decompression mentioned above is an important consideration when determining barotrauma susceptibility, because it can affect a physostomous species' ability to expel gas from the swim bladder. Brown et al. (2012e) found that when decompression occurred slowly (0.6–1.0 kPa/s), Chinook Salmon expelled gas more frequently and thus avoided barotrauma when compared to those decompressed at rapid rates (758.4 to 3,874.9 kPa/s; Brown et al. 2012b). Thus, clearly the rate of decompression associated with structure passage is crucial in predicting impacts; however, this information is often lacking and is needed.

Multiple sensor fish surrogates have been widely used to evaluate hydroturbine, spillway, and other fish bypass systems as well as pump storage and irrigation weir facilities. For example, it was deployed at different elevations and operation conditions to evaluate the biological performance of the advanced hydropower turbine (AHT) at Wanapum Dam (Washington State) to support its relicensing application. The AHT was designed to improve operational efficiency and increase power generation while improving the survival for fish passing through the turbines. The multiple sensor fish surrogate measurements confirmed that the AHT provided a better pressure and rate of pressure change environment for fish passage and improved the passage of juvenile salmon at Wanapum Dam (Deng et al. 2010). The multiple sensor fish surrogate is undergoing design changes such as the size, aspects of function, deployment and recovery, availability, and cost to extend its range of use and provide information for the development of fish-friendly hydrosystems internationally.

MODELING THE PROBABILITY OF MORTALITY OR INJURY

Once the range of natural acclimation pressures and the exposure pressures to be expected during passage through the hydraulic structures are determined, laboratory experiments can be conducted to relate the rate and magnitude of decompression to the expected mortality and injury of fish during infrastructure passage (Brown et al. 2009, 2012b, 2012e). These experiments involve exposing fish to pressure profiles that simulate passage through a hydroturbine or irrigation infrastructure under a range of ratios of pressure change. Such a laboratory approach for the simulation of infrastructure passage is being used to great effect to guide engineers when replacing turbines at dams in the Pacific Northwest of the United States (Brown et al. 2012a). However, a relationship between ratio pressure change and mortality and injury has only been determined for one species and life stage—juvenile Chinook Salmon (Brown et al. 2012b, 2012c)—and is likely to be species and life stage specific.

The type of equipment needed to simulate the different types of infrastructure passage can vary. Simulation of rapid decompression associated with hydroturbine passage requires sophisticated pressure chambers such as those described by Stephenson et al. (2010). These chambers are able to replicate the large ratio of pressure changes commonly observed during hydroturbine passage, which include nadirs well below atmospheric to pressures approaching 0 kPa. However, systems that only need to simulate smaller ratio pressure changes with nadirs of surface pressure (as may be characteristic of irrigation structures) or fairly slow pressure changes may be comparably simpler and inexpensive to construct. Simple systems could also be used in the laboratory to increase and decrease pressures to examine the capacity of fish to regulate their buoyancy.

The ultimate goal of this type of laboratory work should be to model the relationship between the ratio of pressure change fish are exposed to and the probability of injury or mortality. For all of the reasons previously mentioned, the ranges of ratio pressure change to be tested should be informed through careful consideration of the acclimation pressures prior to passage and the range of nadir pressures a fish is likely to be exposed to when it encounters various infrastructures throughout its life history. Once a relationship between mortality and pressure change is established with suitable statistical rigor, it is theoretically possible to predict the mortality of that species and life stage to any passage scenario, and it is only necessary to know the acclimation depth of the fish prior to passage and the nadir pressure expected at the hydropower or irrigation structure.

It is rarely practical to hold fish for extended periods following experimentation, and these holding conditions could vary widely and not represent field conditions. For these reasons, it may be possible to infer delayed mortality from the injuries immediately evident following rapid decompression during laboratory studies. McKinstry et al. (2007) combined the likelihood that fish had certain injuries present following simulated turbine passage with the likelihood of mortality to establish a mortal injury metric. Brown et al. (2012b, 2012c) subsequently determined that the likelihood a fish will be mortally injured relates to pressure exposure using the following equation:

$$\text{Probability of mortal injury} = \frac{e^{-5.56+3.85*LRP}}{1+e^{-5.56+3.85*LRP}},$$

where LRP is the natural log of the ratio of pressure change (acclimation/nadir pressures) to which the fish are exposed.

Techniques similar to those used by McKinstry et al. (2007) and Brown et al. (2012b) could be used to derive mortality metrics for other species. Brown et al. (2012e) determined that the ability of physostomes to expel gas from their swim bladder increases the variability in mortality when they are exposed to pressure changes. However, because physoclists cannot expel gas when rapidly decompressed, the anticipated level of variation is expected to be lower. Consequently, though Brown et al. (2012b) tested over 5,000 juvenile Chinook Salmon to determine the relationship between pressure change and fish damage, smaller sample sizes will likely suffice for physoclistous fish. However, to guide the international development of a broad range of sustainable hydro and irrigation structures, it is important to characterize the effect of pressure changes on a diverse range of physostomous and physoclistous species at different life history stages.

Laboratory experiments to determine the relationship between pressure changes and fish damage must take into consideration the depth to which fish are acclimated prior to water infrastructure contact, as well as the limits of fish buoyancy compensation. Researchers, managers, and turbine designers should be very careful when interpreting existing literature related to barotrauma in fish. Even 40 years ago, researchers like Tsvetkov et al. (1972) were concerned about the underestimation of fish injury associated with pressure changes due to methodological problems and inaccuracies. Examples provided by Tsvetkov et al. (1972) include tests where fish were not allowed to properly acclimate before being exposed to pressure reductions, such as placing physoclistous fish under high pressure and allowing them inadequate time to acclimate (just a few minutes, which is not adequate time for the swim bladder to be filled by the retia). They also highlighted studies of physostomous species where fish were acclimated to high pressures without access to air, thus not allowing fish to acclimate and fill their swim bladder.

These types of problems are not uncommon and also exist with a series of early experiments conducted by Abernethy et al. (2001, 2002, 2003). In these studies, juvenile Rainbow Trout and Chinook Salmon were placed into pressure chambers and held at surface pressure (101 kPa) or the pressures present at 19 m (191 kPa) of depth for 16–22 h. Fish were then exposed to rapid pressure reduction to pressures approximately in the range of 2–10 kPa (although the actual lowest pressures fish were exposed to during all tests were not noted). However, because the fish held at 191 kPa were not provided with an air surface, they could not fill their swim bladder and become neutrally buoyant. Thus, results indicated that fish approaching turbines at 19 m would have the damage similar to that of fish approaching at surface pressure. However, these unrealistic results were part of a chain of research that developed into the understanding of the importance of acclimation in barotrauma experiments (Stephenson et al. 2010).

Caution should also be taken when interpreting some field-based research and scale-model investigations of turbines. For many studies of turbine passage survival, balloons and radio transmitters are attached to fish to aid in their retrieval (see Mathur et al. [1996] for an example). Before release from the surface of a dam, the balloons are injected with a liquid, leading to a chemical reaction that creates gas. This allows fish to pass through the turbine while the balloons are deflated and then be recaptured in the tailwater of a dam after the balloons have inflated. Though these studies have provided a large amount of valuable data on the effects of turbine passage, the information they provide related to barotrauma is likely a best-case scenario because fish are typically injected into turbine entrances from surface pressure. In addition, some studies done on scaled models of turbines (Cook et al. 2003; Electric Power Research Institute and U.S. Department of Energy 2011), which hold promise for reducing strike and shear injuries to fish, were conducted by releasing fish into the scale turbine at surface pressure. Thus, these studies also likely provide a best-case scenario for barotrauma-related injuries.

FIELD VALIDATION OF MODELED MORTALITY RELATIONSHIPS

Any modeled data will benefit from ground-truthing to ensure that the predictions generated in the laboratory adequately reflect the complexities experienced in real-world systems. The mortality models described above are no exception. When possible, estimates made in the laboratory can be verified on existing or pilot hydroturbine or weir structures. The development of new designs is progressing at a rapid rate, particularly in the small-scale hydropower market (Baumgartner et al. 2012). Therefore, there are great opportunities for researchers to work with developers to validate the predictions made in the laboratory when assessing the suitability of pilot projects. In some parts of southeastern Australia, state fisheries management agencies are already requiring developers to initiate field validation of new small-scale hydro designs as a preferred intermediate step between laboratory studies and possible large-scale adoption of any technology (Baumgartner et al. 2012). Field validations may involve running live fish through facilities in parallel with multiple sensor fish surrogates, with the measured mortality rates and ratio of pressure changes compared with laboratory modeling. In the end, this will improve the confidence that developers and fisheries management agencies have in laboratory generated predictions.

Another factor that is critical for increasing the confidence in field results is to design experiments so that injury and mortality estimates are not biased. One important consideration is to ensure that all fish are acclimated to appropriate depths (corresponding to natural migration behavior) prior to being exposed to infrastructure passage. This has often not been the case in field examinations, as pointed out by Stephenson et al. (2010).

Another consideration involves the use of telemetry tags to estimate the route of passage and survival of fish. The mass of the tag relative to the mass of the fish (referred to as “tag burden”) has been shown to influence growth, behavior, swimming performance, and survival for tagged fish when compared to untagged conspecifics (Zale et al. 2005; Brown et al. 2010), and is of particular importance for fish exposed to rapid changes in pressure. Carlson et al. (2012) demonstrated that for juvenile Chinook Salmon exposed to rapid decompression associated with simulated turbine passage, the probability of injury and mortality increased as tag burden increased. Fish carrying a negatively buoyant telemetry tag increase the amount of gas forced into the swim bladder to offset the additional mass and achieve neutral buoyancy, making them more susceptible to barotrauma (Gallepp and Magnuson 1972; Perry et al. 2001). In addition, having a telemetry transmitter inside the body cavity may limit the amount that a swim bladder can expand before it ruptures or causes compression-related injuries. Therefore, field estimates of mortality that are based upon tagged fish have the potential to overestimate the severity of barotrauma injury. To overcome this, we recommend using the smallest tag possible to minimize tag burden or a neutrally buoyant, externally attached tag (tag burden of 0%; Deng et al. 2012; Janak et al. 2012; Brown et al. 2012d, 2013b), when examining survival of fish exposed to rapid decompression associated with infrastructure passage.

AN ADAPTIVE APPROACH TO SUSTAINABLE DEVELOPMENT

Recently there have been renewed global efforts in the expansion of hydropower projects. The retrofitting of new hydro projects to existing structures has also been encouraged by the U.S. Department of Energy to increase the output of American hydropower capability (Hadjerioua et al. 2012). In some parts of the world, established irrigation networks are being explored for their potential to support new economies relating to power generation (Botto et al. 2010). In many other regions, new dams are being planned. As part of Brazil’s decennial plan (MME/EPE 2011), 48 hydropower dams are proposed for construction by 2020. Most of these would be in the Amazon and Tocantins-Araguaia hydrographic regions. These dams are likely to threaten fish diversity of the Amazon (20% of the world’s freshwater fishes, representing about 1,400 species) by regulating flows and disrupting important fish migrations (Rosa and Lima 2008). It is a similar story for the world’s largest inland fishery in the Lower Mekong River, where it is predicted that construction of 11 mainstem dams will lead to a major decline in fish populations, significantly compromising food security (Halls and Kshatriya 2009). If these dire scenarios are to be avoided, it will be necessary to ensure safe fish passage at new and existing structures, with management decisions underpinned by rigorous science.

Based on the information provided in this review, we recommend a logical staged approach to conducting the barotrauma research that will be necessary for refining infrastructure design throughout the world (Figure 8).The first stage involves conducting the field or desktop investigations necessary to determine which species and life history stages are of interest. The majority of barotrauma research to date has focused on the susceptibility of juvenile Chinook Salmon, largely driven by the legislative need to protect this threatened species during its critical seaward-migration in the U.S. Pacific Northwest, where a large number of hydropower facilities could negatively influence their survival. In other large river systems of the world, including the Mekong River in Southeast Asia, the Amazon in South America, and the Murray-Darling River in Australia, a diverse range of species and life history stages undertake downstream migrations (Barthem et al. 1991; Araujo-Lima and Oliveira 1998; Humphries and King 2004; Lintermans and Phillips 2004; Baran and Myschowoda 2008) and are therefore at risk of injury and mortality at existing and proposed hydropower and irrigation operations. For fisheries scientists wishing to embark on barotrauma-related research in these regions, the decision regarding which species and size classes to prioritize for study is daunting. Such decisions could be aided by considering the many factors associated with the susceptibility to barotrauma (see Table 1), including both ecological and biological considerations. By assigning weighted scores corresponding to the factors for each species in an assemblage of fish, multivariate classification approaches could be used to identify key groupings of fish based upon similarity in barotrauma vulnerability (see Table 1). Choosing some fish and life history stages from the higher vulnerability groupings may provide a good starting point for experimentation.

Once the species of study have been selected, a combination of field and lab testing and modeling can both determine the depth of neutral buoyancy as fish approach structures during migration (or acclimation depth) and the expected range of exposure pressures during infrastructure passage (Figure 8). This will provide a range of ratio pressure changes that fish can be subjected to in experimental pressure chambers and, from this, injury or mortality relationships can be modeled. Care must be taken during this experimentation to ensure that fish are properly acclimated (acclimated to the range of pressures that reflect depths where fish are neutrally buoyant as they approach structures). Fish acclimated to surface pressures are likely to provide results that are not necessarily representative of fish in the natural environment because acclimation depth is a very important parameter (Tsvetkov et al. 1972; Stephenson et al. 2010).

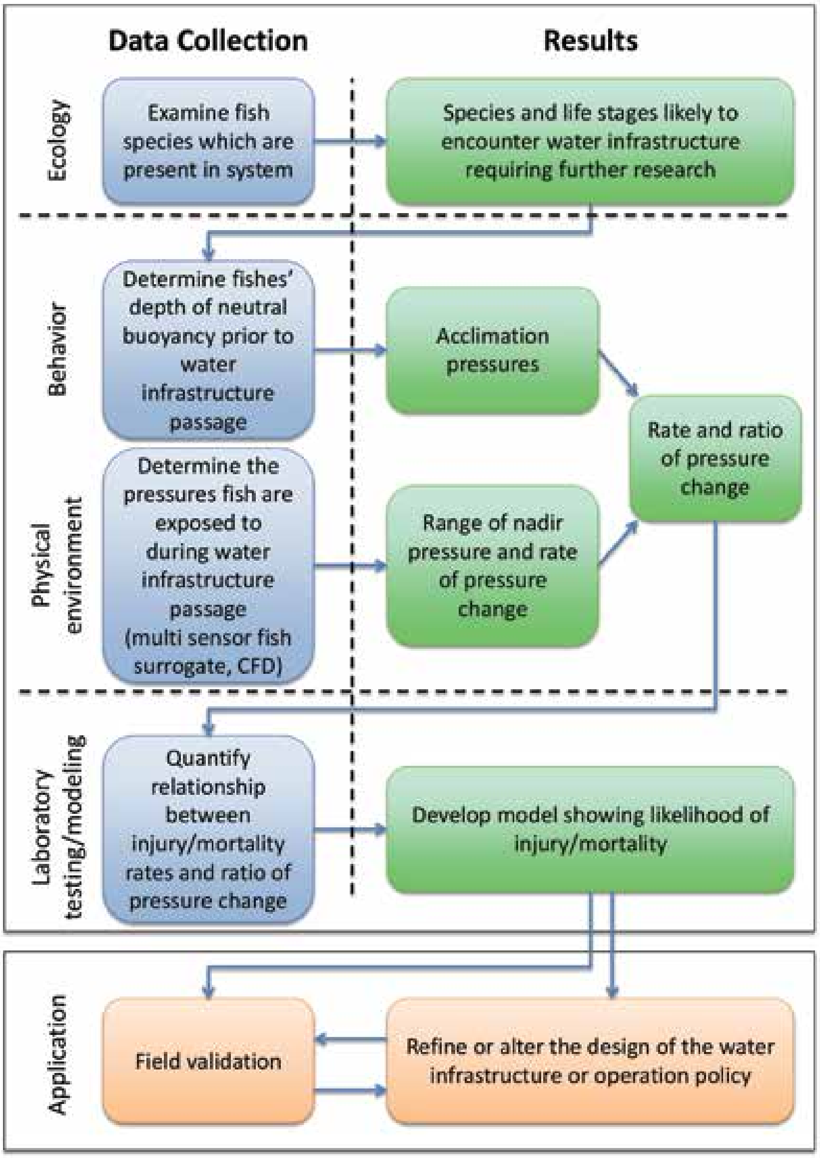

Figure 8. Recommended barotrauma research framework showing logical flow of activities and linkages with industry under an adaptive management model.

The models generated by laboratory experiments can then be used to refine infrastructure design, with models and designs further validated during pilot field trials. This field validation and testing is seen as a critical link in the adaptive management loop that will ensure that fisheries scientists and engineers keep the research and development applied and ultimately targeted on the goal of promoting sustainable water resource development.

Minimizing fisheries losses at water infrastructure is a global problem, and major investment will be needed to promote innovative technology if the current fisheries losses throughout the world are to be abated. A global problem requires a global solution, and we therefore encourage international cooperation in future research efforts. There are many similarities in fish species among different regions of the world and, thus, international collaboration will greatly reduce redundancy. For example, catfish species are common throughout North and South America, Asia, and Europe, and sturgeons (a type of fish with drifting larval stages) are common in North America, Asia, and Europe.

Similarly, larval drift will be a key consideration in many parts of Australia, Asia, and South America and also occurs among North American species. We are at a time where technology allows us to initiate downstream passage research among many species at a global scale using standardized approaches. Such a global approach could provide a more rapid advancement of science and engineering while minimizing duplication of effort.

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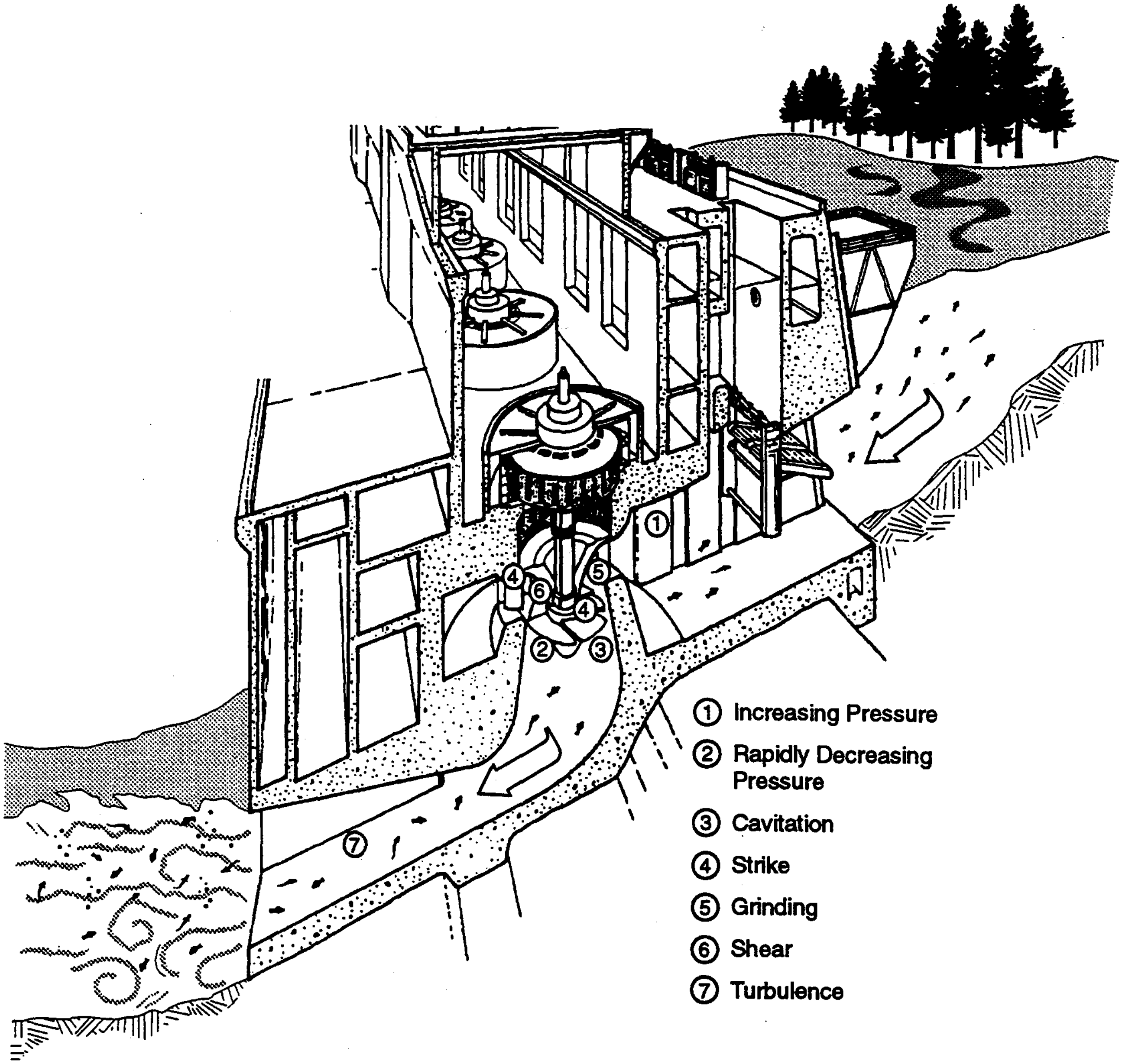

U.S. Department of Energy
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Executive Summary

A review of the literature related to turbine-passage injury mechanisms suggests the following biological criteria should be considered in the design of new turbines:

Pressure - Pressure increases of the magnitude found in hydroelectric turbines do not appear to cause direct damage to entrained fish. Pressure decreases within the turbine are a greater concern. Because the decrease to subatmospheric pressures is virtually instantaneous, fish with swim bladders will be unable to vent gas from the rapidly expanding swim bladder. The swim bladder may distend or rupture, causing direct mortality or reduced ability to escape predators in the tailrace. Allowing minimum pressures within the turbine to fall to no less than 60 percent of the value to which fish are acclimated should protect most fish from direct effects of low pressures.

Cavitation - Turbine designs that minimize pressure reductions to no greater than 60 percent of ambient will not cavitate, and cavitation-related injury to fish will not occur. If cavitation cannot be entirely prevented, introduction of air or oxygen bubbles may serve to mitigate adverse effects by cushioning the shock waves created by the collapsing water vapor bubble.

Shear and Turbulence - Laboratory studies to date have exposed fish to a high-velocity water jet in a static tank. These tests examine the injury and mortality rates of fish in which high shear values are applied to only a part of the fish's body. Other, larger-scale effects of shear on entrained fish, including elongation, compression, torsion, rotation, and deformation, have only been studied for fish eggs and larvae. At high levels, these forces could cause injury and mortality among larger fish. At lower, non-injurious levels, fish would be disoriented by shear and turbulence and may suffer greater indirect mortality (predation) below the turbine discharge.

Mechanical Injury - Because of numerous variables related to the entrained fish (e.g., individual size, condition, and behavior) and the relationship of the fish to the runner and other turbine structures (e.g., region of passage, orientation, and relative velocity), the probability of injury from strike and grinding cannot be precisely estimated for any turbine. Some strictly biological factors, such as the species, length, and mass of entrained fish, influence the injury/mortality rate but cannot be altered by the turbine designer. Aspects of the turbine system that could be modified in order to minimize strike injury are discussed.

Among the injury mechanisms considered in this report, the effects of water pressure on fish seem to be the best understood. The influence of pressure increases and decreases have been studied for a variety of species, so that reasonable biological criteria that will protect turbine-passed fish can be determined. Strike and cavitation appear to be similar in that the effects are probabilistic; it is generally accepted that collision with the blade at sufficient velocity or proximity to a collapsing cavitation bubble will cause injury and death. Expanding this database with new information collected under controlled laboratory conditions would not be difficult. The greatest uncertainties associated with strike and cavitation deal with understanding how fish behavior can alter the risk of injury. We do not know whether behavioral responses to stimuli (changes in illumination, sounds, and flow fields) lead fish into areas within the turbine of lesser or greater risk, or whether the behavioral response is reliable enough to point toward turbine design changes. Least understood are the effects of shear forces and turbulence on fish.

Adverse water quality may also alter the effects of the physical injury mechanisms considered in this review. The mortality ultimately resulting from physical stresses such as pressure changes or strike may be increased by suboptimal water temperatures (either high or low), low dissolved oxygen concentrations, supersaturated nitrogen gas, and high levels of debris and other suspended materials. These water quality factors are usually optimized in laboratory studies. At operating turbines water quality problems may add to the overall level of stress and may contribute to greater-than-expected turbine passage mortality.

Most of the studies of turbine-related injury mechanisms have examined only direct mortality. Much less is known about indirect mortality, i.e., the influence of sublethal turbine-passage stresses on later mortality due to predation or disease. Further investigations would be useful to ensure that reductions in direct mortality due to turbine design changes are not nullified by high levels of indirect mortality.

Coordinated laboratory and field studies are needed to reduce uncertainties about the relative importance of the potential injury mechanisms associated with turbine passage. Pressure changes are easy to study in the laboratory under controlled conditions. The rapid pressure increases and decreases experienced by an entrained fish can be reliably simulated in the laboratory, and as a result more is known about this stress than any other. On the other hand, techniques for studying fluid shear stresses and turbulence are not well developed. Shear and turbulence have been difficult to recreate in laboratory experiments, and little is known about the levels of injury, direct mortality, and indirect mortality (predation and disease) that may result from exposure to these stresses in a hydroelectric turbine.

The substantial developments in video and hydroacoustics techniques in recent years might used to visualize the path taken by entrained fish in an operating turbine. This information is needed to develop a better understanding of the risk of strike and grinding, as well as the pressure vs. time, shear vs. time, and turbulence vs. time histories experienced by fish passing through existing and advanced turbines. Low-light sensitive video cameras, perhaps in conjunction with light-emitting tags attached to the fish, show promise for tracking the path of entrained fish. Split-beam hydroacoustics techniques can potentially detect and record a fish's movements in three dimensions with little concern about altering the fish's behavior. However, the ability of hydroacoustics to track fish reliably inside of a turbine, under conditions of high velocities, high turbulence, crowding of entrained fish, and electronic interference, has yet to be demonstrated.

We evaluated the literature on fish behavior as it relates to passage of fish near or through hydropower turbines. The goal is to foster compatibility of engineered systems with the normal behavior patterns of fish species and life stages such that entrainment into turbines and injury in passage are minimized. In particular, we focused on aspects of fish behavior that could be used for computational fluid dynamics (CFD) modeling of fish trajectories through turbine systems. Downstream-migrating salmon smolts are generally surface oriented and follow flow. They can be diverted from turbines by surface spills, with varying degrees of effectiveness. Smolts orient to the ceilings of turbine intakes but are horizontally distributed more evenly, except as affected by intake-specific turbulence and vortices. Smolts often enter intakes oriented head upstream, but may change orientation in the flow fields of the intake. Non-salmonids are entrained most often from vicinities of shorelines and episodically, suggesting accidental capture of schools (often of juveniles or in cold water) and little behavioral control during turbine passage. Models of fish trajectories should not assume neutral buoyancy throughout the time a fish passes through a turbine, largely because of pressure effects on swim bladders. Fish use their lateral line system to sense obstacles and change their orientation, but this sensory-response system may not be effective in the very rapid passage times of turbine systems. Effects of pre-existing stress levels on fish performance in turbine passage (especially as they affect trajectories) are not well known but may be important. There are practical limits of observation and measurement of fish and flows in the proximity of turbine runners that may inhibit development of much information that is germane to developing a more fish-friendly turbine.

Based on our review of fish behavior in relation to hydropower facilities, we provide the following recommendations to guide both turbine system design and additional research:

1. The first priority for a fish-friendly turbine *system* in migratory salmonid waters should be one that bypasses as many downstream-migrating fish as possible along these fish's natural surface-oriented migration pathway away from deep turbine intakes.

2. Further report evaluation and data collection and analyses are needed to specify fish cross-sectional distribution in a mathematically rigorous way for species, sizes, and intake geometries in order to specify quantitatively the fish trajectories through turbines.

3. Further analysis is needed using hydroacoustic and underwater television data, both new and as related to submerged traveling screens, as indicators of species- and size-specific fish orientation as they enter turbines.

4. Considerably more justification would be needed for commitment of major expenses for fish-friendly turbines in freshwaters occupied by non-migratory species.

5. Simulation of many non-salmonids as passive objects in CFD modeling seems appropriate.

6. The significance of differences from neutral buoyancy and of changes in buoyancy during fish trajectories through a turbine should be established from modeling studies of fish with a range of constant and changing densities.

7. Further studies of fish's reaction times to structures or high shear/turbulence areas within the turbine passage are needed. Models can tentatively assume that orientation of fish as they enter the scroll case will be retained as they transit the turbine itself (or at least that the fish will not be able to control its orientation in a turbulent environment), under

the assumption that reaction times are too long for the rapid flow rates.

8. Research on the orientation in and use of unsteady flows by migrating juvenile salmonids is needed.

9. Testing of fish behavior in turbines should include background information on pre-existing stress levels, and experiments should use fish in both test and control lots that have been given known amounts of prior stress.

10. Innovative means for obtaining information on fish behavior near turbine runners should be pursued, but there should be realistic expectations about the feasibility of this research.

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1. Introduction

Hydroelectric power plants can impact fish populations by interfering with both upstream and downstream movements. These impacts are most serious for anadromous fish species, such as salmon, steelhead, and American shad, whose life histories require passage between marine and freshwater environments. Fish ladders or lifts are commonly installed to provide for upstream movements around dams, whereas a wide variety of screens and other mitigative measures have been employed to reduce the numbers of downstream-migrating fish that are entrained in the intake flow and pass through the turbines (Sale et al. 1991; Čada and Sale 1993; Francfort et al. 1994; OTA 1995).

Turbine intake screens and other related measures have had mixed success in promoting safe downstream passage of fish. At some hydropower plants, these measures have significantly reduced turbine entrainment, but at other plants unacceptably large numbers of fish still suffer turbine-passage mortality or are harmed by the fish passage mitigation measure itself (Čada and Francfort 1995; Čada In Press). Even effective, well-designed screening and bypass systems may protect only a portion of the fish entrained in the intake flows; the remainder will pass through the turbines. Hence, there is a need not only to develop fish screens to reduce turbine passage, but also to develop turbines that increase the survival of fish that are entrained.

Recognizing the need for multiple solutions to the downstream fish passage problem, the U.S. Army Corps of Engineers conducts research aimed at reducing mortality of fish (especially salmon) caused by passage through Kaplan turbines at their hydropower plants (USACE 1995). On a wider scale, the U.S. Department of Energy, through its Advanced Hydropower Turbine System Program, supports the development of "environmentally friendly" turbines, i.e., turbine systems in which environmental attributes such as entrainment survival are emphasized (Brookshier et al. 1995). Advanced turbines would be suitable for installation at new hydropower facilities and to replace aging turbines at existing plants. It is expected that these turbines could permit the efficient generation of electricity while minimizing the damage to fish and their habitats.

Development of advanced, environmentally friendly hydroelectric turbines requires knowledge of the physical stresses (injury mechanisms) that impact entrained fish and the fish's tolerance to these stresses. Possible causes for entrainment mortality, physical injuries, sublethal physiological stress, and disorientation are many and varied; a recent workshop (USACE 1995) concluded that entrainment injuries could result from rapid and extreme water pressure changes, cavitation, shear, turbulence, and/or mechanical injuries (strike, grinding and abrasion). Instrumentation of turbines and the increasing use of Computational Fluid Dynamics (CFD) modeling can provide considerable information about the levels of each of these potential injury mechanisms that can be expected within the turbine. Frequently missing, however, are data on the responses of fish to these levels of stress. For example, the sensitivity of fish to the levels of shear or turbulence that are predicted to occur in a turbine is not well understood, and as a result we do not know what effect altering the amount of shear in a new turbine design will have on survival. Passage through different regions of the turbine (e.g., close to the blade hub or out near the blade tip) will entail exposure to different pressure, shear, and turbulence regimes and different probabilities of mechanical injuries. The behavior of the fish while passing through the turbine may alter the passage route, leading to greater or lesser exposure to these injury mechanisms than would be expected from consideration of the entrained organism as a passive object.

The purpose of this report is to review published laboratory bioassays and similar studies of the responses of fish to the component stresses of turbine passage: pressure, cavitation, shear, and blade strike (*Section 2*). We have examined each of these component stresses of turbine entrainment with the goal of deriving biological criteria for the turbine designers. In many cases there are few or no data to support quantitative biological criteria, so in *Section 3* we describe laboratory and field experimental techniques that could be used to fill gaps in existing information. Finally, we examine the role of behavior in mediating the effects of turbine passage stresses (*Section 4*). Entrained fish are not necessarily passive objects; by their behavior during turbine passage they may be able to swim out of (or into) areas of the turbine that cause damage. The published literature on fish behavior may suggest whether particular species or sizes of fish are likely to exhibit predictable, directed movements, knowledge of which would be useful to turbine designers.

2. Review of Literature Relating to Injury Mechanisms Associated with Turbine Passage

Phases I and II of the U.S. Department of Energy's Advanced Hydropower Turbine System Program (AHTSP) involve considerable Computational Fluid Dynamics (CFD) modeling and engineering design studies to develop novel designs for fish-friendly turbines, i.e., turbines in which mortality of entrained fish is small. In order to accomplish this, the designers need quantitative environmental criteria as input. That is, the engineers need numbers which define a "safety zone" for fish within which all of the injury/mortality mechanisms experienced by turbine-passed fish (water pressure changes, shear forces, cavitation, and chance of mechanical strike) are at acceptable (definable) levels for survival. If one of these injury mechanisms has over-riding importance compared to others, the designers could focus their efforts to "design out" this stress in the new generation of turbines.

Literature reviews of turbine-passage mortality studies have often focused on field studies of "high-mortality" and "low-mortality" turbines in an attempt to discern the design causes for the differences in mortality. Such studies of the whole-system performance have the advantage of addressing real-world conditions and will provide necessary base case numbers for turbine-passage mortality at existing turbines. Field studies of entrainment mortalities at particular sites are limited because they reflect the impacts on survival of all of the injury mechanisms together, but cannot distinguish effects of individual stresses. The relative importance of each of these stresses is difficult to discern from field studies at hydropower plants, especially when the observable physical damage to fish is similar for many of the stresses

This first section of the report examines the injury/mortality mechanisms associated with turbine entrainment, as studied separately under controlled conditions in the laboratory and field rather than in combination at hydropower sites. This literature can be used to derive biological criteria for the design engineers. For example, pressure zones are defined within which fish would not be harmed, and outside of which pressures could cause mortality. These pressure values would provide the CFD modelers and turbine design engineers with target values for their design work. "Safety zones" for other components of entrainment (e.g., shear, cavitation, blade strike) are also proposed where sufficient information exists in the literature. Gaps in available information are identified in order to direct future investigations.

Injuries and mortalities among fish passing through a hydroelectric turbine can result from several mechanisms, including rapid and extreme water pressure changes, cavitation, shear, turbulence, and mechanical injuries (USACE 1995). Each of these mechanisms can cause *physical* injuries that are severe enough to kill the fish directly; these include descaling; loss of the protective mucous layer; torn gill covers; decapitation; bruises; burst swim bladder; hemorrhaging; and other internal injuries. If the entrainment stresses are not immediately lethal, the fish may nonetheless be *physiologically stressed* and *disoriented*, so that they are more susceptible to predation in the tailwaters

(Mesa 1994) or disabled so that they later succumb to disease (indirect mortality). The following sections review literature relevant to understanding the importance of each of these factors to turbine-passage mortality. Most of the studies examine only direct mortality; much less is known about the effects of sublethal injuries on indirect mortality.

2.1 Pressure Effects

Pressure at any point is the force per unit area acting upon the point. Pressure is commonly expressed as pounds per square inch (psi) in the English system and as newtons per square meter (N/m^2) in the International (SI) system. An alternative unit of pressure in the SI system is the pascal (Pa); one pascal is equal to one N/m^2. Water pressures normally experienced by fish are most easily expressed as kilopascals (kPa). Pressures have been expressed in a variety of units in the studies reviewed in this report. Wherever possible, pressures have been converted to the SI system and expressed as kPa, followed by psi in parentheses. For example, water pressure at one atmosphere is equivalent to 101.325 kPa (14.73 psi).

Among fish with swim bladders, the response to rapid pressure changes encountered within a turbine is affected by whether the fish is *physostomous* or *physoclistous*. Physostomous fish have a duct, the pneumatic duct, which connects the swim bladder with the esophagus (Lagler et al. 1962). Gas can be quickly taken into or vented from the swim bladder in these species through the mouth and pneumatic duct, so that adjustment to changing water pressures can take place rapidly, often on the order of seconds. As a general rule, physostomes include the soft-rayed fishes like salmon, trout, catfish, minnows, and gar. On the other hand, physoclists lack a direct connection between the swim bladder and the esophagus. In these fish the contents and pressures within the swim bladder must be adjusted by diffusion into the blood, a process measured on the order of hours. Physoclistous fish include many of the spiny-rayed fishes such as perch, bass, and sunfish.

Once inside a turbine, physoclistous fish cannot adjust the volume of their swim bladder rapidly enough to compensate for changing water pressures; the swim bladder will be compressed and the fish will become more dense under increasing water pressures. Conversely, in a region of low pressure, downstream from the turbine blades, the swim bladder will expand, potentially to the point of bursting. Physostomes have more control over the volume of gas in the swim bladder than physoclists. If a deep-water-adapted physostome is drawn toward a surface intake, decreasing water pressure will cause the swim bladder to expand. Excess gas can be vented if the rate of ascent is sufficiently slow. However, even physostomous fish may not be capable of venting excess gas in response to the rapid pressure reductions (often less than 1 sec) that occur within the turbine and draft tube.

Harvey's (1963) work with sockeye salmon reinforced numerous other studies that found fish can tolerate very high hydrostatic pressures. He exposed sockeye fry and smolts to pressures in test chambers as high as 2064 kPa (300 psi) with no significant mortality. However, the rate of pressure increase in the test chambers during most tests was slow (1 psi/sec), so that maximum pressures were reached only after 5 minutes. This gradual increase in hydrostatic pressure does not duplicate the rapid pressure changes (measured in seconds) experienced by turbine-entrained fish upstream from the turbine blades. Rowley (1955) subjected rainbow trout to a similar pressure regime [gradual increase from atmospheric pressure to as high as 1376 kPa (200 psi) followed by instantaneous release to atmospheric pressure] and observed no detrimental effects.

Foye and Scott (1965) exposed six species of freshwater fishes (chain pickerel, yellow perch, fallfish, common shiners, lake trout, and Atlantic salmon) to instantaneous pressure increases to 2064 kPa (300 psi), followed by decompression back to atmospheric pressure over a 10-minute period. No mortality was observed among salmon, lake trout, or fallfish (a minnow species) over the subsequent 7-day holding period. Long-term mortalities among the other three species showed considerable variation, but inadequate controls precluded a quantification of mortality or, indeed, a determination that mortality among test fishes was caused by the pressure increases.

Of greater relevance to hydroelectric turbine passage, Harvey (1963) also measured the effects of decompression experienced at rapid rates. In these decompression studies he lowered hydrostatic pressures in as little as 0.1 seconds (decompression rates as high as 7,500 psi/sec) to pressure values as low as 1.6 kPa (0.23 psi). An initial series of tests indicated that briefly increasing the pressure above atmospheric (to 50 or 300 psi) before decompression did not affect mortality rates. Rather, increasing vacuums led to increasing mortalities. At pressures less than 84.6 kPa (12.3 psi) mortalities of test fish exceeded controls; sockeye mortalities averaged about 2 percent following brief exposures to 17.2 kPa (2.5 psi). The rate of decompression was important. Smolts gradually exposed to a reduction in pressure below atmospheric showed no apparent ill effects, even at pressures as low as 16.5 kPa (2.4 psi).

In another series of tests, smolts acclimated to surface waters, brief high pressures, and then rapid decompression to subatmospheric pressures experienced little mortality (Harvey 1963). However, outmigrating smolts that had been acclimated to deep water of a lake before being exposed to a sudden reduction in pressures suffered mortalities as high as 35 percent. Death was due to minute gas emboli, most commonly lodged in the heart or ventral aorta. Sockeye smolts held at a lake depth of 35 feet for 7 days (i.e., acclimated to a pressure of about 30 psi) suffered 21 percent mortality following decompression tests.

Harvey (1963) concluded that sockeye juveniles exhibited a tolerance to pressure increases, but could succumb to rapid decompression under conditions permitting swim bladder gas to appear as emboli in the blood stream. Compared to other species of fish, particularly physoclistous fish, sockeye are less susceptible to adverse pressure changes because the volume of the swim bladder is a small percentage of the fish's volume, the bladder is very extensible, and gas can be readily released through the pneumatic duct under slowly decreasing hydrostatic pressures. However, very rapid decompression, such as that experienced in fractions of a second downstream from turbine blades, may not permit the escape of swim bladder gas even in physostomous fish like salmon, and swim bladder damage and mortality among depth-acclimated fish would occur. Histological examination showed that the pneumatic duct is poorly adapted to rapid release of gas [Harvey and Hoar, unpublished manuscript, cited in Lucas (1962)], so that swim bladder rupture under severe vacuum conditions is possible even for physostomous fish like sockeye salmon. Harvey's (1963) belief that swim bladder damage was an important cause of pressure-related death was supported by his limited series of pressure tests on sculpins, which do not have swim bladders. The sculpins evidenced little discomfort and no mortality upon sudden exposure to vacuum conditions.

Similar conclusions were drawn by Tsvetkov et al. (1972), who examined pressure effects on a variety of freshwater salmonids, minnows, sturgeons, and perch. In their laboratory experiments fish were allowed to acclimate to excess pressures (up to 608 kPa; 88 psi) before being rapidly depressurized to atmospheric pressure (depressurization rates as high as 608 kPa/s). This technique was used to mimic the experience of depth-acclimated fish exposed to rapid depressurization downstream of hydroelectric turbines. Physostomous fish survived far better than physoclistous fish, but even physostomous fish were killed at decompression rates greater than 91 kPa/s. Because larvae and fingerlings of physostomous fish released swim bladder gases with greater difficulty than older fish, they were killed by relatively lower absolute pressure decreases and lower rates of decompression. The two species of sturgeon tested by Tsvetkov et al. (1972) were resistant to pressure effects. Despite many hours of exposure to pressures up to 608 kPa (88 psi), the investigators were unable to determine whether the sturgeon became acclimated to increased pressure. Subsequent rapid decompression was not lethal.

Feathers and Knable (1983) acclimated largemouth bass to elevated pressures (191, 280, and 369 kPa), then reduced the pressure to atmospheric (101 kPa; 14.7 psi) in less than one minute. Mortality was directly related to the magnitude of depressurization, ranging from an average of 25 percent at an acclimation pressure of 191 kPa (27.8 psi) to an average of 46 percent at an acclimation pressure of 369 kPa (53.8 psi). Depressurization mortality commonly occurred within 1 hour at the higher acclimation pressures, whereas mortality occurred over a 5-day period as a result of depressurization from the 191 kPa acclimation pressure. These tests indicate that relatively small but rapid pressure decreases can be harmful to physoclistous fish. Mortalities following depressurization from 191 kPa were largely attributed to respiratory failure and the stress of floating on the surface due to an expanded swim bladder. On the other hand, rapid depressurization to atmospheric pressure from 280 and 369 kPa caused severe hemorrhaging and large gas-bubble formation, especially in the areas of the heart and associated blood vessels, gills, and the brain.

Hogan (1941) exposed freshwater fishes to the types of vacuum conditions experienced within siphons used to transfer water over levees. The general procedure was to acclimate fish in an aquarium to atmospheric pressures, reduce the pressure to about 17 kPa (2.5 psi) in 15 seconds, hold the fish at the subatmospheric pressure for 10-30 seconds, and allow the pressure to return to atmospheric in 15 seconds. This was believed to simulate the time-pressure history experienced by fish entrained in the siphons. As a general rule, physostomous fish (golden shiners, carp, bullhead catfish, and long-nosed gar) survived the tests better than physoclistous fish (bluegill sunfish, crappie, and largemouth bass). Most physoclists were killed or showed obvious physical distress from these pressure changes; the longer the exposure to subatmospheric pressures, the greater the mortality. On the other hand, none of the physostomes died, although many temporarily lost equilibrium. Hogan (1941) observed minnows and gar discharging air through the pneumatic duct as vacuum was applied. He believed that this explained the relatively greater resistance of the physostomes to subatmospheric pressures.

Turnpenny et al. (1992) tested a variety of marine fishes under pressure regimes likely to be experienced during passage through a low-head tidal power turbine. All fish were acclimated to ambient pressure (ca. 101 kPa), then exposed to one of three pressure series. In the first series, the Protracted Low Pressure Series, test fish were raised to a pressure of 405 kPa (59 psi) in 10 seconds, then decompressed in 0.1 second to pressures ranging from 15 kPa to 101 kPa for 30 seconds. In a second series of tests, the Protracted High Pressure Series, pressures were raised from 101 kPa to as high as 405 kPa in 5 seconds, held at the increased pressure for 15 seconds, then returned to atmospheric pressure. Finally, tests under the Simulated Operating Regime Series were designed to mimic the pressure regime experienced by fish entrained in low-head turbines; surface- (101 kPa) or midwater- (202 kPa) acclimated fish were exposed to pressures as high as 345 kPa (50 psi), decompressed to subatmospheric pressures in a fraction of a second, then quickly returned to near atmospheric conditions; total exposure time to pressure fluxes in this third series was less than 5 seconds.

Atlantic salmon smolts, brown trout, and rainbow trout were generally tolerant to the pressure regimes tested by Turnpenny et al. (1992). No external damage (e.g., popped eyes, superficial hemorrhaging) was observed, and internal damage was restricted to swim bladder rupture among approximately 10 percent of the fish exposed to the most widest range of rapid decompression. Similarly, clupeids (herring and shad) and eels were tolerant of the pressure fluxes. The authors attributed this pressure tolerance to the ability of the physostomous salmonids and clupeids to rapidly vent excess gas from their swim bladders under decompression conditions (and the absence of an inflated swim bladder in the eels). On the other hand, physoclistous fish (e.g., seabass) that were unable to reduce the swim bladder volume quickly suffered high rates of swim bladder rupture and mortality. In separate tests, Turnpenny et al. (1992) estimated that for physoclistous species under sustained decompression, swim bladder rupture occurs at around a doubling of the swim bladder volume (or a halving of the acclimation pressure). Although physostomes were much more resistant of decompression, the most rapid and extreme pressure drops (surface-acclimated salmonids exposed to an equivalent of an eight-fold increase in swim bladder volume in 0.1 second) exceeded the response rate of the venting system and caused rupture of the swim bladder.

Traxler et al. (1993) subjected caged freshwater fishes (largemouth bass, bluegill sunfish, and channel catfish) to underwater explosions. Pressure fluxes resulting from the explosions were low, never exceeding 37 kPa (5.4 psi). No adverse effects on the fishes were observed.

The general conclusion that can be drawn from these studies is that pressure increases of the magnitude found in hydroelectric turbines are unlikely to injure or kill entrained fish. Rapid, brief pressure increases caused little or no direct mortality in a variety of studies using a variety of fish. However, high pressures may alter the behavior of fish such that they may have increased susceptibility to other, non-pressure-related sources of mortality. Some investigators have noted that fish exposed to high pressures were momentarily stunned. Although the test fish fully recovered in the laboratory holding tanks, temporarily stunned fish may be more susceptible to predators in the tailwaters of a hydroelectric dam. Further, in response to increasing pressures fish may actively swim within the turbine to areas that would not be predicted based on modeling of flow fields and neutrally buoyant objects. Harvey (1963) observed an increase in the rate of pectoral fin movements and angle of the body (head upwards) among sockeye salmon in response to pressure increases. Many investigators have observed a tendency for salmonids to swim downwards (sound) in response to increased pressure (Harvey 1963; Muir 1959). This sounding behavior would reinforce the natural tendency of the fish to sink under increased pressures (because the swim bladder becomes compressed). Consequently, actively swimming salmonids may not act like neutrally buoyant objects within the high-pressure region of turbines, but rather may move to regions of the turbine that pose relatively greater or lesser risk. The effects of the combination of increased density, sounding behavior, and other directed and random fish movements on turbine-passage mortality is unknown.

From a direct mortality standpoint, laboratory studies indicate that the brief exposure to subatmospheric pressures within the turbine are more likely to be damaging to fish with swim bladders. Table 1 and Figure 1 display mortalities that have been observed following exposure in the laboratory to rapid and brief pressure reductions. These data were selected using the following criteria: (1) fish had been held at a particular pressure (usually atmospheric pressure) long enough to become acclimated; (2) reduction from acclimation pressure (P_a) to exposure pressure (P_e) was rapid and brief; i.e., no more than a few seconds, in order to simulate the duration of low pressure exposure within a turbine. This second criterion was relaxed somewhat for studies which used physoclistous species. Physoclistous fish do not have a pneumatic duct, so they cannot rapidly vent gases from the expanding swim bladder. Consequently, exposure studies with more gradual pressure reductions (on the order of 10 to 15 seconds) were plotted in Figure 1 for physoclistous species. For example, Hogan (1941) exposed both physoclistous fish (largemouth bass, bluegill sunfish, and crappie) and physostomous fish (minnows, catfish, and gar) to pressure reductions that took 15 seconds to achieve. For physoclistous species that take many minutes to adjust to changing water pressures, this gradual pressure reduction adequately mimics the virtually instantaneous pressure reduction in a turbine. On the other hand, under the assumption that the rate of pressure reduction was sufficiently slow to allow the physostomes to vent expanding gases from the swim bladder, Hogan's tests would not reproduce relevant turbine conditions for physostomes, and these data are not plotted.

Figure 1 plots the percent mortality among test fishes versus the ratio of exposure pressure to acclimation pressure, P_e/P_a (see

Table 1. Mortality of fish exposed to rapid and brief pressure reductions in laboratory test chambers.

Test	Species	Acclimation pressure, P_a (kPa)	Exposure pressure, P_e (kPa)	P_e/P_a	Mortality (%)	Source
1	sockeye salmon	101	67	0.66	0	Harvey (1963)
2	sockeye salmon	343	101	0.29	0.5	Harvey (1963)
3	sockeye salmon	101	67	0.66	2	Harvey (1963)
4	sockeye salmon	205	67	0.33	21	Harvey (1963)
5	perch	303	101	0.33	70	Tsvetkov et al. (1972)
6	largemouth bass	101	101	1.00	0	Feathers and Knable (1983)
7	largemouth bass	191	101	0.53	25	Feathers and Knable (1983)
8	largemouth bass	280	101	0.36	41.7	Feathers and Knable (1983)
9	largemouth bass	369	101	0.27	45.8	Feathers and Knable (1983)
10	bluegill sunfish	101	17	0.17	33	Hogan (1941)
11	bluegill sunfish	101	17	0.17	50	Hogan (1941)
12	crappie	101	41	0.40	100	Hogan (1941)
13	crappie	101	17	0.17	50	Hogan (1941)
14	largemouth bass	101	17	0.17	80	Hogan (1941)
15	largemouth bass	101	17	0.17	100	Hogan (1941)
16	largemouth bass	101	17	0.17	50	Hogan (1941)
17	Atlantic salmon, brown trout, rainbow trout	101	15	0.15	0	Turnpenny et al. (1992)
18	brown trout	343	30	0.09	10	Turnpenny et al. (1992)
19	rainbow trout	343	30	0.09	0	Turnpenny et al. (1992)
20	herring	343	30	0.09	4	Turnpenny et al. (1992)
21	coho salmon	101	7	0.07	0	Muir (1959)
22	coho salmon	101	7	0.07	10	Muir (1959)

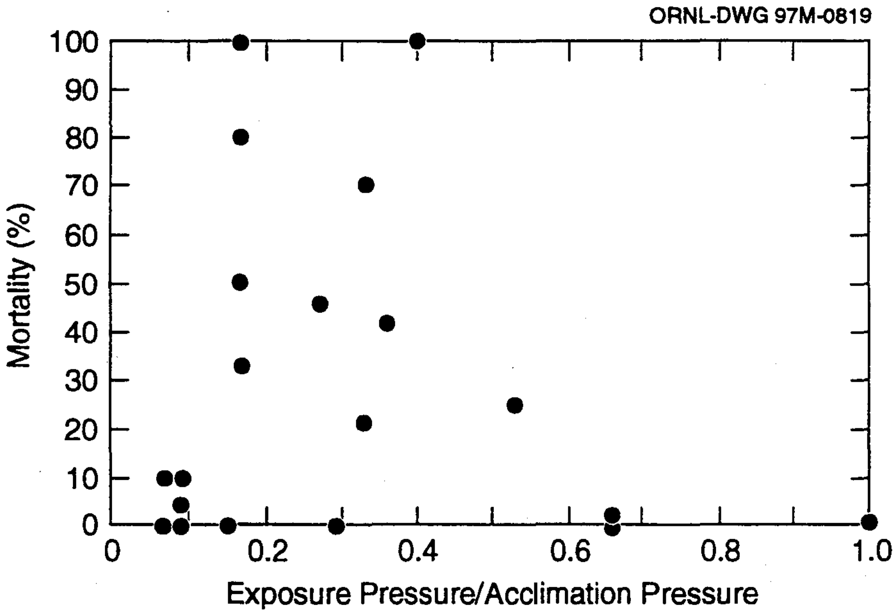

Figure 1. Fish mortalities following exposure in the laboratory to brief and rapid pressure reductions. See text and Table 1 for description of studies.

Table 1 for data). P_e/P_a is an indication of the severity of pressure reduction - the lower the value of the ratio, the greater the reduction in water pressure from that to which the test fish were acclimated. Many of these studies are old, poorly documented, have inadequate or no controls, and used only small numbers of fish. Not surprisingly, Figure 1 indicates that there is considerable variation in the response of fish to pressure reductions. However, the highest mortalities occurred when the pressure reduction was greatest, i.e., when the exposure pressure was a relatively small proportion of the acclimation pressure. There are few data above a P_e/P_a ratio of 0.40, but the three tests in which exposure pressure was greater than 60 percent of the acclimation pressure (P_e/P_a ratio > 0.60) resulted in little or no mortality. Below a P_e/P_a of 0.40 the highest mortalities were recorded among physoclistous fish (bass, bluegill, crappie, perch); this is consistent with the observations of Jones (1951) that a 60 percent reduction in pressure ($P_e/P_a = 0.40$) burst the swim bladders of perch. The higher survival of physostomes may indicate that these fish have greater resistance to swim bladder expansion and/or some ability to vent swim bladder gases even under conditions of very rapid pressure reductions. These sparse data indicate that pressures within the turbine should fall to no less than 60 percent of the value to which entrained fish are acclimated. For surface-oriented fish, a pressure of 60 kPa (8.8 psi) or greater at all points within the turbine and draft tube would be expected to protect most fish from direct mortality of low pressures.

Based on a consideration of salmonid data in USACE (1991), ARL (1996) suggested that minimum pressures within the turbine be no less than 30 percent of the fish's initial acclimation pressure. For fish distributed within the top 34 feet of water, this would dictate a minimum pressure of about 10 psi (69 kPa). This suggested minimum pressure criterion (30 percent of acclimation) is less restrictive than the "60 percent of acclimation" criterion suggested above. Whereas it may protect deep-adapted salmonids (and other physostomes) that are able to vent some of the expanding gases in the swimbladder as they are drawn upwards toward the intake, the 30 percent criterion may not be sufficient to protect other species of physoclistous fish.

2.2 Cavitation Effects

Cavitation is the process of formation of gas bubbles in a liquid caused by a localized reduction in pressure to a point at or below the vapor pressure (Turnpenny et al. 1992). In a turbine, cavitation can occur in areas of low pressure (e.g., downstream of the turbine blades), increasing local velocities, abrupt changes in the direction of flow, roughness or surface irregularities, and under certain conditions of water temperature and air content (USACE 1995). Once formed, cavitation bubbles stream from the area of formation (e.g., the blade surface) and travel with the flow to regions of higher pressure, where they collapse. The violent collapse of cavitation bubbles creates shock waves, the intensity of which depends on many factors, including bubble size, water pressure in the collapse region, dissolved gas content, and the presence of air (not water vapor) bubbles. Forces generated by cavitation bubble collapse may reach tens of thousands of kilopascals at the instant and point of collapse (Hamilton 1983a; Rodrigue 1986). These pressure waves decrease rapidly from the center of collapse, but nearby fish could be injured.

Muir (1959) simulated cavitation effects in a laboratory device. Brief exposure of 1.5- to 4-inch-long coho fingerlings to hydrostatic pressures equal to the vapor pressure of water caused no mortality. However, in other tests fish were rapidly decompressed to vapor pressure for 0.4 seconds, then returned instantaneously to atmospheric pressure. The vapor pocket that had formed in the test chamber collapsed, resulting in the death of 12 of the 20 test salmon (60 percent mortality). Microscopic examination of the fish revealed hemorrhaging of the eyes and gill plates. Muir (1959) concluded that it was the rapid, high-pressure shock waves associated with collapse of the cavitation bubble that caused the observed mortality. Hubbs and Rechnitzer (1952) also reported on the lethality of instantaneous shock waves (in this case caused by underwater explosions) to caged marine fishes. Less abrupt pressure waves of equal or greater magnitude caused no mortality.

The nature of cavitation bubble collapse and its likely effects on turbine-passed fish was discussed by Turnpenny et al. (1992). They pointed out that a bubble collapsing in midwater, away from any surface, will have the viscous forces resisting collapse distributed symmetrically around the bubble and therefore will tend to collapse symmetrically; the resultant shock wave will emanate more or less spherically from the point of collapse. On the other hand, a bubble collapsing near a surface (e.g., turbine blade, wall, fish's body) will not have the viscous forces distributed symmetrically. Collapse near a rigid surface will pull in water preferentially from the side away from the surface (distal side), causing the bubble to flatten out and collapse toward the surface (proximal direction), sometimes accompanied by the formation of a high-velocity microjet. Conversely, cavitation bubble collapse near an elastomeric (flexible) surface or a free-surface (e.g., air-water interface) will tend to be in the distal direction, moving away from the surface.

Turnpenny et al. (1992) recognized the importance of determining whether a fish acts as a rigid surface or an elastomeric surface in assessing the risk of damage from cavitation. They developed a laboratory apparatus that enabled them to create a cavitation bubble, photograph the bubble's collapse, and observe the effects of bubble implosion on fish tissue. A series of tests with brass plates as controls supported the idea that cavitation bubbles generated near a rigid surface would collapse asymmetrically, with the implosion being directed towards the metal plate. Subsequent tests with recently killed fish led to similar results - bubble collapse was asymmetrical and directed toward the surface of the fish in 33 of 35 (94 percent) of the exposures. However, no evidence of tissue damage was found on any fish as a result of the bubble implosions. This limited set of tests did not examine mortality and did not quantify the forces associated with cavitation bubble collapse. Turnpenny et al. (1992) pointed out that although their bubble collapse experiments did not cause any apparent tissue damage, fish are not safe from cavitation damage during turbine passage because the energy levels in a turbine may be vastly higher. They assumed that cavitation that can damage turbo-machinery can also damage fish, and that the closer fish passes to a vapor cavity the greater the probability of injury.

As noted in the section on pressure effects, decompression can be harmful to turbine-passed fish even if water pressures do not drop below vapor pressure. If turbines are designed and operated so that water pressures do not drop below 60 percent of ambient pressure anywhere in the turbine, cavitation will not occur and there will be no injury to fish (or damage to turbomachinery) from the collapse of cavitation bubbles. If cavitation cannot be eliminated entirely, another mitigation alternative is to introduce air into the turbine to reduce the effects of cavitation on noise, vibration, and damage to fish and machinery (Daily 1986; Hamilton 1983b, 1984). Entrained air can ameliorate the shock waves created by cavitation because (1) any air present in the vapor cavities will cushion the cavity collapse and reduce the resulting water hammer pressure, and (2) the presence of air bubbles will reduce the speed of the shock wave, and hence the magnitude of the shock waves on a surface (Chanson 1989). Turbine designs that introduce air or oxygen bubbles into the flow for tailwater aeration could have the additional benefit of mitigating some of the fish mortality resulting from cavitation.

2.3 Shear Stress Effects

Shear stress, like pressure, is force per unit area. The difference between pressure and shear stress is the direction in which the force is applied. In pressure the force acts *perpendicular* to the surface, whereas a shear force acts *parallel* to it (Gordon et al. 1992). Shear stress has the same units as pressure, N/m^2. In this report, studies of shear stress have been expressed wherever possible as N/m^2 and kPa, where one kPa equals 1,000 N/m^2.

Groves (1972) exposed juvenile salmon (total lengths ranging from 3.5 to 13.5 cm) to a water jet submerged in a tank of static water. In his experimental protocol the jet was brought to full speed [mean calculated velocities ranged from 9 to 37 m/s (30 to 120 ft/s)] and the fish were immediately introduced to the tank near the nozzle. Each test lasted only for the time needed to introduce the fish, usually less than a second. Thus, exposure to shear in this experiment was a brief, one-time exposure to high velocity water at the edge of the jet. The actual velocities and shear stresses experienced by fish were not measured. Some of the tests included high speed photography to track the fishes' movements, and all tests examined the resultant types of injuries and mortality.

Juvenile salmon were unaffected by exposure to the lowest velocity jet tested, 9 m/s (30 ft/s). As jet velocities increased the rates of disorientation, visible injury, and mortality also increased (Groves 1972). Fish disabled (disoriented) but without visible injury usually regained normal capacities in 5 to 30 minutes. Visible injuries were mostly in the head region and included bulged or missing eyes, broken and ripped gill covers, and torn gills. Whereas visible injuries and mortalities were zero at 9 m/s, velocities of 15 m/s (50 ft/s) caused injuries in 2 to 59 percent of the fish in the test batches (Tables 2 to 4). At any given jet velocity, injury rates were inversely related to the size of the fish, i.e., 3-cm salmon were more often injured than 13-cm-long salmon.

Injury from the water jet was related to the part of the fish contacted and to the position of the fish relative to the jet flow direction at the time of contact (Groves 1972). Greatest injuries occurred when the jet contacted the head region and was moving from the rear towards the head of the fish. Larger fish were less affected if the jet initially contacted some other portion of the body than the head, or if the fish was facing into the jet stream. On the other hand, smaller fish were damaged irrespective of their orientation. Groves attributed this size-related difference in injury rates to the proportion of the fish's surface area struck by the jet. The jet struck a relatively larger portion of a small salmon's body, and at the higher velocities some were literally torn apart. Larger fish had a proportionately small portion of their bodies contacted by the margin of the jet, so injuries tended to be more frequent when initial contact was with more protruding or less rigidly attached parts of their head region, such as the gill structures and eyes.

Morgan et al. (1976) used rotating concentric cylinders to create shear zones in 30.5-cm-diameter chambers. Striped bass and white perch eggs and larvae were introduced into the layer of water between the cylinders, and consequently exposed to calculated shear forces ranging from 76 to 404 dynes/cm^2 (7.6 to 40.4 N/m^2; 0.0076 to 0.040 kPa) for periods of 1 to 20 minutes. Both eggs and larvae were sensitive to these low levels of shear. For example, shear forces of 350 dynes/cm^2 (35 N/m^2 ; 0.035 kPa) killed an average of 38 percent of the white perch larvae in 1 minute, 52 percent in 2 minutes, and 75 percent in 4 minutes. The authors developed a set of regression equations which related the amount of shear to expected mortality among these fish early life stages.

McEwen and Scobie (1992) estimated that shear forces within a reference turbine could average over 500 N/m^2 (0.5 kPa); maximum values were estimated to be 3,740 and 5,421

Table 2. Effects of exposure of juvenile coho salmon to the margins of water jets moving at various calculated velocities. Fish ranged from 8.5 to 11 cm in size (mean = 10 cm). Test series 1 from Groves (1972).

Jet velocity, fps	Number of fish	Percent disoriented, injured, and/or killed	Percent visibly injured	Percent dead after 48 hours
30	50	0	0	0
50	50	18	8	2
70	50	42	28	8
90	50	56	24	16
100	50	62	20	22
120	50	74	14	32

Table 3. Influence of juvenile salmon size on the effects of water jets moving at various calculated velocities. Test series 2 from Groves (1972).

Jet velocity, fps	3 to 6 cm long			9 to 13 cm long		
	Number of tests	Number of fish	Percent injured	Number of tests	Number of fish	Percent injured
30	1	10	0	6	27	0
50	4	32	59	7	31	16
70	1	5	100	7	34	38

Table 4. Influence of juvenile salmon size on the effects of water jets moving at various calculated velocities. Test series 3 from Groves (1972).

Jet velocity, fps	3.5 to 5 cm long			6 to 8 cm long			9.5 to 13.5 cm long		
	No. of tests	No. of fish	Percent injured	No. of tests	No. of fish	Percent injured	No. of tests	No. of fish	Percent injured
30	3	75	0	6	50	0	10	50	0
50	3	75	37	13	174	26	15	75	9
70	7	164	52	31	201	35	14	100	29

N/m^2 (3.7 and 5.4 kPa) for "on-design" and "off-design" conditions, respectively. On the basis of these calculations, Turnpenny et al. (1992) designed a laboratory apparatus that could expose fish to localized shear forces of this magnitude. They introduced fish into a high-velocity water jet submerged in a tank of static water, then examined the fish for injuries and long-term mortality. Jet velocities tested ranged from 5 to over 21 m/s (16 to 69 ft/s), resulting in maximum shear stresses ranging from 206 to 3410 N/m^2 (0.2 to 3.4 kPa).

Salmonids (Atlantic salmon, rainbow trout, and brown trout) tested at the lowest shear stresses (maximum values of 206 and 774 N/m^2) experienced little scale loss, no loss of mucous coating, no other apparent injuries, and no mortality up to 7 days after the single exposure (Table 5). Greater jet velocities and shear stresses resulted in more injuries and lower long-term survival (Turnpenny et al. 1992). For example, at the highest shear stresses tested (maximum value near the jet of 3410 N/m^2), localized loss of mucous cover and some eye damage (corneal rupture; pop-eye; hemorrhaging in the eye) was noted; survival was around 90 percent 7 days after the test. Fish that died after exposure to the higher shear stress levels were heavily coated with fungus, probably because the loss of mucous increased their susceptibility to fungal infections.

Clupeids (shad, herring) were much more susceptible to shear stresses in the experiments of Turnpenny et al. (1992). All fish tested in the apparatus, even at the lowest maximum shear stress of 206 N/m^2 (0.2 kPa), died within 1 hour (Table 5). Many clupeids suffered eye damage, eye loss, torn and bleeding gills, and substantial loss of scales and mucous layer. At the other end of the scale, eels suffered no evident damage, other than some loss of mucous coating, and no 7-day mortality even at the highest shear stress levels tested.

Turnpenny et al. (1992) observed visible creases on the body surfaces of some fish entrained in the turbulent jet, which led to crushing of internal organs and internal hemorrhaging. Eye damage (corneal rupture, pop-eye, or red-eye) or eye removal were also common injuries among the fish exposed to these localized shear forces. Finally, osmotic imbalance caused by loss of much of the mucous layer and underlying scales is believed to be the reason for the sensitivity of clupeids to even low levels of shear. Eels, which have substantial mucous layers, were not injured by high shear forces.

Turnpenny et al. (1992) noted that their experimental apparatus demonstrated the effects of contact of part of the fish's body with a small zone of high shear stress, i.e., small-scale effects. Groves' (1972) experiments were also similarly limited. Larger-scale effects of shear and turbulence, in which the entire fish is additionally subjected to forces of elongation, compression, and torsion, were not adequately modelled in their studies. Although Morgan et al. (1976) only examined sensitive fish eggs and larvae, the experimental protocol enabled them to take into account the mortality caused by these other, larger-scale effects, i.e., the rotational and deformational components of shear that impact the entire animal. At some level these additional stresses might also cause physical damage to fish, while lower, non-injurious levels of rotation and deformation would be expected to disorient the fish, such that it would be hindered in its ability to escape predators in the tailrace.

Table 5. Effects of exposure of various fish to the margins of water jets moving at different velocities. Modified from Turnpenny et al. 1992.

Species	Jet velocity (m/s)	Maximum shear stress (N/m²)	Age Group	Survival at 7 days (%)	Mean scale loss (% per fish)	Eye damage (% of fish)	Gill damage (% of fish)
Atlantic salmon (*Salmo salar*)	0	0	2	96	5.8	0	0
	5.4	206	2	100	5.7	0	0
	10.4	774	2	100	4.4	0	0
	16.4	1920	2	92	8.0	28	0
	>20.9	3410	2	88	4.6	32	0
Rainbow trout (*Onchorhynchus mykiss*)	0	0	1	-	3.3	0	0
	16.4	1920	1	-	3.8	0	-
	>20.9	3410	1	-	5.0	0.3	2.0
Brown trout (*Salmo trutta*)	0	0	1/2	100	0	0	0
	10.4	774	1/2	100	0	0	0
	16.4	1920	1/2	80	5	10	0
	>20.9	3410	1/2	90	5	10	10
Atlantic herring (*Clupea harengus)*	0	0	0	100	5.0	18	0
	5.4	206	0	0	8.2	30	0
	10.4	774	0	0	24	60	0
	16.4	1920	0	0	58	60	40
	>20.9	3410	0	0	90	60	20
Twaite shad (*Alosa fallax*)	0	0	0	100	5.0	0	0
	>20.9	3410	0	0	90	40	20
Eel (*Anguilla anguilla*)	0	0	-	100	-	0	0
	5.4	206	-	100	-	0	0
	10.4	774	-	100	-	0	0
	16.4	1920	-	100	-	0	0
	>20.9	3410	-	100	-	0	0

The Groves (1972) and Turnpenny et al. (1992) high-velocity water jet studies noted size- and species-specific differences in sensitivity to brief exposure to shear stresses. Groves pointed out that smaller salmon (ca 3 cm long) suffered greater injury and mortality rates than larger salmon (up to 13.5 cm long), probably because of lesser tissue strength and exposure of a greater proportion of the body to initial contact with the jet. Turnpenny et al. observed little effect among eels (which may be resistant to shear because of their substantial mucous coating) and high sensitivity among clupeids (whose mucous coating and scales were readily lost). Salmon and trout appeared to be intermediate in their sensitivity to the shear created by the high-velocity jet.

2.4 Turbulence Effects

Turbulent flow occurs when fluid particles move in a highly irregular manner, even if the fluid as a whole is traveling in a single direction. That is, there are intense, small-scale motions present in directions other than that of the main, large-scale flow (Vogel 1981). Unlike laminar flow, which can be described by a linear equation, turbulent flow can only be defined statistically (Gordon et al. 1992); descriptions of the overall motion within turbulent flows cannot be taken as describing the paths of individual particles. Turbulence exists at all scales in nature, from the swirling motion created when a salmon scoops out a redd (scales smaller than the size of the fish) to large whirlpools in a river (scales much larger than a fish). Similarly, within a hydropower turbine turbulence occurs at different scales. Smaller-scale turbulence, which occurs throughout turbine passage, can distort and compress portions of the fish's body. Large-scale turbulence, which may be most pronounced in the draft tube, creates vortices (swirl) which spin the fish and may cause disorientation. It is believed that this turbulence-caused disorientation, while perhaps not injuring the fish directly, may leave turbine-passed fish more susceptible to predators in the tailrace.

The effects of turbulence on survival of paddlefish yolk-sac larvae was examined in the laboratory by Killgore et al. (1987). Paddlefish larvae were placed in circular containers and exposed to differing frequencies and intensities of turbulence created by water jets. Turbulence in the laboratory chambers was expressed in terms of both water velocities (cm/s) and pressures (dynes/cm^2). The investigators found that turbulence intensity was more lethal than frequency of disturbance. Low turbulence (1,774-1,902 dynes/cm^2; 21.5-22.8 cm/s) caused 3 and 13 percent short-term mortality, whereas high turbulence (6,219-6,421 dynes/cm^2; 56.5-59.3 cm/s) resulted in 87 and 80 percent short-term mortality. Longer-term direct mortality, indirect mortality, and physiological stress were not examined. Based on these laboratory studies and field measurements of pressures near commercial barges (which sometimes exceeded 50,000 dynes/cm^2 near the propellers), Killgore et al. (1987) suggested that turbulence generated in the immediate vicinity of commercial vessels could cause mortality among paddlefish larvae.

2.5 Mechanical Effects (Strike and Grinding)

Damage to turbine-passed fish can occur if they collide with structures within the turbine systems, including fixed guide and stay vanes, moving runner blades, and flow-straightening walls in the draft tube. This mechanism is called *strike*. The probability of a fish being injured or killed by mechanical strike is a complicated function of characteristics of the fish (species, age, length, mass, condition), the turbine (number of runner blades, size of the openings between vanes and blades, sharpness of the blade edges, revolution rate, blade velocity), and the relationship between the fish and the turbine (e.g., the region of fish passage relative to the runner hub, orientation of the fish's longitudinal axis relative to the blade edge, and the fish's velocity relative to the blade velocity)(USACE 1995).

Mechanical injury can also be caused by *grinding*, in which the fish is drawn through narrow openings or clearances (gaps) between structures in the turbine passageway (USACE 1995). Within Kaplan turbines, the smallest clearances are gaps between adjustable turbine blades and the hub, between blade tips and the discharge ring, and between the top and bottom of the wicket gate seal plates when gates are set at higher openings. Grinding injury is most often evidenced as localized bruises that result from the fish being squeezed through the narrow gaps. However, grinding may also cause deep cuts and decapitation.

Turnpenny et al. (1992) noted that theoretically the probability of strike can be estimated from information on water velocity through the turbine, blade and guide vane angle, blade rotational speed, and fish lengths; these ideas have been explored by von Raben (1957), Montén (1985) and Solomon (1988). However, Solomon (1988, as cited in Turnpenny et al. 1992) pointed out that this probabilistic approach to estimating strike relies on several assumptions, including:

(a) the distribution of fish is either random or can be specified (this assumption is important because the probability of injury is higher towards the runner tip due to higher collision velocity);

(b) the fish enter the turbines randomly with respect to time, or else according to a specifiable temporal pattern;

(c) the fish either move passively through the turbine or attempt to resist entry by swimming at a known rate (active swimming against the flow of water reduces the rate of passage and thereby increases the risk of the fish being caught by the blade sweep; alternatively actively burst swimming at an angle to the flow could carry the fish into or out of regions of high strike probability);

(d) the fish are aligned randomly or else are aligned along the streamlines (this affects their effective length relative to the probability of striking a moving blade); and

(e) the consequences of strike are the same, irrespective of where or with what force the fish is struck.

Most of these simplifying assumptions are difficult to prove (or specify reliably) in a general sense because they may vary based on site- and species-specific conditions. In addition, some of these factors are greatly affected by the behavior of individual fish; one fish may pass through the turbine like a rigid, immobile, neutrally buoyant object aligned with the stream flow, whereas the next fish (of the same species and size, and entering the intake at the same location) may elect to change positions near the runner blade by active swimming movements. Consequently, estimates of the probabilities of strike and strike-related injury/mortality may have wide confidence boundaries because of the often unpredictable behavior of individual fish.

Recognizing that most of the assumptions listed above are site- and species-specific, Turnpenny et al. (1992) concentrated on investigating in the laboratory the process of approach and collision between fish and various blade profiles. In addition, they attempted to establish how fish size, orientation, and position relative to the blade influence injury and mortality. Their laboratory apparatus consisted of a portion of a turbine blade attached to a set of rails within a glass viewing tank. By means of springs, the blade section could be moved rapidly along the rails in order to strike a test fish positioned in the tank. Four blade tip profiles were used (Figure 2), ranging from blunt (near the hub) to narrow (near the blade tip). The blade section struck the test fish at a velocity of around 5-7 m/s, which was comparable to the calculated collision velocity near the hub of a full-sized turbine. The experimental apparatus was unable to reproduce the estimated peripheral runner collision velocity of around 20 m/s. Strike experiments included estimation of survival, investigation of the effects of the "bow wave" from the blade pushing fish to one side or another, and the effects of fish length, mass, and orientation on strike probabilities.

At a collision velocity of 5.2 m/s and a wide (hub) blade profile, little damage and no mortality was observed among brown trout, sea bass, or eel (Turnpenny et al. 1992). This experiment reproduced the expected conditions associated with a fish striking a turbine blade near the hub. On the other hand, strikes from the three narrower blade profiles, even at relatively low velocities of 6.9-7.1 m/s, caused severe damage to test fish in most cases. Principal symptoms were scale and mucous loss, bruising, eye damage, and internal bleeding. Some fish had broken spinal columns or deep grooves left by the blade impact.

In tests with freshly killed fish, Turnpenny et al. (1992) noted that mass and center of gravity (orientation) relative to the blade had important influences on the probability of strike. In general terms, water approaching the turbine blade divides and moves laterally to pass around the blade. Small objects suspended in the water (e.g., small fish and plankton) are often swept around the blade with the water flow and do not collide with the leading edge. However, larger fish, because of their inertia, tend not to follow the streamlines along the blade but rather follow their original trajectory. Whether or not a large fish collides with the leading edge of the blade depends upon the balance between sideways drag of the water and the inertia of the fish. The investigators found that small fish (<20 g) were generally swept aside by the water moving around the blade unless their center of gravity fell within the direct path of the blade. Even then only a small percentage of small fish (13.7%) were struck. Heavier fish had a greater probability of collision owing to the inertial effect. Fish with a body mass of up to 200 g had a 75% change of being struck when the center of gravity fell within the path of the blade, and heavier fish had a 100% chance. As a large fish's center of gravity was increasingly offset from the blade centerline, flexibility and tendency to follow the streamlines reduced the chance of a strike. For example, if a large fish was offset from the blade centerline by 0.4 body length the probability of collisions dropped to near zero.

Turnpenny et al. (1992) used these experimental observations to develop equations that were used to calculate the

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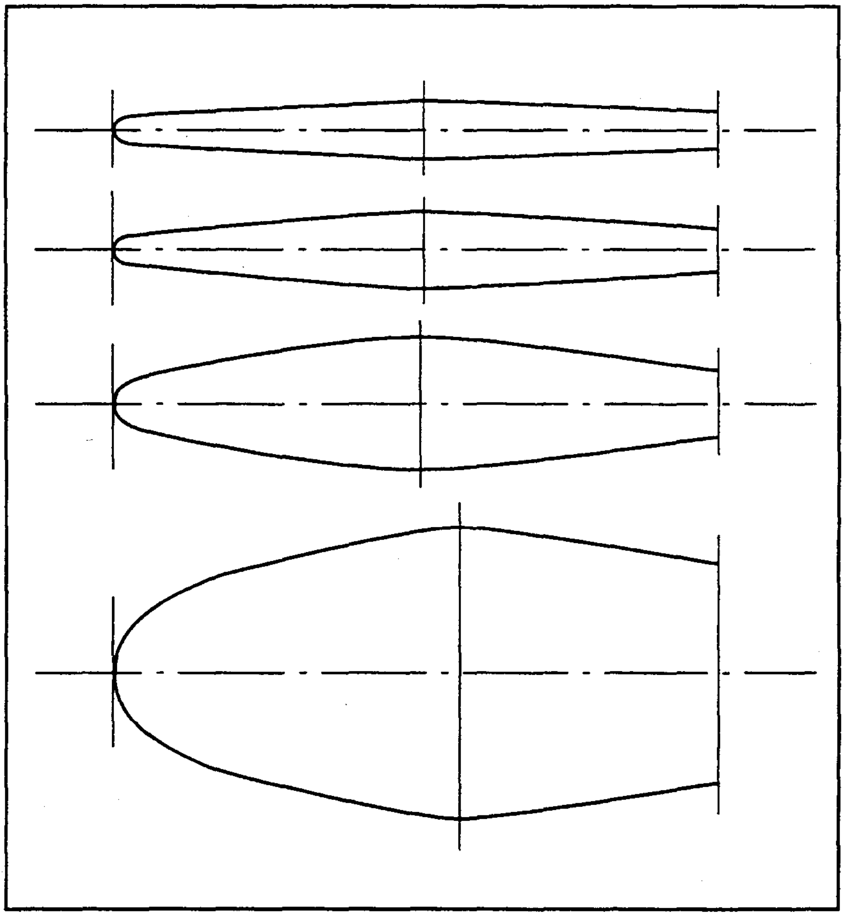

Figure 2. Four blade tip profiles used for strike experiments of Turnpenny et al. (1992). Top profile represents the blade leading edge near the tip; bottom profile represents the blade leading edge near the hub.

probability of blade strike for a low-head, axial flow tidal power turbine and a variety of fish weights and lengths. The equations take into account the effects of fish length, fish location, fish orientation, fish swimming speed, water velocity, open space between blades, blade thickness, and blade speed. Separate probabilities were calculated for fish oriented randomly with the flow, fish aligned with the flow but not swimming, and fish aligned with the flow and swimming against the flow at 6.5 body lengths/s (which increases the exposure time and thus the chance of blade strike). Estimated strike probabilities for this turbine ranged from as little as 0.32% for small fish to as much as 86% for large fish (Table 6).

A recent turbine passage survival workshop (USACE 1995) noted that turbine designers have a number of options that can affect the incidence of strike, including altering the number of blades, length of blades, area per blade channel, thickness and bluntness of blade entrance edges, and blade tilt. Optimizing these factors for fish passage survival may have power production consequences. Blade speed can also have an important influence of probability of strike, and is a factor in the strike probability equations developed by von Raben (1957), Montén (1985) and USACE (1991). Based on a plot of fish mortality vs. tip speed (peripheral runner velocity) in Francis turbines (EPRI 1987), ARL (1996) suggested that peripheral runner velocities of 40 ft/s or less would have a low potential for causing strike-related mortality.

There are no data to assess the relative importance of grinding as a contributor to mechanical injuries in hydropower turbines. Participants in the USACE (1995) workshop felt that grinding injuries could occur among fish entrained in water leaking through gaps between the turbine blade leading edge and the hub, the blade tips and the throat ring, the wicket gates and stay vanes, and the wicket gates and distributor ring. ARL (1996) suggested that grinding injuries could be prevented by limiting clearances between rotating and stationary turbine components to no greater than 2 mm. Limiting clearances to this small size would preclude all but the smallest fish from passing through gaps. The suppositions of the USACE (1995) workshop participants and ARL (1996) about the potential effects of gaps on turbine-passage injuries are reasonable. However, because this issue has not been adequately studied there is presently no basis in the literature to support the need for such narrow clearances or, indeed, whether reductions in gaps will significantly reduce turbine passage mortality.

2.6 Conclusions and Recommendations

2.6.1 Biological Criteria for New Turbine Designs

A review of the literature related to typical turbine-passage injury mechanisms suggests the following biological criteria should be considered in the design of new turbines:

Pressure

Pressure **increases** of the magnitude found in hydroelectric turbines do not appear to cause direct damage to entrained fish. Rapid pressure increases much higher than those found within a turbine did not result in mortality. One possible area of concern regarding pressure increases is the resultant increase in density of the fish. Rapid pressure increases will compress the swim bladder, making the fish more dense and causing it to sink. This would change the flow path of fish

Table 6. Calculated probabilities (expressed as percentages) that fish of various lengths and weights will be injured by blade strike in a low-head, axial flow tidal power turbine. Probabilities do not include collision with the widest blade profile near the hub. Modified from Turnpenny et al. (1992).

Fish weight (g)	Fish standard length (mm)	Random orientation of fish				Fish aligned with flow, not swimming				Fish aligned with flow, swimming against flow at 6.5 body lengths/s			
		3 m head, 382 m^3/s	5 m head, 507 m^3/s	6 m head, 739 m^3/s	8 m head, 554 m^3/s	3 m head, 382 m^3/s	5 m head, 507 m^3/s	6 m head, 739 m^3/s	8 m head, 554 m^3/s	3 m head, 382 m^3/s	5 m head, 507 m^3/s	6 m head, 739 m^3/s	8 m head, 554 m^3/s
<20	25	0.57	0.45	0.32	0.41	0.86	0.66	0.47	0.61	0.88	0.67	0.48	0.62
	50	1.1	0.83	0.59	0.76	1.7	1.3	0.89	1.2	1.7	1.3	0.91	1.2
	75	1.6	1.2	0.85	1.1	2.5	1.9	1.3	1.7	2.7	2.0	1.4	1.8
	100	2.1	1.5	1.0	1.4	3.3	2.5	1.7	2.3	3.6	2.7	1.8	2.4
20 to 200	100	2.1	1.5	1.0	1.4	3.4	2.6	1.8	2.4	3.7	2.8	1.9	2.6
	150	2.8	2.0	1.3	1.8	4.9	3.7	2.6	3.4	5.7	4.2	2.8	3.8
	200	3.6	2.6	1.6	2.3	6.4	4.8	3.3	4.4	8.0	5.7	3.7	5.1
	250	4.4	3.1	1.9	2.8	7.9	5.9	4.1	5.4	11	7.3	4.7	6.6
>200	250	4.8	3.6	2.4	2.9	8.0	6.2	4.4	5.7	11	7.5	5.0	6.7
	500	8.7	6.3	4.0	5.7	15	12	7.9	11	30	18	11	16
	750	13	9.0	5.6	10	22	17	11	15	86	38	19	32
	1000	17	12	7.2	13	30	22	15	20	1857?	86	31	64

within a turbine compared to a neutrally buoyant object. Within limits the fish can counteract this tendency to sink by active swimming, but it is not known whether a fish would do this within a turbine environment.

Pressure **decreases** within the turbine are a greater concern. The problem is not so much a matter of the lowest pressure experienced by fish in the turbine as it is the magnitude and rate of change from the fish's acclimation pressure. For example, a fish acclimated to surface water (101 kPa) may be unaffected by brief passage through a region of low pressure (say 60 kPa) in the turbine. On the other hand, a fish acclimated to deep water (300 kPa) will experience a relatively large pressure decrease passing through the same region of the turbine. Because the decrease is virtually instantaneous, all fish with swim bladders (even physostomous fish with pneumatic ducts) will be unable to vent gas from the rapidly expanding swim bladder. The swim bladder may distend or rupture, causing direct mortality or reduced ability to escape predators in the tailrace. Studies of swim bladder rupture and fish mortality following rapid decompression indicate that allowing minimum pressures within the turbine to fall to no less than 60 percent of the value to which fish are acclimated should protect most fish from direct effects of low pressures. As with compression, sublethal decompression may momentarily stun the entrained fish or otherwise alter its behavior so that its susceptibility to predation in the tailwaters could be changed.

Cavitation

Turbine designs that minimize pressure reductions to no greater than 60 percent of ambient (see above) will not cavitate, and cavitation-related injury to fish will not occur. If cavitation cannot be entirely prevented, introduction of air or oxygen bubbles may serve to mitigate adverse effects by cushioning the shock waves created by the collapsing water vapor bubble. This measure would have the additional advantage of aerating water that is discharged from the turbines.

If cavitation does occur, the consequences could be predicted in a similar way to those of mechanical strike. The probability of injury from cavitation could be calculated from information about the magnitude and areal extent of cavitation and the likelihood that fish will pass near enough to be affected by the pressure waves and/or high-velocity microjet. Presently, there is insufficient information in the literature to predict how close to areas of cavitation bubble collapse fish can pass without injury.

Shear

The effects of shear within the turbine and draft tube environment have not been adequately studied. The best available information comes from laboratory studies in which the fish is exposed to a high-velocity water jet in a static water tank. These tests examine the injury and mortality rates of fish in which high shear values are applied to only a portion of the fish. Shear effects are both species and life-stage specific:

- 3,410 N/m^2 (34,100 dynes/cm^2; 3.4 kPa) caused no apparent injury and no mortality among eels
- 1,920 N/m^2 (19,200 dynes/cm^2; 1.9 kPa) caused low levels (~ 10%) of injury and mortality to juvenile salmonids
- 206 N/m^2 (2,060 dynes/cm^2; 0.2 kPa) can cause complete mortality in clupeids,

apparently due to loss of scales, epithelium, and mucous layers.

- 35 N/m^2 (350 dynes/cm^2; 0.035 kPa) caused an average of 38 percent mortality among white perch larvae in 1 minute, 52 percent in 2 minutes, and 75 percent in 4 minutes. Striped bass larvae were nearly as sensitive.

Other, larger-scale effects of shear on entrained fish, including elongation, compression, torsion, rotation, and deformation have only been studied for fish eggs and larvae. At high levels, these forces could cause injury and mortality among larger fish. At lower, non-injurious levels, fish would be physiologically stressed and disoriented by shear and turbulence and may suffer greater indirect mortality (predation) below the turbine discharge.

Strike

Because of numerous variables related to the entrained fish (e.g., individual size, condition, and behavior) and the relationship of the fish to the runner and other turbine structures (e.g., region of passage, orientation, and relative velocity), the probability of injury from strike and grinding cannot be precisely estimated for any turbine. Some strictly biological factors, such as the species, length, and mass of entrained fish, influence the injury/mortality rate but cannot be altered by the turbine designer. Other biological factors may be influenced by turbine design (fish swimming behavior and orientation during turbine passage), but we do not know how design changes could be made to accommodate these factors. All else being equal, qualities of the turbine system that could be considered in order to minimize strike injury include:

- reducing the number of blades or amount of blade leading edge will reduce the probability of contact;

- maximizing the open space between blades and other structures will provide the largest routes of safe passage for entrained fish;

- blunt leading edges will cause less injury than sharp leading edges;

- lower runner speeds (blade rotational speeds) result in lower collision velocities and lower injury rates;

- fish struck by the blade near the hub will experience fewer injuries than fish struck near the blade tip because of reduced collision velocities. Consequently, turbine designs that direct entrained fish away from the runner periphery and towards the hub may cause lower injury rates. Note, however, that recent studies at Wanapum Dam suggest that greater turbulence and cavitation near the hub, as well as the possibility of grinding injuries in the blade-hub gaps, may lower survival of fish that pass through the turbine near the hub;

- Gaps between fixed and moving parts of the turbine should be minimized to reduce injury and mortality due to the mechanism of grinding.

2.6.2 Relative Importance of Turbine-Passage Injury Mechanisms

The relative importance of these mechanisms will depend on the species, size, and life stage of entrained organisms. For example, Dadswell and Rulifson (1994) published a hypothetical distribution of mortality mechanisms among marine animals passing through low-head hydropower turbines (Figure 3). In their conceptualization, mortality resulting from mechanical strike increased with increasing length of the entrained animal, being very low among 2-cm-long juveniles and approaching 100 percent in animals 2 m long or greater. Shear-related mortality is relatively low for all sizes of animals; it is highest among 20-cm-long juveniles and less damaging to smaller and larger fish. They hypothesized that mortalities from cavitation were constant over a wide size range, but that pressure effects were greatest among the smallest organisms and declined precipitously with size.

Many of the trends in Dadswell and Rulifson's (1994) hypothetical distribution are reasonable, based on the present review of literature. Certainly, the probabilities (and consequences) of mechanical strike will increase with increasing fish size. Also, cavitation can cause point-source injuries (from the microjet) or shock wave-caused mortality that would likely affect a wide size range of fish equally. For most turbines, cavitation occurs in a limited area, and therefore cavitation-caused mortality should also occur among a limited proportion of entrained fish.

There is less support from laboratory and other controlled studies for the shear and pressure trends shown in Figure 3. Shear has been shown to have a significant species-specific component unrelated to length; for example, eels with thick layers of mucous are much more resistant to shear forces than shad. Relatively low levels of shear and turbulence can be very damaging to fish eggs and larvae. Definitive studies of the effects of shear stresses and turbulence on fish are needed, but the few studies that have been conducted indicate that, for a particular species, mortality due to shear may be similar to the pressure line in Figure 3, i.e., high mortality among smaller, more fragile life stages and decreasing mortality with increasing size.

The present review of literature indicates that mortality resulting from the pressure-related component of turbine passage may be lowest among the smallest fish and increase to a relatively constant level in medium and large-sized fish. Fish of all sizes appear to be resistant to rapid and large pressure increases. Rapid pressure decreases, on the other hand, can be damaging, and the extent of the damage appears to be related to the tolerance of the fish to the rapid swim bladder inflation that occurs at lowered pressures. Fish larvae, early juveniles, and some species of adult fish (e.g., sculpins) do not have developed swim bladders, and these fish appear to have resistance to lowered pressures as well. Most fish have developed swim bladders at a length of a few centimeters; these fish could experience burst swimbladders in the areas of subatmospheric pressure downstream of the turbine blade. It is possible that physostomous fish (that can vent expanding gases from the swim bladder through the pneumatic duct) and physoclistous fish (that cannot vent gases) may have different sensitivities. However, the pressure drops occur so rapidly in a turbine that it is unlikely that physostomous fish can completely accommodate the changes.

Adverse water quality may also alter the effects of the physical injury mechanisms considered in this review. The mortality ultimately resulting from physical stresses such as pressure changes or strike may be increased by suboptimal water temperatures (either high or low), low dissolved oxygen concentrations, supersaturated nitrogen gas, and high levels of debris and other suspended materials. These

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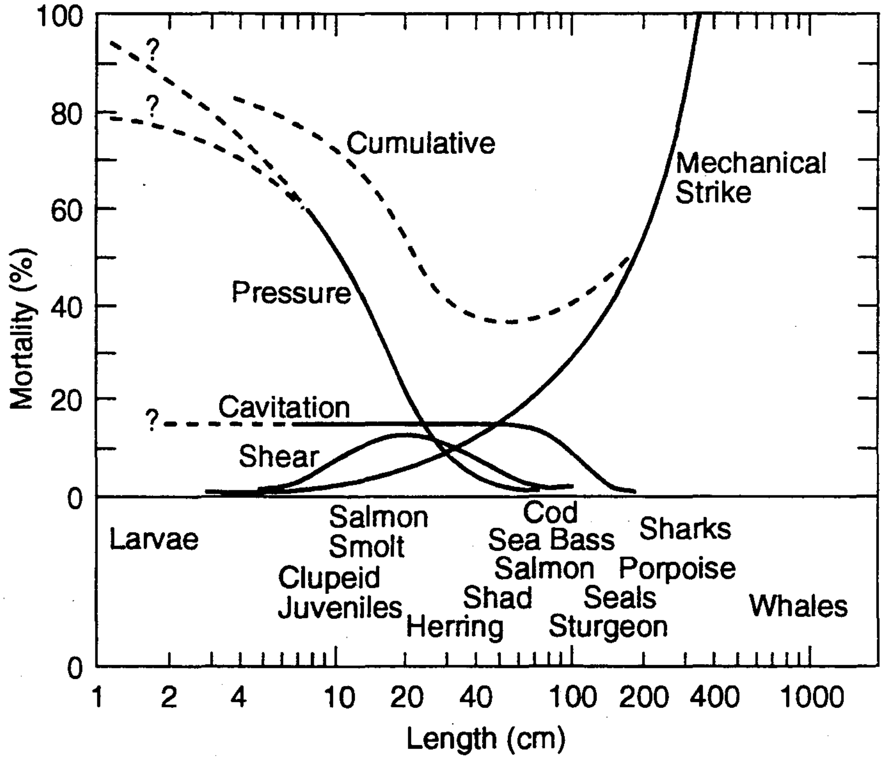

Figure 3. Hypothetical distribution of mortality and its causes from passage through hydraulic, low-head turbines in relation to body length of aquatic organisms. From Dadswell and Rulifson (1994).

water quality factors are usually optimized in laboratory studies. At actual operating turbines water quality problems may add to the overall level of stress and may contribute to greater-than-expected turbine passage mortality.

One of the drawbacks of examining individual injury mechanisms in the laboratory under controlled, optimal water quality conditions is that no information is developed about possible synergistic or antagonistic effects of multiple stresses. Synergistic effects occur when the mortality resulting from several stresses applied simultaneously is greater than would be expected from summing the expected mortalities from each of the separate stresses. Adverse synergistic effects might occur, for example, when a fish that is already stressed by high water temperatures dies after exposure to levels of shear that are considered to be sublethal from laboratory studies. Conversely, antagonistic effects occur when the combined effect of multiple stresses is lower than would be expected from summing the separate effects (you can't kill a fish twice, so a fish that is killed by blade strike will not be killed subsequently by lethal levels of cavitation). Examples of both synergistic and antagonistic effects of multiple contaminants are well known in the toxicology literature, but they have not been widely studied for the types of stresses considered in this report. Laboratory studies conducted by Čada et al. (1980) suggested that combined effects of thermal shock, shear, pressure changes, and pump passage had synergistic effects for some freshwater fish species. Multiple disturbances (handling stresses) have been shown to have a significant cumulative effect on physiological stress responses in juvenile chinook salmon (Barton et al. 1986), which in turn may result in increased losses to predation (Mesa 1994). Uncertainties about the possible cumulative effects of multiple stresses were discussed in USACE (1995).

Finally, most of the studies of turbine-related injury mechanisms have examined only direct mortality (USACE 1995). Much less is known about indirect mortality, i.e., the influence of sublethal turbine-passage stresses on later mortality due to predation or disease. Figure 3 could be revised to include additional, indirect mortality. However, the revised figure could conceivably look several ways: (1) identical to Figure 3 because indirect mortality is insignificant; (2) all lines depict 100 percent mortality at all fish lengths because the eventual mortality among turbine-passed fish from predation and disease is complete; or (3) some intermediate condition. Some attempts have been made to examine long-term mortality among turbine-passed fish. For example, Ferguson (1991) investigated long-term survival by comparing the numbers of turbine-passed juvenile salmon with subsequent adult returns. Further investigations of this type would be useful to ensure that reductions in direct mortality due to turbine design changes are not nullified by high levels of indirect mortality.

2.6.3 Need for Additional Studies

The disparities between the hypothetical mortality distributions of Dadswell and Rulifson (1994) and the distributions that could be drawn based on the studies reviewed in this report may be due in part to differences in turbine design. Different turbine designs will have different pressure regimes, shear regimes, and probabilities of strike. However, some of the disagreement about probable causes of mortality is due to the lack of reliable information about the importance of each of injury mechanisms associated with hydropower turbine passage. Most turbine-passage studies to date have been carried out at operating hydropower sites (see EPRI 1987). While these studies are necessary to estimate overall mortality associated with turbine passage for those particular sites and species, they are not very useful for determining the relative importance of the different injury mechanisms. Most field studies that have attempted to partition the observed injuries among the possible mechanisms have been frustrated by the fact that different stresses can cause the same injuries (USACE 1995; Voith Hydro 1996; ARL 1996). Turnpenny et al. (1992) summarized a series of single-mechanism laboratory studies (Table 7) and noted considerable overlap in injury symptoms. They found for example, that eye hemorrhaging can be caused by both pressure changes and shear forces, whereas scale and mucous loss can be caused by both shear and blade strike. Because of the overlap in injury symptoms, reliable biological criteria for the turbine designers will need to be based on controlled laboratory or field studies in which each injury mechanism is examined separately.

Table 7. Summary of primary injuries to fish observed in laboratory studies by Turnpenny et al. (1992) of pressure, shear, and blade strike.

Cause/symptom	Pressure	Shear	Blade strike
Ruptured swimbladder	yes	no	no
Eye hemorrhaging	yes	yes	no
Corneal rupture/eye loss	no	yes	yes
Scale loss	no	yes	yes
Mucous loss	no	yes	yes
Internal hemorrhaging	no	yes	yes
Egg loss	yes	no	no
Gill/operculum damage	no	yes	no

Among the injury mechanisms considered in this report, the effects of water pressure on fish seem to be the best understood. The influence of pressure increases and decreases have been studied for a variety of species, so that reasonable biological criteria that will protect turbine-passed fish can be determined. Strike and cavitation appear to be similar in that the effects are probabilistic; it is generally accepted that collision with the blade at sufficient velocity or proximity to a collapsing cavitation bubble will cause injury and death. Expanding this database with new information collected under controlled laboratory conditions would not be difficult. The greatest uncertainties associated with strike and cavitation deal with understanding how fish behavior can alter the risk of injury. We do not know whether behavioral responses to stimuli (changes in illumination, sounds, and flow fields) lead fish into areas within the turbine of lesser or greater risk, or whether the behavioral response is reliable enough to point toward turbine design changes.

Least understood are the effects of shear forces on fish. Several experiments have investigated the effects of localized shear by causing the fish to be struck on a portion of its body by a high-velocity water jet. These experimental conditions can be used to develop biological criteria. Of perhaps greater relevance to turbine passage, however, are the rotational and deformational forces experienced by the entire fish as is passes through highly turbulent areas of the turbine, draft tube, and tailrace. These effects have been shown to be damaging to fish eggs and larvae, but have not been adequately studied in larger fish. Even if these aspects of shear and turbulence cause little direct mortality, they are known to disorient the fish so that they may have increased susceptibility to predators.

3. Laboratory and Field Techniques for the Study of Injury Mechanisms Associated with Turbine Passage

Biological criteria can be developed through the use of both laboratory studies and field studies. The primary advantage of laboratory studies is that individual injury mechanisms can be isolated and examined under controlled conditions. For example, the effects of pressure changes on injury/mortality can be examined by itself, with all other stresses minimized. The biological response to a range of pressure changes can be quantified, and this response should apply to any turbine in any river system that exhibits these pressure changes. Also, the relative importance of the injury mechanisms can be determined if tests of each mechanism are conducted in similar ways and results are expressed in comparable terms. Turbine designers can focus on reducing the values of those individual injury mechanisms that have been shown to cause the greatest effect in controlled laboratory conditions. If tradeoffs are required (e.g., increasing the pressure changes in order to decrease shear stresses), laboratory studies of each mechanism are needed to predict the ultimate effect on fishes.

On the other hand, field studies have the advantage of replicating the actual entrainment experience. Turbine-passed fish are exposed not just to shear stresses or pressure changes, but rather to combinations of all injury mechanisms (pressure, shear, and mechanical injury) simultaneously. There is a potential for non-additive effects among these mechanisms, i.e., the combined mortality rate may be greater than (synergistic) or less than (antagonistic) the sum of the mortalities estimated from the individual mechanisms examined separately. Effects of combined stresses are extremely difficult to study in the laboratory. Field studies have the advantage of creating realistic combinations of stresses. The primary disadvantage of field studies is their site-specificity. One field site with 10 m of head may not be able to produce turbine passage conditions that are relevant to another field site with 30 m of head. It is not possible to test levels of each of the injury mechanisms beyond those provided by the particular turbine, and these levels are relatively uncontrolled. Fish passing through one region of the turbine are exposed to a different combination of pressure, shear, and mechanical stresses than fish passing through a different region of the same turbine. Consequently, it has been difficult to develop biological criteria from field studies of turbine-passed fish that can be reliably applied to the prediction of injury/mortality at other turbines. Field studies can provide very good information about entrainment mortality at that particular site, but relatively little information that is relevant to different hydropower sites or that could be used to make turbine design tradeoffs.

There is considerable value to conducting both laboratory and field studies for developing biological criteria in support of advanced turbine designs. Laboratory studies are needed to examine each of the injury mechanisms under controlled conditions. The biological criteria resulting from these studies are not site-specific, and thus provide basic information that can be applied to a wide variety of turbines. Field studies provide the evidence that biological responses observed in the laboratory are representative of real-world conditions, where such factors as temperature, turbidity, or dissolved gas concentrations may be sub-optimal. Further, field studies reflect the simultaneous exposure of fish to multiple stresses that, when compared to laboratory tests, allow the detection of unexpected non-additive cumulative effects.

The following sections review literature that describe laboratory and field techniques that could be brought to bear on the turbine-passage problem.

3.1 Laboratory Techniques

The purpose of this section is to describe techniques and experimental apparatuses that have been used to examine injury mechanisms associated with turbine passage (strike and grinding, pressure, cavitation, shear, and turbulence). The reader is referred to Part 1 of this review for a discussion of the injury mechanisms and the conclusions of these studies.

3.1.1 Mechanical Injury (Strike and Grinding)

Although strike has always been considered one of the most obvious and major causes of injury among turbine-passed fish, there have been surprisingly few attempts to study this mechanism under controlled conditions. Most investigations of strike have focused on estimating the probabilities that fish will contact some part of the turbine machinery, especially the blades and wicket gates (von Raben 1957; Montén 1985; Solomon 1988; Nece 1991). Some of these analyses assume that any contact will cause serious injury or death, or else assume that a constant percentage of fish striking the blade will be killed. In fact, Bell and Kidder (1991) pointed out that not all fish that collide with runner blades and vanes are killed; the lethal rate of strike is variable. Laboratory tests by Turnpenny et al. (1992) found that even at rapid velocities (5.2 m/s), collision with the blunt leading edge of the runner blade (e.g., near the hub) caused little damage and no mortality among several species of fish. Collision with a narrower blade profile, as occurs near the blade tip, caused severe injury. Consequently, it is important not only to estimate the probability of contact with the turbine machinery, but also the probability of injury once that contact is made.

Turnpenny et al. (1992) examined the assumption that the consequences of a strike are the same irrespective of where the fish is struck. They constructed an experimental apparatus that allowed them to assess injury resulting from different blade leading edge profiles and collision velocities, both of which become more injurious with increasing distance from the hub. In addition, effects of collision of mechanical structures with different parts of the fish's body were examined.

A short section of the leading edge of a turbine blade was mounted to a set of springs in the test tank (Figure 4). The blade was moved along tubular rails to one side of the tank with a pneumatic ram and held in place with levers. When the lever was released, the blade was fired at either lightly anaesthetized, free-swimming fish or freshly killed fish suspended in the path of the blade. Actual velocities of the blade, which ranged from 5.2 to 7.4 m/s, were measured by a chopped light beam detector, and were precisely reproducible (Turnpenny et al. 1992). Collisions were recorded by a video camera. Because the collisions occurred very rapidly

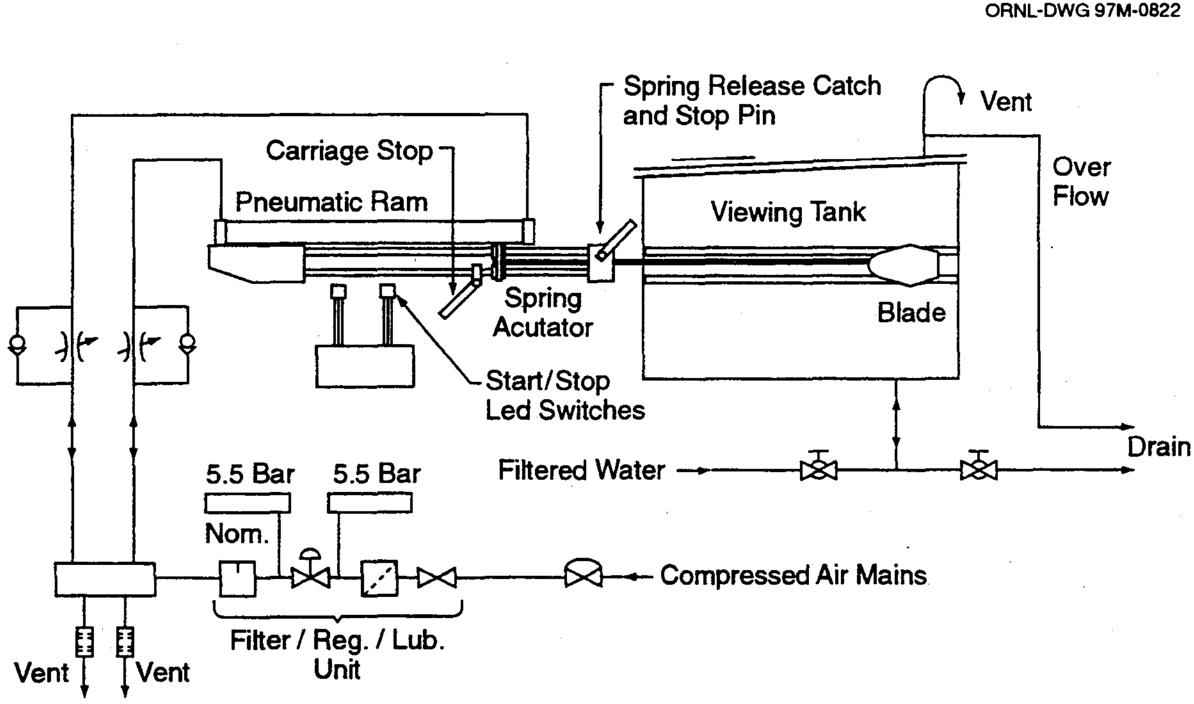

Figure 4. General arrangement of the fish/blade strike simulator used by Turnpenny et al. (1992).

(on the order of 60 milliseconds), there was no evidence that the live fish actively responded to the approaching blade, and most tests were done with freshly killed fish. Post-test analyses included measurements of the fish's standard length, mass, center of gravity (which had an important bearing on likelihood of injury), fish-to-blade angles in two planes, and the distance between the fish's center of gravity and the center of gravity of the blade section. Fish in the path of the blade that were deflected to one side by the blade or its "bow wave" (as opposed to being folded around the blade) were recorded as non-strikes.

Injuries to fish struck by the blade section included scale and mucous loss, bruising, eye damage, and internal bleeding (Turnpenny et al. 1992). It was found that small fish (of a few grams weight) were swept around the front of the blade along with the streamlines of the water. Larger fish, owing to their greater inertia, have a higher probability of colliding with the blade. The probability of collision in this laboratory experiment was expressed as the ratio of two measurements: the shortest distance between the fish's and the blade's centers of gravity and the fish's body length. Their studies affirmed the importance of fish weight when calculating strike probabilities; very small fish (<20 g) in the path of the blade virtually avoided collision, whereas larger fish (>200 g) had nearly a 100 percent chance of being struck.

Mechanical injury to turbine-passed fish can also be caused by grinding, in which fish are drawn through narrow openings or gaps between structures in the turbine passageway. There have been no studies to assess the importance of grinding as a factor in turbine-passage mortality.

3.1.2 Pressure

Studies of the effects of pressure on fish have been carried out for nearly a century, mainly with the goal of understanding the physiology of fish living at great depths in the sea. For example, Sébert et al. (1990) described a hyperbaric chamber that allows fish to be held at pressures of up to 101 atmospheres (atm) for at least one month; this apparatus was used to study the physiological adaptations of eels to vertical migrations in the sea. Of greater interest here are the relatively recent studies of rapidly varying pressures that have been done to assess the effects of explosions, pump passage, or turbine passage. For example, Rowley (1955) put rainbow trout into a small lucite chamber, increased pressure with a hand pump, and, after an exposure of less than 1 minute, released the pressure instantaneously. This time-pressure regime simulated pressure changes in a hydropower penstock, but was not similar to that experienced by turbine-passed fish. Foye and Scott (1965) exposed fish to rapid pressure increases (atmospheric to 2065 kPa instantaneously, followed by a 10-minute period of pressure decrease back to atmospheric) in a 102 cm X 30 cm cylindrical steel tank. This regime was designed to mimic pressures experienced by fish entrained during the pump cycle of a pump storage project. In order to better simulate turbine passage, Muir (1959) constructed a test apparatus that increased the hydrostatic pressure in a small (20 cm X 10 cm) cylindrical chamber to about 570 kPa (5.6 atm) in a few seconds, then reduced the pressure to 7 kPa (0.07 atm) in 0.01 seconds.

Harvey (1963) studied the effects of increased water pressures on sockeye salmon fry and smolts using a cylindrical steel chamber, 91 cm (3 ft) long and 30 cm (1 ft) in diameter. One end was fitted with a removable flange secured by bolts. Pressure was applied by means of a pump and regulated with valves and a bypass over the range of 101 to 2165 kPa (1 to 21 atmospheres). The most rapid rate of pressure increase achievable with this apparatus was about 69 kPa per second, but pressure could be returned to atmospheric instantaneously. Subatmospheric pressures were investigated with a smaller cylinder in which pressures were reduced by means of a vacuum pump. Pressures as low as 2 kPa (0.02 atmospheres) were achieved in this test chamber. Pressures were measured with a transducer and recorded on an oscillograph. Ends of the chamber were fitted with plastic ports in order to observe fish behavior. Harvey (1963) did not report the variability in actual pressures achieved in the chambers, but noted that it was not possible to control precisely the desired vacuum (subatmospheric) conditions.

Knable and Feathers (1983) pointed out that many of the early studies used compressed air to increase pressure in the test chambers. This technique could result in supersaturation of gases in the water and tissues of the test organism, which in turn could cause gas embolisms (gas bubble trauma) during subsequent decompression. They developed a large (200 cm X 70 cm) test chamber that could maintain pressures of 520 kPa for at least 24 h with a continuous exchange of water. Although the complicating effects of supersaturated gases on pressure responses were eliminated, the chamber was not designed to recreate the rapid pressure increases and subatmospheric pressures that are common to hydropower turbines.

Turnpenny et al. (1992) constructed a 140-L pressure flux vessel in which pressure could be adjusted from 10 to 400 kPa (0.1 to 3.9 atm) by means of a piston. Control of the piston was achieved with a computer-generated signal to a hydraulic actuator. A control program on the computer allowed the desired pressure time series to be defined and stored, in order to generate repeatable time-pressure patterns. Their calculations took into account the compressibility of the fish's swim bladder in determining the amount of piston movement needed to create the desired pressure change. Provided that the weight of fish (and therefore size of swim bladder) introduced to the chamber did not exceed the design limit and the pressure vessel was properly sealed, time-pressure curves were achieved within 5 percent of target values throughout the run. In addition to the main pressure chamber, which was constructed of stainless steel, an accessory plexiglass chamber connected directly to the main chamber allowed observation of the behavior of individual fish.

Montgomery Watson (1995) exposed smolt-sized rainbow trout to different levels of water pressure and dissolved gas saturation in laboratory chambers. The pressure exposure system consisted of two acrylic cylinders, each 55 cm (22 in) long and 27.5 cm (11 in) in diameter, connected to a system of hydraulic and pneumatic cylinders and their controls and water supply (detailed schematics are provided in the report). The chambers were connected to hydraulic cylinders which in turn were connected to pneumatic cylinders. A computer-controlled gas pressurization system caused the pneumatic cylinders to change the position of the hydraulic cylinders, thereby pressurizing or depressurizing the test chambers while maintaining control over dissolved gas concentrations. Pressure could be dropped from 300 kPa (100 feet of head or 3 atm) to the vapor pressure of water in 0.1 seconds.

Groups of Age 0, 9 to 10 cm-long rainbow trout were exposed to the following pressure regime in the test chambers: Initial Pressurization Phase (atmospheric pressure to 300 kPa in 30 to 60 seconds); Transient Phase (drop to the vapor pressure of water, 2 kPa, in 0.10 seconds); Low Pressure Phase (close to the vapor pressure of water for 0.25 seconds); and Recovery Phase (return to 115 to 120 kPa in 30 to 60 seconds) (Montgomery Watson 1995). This was estimated to be the worst case pressure condition for a fish passing close to a turbine blade at McNary Dam. Groups of 20 test fish in each chamber were exposed to the pressure transients (and different gas saturations) and held in the chambers for an additional 30 minutes. After the 30 minutes were up, treatment and control fish were removed from the chambers, combined, and introduced to a tank containing adult rainbow trout predators. After 25 minutes, survivors were removed from the predation tank.

Montgomery Watson (1995) also established the performance characteristics of the pressure test chamber system by running ten pressure cycles on each chamber and measuring the actual pressures achieved. An example of the repeatability of the pressure regimes created in the chambers is shown in Figure 5. Although there was some variability

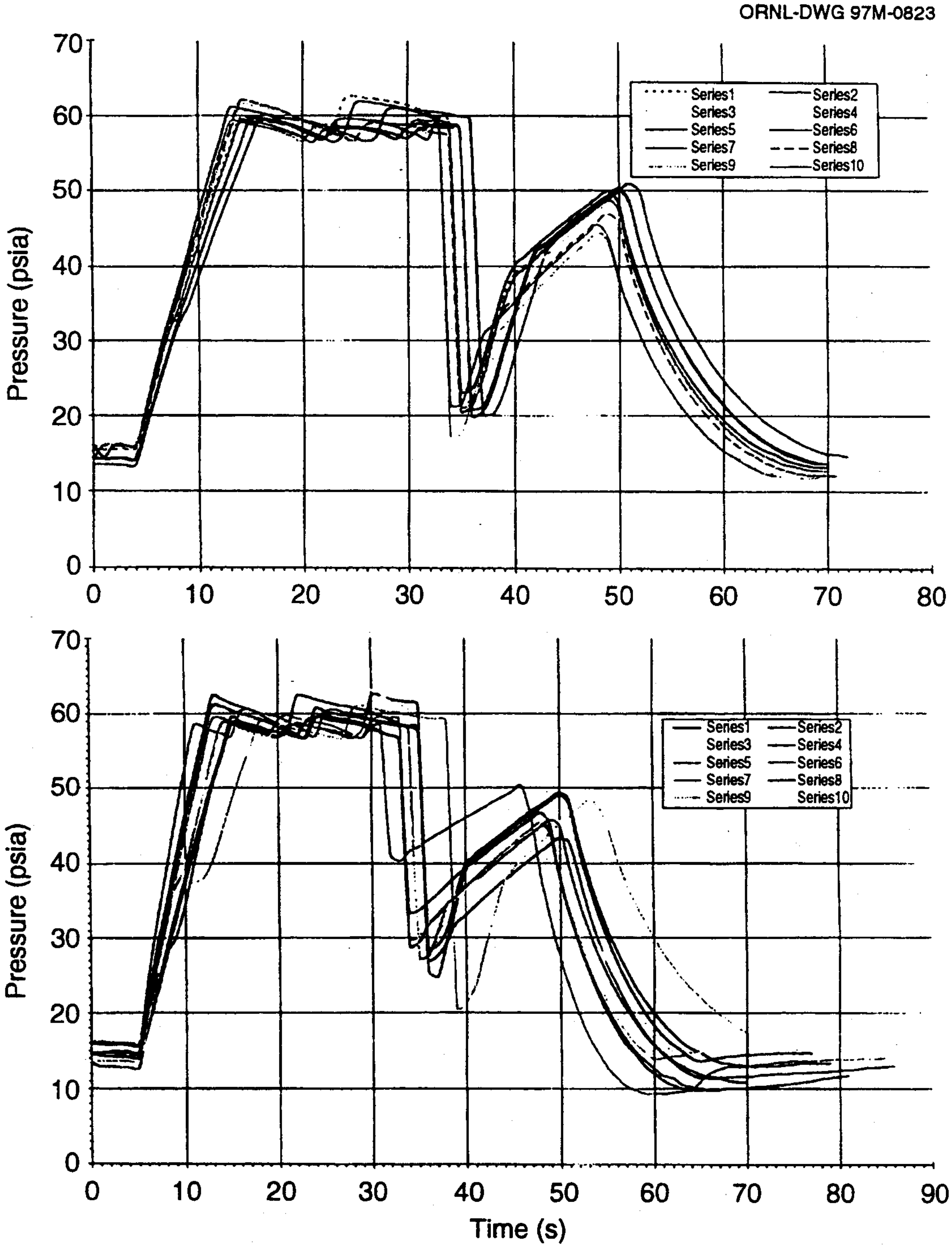

Figure 5. Example of the repeatability of the pressure vs. time regimes created in the test chambers used by Montgomery Watson (1995).

in the time-pressure histories, the absolute positive and subatmospheric pressures achieved were similar among the replicates.

In summary, nearly all of the pressure studies have been carried out by placing the fish in cylindrical chambers and exposing them to the desired time-pressure regime. A variety of response variables have been examined, ranging from swim bladder injury to direct mortality to changes in susceptibility to predation. Recent studies have been more conscious of the complicating effects of dissolved gases in the test chamber. Fish held in a static chamber may consume enough of the dissolved oxygen to be affected, whereas supersaturation of nitrogen may lead to gas bubble trauma when the chamber is decompressed. Refinements in equipment have enabled investigators to control dissolved gas concentrations and to test rapid and extreme pressure changes (similar to those experienced by turbine-passed fish) in precise, repeatable ways.

3.1.3 Cavitation

The importance of cavitation as a possible source of turbine-passage mortality was recognized early. For example, Muir (1959) noted that fish passing through a region of cavitation will be subjected not only to the stresses associated with a partial vacuum but also to pressure intensities resulting from the collapse of the vapor pockets. He exposed fish to cavitating conditions in the laboratory with a water hammer apparatus; water moving rapidly through a pipe between two tanks was abruptly stopped by the rapid closing of a check valve. A wave of reduced pressure, starting at the check valve, swept downstream through the pipe to a pipe riser containing the experimental fish. Pressure in the riser was reduced to the vapor pressure of water, as evidenced by a transducer and the formation of a vapor pocket. The development of a vacuum was followed by a rapid opening of the check valve, which increased the pressure again and caused the vapor pocket in the riser to collapse. Test fish were examined microscopically for evidence of hemorrhaging.

Ramamurthy et al. (1984) described an apparatus for generating cavitating conditions in the laboratory and studying its erosive effects on different materials. The apparatus consisted of a 61-cm-diameter circular disk in a closed, water-filled chamber. The disk was mounted on the shaft of a motor and rotated in the chamber at 1800 rpm. Equilateral triangular prisms were mounted on the surface of the disk to form the cavitating source, and the material to be tested (e.g. strips of aluminum) was also fixed on the disk in the wake region formed by the prism. As the disk spun rapidly, the prism generated cavitation bubbles which were swept toward the nearby test material. Although this rotating disk apparatus is widely accepted as a device to study the resistance of materials to cavitation, it does not appear to be adaptable to assessing cavitation damage to fish. The effects of spinning and turbulence would be harmful to the fish as well, and these would be difficult to separate from the effects of cavitation.

Turnpenny et al. (1992) used an underwater spark generator to create individual cavitation bubbles in a static water tank. The vapor bubble created by a spark in the 0.5-mm electrode gap reached its maximum size of 8-10 mm within 1.4 milliseconds, then collapsed in less than 0.1 milliseconds. The electrode gap was surrounded by a brass cage, within which freshly killed fish were held during bubble collapse experiments. Fish were mounted on a wooden splint in order to ensure a replicable geometry between the fish body surface and the spark gap. Individual fish were exposed to a series of five successive bubble implosions positioned at intervals along the head and body. Fish were photographed during the cavitation bubble implosions, and were subsequently examined for tissue damage. Although no tissue damage was observed, Turnpenny et al. (1992) cautioned that the results of these limited tests should not be interpreted to mean that cavitation is not a problem in operating turbines. Pitting damage is often seen on the runners of cavitating turbines, but such effects were not observed on brass plates exposed to collapsing bubbles in these laboratory studies. Energy levels associated with cavitation bubble collapse must be vastly higher than those that Turnpenny et al. were able to generate with their experimental apparatus.

3.1.4 Shear Stress and Turbulence

Johnson (1970a,b; 1972) reported on a series of tests to examine the injury and mortality among juvenile salmonids entering a tank of water through a submerged, high-velocity jet. The motivation for the tests was to determine whether fish would be injured in the high-velocity flows associated with slotted bulkhead downstream fish bypass systems at Columbia River Basin dams. Juvenile coho, chinook, and steelhead were introduced into a 36-cm (14-inch) supply line which narrowed to either a 10-cm or 15-cm (4-inch or 6-inch) nozzle. The nozzle was submerged in a water-filled test tank that measured 12 m long, 6 m wide, and 2 m deep (40 ft X 20 ft X 6 ft). Depending on the test, the water jet coming from the nozzle had a velocity of 17.5, 20.4, 23.6, or 28.0 m/s (57.5, 67, 77.5, or 92 ft/s). Most test fish entered the supply line from the lock in which they were held within 10 seconds, then rapidly passed through the nozzle into the tank. The jet was left in operation for three to four minutes after all fish had left the lock to ensure that they had passed through the nozzle. The pump was shut off, the tank drained, and the fish collected for post-test observation. High-speed cameras (1200 frames per second) recorded movements of the fish as they were ejected from the nozzle. Later examination of the film at slower speed (16 frames per second) provided a minimum viewing time of 5.25 seconds for each fish as it traveled in the jet.

Johnson (1970a) observed no mortality at the lowest velocity tested, 17.5 m/s. Mortality averaged 2.4, 7.2, and 31.0 percent at jet velocities of 20.4, 23.6, and 28.0 m/s, respectively. Johnson (1970b) pointed out several other possible causes for the observed mortalities, some of which he was unable to rule out completely with the experimental apparatus. Possible alternative causes for fish mortality include mechanical damage to 20- to 23-cm-long fish when forced sideways through a 10-cm-diameter nozzle, and the sudden pressure drop that occurred when fish passed through the nozzle. Although lowest pressures experienced by test fish were estimated remain above atmospheric, Johnson (1972) reported an intense plume of cavitation near the nozzle at the two highest velocities that may have injured fish which exited the jet within 1 m of the nozzle. A final complication of these studies is that the experimental apparatus didn't allow for precise control and measurement of shear forces experienced by fish. The location of fish in the jet, orientation of fish as they exited the nozzle, and location where they exited the jet into the relatively static water tank could not be controlled or replicated. Finally, fish occasionally re-entered the jet due to water circulation patterns in the tank; the shear that was experienced when fish already in the tank are drawn back into the jet and instantly accelerated from zero to nearly 28 m/s added an unquantified stress that may have been reflected in the mortality.

Groves (1972) used a modification of the water jet technique to study the effects of shear on juvenile coho, chinook, and sockeye salmon. Unlike the approach described by Johnson (1970a), test fish did not pass through the jet's nozzle. Rather, fish were flushed into the water tank through an angled tube that was positioned so that they would strike the jet within 7.6 cm (3 inches) of its emergence from the nozzle. Jet velocities ranging from 9 to 37 m/s (30 to 120 ft/s) were tested, although the exact velocity of the boundary of the jet that fish actually contacted was not known. Although the water in the center of the jet was moving at speeds approximating the calculated velocities, fish contacted on the outer margins of the stream where the water was slower. Further, the actual shear forces experienced by fish striking the jet were not calculated. High-speed photography (1,600 frames per second) allowed subsequent analysis of the path of fish entrained in the jet and the cause of injuries. Groves (1972) concluded that fish could be injured in any high energy flow situation that creates momentary (as low as 1 millisecond), localized points of sharp velocity change. He noted that such rapid, transitory events would be difficult to pinpoint in specific field conditions, and impossible for fish to detect or avoid.

Killgore et al. (1987) exposed paddlefish larvae to turbulence with an experimental apparatus that was essentially a small version of the one used by Groves (1972). A jet of water was pumped into a circular, 27-cm-diameter bucket. In the center of the bucket was an 11-cm-diameter pipe, which created a circular raceway. The jet caused water to move in a circular fashion within the bucket, at velocities of 22 to 59 cm/s. Turbulence was quantified by measuring pressure changes at four locations within the bucket. Pressures (which were equated with levels of turbulence by the authors) ranged from 1,774 to 6,421 dynes/cm^2. Paddlefish yolk-sac larvae were exposed to a particular time-turbulence regime and examined immediately afterward to assess survival.

Turnpenny et al. (1992) also tested the effects of shear using the Groves (1972) experimental approach. Water was discharged into a large flume tank (8 m long X 1.5 m wide X 1 m high; 0.6 m water depth) through a jet nozzle at velocities of 5, 10, 15, 19, and 20 m/s. The calculated relationship of shear stress (expressed as N/m^2, where 1 N/m^2 equals 10 dynes/cm^2) to flow rate of the jet is shown in Figure 6. The calculated variation in shear across the jet centerline with distance is shown in Figure 7. Fish were individually fed into the water jet through an introduction tube, entrained into the jet stream, swept to the quiet area of the tank, and were netted out. Fish were immediately examined for damage, then held for 7 days to assess long-term survival. High-speed photography of the fish's movements showed that upon entering the tank fish were immediately drawn into the center of the jet and then "pirouetted" along the tank in a circular motion. The resulting bending motion cause visible creases on the outside of the body of some fishes and crushed the internal organs of others.

Turnpenny et al. (1992) regarded their approach to studying shear as the most relevant to turbine passage because it can

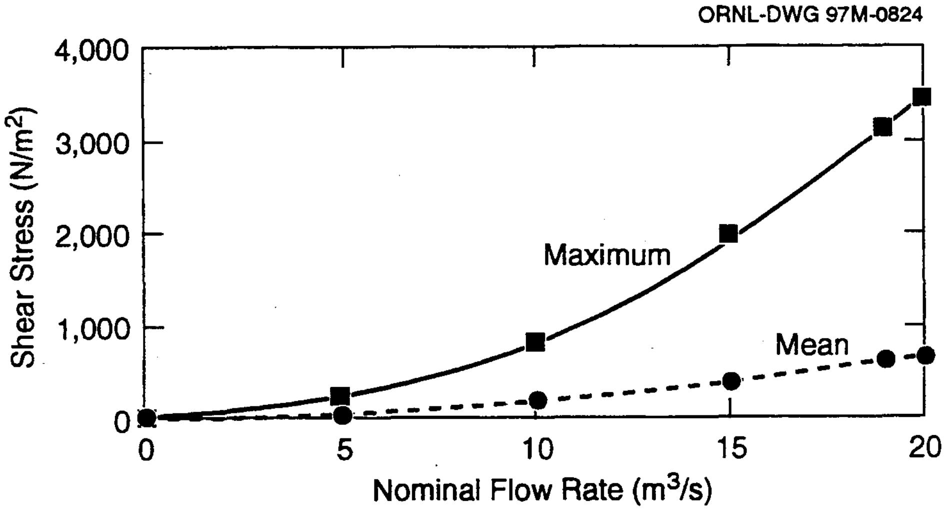

Figure 6. Calculated relationship between shear stress (N/m^2) and flow rate of the jet used in the shear stress studies of Turnpenny et al. (1992).

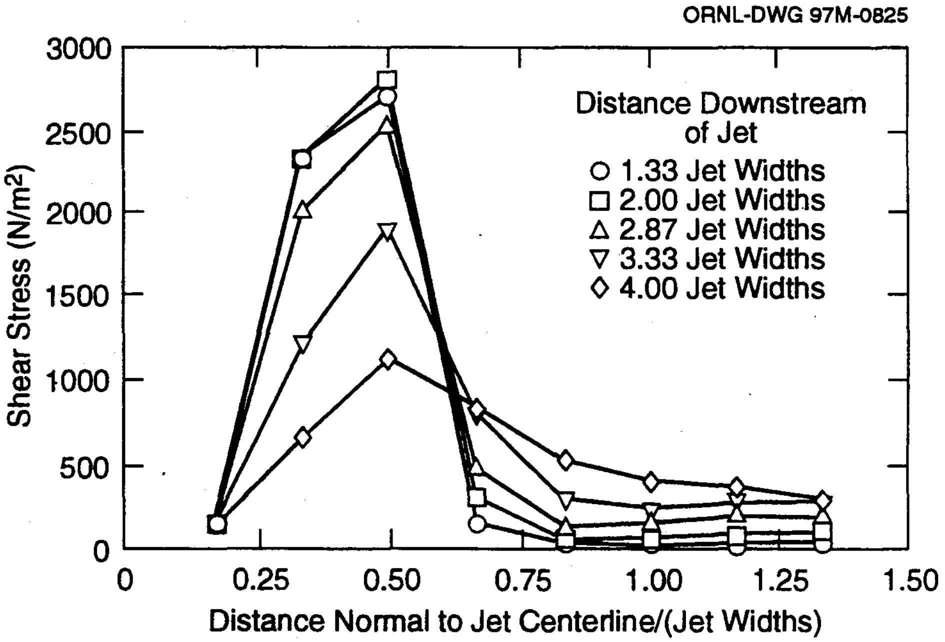

Figure 7. Calculated variation in shear across the jet centerline with distance for the Fawley nozzle used in the shear stress studies of Turnpenny et al. (1992).

produce the effects of localized shear stresses on the fish's body which lead to scale loss, eye damage, and gill damage. They felt, however, that this technique does not adequately reproduce the forces of elongation, compression, and torsion that a fish would experience within a turbine when different parts of its body enter regions of differential velocity. Such forces might lead to creases and internal organ damage seen in some of the fish. They suggested that such effects would be worth examining in future studies.

Shtaf et al. (1983) examined the effects of turbulence on fish swimming behavior in small laboratory flumes. Because turbulence was generated by placing screens and other obstructions in the flume it was not strictly predictable or reproducible. The investigators were interested in studying swimming behavior in natural waters, so water velocities in the flume were low (13 and 18 cm/s) and the resulting turbulence was not damaging. Degree of turbulence was expressed as the standard deviation of instantaneous water velocity divided by the mean water velocity. Hence, the greater the deviation from mean water velocity in the flume, the greater the degree of turbulence. This formulation is useful for comparing relative turbulence associated with different flows and structures within the same experimental flume, but does not provide an absolute expression of turbulence and shear forces to which the fish were exposed.

Morgan et al. (1976) investigated the effects of shear stresses on striped bass and white perch eggs and larvae. They were particularly interested in reproducing the rotational and deformational forces that are exerted on a fish egg exposed to adjacent flow fields of different velocities. They developed a shear stress exposure chamber which consisted of two fixed concentric plexiglas cylinders, 20.3 and 30.5 cm in diameter. A third, 25.4-cm-diameter cylinder was placed between these two water-filled cylinders and rotated at speeds ranging from 14 to 231 rpm.

The shear stresses experienced by fish in the experimental apparatus was a function of the speed of rotation of the middle cylinder (Morgan et al. 1976). In addition, flows in the water space between the small fixed cylinder and the rotating cylinder (inner annulus) were different from those in the space between the large, outer fixed cylinder and the rotating cylinder (outer annulus). In the inner annulus, the centrifugal forces are in the direction of increasing radial velocity (i.e., nearest the inner wall of the rotating cylinder). This stabilizes the flow in the inner annulus into a circularly annular pattern (Covette flow). In the outer annulus, however, faster moving flow near the wall of the rotating cylinder is pushed radially outward by centrifugal forces, resulting in turbulent mixing (Taylor instability). Shear stresses at the wall of the inner annulus were calculated; at the lowest rpm, shear was estimated to be 0.052 dynes/cm^2. The authors would not determine analytically the shear stresses in the turbulent outer annulus, but instead assumed a Reynolds number of 4,738 and a corresponding shear stress value of 0.64 dynes/cm^2. Shear forces in the outer annulus were an order of magnitude greater than those in the inner annulus at the lowest rotational speed, and the discrepancy increased with higher rpm. Fish eggs and larvae were introduced into the outer, turbulent annulus and exposed to shear stresses for various periods of time. Short-term studies used shear stresses between 76 and 404 dynes/cm^2 for 1 to 20 minutes. Longer term studies exposed eggs and larvae to shear stresses between 0.64 and 86 dynes/cm^2 for 2 or 3 days. Based on these studies, Morgan et al. (1976) developed regression equations that related mortality to shear level.

It is clear from these experiments that the potential injury mechanisms of shear stress and turbulence have proven difficult to study in the laboratory. The concept of shear and precisely how it might affect turbine-entrained fish have been difficult to describe or even to express quantitatively. As a result, the shear stress experience has not been reliably simulated in a quantitative and reproducible manner. Similarly, turbulence has not been rigorously examined. Severe turbulence in a hydroelectric turbine system is believed by some to have adverse effects, but, like shear, the mechanism has been difficult to express, quantify, or apply in controlled studies. The spinning and buffeting associated with turbulence in the draft tube and tailrace are less likely to cause injury and direct mortality than they are to disorient the fish so that it is more susceptible to indirect mortality (predation). Laboratory studies are needed to expose fish to the levels of turbulence that occur in a turbine system and to assess the consequent direct and indirect mortality.

3.2 Field Techniques

A variety of techniques are available for studying turbine passage rates and mortalities of entrained fish, including tailrace netting, Turb'N tags, Passive Integrated Transponder (PIT) tags, or hydroacoustics. These techniques are presently used to quantify the numbers of fish entering and leaving the turbine (and the consequent injury and mortality), but they do not provide any information about the behavior of fish within the turbine. For the purpose of improving turbine designs, there is a need to go beyond these applications and to develop an understanding of the precise path taken by turbine-passed fish. Flow path visualization techniques are being explored in order to define exactly those areas of the turbine that fish pass through and the mortalities associated with these areas. For injury mechanisms such as mechanical damage (blade strike, grinding, or contact with walls and other obstructions) or cavitation, some of these visual techniques could be used directly. For other injury mechanisms (pressure and shear), visual observations of the flow path of entrained fish would need to be accompanied by estimates or measurements of the levels of these stressors throughout turbine passage. Some of these techniques have been employed at hydroelectric power plants, whereas others have not but may be adaptable. The two most readily adapted techniques for visualizing the flow path of individual entrained fish are low-light-sensitive video cameras and hydroacoustics.

3.2.1 Low-Light-Sensitive Underwater Video Camera

Nestler and Davidson (1995a) used underwater video cameras to study the effects of bypass screens on water flows and smolt behavior at McNary Dam. Three different camera types were used (specifications are shown in Table 8), but no comparisons among cameras were reported. Cameras were mounted on the screens and aimed laterally to look across the surface of the screen. A 120-W incandescent light source was attached to the camera housing and aimed in the same direction as the camera. Camera depth of view was about 0.6 to 0.9 m (24 to 36 inches) for the screen surface, but when illuminated the highly reflective bodies of smolts could be detected at a distance of about 1.2 m (48 inches). During imaging, each camera was connected to a video cassette recorder and a television monitor.

Table 8. Specifications of low-light sensitive underwater video cameras used at McNary Dam by Nestler and Davidson (1995a).

Camera type	Sensitivity	Lens	Power	Size	Weight in water (kg)	Cost ($)
Underwater CCD Monochrome Television Camera OE 1359	0.03 lux on the sensor	3.7 mm, f/1/6-f/300 Auto Iris	16-24 V d.c. at 200 mA maximum	152 mm length - 53 mm diameter	0.27	10,500
DeepSea Power & Light Micro-SeaCam Underwater Video Camera	1 lux	60 degree angular field of view in water	12 VDC at 140 mA maximum	122 mm length - 36 mm diameter	0.3	5,200
Silicon-Intensified-Target (SIT) TV Camera SL-99	1,000 times greater than a standard vidicon	12.5 mm, f/1.4	12.7 VDC at 850 mA ± 50 mA	356 mm length - 95 mm diameter	1.8	10,500

Nestler and Davidson (1995a) recognized that the presence of the video camera, illumination system, and mounting hardware would produce significant hydraulic anomalies that could influence fish behavior. In addition, the illumination field required for video imaging could also attract or repel smolts. From studies designed to quantify the effects of these potential biases, they concluded that smolts did not concentrate in the wake of the camera mounting system. The fish swam around the mount without apparent response other than to avoid contact with the structure. Different illumination intensities influenced the number of fish imaged, but did not appear to alter the behavior of fish relative to impingement on the screen. Because fish behavior could not be observed with video cameras without some minimum level of illumination, a "no illumination" condition could not be examined and the biasing effects of artificial illumination on entrained fish behavior could not be completely resolved.

Moore and Scott (1988) also used a Silicon Intensifier Target underwater camera in their studies of the behavior of recently emerged trout fry. Because these fry emerge from the redds only at night, a low-light camera was needed to record their activities. The camera was housed in a special support and placed in the stream immediately downstream from the redd. The stream bed was illuminated from above the water surface with an artificial light source, the intensity of which was equivalent to full moonlight. The authors did not report the type of light source, exact illumination intensity, or the camera's viewing range. By means of the camera they were able to observe and videotape the swim-up and rapid downstream movement of trout fry under natural nighttime light levels.

Vaughn (1995) described a prototype underwater camera system that was used to inspect submerged traveling screens at the John Day Dam on the Columbia River. The monochrome cameras required a minimum scene illumination of 0.9 lux and had a fixed focus (0.1 meters to infinity). Illumination was provided by 250-watt submergible lights with variable intensity control (range of light intensities was not reported). Depending on water clarity, visibility with this video system ranged from 0 to 1.5 meters, but typically was about 0.3 meters.

Because low-light sensitive underwater cameras can directly observe (and record for later analysis) objects moving through the turbine, they have considerable value for understanding whether fish behavior significantly influences injury rates. For example, video imaging may be the most reliable technique for assessing any tendency of fish to swim away from or towards obstructions or areas of cavitation and shear stress. The movements of live fish within the turbine environment could be compared to those of dead fish or other neutrally bouyant objects to determine whether such mechanisms as blade strike have a significant behavioral component.

These low-light sensitive video camera studies all relied on external lights to illuminate a darkened area. The limited viewing range and potential effects of illumination on fish behavior are major limitations to the use of those techniques inside of a turbine. However, it may not be necessary to illuminate the turbine passageways if the entrained fish is fitted with a light-emitting tag, such as a light-emitting diode (LED), that could be detected by the camera. A single light-emitting tag could be used to estimate the fish's rate of passage through the turbine and a rough estimate of the actual flow path. Further, attaching two LED tags with different colors or different blinking rates could allow estimates of the orientation and path of the fish in three dimensions. LEDs have been incorporated into instrument packages used to sense and record depths achieved by marine diving birds (Wilson et al. 1989; Croll et al. 1992). However, these packages are still too large (9 cm X 1.5 cm; 6 to 11 g) to be attached to turbine-entrained juvenile fish. The key to the use of this technique is to develop a light-emitting tag that is small, lightweight, and can be detected at reasonable distances in turbid water and in a darkened turbine passageway.

The value of low-light video imaging technology to visualize the flow paths of turbine-passed fish is presently limited by (1) the camera's viewing range and (2) potential biases associated with the unnatural hydraulic and illumination conditions caused by the presence of the camera. At best, the cameras used by Nestler and Davidson (1995a) were only able to detect fish passing within 1.2 m (4 feet). Visualizing a long flow path taken by an individual fish would require a network of integrated, closely spaced cameras. Increasing the intensity of conventional illumination to extend to viewing range of the camera could alter the fish's behavior and bias the results. Consequently, video imaging may be most useful for studying the passage of fish through relatively small areas such as gaps between blade and hub that have been suggested as likely sites for grinding injuries. The flow fields created by the camera and its mounting bracket and light source could be eliminated by installing all equipment outside of the turbine and imaging the fish through viewing ports. Fish behavior changes caused by illumination could be reduced or eliminated by using cameras that are sensitive to wavelengths not perceived by fish or the development of a small light-emitting tag.

3.2.2 Hydroacoustic Techniques

A variety of hydroacoustic techniques have been developed to study the movements of fish near hydropower projects (Thorne and Johnson 1993). Unlike hydroacoustic equipment mounted on commercial fishing vessels to monitor the movements of schools of fish in the open sea, measurements near a hydropower plant can be made from a fixed location, e.g., the dam or a stationary floating platform near the forebay. The general approach for fixed-location acoustic studies is to place one or more transducers on a fixed structure, aim the acoustic volume toward an area of interest (e.g., horizontally out into the reservoir), and sample fish as they pass through the ensonified acoustic beam (Steig and Johnston In Press). Fish passing through the beam produce echoes that can be tracked over successive ensonifications (pulses of the acoustic beam). Three general techniques have been developed: single-beam, dual-beam, and split-beam hydroacoustics.

Single-beam hydroacoustics - The simplest echosounders transmit sound in a single beam, which permits the range, but not the direction, of targets to be determined (MacLennan and Simmonds 1992). Ransom and Steig (1995) summarized the findings of numerous evaluations of spillway and sluiceway bypass effectiveness at Columbia River basin dams. Nearly all of these evaluations used single-beam hydroacoustics techniques to obtain relative estimates of fish passage rates. Typically, the transducers were placed on a fixed structure (e.g., intake wall or trash rack) and sampled salmon smolts as they passed through the ensonified beam. The focus of these studies was the movements of smolts immediately upstream of the dam or at the intake entrance; there is no indication that these techniques were used in the deeper turbine passages.

Dual-beam hydroacoustics - Whereas single-beam techniques allows relative numbers of fish to be estimated, the dual-beam technique can be used to estimate directly the acoustic target strength, which in turn can be related to the length of individual fish or the biomass of schools of fish (Love, 1971; Johnston et al. 1993). Johnston et al. (1993) used dual-beam hydroacoustic techniques to estimate target strengths (and fish lengths) of fish entrained at two hydroelectric dams.

Split-beam hydroacoustics - This recently developed technique has the ability to estimate the absolute velocity and three-dimensional paths of individual fish passing through the beam. In addition, the individual fish's target strength can be measured, from which estimates of size or mass can be made. This technique has been employed at the entrance to hydropower dams to monitor that movement patterns of downstream-migrating fish within the hydropower reservoir.

For example, Steig and Johnston (In Press) described an application of split-beam hydroacoustic techniques to the study of fish movement patterns in the forebay of Rocky Reach Dam in Washington. An elliptical-beam transducer was mounted on each of the four corners of a barge and aimed downward and out into the forebay. Fish were detected in cells within the ensonified volume. Each cell was 5 m long (measured outward from the transducer), but had a volume that increased with distance from the transducer, owing to the elliptical shape and increasing width of the beam with distance. The split-beam technique was capable estimating the numbers of fish in each cell (and thus density), acoustic size estimate (target strength) of each fish, and the three-dimensional trajectory of each fish. Precision of the estimates was not given, and results were presented only for average density, target strength and trajectory of all fish in a given cell. Fish movement patterns in the lower reservoir and forebay indicated that fish tended to follow bulk flow near the powerhouse.

One of the prerequisites for estimating target strength (fish size) *in situ* is the ability to separate single target echoes from multiple echoes. That is, two small fish moving close together should not be interpreted as a single, large fish. Sole et al. (1995) examined these potential biases in a laboratory test tank with a Simrad EK500 split-beam echo-sounder. They concluded that (1) the single-fish discriminator software showed a bias against accepting weaker targets, and (2) multiple echoes from targets as far as 0.7 m apart were falsely accepted as single echoes. The authors cautioned that these discriminators may be unreliable for estimating target strength of pelagic organisms, unless fish are widely separated and differ little in target strength. Biases such as these will have to be corrected in order for split-beam hydroacoustics techniques to be successfully applied to visualizing the flow path of fish within a turbine.

Ransom and Steig (1994) listed the advantages and disadvantages of hydroacoustics techniques for fisheries studies. The advantages include:

1. Hydroacoustics readily provide estimates of fish entrainment rates and abundance.

2. High sampling power and relatively low manpower requirements reduce overall study costs.

3. Hydroacoustic techniques do not harm the sampled fish or alter their behavior.

4. Because large quantities of data can be easily acquired, statistical comparisons and interpretations are facilitated.

5. Net avoidance and other netting bias problems are avoided.

6. Real-time data analysis is possible.

7. Hydroacoustic techniques allow documentation of fish behavior. For example, split-beam acoustic techniques can directly estimate fish velocity and three-dimensional movements.

8. Hydroacoustics have been used extensively at power plants throughout the world for nearly 20 years.

Ransom and Steig (1994) also pointed out the disadvantages of hydroacoustics studies:

1. Direct species identification is not yet possible.

2. Specialized, costly equipment is needed.

3. Specialized training is required.

4. If very small fish are to be monitored, the technique may be susceptible to background interference. At some dams, excessive turbulence, entrained air, and electronic interference can limit the usefulness of hydroacoustics.

This last disadvantage may be the greatest problem associated with using hydroacoustics techniques to visualize the flow path of fish through the turbine. Hydroacoustics studies at hydropower plants have been oriented toward monitoring the movements of fish in the lower reservoir, forebay area, trash racks, or in the vicinity of the submerged screens (see, for example, Matousek et al. 1995; Williams et al. 1995). There do not appear to be any applications of these techniques to the interior of the turbine or draft tube, where turbulence and electronic interference are greatest. Entrained fish are most concentrated in these areas, such that the problem with discriminating multiple echoes (Sole et al. 1995) may be difficult to overcome. Finally, compared to the reservoir and forebay areas, fish move very rapidly through the turbine and draft tube. Adequate detectability requires the correct combination of ping rate and beam width, relative to the fish's velocity. The ability of split-beam hydroacoustics to estimate velocity and three-dimensional movements under these conditions may be exceeded.

3.3 Conclusions and Recommendations

Coordinated laboratory and field studies are needed to understand the relative importance of the potential injury mechanisms associated with turbine passage. Pressure changes are easy to study under controlled conditions. The rapid pressure increases and decreases experienced by an entrained fish can be reliably simulated in the laboratory, and as a result more is known about this stress than any other. At the other end of the scale, techniques for studying fluid shear stresses and turbulence are not well developed. Shear and turbulence have been difficult to recreate in laboratory experiments, and little is known about the levels of injury, direct mortality, and indirect mortality (predation and disease) that may result from exposure to these stresses in a hydroelectric turbine.

There have been substantial developments in both video and hydroacoustics techniques in recent years that might used to visualize the path taken by entrained fish in an operating turbine. This information is needed to develop a better understanding of the risk of strike and grinding, as well as the pressure vs. time, shear vs. time, and turbulence vs. time histories experienced by fish passing through existing and advanced turbines. Low-light sensitive video cameras, perhaps in conjunction with light-emitting tags attached to the fish, show promise for tracking the path of entrained fish. Split-beam hydroacoustics techniques can potentially detect and record a fish's movements in three dimensions with little potential for altering the fish's behavior. However, the ability of hydroacoustics to track fish reliably inside of a turbine, under conditions of high velocities, high turbulence, crowding of entrained fish, and electronic interference, has yet to be demonstrated.

4. Fish Behavior in Relation to Entrainment in Hydropower Turbines

The literature on fish behavior as it relates to passage of fish near or through hydropower turbines is reviewed in this section of the report. An evaluation was stimulated by the need to develop more "fish-friendly" turbine systems for hydropower facilities (Brookshier et al. 1995). One aspect of "friendliness" is compatibility of engineered systems with the normal behavior patterns of fish species and life stages in the vicinity of the generation facilities such that entrainment into turbines and injury in passage are minimized.

Turbine modelers and designers need to know how fish move into and through turbines in order to develop novel designs that are less damaging to fish. Biologists need to define whether fish can be simulated in computer and physical models as passive, neutrally buoyant particles distributed throughout the water mass entering a turbine or if they must be represented in ways that reflect specific fish distribution patterns, physical orientations, and directed swimming movements. Fish distribution patterns in a turbine intake would influence the parts of the turbine through which the fish pass (e.g., near the hub or near the blade tips). Physical orientations would affect the likelihood of being struck by a turbine blade. Capabilities of fish for directed swimming movements in the high water velocities of a turbine intake would influence the constancy of distribution patterns and orientations as fish approach the turbine runner. This report evaluates the knowledge and importance of these considerations.

Physical damage to fish that pass through hydropower turbines is a major source of mortality for many fish populations in the vicinity of hydropower projects (OTA 1995; NRC 1996). This is especially true for migratory species such as salmon for which the dam is a barrier to movement that must be traversed or the population spawning upstream perishes. Although successful technologies have been developed for passing adult salmon upstream over dams (through simulation in fish ladders of the features of the normal migratory habitat), passage of downstream-migrating juveniles has been difficult to manage and generally not very successful (NPPC 1994; Čada et al. 1994; Francfort et al. 1994).

Both guidance away from turbine intakes and injuries inflicted by the turbine system (including hydrodynamic aspects of the scroll case and draft tube) are influenced, if not determined, by the size-dependent behavior patterns of the entrained species. Most bypass systems for juvenile salmon at major hydroelectric facilities, which involve screening juveniles from deep turbine intakes, seem to have been designed to oppose normal fish behavior in dam forebays. Normal behavior is surface oriented and in the direction of flow (Williams et al. in press). The development of intake screening arose from the observations that fish pulled to unnatural depths of turbine intakes accumulated in the gatewells associated with the tops of the intakes. Recent success with surface flow bypasses (Johnson et al. 1992; Skalski et al. in press) can be attributed to those facilities' closer matches to normal migration behavior (Williams et al. in press).

Damage to fish in turbines is not restricted to species that migrate between fresh water and the ocean. Many freshwater residents undergo extensive movements over the course of the seasons. Some of these movements are necessary for successful completion of the life cycle in different portions of a river system. Dams can create obstacles to population success similar to those for ocean-going species. In other cases, local resident fishes in impoundments can be drawn into turbines accidentally as a consequence of their normal feeding and rearing processes in the vicinities of turbine intakes. Thus, it may be useful to consider a diversity of fish behaviors to minimize turbine-induced damages under a wide range of hydropower installations.

In this section we briefly introduce the sources of fish mortality from turbine passage, give a synopsis of earlier literature reviews of fish behavior near turbines with their conclusions, review relevant and current basic scientific information about fish physiology and behavior, review on-site data at dams, and finally provide generalizations and implications for improved design of turbine systems. Because the majority of *in situ* studies have been conducted with salmonids, this fish group necessarily dominates the empirical aspects of this evaluation. Academic research on the physiology and behavior of fish, in general, provide additional guidance. The primary technological focus is on fixed- or variable-blade, Kaplan-type, vertical shaft propeller turbines, the type found most commonly in the Columbia River basin and at other large hydropower installations.

4.1 Sources of Mortality in Turbines

Although the need for technologies for passing adult salmon upstream past dams on rivers such as the Columbia was obvious and led to early legislative mandates (e.g., the Federal Power Act of 1920, which provided that the Secretary of Interior may require fishways at all federally licensed hydropower projects) (OTA 1995), the need to provide downstream passage for salmon smolts was controversial (Mighetto and Ebel 1994). The need was not clearly documented until Harlan Holmes conducted a set of experiments at the newly completed Bonneville Dam on the Columbia River (H. Holmes papers on file at the University of Washington, Seattle; Bell et al. 1967). Holmes estimated that between the years 1938 and 1948 there were losses of 11 to 14% of juvenile salmon in passing through the turbines. These estimates were derived from the experimental release of several paired groups of marked juvenile chinook salmon *Oncorhynchus tshawytscha*, one group of each pair being released so the fish would pass through the turbines and the other released in the tailrace, with conclusions about turbine-caused mortality being based on differential return of adults in subsequent years.

Later studies measured losses by recovering fish released in turbine intakes in nets suspended in the tailrace (Schoeneman et al. 1961). Bell (1981) summarized studies conducted to 1980 (Kaplan-type turbines), predominantly at mainstem dams on the Columbia and Snake rivers, with a range of turbine-induced losses from 6 to 32% of juveniles. Recently, studies have been focused on new turbine technologies and attempts to isolate direct turbine-caused mortality with fish released and collected individually with "balloon tags." Vertical axis turbines at Rocky Reach Dam showed about 4% mortality with fixed blades and 7% with variable blades (RMC and Skalski 1994). Kaplan turbines at Lower Granite Dam showed 5.2% mortality directly from turbines (RMC and Skalski 1995). Mathur et al. (1996) estimated 7%

short-term (turbine passage only) mortality in Kaplan turbines of Rocky Reach Dam when test fish were released near the intake's ceiling and 5.3% when they were released near the centerline.

The early findings stimulated engineering studies designed to identify factors responsible for turbine-induced mortalities and to seek engineering solutions. Physical models of turbines and turbine facilities were used (Cramer 1965; Cramer and Oligher 1960, 1961). These studies led to generalizations that have guided turbine design and operation ever since, not only in the Columbia River basin but elsewhere (Bell 1981; Turbak et al. 1981; Lucas 1962). In general, it was concluded that fish survival follows the efficiency curve of Kaplan turbines (the most common type in Columbia River system dams) with highest survival occurring at highest efficiency; turbines with negative pressure in the draft tube have a higher kill rate than those with positive pressure, pointing to the importance of maintaining an optimum tailwater elevation; and larger fish suffer greater mortality than smaller fish. Although early physical model studies could not establish realistic effects of clearances between parts such as runners, wicket gates, and hub, much of the mortality was presumed to occur at those interfaces because of the demonstrated importance of fish size. Recent studies with marked fish in actual turbines (using balloon tags) have confirmed the importance of these interfaces (work underway by Mid-Columbia Public Utility Districts). The studies have also indicated that submerged traveling screens installed in turbine intakes to bypass fish through gatewells instead of allowing them to pass through turbines are themselves a sizable source of biological damage to downstream migrating salmon (Koski et al. 1986; Wik and Barila 1990; Peven 1993; Nestler and Davidson 1995a). Spiral flow and pressure regimes in the draft tube also present concerns. Individual injury mechanisms associated with turbine passage were considered in detail in a Corps of Engineers workshop (USACE 1995).

4.2 Behavior of Salmonids

The early studies of fish mortalities at dams also stimulated studies of the behavior of salmonids. Biologists associated with hydropower facilities sought primarily to find ways to direct juveniles away from intakes. They examined the locations of fish in dam forebays (the water just upstream of a dam) and the relationships between fish passage and the depths of intakes. Natural and artificial cues (lights, bubble curtains, electric fields, and sound) were evaluated as guidance mechanisms. Early studies established the fundamental behavior pattern of juvenile salmonids as being surface-oriented and following flow. No amount of artificial stimulus has been shown to be sufficiently effective in guiding fish movements otherwise to justify full-scale or prototype testing in the field for application at large hydroelectric projects (Ebel 1981; Mighetto and Ebel 1994; OTA 1995). Surface-flow bypasses mentioned above rely on the natural stimuli of surface orientation for effectiveness.

Basic research on behavior of juvenile salmonids was also underway during the same time, although often independent of the applied studies (Hoar 1954; McDonald 1960; Arnold 1974; Thorpe 1982; Fangstam et al. 1993). Descriptions of swimming behavior in water flow, orientation of movements, flow cues to migration, and swimming speeds in different environmental situations occupied the interests of these basic researchers.

Intensive research on salmonids, both basic and applied, has shown several important considerations for understanding fish behavior as it affects entrainment injury and mortality at turbines. These considerations are: orientation with bulk water flow (toward turbines or alternative pathways), surface orientation of salmonid downstream migrants (the most studied) and other orientations of other species, and body orientation in flow that affects the likelihood of striking a turbine blade or other structures. Basic studies of fish behavior, described in the following section, suggest other important considerations, such as buoyancy and stability, obstacle recognition and avoidance, the sensing of acceleration in relation to fish orientation and directed movements, behavior in turbulent flow, and stress responses that may modify normal behavior.

4.2.1 Orientation with Bulk Flow

That downstream-migrating juvenile salmonids or other anadromous species should follow downstream water movement seems axiomatic. However, the degree to which this relationship holds in relation to fish entering turbines or guided to other pathways has been the subject of much study.

Spill is an alternative pathway for water and fish movement that has provided evidence of the complexity of flow-following by juvenile salmonids (Williams et al. in press). Spill refers to the release of water over dam spillways rather than through turbines (Figure 8). On the Columbia River, spillways are not at the surface, generally, but their crests can be as deep as about 50 feet (15 meters) for the typical Tainter gate-equipped spillways. Spill

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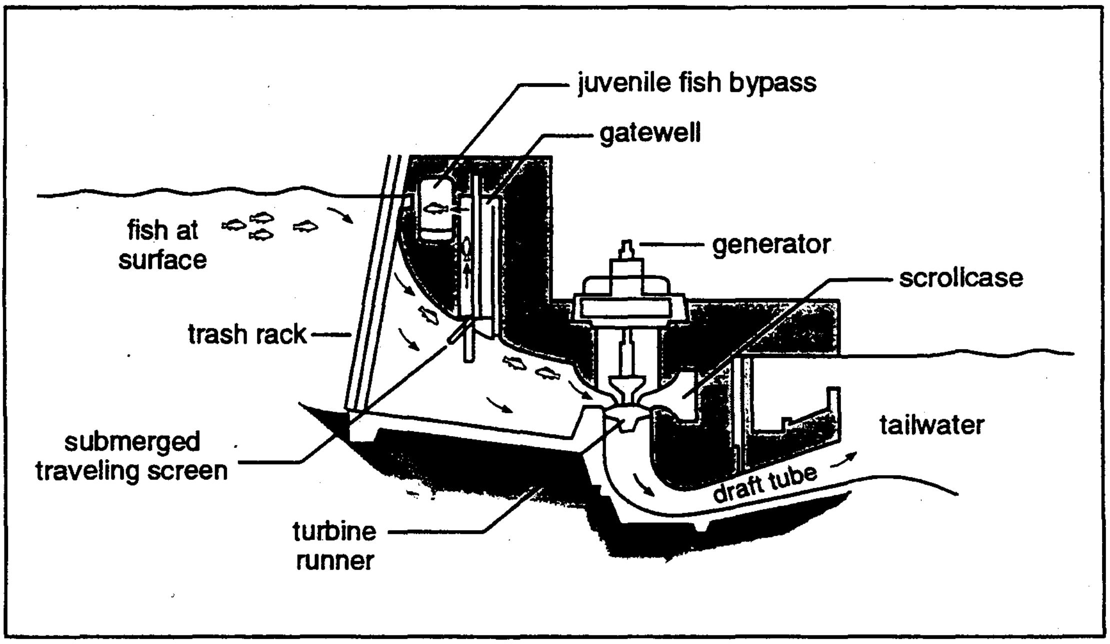

Figure 8. Generalized hydropower facility showing alternative water pathways through powerhouse turbines or spillway. Insets show cross-sections of a typical spillway and a spillway modified as a surface spillway.

volume can be small or large in relation to river discharge and turbine passage, depending on natural river discharge conditions (turbine capacity may be exceeded by total river discharge during high-runoff events) or discretionary and regulated operations that induce spill.

Because spill is recognized as being a more benign means of passing fish than through turbines [0 to 4% mortality (FPC 1994; NMFS 1995), but typically 0 to 2% for standard spill bays], extensive studies were conducted at mid-Columbia public utility district projects to define the relationships between spill volume relative to river flow and the resulting percentages of juvenile fish passed in spill (Biosonics 1983a,b, 1984; Raemhild et al. 1985). Spill volume was varied experimentally from 20 to 85% spill relative to river discharge. Non-linear response curves were found. For example, at Wanapum Dam in the spring of 1983, night-time spill of 20% of the instantaneous flow passed about 45% of the fish, while 50% spill passed 60% of the fish (Biosonics 1983b). In contrast, at Rocky Reach Dam during the spring of 1983, night-time spill amounting to 20% of the river flow was estimated to pass about 16% of the fish, spill of 50% passed about 30% of the fish, and spill of 80% passed about 55% of the fish (Biosonics 1984).

Similarly, studies were conducted at the federal Columbia River basin projects by the National Marine Fisheries Service as an aid to increasing smolt passage rate over spillways (Giorgi and Stevenson 1995). Numerous studies since 1983 at spill percentages between 37 and 66% (Kuehl 1986; Johnson and Wright 1987; Magne et al. 1987a,b; Oullette 1988; McFadden and Hedgepeth 1990) were evaluated by Giorgi and Stevenson (1995). At John Day Dam, spill effectiveness ratios (i.e., the relative number of fish following spill in relation to water flow) was 1.3 in 1987, 1.1 in summer 1988, and 1.4 in 1989. Giorgi and Stevenson (1995) concluded that the scattergram of data from 1993 showed a ratio of essentially 1. Our evaluation of these data suggests that averaging over seasons and a fairly limited range of spill percentages has obscured the underlying curvilinear nature of the response. That is, certain amounts of spill under the right conditions are likely to be more effective in passing fish than is indicated by the sheer bulk of flow. Detailed evaluation of research on spill effectiveness is beyond the scope of this review, but the point can be made that juvenile salmonids will use alternate pathways in lieu of turbine passage and not be governed by just water flow.

Spillway depth appears to influence spill effectiveness in passing fish. Raymond and Sims (1980) suggested that surface spill would be more effective than standard spill. They placed stoplogs in the spillway of John Day Dam to create a surface skimming of water for the spillway and found an enhanced number of juvenile salmon for the amount of water passed (Figure 8, right inset). Willis and Uremovich (1981) and Willis (1982) evaluated the ice and trash sluiceway at Bonneville Dam as a bypass system for juvenile salmonids, and found it passed about 40% of the fish approaching the project when there was no spill. Willis (1982) produced an estimate of spill effectiveness while studying the efficiency of the surface ice and trash sluiceway for passing fish at The Dalles Dam. At spills of about 10 to 60%, he found high fish passage (spill effectiveness) at low spill levels. Also aiding spillway passage is the fact that the spillway at The Dalles is aligned with the natural course of the river whereas the powerhouse is at right angles to river flow. Magne et al. (1987a,b) found that the ice and trash sluiceway at the second powerhouse of Bonneville Dam passed an estimated 81% of smolts passing the powerhouse in daytime and 30% at night. The efficiency of surface sluiceways in diverting fish from turbine intakes was generally in the neighborhood of 20 to 40% (Williams et al. in press). Success with surface spill and surface flow bypass systems at Wells Dam (89%; Skalski 1993) provide the rationale for a new generation of juvenile salmon bypass systems using surface flows (Johnson et al. 1992; Skalski et al. in press).

We conclude that studies with spill in conventional Columbia River spillways affirm the basic flow-following response of juvenile salmonids. However, for any specific spillway or set of spillways at a dam, the particular physical configuration will affect the percentage of fish that follow a water mass. There also tends to be a curvilinear response at any particular site of spill effectiveness in passing fish at different flows. The converse of this is also true, that is, there will be a curvilinear response for the percentage of fish that enter the turbine intakes when spill is occurring. A major factor affecting whether fish follow bulk water flow is the depth of withdrawal, with surface water having a greater likelihood of carrying fish than deep water, as we discuss below.

4.2.2 Surface orientation

There is a preponderance of evidence that juvenile salmon migrating downstream are oriented to the upper portion of the water column. Giorgi and Stevenson (1995) have reviewed much of the evidence, which includes numerous depth ranges and locations. The highly applied research on spill effectiveness at certain Columbia River dams, noted above, has reinforced this generalization. Entry into deep turbine intakes is thus a passage of last resort, rather than a preferred mode of migration.

Ice and trash sluiceways, located at the surfaces of dams, were studied in more detail recently. At Rock Island Dam, spill that was split equally between deep and shallow spill yielded 87% of the fish passing in shallow spill (Ransom et al. 1988). At Wanapum Dam, 4% of the total fish passing the dam passed through the sluiceway in 0.5% of the river discharge on a 24-hour basis (Ransom and Malone 1990). At Priest Rapids Dam in spring, a sluiceway that passed only 1.3% of the river flow passed 3% of the fish (McFadden et al. 1992). In summer, it passed 4% of the fish in 2% of the water. Spill in the sluiceway was judged to be twice as effective as spill in the typical, deeper spillway.

Studies at several dams have shown that juvenile salmon do not generally descend to significant depths unless no alternative is presented (Wagner and Ingram 1973; Dunn 1978). Field studies were reviewed by Eicher (1988). For example, in the forebay of Lower Granite Dam (Snake River), 92% of the smolts were found to be in the upper 36 feet of the water column.

Further evidence of surface orientation in the vicinity of turbine intakes comes from the fact that smolts are observed to accumulate in gatewells of unscreened turbine intakes (Long 1968; Long et al. 1970). When drawn by currents to intake depths, the fish orient to the ceilings of the intakes and seek openings (gatewells) to return to the surface.

Early studies of fish distribution in turbine intakes (e.g., Long 1968; Long et al. 1970) were conducted mostly with fyke nets suspended in the turbine intakes, which may affect fish distribution. Video imaging has indicated that fyke nets suspended in turbine intakes have a large, significant effect on almost all fish behavioral variables and some hydraulic variables (Nestler and Davidson 1995a). Thus, data obtained with fyke nets or at screens when fyke nets are operated may not represent both fish-behavioral and hydraulic features of an unobstructed intake. However, Raemhild et al. (1985) used hydroacoustic methods and found about 80% of the emigrating salmon smolts entered the turbine intake of Rocky Reach Dam on the Columbia River within 6.1 m (20 feet) of the intake ceiling, with the remainder passing in the lower 9 m (fish were somewhat less clumped near the ceiling at night, suggesting a partial breakdown of the surface orientation tendency at night).

Numerous hydroacoustics studies at each of the five mid-Columbia projects showed that smolts were concentrated in the upper portion of the water column, generally the upper one-third (several Biosonics reports). For example, Ransom et al. (1988) found that fish approaching Rock Island Dam were surface oriented. These data sets and reports should be analyzed further for information specific for fish species, life stages, and turbine intake arrangements.

High fish abundance near ceilings of intakes is the basis for current juvenile fish bypasses at most Columbia River basin dams, which use traveling screens extending from the bottoms of gatewells and into the turbine intake to enhance the numbers of fish that find the gatewell (Mighetto and Ebel 1994) (Figure 9). Fish are not all at the entrance ceiling, however, but extend into the center of flow. This is evident in the fact that fish guidance to gatewells by intake screens has been improved by extending the initial lengths further into the turbine intakes (Gessel et al. 1995). Extended length screens, still occupying only the upper portion of an intake, have been able to capture near or over 80% of the yearling chinook salmon migrants entrained in the turbine intake at McNary Dam (McComas et al. 1994) and Little Goose Dam (Gessel et al. 1995). For steelhead*Oncorhynchus mykiss*, fish guidance efficiency has exceeded 90%. The screens have altered local fish distribution, however, causing more water (and presumably fish) to flow in the lower portions of the intakes (Turner et al. 1993).

Despite vertical differences in fish distribution, juvenile salmonids often appear to be equally distributed horizontally within turbine intakes, as determined by studies at several Columbia River basin projects between 1977 and 1982 (Gessel et al. 1991). Johnson (1996), however, found statistically significant differences in horizontal distribution at Lower Granite Dam on the Snake River. It is likely that specific geometries of turbine intakes and screens, which differ among projects, can be used to estimate the percentages of fish entrained in different portions of the cross sections of the intakes, although such a complete analysis has not been done with these data.

Radiotelemetry studies of salmon smolts as they encounter dams in their downstream migration show fish near the surface and unable to orient to deep currents that would take them to the deep turbine intakes. The general pattern is for migrations of these fish to be delayed at the forebay (Giorgi et al. 1986, 1988a,b; Snelling and Schreck 1995). The transmitter-equipped juvenile salmonids move laterally back and forth along the dam or just upstream of it, apparently searching for a surface outlet. When none is found, the fish

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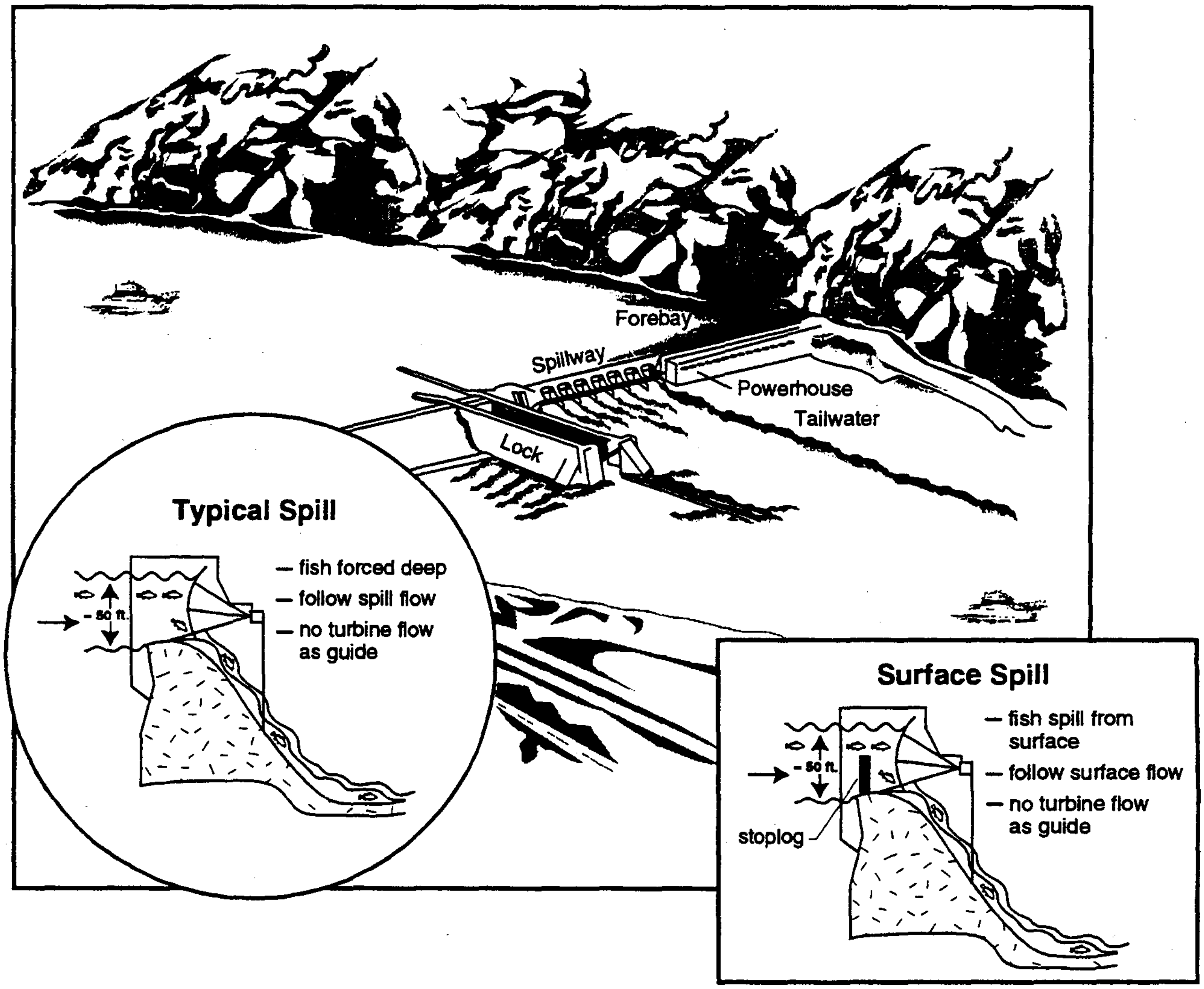

Figure 9. Generalized cross section of a Columbia River basin hydropower powerhouse, showing distribution of downstream-migrating juvenile salmon and fish-passage devices.

will descend (most often at night) to pass the dam through the turbines or turbine gatewell bypasses.

Several studies have suggested an increase in buoyancy with advancing stage of smoltification. Smoltification is the set of progressive physiological and behavioral modifications that occur in juvenile salmonids as they change from the parr stage (freshwater resident) to the smolt stage (migratory and in preparation physiologically for transition from fresh to salt water). Increased buoyancy is believed to assist a fish in attaching to currents for downstream migration with minimal energy expenditure. This feature would also tend to increase their use of surface layers in contrast to deep waters.

We conclude that the basic surface orientation of migrating juvenile salmonids has been abundantly demonstrated, although precise depth ranges vary locally. Entry to turbines at great depths is a last resort for continuing their migration. If an alternative at shallower depths is available, they will preferentially take it. Once in a turbine intake, fish orient to the upper portion of the water mass, often passing along the ceiling where traveling screens have been effective in removing them from the flow. Thus, their entry to the turbine itself will not be uniform across the water mass entering the turbine. Further report evaluation and data analyses will be needed to specify this distribution in a mathematically rigorous way for species, sizes, and intake geometries. Maximum rigor will be attained when each hydroelectric project is evaluated individually.

4.2.3 Body Orientation in Flow

Physical damages to fish in turbine systems (intake, turbine, draft tube) may depend on the fish's orientation as they enter. The simplest concept is of a fish moving passively with the water, and with no particular orientation. This "inanimate, neutrally buoyant object" model was dispelled early in studies of salmon migration in rivers. Ratter (1902) concluded that juvenile salmon in the Sacramento River, California, drifted downstream tail first, keeping their heads upstream to promote water passing through gills and for catching food. Smith (1982) used experimental observations of coho salmon *Oncorhynchus kisutch* to support the idea of fish orienting mostly upstream while drifting seaward. Recent laboratory flume studies by Nelson et al. (1994) have confirmed head-upstream swimming by chinook salmon underyearling migrants. Fish swam upstream at about one body length per second against the current as they either maintained position in the experimental flume or were swept downstream tail-first by higher velocities. Active swimming downstream was observed only in very low velocities. Sockeye salmon *Oncorhynchus nerka* smolts (yearlings or older), on the other hand, showed active downstream swimming in rivers that was not simply a matter of following currents (Brett and MacKinnon 1953; Groot 1965). Active swimming appeared to follow a compass orientation related to river and lake geography. Smolts entering a river from a lake swam actively with the currents (Groot 1982), a pattern that they may follow as they leave a reservoir and enter the flow of a turbine intake. Rainbow trout (steelhead) have been observed with infrared light to swim actively downstream at rates greater than water movement (Northcote 1962).

There have been some direct observations of fish orientation entering turbine intakes. Coho salmon yearling smolts approached an inclined plane screen installed in a penstock mostly oriented head upstream, based on visual observations and videotapes made through a viewing port of fish released into the penstock (Winchell et al. 1991). Video imaging of natural-run salmonid smolts (unidentified) in spring at McNary Dam showed an average of only about 3% (standard deviation 7.9) oriented head downstream with the current (range 0% to 33%) as they approached submerged traveling screens in turbine intakes with different camera locations and screen types (Nestler and Davidson 1995a). In summer, the percentage averaged 17.5 (SD 18; range 0-100), a significant seasonal difference. Few fish (about 8%) exhibited no control over their movements. At The Dalles Dam, however, Nestler and Davidson (1995b) found 42% of the salmonid smolts oriented head downstream as they approached the submerged traveling screens. Johnson (1996) indicated that hydroacoustic studies at Lower Granite Dam showed most smolts oriented head upstream and toward the surface. In contrast, Montén (1985) reported studies in which cuts on fish passing through turbines were tallied with the conclusion that the fish were oriented randomly as they approached the blades.

Flow instabilities in the intakes may greatly affect orientation of fish as they enter turbines. Nestler and Davidson (1995a,b) reported large flow instabilities in turbine intakes at McNary and The Dalles dams. Plotsky and Johnson (1996) reported transient vortices in intakes at Bonneville Dam that made hydroacoustic sampling impossible despite the desirability of testing the assumption that the vortices carry large numbers of smolts past screens and into turbines. The prevalence and effects of such vortices need to be established before generalizations can be made about fish orientations in turbine intakes.

Many basic studies have been conducted on Atlantic salmon *Salmo salar* in Europe and the northeast United States, with the prevailing view that migration is passive (Fried et al. 1978; McCleave 1978; Thorpe and Morgan 1978; Thorpe et al. 1981). Thorpe (1982) reasoned that there should be little advantage in a migrant expending scarce energy reserves by actively swimming. However, Arnold (1974) pointed out that migration is a complex response to currents, with a mix of passive and oriented movements. Recent experiments with Atlantic salmon have shown that active swimming is used for a considerable portion of the distance traveled even though it is a small proportion of the time (Fangstam et al. 1993). Thus, young Atlantic salmon may swim headed downstream while moving, but rest in backwaters for much of the outmigration period (Williams et al. in press).

Travel time studies in the Columbia River system, made possible with PIT-tag technology (Prentice et al. 1993), have shown marked species differences (Berggren and Filardo 1993), which may relate to migratory orientation and behavior (Williams et al. in press). Steelhead, in particular, have shown tendencies to migrate faster than the average velocity of the watermass in which they move, suggesting active downstream swimming at least part of the time.

Radiotelemetry studies by Schreck et al. (1995) in the Willamette River, Oregon showed yearling chinook salmon would exhibit directed downstream swimming in the faster reaches of the river whereas they moved more slowly (passively?) in slow reaches. When groups of fish were tagged and followed together, individuals in the "pack" exhibited numerous changes in relative longitudinal position, suggesting that individual fish migrate in spurts with periods of slower movement or rest.

Accelerating flow seems to influence the speed, and likely the orientation, of migration. Mundy et al. (1995 draft) related episodic high movement rates of yearling chinook salmon past the Prosser Dam on the Yakima River, Washington, to concurrent accelerations in flows. When the seasonal data on daily flows and daily fish movements were compared for the specific dates of fish presence, the fish appeared to be moving when flows increased. Achord et al. (1995) also noted a historical pattern of increased migration of chinook salmon yearlings on rising water flow. Our review of PIT-tag data from other studies (FPC 1994; Buettner and Brimmer 1995) suggests that this phenomenon is common.

If accelerating flow stimulates more active movement generally, it may signal a transition to head-first, downstream swimming. This hypothesis has been raised as potentially important for juvenile salmon migrations in the Columbia River basin (Williams et al. in press) but has not been tested. This phenomenon may also occur in the intakes of turbines, where the relatively quiescent waters of the reservoir are replaced by accelerating velocities in the intake and scroll case. Generally, the velocity at the upper end of a typical turbine intake on the Columbia and Snake rivers is 3-4 ft/s (about 1 m/s)(USACE 1995). The velocity gradually increases to 6-9 ft/s (about 2 m/s) as it enters the scroll case. The flow then rapidly accelerates to >20 ft/s (>3.5 m/s) through the wicket gates, and then to 50-60 ft/s (15-18 m/s) as it passes the blades and hub. The water then decelerates rapidly to 12-20 ft/s (3.6-6 m/s) as it turns to enter the draft tube. It continues to decelerate to 8-12 ft/s (2.4-3.6 m/s) in the tailrace. Because body orientation is likely important for estimating likelihood of strikes in turbines (Turnpenny et al. 1992), a possible relationship between body orientation and flow acceleration in the turbine intake warrants more detailed study. Also, extreme turbulence that may accompany these highest velocities immediately in front of the turbine runner may disorient fish, so that the "usual" behavior could be stymied and fish orientations suddenly become randomized (as suggested by the body-strike results of Montén (1985).

As indicated earlier in this report, variability among individual fish may be important. Although one fish may pass through a turbine aligned with the water flow, another may respond to changing fluid dynamics by altering its orientation. Consequently, estimates of the probabilities of injury (e.g., from blade strikes) may have wide confidence intervals. These confidence intervals may greatly exceed the average gain in survival by structural modifications to the turbine system, and thus be difficult to identify, test, and evaluate.

We conclude from this review that juvenile salmonids entering turbine intakes may be oriented in several ways, depending on the species and the migration tendencies of the fish at the time. The majority of underyearling chinook salmon (the smallest migrants) appear to move in a head-upstream manner. They likely maintain that attitude as they enter turbines. Most yearlings (the larger fish), especially steelhead, appear to swim rapidly, directed downstream in the riverine environment but oriented head upstream near bypass screens in turbine intakes. Yearling chinook salmon may show both types of orientation, but could be oriented head downstream in the accelerating flows of a turbine intake. All of these behaviors may be negated by rapid flows and turbulence at the entrance to turbine runners. Further analysis is needed of hydroacoustic and underwater television data related to submerged traveling screens as indicators of fish orientation as they enter turbines. These technologies need to be applied also at the entrances of turbine runners.

4.3 Behavior of Non-Salmonids

The relationships of the salmonid information to behavior of non-salmonids and resident fishes, including salmonids is problematical. Juvenile salmonids are attempting to move downstream, and passage through turbines is one route. Resident fishes without the migration urge likely are adapted to resist currents and water flow, the agents that would displace them from their normal habitats. However, some non-anadromous fish have extensive migrations within fresh waters that are intercepted by hydropower facilities, and thus some of their entrainment may be analogous to downstream-migrating salmon. Entrainment of non-migratory species is likely accidental and may relate to the degree to which each species uses habitats closest to the turbine intakes (FERC 1988, 1995). Entrainment probability and fish behavior for resident fishes is likely to be highly site-specific, depending on the habitats and species encountered.

The Federal Energy Regulatory Commission has begun to synthesize information obtained in entrainment monitoring studies it has required at small hydropower sites dominated by non-salmonids (FERC 1995). Its staff and contractor, Stone and Webster Environmental Technology and Services, Inc., surveyed limited-distribution reports of 45 studies east of the Mississippi River, predominantly in Michigan and Wisconsin, but also including sites in South Carolina, West Virginia, Ohio, Pennsylvania, and New York. The facilities were mostly small (less than 5 MWe), but some were up to 102 WE in electrical generating capacity. Although the emphasis of the review was on species and numbers of fish entrained and the factors affecting entrainment, some information on fish behavior in intakes was gleaned. Species included gamefishes (e.g., smallmouth bass *Micropterus dolomieui*, and walleye *Stizostedion vitreum*), panfishes (e.g., yellow perch *Perca flavescens*, and black crappie *Pomoxis nigromaculatus*), and forage fish (e.g., alewife *Alosa pseudoharengus*, white sucker *Catostomus commersoni*, gizzard shad *Dorosoma cepedianum*, and threadfin shad *Dorosoma petenense*). Most fish entrained were small, entrainment was episodic (brief periods of large numbers of fish entrained, with long intervals of low entrainment), and there was high variability in diel, seasonal, spatial, and species-specific entrainment rates.

The FERC analysis evaluated the cross-sectional distribution of fish drawn into intakes (FERC 1995). There were no consistent trends in vertical distribution (among sites or species). However, horizontal distribution was generally not uniform. About 80% of the fish occurred near the side walls and about 20% in the centers. Proximity to the shoreline was often a major factor determining relatively high fish occurrence, both among multiple intake bays and for locations in a single intake bay. This suggests an important tendency of entrained non-salmonids to follow shorelines in their normal behavior, which affects vulnerability to intakes and suggests proximity to sidewalls as consistent routes of passage through turbines.

The episodic nature of entrainment relates to both seasonality of life cycles and to seasonal cold stress (FERC 1995). Juvenile fish appear to be especially vulnerable, perhaps because of abundance and tendency to disperse, and perhaps because of poorly developed sensory abilities. They may also lack the strength or stamina to escape flows. Entrainment episodes often involve schools of juveniles, especially shad species. Even without cold stress (see below), juvenile non-salmonids may have poor orientation capabilities in currents that may suggest movement through turbines without directed avoidance behavior.

Fish guidance experiments with louvers indicate behavioral responsiveness of larger sizes of non-salmonids but poor guidance by small sizes (EPRI 1986). There was generally good guidance by fish larger than 1.2 to 2.4 in (3-6 cm) long. Guidance decreased rapidly for striped bass *Morone saxatilis* less than 1.2 in (3 cm) and for white catfish *Ameiurus catus* less than 3.6 in (9 cm). High proportions (>75%) of anadromous American shad *Alosa sapidissima* and blueback herring *Alosa aestivalis* were guided by louvers in the autumn at Holyoke Dam on the Connecticut River (Harza and RMC 1993).

Entrainment of resident fishes in bulb turbines has received more attention than other turbine types, largely because of the use of this technology in inland rivers such as the Ohio River (FERC 1988) and in tidal hydropower (Dadswell et al. 1986). Regardless of turbine type, the behavior patterns that lead to initial entrainment are germane. The results of studies of turbine-induced fish mortality in non-salmonid waters are highly varied. Spectacular damages were suffered by high numbers of large and important fish species (American shad, striped bass) in the Bay of Fundy (Dadswell et al. 1986). Other facilities, such as the Racine and Greenup/Vanceburg projects on the Ohio River have had few occurrences of entrainment damages (WAPORA, Inc. 1987; Olson et al. 1987; Olson and Kuehl 1988). Entrainment injury and mortality rates in tidal waters are clearly affected by fish size (larger ones are more susceptible to damage), species (clupeid fishes of the herring family are most sensitive), and schooling behavior (herrings that moved in and out of the tidal embayment on a daily cycle were badly affected). In freshwaters of the Ohio River, few game fish are entrained, but there were many schooling gizzard shad (clupeids) and freshwater drum *Aplodinotus grunniens*. In general, the few larger gamefish that were entrained in Ohio River facilities suffered high mortality.

The susceptibility of fishes to entrainment because of biological behavior varies seasonally and among species and life stages. Holland et al. (1984) summarized existing information on adult fish movements through dams on the upper Mississippi River. These data were further analyzed by Normandeau Associates, Inc. (1986). The movements of most gamefishes do not take them through dams and most interpool movement occurs in high flows when considerable water is spilled. Studies at the Racine project (Ohio River) showed gamefishes were entrained only occasionally. Early life stages (eggs, larvae, and pelagic juveniles) of several species are essentially planktonic and they drift with water during the spring and summer spawning periods. Water bodies with large numbers of species with these life-history patterns can be expected to show large numbers of fish entrained in turbines. Survival of these early life stages is high, however. As these fish grow, schools of juveniles occupying the open waters (especially gizzard shad and freshwater drum) remain susceptible, and entrainment damages to these ages are higher as their size increases (WAPORA, Inc. 1987). Gizzard shad schools usually occur in the top 10 feet (3 m).

There is some indication that resident fishes are more vulnerable in autumn and winter than in warm seasons (FERC 1995). Extreme cold or sudden temperature declines can make fishes comatose and they will drift into intakes. This is a common problem at steam electric generating stations, which entrain large numbers of threadfin shad on intakes in cold winters (McLean et al. 1980). A high percentage of alewife entrained annually at one hydropower facility occurred during one 1-week period in early January, and this was accompanied by a high entrainment of walleye (due either to vulnerability while feeding on moribund alewife or because of their own debility). FERC (1995) suggested that there is sufficient information about the occurrence of entrainment during periods of cold stress that these episodes could be predicted from weather data. Comatose or moribund fish are unlikely to exhibit any avoidance reactions or controlled body orientation that would cause them to differ from passive particles in transit through turbines.

We conclude that schooling behavior of juvenile fishes in habitats near turbine intakes is a major factor in susceptibility of non-salmonid and resident species of fish to entrainment. They probably exhibit little avoidance or orientation behavior once entrained except for proximity to walls of the turbine intakes that reflects nearby habitat and shoreline-oriented movements. Cold water temperatures sufficient to make fish comatose will increase vulnerability to being entrained and result in poor or no orientation and avoidance behavior during transit through turbines.

4.4 Basic Studies of Fish Behavior

Few non-salmonids have been studied in actual turbine intakes the way salmonids have been. Therefore, our discussion centers around basic features of the morphology, physiology and behavior of fishes as a group that can affect their responses to being passed through turbine systems. We emphasize features that could affect computational fluid dynamic modeling of fish movement, particularly deviations from movements projected for a neutrally buoyant particle.

4.4.1 Buoyancy and Stability

The risk of mechanical injury to fish by being struck by a turbine runner blade appears to be related to the zone of fish passage through the turbine in relation to the hub (USACE 1995). This location is, in turn, related to the position of the fish in the water column of the turbine intake, which depends partly on the buoyancy of the fish. The Corps of Engineers workshop (USACE 1995) considered the most critical uncertainties regarding runner-blade strikes to be whether or not fish remain neutrally buoyant within the turbine, whether buoyancy is species- and size-specific, and whether it is affected by pressure changes in the intake. Computational fluid dynamics modeling should take any differences from neutral buoyancy into account.

Fish are denser than the water they live in, and unless they have some mechanism for compensating for this difference, they sink (Alexander 1993). Buoyancy adaptations include hydrodynamic forces during swimming, fats and oils in specific tissues such as the liver, and gas-filled swim bladders. Densities of most fishes are reduced to within 1% of the surrounding water by these adaptations. Because tissues of different density are distributed nonuniformly in the body, the center of gravity (for sinking) is usually different from the center of buoyancy. The centers of buoyancy of fishes with swim bladders are generally slightly below the center of gravity, making the equilibrium of fish unstable; this is why dead and comatose fish float upside down. These points are important for considering whether fish can be modeled as neutrally buoyant particles for evaluations of turbine passage.

The most commonly found fish species have swim bladders, which are gas-filled floats that can match the densities of fishes to that of the water in which they swim to within about 0.5%. Use of low-density gas is an exceedingly efficient way of balancing density of bones and other dense tissues (Alexander 1993). Gas bladders occupy only about 7% of the volume of a freshwater fish and 5% of a marine one.

Although efficient for equilibrating buoyancy at a constant depth, gas bladders offer severe disadvantages for rapid changes in depth, such as occur when fish are drawn into deep turbine intakes from surface waters. Swim bladders expand when fish swim nearer the surface, where the pressure is less, and are compressed when it swims deeper. In accord with Boyle's Law, a swim bladder at a depth of 10 m is compressed to half its volume at the surface and it shrinks by half again at 20 m. Thus, the density of a fish with a swim bladder matches that of the water only at one depth, unless the quantity of gas in it is adjusted. As a fish rises a little, its density will decrease, making it tend to rise further. Conversely, if it sinks a little its density will increase and it will sink more. Most fish hover (use their pectoral fins in a back and forth motion) to make adjustments to differences in density within a fairly narrow bound. Jones (1952) found perch could not hover by fin movements alone at pressures more than 16% beyond the pressure to which they were adapted. Active swimming to desired depths, which many fish do, is required beyond this point.

Slow depth adjustment of buoyancy is made possible by gas secretion and absorption across specialized tissues of the bladder membranes, a process that has been studied for over 100 years. This equilibration process generally occurs at a rate equivalent to a few meters of depth per hour (the fastest appears to be about 2.5 m/h; Alexander 1993). Therefore, a fish with a gas bladder that is drawn rapidly (within seconds) into a turbine intake will be increasingly more dense than a neutrally buoyant particle as the depth and pressure increase. It will become more buoyant again as it passes through a draft tube and enters the tailrace at near-surface pressures. Fish drawn from mid-depths of the forebay and released at essentially surface pressures in the tailrace may be over-buoyant and float to the surface. These changes can be calculated based on Boyle's Law and a knowledge of the depth from which the fish originated.

A complicating factor in gas-bladder-induced buoyancy is the ability of some fish to evacuate gas from their bladders, usually by way of a vent to the mouth area. Salmonids have such a vent (are physostomous); the freshwater basses, for example, do not (are physoclistous). When external pressure drops rapidly, as it does in the exit of a turbine, gas in the expanding swim bladder may be released, allowing the fish to become more dense rapidly. Although such gas evacuation would not affect buoyancy in the turbine intake or turbine itself, it would affect buoyancy in the draft tube and tailwater. In principle, gas evacuation and rapid change in buoyancy would be most likely to occur in fish that have been acclimated to high pressures of deep water in the forebay of a dam. Computational fluid dynamics models that seek to understand the delivery of fish downstream of a dam may need to take such changes into account. There is no direct information available on whether such gas-bladder evacuation actually occurs in turbine passage, the pressure changes that would induce it, and the rapidity with which it would occur. Harvey (1963) suggests that the pneumatic duct connecting the bladder with the outside becomes constricted under rapid pressure drops and gas is not expelled. Indirect evidence from generally better survival of physostomous fishes in turbine passage suggests that it may be occurring.

Buoyancy mechanisms other than swim bladders may be important for fish in some turbine intakes. Dense fish without swim bladders generate upward hydrodynamic forces as they swim equal to the difference between weight and buoyancy upthrust. Sharks and sturgeons accomplish this with large pectoral fins that cannot be folded and asymmetrical tails that generate an upward thrust. Tunas have symmetrical tails but a prominent caudal peduncle and the same stout pectoral fins that provide upthrust (Magnuson 1978). Paddlefish *Polyodon spathula* have an added planing surface in the form of a large snout (which compensates for the high drag of a large mouth gape used for plankton feeding).

The freshwater pelagic (open water column) members of swim-bladderless fish, especially paddlefish (sturgeons are largely bottom dwellers), are at special risk of being entrained in turbine intakes because they must cruise constantly in the water column to keep from sinking. Once they have lost hydrodynamic control (as they probably do in a turbine intake where water may drag them along), they will sink. The point of loss of hydrodynamic control may be calculated. Like airplanes, fishes that use fins as fixed hydrofoils have a minimum speed (the stalling speed) below which the fins cannot generate the required lift. This speed can be calculated for species and individuals of different sizes from standard equations, and has been for selected examples (Alexander 1990 and textbooks on aerodynamics). Thus, the degree of hydrodynamic control, the location in the intake where this control is lost, and therefore the tendency for sinking (and resultant trajectory through a turbine) can, in principle, be estimated for these entrained fish. The location in the draft tube where there is sufficiently low turbulence for re-establishment of hydrodynamic control (and near-neutral buoyancy) may also be important. Although such an analysis need not be carried out for all fishes, the approach may aid in resolving site- and species-specific problems.

Fish that compensate for their otherwise high density by using fats, oils, or (in some cases) especially watery tissues and poorly ossified bones are of little concern for hydropower turbines. Their circum-neutral buoyancy will remain constant through turbine passage, and they can be modeled as such. Most such fishes are marine (Alexander 1993), where they may be of concern only for tidal hydropower. Sharks have especially large and oily livers as well as using hydrofoils for depth control.

We conclude from this review that models of fish trajectories cannot assume neutral buoyancy throughout the time a fish passes through a turbine. Fish without swim bladders that depend on activity to maintain themselves will likely lose control and be negatively buoyant. With numerical values depending on initial depth in the forebay, fish with swim bladders will become progressively more dense as they descend to the turbines and then positively buoyant as they are discharged to the draft tube and dam tailwater. Whether these differences will be significant for modifying fish trajectories should be established from the computational fluid dynamics modeling studies using known pressures in each part of the turbine and Boyle's Law acting on fish with gas bladders.

4.4.2 Obstacle Recognition and Avoidance

Turbines, especially wicket gates and rotating blades, are physical obstacles in the path of a moving fish. Turbine housings are solid walls, although Bell (1981) notes that there is a hydraulic "cushion" of water moving laterally after impact.. Both physical contact and shear at the surfaces of these structures can be damaging to fish. The degree to which fish are able to detect and avoid the physical obstacle in the brief time frame of passage will affect the likelihood of damage and the ability of computational fluid dynamics models to predict travel pathways (trajectories). It is also possible that changes in water flow patterns in the intakes will be perceived as an "obstacle" and the fish may initiate avoidance before the physical structure of the turbine itself is reached.

An avoidance reaction can possibly remove a fish from danger. On the other hand, a rapid change in orientation (e.g., the angle of approach to a turbine blade) can influence the damage inflicted, perhaps detrimentally. The Turbine Passage Survival Workshop (USACE 1995) identified an understanding of how fish detect velocity changes and subsequently control their vertical movements as a critical uncertainty. Aside from vision, which is likely unavailable in a dark turbine intake, fish sense obstacles through the lateral line system.

The lateral line system in fishes is a sensory pathway for detection of fluid movement for which humans have no counterpart (Bleckman 1986; Popper and Platt 1993). A system of tubes beneath the skin of the lateral musculature and head is connected to the outside water and contains sensory cells for the detection of motion in the enclosed fluid. Water displacements and pressure waves that are formed by any pulsating, vibrating, or moving object are detected by the lateral line. The lateral line is especially adapted to detecting the low-frequency pressure waves that may differ in timing from one end of a fish to the other (more typical "sound" that affects the whole body simultaneously is detected by the ear). The ear and lateral line form a continuity of perception for a broad spectrum of frequencies and forms of pressure waves. The lateral line is the organ fish use for the identification and localization of stationary and moving objects, in conjunction with or in lieu of (in darkness) sight.

Because any object moving through water (or water passing an object) creates a set of pressure waves, the aquatic environment is a collage of waves. A moving fish creates waves that are reflected from other objects and perceived by the fish's lateral line. The fish thus recognizes that an object is present and apparently develops an understanding of the shape, position, and motion of an object encountered repeatedly (for blinded fish can be trained to recognize specific object stimuli such as different-sized glass disks as cues for punishment or reward). Other moving objects (or stationary objects in moving water) create their own waves, which are received by a fish's lateral line system. The lateral line has been documented to be an important sensory component of prey recognition, feeding, predator recognition, predator avoidance, avoidance of physical structures, and shoaling (schooling) behavior. It is the most likely sensory system for fish to use in identifying and reacting to structural features of a turbine system. It now seems clear that the lateral line is primarily used in hydrodynamic interactions at very short distances, on the order of the body length of the receiver (Kalmijn 1989). The system responds to a range of about 10 to 200 Hz.

Turbine blades and wicket gates would perhaps be most analogous to a predator, for which the lateral line system of the prey gives warning of imminent capture, allowing quick movement sufficient to move the target safely away from the predator. The ability of a fish to detect and avoid obstacles in a turbine sufficient to affect likelihood and geometry of strikes is questionable, considering the short distance for reception and the rapid rate at which fish encounter the obstacles (because of both rapid water flow and movement of the runner blades). An ability to make avoidance reactions also would depend on the degree of hydrodynamic control being maintained by the fish at the moment. On the other hand, any rapid detection and avoidance responses on the part of a fish as it encounters a wicket gate or runner blade could alter the ability of the fish to pass around the object, thus causing it to deviate from the theoretical flow lines expected in CFD models. A more blunt object would project more prominent waves, thus enhancing a fish's ability to detect it. Turnpenny et al. (1992) showed less damage by the blunt faces of the thicker portions of turbine blades near the hub than the slimmer blade tips. Establishment of sensation as a cause for fewer strikes must await studies of live fish to compare with those of freshly killed fish, as used by Turnpenny et al. (1992).

We conclude that the lateral line has great sensitivity over relatively short distances; it can induce burst swimming and changes in fish orientation, and these may affect fish orientation in a fluid dynamics model. These effects may occur near the walls of turbine intakes at a distance from the runner. It is unlikely, however, that lateral-line sensing of obstacles in turbines themselves occurs fast enough to affect fish orientation markedly in the very rapid passage times.

4.4.3 Sensing acceleration

Fish moving with a mass of water and out of sight and lateral-line sensing of walls or other boundaries might be viewed as unaware of their net displacement. In a perfectly steady flow this may be true. However, if the water accelerates or slows, the fish is able to sense this change in rate of movement (linear acceleration) by means of the inner ear. Sensing acceleration or deceleration may be important as cues for body orientation and mode of swimming, as noted above (where accelerating flows tended to cause more rapid fish passage and possibly downstream swimming). Increased turbulence would be recognizable as angular accelerations, and perhaps stimulate changes in body orientation, also.

The inner ear of vertebrates is involved with both auditory (hearing) and postural (body orientation) senses (Popper and Platt 1993). Although the semicircular canals and the utricle of the inner ears have been believed to be the structures that sense acceleration, the current view is that all of the component organs of the inner ear provide major inputs to both postural control and hearing. Portions (the semicircular canal organs) tend to respond to angular accelerations whereas other portions (the otolithic organs and the non-otolithic macula neglecta) respond to linear accelerations. Otolith organs are liquid-filled pouches that each contain a dense mass of some crystalline forms of calcium minerals (called otoliths--earstones--, which are commonly used in aging fishes because the minerals are deposited concentrically as fish age). Sensory cells detect movements of the otoliths within the pouches in response to changes in acceleration. Much is known about the anatomy and function of these organs (Popper and Platt 1993), which undoubtedly come into play as fish follow their trajectory through a turbine system.

There is little firm evidence on which to form conclusions about the effects of sensing accelerations, other than what was discussed earlier. The degree to which fish change orientation in response to changes in linear and angular acceleration in a turbine system is not known but could be of great significance to computational fluid dynamics models of fish trajectories.

4.4.4 Behavior in Turbulent Flow

Once a fish has left the turbine and initial part of a draft tube, it enters a zone of high turbulence in the lower draft tube and tailrace. If the goal of a fish-friendly turbine system is to deliver fish into a normal migration environment downstream of the dam, then reorientation of fish in the turbulent tailrace will be necessary. Disorientation and physiological stress there may be a major cause of mortality through predation (Long et al. 1968). Few studies have been conducted of fish behavior in high turbulence. Passive movement of positively or negatively buoyant objects may be the best model (see above), especially in the most turbulent zones. However, Shtaf et al. (1983) attempted to define experimentally the influence of lesser amounts of turbulence on fish behavior and showed several types of behavioral responses in the roach *Rutilus rutilus* and minnow *Phoxinus phoxinus*. The existing experimental work is embryonic, at best, and it is now of little practical use for turbine system designs or modeling. Nonetheless, the work suggests experimental techniques and approaches that may be usefully explored to establish whether fish in the turbulent draft tube and tailrace modify their trajectories based on turbulence.

Work is currently underway to develop theoretical concepts of fish migration in rivers that take into account a probable use of turbulent flows to assist downstream movement beyond passive attachment to bulk flow (Williams et al. in press). The notion is that migrating juvenile salmon probably use features of unsteady flow in rivers such as turbulent bursts, vortices, and waves to find regions of relatively high velocity to speed their migration. This use would imply a sensory ability to detect these features and behavior to establish a beneficial orientation in them. Turbulence in a draft tube and tailrace is likely much greater than in a natural river except at waterfalls. Because the outlet of a turbine imparts a “whirl” component (USACE 1995), fish may sense this whirl as a natural vortex and orient to it in ways that move them rapidly toward the periphery. The periphery, however, is a draft tube wall, which may impart abrasions on impact.

We conclude that the use of unsteady fluid flow in migrations and turbine intakes is speculative at this point, but may lead to focused research of value to design of turbine systems that better match the natural behavior of juvenile salmonids.

4.4.5 Stress

Stressed fish may not behave normally, which could affect their performance in turbine systems. In the context of turbine-effect studies, stress has been used in a general sense to describe disorientation, loss of equilibrium, stunning, abnormal swimming behavior and energy depletion, usually as a *result* of turbine passage (USACE 1995). In other contexts, stress is related to specific physiological changes in enzyme systems and measures of blood chemistry (Adams 1990). Examples of sensitive indicators of stress are the capacities of fish to osmoregulate, mount an immune response, resist disease, respond physiologically to another stressful factor, swim, avoid predators, and learn (Schreck 1990). Numerous stress-induced physiological events alter the capacity of fish to perform various physiological and behavioral operations or functions. The degree of pre-existing stress in fish that enter a turbine intake may alter many of the behavioral features discussed in the preceding sections.

Juvenile salmonids that pass through turbines in the Columbia River, for example, are often under some degree of stress (NRC 1996; Williams et al. in press). For example, gas bubble trauma affects juvenile migrants at times when large amounts of water are spilled at dams and atmospheric gases become supersaturated (Bouck 1980; Weitkamp and Katz 1980). Bubble formation in tissues likely affects buoyancy to some degree and also the changes in buoyancy that might be expected during turbine passage. Various infectious diseases are present in migrating salmonids. High water temperatures in reservoirs result in migrants being exposed to temperatures considerably above their physiological optima and often close to lethal levels. Recovery from the physical trauma of turbine passage at upstream dams is not likely to be complete when fish reach the next dam in the series of eight from the middle Snake River to the mouth of the Columbia River.

The episodic occurrence and seasonal timing of entrainment of many non-salmonids in turbines leads to the conclusion that cold stress is a significant factor, as discussed above. A comatose or moribund fish is unlikely to exhibit avoidance or orientation behavior in a turbine intake that would affect turbine-induced mortalities. Whether cold stress makes fish more or less vulnerable to physical damages from turbine passage is not known.

The importance of preexisting stress levels for fish performance in turbine passage (especially as it affects trajectories) is not well known (Schreck et al. 1984; Bjornn 1992), except for strong inferences about effects of cold stress. Attempts to relate trajectories and injuries to most preexisting stresses have generally been inconclusive in numerous hydroacoustic studies in the mid-Columbia River (several studies by Parametrix, Inc. for the Grant County Public Utility District). Suggestions have been made, however, that testing of fish behavior in turbines should include background information on preexisting stress levels, and perhaps experiments should use fish in both test and control lots that have been given known amounts of prior stress (USACE 1995). We agree with these suggestions, for it is important for modeling of fish trajectories to know whether the behaviors modeled and responses seen are representative or skewed by virtue of a preexisting stress.

4.5 Measurement Concerns

Although it is desirable to have more accurate information on fish behavior and orientation in turbine intakes, especially as fish approach the turbine runner, there are important limitations for making observations. Realistic expectations of further research are necessary.

Direct observation in physical models is hampered by elements of scale. Although the turbine system can be scaled to a smaller size, the fish cannot. The types of behaviors examined in this report are often not only species-specific, but also size-specific within a species. Use of very small fish (e.g., fry or aquarium species) as surrogates for larger ones compromises the need to observe relevant behavior.

Video observation and recording of fish positions in actual turbine systems seems feasible based on experiences viewing juvenile salmonids at traveling screens at fish bypass systems at Columbia River Basin dams (Nestler and Davidson 1995a,b). The technique has obvious limitations in turbid water, but would be useful in clear-water sites where representative fish species are entrained. However, positioning cameras in the extremely high velocities near the turbine runners without disrupting the fish and water flows that are of interest may prove to be infeasible. Nestler and Davidson (1995a,b) relate placement difficulties with even the slower velocities at the bypass screens.

Hydroacoustics has provided valuable data in turbine intakes at a distance from the runner, but turbine "noise" affects data analysis increasingly as hydrophones are placed near or directed toward the turbine (FWS 1992). The background noise affects the detection of small fish most strongly, and these are the sizes often of concern. Experimentation with different sound frequencies may be necessary before hydroacoustic detection can be used in close proximity to the turbine runners.

Forensic analysis of fish that have been passed through turbines experimentally (subsequent recovery often facilitated by use of balloon tags) may be improved to the point where location and orientation can be inferred more accurately. Balloon tag studies at Rocky Reach Dam on the Columbia River were able to resolve a difference of 1.7% in mortality of smolts passing through turbines with fixed versus variable blades, leading engineers to conclude that the additional injury rate was induced by a small gap between the hub and the blade of the variable pitch turbine (RMC and Skalski 1993). However, many sources of physical damage in turbines result in similar pathologies. The limits of inference may be too severe for meaningful engineering redesign of turbines.

Without implying too much pessimism, we conclude that the practical limits of observation and measurement of fish and flows in the proximity of turbine runners using existing technologies may inhibit development of much information that is germane to developing a more fish-friendly turbine.

4.6 Conclusions and Recommendations

1. Studies with spill in conventional Columbia River spillways affirm the basic flow-following response of juvenile salmonids. There will be a curvilinear response for the percentage of fish entering the turbine intakes when spill is occurring. A major factor affecting whether fish follow bulk water flow is the depth of withdrawal, with surface water having a greater likelihood of carrying fish than deep water.

Recommendation: The first priority for a fish-friendly turbine *system* in migratory salmonid waters should be one that bypasses as many downstream-migrating fish as possible along these fish's natural surface-oriented migration pathway away from deep turbine intakes.

2. The basic surface orientation of migrating juvenile salmonids has been abundantly demonstrated. Once in a turbine intake, fish orient to the upper portion of the watermass, often passing along the ceiling where traveling screens have been somewhat effective in removing them from the flow. Horizontal distribution is more uniform, but probably is affected by vortices and other flow instabilities characteristic of a site. Thus, juvenile salmonid entry to the turbine itself will not be uniform across the cross-section of the watermass entering the turbine.

Recommendation: Further report evaluation and data collection and analyses are needed to specify fish cross-sectional distribution in a mathematically rigorous way for species, sizes, and intake geometries in order to quantitatively specify fish trajectories through turbines.

3. Fish entering turbine intakes may be oriented in several ways, depending on the species and the migration tendencies of the fish at the time. Underyearling chinook salmon (the smallest migrants) appear to move in a head-upstream manner. They likely maintain that attitude as they enter turbines. Yearlings (the larger fish), especially steelhead, appear to swim rapidly, directed downstream. Yearling chinook salmon may show both types of orientation, but could be oriented head downstream in the accelerating flows of a turbine intake.

Recommendation: Further analysis is needed using hydroacoustic and underwater television data, both new and as related to submerged traveling screens, as indicators of species- and size-specific fish orientation as they enter turbines.

4. Schools of juvenile non-salmonid fishes that reside in the open waters of large rivers or tidal estuaries are most vulnerable to entrainment in turbine intakes. Their entrainment is accidental and not related to flow-following behavior. Particularly susceptible freshwater fishes are juvenile gizzard shad and freshwater drum. Few adult gamefishes, which are more oriented to bottoms and shorelines, are vulnerable. Horizontal distribution of entrainment is often not uniform for these species. Susceptible freshwater fishes are generally forage species with high reproductive potential. There has been no special effort to study the orientation of these fishes in turbines.

Recommendation: Considerably more justification would be needed for commitment of major expenses for fish-friendly turbines in freshwaters occupied by non-migratory species.

5. A high percentage of non-salmonid entrainment in hydropower turbines, as in steam electric power station intakes, is of forage species that are made comatose by rapid temperature declines or prolonged cold weather in autumn and winter. Fish in these conditions are not likely to exhibit avoidance or orientation behaviors that would cause them to differ from passive particles during transit through turbines.

Recommendation: Simulation of many non-salmonids as passive objects seems appropriate.

6. Models of fish trajectories cannot assume neutral buoyancy throughout the time a fish passes through a turbine. Fish without swim bladders that depend on activity to maintain themselves will likely lose control and be

negatively buoyant. With numerical values depending on source depth in the forebay, fish with swim bladders will become progressively. more dense as they descend to the turbines (the swim bladder is compressed as water pressures increase) and then positively buoyant as they are discharged to the draft tube and dam tailwater.

> ***Recommendation:*** The significance of differences from neutral buoyancy and of changes in buoyancy during fish trajectories through a turbine should be established from modeling studies of fish with a range of constant and changing densities.

7. Lateral-line sensing of obstacles occurs rapidly and can affect fish orientation . However, it is unclear whether sensations in turbines will affect fish orientation markedly in the very rapid passage times.

> ***Recommendation:*** Further study of reaction times is needed. Models can tentatively assume that orientation of fish as they enter the scroll case will be retained as they transit the turbine itself (or at least that the fish will not be able to control its orientation in a turbulent environment), under the assumption that reaction times are too long for the rapid flow rates.

8. The use of unsteady fluid flow by fish in migrations is speculative at this point, but may lead to focused research of value to the design of turbine systems, especially draft tubes and tailwaters, that better match the natural migratory behavior of juvenile salmonids.

> ***Recommendation:*** Research on the orientation in and use of unsteady flows by migrating juvenile salmonids is needed.

9. The importance of pre-existing stress levels for fish performance (especially as they affect trajectories) in turbine passage is not known. It is important for modeling of fish trajectories to know whether the behaviors modeled and responses seen are representative or skewed by virtue of a pre-existing stress.

> ***Recommendation:*** Testing of fish behavior in turbines should include background information on pre-existing stress levels, and experiments should use fish in both test and control lots that have been given known amounts of prior stress.

10. Practical limits of observation and measurement of fish and flows in the proximity of turbine runners may inhibit development of much information that is germane to developing a more fish-friendly turbine.

> ***Recommendation:*** Innovative means for obtaining information on fish behavior near turbine runners should be pursued, but there should be realistic expectations about the feasibility of this research.

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Conservation status and population trends of the American eel in Canada

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* This series documents the scientific basis for the evaluation of fisheries resources in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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1. INTRODUCTION

The American eel is the West Atlantic representative of the worldwide genus *Anguilla*, whose members spawn in ocean waters, migrate to coastal and inland continental waters to grow, and then return to ocean spawning areas to reproduce and die. Several *Anguilla* species have shown sharp population declines, spurring international conservation concerns (Anon. 2003, Dekker 2003, Tsukamoto 2003).

Concerns have also been raised about American eel populations. The collapse of the formerly large eel population of Lake Ontario, and decreasing indicators elsewhere, have been taken as evidence of a species-wide decline (Haro et al. 2000, Richkus and Whalen 2000, Casselman 2003). Overfishing, migration obstacles, turbine mortality in hydro dams, pollution, habitat degradation, and ocean changes have been proposed as reasons for the population changes, but no clear cause has been identified (Castonguay et al. 1994a, 1994b).

In spring 2005 the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) initiated a review of American eel status Canada in view of a possible listing under the Species at Risk Act (SARA). Part of this process requires the assembly and review of pertinent data on population status and trends by the responsible federal department. The federal department in this case is the Department of Fisheries and Oceans (DFO). However, DFO directly manages eel fisheries only in the Atlantic Provinces. Eel fisheries in Ontario are managed by the Ontario Ministry of Natural Resources and eel fisheries in Quebec are managed by the Quebec Ministère des ressources naturelles, de la faune et des parcs. Hence the task of assembling and reviewing data is a joint one between federal, Ontario, and Quebec biologists.

The central purpose of this paper is to identify the best and most reliable indicators of eel populations in Canada, and to relate them to criterion sets that COSEWIC uses to determine listing categories. To this end the paper reviews the biology and distribution of the American eel in Canada, it presents data on status and population indicators for Ontario, Quebec, and the Atlantic Provinces, and it compares indicators on regional and national scales. To allow direct comparisons, population indicators are converted to a "common currency" by setting them against the year of recruitment from the ocean. Possible causes of changes in eel abundance are also examined, in order to identify threats which impinge on eel populations in Canada.

2. TERMS OF REFERENCE

Review of information for the American eel (*Anguilla rostrata*) prior to assessment by COSEWIC

October 11-12, 2005
Quebec City, Quebec

Chairperson: Nicholas Mandrak

A. Background

The implementation of the federal Species at Risk Act (SARA), proclaimed in June 2003, begins with the assessment of a species' risk of extinction by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), an arm's-length scientific advisory body. This assessment initiates the regulatory process whereby the competent Minister must decide whether to accept COSEWIC's designation and add a species to Schedule 1 of SARA, which will result in legal protection for the species under the Act.

DFO, as the primary generator and archivist of information on aquatic species, will be expected to support the work of COSEWIC by providing the best information available on the status of a species to be assessed. DFO benefits from this activity as it allows COSEWIC to most accurately assess the status of a species when all relevant information is made available to those undertaking the assessment.

A Zonal Peer Review of the available information for the American eel (*Anguilla rostrata*), recently listed on COSEWIC's Call for Bids (January 2005), is scheduled for October 11-12, 2005 in Quebec City, Quebec.

B. General objectives

This advisory meeting is being held to undertake a science-based peer review of information (both DFO and non-DFO) that would be relevant to determining a COSEWIC status designation for the American eel.

The intent of this meeting is to have on the science record:

a) What information is available related to the status and trends of, and threats to, American eel in Canadian waters;
b) The strengths and limitations of the information; and,
c) What the meeting participants think are legitimate uses of the information, and why.

C. Specific objectives

The purpose of the meeting is to ensure that the species information held by DFO is made available to COSEWIC, including the authors of the status report, and the Chairs of the COSEWIC Freshwater Fishes Species Specialist Subcommittee.

For this species, the meeting will review information on life history characteristics, distribution and abundance, and threats in Canadian waters, which could be used by COSEWIC to determine, following its assessment guidelines and criteria, the appropriate risk category. Discussion will also consider the available information on designatable units (DUs), which could support a COSEWIC decision whether or not to consider DUs below the species' level that would be suitable for assessment and designation.

Documentation produced by this part of the meeting will include Research Documents summarising the available information on these species, and a Proceedings Document summarizing discussions at the meeting.

The following information will be reviewed to the extent that it is available:

1. Review life history characteristics

- Growth parameters: age and/or length at maturity, maximum age and/or length
- Fecundity
- Early life history pattern (e.g. duration of planktonic larval life, and major egg, larval, and juvenile transport mechanisms)
- Specialised niche or habitat requirements

2. Review designatable units — see COSEWIC 2005 "Guidelines for Recognizing Designatable Units below the Species Level" (attached).

3. Apply COSEWIC criteria for species in Canada as a whole, and for designatable units identified in 1 (if any), using information in the most recent assessments:

COSEWIC Criterion— Declining Total Population

Summarize overall trends in population size (both number of mature individuals and total numbers in the population) over as long a period as possible and in particular for the past three generations (taken as mean age of spawners). Additionally, present data on a scale appropriate to the data to clarify the rate of decline. Calculate rate of decline over last 10 years or three generations, whichever is greater.

Identify threats to abundance— where declines have occurred over the past three generations, summarize the degree to which the causes of the declines are understood, and the evidence that the declines are a result of natural variability, habitat loss, fishing, or other human activity

Where declines have occurred over the past three generations, summarize the evidence that the declines have ceased, are reversible, and the likely time scales for reversibility.

COSEWIC Criterion— Small Distribution and Decline or Fluctuation: by stock, for species in Canada as a whole, and for designatable units identified in 1 (if on a scale finer than stocks) and using information in the most recent assessments:

a. Summarise the current extent of occurrence (in km^2) in Canadian waters
b. Summarise the current area of occupancy (in km^2) in Canadian waters
c. Summarise changes in extent of occurrence and area of occupancy over as long a time as possible, and in particular, over the past three generations.
d. Summarise any evidence that there have been changes in the degree of fragmentation of the overall population, or a reduction in the number of meta-population units.
e. Summarise the proportion of the population that resides in Canadian waters, migration patterns (if any), and known breeding areas.

COSEWIC Criterion— Small Total Population Size and Decline and Very Small and Restricted: by stock, for species in Canada as a whole, and for designatable units identified in 1 ~~(if on a scale finer than stocks)~~, and using information in the most recent assessments:

a. Tabulate the best scientific estimates of the number of mature individuals;
b. If there are likely to be fewer than 10,000 mature individuals, summarize trends in numbers of mature individuals over the past 10 years or three generations, and, to the extent possible, causes for the trends.

1. Summarise the options for combining surveys to provide an assessment of status, and the caveats and uncertainties associated with each option.

2. For transboundary stocks, summarise the status of the population(s) outside of Canadian waters. State whether rescue from outside populations is likely.

As time allows, review status and trends in other indicators of the status of the species that would be relevant to evaluating the risk of extinction of the species. This includes the likelihood of imminent or continuing decline in the abundance or distribution of the species, or that would otherwise be of value in preparation of COSEWIC Status Reports.

D. Documentation

The meeting will produce the following documentation:

1. At least one Research Document that summarises the overall status of the species and the data and information held by DFO and the Provinces which manage American eel (Ontario and Quebec), which could be used by COSEWIC in assessing a status designation. Additional Research Documents that focus on regional abundance indices and fisheries will be produced as well. These reports will cover the information identified above.
2. Proceedings summarising the decisions, recommendations and major points of discussion at the meeting, including a reflection of the diversity of opinion present in the discussions.

3. BIOLOGY AND DISTRIBUTION

3.1 Species information

3.1.1 *Classification and genetics*

The American eel, *Anguilla rostrata* LeSueur 1817, is a member of the order Anguilliformes, family Anguillidae. Members of the genus *Anguilla* are termed fresh water eels, although some species (including the American eel) are able to complete their life cycle in salt water (Tsukamoto et al. 1998, Arai et al. 2004, Lamson et al. submitted). The American eel is the only North American species of the genus.

All fresh water eels belong to the genus *Anguilla*. Anguillid eels of the North Atlantic Ocean have been divided into two species based on morphological characters (Ege 1939, Tesch 1977), and on molecular phylogeny (Avise et al. 1986, Aoyama et al. 2001, Wirth and Bernatchez 2003). The American eel inhabits continental waters on the western side of the Atlantic Ocean, while the European eel *Anguilla anguilla* is found in continental waters on the eastern side of the Atlantic. Both species reproduce in the Sargasso Sea in the southern North Atlantic.

Panmixia refers to a breeding system in which all members of a species mate randomly as a single breeding population. In panmictic species, genetic structure shows no geographical heterogeneity. Recent evidence suggests that the European eel is not fully panmictic (Wirth and Bernatchez 2001, Daemen et al. 2001), but this interpretation has been contested by Dannewitz et al. (2005). The American eel is considered panmictic on the basis of DNA analysis of eels sampled from the eastern United States and the Gulf of St. Lawrence (Avise et al. 1986, Wirth and Bernatchez 2003). However, no genetic analysis has been conducted on eels from the Lake Ontario-Upper St. Lawrence eel population. Without samples from this important population, the conclusion of panmixis must be viewed as provisional.

American eels spawn in the Sargasso Sea. Leptocephali larvae are dispersed widely by ocean currents, including the Florida Current, the Gulf Stream and the North Atlantic Current, to western shores of the Atlantic Ocean. Because American eels are considered to belong to a single panmictic breeding population, they have to be managed as a single stock (Castonguay et al. 1994a, Haro et al. 2000, Casselman 2003).

3.1.2 *Designatable units*

The panmictic nature of American eel life history implies that factors affecting any life stage, in any of the geographic areas of the range, and in any array of habitats, have the potential to affect the abundance of all life stages of the species throughout the range. Nevertheless, geographic trends in abundance may vary among subpopulations. American eels occupy five ecological areas as recognized by COSEWIC, which are termed Designatable Units (DUs). These are DU1, Great Lakes-Western St. Lawrence (Ontario and western Québec); DU2, Eastern St. Lawrence (eastern Québec); DU3: Maritimes (New Brunswick, Nova Scotia, Prince Edward Island); DU4, Atlantic Islands (Newfoundland); and DU 5, Eastern Arctic (Labrador). In this report, that part of DU3 that drains into the Atlantic Ocean and the Bay of Fundy is referred to as Scotia-Fundy.

3.2 Distribution

3.2.1 *Global range*

The American eel is widely distributed in the fresh waters (streams and lakes), estuaries and coastal marine waters along more than 50 degrees of latitude (from 5° to 63°) of the western North Atlantic Ocean coastline, from Venezuela to Greenland (Scott and Crossman 1973, Tesch 1977; Helfman et al. 1987).

3.2.2. *Canadian range*

The historic Canadian range encompasses all accessible fresh water, estuaries and coastal marine waters connected to the Atlantic Ocean of Canada, up to the mid-Labrador coast (Fig. 3.1). Continental shelves are used by juvenile eels arriving from the spawning grounds, and by silver eels returning to the spawning grounds. Hamilton Inlet-Lake Melville, Labrador, is usually taken as the species' northern limit in Canadian waters (Scott and Crossman 1973). The presence of eels in electrofishing surveys in the English River, near Postville, Labrador (D. Reddin, DFO, pers. comm.) indicates that the species somes strays north of Hamilton Inlet-Lake Melville. Postville is about 100 km north of Hamilton Inlet.

Niagara Falls is the natural limit of the American eel's distribution in the Great Lakes. Occurrences reported in the upper Great Lakes watersheds (Lakes Erie, Huron and Superior) are the result of recent dispersal through the Erie and Welland canals (Scott and Crossman 1973). Such records should be considered vagrant (Fig. 3.1).

Relative abundance of eels is poorly known in much of the species' range in Canada. In areas with active current commercial fisheries (Figs. 3.2 and 3.3), eels must be present in at least moderate abundance.

3.3 Habitat

The American eel is said to use the broadest diversity of habitats of any fish species (Helfman et al. 1987). During their spawning and oceanic migrations, eels occupy saltwater and, in their continental phase, they use all salinity zones. During the continental phase, marine habitat use is limited to shallow protected waters. Survival is affected by environmental conditions in any habitat (oceanic, estuarine, fresh water) utilized during any life cycle phase, and by anthropogenic factors such as hydro-dams, habitat modification and fisheries.

Growing eels are primarily benthic, utilizing substrate (rock, sand, mud) and bottom debris such as snags and submerged vegetation for protection and cover (Scott and Crossman 1973, Tesch 1977).

Eel densities typically diminish with distance from the sea in medium and large rivers (Smith and Saunders 1955, Gray and Andrews 1971, Smogor et al. 1995). However, this pattern may be altered by natural or artificial obstacles. In a population of European eels, White and Knights (1997) reported that barriers to upstream migrants had a greater effect on eel densities than distance from the ocean. Ability to overcome obstacles is size-dependent. Small eels (less than 10 cm long) are able to creep up damp vertical barriers (Legault 1988), but larger eels are generally unable to bypass large waterfalls and dams (McCleave 1980, Barbin and Krueger 1994). Hence, to avoid size-dependent settlement, larger eels attempting to move upstream require unobstructed passage (Moriarty 1987).

Survival of maturing eels in their seaward migration is reduced by passage through hydroelectric turbines (Desrochers 1995, Normandeau Associates and Skalski 2000), fisheries (Castonguay et al. 1994a, Caron et al. 2003, Verreault et al. 2003), and by obstructions which produce free falls of more than 13 m (Larinier and Travade 1999).

Continental-phase American eels are highly plastic in their habitat use. In streams, eels generally do not show consistent preferences for habitat type, cover, substrate, water temperature, and density of predators (Hawkins 1995, Smogor et al. 1995), but there is some association between eel densities and diversity of depth-velocity regimes (Wiley et al. 2004). In Prince Edward Island, eels are abundant in fresh water ponds formed by dams but are rare in most fresh water streams (Cairns et al., submitted).

Some continental-phase eels are predominantly sedentary but others are predominantly migrant (Feunteun et al. 2003). Since otolith is essentially calcium carbonate in an organic proteinaceous matrix, Casselman (1982) interpreted otolith banding as indicating migration stages. Recent investigations using otolith microchemistry (Jessop et al. 2002, Cairns et al. 2004, Thibault et al. 2005, Lamson et al. submitted) report at least three distinct movement behaviours: salt water residency, fresh water residency, and inter-habitat shifting. In the St. Jean River on the Gaspé Peninsula, Thibault et al. (2005) also found that some fresh water resident eels performed very short intrusions into brackish or salt water. Inter-habitat shifting is more frequent in systems where dams do not hinder movements (Jessop et al. 2002, Morrison et al. 2003). Catadromy is no longer seen as obligate for eels, but rather is a facultative life history option (Tsukamoto et al. 1998, Morrison et al. 2003, Arai et al. 2004, Lamson et al. submitted). Seasonal local movements associated with wintering could also involve habitat needs in terms of water temperature, oxygen concentration and water quality, but winter habitat requirements are poorly known (Tesch 1977, Feunteun et al. 2003).

Eels spawn in the Sargasso Sea (Schmidt 1922), east of the Bahamas and south-west of Bermuda (25 °N; 60 °W; McCleave et al. 1987), but habitat requirements for spawning and incubating are unmeasured and poorly understood. Kleckner and McCleave (1988) related the northern limit of spawning by Atlantic eels (*Anguilla* spp.) in the Sargasso Sea to thermal fronts and surface water masses. Spawning would occur south of east-west thermal fronts separating southern Sargasso Sea surface water from mixed Subtropical Convergence Zone water to the north.

3.4 Life cycle

3.4.1 *Overview*

The life history of the American eel encompasses oceanic, coastal, estuarine and fresh water environments. Spawning and hatching take place in the Sargasso Sea (Schmidt 1922). Larvae are transported to coastal waters by the Gulf Stream system formed by the Florida Current, the Gulf Stream and the North Atlantic Current. Some arriving juvenile eels migrate up rivers to become resident yellow eels of fresh water habitats; whereas, others remain in brackish or salt waters, and still others show inter-habitat movement patterns (Jessop et al. 2002, Cairns et al. 2004, Thibault et al. 2005, Lamson et al. submitted). After a number of years (mean 8 to 23; this report) in growth habitats, adult eels mature into silver eels that migrate back to the spawning grounds. Spawning occurs once; therefore, the American eel is a semelparous species (Helfman et al. 1987) and every mortality in continental waters is a pre-spawning mortality. The terminology of eel life history differentiates stages according to migration patterns and morphological characteristics. Life stages are detailed below.

3.4.2 *Egg*

The egg probably hatches within a week of deposition in the Sargasso Sea. McCleave et al. (1987) suggested that hatching peaks in February and may continue until April. According to otolith back-calculations (Wang and Tzeng 2000), hatching occurs from March to October and peaks in August. However, Cieri and McCleave (2000) argued that such back-calculated spawning dates do not match collection evidence and may be explained by resorption.

3.4.3 *Leptocephalus*

The leptocephalus is the larval form. Leptocephali are transparent willow leaf-like, laterally compressed larvae (Figure 3) that are passively transported west and north to coastal waters on the eastern coast of North America, by the surface currents of the Gulf Stream system (Schmidt 1922, Tesch 1977, Kleckner and McCleave 1982). This stochastic larval distribution is completed in 7 to 12 months (Kleckner and McCleave 1985, Wang and Tzeng 2000). Vertical distribution is usually restricted to the upper 350 m of the ocean (Kleckner and McCleave 1982, Castonguay and McCleave 1987). Growth has been evaluated at about 0.21 to 0.38 mm per day (Kleckner and McCleave 1985, Castonguay 1987, Tesch 1998, Wang and Tzeng 2000).

3.4.4 *Glass eel*

Upon entering continental shelf waters, leptocephali metamorphose into glass eels (Figure 4), which have the typical elongate and serpentine eel shape (McCleave et al. 1987). The term glass eel refers to all developmental stages from the end of metamorphosis in the leptocephalus to pigmentation (Tesch 1977). Metamorphosis occurs when leptocephali are about 55 to 65 mm long (Kleckner and McCleave 1985). Mean age at this metamorphosis has been evaluated at 200 days and estuarine arrival at 255 days; giving 55 days between glass eel metamorphosis and estuarine arrival (Wang and Tzeng 2000). During the elver run in the East River (Nova Scotia), the degree of elver pigmentation increased progressively over the run, and glass eels were rarely found after the end of May (Jessop 2003). On the north shore of the Gulf of St. Lawrence, in the Petite Trinité River, glass eels occurred in the second half of June and were rare compared to elvers (Dutil et al. 1989).

Glass eels use of selective tidal stream transport to reach the shore and move upstream (Kleckner and McCleave 1982). This post-larval metamorphosis essentially transforms the eel from a pelagic organism to a benthic organism.

3.4.5 *Elver*

Glass eels become progressively pigmented as they approach the shore; these eels are termed elvers. The melanic pigmentation process (Bertin 1951, Élie et al. 1982, Grellier et al. 1991) occurs when the young eels are in coastal waters. At this phase of the life cycle, the eel is still sexually undifferentiated. The elver stage lasts about three to twelve months. Elvers that enter fresh water may spend much of this period migrating upstream (Haro and Krueger 1991, Jessop 1998). Elver influx is linked to increased temperature and reduced flow early in the migration season, and to tidal cycle influence later on (Tesch 1977, Kleckner and McCleave 1982, Martin 1995).

Elver length and arrival date increase from south to north along the Atlantic coast of North America (Vladykov 1966, Haro and Krueger 1988). In Atlantic coastal Nova Scotia, elver migration peaks between late April - early May and late June, although small numbers of elvers may continue to enter rivers until mid - August (Jessop 1998). Total length averaged 60.14 ± 0.17 mm (50.4 – 70.5 mm) in 2000 on the East River Chester, Nova Scotia (Jessop 2003). On the Murray River (Prince Edward Island), elvers were caught between the end of June and the end of August (Cairns et al., submitted). In the Petite Trinité River (north shore of the Gulf of St. Lawrence), most individuals were already pigmented (elvers) in early July but arrived until the end of July (Dutil et al. 1989) and averaged 62.4 mm (59 69 mm).

3.4.6 *Yellow eel*

The yellow stage is the growth phase of the species. The skin color varies from yellowish to greenish or olive-brown, with the back darker than the belly (Scott and Crossman 1973, Tesch 1977). The skin is thick and tough and may secrete copious amounts of slimy mucous, which acts as a protective cover. Unlike the well-developed scales of most other fishes, eel scales are rudimentary and embedded deeply within the skin.

Sexual differentiation occurs during the yellow stage and appears to be determined by environmental parameters (Krueger and Oliveira 1997, Oliveira 1997). In most Canadian waters more than 95% of sexually differentiated eels are female (Gray and Andrews 1970, Dolan and Power 1977, Dutil et al. 1985, Jessop 1987, Fournier and Caron 2005). Males appear to be more common in the Scotia-Fundy area than elsewhere in Canada. In the Saint John River, males were 7.4% of a sample of 970 eels (Ingraham 1999). Eels which had been captured as elvers in the Bay of Fundy and stocked in a lake on the south shore of the St. Lawrence River (DU2) contained 27.2% males after four years of growth (Verreault et al. submitted).

The yellow eel may continue to migrate upstream for many years. Juveniles passing the eel ladder at the Moses-Saunders dam near Cornwall (Ontario) in the 1970s and 1980s ranged in age from 3 to 7 years, averaging 5 years (Casselman et al. 1997), but now are much older, ranging in age from 10 to 14 years, averaging 12 years (J.M. Casselman, OMNR, pers. comm.). These eels are headed for Lake Ontario, the largest single growth habitat for the American eel within its distribution range.

In the Sud-Ouest River (south shore of the St. Lawrence) and the Petite Trinité River (north shore of the Gulf of St. Lawrence), upstream migrations occurred between June and August (Dutil et al. 1989, Verreault 2002, Fournier and Caron 2005). At Chambly, Beauharnois and Moses-Saunders dams, migration generally peaks in July and August (Casselman et al. 1997, Verdon et al. 2003, Bernard and Desrochers 2005). In subpopulations that have an easy access to brackish or salt waters, eels are reported to move from fresh water to estuaries in the spring, and move back to fresh water in the fall (Medcof 1969; F. Caron, Faune Québec, pers. comm.).

Home ranges occupied by individual yellow eels vary with habitat type (stream, lake, tidal, creek, marsh, estuary) and movement patterns (fresh water residency, salt water residency, inter-habitat shifting). Home range was estimated to be relatively small (up to 2 ha) in estuarine habitats such as salt marshes (Ford and Mercer 1986) and tidal streams (Bozeman et al. 1985, Dutil et al. 1988), but in Lake Champlain, LaBar and Facey (1983) reported home ranges up to 65 ha.

Growth patterns are affected by habitat type. Eels utilizing brackish and salt waters grow much more rapidly than those in fresh water (Jessop et al. 2002, Cairns et al. 2004). Disparities exist in fresh water habitats as well. Annual growth increments of eels translocated in a watershed originally without eels (see Stocking section) were much lower in rivers (40 mm per year) than in a lake (112 mm per year; Verreault et al., submitted).

In Canadian waters, American eels hibernate in mud during the winter. Wintering areas include fresh water, brackish estuaries, and bays with full strength salt water (Smith and Saunders 1955, D.K. Cairns, pers. obs.). Eels in estuaries often concentrate in mud that has upwelling fresh water, but they may also be found in bottoms that have no fresh water influx (D.K. Cairns, pers. obs.). Eels are reported to enter torpor at temperatures below 5 - 10°C (Walsh et al. 1983). However, Smith and Saunders (1955) caught two eels in a stream trap in Prince Edward Island in January and February. Eels speared through the ice in a Prince Edward Island estuary in January had stomachs bulging with fresh and undigested silversides (*Menidia menidia*), indicating recent feeding activity (D.K. Cairns, pers. comm.). These observations suggest that eels are occasionally active during the normal period of hibernation.

3.4.7 *Silver eel*

As the maturation process proceeds, the yellow eel metamorphoses into a silver eel. The silvering metamorphosis results in morphological and physiological modifications that prepare the animal to migrate back to the Sargasso Sea. The eel acquires a greyish colour with a whitish or cream coloration ventrally (Gray and Andrews 1971, Scott and Crossman 1973, Tesch 1977). The digestive tract degenerates (Pankhurst and Sorensen 1984, Durif 2003), the pectoral fins enlarge to improve swimming capacity (Pankhurst 1982a, McGrath et al. 2003, Durif 2003), eye diameter expands and visual pigments in the retina adapt to the oceanic environment (Vladykov 1966, Pankhurst 1982b, McGrath et al. 2003), the integument thickens (Tesch 1977, Pankhurst and Lythgoe 1982), percentage of somatic lipids increases to supply energy for migrating and spawning (Larsson et al. 1990, Tremblay 2004), gonadosomatic index (Verreault 2002, McGrath et al. 2003, Tremblay 2004) and oocyte diameter increase (Couillard et al. 1997), gonadotropin hormone (GTH-II) production increases (Durif et al. 2005), and osmoregulatory physiology changes (Dutil et al. 1987).

Distance traveled to reach the Sargasso Sea during seaward migration varies substantially over the geographic range of the American eel. A silver eel from the most distant growth habitat, western Lake Ontario, must migrate more than 4 500 km to reach the spawning grounds, whereas a maturing eel from the closest Canadian growth habitat, southern Nova Scotia, migrates about 2 000 km. Disparities in migration distance generate disparities in the timing of the onset of the migration, probably related to synchronous arrival in the Sargasso Sea, permitting spawning between February (peak) and April (Kleckner et al. 1983, Kleckner and McCleave 1985, Helfman et al. 1987), maturing eels from Lake Ontario must begin outmigrating in mid- to late June (McGrath et al. 2003, J.M. Casselman, OMNR, pers. comm.), whereas silver eels from Nova Scotia outmigrate until November (Jessop 1987). Downstream migration occurs primarily at night.

There are major disparities in reproductive characteristics of silver eels across the species' range (Nilo and Fortin 2001). Eels from northern subpopulations show slower growth and greater length, weight, and age at migration (Hurley 1972, Facey and LaBar 1981, Helfman et al. 1987). For a given subpopulation, size, rather than age, appears to be the main cue triggering maturation and migration (Helfman et al. 1987, Oliveira 1999, Verreault 2002, Tremblay 2004). A substantial variability in length of female silver eels is observed within the St. Lawrence River watershed (Table 3.1). Age data are scarce and restrict our knowledge of long-term recruitment indices. However, a generation time for the Lake Ontario-St. Lawrence River stock is approximately 20 years, much longer than in the southern part of the species range along the Atlantic seaboard of the United States (approximately 6-14 years, J.M. Casselman, OMNR, pers. comm.).

American eel fecundity varies with body size (Wenner and Musick 1974, Barbin and McCleave 1997, Tremblay 2004). Female silver eels from the upper St. Lawrence River, the Richelieu River and the Sud-Ouest River are generally much larger than those from elsewhere in Canada (Table 1). In five subpopulations within the St. Lawrence River and Gulf, Tremblay (2004) found that absolute fecundity varies, ranging from 3.4 to 22 million eggs for body lengths ranging from 53.2 to 111.0 cm and body weight ranging from 260 to 3340 g. Large-bodied eels average about 6.5 million oocytes / kg; while small-bodied eels have more than 10 million oocytes / kg.

Thus, maturing eels from the upper St. Lawrence River – Lake Ontario and from the Sud-Ouest River are generally much larger (Table 1). Therefore, the importance of the reproductive capacity of eels of these subpopulations from DU1 and DU2 to the overall fecundity of the species must be considered.

3.5 Nutrition

3.5.1 *Leptocephalus*

Little is known about the food habits of leptocephali. Recent studies on other eel species (Otake et al. 1993, Mochioka and Iwamizu 1996) suggest that leptocephali do not feed on zooplankton but rather feed on detrital particles such as marine snow and fecal pellets or particles such as discarded larvacean houses.

3.5.2 *Elver*

There are conflicting data regarding which elver pigmentation stage begins feeding. Based on laboratory experiments on European eels, Lecomte-Finiger (1983) reported that elvers were morphologically and physiologically unable to feed, whereas Tesch (1977) found that elvers on stage VIA4 (Élie et al. 1982) were feeding. Stomach examination of elvers caught during their upstream migration in the Petite Trinité River on the north shore of the Gulf of St. Lawrence revealed that elvers fed primarily on insect larvae (Dutil et al. 1989).

3.5.3 *Yellow eel*

The yellow eel is essentially a benthic omnivore foraging on bottom-living organisms at night. Prey includes fishes, molluscs, crustaceans, insect larvae, surface-dwelling insects, worms and plants. The eel prefers small prey animals which can easily be attacked (Tesch 1977). Food type varies with body size (Ogden 1970, cited in Tesch 1977). The stomach contents of eels less than 40 cm and captured in streams consisted mainly of aquatic insect larvae, whereas larger eels fed predominantly on fishes and crayfishes. In Lake Champlain, food sources were mainly fish (38%), decapods (30%) and insects (10%). Insect abundance decreased in larger eels (Facey and LaBar 1981). The eel diet adapts to seasonal changes and the immediate environment. Feeding activity decreases or stops during the winter, and food intake ceases as eels physiologically prepare for the spawning migration.

Table 3.1
Migration periods, mean length and age of female silver eels exiting Canadian fresh water systems.

Site	DU[a]	Migration period	N	Length (cm)	Age	Reference
Upper St. Lawrence River, St. Francis and St. Lawrence lakes, Moses-Saunders, Vicinity of Iroquois dam	1	Jun–Oct	200	91.5	20	Casselman 2003
			53	97.6		McGrath et al. 2003
			30	100.1	21	Tremblay 2004
Richelieu River	1	Jun-Oct	494	101.9		Dumont et al. 1998
St. Lawrence estuary	1	Sep-Oct	474	84.0		Couillard et al. 1997
			4529	85.3		Verreault et al. 2003
			30	83.7	20	Tremblay 2004
Sud-Ouest River (South shore of the St. Lawrence)	2	Aug-Nov	107	102.6	21	Verreault 2002
			30	104.3	23	Tremblay 2004
Petite Trinité River (North shore of the St. Lawrence)	2	Aug-Oct	424	65.0		Fournier and Caron 2005
			30	67.9	20	Tremblay 2004
Long Pond (Prince Edward Island)	3	Aug-Oct	30	69.3	20	Tremblay 2004
Topsail and Indian ponds, Salmon River, Burnt Berry Brook, Topsail Barachois (Newfoundland)	4	Aug-Oct	92	69.4	12	Gray and Andrews 1971
LaHave River (Nova Scotia)	3	Aug-Nov	346	61.1	19	Jessop 1987

[a] Designatable units: 1) Great Lakes-Western St. Lawrence; 2) Eastern St. Lawrence); 3) Maritimes; 4) Atlantic Islands

Table 3.2

Age frequency distributions of American eels in Canada. Eels are considered to be age 0 in the year of arrival in continental waters.

Age	Province	Ontario	Ontario	Ontario	Ontario	Quebec	Quebec	Quebec	Quebec	Quebec	Quebec	Quebec	Quebec	Quebec
	Region						N. Shore	N. Shore	N. Shore	NW Gulf	NW Gulf	Gaspé Pen.	Gaspé Pen.	Gaspé Pen.
	System	Lake Ontario	St. Lawrence R.	St. Lawrence R.	Ottawa R.	St. Lawrence estuary	Petite r. de la Trinité	Petite r. de la Trinité	Petite r. de la Trinité	Matamek R.	Ruisseau Sylvie	St. Jean	St. Jean	St. Jean
	Sites	Eastern Lake	Moses-Saunders dam	Moses-Saunders dam	Chats Falls	Various				Bill L.				
	Habitat	Lake	River	River	River	Estuary	River	River	Lake	Lake	Estuary	River	Lake	Estuary
	Stage	Yellow	Ascending juveniles	Ascending juveniles	Upstream migrants	Silver	Silver	Yellow, mig. upstream	Yellow			Yellow, mostly mig. downstream	Yellow	Yellow
	Year	1958-1966	1970s-1980	1990s	1963-1964	1970	1999-2001	1999-2001	2000-2001	1970	1973	2004	2004	2004
	Gear	Various	Ladder trap	Ladder trap	Trap	Trap				Fyke	Rotenone			
	Exploited	Yes			No	Yes	No	No	No	No	No	No	No	No
	Size sampled	All			All	All	All	All	All	All	All	All	All	All
	Source	Hurley 1972			Hurley 1972	Larouche et al. 1974	Caron et al. unpubl.	Caron et al. unpubl.	Caron et al. unpubl.	O'Connor & Power 1973	Dolan 1975	Caron & Thibault unpubl.	Caron & Thibault unpubl.	Caron & Thibault unpubl.
Mean		11.49	5.60	11.90	4.53	16.83	19.22	6.85	10.80	10.58	2.23	8.68	11.22	7.35
SD		2.66			2.24	2.88	4.01	4.33	6.57	2.32	2.74	3.81	4.30	2.18
N		195			17	2931	74	133	45	19	69	78	36	48
0											23			
1								8			16			
2					3			17	6		5			
3					5			18	4		12	1		
4		3			2			9				7		7
5					1			7	2		2	6		5
6		4			2			11	2		2	10	5	5
7		10			2	2		9	2		1	5	6	3
8		12			1	4		10	1	2	2	13		17
9		10			1	5		6	2	7	3	6	4	2
10		27				20		4	2	3	1	13	2	4
11		25				41	1	6	3	1	1	8	4	5
12		27				87	1	7	3	2	1	3	3	
13		33				168	4	11	2	2		1	1	
14		23				264	1	5	2			1	2	
15		12				358	6	2	3	1			2	
16		5				475	5	3	3	1		1	3	
17		3				409	7		2				1	
18		1				358	7		1			1		
19						189	9		2					
20						199	5						3	
21						152	5		1					
22						106	7							
23						58	2							
24						30	7					2		
25						6	4		1					
26							1							
27							1							
28									1					
29														
30							1							

Table 3.2
continued

Age	Province	NB	NB	NB	NB	NB	NB	NS	PEI	PEI	PEI	PEI	PEI	PEI	PEI
	Region	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf	Gulf
	System	Restigouche R.	Miramichi Bay	Miramichi Bay	Miramichi Bay	Miramichi R.	Miramichi R.	Margaree R.	Various	Various	Various	Various	Various	Wheatley River	Brackley Bay
	Sites		Oak Pt.	Oak Pt.	Oak Pt.	Various	Rocky & Clearwater Brks.	Below Seal							
	Habitat	Stream	Bay	Bay	Bay	Stream	Stream	River	Bays & estuaries	Bays & estuaries	Bays & estuaries	Bays & estuaries	Bays & estuaries	Estuary bottom mud	Bay bottom mud
	Stage	Yellow				Yellow	Yellow	Yellow	Mostly yellow	Mostly yellow	Mostly yellow	Mostly yellow	Mostly yellow	Yellow	Yellow
	Year	2004	2003	2003	2003	2003	2003	2004-2005	1999-2003	1999-2003	1999-2003	1999-2003	1999-2003	Wint. 2004-05	Wint. 2004-05
	Gear	Rot. Screw	Fyke	Fyke	Fyke	E-fishing	Rot. Screw	Rot. Screw	Fyke	Fyke	Fyke	Fyke	Fyke	Mud spear	Mud spear
	Exploited	No	Yes	Yes	Yes	No	No	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes
	Size sample	All	>=20 cm	>=38.1	>=46 cm	All	All	All	All	>=46 cm	>=50.8 cm	<46 cm	<50.8 cm	All	All
	Source	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO	DFO
Mean		5.41	7.00	7.12	7.29	5.86	6.07	8.37	5.67	6.04	6.21	4.54	4.86	7.15	6.50
SD		3.69	0.97	0.86	0.76	2.49	1.36	3.35	1.29	1.05	1.03	1.29	1.21	1.54	1.57
N		46	74	68	56	29	28	107	370	278	222	92	148	53	22
0															
1									1			1	1		
2						2			2			2	2		
3		14				2		1	15	1		14	15	1	
4		6				5	2	5	40	8	4	32	36	3	4
5		8	1	4	2	4	9	8	102	80	51	22	51	5	1
6		8	6	6	1	5	8	12	125	109	91	16	34	6	5
7		6	7	4	35	7	6	23	58	54	51	4	7	10	6
8		2	41	16	15	1		24	22	22	21		1	19	4
9			16	3	3	1	3	8	4	3	3	1	1	9	2
10		1	3			1		8	1	1	1				
11								6							
12								1							
13								2							
14						1		3							
15								1							
16								1							
17															
18								2							
19								1							
20															
21															
22															
23															
24								1							
25															
26															
27		1													
28															
29															
30															

Table 3.2
continued

Age	PEI	PEI	NS	NS	NS	NS	NS	NS	NS
	Gulf	Gulf	Atlantic	Atlantic	Atlantic	Atlantic	Atlantic	Atlantic	Atlantic
	Brackley-Covehad	Long & Campbells Ponds	Bridge End & Wentzells Lakes	Eel Brook	Medway River	LaHave River	Saint John	Saint John	Saint John
	Various	Outlets to sea					Mactaquac below dam	French-Indian L.	Grand Lake
	Fresh impoundments	Stream	Lake	Stream	River	River	River	Lake	Lake
	Yellow	Silver	Yellow	Yellow	Silver females	Silver females	Yellow, silver	Yellow, silver	Yellow, silver
	2003	2002-2003	1979	1983	1983	1983	1991	1991	1991
	Fyke	Fyke	Pots	Weir	Weir	Weir	Fyke		
	No	No	?	Yes	Yes	Yes	Yes		
	All	All	All	All	All	All	All	All	All
	H. Lamson unpubl.	DFO	Hutchison & Taylor 1980	Jessop 1987	Jessop 1987	Jessop 1987	B. Jessop DFO unpubl.	B. Jessop DFO unpubl.	B. Jessop DFO unpubl.
Mean	10.12	17.67	9.74	12.90	19.20	19.40	13.84	15.94	14.84
SD	5.30	4.37	3.78				2.80	4.24	3.63
N	65	88	19	120	272	352	51	51	211
0									
1									
2									
3	1								
4	3		2						
5	12								
6	10	4	3						
7	7	1						1	1
8	5		2				1		4
9		1	5				1		3
10							3	2	7
11	1	1					5	6	18
12	1	2	1				7	3	23
13	3	4	2				9	2	29
14	6	5	2				6	4	28
15	2	2					4	8	14
16	2	5	2				8	4	20
17	4	8					1	6	23
18	2	13					3	4	14
19	2	9					1	2	8
20	2	14					1	1	4
21	1	8					1	3	8
22		7							
23	1	1							1
24								3	2
25								1	
26		2						1	2
27									1
28									
29									
30		1							1

Table 3.2
continued

Province	Age	Nfld	Nfld	Nfld	Nfld	Nfld	Nfld	Nfld
Region		Atlantic	Atlantic	Atlantic	Atlantic	Atlantic	Atlantic	Atlantic
System		Salmon R.	Burnt Berry Brk.	Indian Pond	Dog Bay	Topsail Barachois	Topsail Pond	Holyrood Bay
Sites								
Habitat		Stream	Stream	Brackish pond	River mouth	Brackish pond	Fresh pond	River mouth
Stage		Yellow	Yellow	Yellow	Silver	Yellow	Silver	Silver
Year		1968	1967	1967	1981	1967	1967	1981
Gear		Smolt trap	E-fishing	Pots	Fyke	Pots	Fyke	Fyke
Exploited		?	?	?	Yes	?	No	Some
Size sample		All	All	All	All	All	All	All
Source		Gray & Andrews 1975	Gray & Andrews 1975	Gray & Andrews 1975	Bouillon & Haedrich 1985	Gray & Andrews 1975	Gray & Andrews 1975	Bouillon & Haedrich 1985
	Mean	6.66	5.67	8.75	12.97	8.03	12.29	12.90
	SD	1.48	2.05	1.64	1.57	1.54	1.64	1.64
	N	38	66	75	94	135	92	90
	0							
	1							
	2		2					
	3		6					
	4	3	13			2		
	5	7	13	2		5		
	6	6	15	6		19		
	7	9	3	10		20		
	8	11	6	13		27		
	9	1	5	15	1	45	5	2
	10	1	2	20	3	12	5	3
	11		1	7	11	4	13	11
	12			2	21	1	34	21
	13				27		20	22
	14				16		6	20
	15				12		6	5
	16						1	3
	17				2		1	3
	18				1		1	
	19							
	20							
	21							
	22							
	23							
	24							
	25							
	26							
	27							
	28							
	29							
	30							

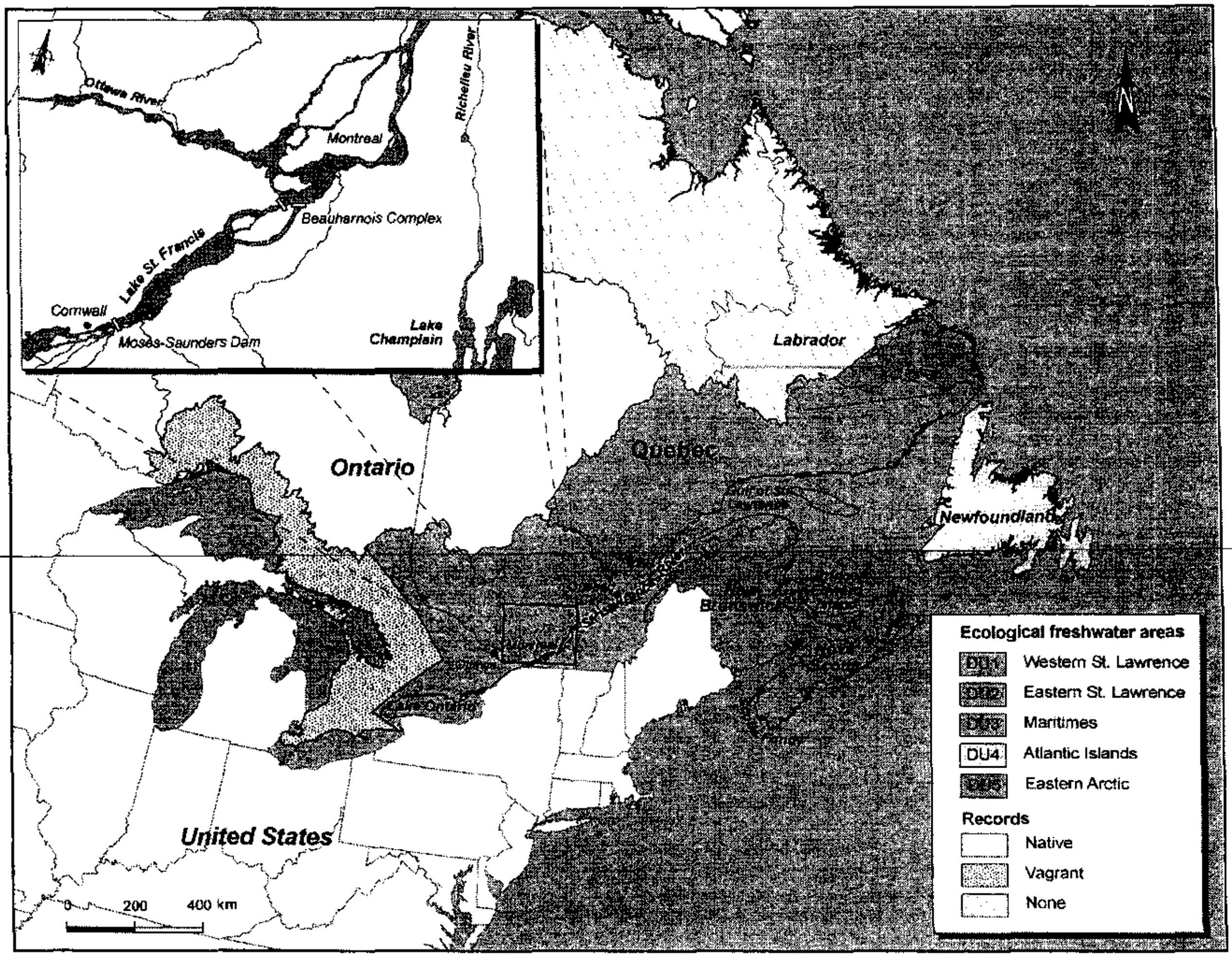


Fig. 3.1
Canadian geographic range of the American eel (vagrant records from N. Mandrak, DFO).

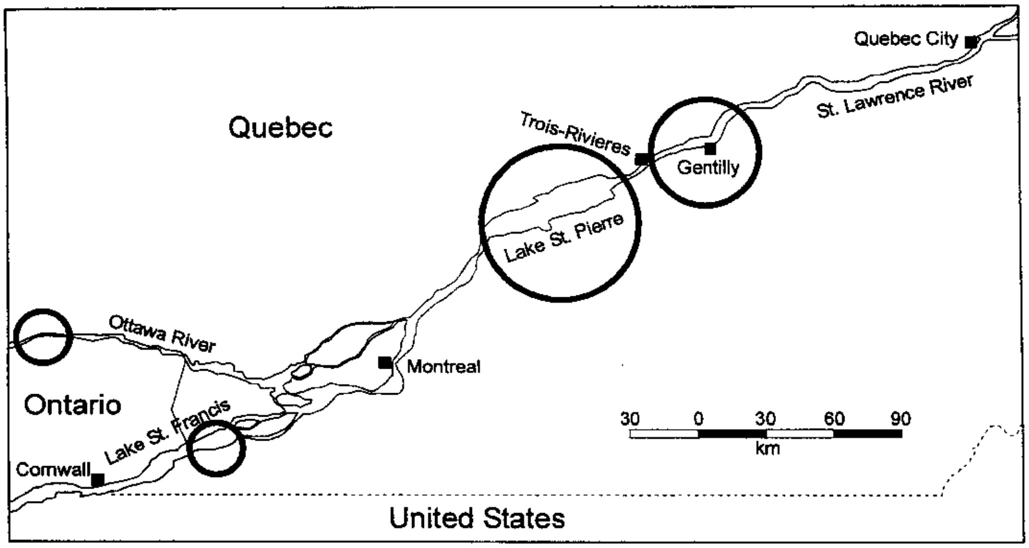


Fig. 3.2
Areas of commercial fishing for locally produced yellow eels (circles) in Quebec west of Quebec City. Data sources: Yves Mailhot, Pierre Dumont, Pierre Pettigrew, Guy Verrault; Faune Québec.

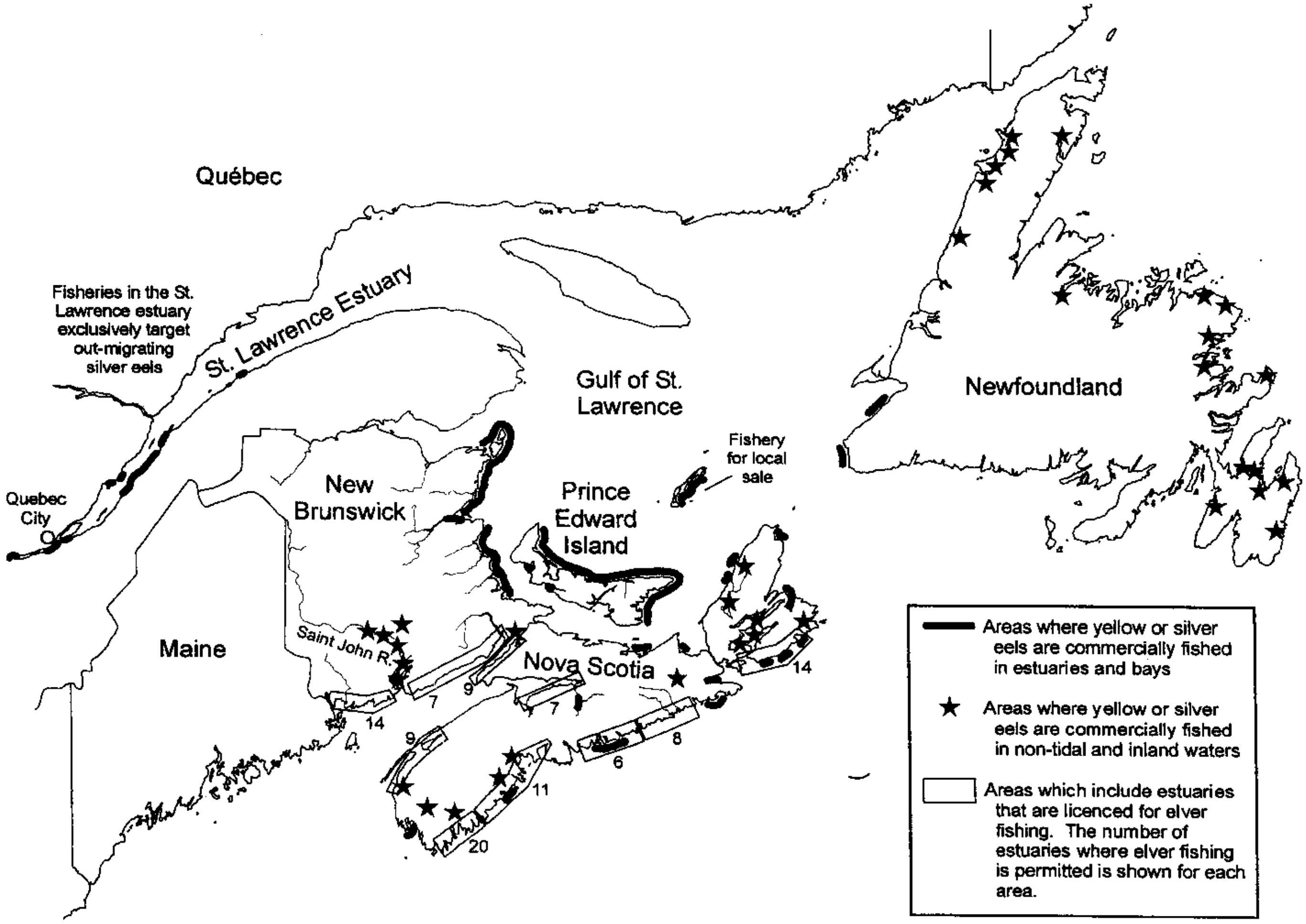


Fig. 3.3
Areas of commercial eel fishing in Canada east of the Quebec City area. Data sources: Mitchel Feigenbaum and Paul Firminger, South Shore Trading Limited; François Caron, Stan Georges, and Rémi Tardif, Faune Québec; David Cairns, Brian Jessop, and Greg Stevens, Department of Fisheries and Oceans.

4. EEL ABUNDANCE INDICATORS IN ONTARIO

4.1 Background

Eels in Ontario are part of the Upper St. Lawrence - Lake Ontario (USLLO) eel population, which occupies DU1 (Great Lakes and Western St. Lawrence, Fig. 3.1). Waters of eastern Ontario are the most distant of any major rearing area from the American eel's spawning grounds. The principal historic areas used by eels in Ontario are the main stem of the St. Lawrence River, Lake Ontario (especially the eastern part), and the Ottawa River (Verreault et al. 2004). Accessible tributaries to these water bodies were also occupied.

Eels were present in eastern Ontario waters in prehistoric time (from archaeological digs) and in early historic time (from contemporary reports) (Casselman 2003). Data from these sources suggest that eels were abundant and were a significant food source for native peoples (Casselman 2003).

Population status, fisheries, and management of eels in Ontario waters have been described by Hurley (1972, 1973), Casselman et al. (1997a,b), Stewart et al. (1997), Casselman (2003), Marcogliese and Casselman (submitted), and Mathers and Stewart (submitted).

4.2 Population indicators

4.2.1 *Resident populations*

Reported landings in Ontario waters increased gradually from the middle of the 20th century to peak in the late 1970s, and then declined (Table 4.1, Fig. 4.1). The increase in the 1970s occurred at a time of rising prices and the opening of new markets (Mathers and Stewart submitted). The sharp drop in 1982 was due to the closure of European markets because of contaminant levels. In 2004 the fishery was closed and landings ceased.

A trawl survey has been conducted in the Bay of Quinte, an arm of Lake Ontario, since 1972 (Casselman 1997b). The index shows very high interannual variability (Table 4.2, Fig. 4.2), but long-term trends are nevertheless evident. The index declined irregularly from the early 1980s to the early 2000s, when catches per trawl were virtually nil.

A survey using commercial electrofishing gear in eastern Lake Ontario (Casselman 1997b) showed relatively stable catches per hour from 1984 to 1989 (Table 4.3, Fig. 4.3). The index then declined to very low levels. Eels caught per hour in 2003 and 2004 were less than 1% of values recorded in the 1980s.

4.2.2 *Recruits*

Eels destined for Lake Ontario must pass the Moses-Saunders hydroelectric dam which spans the St. Lawrence River between Cornwall, Ontario, and Massena, New York. This dam and the navigation facilities of the St. Lawrence Seaway were constructed in 1954-1958 (Marcogliese and Casselman submitted). Prior to 1974, eels reaching the Moses-Saunders headpond must have done so via the Seaway navigation locks. In August 1974, an eel ladder was constructed at Moses-Saunders. Mean daily counts of eels ascending the ladder during the 31 day peak migration period provide an index of eel migration around the dam (Casselman et al. 1997a, Casselman 2003, Marcogliese and Casselman submitted). The index rose from its inception in 1974 to peak in the early 1980s. The index declined steeply from 1983 to 1986, then less steeply from 1987 to 1995. The index has been stable at a very low level since 1998. Mean index values in 2000-2004 were 99.7% lower than mean index values in 1974-1983.

Sizes of eels sampled at Moses-Saunders have shifted upwards over the years of ladder operation (Casselman 2003). Mean lengths increased in linear fashion from about 32.5 cm in the mid-1970s to about 47.5 cm in the 2000s (Marcogliese and Casselman submitted). Mean ages of ascending eels increased from 5.6 years in the 1970s-1980 to 11.90 years in the 1990s (Table 3.2). With these shifts in size and age, ascension of small, young eels above Moses-Saunders has virtually ceased.

Historically, numbers of eels ascending the ladder in fall were a very small fraction of the total run (Marcogliese and Casselman submitted). By the 2000s, eels ascending in fall were a substantial portion of the total run.

4.2.3 *Indicators against recruitment year*

Figs. 4.1-4.4 plot indices against the year in which recruiting eels arrived in continental waters, based on mean ages given in Table 3.2. The declines in the Lake Ontario electrofishing and the Moses-Saunders ladder indices began in the late 1970s recruitment years (Figs. 4.3-4.4). A starting point for the decline in the Bay of Quinte trawl index cannot be defined due to that index's great inter-annual variability (Fig. 4.2).

Table 4.1
Reported American eel landings in Ontario, 1950-2005, against year of landings and against year of recruitment. Landings data are from Ontario Ministry of Natural Resources. Year of recruitment is based on a mean age of 11 years (Hurley 1972, see Table 3.2)

Landing year	Recruitment year	Landings (t)
1950	1939	12.92
1951	1940	21.42
1952	1941	29.21
1953	1942	25.61
1954	1943	35.10
1955	1944	30.60
1956	1945	18.64
1957	1946	44.60
1958	1947	53.12
1959	1948	55.39
1960	1949	49.65
1961	1950	58.63
1962	1951	48.83
1963	1952	76.32
1964	1953	110.57
1965	1954	84.92
1966	1955	64.39
1967	1956	61.04
1968	1957	77.82
1969	1958	76.07
1970	1959	65.63
1971	1960	75.62
1972	1961	122.09
1973	1962	84.74
1974	1963	99.86
1975	1964	166.29
1976	1965	153.96
1977	1966	186.39
1978	1967	228.70
1979	1968	221.53
1980	1969	164.03
1981	1970	107.90
1982	1971	29.02
1983	1972	75.56
1984	1973	122.40
1985	1974	103.95
1986	1975	116.10
1987	1976	102.90
1988	1977	105.30
1989	1978	121.49
1990	1979	119.04
1991	1980	116.96
1992	1981	123.02
1993	1982	104.96
1994	1983	82.31
1995	1984	62.17
1996	1985	56.75
1997	1986	43.38
1998	1987	21.48
1999	1988	20.52
2000	1989	29.46
2001	1990	28.51
2002	1991	12.35
2003	1992	13.35
2004	1993	0.00
2005	1994	0.00

Table 4.2
Geometric mean number of eels per trawl haul, Bay of Quinte, 1972 to 2004, against survey year and against year of recruitment. Year of recruitment is based on a mean age of 11 years (Hurley 1972, see Table 3.2).

Survey year	Recruitment year	Geometric mean of eels per trawl
1972	1961	1.873
1973	1962	1.620
1974	1963	0.997
1975	1964	1.543
1976	1965	1.286
1977	1966	1.064
1978	1967	0.417
1979	1968	0.767
1980	1969	0.252
1981	1970	1.530
1982	1971	1.884
1983	1972	0.557
1984	1973	0.330
1985	1974	0.778
1986	1975	0.865
1987	1976	1.552
1988	1977	0.299
1989	1978	0.952
1990	1979	0.356
1991	1980	0.454
1992	1981	0.585
1993	1982	0.434
1994	1983	1.157
1995	1984	0.091
1996	1985	0.356
1997	1986	0.085
1998	1987	0.123
1999	1988	0.074
2000	1989	0.053
2001	1990	0.006
2002	1991	0.013
2003	1992	0.000
2004	1993	0.000

Table 4.3
Mean number of eels electrofished per hour, Eastern Basin Lake Ontario, 1984 to 2004, against survey year and against year of recruitment. Year of recruitment is based on a mean age of 11 (Hurley 1972, see Table 2.1).

Survey year	Recruitment year	Eels electrofished per hour
1984	1973	85.60
1985	1974	63.10
1986	1975	82.90
1987	1976	89.00
1988	1977	68.80
1989	1978	93.00
1990	1979	64.10
1991	1980	38.50
1992	1981	44.40
1993	1982	22.70
1994	1983	30.00
1995	1984	10.50
1996	1985	14.90
1997	1986	7.30
1998	1987	12.90
1999	1988	21.60
2000	1989	9.37
2001	1990	6.82
2002	1991	3.36
2003	1992	0.65
2004	1993	0.52

Table 4.4
Mean number of eels per day ascending the ladder at the Moses Saunders dam, Ontario, 1974-2004, during 31 day peak mid-summer migration period[a], against the year of the survey and against the year of recruitment. Year of recruitment is based on a mean age of 6 in the 1970s and 1980s and a mean age of 12 beginning in the 1990s (Table 3.2).

Year	Eels per day against survey year[b]	Eels per day against recruitment year[c]
1968		7934
1969		14403
1970		10363
1971		20013
1972		16448
1973		18977
1974	7934	9046
1975	14403	13473
1976	10363	27489
1977	20013	26103
1978	16448	9074
1979	18977	9868
1980	9046	2828
1981	13473	4755
1982	27489	5220
1983	26103	3233
1984	15051	
1985	18510	144
1986	5380	57
1987	9277	27
1988	5442	52
1989	5795	18
1990	3096	55
1991	1226	40
1992	277	53
1993	232	
1994	4998	
1995	671	
1996		
1997	144	
1998	57	
1999	27	
2000	52	
2001	18	
2002	55	
2003	40	
2004	53	

[a] In 2004, the fall peak (mean 274/day) surpassed the mid-summer peak for the first time. Fall eels are larger thann summer eels.

[b] Data not available for 1996

[c] For years 1978-1993, the ladder index was taken as the mean of that calculated using a mean age of 6 and that calculated with a mean age of 12.

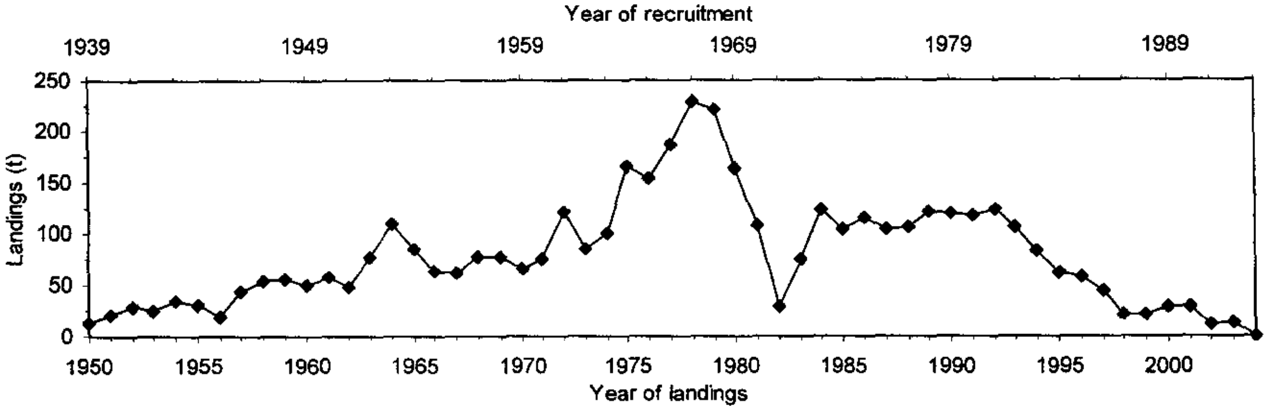


Fig. 4.1
Reported American eel landings in Ontario, against year of landings and against year of recruitment. Year of recruitment is based on a mean age of 11 (Table 3.2).

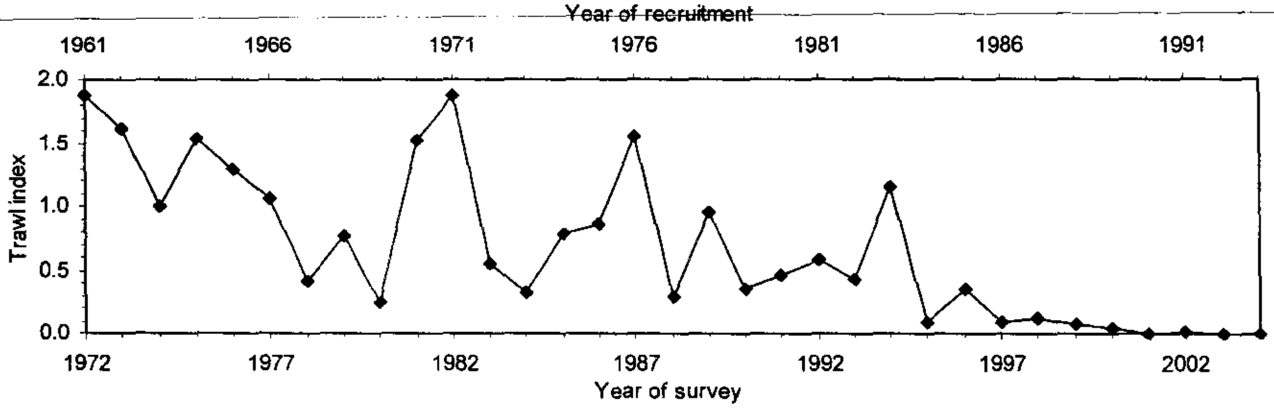


Fig. 4.2
Geometric mean number of eels per trawl haul, Bay of Quinte, 1972 to 2004, against survey year and against year of recruitment. Year of recruitment is based on a mean age of 11 years (Hurley 1972, see Table 3.2).

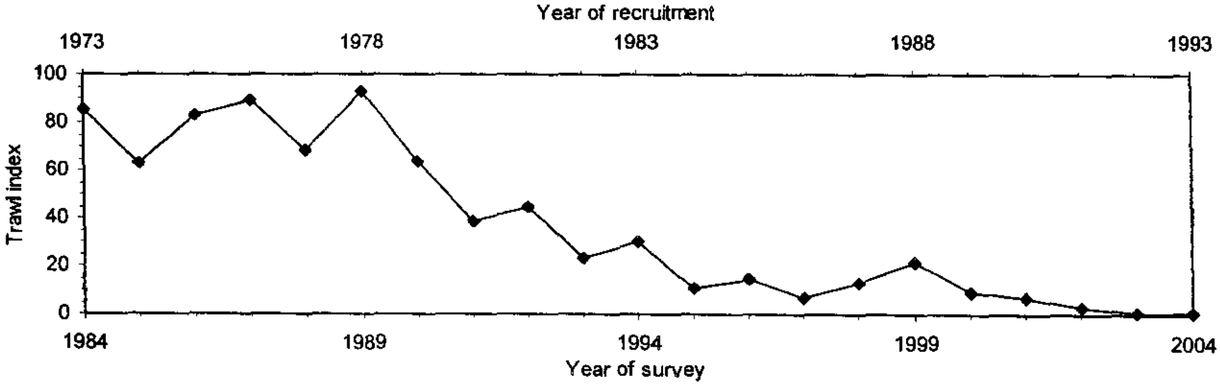


Fig. 4.3
Mean number of eels electrofished per hour, Eastern Basin Lake Ontario, 1984 to 2004, against survey year and against year of recruitment. Year of recruitment is based on a mean age of 11 (Hurley 1972, see Table 2.1).

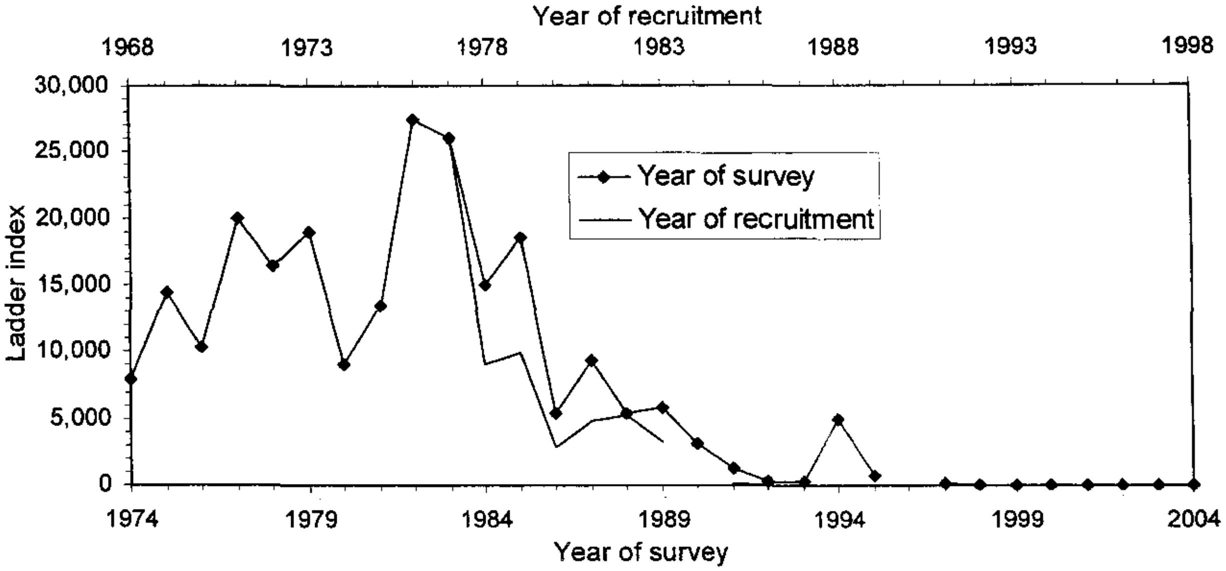


Fig. 4.4
Mean number of eels per day ascending the ladder at the Moses Saunders dam, Cornwall, Ontario, Ontario, 1974-2004, during 31 day peak mid-summer migration period, against the year of the survey and the year of recruitment. Year of recruitment is based on an mean age of 6 in the 1970s and 1980s and a mean age of 12 beginning in the 1990s (Table 3.2).

5. EEL ABUNDANCE INDICATORS IN QUEBEC

5.1 Background

Eel habitat in Quebec is found primarily in DU1 (Western St. Lawrence) and DU2 (Eastern St. Lawrence) (Fig. 3.2). The central and southern part of the Gaspé Peninsula and the Magdalen Islands are in DU3 (Maritimes).

Eels must travel through the eastern St. Lawrence (DU2) to reach rearing habitat in DU1 or US waters in the upper St. Lawrence drainage and Lake Ontario. Eels that leave these upstream waters on their spawning migration must again pass through DU2 on their way to the sea.

5.2 Population indicators

5.2.1 *Commercial fisheries*

Virtually all eels that are commercially fished in Quebec have been reared in DU1 or in US waters adjacent to DU1. Yellow eels are fished in Lake Francis (the headpond of the Beauharnois dam), and a mix of yellow and silver eels is fished in Lake St. Pierre and the Gentilly area of the St. Lawrence River (Fig. 3.2). A very small fishery also exists on the Ottawa River. Downstream migrating silver eels are subject to a major fishery in the St. Lawrence estuary (Fig. 3.3). There is a small fishery in the Magdalen Islands. There are two eel fishing permits for the south shore of the Gaspé Peninsula, but it is not known if these permits are active.

Reported eel landings from 1920 to 2003 are presented in Table 5.1 and 5.1. Silver eels taken in the estuary comprise the bulk of the landings. Reported landings peaked at over 1,000 tonnes in the 1930s. Landings in the estuary fishery were relatively stable from the 1950s to about 1990. Estuary landings have declined by about two thirds since 1990 (Fig. 5.2). Landings of silver eels from Lake St. Pierre and west, have also shown steady decreases since 1990, but landings of yellow eels in Lake St. Pierre and west have been relatively stable.

5.2.2 *Interception indices*

This section covers research gear, including estuary traps and fishway traps, which intercept moving eels.

Eels have been counted at an experimental eel trap at St. Nicholas near Quebec City since the late 1960s (de Lafontaine submitted) (Table 5.3, Fig. 5.3). This trap captures primarily yellow eels during the summer and silver eels in the fall. Fall capture numbers have fluctuated in a climbing trend since about 1990. Summer numbers have declined in approximately linear fashion since the 1970s.

Data on size of eels in the St. Nicholas trap have been collected since 1994. Mean size of fall-caught eels increased from about 77 cm in the mid-1990s to about 87 cm in 2002 (de Lafontaine submitted).

St. Lawrence estuary silver eel landings and fall captures at the St. Nicholas trap are plotted against year of recruitment from the ocean in Figs. 5.1 and 5.2. Eels are assumed to be 17 years old, based on ages of silver eels captured in the estuary in 1970 (Table 3.2). No other ages are available for silver eels from the St. Lawrence estuary. The 17 year de-lag factor must be considered approximate, especially for the St. Nicholas captures which are showing marked increases in size, and presumably age.

Reported landings plotted against recruitment with a 17 year de-lag indicates that the decline in St. Lawrence estuary silver eel landings began about recruitment year 1973 (Fig. 5.2). In the St. Nicholas series, recruitment year 1973 was the approximate start of a period of increase (Fig. 5.3).

Abundance indicators from other interception fishing gear are presented in Table 5.3 and Fig. 5.4. In general these series are of short duration and have variable methodologies which make temporal comparisons difficult.

Counts of eels creeping up rock faces in the Petite rivière de la Trinité, near the mouth of the St. Lawrence, were conducted in the mid 1980s and again in the mid-1990s. Mean counts in the two periods were similar (Fig. 5.4).

Yellow eels ascending the Sud-Ouest River were counted visually as they climbed a vertical rock face at a waterfall in 1996 and 1998. These counts represent an unknown proportion of the total upstream run. In 1999, a fishway with a counting trap was installed, which captured all ascending eels. Counts in 1996-1998 (16,617, 2,280) were much higher than those of 1999-2004 (285-570). The decline is considered to be real despite the change in methodology, because the fishway trap is a more efficient counting method than the visual counts.

Yellow eels counted at the Chambly fish ladder on the Richelieu River were high in the first two years of ladder operation, and then fell sharply (Fig. 5.4). This pattern has been attributed to pent-up demand by eels that had accumulted below the dam for access to upstream waters (Caron et al. submitted).

5.2.3 *Electrofishing*

Electrofishing surveys are conducted in numerous Quebec rivers to assess populations of salmonids, particularly Atlantic salmon. Counts of eels recorded as bycatch in these surveys may serve as indicators of eel abundance.

Electrofishing surveys in the Saint-Jean, Trinité, and Bec-Scie Rivers have been conducted with three sweeps per site. Because of low eel captures, it is generally not possible to estimate densities by the depletion method. Data are therefore presented as summed counts over the three sweeps per 100 m^2 (Tables 5.4-5.6, Fig. 5.5). The indices show high interannual variation, and no long-term trends are evident.

Eel indices have also been calculated for numerous other rivers in eastern Quebec (Table 5.7). Analysis methods for these data series have not yet been documented. In some cases methods changed in the course of the series.

Table 5.1

Reported landings (metric tonnes) of American eels in Quebec, by region and by life stage. Data from Faune Québec

Year	Lake St. Francis yellow	St. Lawrence Montreal yellow	Richelieu River Silver	Lake St. Pierre			St. Lawrence estuary							All Quebec		
				Yellow	Silver	Total	Fluvial estuary			Upper estuary silver	Lower estuary silver	Total silver	Total	Total yellow	Total silver	Total
							Yellow	Silver	Total							
1920		4.4	53.8	1.5	3.2	4.7	2.2	153.4	155.6	54.7	1.4	209.5	211.7	8.1	266.5	274.6
1921	0.5	3.5	52.7	3.7	8.0	11.7	2.1	150.3	152.4	86.5	2.8	239.5	241.7	9.8	300.3	310.1
1922	0.5	15.6	49.7	3.0	6.6	9.6	5.0	349.4	354.4	20.7		370.2	375.1	24.1	426.4	450.5
1923	0.8	43.9	39.7	4.5	9.7	14.2	6.1	432.1	438.3	18.4		450.5	456.7	55.2	500.0	555.2
1924	0.9	70.1	40.6	3.9	8.4	12.3	5.1	356.8	361.9	44.3	6.6	407.7	412.8	79.9	456.8	536.7
1925		33.6	47.3	6.7	14.5	21.2	4.9	342.5	347.4	77.9	4.7	425.1	430.0	45.2	486.9	532.1
1926	1.9	31.6	54.6	8.0	17.3	25.3	10.1	714.6	724.8	104.5	9.3	828.5	838.6	51.6	900.3	951.9
1927	1.3	32.0	45.4	9.8	21.2	31.0	5.8	411.3	417.1	41.6	30.5	483.4	489.2	48.9	550.0	598.9
1928	1.6	42.5	46.0	13.3	29.0	42.3	10.9	765.7	776.5	42.4	30.4	838.4	849.3	68.3	913.5	981.8
1929	0.9	25.2	42.5	15.7	34.1	49.8	4.8	335.6	340.3	44.5	17.5	397.5	402.3	46.6	474.1	520.7
1930	4.5	39.3	38.0	17.4	37.9	55.3	5.2	367.8	373.0	43.4	24.0	435.1	440.4	66.5	511.0	577.5
1931	1.0	42.0	48.5	25.6	55.7	81.4	7.4	523.2	530.6	51.5	13.8	588.5	595.9	76.1	692.8	768.8
1932	1.0	28.7	56.7	51.3	111.6	162.9	7.2	505.7	512.9	55.4	25.4	586.5	593.7	88.1	754.8	842.9
1933	1.3	32.3	59.6	52.8	114.8	167.6	10.2	717.1	727.3	71.0	28.7	816.8	827.0	96.6	991.2	1087.8
1934	0.9	34.4	63.7	58.5	127.3	185.8	9.0	631.8	640.8	53.8	28.9	714.5	723.5	102.8	905.5	1008.3
1935	0.8	32.5	64.0	58.7	127.6	186.3	9.0	631.4	640.4	55.3	28.5	715.3	724.2	100.9	906.9	1007.9
1936	0.8	31.6	64.0	55.6	120.9	176.5	8.6	603.5	612.0	55.4	28.6	687.5	696.0	96.6	872.4	968.9
1937	0.7	14.5	72.6	33.7	73.3	107.0	7.7	542.1	549.8	49.9	30.2	622.1	629.8	56.6	768.0	824.6
1938	0.9	15.4	74.8	35.0	76.1	111.1	7.7	544.7	552.5	52.6	29.3	626.6	634.3	59.1	777.6	836.6
1939	1.4	14.3	72.6	14.6	31.7	46.3	7.8	550.6	558.4	49.9	25.0	625.4	633.2	38.1	729.7	767.8
1940	0.9	4.5	54.4	12.0	26.1	38.1	3.8	269.5	273.3	9.1	18.1	296.7	300.5	21.3	377.2	398.5
1941	0.7			15.1	32.9	48.1				12.2	9.5	21.8	21.8	15.8	54.7	70.5
1942	1.1	4.1		16.7	36.3	53.0	5.4	381.3	386.7	12.0	11.3	404.6	410.0	27.4	440.9	468.3
1943	2.0	15.0		26.6	57.9	84.6	7.1	496.9	503.9	11.5	18.1	526.6	533.6	50.7	584.5	635.2
1944	1.1	4.2		12.0	26.0	38.0	3.1	221.3	224.5	13.0	11.8	246.1	249.2	20.5	272.1	292.5
1945	1.9	6.3	23.0	27.3	59.3	86.6	3.7	261.4	265.1	1.0	0.9	263.4	267.1	39.2	345.7	384.9
1946	1.0	3.2	21.7	15.9	34.6	50.4	3.4	236.1	239.5	5.2		241.3	244.7	23.4	297.6	321.0
1947	0.9	2.6	1.0	15.0	32.6	47.6	3.7	263.0	266.7	2.3	0.7	265.9	269.7	22.2	299.5	321.6
1948	0.3	1.5	8.8	7.5	16.2	23.7	2.4	172.0	174.4	13.3		185.3	187.7	11.8	210.3	222.1
1949	0.4	1.6	22.4	5.9	12.9	18.9	1.4	98.3	99.7	40.4	1.0	139.7	141.1	9.3	175.0	184.3
1950	0.4	1.8	33.3	4.9	10.7	15.6	2.9	203.2	206.1	39.5	2.1	244.8	247.7	10.0	288.8	298.8
1951	0.4	1.2	24.2	6.5	14.0	20.5	3.5	245.8	249.3	54.1	0.6	300.6	304.0	11.6	338.8	350.4
1952	0.5	1.3	12.4	7.0	15.2	22.2	4.2	292.5	296.6	57.2	0.9	350.5	354.6	12.9	378.1	391.0
1953	0.5	0.7	18.6	7.2	15.6	22.8	4.2	293.0	297.2	59.6	3.3	356.0	360.2	12.5	390.1	402.6
1954	0.9	0.8	12.6	9.5	20.7	30.2	3.4	236.6	239.9	56.3	11.5	304.3	307.7	14.5	337.6	352.1
1955	1.1	0.9	16.9	13.1	28.6	41.7	3.8	267.4	271.2	59.2	9.5	336.1	339.9	18.9	381.5	400.4

Table 5.1 (continued)

Year	Lake St. Francis yellow	St. Lawrence Montreal yellow	Richelieu River Silver	Lake St. Pierre			St. Lawrence estuary							All Quebec		
				Yellow	Silver	Total	Fluvial estuary			Upper estuary silver	Lower estuary silver	Total silver	Total	Total yellow	Total silver	Total
							Yellow	Silver	Total							
1956	1.2	1.4	28.3	8.0	17.4	25.4	3.6	251.6	255.1	78.6	4.1	334.3	337.9	14.2	380.0	394.2
1957	1.0	0.9	30.2	7.4	16.1	23.5	5.7	404.2	410.0	92.9	2.0	499.2	504.9	15.0	545.5	560.4
1958	1.2	0.8	18.1	16.8	36.6	53.5	3.8	266.3	270.1	131.5	2.0	399.7	403.5	22.6	454.4	477.0
1959	0.5	0.6	33.1	16.6	36.1	52.8	2.8	195.8	198.5	94.3	8.4	298.5	301.3	20.5	367.7	388.2
1960	1.8	0.7	19.1	15.2	33.0	48.2	2.6	180.7	183.3	205.8	3.2	389.7	392.3	20.3	441.8	462.1
1961	2.0	1.6	17.0	17.1	37.1	54.2	2.0	137.7	139.7	164.2	2.8	304.7	306.6	22.7	358.8	381.5
1962	1.8	6.3	26.3	19.5	42.5	62.0	1.4	100.4	101.8	181.3	3.8	285.5	286.9	29.1	354.2	383.3
1963	1.0	1.2	35.5	24.4	53.0	77.4	2.2	153.9	156.1	191.9	5.2	351.1	353.3	28.7	439.6	468.2
1964	0.3		44.4	28.3	61.5	89.7	1.6	110.6	112.2	196.5	4.0	311.1	312.7	30.2	417.0	447.2
1965	0.1	0.1	51.6	27.6	59.9	87.5	2.1	148.6	150.7	253.9	3.9	406.4	408.5	29.9	517.9	547.8
1966	0.4		49.9	26.5	57.6	84.1	1.5	108.8	110.3	235.2	8.0	351.9	353.5	28.5	459.4	487.9
1967	0.8	0.1	37.2	25.5	55.5	81.1	0.9	64.7	65.6	227.5	23.3	315.5	316.4	27.3	408.2	435.5
1968	0.5		36.0	28.4	61.7	90.0	1.4	100.0	101.4	204.7	65.2	369.9	371.3	30.2	467.5	497.8
1969	0.4		22.8	26.1	56.8	83.0	1.1	75.8	76.8	245.9	83.8	405.5	406.6	27.6	485.1	512.7
1970	0.3		1.1	9.0	19.6	28.7	0.4	31.7	32.1	170.4	81.1	283.1	283.6	9.8	303.9	313.6
1971	0.0		24.4	5.7	12.3	17.9	0.2	13.5	13.7	165.0	92.2	270.7	270.9	5.8	307.4	313.2
1972	1.3	0.4	7.6	28.2	61.2	89.4	0.3	21.9	22.3	153.9	34.0	209.8	210.1	30.2	278.6	308.8
1973	0.7		1.0	21.4	46.5	67.8	0.3	20.7	21.0	138.3	72.0	231.0	231.3	22.3	278.5	300.8
1974	0.6		34.1	27.6	60.0	87.6	0.3	18.6	18.8	190.7	56.9	266.1	266.4	28.4	360.2	388.6
1975	1.1	0.2	41.4	25.2	54.8	80.0	0.5	33.3	33.8	303.9	64.5	401.7	402.1	27.0	497.8	524.9
1976	2.3	0.0	20.0	31.6	68.6	100.2	0.3	20.3	20.6	222.4	52.5	295.1	295.4	34.1	383.8	417.9
1977	0.1	0.1	47.3	23.7	51.6	75.3	0.3	21.1	21.4	282.4	80.3	383.8	384.1	24.2	482.7	506.9
1978	0.3		37.2	28.2	61.3	89.4	0.3	22.7	23.0	295.6	79.9	398.1	398.5	28.7	496.6	525.4
1979	1.1		43.0	26.6	57.9	84.5	0.5	32.1	32.5	256.1	88.2	376.3	376.8	28.2	477.2	505.3
1980	0.9		66.3	24.0	52.2	76.2	0.4	25.5	25.9	314.6	111.5	451.7	452.1	25.2	570.2	595.4
1981	2.1		72.9	28.4	61.7	90.1	0.5	33.0	33.5	320.6	80.9	434.5	434.9	30.9	569.1	600.1
1982	1.6		48.9	22.5	49.0	71.6	0.4	27.1	27.5	153.3	78.4	258.8	259.2	24.6	356.6	381.2
1983	0.9		33.0	23.4	50.8	74.2	0.3	22.5	22.8	174.9	46.0	243.3	243.7	24.6	327.2	351.7
1984	2.4		21.9	28.2	61.3	89.5	0.2	16.1	16.4	253.2	28.4	297.7	297.9	30.8	380.9	411.7
1985			47.5							249.0	93.0	342.0	342.0	0.0	389.5	389.5
1986	4.6		48.1	22.6	49.1	71.7	0.8	57.0	57.9	229.6	84.9	371.6	372.4	28.0	468.8	496.8
1987	4.2		38.0	15.8	34.4	50.1	0.8	53.9	54.7	180.8	97.7	332.4	333.2	20.8	404.8	425.6
1988	5.2		34.4	14.0	30.4	44.4	0.9	61.9	62.8	208.5	78.8	349.3	350.2	20.0	414.1	434.1
1989	9.2		26.2	17.9	38.9	56.7	0.8	58.3	59.1	203.7	78.0	340.0	340.9	27.9	405.1	433.0
1990	15.7		19.0	16.8	36.6	53.4	1.3	88.8	90.0	232.7	62.9	384.4	385.7	33.8	440.0	473.7

Table 5.1 (continued)

Year	Lake St. Francis yellow	St. Lawrence Montreal yellow	Richelieu River Silver	Lake St. Pierre			St. Lawrence estuary							All Quebec		
				Yellow	Silver	Total	Fluvial estuary			Upper estuary silver	Lower estuary silver	Total silver	Total	Total yellow	Total silver	Total
							Yellow	Silver	Total							
1991	12.7		21.9	15.9	34.5	50.4	0.7	50.1	50.8	197.6	62.1	309.8	310.5	29.2	366.2	395.5
1992	6.8		19.7	13.7	29.7	43.4	1.0	68.6	69.6	118.7	60.9	248.2	249.2	21.4	297.6	319.1
1993	2.2		14.1	16.5	35.8	52.2	0.8	55.3	56.1	149.3	54.6	259.1	259.9	19.5	309.0	328.4
1994	5.2		8.4	14.6	31.7	46.3	0.9	60.2	61.1	116.2	45.0	221.4	222.3	20.7	261.5	282.2
1995	10.0		12.6	11.7	25.5	37.2	1.1	74.7	75.7	91.6	51.1	217.4	218.4	22.8	255.4	278.2
1996	14.1		2.1	14.4	31.3	45.8	0.8	55.7	56.5	76.1	31.1	162.9	163.7	29.3	196.3	225.6
1997	15.6		4.7	10.2	22.2	32.4	0.7	47.9	48.6	77.9	22.0	147.8	148.5	26.5	174.7	201.1
1998	10.6			10.9	23.7	34.5	0.8	53.9	54.7	98.7	16.6	169.2	170.0	22.3	192.9	215.1
1999	10.0			8.2	17.7	25.9	0.7	46.4	47.0	68.4	16.0	130.8	131.5	18.8	148.5	167.4
2000	23.3			12.4	27.0	39.4	0.8	53.0	53.8	72.4	13.4	138.8	139.6	36.5	165.8	202.3
2001	25.2			8.1	17.7	25.8	0.7	50.7	51.4	69.9	11.1	131.7	132.4	34.0	149.4	183.4
2002	23.6			9.6	21.0	30.6	0.6	43.0	43.6	65.7	6.5	115.2	115.8	33.8	136.2	170.0
2003	23.6			6.2	13.4	19.6	0.5	33.2	33.7	47.9	12.8	94.0	94.4	30.2	107.4	137.6

Table 5.2
Number of eels caught per day in the experimental trap at St. Nicholas, on the St. Lawrence River near Quebec City. Most eels caught in summer are yellow and most eels caught in fall are silver. Recruitment year for silver eels is based on a mean age of 17 (Table 3.2). Data from de Lafontaine submitted.

Survey year	Recruitment year for silver eels	Number of eels caught Summer (15 May-31 Aug)	Fall (1 Sep-31 Oct)	Total
1968	1951		193	
1969	1952			
1970	1953			
1971	1954		414	581
1972	1955		297	597
1973	1956		225	425
1974	1957		209	383
1975	1958	115	232	347
1976	1959	246	194	440
1977	1960	133	328	461
1978	1961	134	449	583
1979	1962	128	273	401
1980	1963	90	187	277
1981	1964	143	176	319
1982	1965	100	199	299
1983	1966	115	234	349
1984	1967	147	166	313
1985	1968	129	200	329
1986	1969	136	176	312
1987	1970	83	166	249
1988	1971	69	207	276
1989	1972	104	83	187
1990	1973	87	160	247
1991	1974	48	169	217
1992	1975	89	177	266
1993	1976	78	188	266
1994	1977	103	200	303
1995	1978	45	208	253
1996	1979	103	127	230
1997	1980	74	138	212
1998	1981	93	205	298
1999	1982	69	381	450
2000	1983	112	190	302
2001	1984	67	350	417
2002	1985	90	239	339
2003	1986	78	257	335
2004	1987	78	200	278
2005	1988	34		

Table 5.3

Abundance indicators for small eels, yellow eels, and silver eels in Quebec. Data from Caron et al. submitted.

Year	Small eels		Yellow eels					Migrating silver eels				
	Petite Trinité		Sud-Ouest		Richelieu	Beauharnois[c]		St. Lawrence, upstream from Quebec City	Sud Ouest		Petite Trinité	Saint-Jean
	Visual counts of eels creeping up rocks	CMR[a] estimate	Visual count of eels ascending a rock face	Fishway trap at falls	Chambly Ladder count	West eel pass at Group 37	Total, all passes	CMR estimate	Fish fence count with gaps during the season	Full fish fence count	CMR[a] Estimate	Partial estimate
Mean sizes (cm)	12-15		20-29		33	37-47			98-111		63-65	
Age	1-2		2-10		4-12	4-12						
1982	4,027											
1983	3,643											
1984	732											
1985	581											
1993	1,178											
1994	488		16,617			24,721	52,483					
1995	3,440					17,072						
1996	3,550		2,280					488,000	214			
1997					10,863 [b]			397,000				
1998					9,875	5,441						
1999		13,912		407	3,685	10,692				315	2,309	
2000		19,829		285	239	6,881			34		3,019	
2001		17,534		435	359	13,099				108	2,855	
2002					240	10,503	43,111			68		
2003				570	3,336	32,684	59,969			60		
2004				407	727		58,586		108			2,004

[a] Capture-mark-recapture

[b] 1997 was the first year of ladder operation. Most eels in 1997 were released below the dam to test the efficiency of the eel pass.

[c] In 1994, 4 eel fishways with traps were installed. The trap at Beauharnois, west side, Group 37, caught 24,721 eels. The trap at Beauharnois, west side, Central 3, caught 9,332 eels. The trap at Beauharnois, east side, Central 1, caught 15,134 eels. The trap at the Pointe-des-Cascades caught 3,186 eels. The fishway trap at Group 37 was operated in 1995 and 1998-2001. It was replaced by a permanent eel pass in 2002. An eel fishway with a trap was installed on the east side of the dam in 2002. This was replaced by a permanent eel pass in 2004.

Table 5.4
Numbers of eels caught in electrofishing sweeps and mean number of eels captured per 100 m^2 in the Saint-Jean River, 1975-2000.

Year	Eels counted in Sweep				No. sites	Site areas (m^2)	Eels captured 100 m^{-2}
	1	2	3	Not avail.[a]			
1975						± 100	0.20
1976							
1977				1	20	50-150	0.05
1978				3	19	50-180	0.16
1979							
1980							
1981							
1982							
1983							
1984							
1985							
1986							
1987							
1988	1	0	0		12	105	0.08
1989	3	2	3		13	100	0.62
1990	2	0	0		12	100	0.17
1991	0	0	0		12	100	0.00
1992	2	1	0		12	100	0.25
1993							
1994							
1995							
1996							
1997							
1998							
1999							
2000				2	13	100	0.15

Table 5.5
Numbers of eels caught in electrofishing sweeps and mean number of eels captured per 100 m^2 in the Trinité River, 1984-1992.

Year	Closed stations						Open stations						All stations					
	Sweep			No. sites	Site areas (m2)	Eels captured 100 m^{-2}	Sweep			No. sites	Site areas (m2)	Eels captured 100 m^{-2}	Sweep			No. sites	Site areas (m2)	Eels captured 100 m^{-2}
	1	2	3				1	2	3				1	2	3			
1984	0	0	1	14	100	0.07	0	2	0	53	100	0.04	0	2	1	67	100	0.04
1985	3	0	0	19	100	0.16	3	10	11	49	100	0.49	6	10	11	68	100	0.40
1986	0	0	0	11	100	0.00	4	5	10	46	100	0.41	4	5	10	57	100	0.33
1987	1	0	1	16	100	0.13	3	7	0	53	100	0.19	4	7	1	69	100	0.17
1988	1	0	0	11	100	0.09	3	6	3	53	100	0.23	4	6	3	64	100	0.20
1989	2	0	0	11	100	0.18	9	8	4	53	100	0.40	11	8	4	64	100	0.36
1990	0	0	0	11	100	0.00	6	3	4	52	100	0.25	6	3	4	63	100	0.21
1991	1	0	0	10	100	0.10	4	6	4	52	100	0.27	5	6	4	62	100	0.24
1992	0	0	0	4	100	0.00	2	4	0	28	100	0.21	2	4	0	32	100	0.19

Table 5.6
Numbers of eels caught in electrofishing sweeps and mean number of eels captured per 100 m^2 in the Bec-Scie River and its tributary Ruisseau Castor, Anticosti Island, 1985-1996.

Year	Bec-Scie							Ruisseau Castor							All sites						
	Eels counted in Sweep				No. sites	Site areas (m^2)	Eels captured 100 m^{-2}	Eels counted in Sweep				No. sites	Site areas (m^2)	Eels captured 100 m^{-2}	Eels counted in Sweep				No. sites	Site areas (m^2)	Eels captured 100 m^{-2}
	1	2	3	Not avail.[a]				1	2	3	Not avail.				1	2	3	Not avail.			
1985	17	0	0		6	96-110	2.73								17	0	0		6	96-110	2.73
1986																					
1987	25	0	0		5	91-108	4.02								25	0	0		5	91-108	4.02
1988				200	9	98-108	22.20				75	7	96-375	10.70				275	16	96-375	17.20
1989				143	9	98-108	15.90				25	7	96-150	3.60				168	16	96-150	10.50
1990	88	6			10	96-104	9.40	24	-	-		7	96-104	3.42	112	6	-		16	96-104	6.94
1991				137	9	96-105	15.20				19	7	96-105	2.70				156	16	96-105	9.80
1992				77	9	96-104	9.40				10	7	99-108	1.40				87	16	96-108	5.40
1993																					
1994				241	9	98-110	26.80				13	7	98-108	1.90				254	16	98-110	15.90
1995				95	9		10.60				15	7		2.10				110	16		6.90
1996				223	9		24.80				8	7		1.10				231	16		14.40

[a] Data files do not break down eel counts by sweep number.

Tablle 5.7
Indices of eel abundance from electrofishing surveys in Quebec, 1971-2004. Data from Caron et al. submitted

Year	Bonaventure		Cascapédia		Petite Cascapédia		Dartmouth		York		Grande Rivière		Grande Rivière Est		Grande Rivière Ouest		Petit Pabos		Grand Pabos Nord		Grand Pabos Ouest		Saint-Anne		Cap-Chat	
	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index
1971																										
1972							n.d		n.d	0.3															n.d	0
1973																										
1974					n.d	0																	n.d	0	n.d	0
1975					n.d	0.2			n.d	0.1													n.d	0	n.d	0.2
1976					n.d	0.1	n.d		n.d	0.3													n.d	0	n.d	0.3
1977	n.d.				n.d	0			n.d	0.1	n.d	0.9	n.d	0	n.d	1.5							n.d	0.1	n.d	0.8
1978			n.d.	0.2	n.d	0	n.d		n.d	0.1	n.d	0.4	n.d	0.2	n.d	1.1							n.d	0.1		
1979																										
1980																										
1981																							11	0.2	13	0
1982																							11	0.2	13	0.1
1983																							10	0.2	13	0.1
1984																							10	0.3	13	0.3
1985																							10	0.2	12	0.4
1986																							10	0.2	12	0.5
1987																							10	0.4	13	0.4
1988																							13	0.1	10	0.1
1989																									10	0
1990	25	0.00	17	0.06	32		10		12		5		4		5		9		8	0.00	14	0.00	13	0.15	10	0.31
1991	25	0.00	17	0.12	32				10	0.80	6		4	0.30	6	0.20	8		14	0.00	8	0.00			8	0.38
1992	25	0.04	17	0.06	32						20														6	0.00
1993	25	0.00	16		33						20	0.20													23	0.09
1994					33		4	0.5																		
1995																										
1996																										
1997																										
1998					64																					
1999	38	0.00							12	0.40							10		11	0.00						
2000	25	0.00	17		32		11	0.40	14								11		15	0.06	8	0.00	13	0.00	28	0.16
2001	25	0.00	17		33		11	0.09	14								10	0.10	11	0.10			13	0.00	23	0.05
2002																			15	0	13	0.1	7	0	11	0
2003																										
2004																										

Table 5.7 (continued)

Year	Madeleine		Sud-Ouest		Ouelle		Rimouski		Mitis		Matane		Petite Matane		Port-Daniel Nord		Port-Daniel du milieu		Petite rivière Port-Daniel		Causapscal		Humqui		Patapédia	
	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index	N	Index
1971	n.d								n.d	0																
1972	n.d								n.d	0					n.d	0					n.d	-				
1973	n.d								n.d	0			n.d	0	n.d	2.3					n.d	0	n.d	8.6		
1974	n.d								n.d	0.2			n.d	0	n.d	0.3					n.d	0	n.d	2.1		
1975	n.d								n.d	0.4			n.d	0.2	n.d	0.4					n.d	0	n.d	2.3	n.d	0.3
1976	n.d								n.d	0.4			n.d	0.1	n.d	0	n.d	0			n.d	0.1	n.d	2.7	n.d	0
1977	n.d								n.d	0.7	n.d	0.3	n.d	0.1	n.d	0.4			n.d	1.1	n.d	0.1	n.d	1	n.d	0
1978	n.d								n.d	0.3			n.d	0							n.d	0.2	n.d	0.9	n.d	0
1979																										
1980																										
1981																										
1982	20	0	6	13			4	3.6																		
1983	20	0.1			10	0	4	5.9																		
1984					11	0.09	4	1.6																		
1985	20	0.3	6	8	11	0	4	1.3																		
1986	24	0.3			15	0	4	2.4																		
1987	24	0.2			15	0	4	0.4																		
1988	24	0			16	0																				
1989	22	0			16	0																				
1990					17	0.00	5	0.20																		
1991					17	0.00	5	1.70																		
1992					17	0.00																				
1993					17	0.00	5	1.20																		
1994							3	1.7																		
1995																										
1996																										
1997																										
1998							24	0.3																		
1999			6	10.00	17	0.00																				
2000	23	0.40	6	4.00	17	0.00																				
2001	21	0.10			17	0.00																				
2002	9	0			17	0																				
2003																										
2004					18	0																				

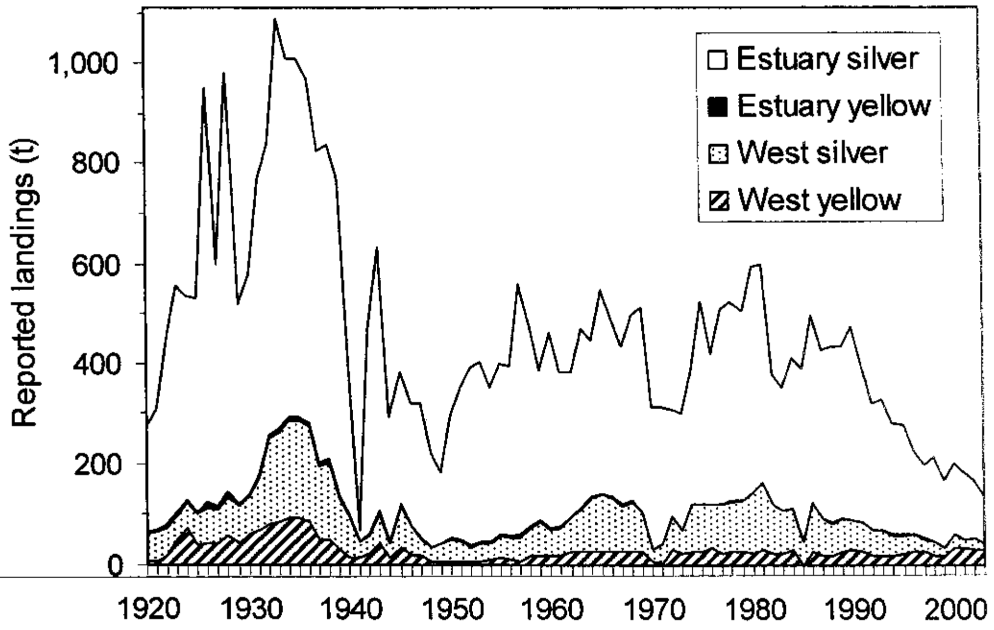


Fig. 5.1
Reported landings of American eels in West Quebec (Lake St. Pierre and west) and in the St. Lawrence estuary.

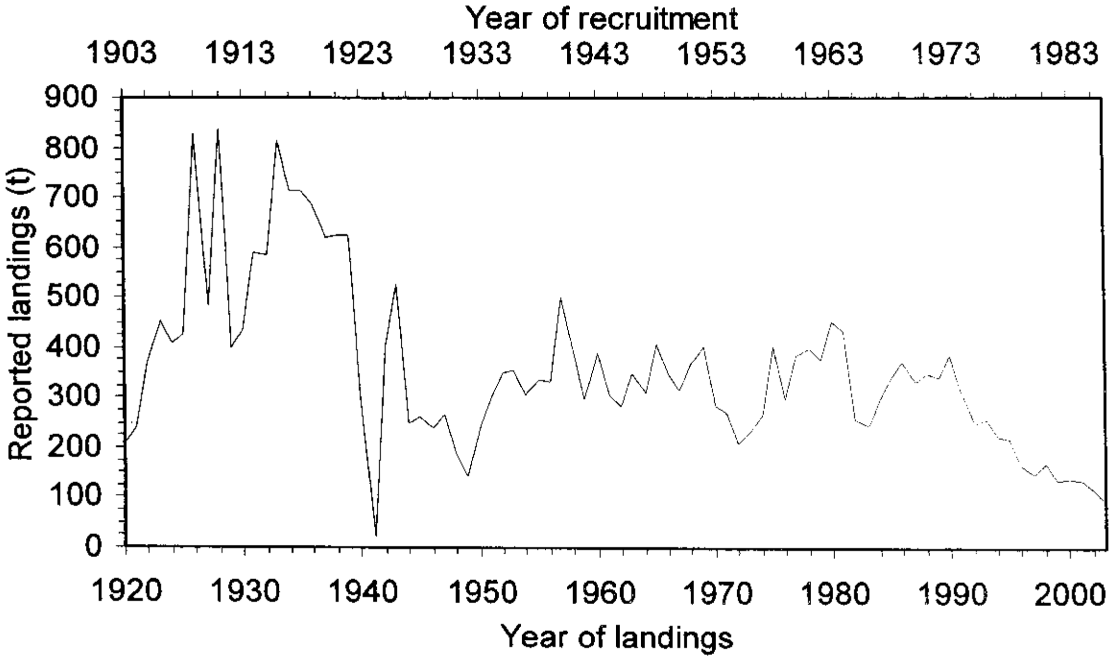


Fig. 5.2
Reported landings of silver American eels in the St. Lawrence estuary, against year of landings and against year of recruitment. Year of recruitment is based on a mean age of 17 (Table 3.2).

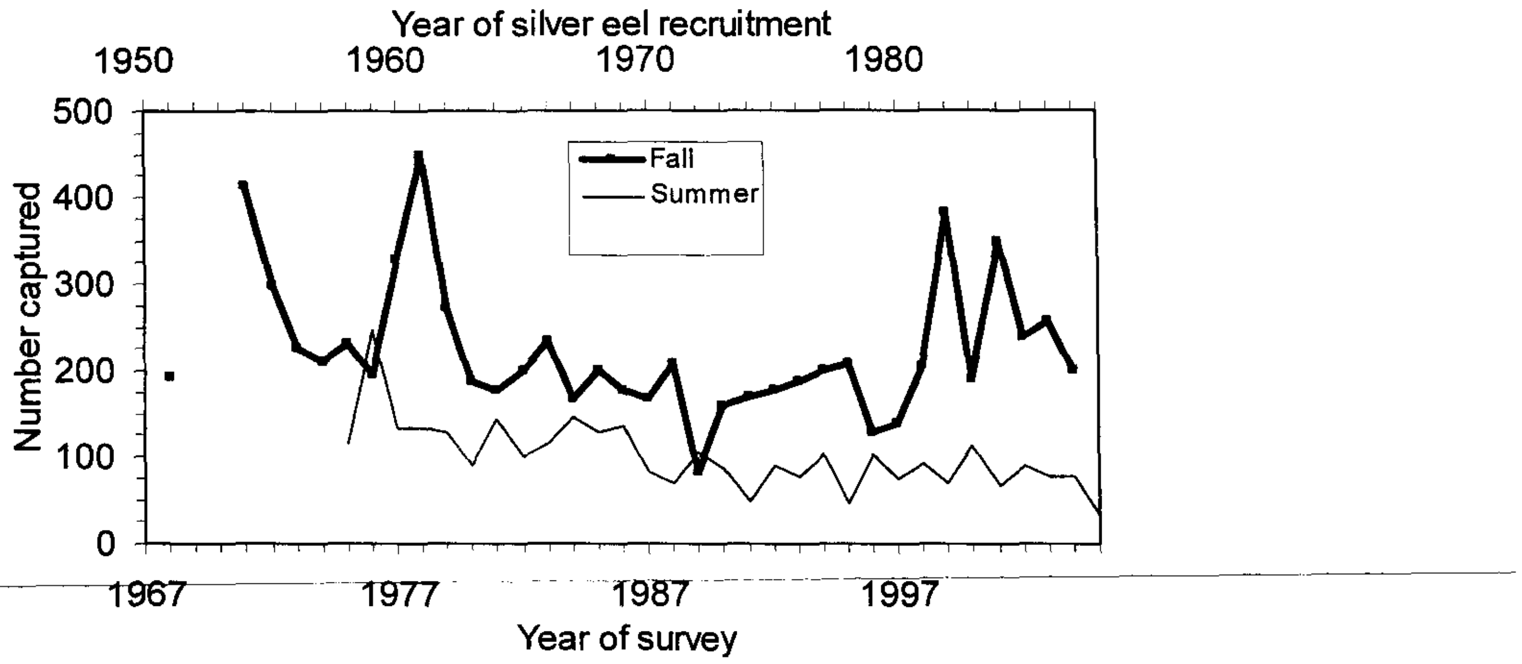


Fig. 5.3
Number of eels caught per day in summer (15 May-31 Aug) and fall (1 Sep-31 Oct) in the experimental trap at St. Nicholas, on the St. Lawrence River near Quebec City. Most eels caught in summer are yellow and most eels caught in fall are silver. Recruitment year for silver eels is based on a mean age of 17 (Table 3.2).

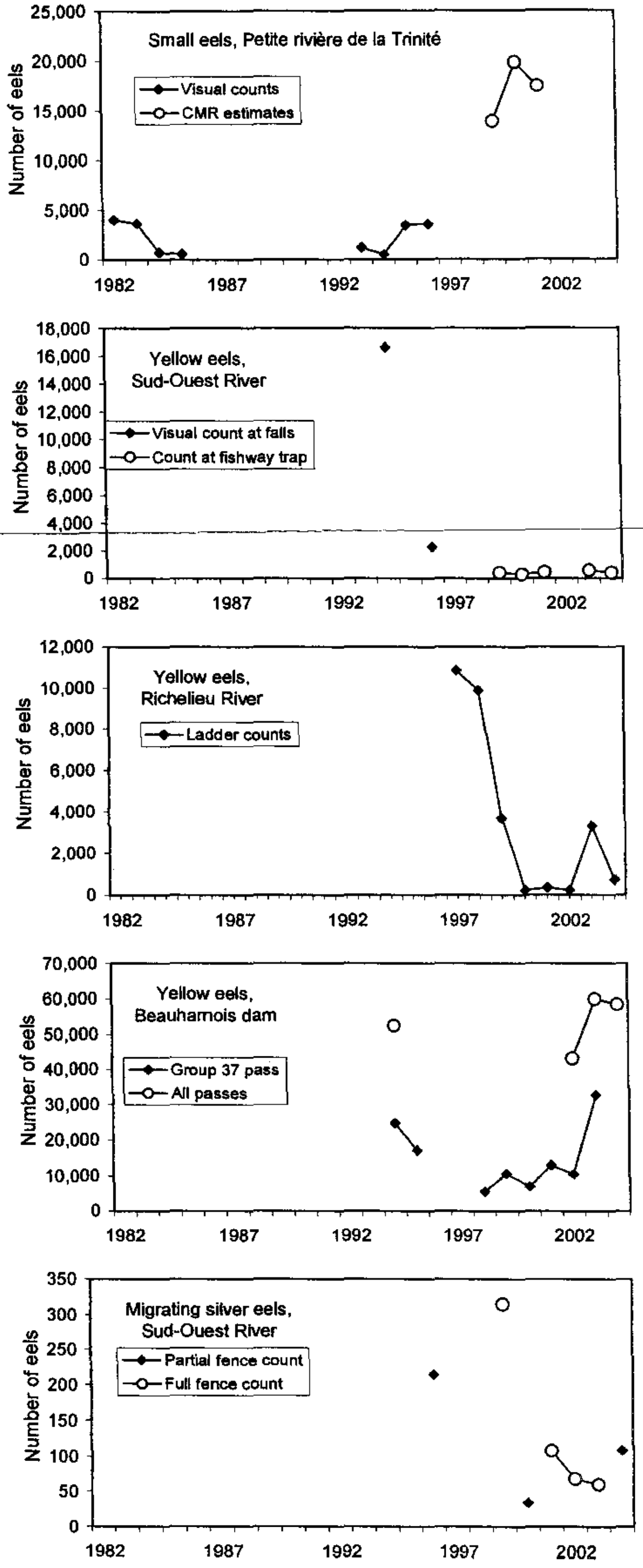


Fig. 5.4
Abundance indicators for small eels, yellow eels, and silver eels in Quebec. Data from Caron et al. submitted.

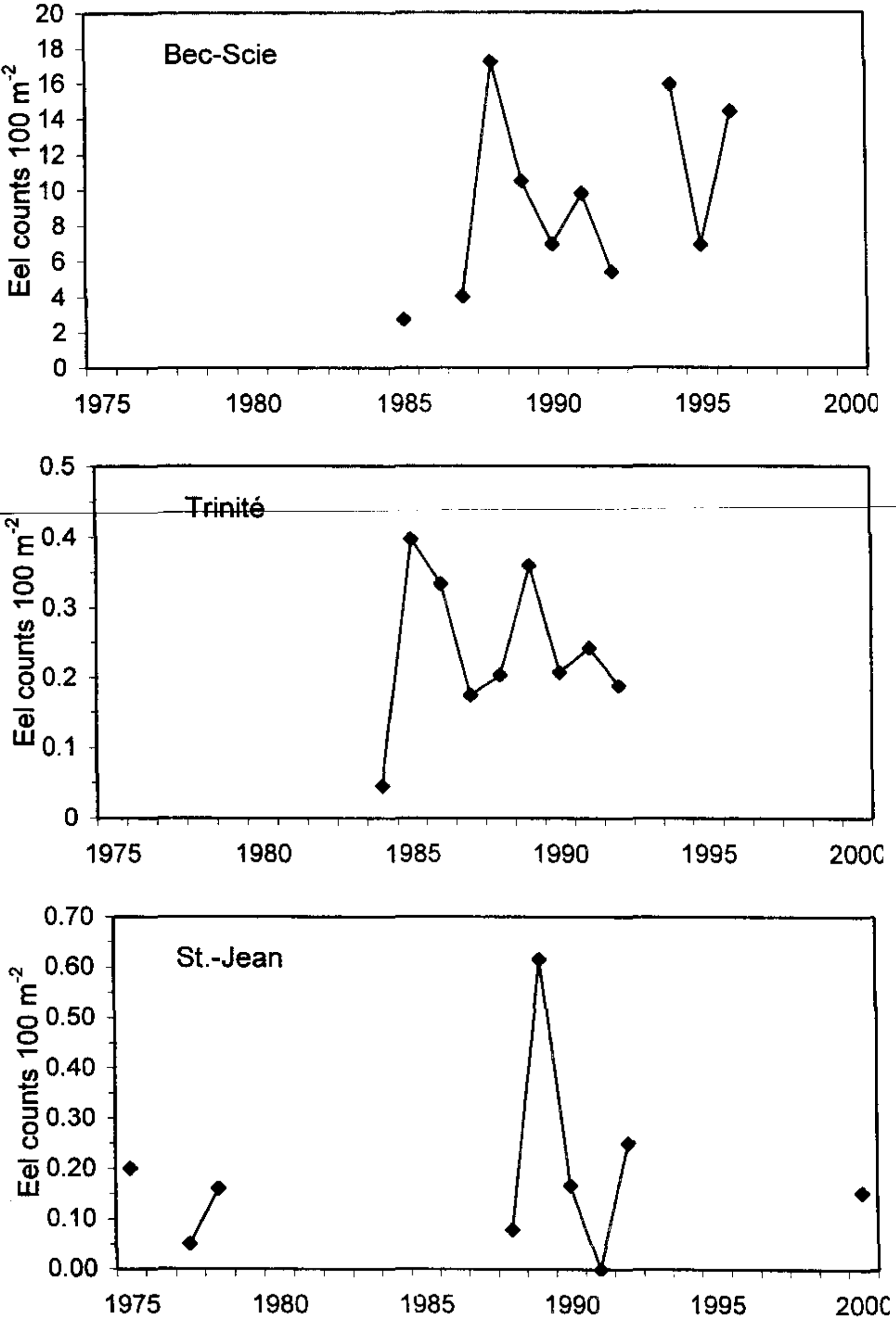


Fig. 5.5
Numbers of eels caught in 3 electrofishing sweeps per 100 m^2 in the Bec-Scie River (including Ruisseau Castor), the Trinité River, and the St. Jean River.

6. EEL ABUNDANCE INDICATORS IN THE SOUTHERN GULF OF ST. LAWRENCE

6.1 Background

The southern Gulf area consists of the Gulf of St. Lawrence drainages of New Brunswick and Nova Scotia, and Prince Edward Island. The southern Gulf area is grouped with the Scotia-Fundy area, and part of the Gaspé Peninsula, to form DU3. The southern Gulf is the only major part of the eel's range in Canada to be nearly devoid of hyro-electric dams.

6.2 Population indicators

6.2.1 *Commercial landings*

The main commercial fisheries in this area are prosecuted in tidal waters of eastern New Brunswick and the north and east coasts of Prince Edward Island (Fig. 3.3). The only commercial freshwater fishery is in western Cape Breton Island. There is little or no fishery in the Bay of Chaleur, in much of Gulf Nova Scotia, and in the western and central parts of the south shore of Prince Edward Island.

Eel landings from 1917 to 1988 were obtained from LeBlanc and Chaput (1991). Data for subsequent years were obtained from DFO Statistics Branch. Licenced fish buyers are required to complete purchase slips for all eels they purchase, and to provide these records to DFO Statistics Branch. Fisheries officers and, in recent years, statistical officers, estimate the amount of eels landed which are not recorded on purchase slips. Such landings include eels retained for personal use or sold to unlicenced buyers, recreational landings, and eels taken illegally. These estimated landings are recorded on forms known as Supplementary B slips. DFO landings statistics include both purchase slip data and Supplementary B data. Data for 2004 are preliminary, and do not include Supplementary B data.

Reported landings in Gulf NB, Gulf NS, and PEI were low until the early 1960s (Table 6.1, Fig. 6.1). The bulk of the landings come from Gulf NB and PEI. In the 1960s a new fishing method (the fyke net) was introduced and new markets were developed. Reported landings rose sharply and peaked in the late 1960s and early 1970s. This peak was followed by a decline, and a period of irregular fluctuations. Reported landings have increased more than two-fold in Gulf NB and PEI since 1997. Landings in Gulf NS remain very low, only a few tonnes per year.

South Shore Trading Co. of Port Elgin, N.B., is the major eel buyer in Gulf Region. Weight of eels purchased by this firm and its agents are compiled from company records. Reported landings and South Shore purchase records are compared in Table 6.1 and Fig. 6.2 for 2000-2004. In Gulf New Brunwick reported landings are greater than reported purchases, but follow parallel trends. In PEI, the two series are close. Purchase figures were larger than landings records for 2004 in PEI, but this might be due to the absence of Supplementary B information from the landings records. In general, the parallel nature of the two data series suggests that reported landings are at least an approximately accurate record of eels which are fished and commercially sold.

6.2.2 *Commercial CPUE*

Table 6.2 shows the sum of purchases and the number of weekly eel pick-ups from fishers in Gulf NB and PEI for 2000-2004. The number of pick-ups is an indication of fishing effort because it reflects the number of active fishers and the duration of their fishing effort. There are sometimes intevals greater than a week between pick-ups. In such cases it is not possible to know whether the fisher was active during the full interval. We calculated effort in two way. Unadjusted fisher-weeks is the total number of pick-ups. Adjusted fisher-weeks is the total number of weekly pick-ups, plus the number of weeks with no pickups where the week was followed by a week with a pick-up.

CPUEs were calculated as kg of eels purchased per fisher-week. CPUES calculated using the two methods were closely parallel (Fig. 6.3). CPUE in Gulf NB showed little variation. CPUE in PEI peaked in 2001, fell to a trough in 2003, and then rose again in 2004.

DFO has operated a compulsory logbook program for commercial eel fishers in Gulf NS since 1997. On PEI, a voluntary logbook program has been in operation since 1996. Three to seven commercial eel fishers participate in the program each year. The logbook contains a line for each day of the fishing season, with spaces to record the number of fyke nets in the water, the catch of legal eels (in pounds), and the number of eels under the minimum legal size (46 cm up to 1997; 50.8 cm in 1998-2004).

CPUE in the Gulf NS series has varied without trend since 1997 (Table 6.3, Fig. 6.4). CPUE of legal-size eels on PEI more than doubled between 1996 and 2004 (Table 6.3, Fig. 6.4). CPUE of sublegal eels peaked in 1999, decreased, and has been rising since 2001.

6.2.3 *Electrofishing*

DFO-Gulf Region conducts electrofishing surveys in rivers draining into the southern Gulf of St. Lawrence to evaluate salmonid populations (Chaput and Claytor 1989; Atkinson et al. 2000; Cairns et al. 2000; Chaput et al. 2000, 2001; Douglas and Swasson 2000; Marshall et al. 2000; O'Neil et al. 2000; Cairns 2002; Atkinson 2004). We estimated eel densities from data gathered during these surveys.

Prior to 1993 all electrofishing surveys were conducted with multiple sweeps in sites bounded upstream and downstream by barrier nets. This method allows calculation of populations by the removal method (Zippin 1958). In 1993, single-sweep electrofishing sessions without barrier nets were introduced to improve survey efficiency. Effort was measured as seconds of electrofishing time. The relation of salmon CPUE to abundance was determined from calibration sites where a single CPUE sweep was made within barriered sites which were subsequently fished with the removal method, by which populations were estimated. Single sweeps without barrier nets are referred to as Sweep no. 0.5. Sweeps with barrier nets are called Sweep no. 1, 2, 3, etc. At some sites in recent years salmonid populations were estimated by mark-recapture, using two sweeps and no barrier nets.

The Zippen depletion method assumes that catch numbers, on average, will decline with successive fishing sessions as the population is depleted by removals. Mean number of eels per sweep in the Restigouche, Miramichi, and Margaree Rivers were 0.84, 0.41, 0.27, and 0.20 for sweeps 1 through 4, respectively (Table 6.4). This suggests that electrofishing in the southern Gulf satisfies the depletion assumption.

We estimated eel populations in electrofishing sites by the Zippin formula when it was possible to do so. At many sites the Zippin formula could not be applied because catches were too low, because catches did not decline with sweep number, or because only one sweep was conducted. Populations at these sites were estimated with the aid of correction factors (Table 6.5). Correction factors were calculated as follows. For each site where a Zippin population estimate was available, the Sweep 0.5 count as a proportion of the Zippin population estimate and the Sweep 1 count as a proportion of the Zippin population estimate were calculated. We also calculated the sum of Sweeps 0.5 to 3, the sum of Sweeps 1 to 3, the sum of Sweeps 1 to 4, and the sums of Sweeps 1 to 5, as proportions of the Zippin estimate. The means of these proportions were calculated per river and across all rivers. Correction factors were the inverse of these means.

To illustrate, consider a 3-sweep electrofishing session on the Miramichi River where Sweep 1 captures no eels, Sweep 2 captures two eels, and Sweep 3 captures three eels. The Zippin algorithm cannot be used because the numbers caught do not decline with sweep number. In the Miramichi, on average, the sum of Sweeps 1 through 3 was 0.800 of the Zippin population estimate for the site. The inverse of 0.800 is 1.25. In our example, five eels were caught in Sweeps 1 to 3. Hence the population estimate for the site is 5 x 1.25 = 6.25.

Density estimates for eels in the Margaree River from 1957 to 1987 are from Chaput and Claytor (1989).

Mean eel densities in the Restigouche, Miramichi, and Margaree Rivers are presented in Table 6.6 and Fig. 6.6. After irregular variation, densities in the Restigouche fell to low levels in the mid-1990s but then increased, showing high peaks in the early 2000s. Densities in the Miramichi varied irregularly in the 1950s and 1960s, then peaked in the early 1970s. They then dropped to low levels by the late 1980s and early 1990s, with a gradual recovery since that time. Densities in the Margaree peaked in the 1960s. They have been very low since 1990.

Table 6.1
Reported American eel landings in the Southern Gulf of St. Lawrence, against year of landings and against year of recruitment. Landings data are from DFO Statistics Branch. Landings for 2004 are preliminary and do not include Supplementary B data. Year of recruitment is based on mean ages given in Table 3.2. Purchases of South Shore Trading Limited are also shown for Gulf NB and PEI for 2000-2004. Data are from company records.

Year	Reported landings (tonnes) Gulf New Brunswick[a]			Reported landings (tonnes) Gulf Nova Scotia[b]			Reported landings (tonnes) Prince Edward Island[c]			Reported landings (tonnes), southern Gulf of St. Lawrence		South Shore Trading purchases against year of purchase (tonnes)	
	Against year of landing	Mean age	Against year of recruitment	Against year of landing	Mean age	Against year of recruitment	Against year of landing	Mean age	Against year of recruitment	Against year of landing	Against year of recruitment	Gulf New Brunswick	Prince Edward Island
1907			51.0			12.6							
1908			61.8			13.1							
1909			75.1			6.3							
1910			24.2			10.7							
1911			40.7			14.8			0.0		55.5		
1912			14.0			7.5			0.0		21.5		
1913			10.2			0.7			0.0		10.9		
1914			10.0			7.5			0.0		17.5		
1915			18.4			4.3			0.0		22.7		
1916			5.4			5.7			0.0		11.1		
1917	51.0	10	1.4	12.6	10	3.5	0.0	6	0.0	63.6	4.9		
1918	61.8	10	16.3	13.1	10	6.6	0.0	6	0.0	74.9	22.9		
1919	75.1	10	5.2	6.3	10	4.5	0.0	6	0.0	81.4	9.7		
1920	24.2	10	11.8	10.7	10	15.2	0.0	6	0.0	34.9	27.0		
1921	40.7	10	18.8	14.8	10	11.9	0.0	6	0.0	55.5	30.7		
1922	14.0	10	9.1	7.5	10	13.8	0.0	6	0.0	21.5	22.9		
1923	10.2	10	11.0	0.7	10	16.7	0.0	6	0.0	10.9	27.7		
1924	10.0	10	11.3	7.5	10	5.8	0.0	6	0.0	17.5	17.1		
1925	18.4	10	7.6	4.3	10	5.2	0.0	6	0.0	22.7	12.8		
1926	5.4	10	4.4	5.7	10	4.1	0.0	6	0.0	11.1	8.5		
1927	1.4	10	5.7	3.5	10	4.2	0.0	6	0.0	4.9	9.9		
1928	16.3	10	9.4	6.6	10	9.7	0.0	6	0.0	22.9	19.1		
1929	5.2	10	11.0	4.5	10	8.7	0.0	6	0.0	9.7	19.7		
1930	11.8	10	4.8	15.2	10	10.6	0.0	6	0.0	27.0	15.4		
1931	18.8	10	3.6	11.9	10	3.4	0.0	6	0.0	30.7	7.0		
1932	9.1	10	12.5	13.8	10	4.8	0.0	6	0.0	22.9	17.3		
1933	11.0	10	14.1	16.7	10	5.0	0.0	6	0.0	27.7	19.1		
1934	11.3	10	13.9	5.8	10	6.9	0.0	6	0.0	17.1	20.8		
1935	7.6	10	14.5	5.2	10	10.7	0.0	6	0.0	12.8	25.2		
1936	4.4	10	29.1	4.1	10	16.7	0.0	6	0.0	8.5	45.8		
1937	5.7	10	31.8	4.2	10	13.6	0.0	6	0.0	9.9	45.4		
1938	9.4	10	29.0	9.7	10	8.7	0.0	6	0.0	19.1	37.7		
1939	11.0	10	29.4	8.7	10	37.6	0.0	6	0.0	19.7	67.0		
1940	4.8	10	22.2	10.6	10	23.6	0.0	6	0.0	15.4	45.8		
1941	3.6	10	15.5	3.4	10	20.9	0.0	6	0.9	7.0	37.3		
1942	12.5	10	15.8	4.8	10	11.9	0.0	6	10.0	17.3	37.7		
1943	14.1	10	13.1	5.0	10	7.7	0.0	6	3.6	19.1	24.4		
1944	13.9	10	33.1	6.9	10	6.4	0.0	6	2.3	20.8	41.8		
1945	14.5	10	48.6	10.7	10	10.5	0.0	6	3.6	25.2	62.7		
1946	29.1	10	10.5	16.7	10	14.6	0.0	6	5.0	45.8	30.1		
1947	31.8	10	8.6	13.6	10	10.1	0.9	6	6.3	46.3	25.0		
1948	29.0	10	14.5	8.7	10	14.1	10.0	6	3.7	47.7	32.3		
1949	29.4	10	23.6	37.6	10	11.4	3.6	6	9.1	70.6	44.1		
1950	22.2	10	30.9	23.6	10	23.6	2.3	6	4.6	48.1	59.1		
1951	15.5	10	57.4	20.9	10	27.8	3.6	6	12.3	40.0	97.5		
1952	15.8	10	81.9	11.9	10	26.4	5.0	6	18.7	32.7	127.0		
1953	13.1	10	53.7	7.7	10	23.6	6.3	6	26.4	27.1	103.7		
1954	33.1	10	56.4	6.4	10	18.8	3.7	6	31.9	43.2	107.1		

Table 6.1 (continued)

Year	Reported landings (tonnes) Gulf New Brunswick[a]			Reported landings (tonnes) Gulf Nova Scotia[b]			Reported landings (tonnes) Prince Edward Island[c]			Reported landings (tonnes), southern Gulf of St. Lawrence		South Shore Trading purchases against year of purchase (tonnes)	
	Against year of landing	Mean age	Against year of recruit-ment	Against year of landing	Mean age	Against year of recruit-ment	Against year of landing	Mean age	Against year of recruit-ment	Against year of landing	Against year of recruit-ment	Gulf New Bruns-wick	Prince Edward Island
1955	48.6	10	62.6	10.5	10	16.3	9.1	6	17.7	68.2	96.6		
1956	10.5	10	99.2	14.6	10	15.0	4.6	6	13.1	29.7	127.3		
1957	8.6	10	108.0	10.1	10	52.3	12.3	6	15.9	31.0	176.2		
1958	14.5	10	150.6	14.1	10	28.3	18.7	6	34.2	47.3	213.1		
1959	23.6	10	214.2	11.4	10	38.1	26.4	6	48.6	61.4	300.9		
1960	30.9	10	294.7	23.6	10	45.4	31.9	6	32.8	86.4	372.9		
1961	57.4	10	319.4	27.8	10	52.1	17.7	6	61.8	102.9	433.3		
1962	81.9	10	272.8	26.4	10	50.3	13.1	6	130.7	121.4	453.8		
1963	53.7	10	220.4	23.6	10	28.0	15.9	6	194.5	93.2	442.9		
1964	56.4	10	156.2	18.8	10	28.3	34.2	6	239.9	109.4	424.4		
1965	62.6	10	120.8	16.3	10	28.6	48.6	6	351.4	127.5	500.8		
1966	99.2	10	118.7	15.0	10	18.0	32.8	6	272.8	147.0	409.5		
1967	108.0	10	110.1	52.3	10	5.9	61.8	6	157.2	222.1	273.2		
1968	150.6	10	81.6	28.3	10	12.3	130.7	6	101.2	309.6	195.1		
1969	214.2	10	102.4	38.1	10	12.6	194.5	6	103.5	446.8	218.5		
1970	294.7	10	150.4	45.4	10	9.5	239.9	6	94.1	580.0	254.0		
1971	319.4	10	191.2	52.1	10	7.5	351.4	6	97.6	722.9	296.3		
1972	272.8	10	159.2	50.3	10	11.3	272.8	6	113.6	595.9	284.1		
1973	220.4	10	97.4	28.0	10	9.6	157.2	6	111.0	405.6	218.0		
1974	156.2	10	122.4	28.3	10	8.9	101.2	6	120.1	285.7	251.4		
1975	120.8	10	202.4	28.6	10	5.1	103.5	6	220.0	252.9	427.5		
1976	118.7	10	230.2	18.0	10	15.6	94.1	6	167.6	230.8	413.4		
1977	110.1	10	171.6	5.9	10	13.2	97.6	6	150.5	213.6	335.3		
1978	81.6	10	233.5	12.3	10	24.7	113.6	6	164.6	207.5	422.8		
1979	102.4	10	209.0	12.6	10	30.2	111.0	6	139.4	226.0	378.6		
1980	150.4	10	149.3	9.5	10	20.8	120.1	6	226.0	280.0	396.1		
1981	191.2	10	130.2	7.5	10	34.8	220.0	6	149.9	418.7	314.9		
1982	159.2	10	119.6	11.3	10	56.0	167.6	6	124.7	338.1	300.3		
1983	97.4	10	88.3	9.6	10	89.2	150.5	6	69.5	257.5	247.0		
1984	122.4	10	68.1	8.9	10	42.3	164.6	6	123.8	295.9	234.2		
1985	202.4	10	60.2	5.1	10	16.3	139.4	6	126.6	346.9	203.0		
1986	230.2	10	48.7	15.6	10	11.4	226.0	6	54.0	471.8	114.1		
1987	171.6	10	36.4	13.2	10	17.2	149.9	6	74.0	334.7	127.6		
1988	233.5	10	49.2	24.7	10	15.0	124.7	6	45.8	382.9	110.0		
1989	209.0	10	47.2	30.2	10	9.0	69.5	6	34.6	308.7	90.7		
1990	149.3	10	92.2	20.8	10	6.9	123.8	6	36.0	293.9	135.1		
1991	130.2	10	115.4	34.8	10	3.4	126.6	6	31.3	291.6	150.1		
1992	119.6	10	143.8	56.0	10	4.2	54.0	6	23.6	229.6	171.6		
1993	88.3	10	122.1	89.2	10	9.3	74.0	6	35.3	251.5	186.9		
1994	68.1	10		42.3	10	2.7	45.8	6	63.5	156.2			
1995	60.2	10		16.3	10		34.6	6	41.2	111.0			
1996	48.7	10		11.4	10		36.0	6	86.4	96.1			
1997	36.4	10		17.2	10		31.3	6	69.6	84.9			
1998	49.2	10		15.0	10		23.6	6		87.8			
1999	47.2	10		9.0	10		35.3	6		91.5			
2000	76.4	10		6.9	10		63.5	6		146.7		55.8	57.9
2001	92.2	11		3.4	11		41.2	6		136.8		64.8	45.3
2002	115.4	11		4.2	11		86.4	6		206.0		68.8	73.4
2003	143.8	11		9.3	11		69.6	6		222.7		90.7	56.6
2004	122.1	11		2.7	11		62.1	6		186.9		89.8	75.0

[a] Minimum sizes in the Gulf NB eel fishery were set at 38.1 cm in 1996, 46 cm in 2001, and 50 cm in 2004.

[b] Minimum sizes in the Gulf NS eel fishery were set at 46 cm in 1996, and 50 cm in 2001. Mean age assumed to be the same as Gulf NB.

[c] Minimum sizes in the PEI eel fishery were set at 46 cm in the 1970s, and 50.8 cm in 1998.

Table 6.2
Purchases, fishing effort in fisher-weeks, and catch per unit effort in the Gulf New Brunswick and Prince Edward Island eel fishery from records of the South Shore Trading Company.

Year	Purchases (tonnes)	Fisher-weeks		CPUE (kg per fisher-week)	
		Unadjusted[a]	Adjusted[b]	Unadjusted[a]	Adjusted[b]
Gulf New Brunswick					
2000	55.8	204	295	273.4	189.1
2001	64.8	286	378	226.7	171.5
2002	68.8	272	357	252.9	192.7
2003	90.7	327	376	277.2	241.1
2004	89.8	386	425	232.7	211.3
Prince Edward Island					
2000	57.9	204	215	284.0	269.5
2001	45.3	96	113	472.3	401.3
2002	73.4	169	209	434.1	351.0
2003	56.6	193	218	293.3	259.7
2004	75.0	205	219	365.6	342.3

[a] Unadjusted fisher-weeks is the summed number of weekly pick-ups from fishers.

[b] For adjusted fisher-weeks, in weeks when there is no pick-ups, fisher-weeks is the number of fisher pick-ups the following week.

Table 6.3
Catch rates of American eels by commercial fishers in Gulf Nova Scotia against year of fishing, and in Prince Edward Island against year of fishing and against year of recruitment. Year of recruitment is calculated from mean ages of 6 for legal eels and 5 for sublegal eels (Table 3.2). Catch rates on PEI are standardized to a minimum legal length of 50.8 cm (see Cairns et al. 2005).

Year	Gulf Nova Scotia			Prince Edward Island			
	Fyke nets (kg net^{-1} day^{-1})	Winter spearing (kg hr^{-1})	Summer spearing (kg hr^{-1})	Legal sized eels (kg net^{-1} day^{-1})		Sublegal eels (numbers net^{-1} day^{-1})	
				Against year of fishing	Against year of recruitment	Against year of fishing	Against year of recruitment
1990					0.32		
1991					0.29		0.99
1992					0.52		1.98
1993					0.92		2.51
1994					0.91		4.04
1995					0.64		1.98
1996				0.32	0.77	0.99	1.15
1997	1.73	2.33	3.73	0.29	0.87	1.98	1.43
1998	0.93	3.30	3.74	0.52	1.15	2.51	1.63
1999				0.92		4.04	2.06
2000	1.23	1.81	3.07	0.91		1.98	
2001	1.34	1.10	1.60	0.64		1.15	
2002	1.74	3.18	2.19	0.77		1.43	
2003	1.31	2.91	0.96	0.87		1.63	
2004	2.42	3.42		1.15		2.06	

Table 6.4
Mean eel counts per sweep in electrofishing surveys conducted in the Restigouche, Miramichi, and Margaree Rivers, by sweep number.

River	Eels per sweep, for sweep no.																				
	0.5[a]			1			2			3			4			5			6		
	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N
Restigouche	0.16	0.92	542	0.48	1.40	755	0.14	0.54	755	0.08	0.35	754	0.04	0.29	582	0.02	0.15	347			
Miramichi	0.17	0.59	727	0.96	2.99	1,716	0.54	1.68	1,641	0.35	1.19	1,641	0.27	1.04	1,527	0.16	0.61	1,111	0.13	0.36	200
Margaree	0.20	1.19	59	0.57	0.84	23	0.13	0.34	23	0.39	1.16	23	0.00	0.00	13						
Weighted mean, sum[b]	0.17		1,328	0.82		2,494	0.41		2,419	0.27		2,418	0.20		2,122	0.13		1,458	0.13		200

[a] Sweep 0.5 is a sweep without barrier nets that is conducted alone, or conducted prior to regular Zippin sweeps.

[b] Mean is weighted by sample size

Table 6.5
Electrofishing counts per sweep and sums of sweep counts as a proportion of Zippin estimates of total population, and correction factors to convert sweep counts to total populations.

River	Sweep 0.5 counts[a]			Sweep 1 counts			Summed counts, sweeps 0.5-3			Summed counts, sweeps 1-3			Summed counts, sweeps 1-4			Summed counts, sweeps 1-5		
	Prop. of Zippin estimate	Correct-tion factor	N	Prop. of Zippin estimate	Correct-tion factor	N	Prop. of Zippin estimate	Correct-tion factor	N	Prop. of Zippin estimate	Correct-tion factor	N	Prop. of Zippin estimate	Correct-tion factor	N	Prop. of Zippin estimate	Correct-tion factor	N
Restigouche	0.000		1	0.499	2.005	83	0.813	1.230	1	0.873	1.145	83	0.906	1.103	81	0.916	1.092	49
Miramichi	0.210	4.767	3	0.340	2.940	450	0.768	1.303	3	0.800	1.250	450	0.856	1.168	443	0.899	1.113	348
Buctouche				0.303	3.301	2				0.738	1.356	2	0.738	1.356	2			
Mainland Gulf NS rivers				0.440	2.270	15				0.862	1.160	15	0.890	1.123	13	0.925	1.081	11
Gulf Cape Breton rivers				0.607	1.646	2				0.965	1.036	2	0.965	1.036	2			
Morell				0.527	1.899	22				0.876	1.142	22	0.913	1.095	17	0.942	1.061	9
Weighted mean/sum[b]	0.157	6.357	4	0.374	2.676	574	0.779	1.284	4	0.816	1.226	574	0.866	1.155	558	0.902	1.108	417

[a] Sweep 0.5 is a sweep without barrier nets that is conducted alone, or conducted prior to regular Zippin sweeps.

[b] Mean is weighted by sample size.

Table 6.6
Mean American eel densities in the Restigouche, Miramichi, and Margaree Rivers, estimated from electrofishing, against year of surveys and against year of recruitment. Year of recruitment is based on mean ages given in Table 3.2.

Year	Density (eels 100m^{-2}) in the Restigouche River		Density (eels 100m^{-2}) in the Miramichi River		Density (eels 100m^{-2}) in the Margaree River	
	Against year of surveys	Against year of recruitment	Against year of surveys	Against year of recruitment	Against year of surveys	Against year of recruitment
Mean age	5		6		8	
1943						
1944						
1945						
1946				0.56		
1947				1.31		
1948				0.30		
1949				0.63		3.39
1950				0.48		1.75
1951				0.34		2.75
1952			0.56	0.55		4.81
1953			1.31	1.47		4.25
1954			0.30	0.85		3.00
1955			0.63	0.37		5.13
1956			0.48	0.13		5.66
1957			0.34	0.43	3.39	2.01
1958			0.55	0.95	1.75	2.98
1959			1.47	0.72	2.75	2.42
1960			0.85	0.62	4.81	1.46
1961			0.37	1.02	4.25	1.11
1962			0.13	0.88	3.00	
1963			0.43	0.52	5.13	
1964			0.95	0.33	5.66	
1965			0.72	1.88	2.01	
1966			0.62	1.44	2.98	
1967		0.43	1.02	1.23	2.42	2.59
1968		0.50	0.88	1.57	1.46	2.41
1969		0.35	0.52	1.23	1.11	2.83
1970		1.01	0.33	1.06		1.76
1971		0.23	1.88	1.24		1.74
1972	0.43	0.37	1.44	0.65		
1973	0.50	0.28	1.23	0.16		0.00
1974	0.35	0.10	1.57	0.15		5.00
1975	1.01	0.48	1.23	0.37	2.59	0.18
1976	0.23	0.09	1.06	0.89	2.41	0.00
1977	0.37	0.09	1.24	0.90	2.83	0.00
1978	0.28	0.51	0.65	0.47	1.76	0.47
1979	0.10		0.16	0.18	1.74	0.79
1980	0.48	0.32	0.15	0.15		
1981	0.09	0.48	0.37	0.17	0.00	
1982	0.09	0.29	0.89	0.27	5.00	
1983	0.51	0.76	0.90	0.07	0.18	1.09
1984		0.60	0.47	0.24	0.00	0.00
1985	0.32	0.36	0.18	0.00	0.00	0.00
1986	0.48	0.21	0.15	0.15	0.47	0.14
1987	0.29	0.06	0.17	0.28	0.79	0.11
1988	0.76	0.00	0.27	0.41		0.00
1989	0.60	0.31	0.07	0.07		0.19
1990	0.36	0.00	0.24	0.30		0.10
1991	0.21	0.00	0.00	0.20	1.09	0.00
1992	0.06	0.00	0.15	0.33	0.00	0.22
1993	0.00	0.12	0.28	0.47	0.00	0.15
1994	0.31	0.25	0.41	0.34	0.14	1.08
1995	0.00	0.27	0.07	0.73	0.11	0.12
1996	0.00	1.18	0.30	0.30	0.00	0.25
1997	0.00	1.40	0.20	0.32	0.19	0.17
1998	0.12	0.24	0.33	0.64	0.10	
1999	0.25	0.49	0.47		0.00	
2000	0.27		0.34		0.22	
2001	1.18		0.73		0.15	
2002	1.40		0.30		1.08	
2003	0.24		0.32		0.12	
2004	0.49		0.64		0.17	

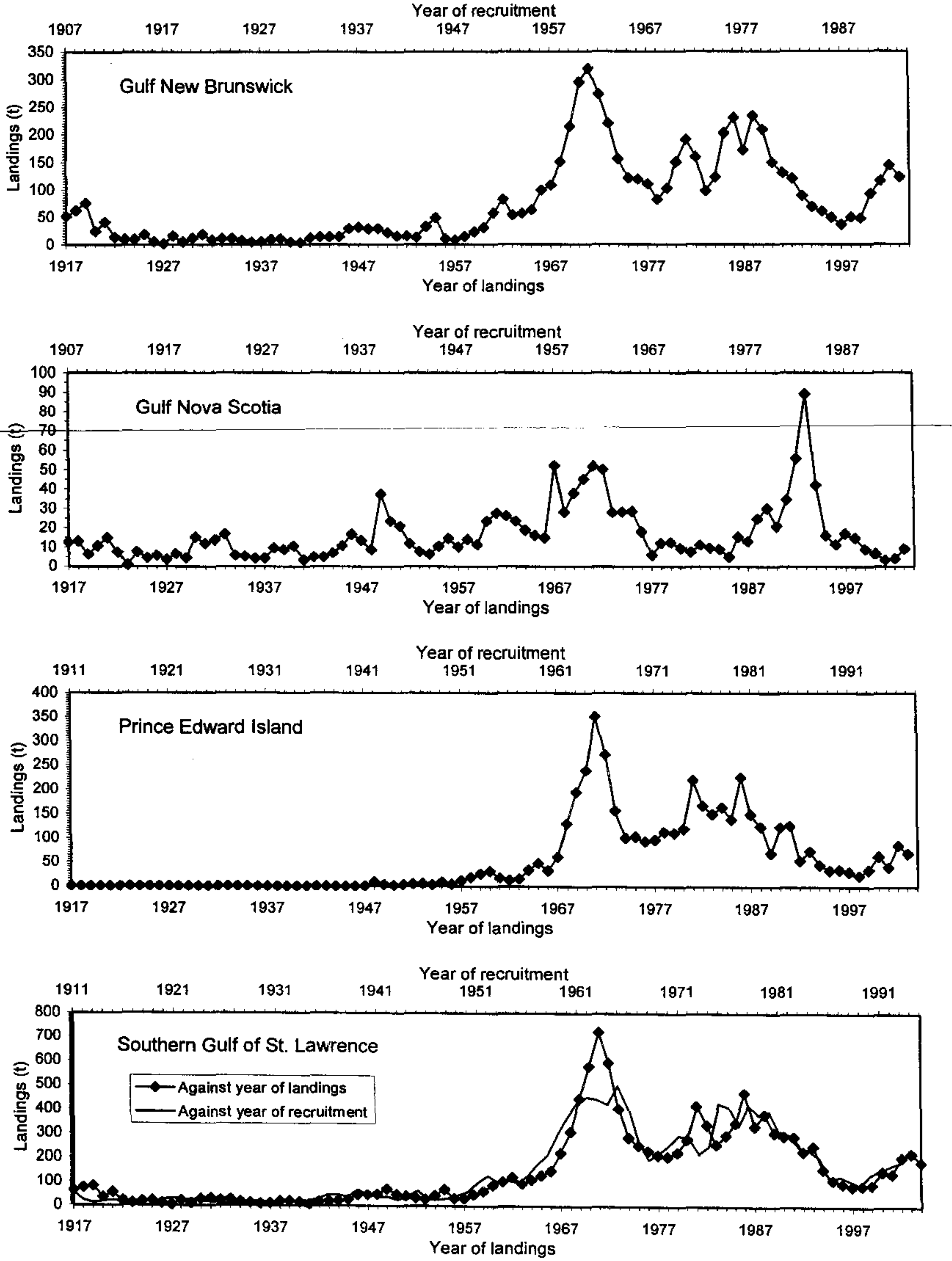


Fig. 6.1
Reported American eel landings in the Southern Gulf of St. Lawrence, against year of landings and against year of recruitment. Year of recruitment is based on mean ages given in Table 3.2. Data for 2004 are preliminary and do not include Supplementary B data.

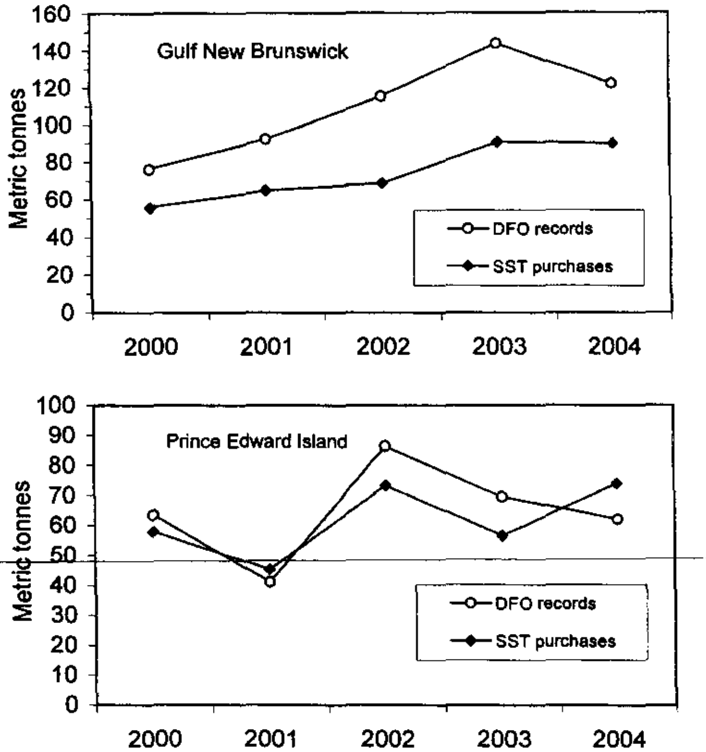


Fig. 6.2
Reported landings of American eels in Gulf New Brunswick and Prince Edward Island, compared to purchases by South Shore Trading Co.

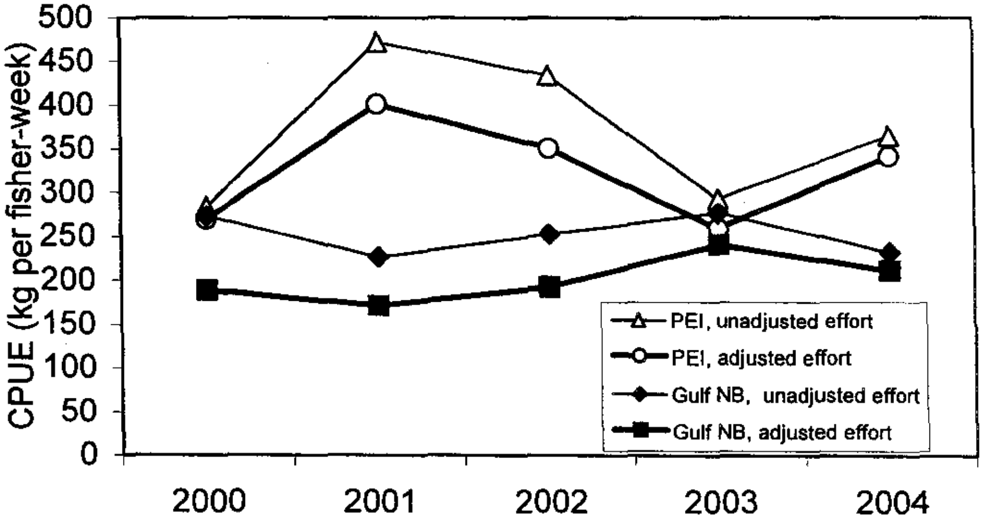


Fig. 6.3
Trends in catch per unit effort in the eel fishery in Gulf New Brunswick and Prince Edward Island, from purchase records of the South Shore Trading Company. See Table 6.3 for definitions of unadjusted and adjusted effort.

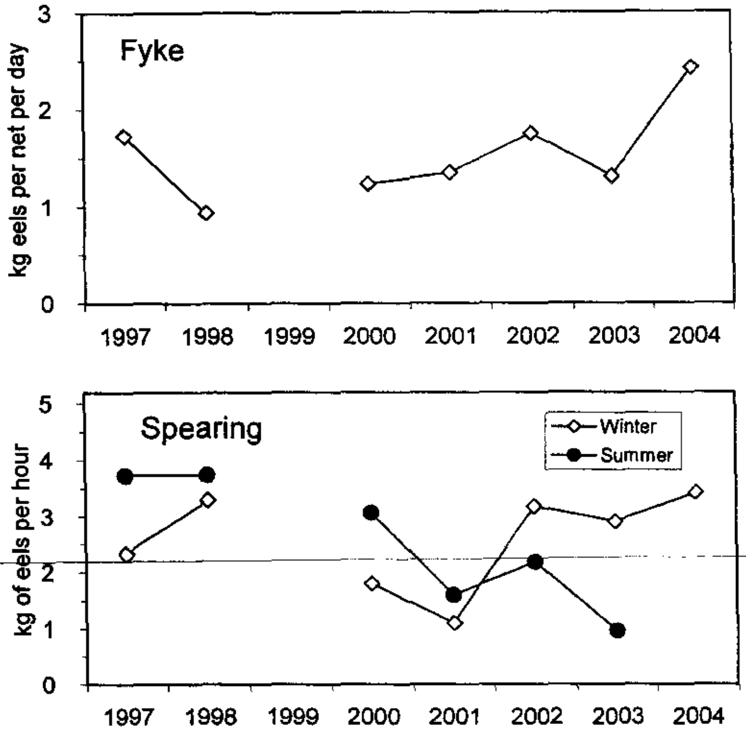


Fig. 6.4
Catch rates of American eels in the fyke net and winter and summer spear fisheries in Gulf Nova Scotia, from fisher logbooks.

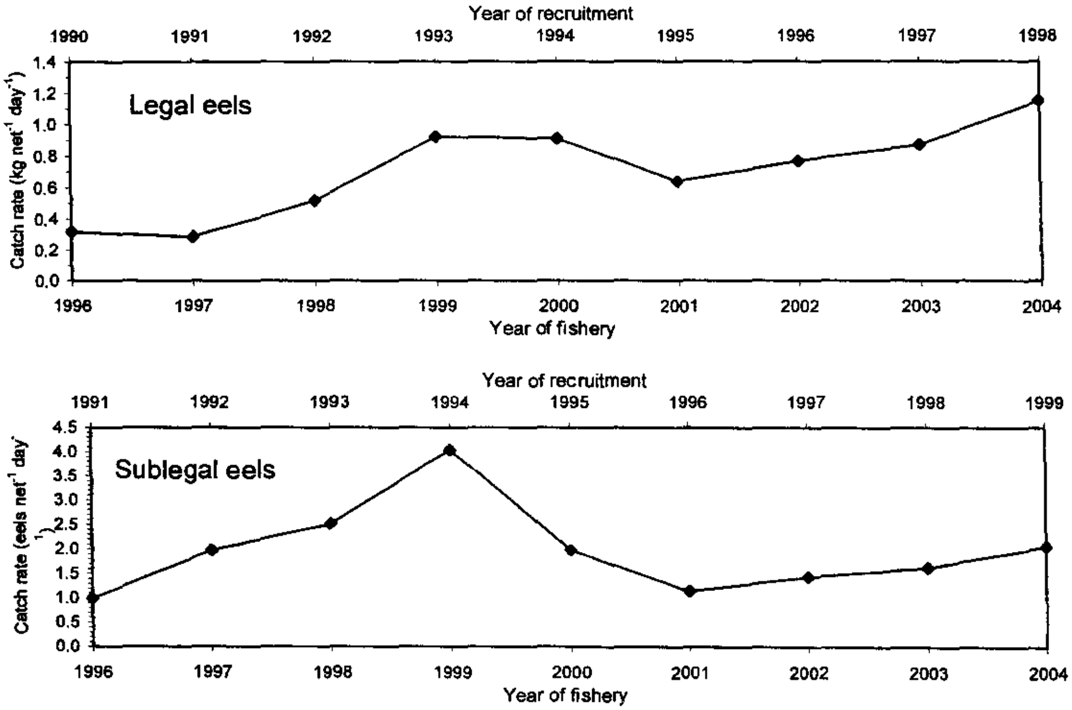


Fig. 6.5
Catch rates of legal and sublegal American eels by commercial index fishers on Prince Edward Island, against year of fishery and against year of recruitment. Year of recruitment is based on mean ages given in Table 3.2 Minimum legal sizes were 46 cm in 1996-1997, and 50.8 cm in 1998-2004.

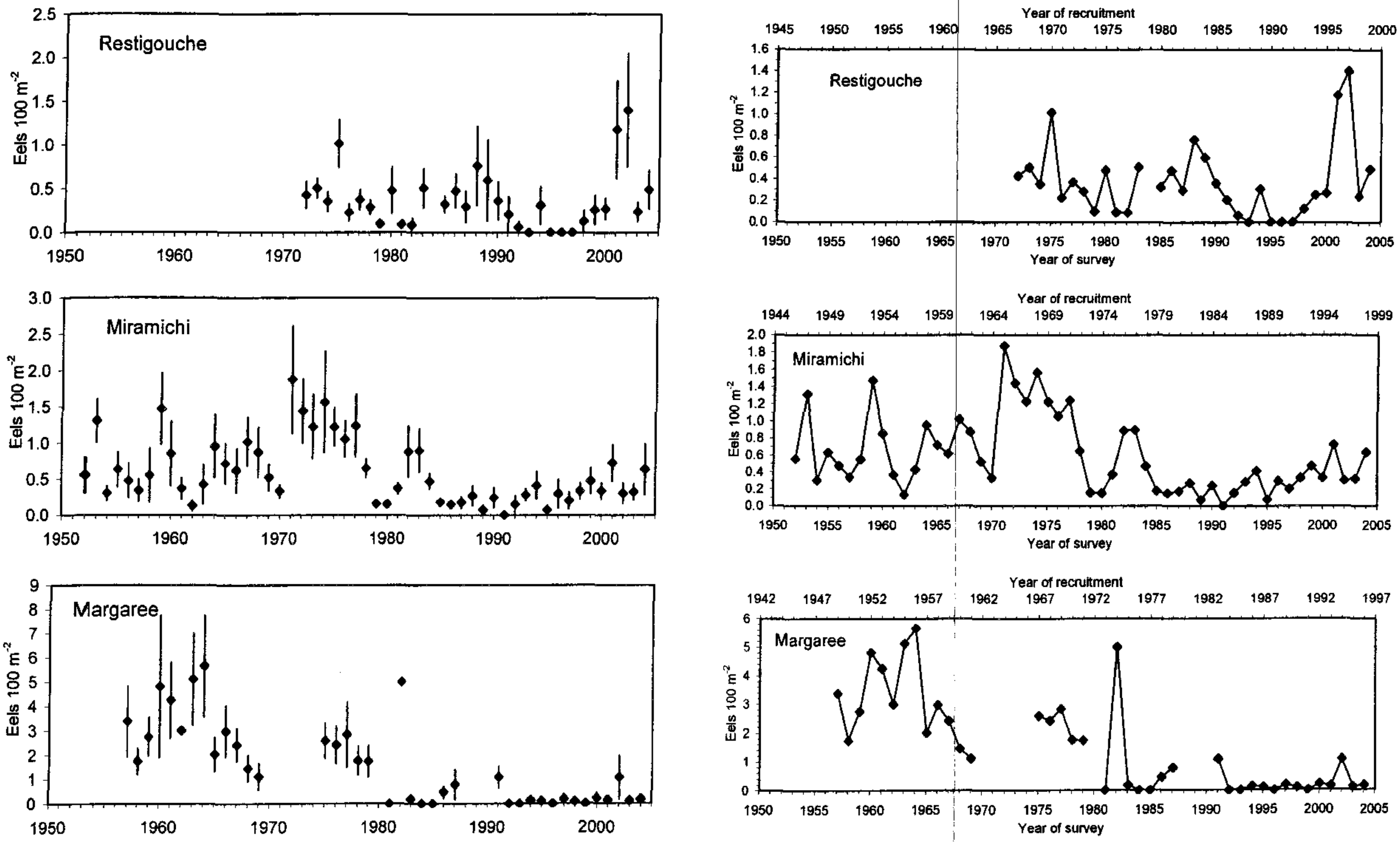


Fig. 6.6
Mean American eel densities in the Restigouche, Miramichi, and Margaree Rivers, estimated from electrofishing. Left panels show means + Standard Error. Right panels show means against year of surveys and against year of recruitment. Year of recruitment is based on on mean ages given in Table 3.2.

7. EEL ABUNDANCE INDICATORS IN SCOTIA-FUNDY

7.1 Background

The Scotia-Fundy area consists of the Atlantic and Bay of Fundy drainages of New Brunswick and Nova Scotia. It is part of DU3.

7.2 Population indicators

7.2.1 *Commercial landings*

The commercial fishery for yellow and silver eels is concentrated in the Saint John River below the Mactaquac dam, and at various inland and coastal locations scattered across Nova Scotia (Fig. 3.3). There is an elver fishery, the only one in Canada, in estuaries of designated rivers along the Bay of Fundy and Atlantic Ocean coasts of New Brunswick and Nova Scotia (Fig. 3.3).

Data on eel landings in Scotia-Fundy are gathered by a logbook system. Reported landings increased in the 1970s in S-F New Brunswick and in the 1990s in S-F Nova Scotia (Table 8.1, Fig. 8.1). Reported landings in the S-F portion of both provinces have been decreasing since the mid 1990s.

Reported landings and South Shore Trading purchase records are compared in Table 8.1 and Fig. 8.2. Landings records which are lower than purchase records in 2004 could be due to the preliminary nature of the landings records.

7.2.2 *Commercial CPUE*

Table 8.2 and Fig. 8.3 show commercial CPUE, in kg of eels landed per fisher-week, from South Shore Trading Company records. The data show a gradual decline in CPUE.

7.2.3 *Elver surveys*

Elvers were counted at the mouths of the East River, Sheet Harbour, in 1996-2002 and the East River, Chester, on the Atlantic coast of Nova Scotia, in 1990-1999. Both series showed fluctuations but no trend (Table 7.3, Fig. 8.4).

7.2.4 *Electrofishing*

Electrofishing surveys are conducted in Scotia-Fundy rivers to assess salmonid populations. Because eel catches were in general insufficient to allow population esimates by the depletion method, results are presented as the sum of eels counted per 100 m^2 (Figs. 8.4 and 8.5). In some cases, electrofishing sessions in which eels were not recorded may have been deleted from the Nova Scotia data set. This would have the effect of upwardly biasing mean counts per 100 m^2.

Table 7.1
Reported landings (metric tonnes) of American eels in the Scotia-Fundy area of New Brunswick and Nova Scotia. Purchases by the South Shore Trading Company are also shown. Data are from company records. Purchases from Cape Breton may include some eels from the southern Gulf area.

Year	Reported landings			South Shore Trading purchases				
	NB	NS	Total	NB	Nova Scotia			Scotia-Fundy total
					Cape Breton	Southern NS	Total	
1950	18.5	20.8	39.3					
1951	2.8	25.3	28.1					
1952	8.6	35.3	43.9					
1953	21.0	40.5	61.5					
1954	9.5	51.3	60.8					
1955	39.6	67.3	106.9					
1956	2.4	48.2	50.6					
1957	5.0	21.9	26.9					
1958	7.1	38.9	46.0					
1959	0.0	20.1	20.1					
1960	0.8	21.5	22.3					
1961	8.6	25.0	33.6					
1962	8.7	24.5	33.2					
1963	13.1	40.5	53.6					
1964	9.7	42.8	52.5					
1965	5.3	26.9	32.2					
1966	19.7	28.2	47.9					
1967	4.1	13.8	17.9					
1968	12.3	22.1	34.4					
1969	60.4	23.6	84.0					
1970	54.3	29.0	83.3					
1971	67.1	49.1	116.2					
1972	35.4	25.0	60.4					
1973	27.1	19.9	47.0					
1974	20.9	13.7	34.6					
1975	51.9	12.4	64.3					
1976	78.0	9.0	87.0					
1977	100.0	9.0	109.0					
1978	44.0	52.0	96.0					
1979	120.0	21.0	141.0					
1980	24.0	30.0	54.0					
1981	35.0	20.0	55.0					
1982	3.0	17.0	20.0					
1983	0.0	19.0	19.0					
1984	3.0	8.0	11.0					
1985	73.0	7.0	80.0					
1986	55.0	6.0	61.0					
1987	49.0	14.0	63.0					
1988	135.0	15.0	150.0					
1989	116.0	6.0	122.0					
1990	90.1	5.0	95.1					
1991	88.0	39.0	127.0					
1992	59.0	62.0	121.0					
1993	116.0	72.0	188.0					
1994	131.0	99.0	230.0					
1995	113.7	115.9	229.6					
1996	63.1	50.3	113.3					
1997	69.2	50.2	119.4					
1998	87.7	73.5	161.2					
1999	63.5	61.5	125.0					
2000	68.9	84.5	153.5	63.4	23.1	38.9	62.0	85.1
2001	64.3	68.2	132.5	35.8	20.2	36.8	57.0	77.2
2002	48.5	42.9	91.3	40.1	22.2	36.7	58.9	81.1
2003	59.0	35.0	94.1	54.1	18.7	11.2	29.9	48.5
2004	51.0	37.0	88.0	74.6	20.9	32.3	53.1	74.0

Table 7.2
Purchases, fisher-weeks, and catch per unit effort in the Cape Breton, southern Nova Scotia, and Scotia-Fundy New Brunswick eel fishery from records of the South Shore Trading Company. Cape Breton data may include some records from the Gulf of St. Lawrence portion of Nova Scotia.

Year	Purchases (tonnes)	Fisher-weeks		CPUE (kg per fisher-week)	
		Unadjusted[a]	Adjusted[b]	Unadjusted[a]	Adjusted[b]
Scotia-Fundy New Brunswick					
2000	63.4	162	204	391.1	310.6
2001	35.8	126	152	284.0	235.4
2002	40.1	130	162	308.5	247.6
2003	54.1	162	190	334.1	284.9
2004	74.6	208	219	358.5	340.5
Cape Breton Island					
2000	23.1	55	98	420.0	235.7
2001	20.2	56	107	361.3	189.1
2002	22.2	64	109	346.7	203.6
2003	18.7	60	105	311.3	177.9
2004	20.9	61	115	341.9	181.3
Southern Nova Scotia					
2000	38.9	45	72	863.8	539.8
2001	36.8	128	165	287.1	222.7
2002	36.7	111	183	330.9	200.7
2003	11.2	47	73	237.8	153.1
2004	32.3	157	259	205.7	124.7

[a] Unadjusted fisher-weeks is the summed number of weekly pick-ups from fishers.

[b] For adjusted fisher-weeks, in weeks when there is no pick-ups, fisher-weeks is the number of fisher pick-ups the following week.

Table 7.3
Size of elver runs at East River-Sheet Harbour and East River-Chester, Nova Scotia.

Year	East River-Sheet Harbour		East River-Chester	
	Estimate	95%CI	Estimate	95%CI
1990	218,300			
1991	376,000			
1992	219,200			
1993	134,100			
1994	309,900	10,900		
1995	101,500	1,600		
1996	336,500	10,100	1,138,100	24,200
1997	467,400	7,000	1,419,000	52,100
1998	109,200	2,000	432,400	8,200
1999	134,600	600	441,700	9,800
2000			791,204	
2001			608,377	
2002			1,715,009	

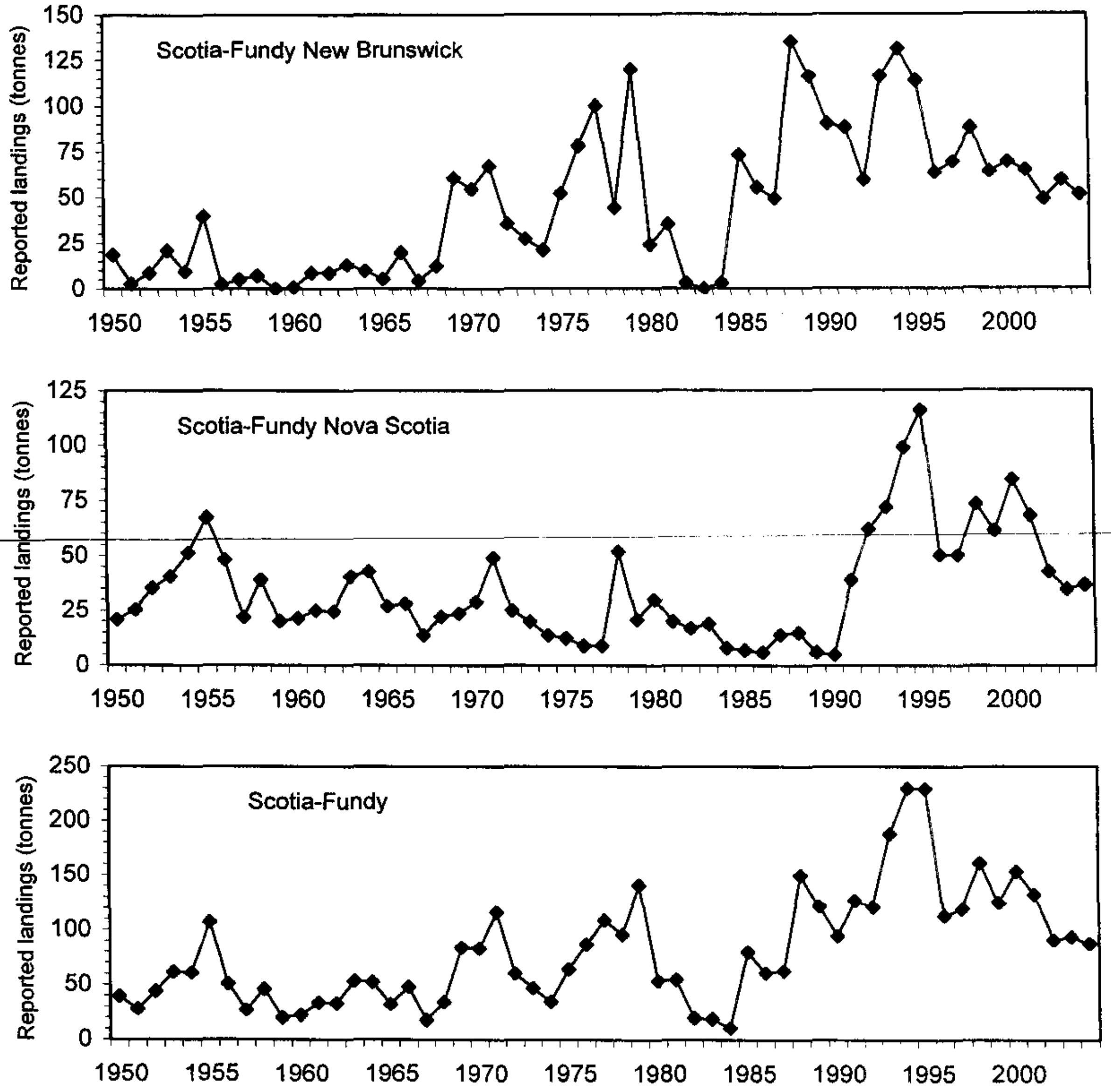


Fig. 7.1
Reported landings of American eels in the Scotia-Fundy area.

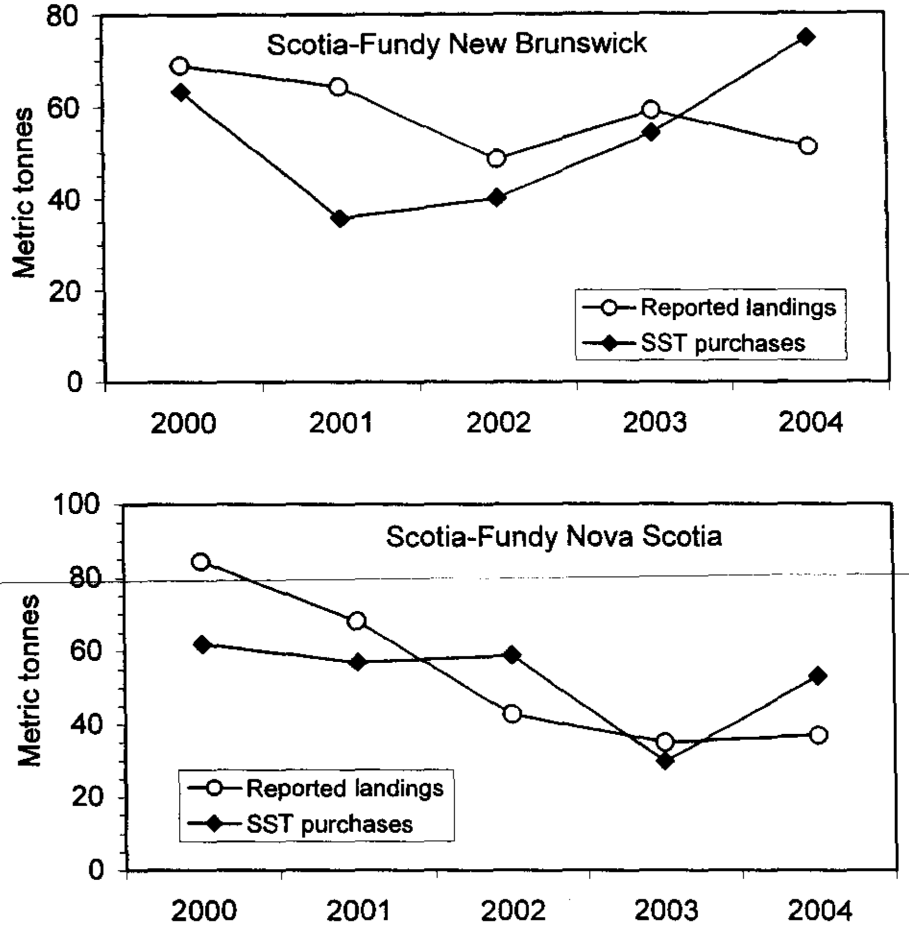


Fig. 7.2
Reported landings of American eels in Scotia-Fundy New Brunswick and Scotia-Fundy Nova Scotia, compared to purchases by South Shore Trading Co. South Shore Trading data for Nova Scotia may include some purchases from the southern Gulf portion of Nova Scotia.

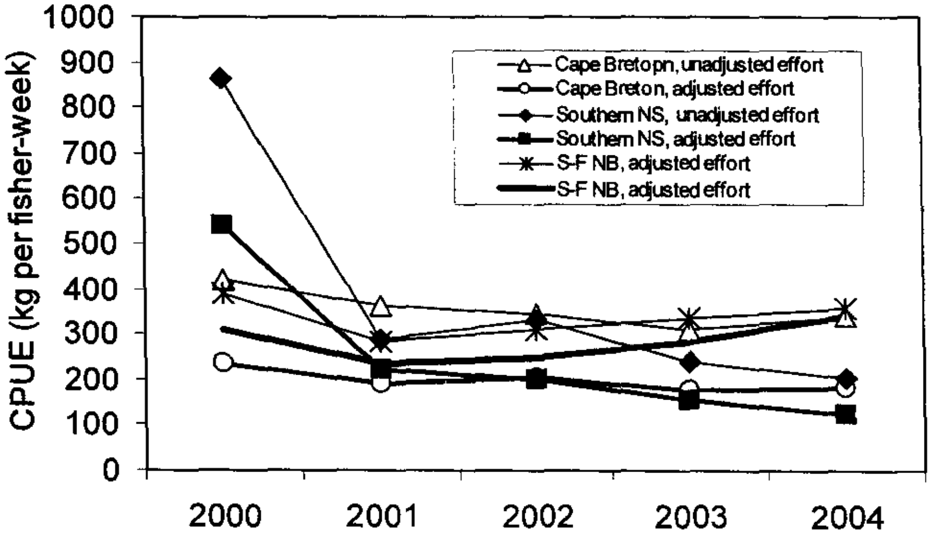
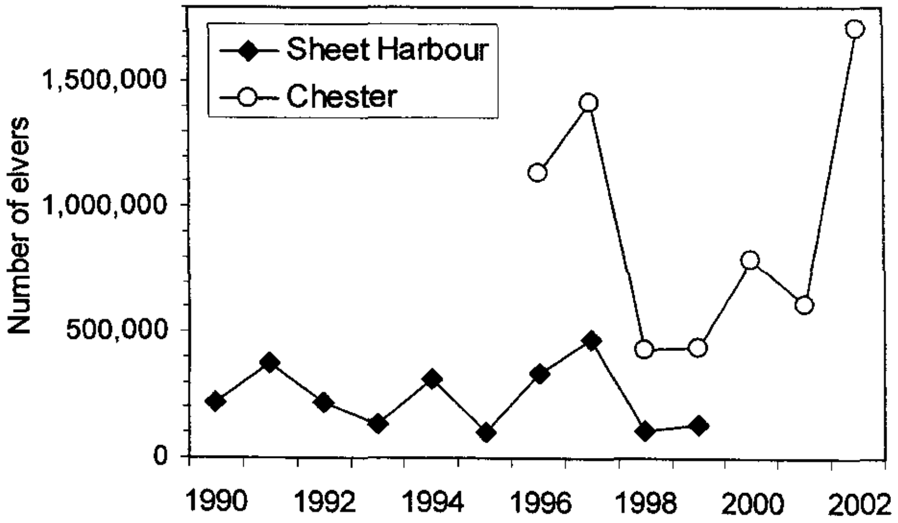


Fig. 7.3
Trends in catch per unit effort in the eel fishery in Cape Breton Island, southern Nova Scotia, and Scotia-Fundy New Brunswick, from purchase records of the South Shore Trading Company. See Table 8.2 for definitions of unadjusted and adjusted effort.



Fi.g 7.4
Size of elver runs at East River-Sheet Harbour and East River-Chester, Nova Scotia.

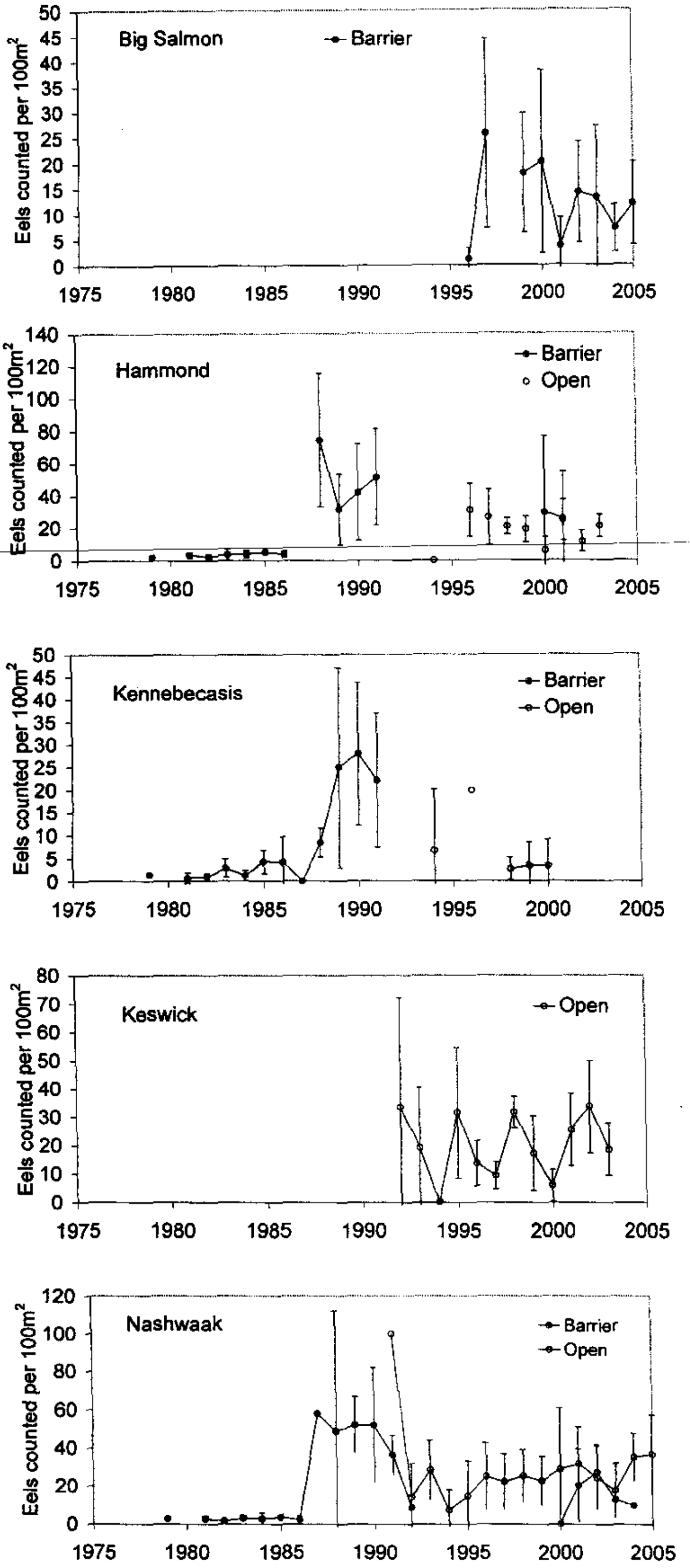


Fig. 7.5
Eels counted per 100 m^2 at barriered and open electrofishing sites in Scotia-Fundy New Brunswick. All sites are in the Saint John River drainage except Big Salmon River.

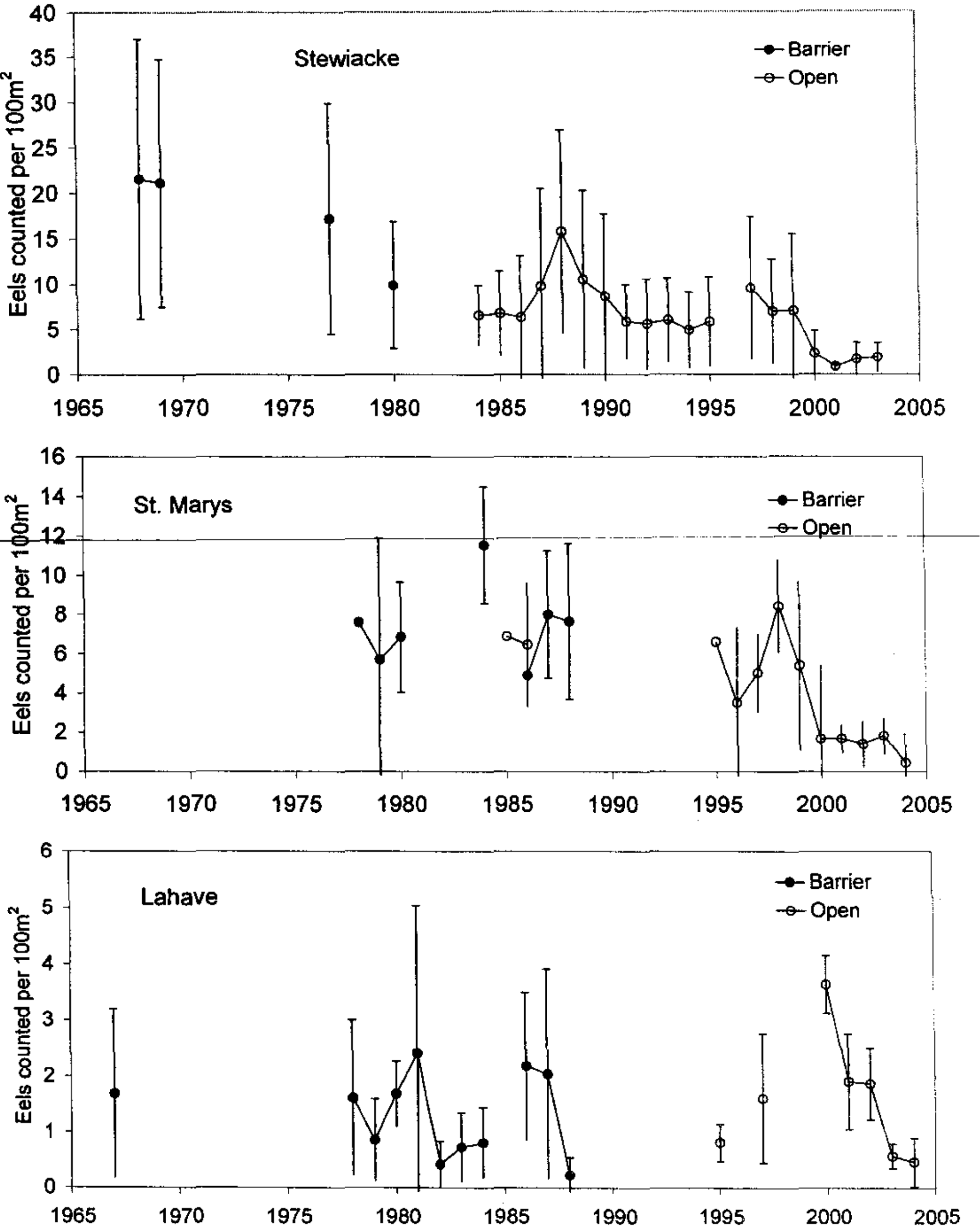


Fig. 7.6
Eels counted per 100 m^2 at barriered and open electrofishing sites in Scotia-Fundy Nova Scotia.

8. EEL ABUNDANCE INDICATORS IN NEWFOUNDLAND AND LABRADOR

8.1 Background

The island of Newfoundland comprises DU4, Atlantic Islands (Fig. 3.1). The eel's range in southern Labrador is split between DU2, Eastern St. Lawrence, and DU5, Eastern Arctic.

8.2 Population indicators

8.2.1 Commercial CPUE

In Newfoundland eels are fished primarily in inland waters (Table 3.2) in the western and northeastern parts of the island. There is minimal commercial fishing on the South Coast and no landings have been reported from Labrador since 1993.

Data on eel landings are collected through a purchase slip system. Total landings showed periodic spikes during the 1960s to the 1980s, and then climbed to the largest peak in 1990 (Figs. 8.2-8.3). Reported landings subsequently declined, and are now less than 1/2 of those of the peak period.

8.2.2 *Electrofishing*

Northeast Brook, located in Trepassey, Newfoundland (46°46' N 53°21' W) was part of the Experimental Rivers Program conducted in Newfoundland from the mid 1980s to the mid 1990s. The main objective of this program was to elucidate the optimum egg deposition rate for juvenile Atlantic salmon given the local habitat conditions. As part of this program salmonid densities were monitored throughout Northeast brook. Methods employed varied but the longest term data set are for riverine stations which were sampled via electrofishing. Each station was isolated with barrier nets and depletion electrofishing conducted (minimum 3 sweeps). During this sampling protocol all species were recorded, including American eel, which was also measured for total length. It is important to note that this area does not have any commercial eel fisheries and thus this data should not be affected by any local harvesting.

While numerous sites were sampled over the course of study, five electrofishing stations were visited on a consistent basis and these have been selected for this trend analysis. Four of the sites are located on the mainstem of Northeast Brook; a fifth (station 8) was located on a small tributary (tributary 1) upstream of Miller's Pond. American eel presence was inconsistent over the study period in this small tributary and although the catch data for station 8 is included below, it does not make up part of the average catch data presented in Fig. 8.3.

The four mainstem stations are described as follows:

- Station 5 is a riffle site located on the lower mainstem. This site was sampled every year from 1984-1996 in summer conditions (July/August).
- Station 18 is a flat type habitat located on the lower mainstem. This site was sampled every year from 1984-1996 in summer conditions (July/August).
- Station 19 is a riffle site located above station 18 but still in the lower mainstem. This site was sampled every year from 1986-1996 in summer conditions (July/August).
- Station 37 is a riffle habitat located on the upper mainstem (i.e. above Miller's Pond). This site was sampled every year from 1984-1996 in summer conditions (July/August).

In addition to these stream sites, Miller's Pond was sampled via Fyke nets from 1986 to 1994 (excluding 1993) during summer conditions. These Fyke net catches utilized a mark recapture methodology to determine salmonid density; again all eels captured were recorded and measured for total length.

Trends: Data are presented as catch per unit effort in (Fig. 8.3) where the effort is one electrofishing station. Given the same stations were sampled in the same manner every year; this method of presentation should be consistent over the time series. American eel abundance was relatively stable within Northeast Brook from 1984 to 1990 (Fig. 8.3). With the exception of 1986, which had low eel abundance for this time period, average catches of eels ranged from a low of 4.5 per station to a high of 8 per station. Data post-1990, with the possible exception of 1992, appears to show lower overall abundance of eels in the electrofishing stations. Average eel catches in this period, again excluding 1992, ranged from 1 to 2.8 eels per station.

American eels also readily utilize lacustrine habitat and although the data collected from Miller's pond does not have as long a time series as the electrofishing data, it is important to consider in the overall abundance for the catchment. Fig. 8.4 presents the total number of eels captured in Fyke nets during salmonid population estimates conducted during summer conditions in Miller's Pond. Again, during the period of 1986 to 1991 captures were consistent, even rising slightly through this period. The data collected in the pond and stream habitat are somewhat inconsistent during 1991 and 1992. Stream captures in 1991 were relatively low while they were the highest observed in Miller's Pond. The opposite was observed in 1992 where no eels were caught in the pond but stream captures were more similar to historical levels. Captures in both major habitat types were low during 1994 and unfortunately the lake work was discontinued after 1994 and was not conducted during 1993.

Individual Stations: Station 5 was the closest site to the ocean that was sampled on a consistent basis. American eel was always present with peak abundances being observed from 1987 to 1990 (Fig. 8.5). The abundances observed during these peak years were generally twice that observed in the other years. Abundance of eels did decline steadily from 1991 through

1994 which consistent with the overall trend in the average data (Fig. 8.3).

Station 18 had the highest abundance of eel in the first two years of sampling (1984 and 1985; Fig. 8.6). With the exception of these years, eel abundance was relatively stable until 1994 when no eels were captured at this site. Capture rates remained low during the subsequent two years (1995 and 1996).

Station 19 was not added to the sample program until 1986 and showed a more sporadic trend in eel abundance than the other two lower mainstem sites (Station 5 and 18; Fig. 8.7). Eels were always captured at this site excepting the last year of study (1996) and while there was no clear temporal trend in the data eel abundance actually tended to be higher during the 1990's than 1980's in the site, which is contrary to the overall trend with respect to the average data presented in Fig. 8.3.

Station 37 was the only station on the mainstem above Miller's Pond that was sampled on a consistent basis. With the exception of 1995 eels were always present at this site and in general eel abundance was slightly higher during the 1980's than the 1990's with the exception of 1992.

Station 8 was also located above Miller's Pond but was on a small tributary flowing from the. Eel presence was sporadic at this station throughout the early sampling period but no eels were caught at this site post 1990. While this data is not included in the average trend data (Fig. 8.3) it lends more evidence to the trend of an overall reduced eel abundance in this catchment during the early part of the 1990's, as compared to that observed during the 1980's.

<u>Summary</u> Overall American eel abundance appears to be lower post 1990 as compared to that observed from 1984 to 1990 in Northeast Brook, Trepassey. Data within the streams and lake are somewhat inconsistent for two years, 1991 and 1992 but the data during the later portion of the sampling was consistently low. It is difficult from this type of data to ascertain what is the 'norm', it is possible that eel abundance was at a natural peak during the mid 1980's but at the same time this abundance trend coincides with many other tends observed in the North Atlantic over the same time period. This data would supply a good backdrop for any future monitoring of eel abundance for this part of their range as the sites are well documented and no fishery exists locally.

Table 8.1

Reported landings (metric tonnes) and landed values of American eels in Newfoundland and Labrador, 1961-2004. Data from DFO Statistics Branch except where noted. Data for 1961-1995 as compiled by Knight (1997).

Landing year	Recruit-ment year[a]	Reported landings (metric tonnes) by district																			
		Northeast and east coasts							South coast				West coast						Labrador	Total	Total[b]
		A	B	C	D	E	F	Total	G	H	I	Total	J	K	L	M	N	Total	O		
1961	1950							0.0				0.0		0.3				0.3		0.3	
1962	1951				12.5	4.6		17.1	3.4	1.8	1.2	6.4						0.0		23.5	
1963	1952			0.2	8.8			9.0	3.0	14.8	10.2	28.0						0.0		37.0	
1964	1953			1.3				1.3		3.7	7.0	10.7	1.0					1.0		13.0	
1965	1954							0.0		2.4	0.9	3.3						0.0		3.3	
1966	1955							0.0				0.0						0.0		0.0	
1967	1956							0.0				0.0						0.0		0.0	
1968	1957							0.0				0.0						0.0		0.0	
1969	1958							0.0				0.0						0.0		0.0	
1970	1959							0.0				0.0						0.0		0.0	
1971	1960							0.0	1.0			1.0	4.0	36.0			3.0	43.0		44.0	
1972	1961							0.0				0.0	17.0	52.0	10.0			79.0		79.0	
1973	1962				4.0			4.0				0.0	5.0	22.0				27.0		31.0	
1974	1963							0.0				0.0	19.0	2.0				21.0		21.0	
1975	1964				0.5			0.5				0.0	7.1	0.3				7.4		7.4	
1976	1965	3.1	1.5					4.6				0.0	6.2	0.5				6.7		11.3	
1977	1966	6.2	0.5				2.8	9.5		2.1		2.1		7.7				7.7		19.3	
1978	1967	0.3				0.9		1.2				0.0		14.5				14.5		15.7	
1979	1968	0.4				1.2		1.6	2.1			2.1		19.6	0.1			19.7		23.4	
1980	1969	0.3	0.3	0.5	3.0	2.9	2.8	9.8	2.4			2.4	0.0	44.9	0.1	25.5		70.5		82.7	
1981	1970		4.6	2.0	0.7	0.5	0.1	7.9	1.1	0.3	1.1	2.5	3.7	13.5	0.1	13.9		31.2		41.5	
1982	1971		8.5	7.1				15.6				0.0		20.3	0.1	0.7		21.1		36.7	
1983	1972		5.4	3.3				8.7				0.0	2.2	16.9	0.2			19.3		28.0	
1984	1973			12.1				12.1				0.0		1.3	0.6			1.9		14.0	
1985	1974		0.4	9.0				9.4		0.1		0.1		2.7	6.9	1.6		11.2	4.3	25.0	
1986	1975		0.2	8.2	0.5	1.8		10.7	0.5			0.5		7.9	0.5	5.3	1.6	15.3		26.6	77
1987	1976		6.5	11.9	0.2	4.1		22.7	1.1	0.1		1.2		5.4	1.1	0.2		6.7		30.6	58
1988	1977	0.1	16.0	27.6	1.0	8.5		53.2		3.4	1.3	4.7		1.8	0.8	0.4		3.0		60.8	116
1989	1978	0.8	11.5	21.5	2.3	4.5		40.6		0.4	0.5	0.9		24.1	0.6	8.1	9.1	41.9		83.5	
1990	1979	5.3	26.1	27.2	4.8	10.3		73.7	0.5	2.4	0.2	3.1	4.1	40.0	1.0	15.7	9.3	70.1		146.6	
1991	1980	3.7	23.0	14.4	3.9	5.8		50.8	0.3	4.2	0.9	5.4	3.3	62.5	0.5	4.6	6.8	77.7		133.9	
1992	1981	0.4	13.6	11.1	5.2	4.0		34.3	1.3	2.9		4.2	5.3	40.0	0.4	4.5	1.3	51.5		89.9	
1993	1982	3.6	18.1	18.9	4.2	6.5		51.3	1.9	1.5	0.4	3.8	22.3	26.3	2.3	6.9	3.0	60.8	0.1	116.1	
1994	1983	3.8	18.4	15.3	6.3	5.4		49.2	2.7	4.7	1.3	8.7	6.0	31.1	0.3	11.1	4.6	53.1		110.9	
1995	1984	3.8	14.7	9.0	6.1	5.2	0.4	39.2	0.8	1.6	1.7	4.1	2.3	23.2	1.3	9.0	6.3	42.1		85.4	
1996	1985	4.18	23.81	9.13	5.65	5.83	0.89	49.48	1.07	3.36	3.00	7.43	3.51	18.34	0.42	7.03	8.21	37.51		94.41	
1997	1986	1.93	14.74	10.00	3.84	6.04	1.36	37.91	0.64	4.02	2.17	6.82	2.27	8.45	1.74	9.43	5.12	27.01		71.74	
1998	1987	2.04	17.47	12.98	4.60	6.35	1.45	44.90	0.35	2.46	1.33	4.13	1.56	9.31	0.14	8.53	4.38	23.92		72.95	
1999	1988	1.39	12.88	12.40	3.40	6.41	1.39	37.86		1.98		1.98	1.88	6.27		3.26	3.27	14.68		54.53	
2000	1989	7.36	11.59	16.58	3.70	4.69	0.72	44.65	0.85	2.81		3.65	2.79	1.40		12.98	4.32	21.49		69.79	
2001	1990	0.31	8.43	9.58	1.97	4.73	0.12	25.14	0.86	0.79		1.66	3.27	4.13		1.63	0.85	9.90		36.69	
2002	1991	1.53	8.77	12.70	1.30	6.37	0.94	31.62	0.32			0.32	3.51	16.60		9.35	4.09	33.55		65.49	
2003	1992	0.00	14.15	5.62	1.60	9.15		30.52	1.03	0.60		1.63	0.19	14.03	10.21	4.63	3.35	32.42		64.57	
2004	1993		9.36	7.64	0.93	3.07		21.01	0.34	1.35		1.68	6.29	12.41			1.85	20.56		43.25	

[a] Based on a mean age of 11 years, which is the mean of mean ages of eels sampled in Newfoundland by pots and fyke nets (Table 3.2)

[b] Data from Newfoundland Department of Fisheries; 1986-1988 only

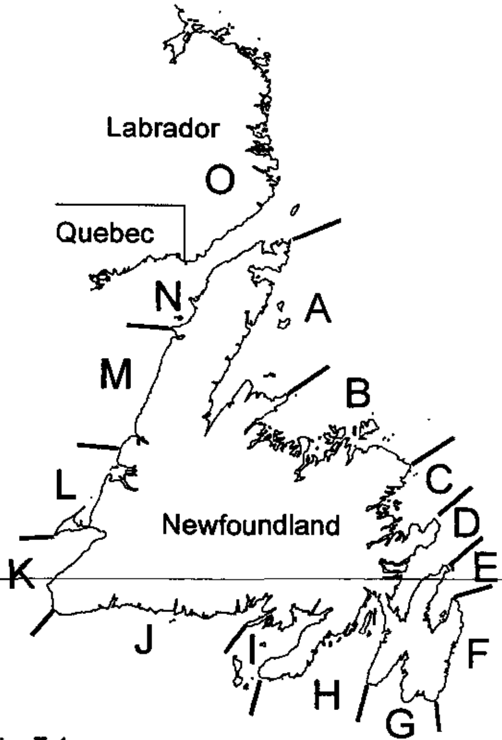


Fig. 7.1
Districts used to record eel landings in Newfoundland and Labrador.

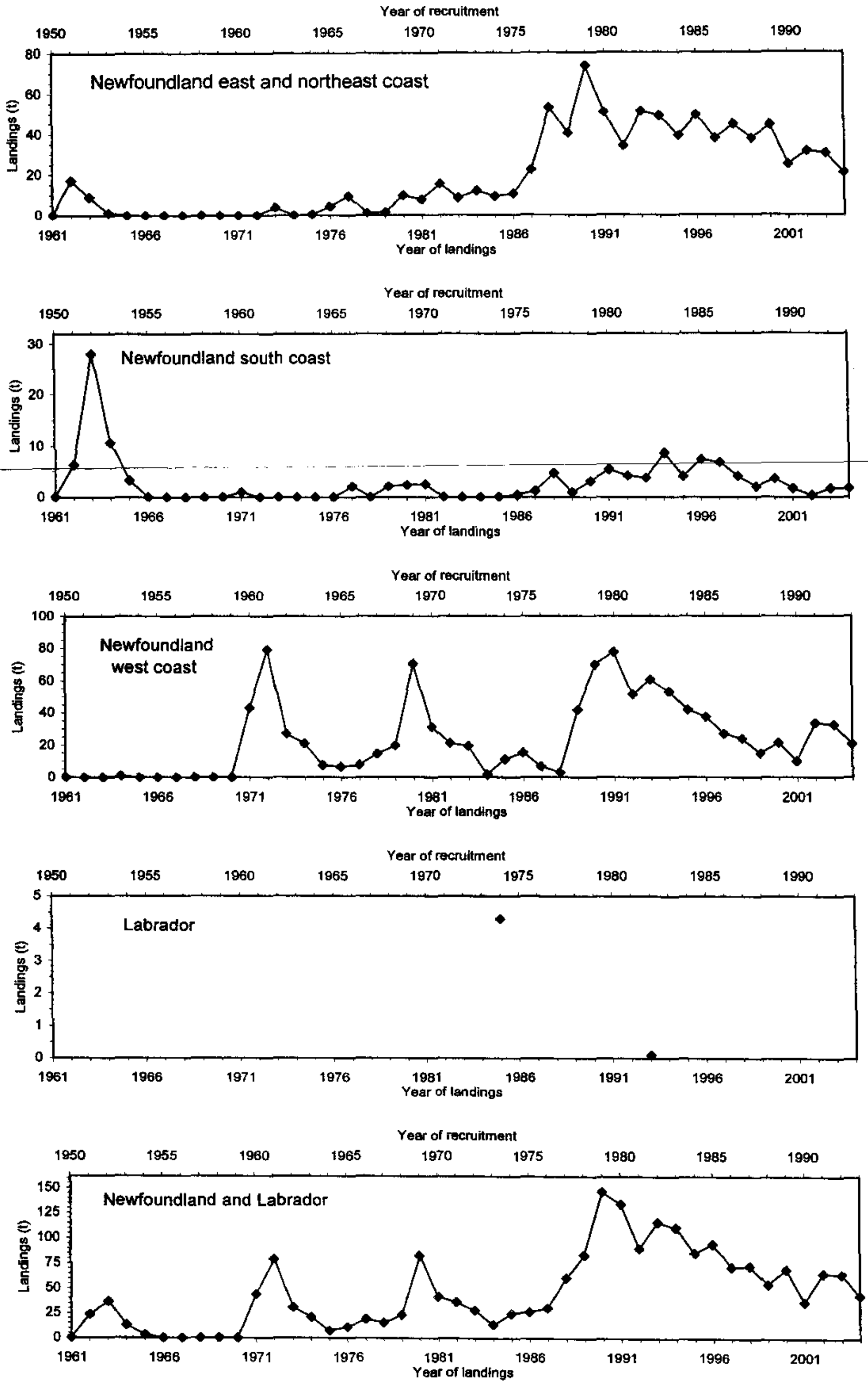


Fig. 7.2
Reported American eel landings in Newfoundland, against year of landings and against year of recruitment. Year of recruitment is based on a mean age of 11 (Table 3.2).

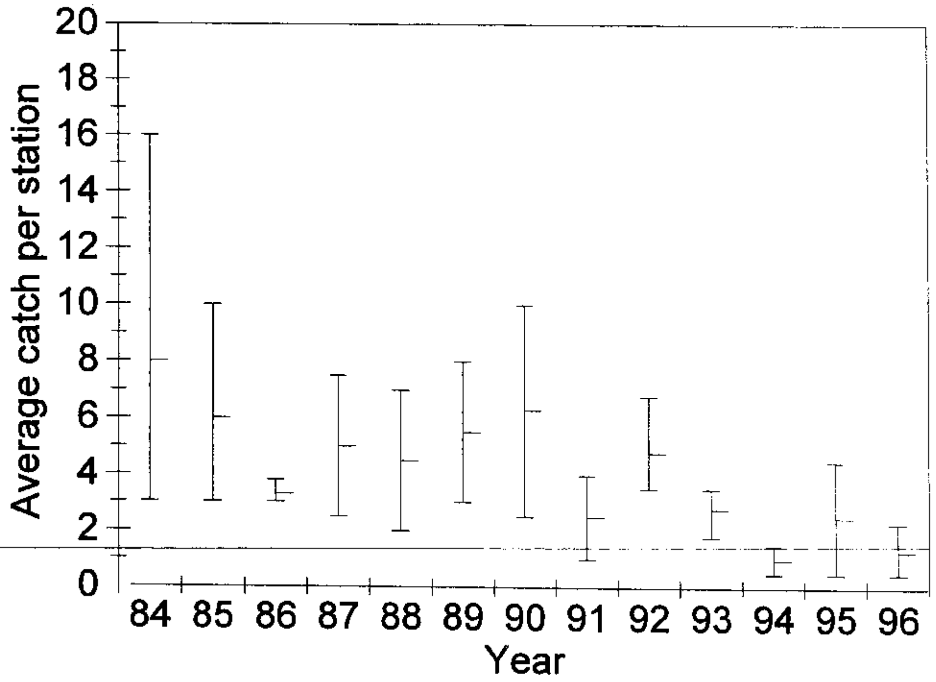


Fig. 8.3:
Average American eel catches in mainstem electrofishing stations of Northeast Brook, Trepassey. Bars are 95% confidence intervals of the mean, calculated via randomization with replacement, 1000 trials.

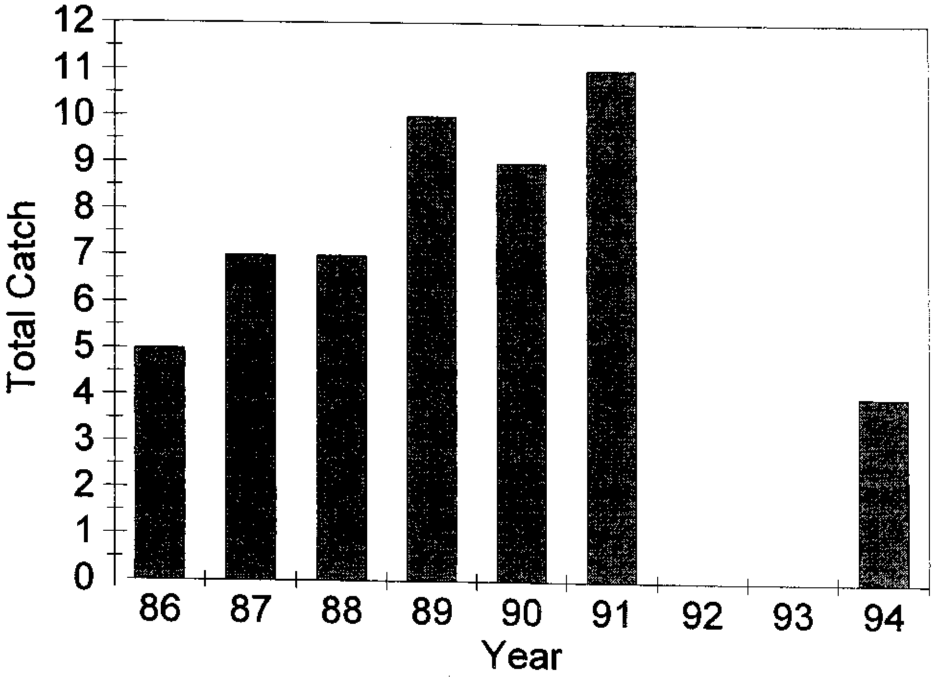


Fig. 8.4:
Total catch of American eel in Miller's Pond during summer population estimates conducted for salmonids. Note estimates were not conducted in 1993, there were not eels captured during sampling in 1992.

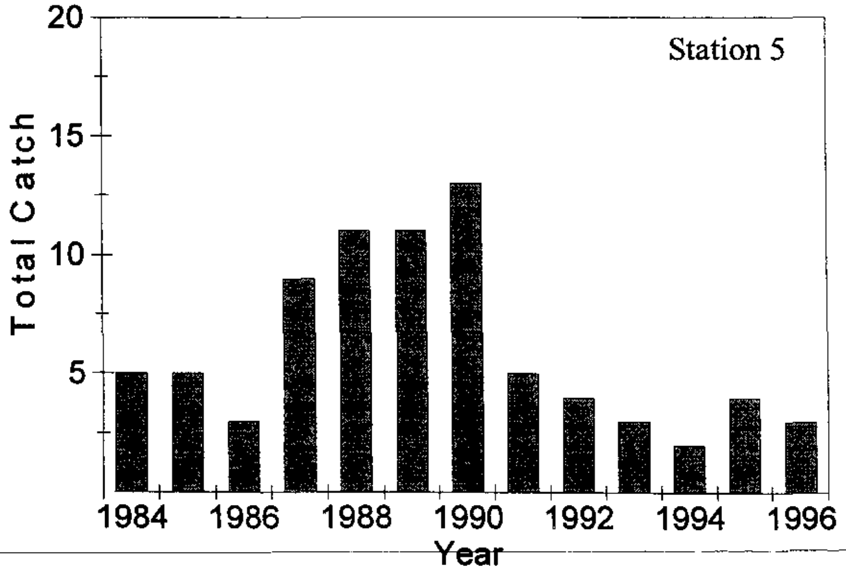


Fig. 8.5:
Total catch data for Station 5.

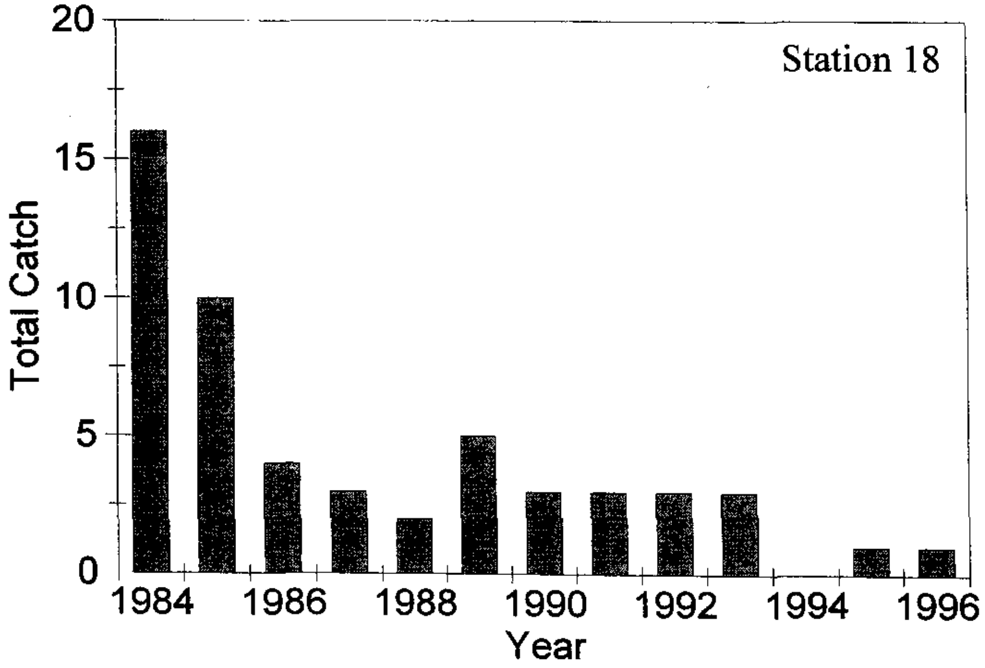


Fig. 8.6:
Total catch data for Station 18.

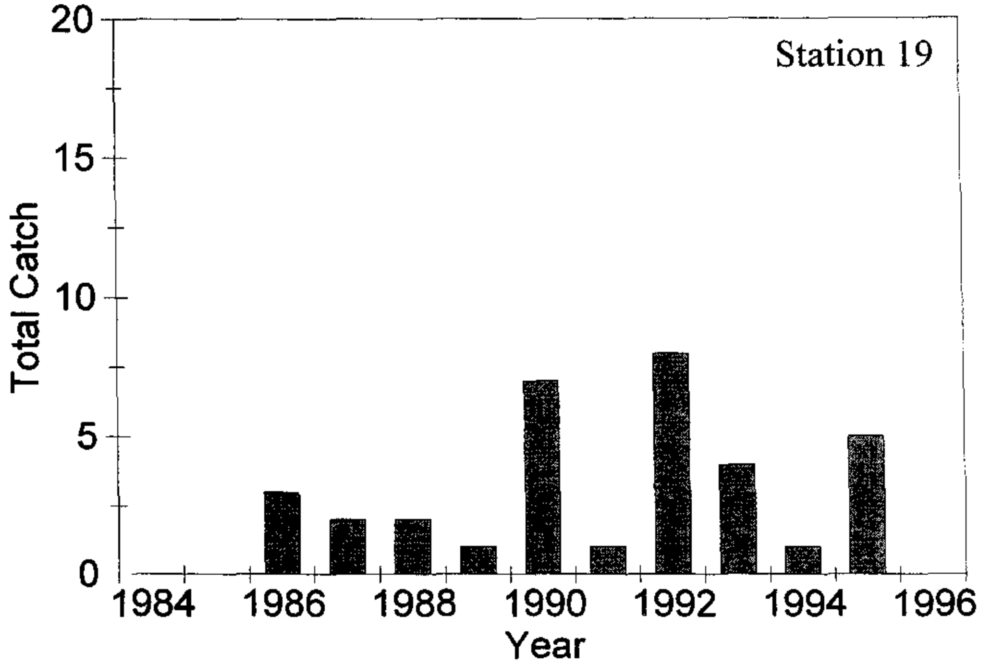


Fig. 8.7:
Total catch data for Station 19.

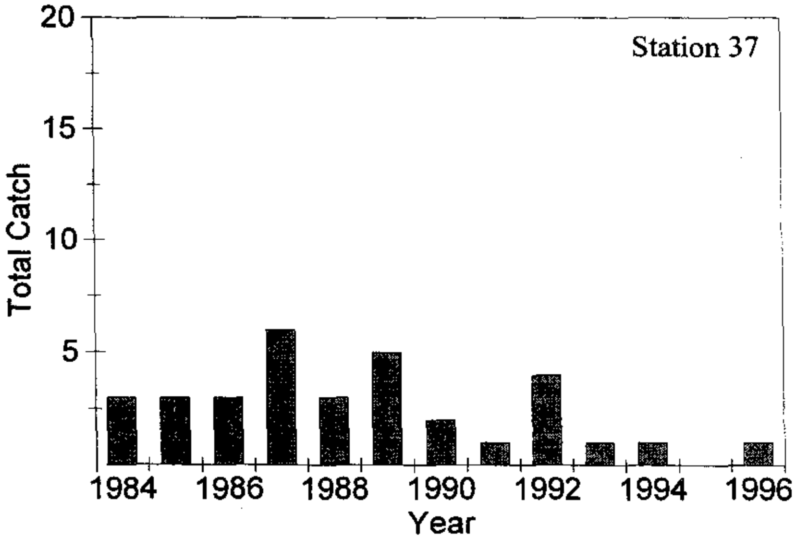


Fig. 8.8:
Total catch data for station 37.

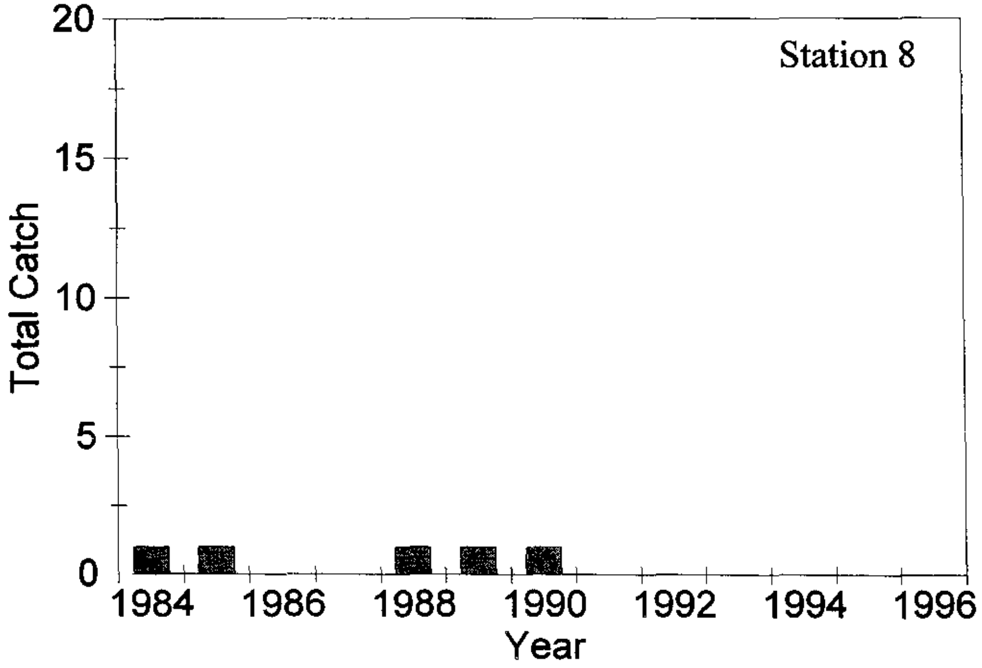


Fig. 8.9:
Total catch data for Station 8.

9. EEL POPULATION TRENDS IN CANADA AND THEIR POSSIBLE CAUSES

9.1 Commercial landings

9.2 Regional and national comparisons of population indicators

9.3 Possible reasons for population changes

Table 9.1

Reported landings of American eels in North America, in metric tonnes. US landings from http://www.st.nmfs.gov/st1/commercial/landings/annual_landings.html. Some data for 2004 are preliminary.

Reported landings of American eels in North America, in metric tonnes. US landings from http://www.st.nmfs.gov/st1/commercial/landings

Year	On-	Quebec			Southern Gulf of St. Law.				Scotia-Fundy			Nfld. & Lab.			Canada	US	North
	tario	Yellow	Silver	Tot.	NB	NS	PEI	Tot.	NS	NB	Tot.	Nfld	Lab	Tot.			America
1917					51	13	0	64									
1918					62	13	0	75									
1919					75	6	0	81									
1920		8	267	275	24	11	0	35									
1921		10	300	310	41	15	0	56									
1922		24	426	450	14	8	0	22									
1923		55	500	555	10	1	0	11									
1924		80	457	537	10	8	0	18									
1925		45	487	532	18	4	0	23									
1926		52	900	952	5	6	0	11									
1927		49	550	599	1	4	0	5									
1928		68	913	982	16	7	0	23									
1929		47	474	521	5	5	0	10									
1930		66	511	578	12	15	0	27									
1931		76	693	769	19	12	0	31									
1932		88	755	843	9	14	0	23									
1933		97	991	1,088	11	17	0	28									
1934		103	906	1,008	11	6	0	17									
1935		101	907	1,008	8	5	0	13									
1936		97	872	969	4	4	0	9									
1937		57	768	825	6	4	0	10									
1938		59	778	837	9	10	0	19									
1939		38	730	768	11	9	0	20									
1940		21	377	398	5	11	0	15									
1941		16	55	71	4	3	0	7									
1942		27	441	468	13	5	0	17									
1943		51	585	635	14	5	0	19									
1944		20	272	293	14	7	0	21									
1945		39	346	385	15	11	0	25									
1946		23	298	321	29	17	0	46									
1947		22	299	322	32	14	1	46									
1948		12	210	222	29	9	10	48									
1949		9	175	184	29	38	4	71									
1950	13	10	289	299	22	24	2	48	19	21	39					955	
1951	21	12	339	350	16	21	4	40	3	25	28					833	
1952	29	13	378	391	16	12	5	33	9	35	44					734	
1953	26	12	390	403	13	8	6	27	21	41	62					641	
1954	35	15	338	352	33	6	4	43	10	51	61					546	
1955	31	19	382	400	49	11	9	68	40	67	107					628	
1956	19	14	380	394	11	15	5	30	2	48	51					656	
1957	45	15	545	560	9	10	12	31	5	22	27					572	
1958	53	23	454	477	15	14	19	47	7	39	46					631	
1959	55	20	368	388	24	11	26	61	0	20	20					603	
1960	50	20	442	462	31	24	32	86	1	22	22					405	
1961	59	23	359	382	57	28	18	103	9	25	34	0	0	0	577	380	957
1962	49	29	354	383	82	26	13	121	9	25	33	24	0	24	610	290	901
1963	76	29	440	468	54	24	16	93	13	41	54	37	0	37	728	440	1,168
1964	111	30	417	447	56	19	34	109	10	43	53	13	0	13	733	472	1,204
1965	85	30	518	548	63	16	49	128	5	27	32	3	0	3	796	709	1,505
1966	64	28	459	488	99	15	33	147	20	28	48	0	0	0	747	580	1,327
1967	61	27	408	436	108	52	62	222	4	14	18	0	0	0	737	724	1,461
1968	78	30	468	498	151	28	131	310	12	22	34	0	0	0	920	769	1,688
1969	76	28	485	513	214	38	195	447	60	24	84	0	0	0	1,120	849	1,969
1970	66	10	304	314	295	45	240	580	54	29	83	0	0	0	1,043	979	2,021
1971	76	6	307	313	319	52	351	723	67	49	116	44	0	44	1,272	1,109	2,381
1972	122	30	279	309	273	50	273	596	35	25	60	79	0	79	1,166	712	1,878
1973	85	22	278	301	220	28	157	406	27	20	47	31	0	31	869	591	1,461
1974	100	28	360	389	156	28	101	286	21	14	35	21	0	21	830	1,388	2,218

Table 9.1 (continued)

Year	On-	Quebec			Southern Gulf of St. Law.				Scotia-Fundy			Nfld. & Lab.			Canada	US	North
	tario	Yellow	Silver	Tot.	NB	NS	PEI	Tot.	NS	NB	Tot.	Nfld	Lab	Tot.			America
1975	166	27	498	525	121	29	104	253	52	12	64	8	0	8	1,016	1,607	2,624
1976	154	34	384	418	119	18	94	231	78	9	87	11	0	11	901	1,117	2,018
1977	186	24	483	507	110	6	98	214	100	9	109	19	0	19	1,035	953	1,988
1978	229	29	497	525	82	12	114	208	44	52	96	16	0	16	1,073	1,615	2,689
1979	222	28	477	505	102	13	111	226	120	21	141	23	0	23	1,117	1,793	2,910
1980	164	25	570	595	150	10	120	280	24	30	54	83	0	83	1,176	1,459	2,635
1981	108	31	569	600	191	8	220	419	35	20	55	42	0	42	1,223	1,369	2,593
1982	29	25	357	381	159	11	168	338	3	17	20	37	0	37	805	1,005	1,810
1983	76	25	327	352	97	10	151	258	0	19	19	28	0	28	732	839	1,571
1984	122	31	381	412	122	9	165	296	3	8	11	14	0	14	855	1,117	1,972
1985	104	0	389	389	202	5	139	347	73	7	80	21	4	25	945	828	1,773
1986	116	28	469	497	230	16	226	472	55	6	61	27	0	27	1,172	1,001	2,173
1987	103	21	405	426	172	13	150	335	49	14	63	31	0	31	957	718	1,675
1988	105	20	414	434	234	25	125	383	135	15	150	61	0	61	1,133	573	1,707
1989	121	28	405	433	209	30	70	309	116	6	122	83	0	83	1,069	747	1,815
1990	119	34	440	474	149	21	124	294	90	5	95	147	0	147	1,129	695	1,824
1991	117	29	366	395	130	35	127	292	88	39	127	134	0	134	1,065	753	1,818
1992	123	21	298	319	120	56	54	230	67	62	129	90	0	90	891	660	1,550
1993	105	19	309	328	88	89	74	252	116	72	188	116	0.1	116	989	712	1,701
1994	82	21	262	282	68	42	46	156	131	99	230	111	0	111	862	706	1,567
1995	62	23	255	278	60	16	35	111	114	116	230	85	0	85	767	594	1,361
1996	57	29	196	226	49	11	36	96	102	72	174	94	0	94	647	459	1,106
1997	43	26	175	201	36	17	31	85	111	64	175	72	0	72	576	427	1,003
1998	21	22	193	215	49	15	24	88	88	75	163	73	0	73	560	461	1,022
1999	21	19	149	167	47	9	35	91	119	76	195	55	0	55	529	459	988
2000	29	36	166	202	76	7	63	147	69	85	153	70	0	70	602	391	992
2001	29	34	149	183	92	3	41	137	64	68	133	37	0	37	518	394	912
2002	12	34	136	170	115	4	86	206	48	43	91	65	0	65	545	291	836
2003	13	30	107	138	144	9	70	223	59	35	94	65	0	65	532	468	1,000
2004	0				122	3	62	187	51	37	88	43	0	43			

Table 9.2

American eel indices against year of recruitment to continental waters. Landings are in metric tonnes. NAO data are from http://www.cgd.ucar.edu/cas/jhurrell/Data/naodjfmindex.1864-2004.xls

Recruitment year	DU1								DU2	DU3								Winter NAO
	Ontario				Quebec				Quebec	Quebec	Southern Gulf					Scotia Fundy		
	Ontario landings	Bay of Quinte geometric mean eels/ trawl	Lake Ontario eels e-fished/ hour	Moses-Saunders eels/ day in peak period	Beauharnois fishway count, Group 37	St. Nicholas trap fall counts	St. Law. estuary silver landings	Sud-Ouest River fish fence counts	Petite Trinité visual counts upstream migrants	St. Jean e-fishing count/ 100 m^2	Restigouche e-fishing eels/ 100 m^2	Miramichi e-fishing eels/ 100 m^2	Margaree e-fishing eels/ 100 m^2	PEI logs legal eels kg/net/ day	PEI logs sublegal eels/ net/ day	East River Chester elver counts	East River Sheet Harbour elver counts	
Stage	Yel.	Yel.	Yel.	Yel.	Yel.	Mostly sil.	Sil.	Sil.	Small yel.	Yel.	Yel.	Yel.	Yel.	Yel,sil	Yel.	Elver	Elver	
Age	11.0	11	11.0	6-12	11	17	17.0	17	2	9	5	6	8	6	5	0	0	
1903							209.5											3.89
1904							239.5											0.23
1905							370.2											1.98
1906							450.5											2.06
1907							407.7											2.06
1908							425.1											1.44
1909							828.5											0.00
1910							483.4											2.10
1911							838.4											0.29
1912							397.5											0.24
1913							435.1											2.69
1914							588.5											1.48
1915							586.5											-0.20
1916							816.8											-0.69
1917							714.5											-3.80
1918							715.3											-0.80
1919							687.5											-0.80
1920							622.1											3.18
1921							626.6											1.63
1922							625.4											1.85
1923							296.7											1.73
1924							21.8											-1.13
1925							404.6											2.39
1926							526.6											0.11
1927							246.1											1.72
1928							263.4											0.63
1929							241.3											-1.03
1930							265.9											0.91
1931							185.3											-0.16
1932							139.7											-0.50
1933							244.8											0.25
1934							300.6											0.86
1935							350.5											0.97
1936							356.0											-3.89

Table 9.2 (continued)

Recruit-ment year	DU1								DU2	DU3								Winter NAO
	Ontario				Quebec				Quebec	Quebec	Southern Gulf					Scotia Fundy		
	Ont-ario land-ings	Bay of Quinte geometric mean eels/ trawl	Lake Ontario eels e-fished/ hour	Moses-Saunders eels/ day in peak period	Beau-harnois fishway count, Group 37	St. Nicholas trap fall counts	St. Law. estuary silver landings	Sud-Ouest River fish fence counts	Petite Trinité visual counts upstream migrants	St. Jean e-fishing count/ 100 m^2	Resti-gouche e-fishing eels/ 100 m^2	Mira-michi e-fishing eels/ 100 m^2	Marg-aree e-fishing eels/ 100 m^2	PEI logs legal eels kg/net/ day	PEI logs sublegal eels/ net/ day	East River Chester elver counts	East River Sheet Harbour elver counts	
1937							304.3											0.72
1938							336.1											1.79
1939	12.9						334.3											0.37
1940	21.4						499.2											-2.86
1941	29.2						399.7											-2.31
1942	25.6						298.5											-0.55
1943	35.1						389.7											1.48
1944	30.6						304.7											0.61
1945	18.6						285.5											1.64
1946	44.6						351.1					0.56						0.27
1947	53.1						311.1					1.31						-2.71
1948	55.4						406.4					0.30						1.34
1949	49.7						351.9					0.63	3.39					1.87
1950	58.6						315.5					0.48	1.75					1.40
1951	48.8						369.9					0.34	2.75					-1.26
1952	76.3						405.5					0.55	4.81					0.83
1953	110.6						283.1					1.47	4.25					0.18
1954	84.9					414	270.7					0.85	3.00					0.13
1955	64.4					297	209.8					0.37	5.13					-2.52
1956	61.0					225	231.0					0.13	5.66					-1.73
1957	77.8					209	266.1					0.43	2.01					1.52
1958	76.1					232	401.7					0.95	2.98					-1.02
1959	65.6					194	295.1					0.72	2.42					-0.37
1960	75.6					328	383.8					0.62	1.46					-1.54
1961	122.1	1.87				449	398.1					1.02	1.11					1.80
1962	84.7	1.62				273	376.3					0.88						-2.38
1963	99.9	1.00				187	451.7					0.52						-3.60
1964	166.3	1.54				176	434.5					0.33						-2.86
1965	154.0	1.29				199	258.8					1.88						-2.88
1966	186.4	1.06				234	243.3			0.20		1.44						-1.69
1967	228.7	0.42				166	297.7				0.43	1.23	2.59					1.28
1968	221.5	0.77		7,934		200	342.0			0.05	0.50	1.57	2.41					-1.04
1969	164.0	0.25		14,403		176	371.6			0.16	0.35	1.23	2.83					-4.89
1970	107.9	1.53		10,363		166	332.4				1.01	1.06	1.76					-1.89
1971	29.0	1.88		20,013		207	349.3				0.23	1.24	1.74					-0.96
1972	75.6	0.56		16,448		83	340.0				0.37	0.65						0.34
1973	122.4	0.33	85.6	18,977		160	384.4				0.28	0.16	0.00					2.52
1974	104.0	0.78	63.1	9,046		169	309.8				0.10	0.15	5.00					1.23
1975	116.1	0.87	82.9	13,473		177	248.2				0.48	0.37	0.18					1.63
1976	102.9	1.55	89.0	27,489		188	259.1				0.09	0.89	0.00					1.37

Table 9.2 (continued)

Recruitment year	DU1								DU2	DU3								Winter NAO
	Ontario				Quebec				Quebec	Quebec	Southern Gulf					Scotia Fundy		
	Ontario landings	Bay of Quinte geometric mean eels/ trawl	Lake Ontario eels e-fished/ hour	Moses-Saunders eels/ day in peak period	Beauharnois fishway count, Group 37	St. Nicholas trap fall counts	St. Law. estuary silver landings	Sud-Ouest River fish fence counts	Petite Trinité visual counts upstream migrants	St. Jean e-fishing count/ 100 m^2	Restigouche e-fishing eels/ 100 m^2	Miramichi e-fishing eels/ 100 m^2	Margaree e-fishing eels/ 100 m^2	PEI logs legal eels kg/net/ day	PEI logs sublegal eels/ net/ day	East River Chester elver counts	East River Sheet Harbour elver counts	
1977	105.3	0.30	68.8	26,103		200	221.4				0.09	0.90	0.00					-2.14
1978	121.5	0.95	93.0	9,074		208	217.4				0.51	0.47	0.47					0.17
1979	119.0	0.36	64.1	9,868		127	162.9			0.08		0.18	0.79					-2.25
1980	117.0	0.45	38.5	2,828		138	147.8		4,027	0.62	0.32	0.15						0.56
1981	123.0	0.58	44.4	4,755		205	169.2		3,643	0.17	0.48	0.17						2.05
1982	105.0	0.43	22.7	5,220		381	130.8	315	732	0.00	0.29	0.27						0.80
1983	82.3	1.16	30.0	3,233	24,721	190	138.8		581	0.25	0.76	0.07	1.09					3.42
1984	62.2	0.09	10.5		17,072	350	131.7	108			0.60	0.24	0.00					1.60
1985	56.7	0.36	14.9	144		239	115.2	68			0.36	0.00	0.00					-0.63
1986	43.4	0.08	7.3	57		257	94.0	60			0.21	0.15	0.14					0.50
1987	21.5	0.12	12.9	27	5,441	200					0.06	0.28	0.11					-0.75
1988	20.5	0.07	21.6	52	10,692						0.00	0.41	0.00					0.72
1989	29.5	0.05	9.4	18	6,881						0.31	0.07	0.19					5.08
1990	28.5	0.01	6.8	55	13,099						0.00	0.30	0.10	0.32			218,300	3.96
1991	12.3	0.01	3.4	40	10,503				1,178	0.15	0.00	0.20	0.00	0.29	0.99		376,000	1.03
1992	13.3	0.00	0.7	53	32,684				488		0.00	0.33	0.22	0.52	1.98		219,200	3.28
1993	0.0	0.00	0.5						3,440		0.12	0.47	0.15	0.92	2.51		134,100	2.67
1994	0.0								3,550		0.25	0.34	1.08	0.91	4.04		309,900	3.03
1995											0.27	0.73	0.12	0.64	1.98		101,500	3.96
1996											1.18	0.30	0.25	0.77	1.15	1,138,100	336,500	-3.78
1997											1.40	0.32	0.17	0.87	1.43	1,419,000	467,400	-0.20
1998											0.24	0.64		1.15	1.63	432,400	109,200	0.72
1999											0.49				2.06	441,700	134,600	1.70
2000																791,204		2.80
2001																608,377		-1.89
2002																1,715,009		0.76

Table 9.2
Hypotheses to explain patterns of population change in American eels in the upper St. Lawrence River and Lake Ontario (USLLO), and the St. Lawrence estuary and Gulf (SLEG). Unless otherwise stated, arguments assume that the American eel is a panmictic species.

Hypothesis	Arguments for	Arguments against
NOA hypothesis Patterns are explained at least in part by changes in the North Atlantic Oscillation index (NOA). High indices depress eel recruitment from the ocean; low indices enhance it.	The NAO has a pervasive influence on biological phenomena in the North Atlantic. Eel recruitment to the Netherlands is negatively correlated with NAO (Knights 2003). A downward trend in eel recruitment in the Gulf (indicated by Miramichi electrofishing data) coincided with an increase in the NAO (Fig. 9.7). The long-term decreasing trend of recruitment destined for the upper St. Lawrence (indicated by the Moses-Saunders index) in the 1970s and 1980s coincided with a long-term increase in the NAO (Fig. 9.7).	There is a reasonable correspondence between Gulf recruitment indicated by Miramichi electrofishing and the NAO. However, there is poor correspondence between recruitment destined for USLLO and the NAO. The greatest decline in recruitment destined for USLLO occurred in the late 1970s and early 1980s, about 10 years after the period of greatest increase of the NAO. Eels destined for both USLLO and SLEG must traverse the North Atlantic. If NAO controls recruitment to both areas recruitment patterns should be similar in the two areas. But the major recruitment decline in the Gulf came a decade earlier than that of the USLLO. Recruitment in the Gulf has been increasing since the 1980s, but recruitment destined for USLLO has remained very low.
Range-end density dependence (REDD) hypothesis Recruitment to the most distant rearing area (USLLO) is sensitive to density of arriving glass eels/elvers. When overall recruitment for the species is depressed, densities of glass eels and elvers in the ocean are low, which weakens the inducement for young eels to travel long distances to rearing grounds. In such circumstances recruitment to the extreme end of the range collapses. depends on density dependency theory	The near-abandonment of USLLO by the American eel can be explained by density-dependence. Eels make density dependent-decisions on what areas to colonize while they are still in the ocean, or as they enter coastal waters and estuaries. There is a generalized decline in the species population, so that densities of glass eels in the ocean, and densities in continental rearing areas, are lower than before. Because habitats close to the spawning ground have low densities, these areas are colonized preferentially. This leaves few eels to go USLLO, which has the greatest distance from the spawning ground of any major rearing area. Density-dependent dispersal in eels has been shown by Tsukamoto et al. [pour Martin à vérifier] and by Lambert [pour Pierre à documenter]. S. Casselman Baltic pulse · densities must have been very high in Gulf] ie '75	The reduction of recruitment to USLLO by > 2 orders of magnitude must be caused by a very powerful factor. If the recruitment collapse in USLLO is due to the area's distance from the spawning ground, then a reduction, lesser but still noticeable, should be evident in the rearing area which is closest to USLLO. Thus, during a period when USLLO recruitment is in collapse, recruitment to SLEG should be depressed. Recruitment destined to USLLO has been in collapse since the early 1980s. Recruitment to the Gulf (as indicated by Miramichi electrofishing) declined, but the decline came a decade before the collapse of recruitment destined for USLLO (Fig. 9.6). Density-dependent effects cannot explain the timing of this decline. If young eels make density-dependent decisions on where to colonize while they are traversing the Gulf, then increasing numbers of recruiting eels in the Gulf should lead to higher numbers entering USLLO. Recruitment to the Gulf, as indicated by the Miramichi data set, has risen 2-3 fold since the early 1980s, and is now about 2/3 the levels found in the 1940s and 1950s. This rise has not been followed by a recruitment recovery in USLLO. This suggests that the recruitment decline in USLLO is not due to range-end density dependence.

in ocean
in Gulf
in Lwr River - more evidence in streams

• "density" may be affected by more than one year class

Zebra mussel hypothesis Recruitment collapse in USLLO is due at least in part to the increased clarity and decreased organic matter content of Lake Ontario that was brought about by zebra and quagga mussels. Such changes to outflowing water causes USLLO to be less attractive to incoming eels.	Eels are attracted to freshwater rearing areas by dissolved and particulate organic matter, and outflows from various streams may differ greatly in their attractiveness (Miles 1968). The invasion of Lake Ontario by zebra and quagga mussels has greatly changed the quality of water in Lake Ontario, and likely reduced its attractiveness to potential migrant eels.	The rapid proliferation of zebra and quagga mussels in Lake Ontario occurred in the early 1990s (Mills 2005), near the end of the period of declining eel recruitment into Lake Ontario. Hence the putative cause (water changes due to mussels) and effect (eel recruitment decline) do not match in time. The zebra mussel has been spreading in Europe at least since 1826, but colonization of new areas is not associated with declines in eel abundance (Minchin). [Guy: pourrais-tu donner la citation?]
Beauharnois hypothesis Recruitment collapse in Lake Ontario is due at least in part to the reduction in ship passage through the navigation locks at Beauharnois Dam.	The first eel fishway was installed at the Beauharnois Dam in 1994. Prior to this, eels moving upstream had to use the locks that permit ships to bypass the dam (Verdon and Desrochers 2003). This access route would be available to eels only when locks were opened to allow ships to enter. It takes about a year for eels to move from the Beauharnois area to the Moses-Saunders Dam (Verdon and Desrochers 2003). Mean daily counts at Moses-Saunders are closely correlated to the amount of ship traffic at Beauharnois during the previous year (r=0.83, P<0.001, Verdon and Desrochers 2003) (Fig. 9.8). This suggests that upstream migration of eels to Lake Ontario is limited by availability of lock openings at Beauharnois.	Mean daily eel counts at Moses-Saunders have declined by more than 99% since the 1970s. Ship passages at Beauharnois have declined by roughly half during the same period (Fig. 9.8). The decline in traffic at Beauharnois can therefore account for only part of the decline in Moses-Saunders counts. Even with the decline in ship traffic at Beauharnois, several hundred ships pass through the locks during each summer eel migration season. In most cases a lock opening admits only one ship (R. Verdon, pers. comm.) This means that eels wishing to move upstream have several opportunities per day to do so. If lock openings at Beauharnois limit upstream passage of eels, then Moses-Saunders counts should begin to increase a year after construction of the first eel fishways in 1994. This is not the case; counts at Moses-Saunders showed no increase in 1995 and subsequently (Table 4.4). This suggests that Beauharnois lock openings do not contrain upstream migration of eels to Lake Ontario.

	For	Against
Hypoxia hypothesis The recruitment collapse in USLLO is due at least in part to the advent of hypoxic bottom water in the St. Lawrence Estuary. Young eels are unable to ascend the estuary because bottom hypoxia prevents them from using selective tidal transport.	Glass eels and elvers ascend an estuary by selective tidal transport (McCleave and Wippelhauser 1987). During flood tide eels rise into the water column to be carried upstream, and during ebb tide they fall to the bottom where they wait for the next tide. A lack of oxygen in bottom water could interfere with this method of travel. Gilbert et al. (2005) reported a long-term decrease in oxygen concentration in bottom waters of the St. Lawrence estuary. Oxygen concentration decreased most abruptly between 1976 and 1983, and showed no apparent change since that time (Fig. 9.9). In 2003, the isoline of an O_2 concentration 60 µmol l^{-1} intersected the bottom at a mean depth of 275 m. Water with less than 62.5 µmol l^{-1} (2 ppm) of O_2 is considered hypoxic. The deep part of the estuary, with depth >300 m, occupies a variable percent of the estuary width, and reaches a maximum at Pointe-des-Monts, where 62% of the estuary width is deeper than 300 m (Fig. 9.10). The steep decline in recruitment at Moses-Saunders affected elvers which arrived in the estuary between 1977 and 1980 (Fig. 9.2). This recruitment decline occurred during the same period as the abrupt drop in oxygen concentration (1976-1983, Fig. 9.9). If elvers attempting to ascend the St. Lawrence use the middle of the estuary, and if they use selective tidal transport, the lack of oxygen in bottom water could hamper or prevent upstream migration. The lack of oxygen does not affect shallow water close to shore. Elvers travelling close to shore can still colonize tributaries in the estuary and near the river mouth. This explains why the Petite rivière de la Trinité, near the mouth of the St. Lawrence, continues to have an adequate supply of recruits.	The drop in oxygen concentration affects only deep water in the lower estuary; concentrations are normal in shallower water. Given that the majority of the estuary has water less than 300 m deep (Fig. 9.10, the oxygen decrease cannot be a barrier to upstream migration. There is no information on how young eels ascend the St. Lawrence estuary. Selective tidal transport has been demonstrated for glass eels and elvers, but not for older eels. Eels arrive at the Sud-ouest River, a tributary of the St. Lawrence estuary, as young yellow eels. Eels may ascend the estuary at this stage. It is not known if young yellow eels use selective tidal transport. If eels ascend the estuary at the small yellow stage, it is likely they would use shallow water near the shoreline, which is close to typical habitat used by eels of this stage. If this is the case, they they would not encounter a lack of oxygen, regardless of where they travelled in the water column. Eel recruitment destined to the USLLO declined steeply in 1977-1980, then declined gradually in 1981-1989. Bottom oxygen concentrations were stable during the second phase of the recruitment decline (Fig. 9.9). Hence the hypoxia hypothesis cannot explain the continuation of the eel decline past 1980.

Dam hypothesis The recruitment collapse in USLLO is due at least in part to dams which partly or wholly block access to upstream waters, and to turbine mortality during downstream passage.	Verreault et al. (2004) reported that the main stem and tributaries of the St. Lawrence River and Lake Ontario were blocked by 8,411 dams over 2.5 m high. 151 of these were hydro dams. Verreault et al. (2004) estimated the production potential of habitat blocked by dams in the St. Lawrence River watershed as 836,545 silver eels, which is much larger than recent estimates of the silver eel run for the entire St. Lawrence system (397,000-488,000, Caron et al. 2003). Mortality of eels passing through the Moses-Saunders and Beauharnois hyrdoelectric facilities ranges from 16 to 26%, depending on turbine type (Verreault and Dumont 2003). Cumulative turbine mortality, combined with fishing mortality, may remove a high proportion of the population that emigrates from Lake Ontario. The combined effect of habitat blockage and turbine mortality is a major constraint on eel survival and production in USLLO, and has contributed to the recruitment collapse.	The current configuration of the St. Lawrence Seaway between Montreal and Lake Ontario was completed by 1965 (Verdon and Desrochers 2003). Most other dams in the USLLO watershed were in place by the 1960s. Recruitment to Lake Ontario was high for about 2 decades after most dams had been built. Dams could not have caused the decline of recruitment in USLLO because there was no spurt of construction that coincided with the decline.
Pollution and habitat hypothesis The recruitment collapse in USLLO is due at least in part to the chemical and physical degradation of rearing habitat.	The drainage area of USLLO is heavily developed for industry, for agriculture, and for urbanization. A wide variety of pollutants enter the system from industrial and agricultural sources and from municipal sewage outlets. Some pollutants have been linked to eel mortalities (Dutil et al. 1987, Castonguay et al. 1994a). Others may have sublethal effects which contribute indirectly to mortality (Couillard et al. 1997). Not all pollutants are likely to be known, especially endocrine disrupting compounds whose effects are often subtle and long-term. There is also extensive modification of aquatic habitat in USLLO due to dredging, sedimentation, shoreline construction, and other causes. Pollution and habitat degradation could have decreased eel survivorship in USLLO and constrained the production capacity of the system. These changes could have contributed to the recruitment collapse in USLLO.	If pollution and habitat degradation was a significant contributor to the collapse of USLLO eel recruitment, there must have been a major increase in the intensity of these factors at the time of the recruitment collapse. This would have been the late 1970s/early 1980s for the arrival phase at the mouth of the St. Lawrence, and the mid 1980s for the transit through Moses-Saunders. No sharp increase in pollution or habitat degradation is known for this period (Castonguay et al. 1994a). Instead, pollutants in the region showed a general declining trend (Hodson et al. 1994).

Fishery hypothesis The recruitment collapse in USLLO is due at least in part to over-fishing.	The exploitation rate of silver eels descending the lower St. Lawrence River was estimated at 19% in 1996 and 24% in 1997 (Caron et al. 2003). Fishing mortality in the estuary and in upstream waters, combined with mortality from hydroelectric turbines, may represent a high proportion of the population. Price per kg in constant dollars increased substantially in the second half of the 20th century, increasing the incentive to fish (Robitaille et al. 2003).	Reported landings in the St. Lawrence estuary silver eel fishery were relatively stable until about 1970, and have since declined (Fig. 8.3). If fishing pressure was not sustainable, then landings should have declined earlier. If fisheries caused the Moses-Saunders decline, there should have been an increase in landings prior to the decline. No such increase occurred. Instead, the decline in estuary landings, adjusted for recruitment lag, began before the decline of the Moses-Saunders index (Fig. 9.2).
Non-panmixis hypothesis Eels in USLLO are a separate spawning population from eels that rear elsewhere. The recruitment collapse is due to mortality factors and production constraints within USLLO (dams, habitat and pollution, fisheries) which have reduced spawning escapement, and therefore production of returning juveniles.	Studies of variation in American eel allozymes (Williams et al. 1973, Koehn and Williams 1978) and molecular genetics (Avise et al. 1986, Aoyama et al. 2001, Wirth and Bernatchez 2003) are based on samples primarily from the United States. The only samples from the Gulf of St. Lawrence are from Petite rivière de la Trinité in the northwestern Gulf and from Prince Edward Island (Wirth and Bernatchez 2003). The genetics of eels from USLLO have not been examined. There is thus no genetic evidence to support the notion that eels from USLLO belong to a common spawning population that includes all American eels. Eels in USLLO are very different in size and have shown very different patterns of recruitment change than eels elsewhere in North America. These differences can be more easily explained if USLLO eels are genetically distinct. If USLLO eels form a separate spawning population, their recruitment may vary independently of trends in other areas. If young eels headed for USLLO come from a different spawning population than those headed for SLEG, there is no longer a need to explain why the recruitment patterns of these two groups have differed so greatly. Eels in USLLO are subject to heavy mortality and production constraints due to barriers to upstream migration, turbine mortality, habitat alteration, pollution, and fisheries. They may also be affected by oceanic changes (NOA) and bottom hypoxia in the estuary. Such factors could reduce the chance of new recruits finding their way to suitable growth habitat in USLLO. They would also reduce spawning escapement of silver eels, which in turn would further reduce recruitment. The cumulative effect of these factors could have led to the observed recruitment collapse.	Eels in USLLO are >99% female. USLLO eels cannot be a distinct spawning population because the region does not produce enough males to sire the young. Panmixis in American eels is supported by genetic studies of eels sampled from Puerto Rico north to the northwestern Gulf of St. Lawrence and Newfoundland. European eels show weak genetic differentiation (Wirth and Bernatchez 2001). Assume for a moment that USLLO and non-USLLO American eels have the same degree of genetic differentiation as northern and southern European eels. Under this scenario, there would be some, although not complete, genetic interchange between USLLO and non-USLLO eels. This means that recruitment between USLLO and non-USLLO eels would still be linked. Hence under the non-panmixis hypothesis an explanation would still be required for the great difference in patterns of recruitment change between USLLO and SLEG.

Table 9.3
Role of hypothesized factors (see Table 9.2) in causing the decline of eels in Lake Ontario-Upper St. Lawrence (USLLO), and inhibiting their recovery.

Hypothesis	Could hypothesized factor have caused the abrupt decline in eel recruitment to USLLO?	Is the hypothesized factor acting to inhibit the recovery of USLLO eels?	Comments
NOA hypothesis	No	No	The NOA index is currently low, which is favourable for eel recruitment
Range-end density dependence (REDD) hypothesis	Maybe	Maybe	
Zebra mussel hypothesis	No	Maybe	
Beauharnois hypothesis	No	No	Beauharnois Dam now has 2 eel ladders, so passage through locks is not necessary
Hypoxia hypothesis	Maybe	Maybe	
Dam hypothesis	No	Probably	
Pollution and habitat hypothesis	No	Probably	
Fishery hypothesis	No	Probably	
Non-panmixis hypothesis	Maybe	Maybe	

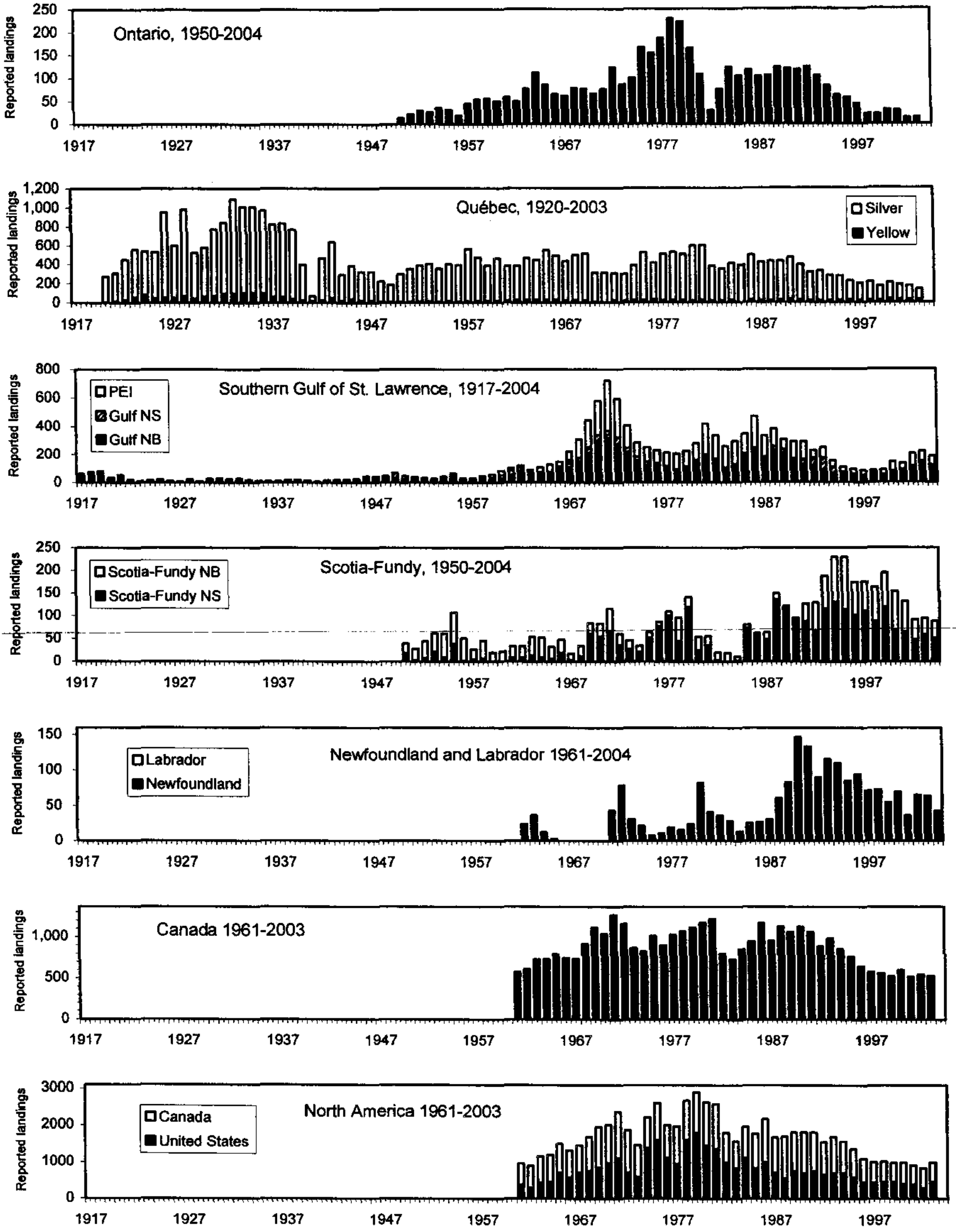


Fig. 9.1
Reported landings of American eels in North America, in metric tonnes. US landings from http://www.st.nmfs.gov/st1/commercial/landings/annual_landings.html. Some data for 2004 are preliminary.

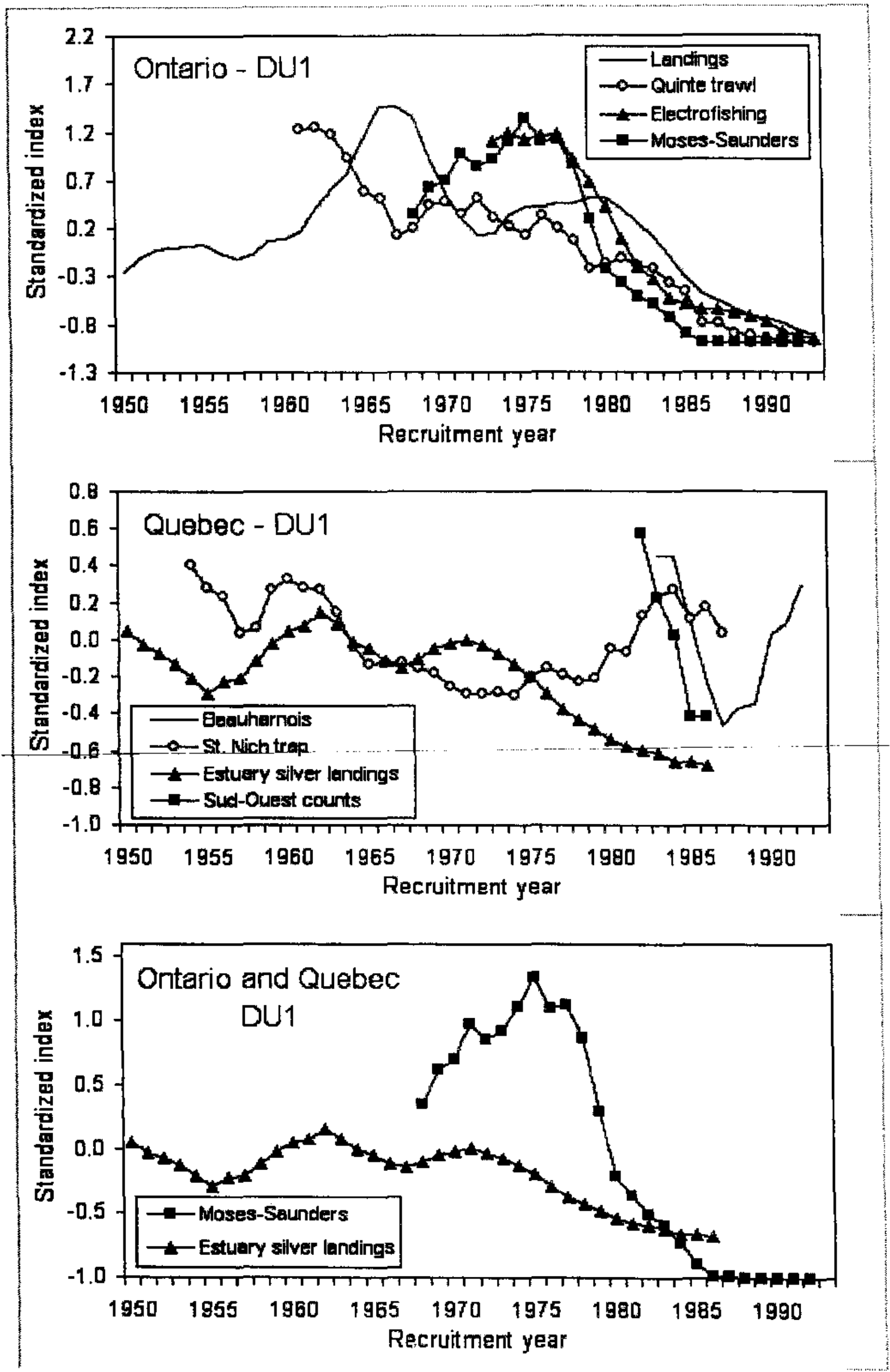


Fig. 9.2
Comparison of eel indices in Designated Unit 1, against recruitment year. Plots show 5 year running means as a proportion of the series mean, and standardized to have a mean of 0.

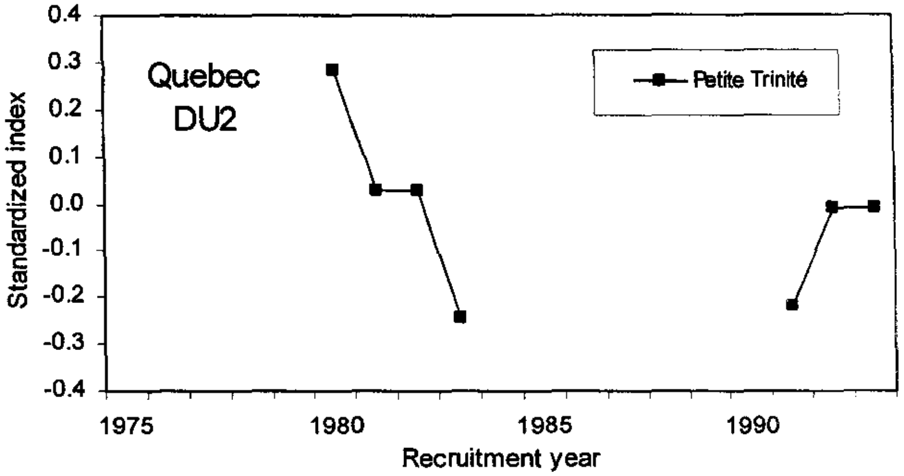


Fig. 9.3
Index of eel abundance at Petite Rivière de la Trinité in Designated Unit 2, against recruitment year. The plots shows a 5 year running mean as a proportion of the series mean, and standardized to have a mean of 0.

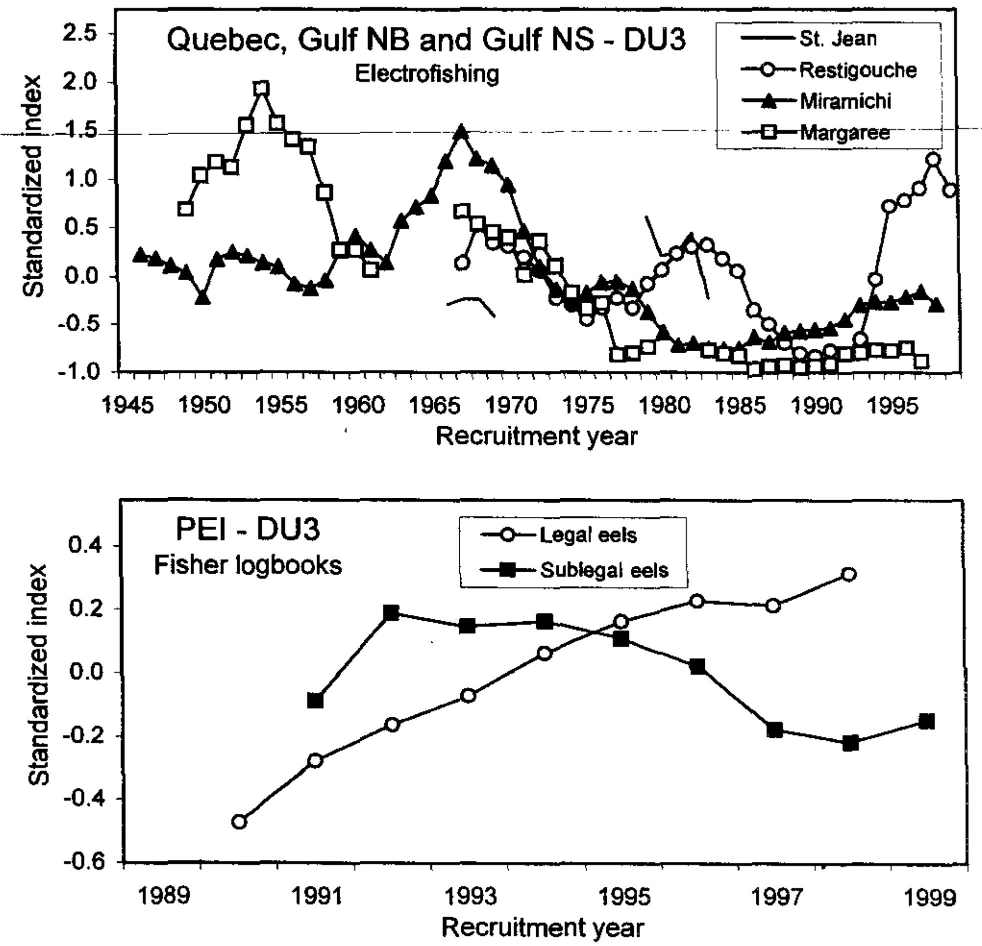


Fig. 9.4
Comparison of eel indices in Designated Unit 3., against recruitment year. Plots show 5 year runnings means as a proportion of the series mean, and standardized to have a mean of 0.

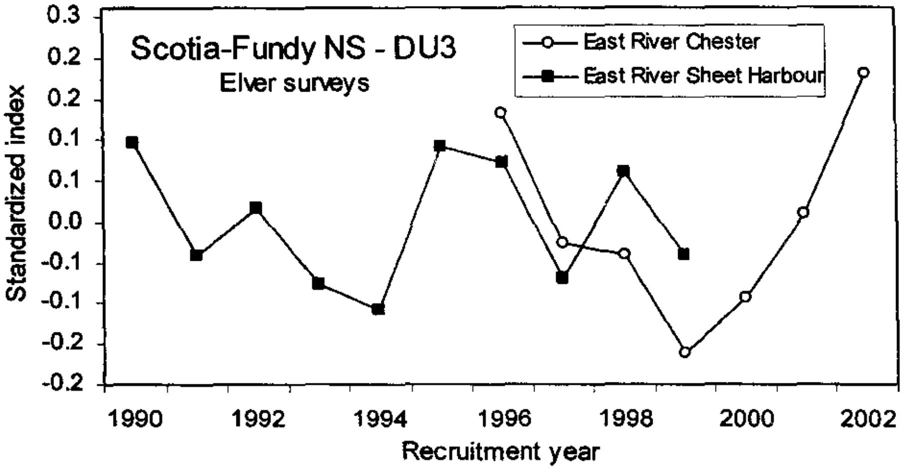


Fig. 9.5
Comparison of elver indices in Designated Unit 3. Plots show 5 year running means, standardized to have a mean of zero.

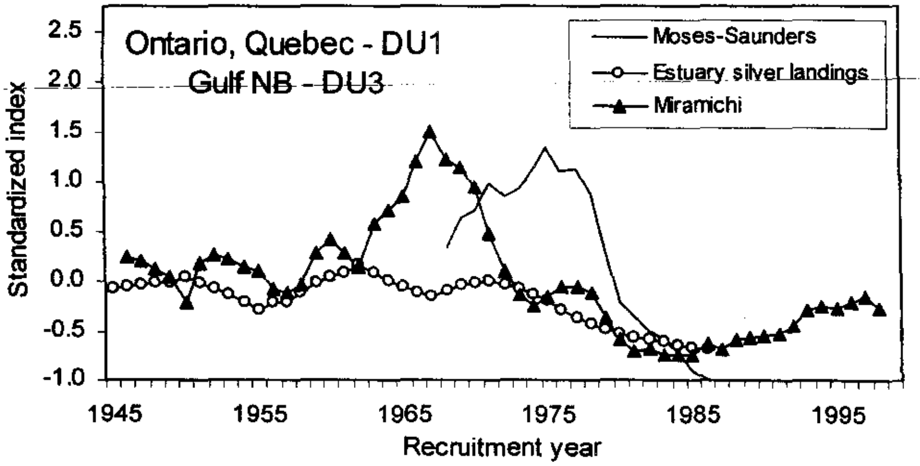


Fig. 9.6
Comparison of eel indices from Moses-Saunders dam, St. Lawrence estuary silver eel landings, and Miramichi River densities. Plots show 5 year runnings means as a proportion of the series mean, and standardized to have a mean of 0.

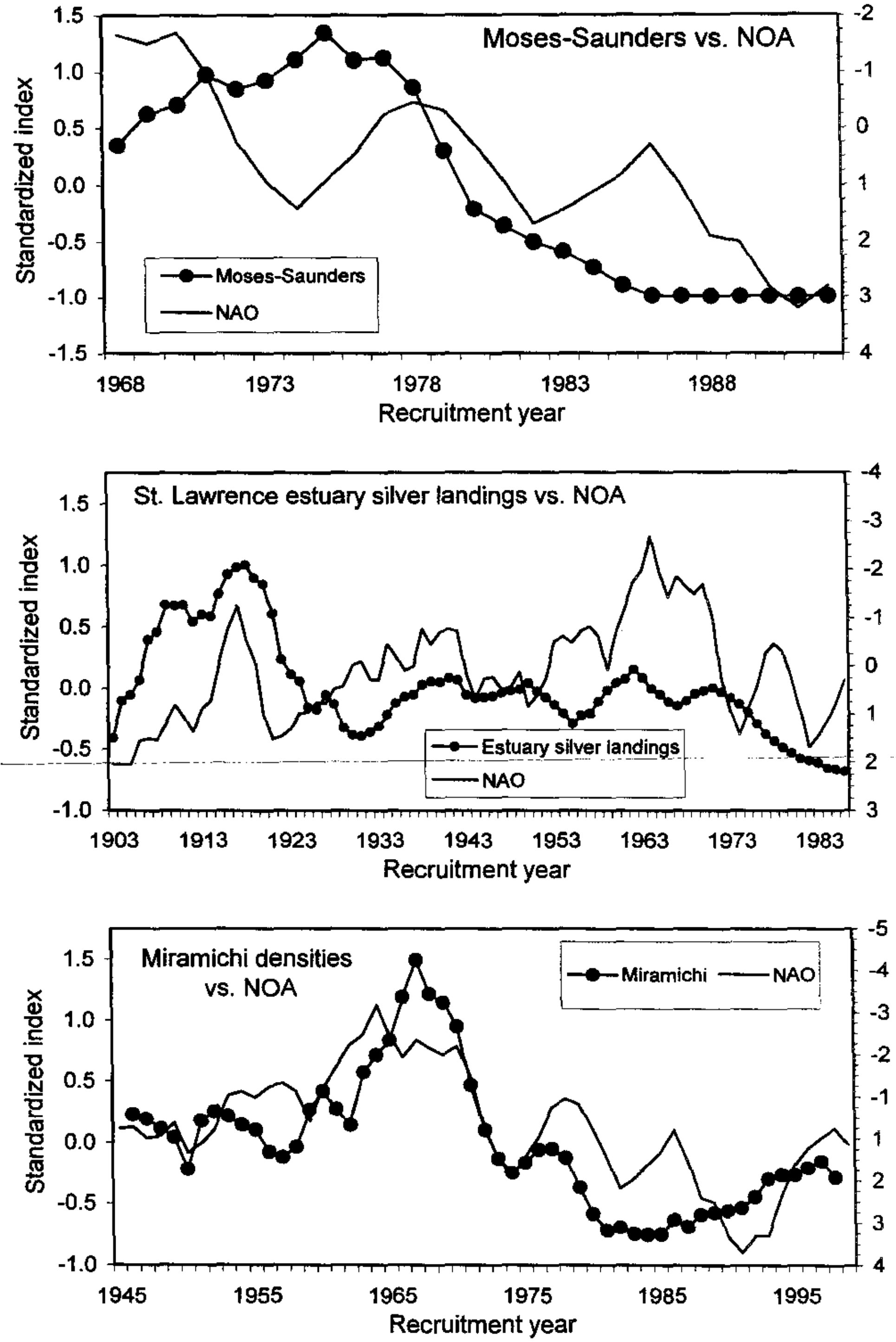


Fig. 9.7
Comparison of North Atlantic Oscillation index with the eel ladder index from Moses-Saunders dam, St. Lawrence estuary silver eel landings, and Miramichi electrofishing densities. Data are presented as 5 point running means. The eel series are standardized by dividing by the series mean and subtracting 1. The NAO is inverted (lowest values at the top).

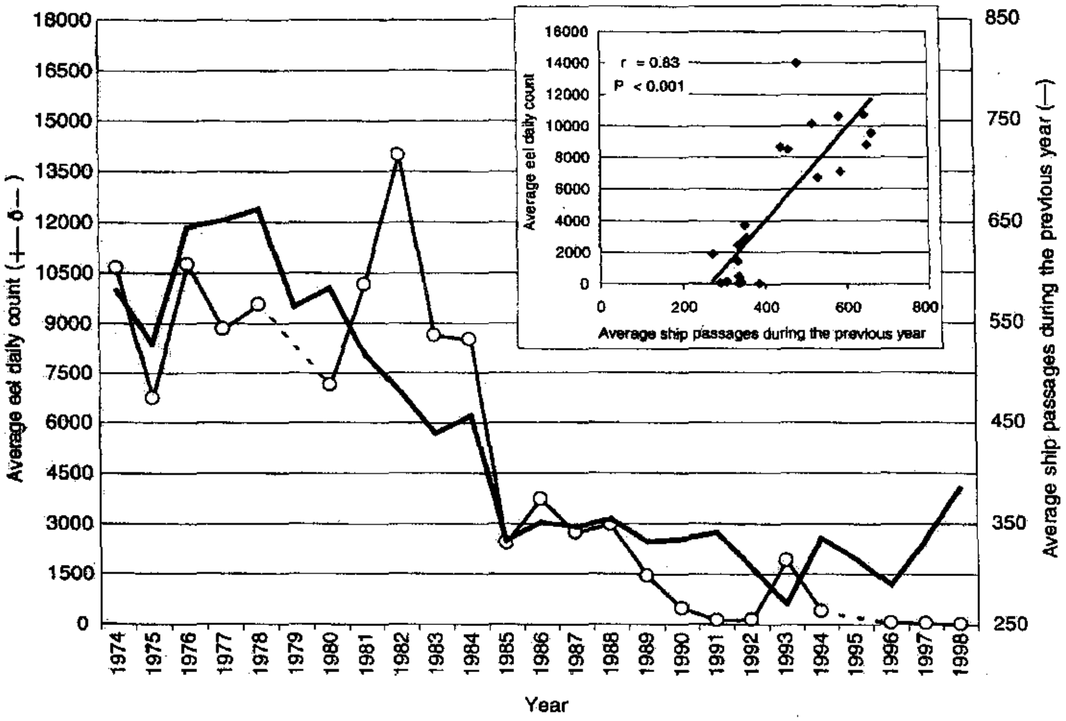


Fig. 9.8
Relation between average daily counts at the Moses-Saunders eel ladder (1975-1999, except 1980 and 1996) and average ship passages for June to August of the previous year in the St. Lawrence Seaway. From Verdon and Desrochers 2003.

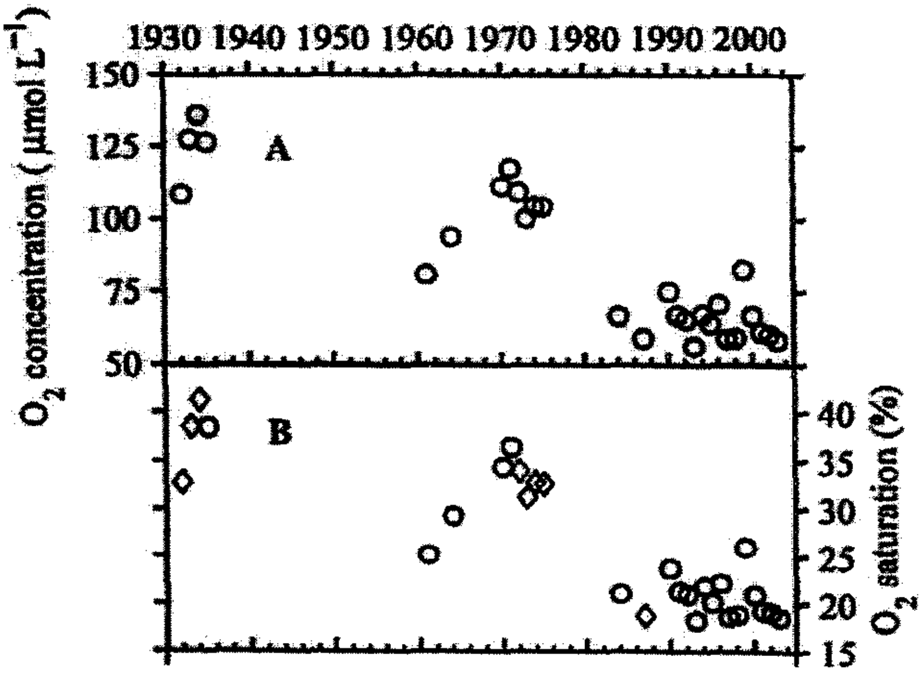


Fig. 9.9
Time series of oxygen concentration and oxygen saturation between 295 m and 355 m depth in the lower St. Lawrence Estuary. From Gilbert et al. 2005.

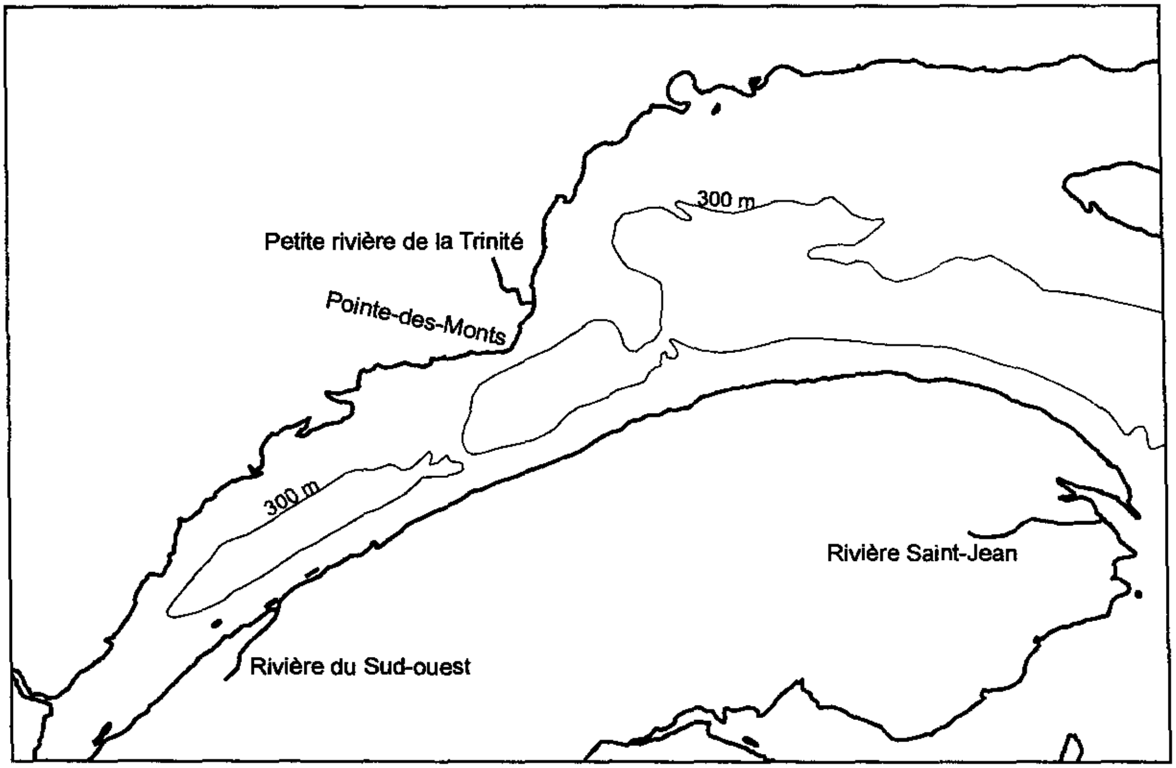


Fig. 9.10
The lower St. Lawrence estuary, showing the 300 m bottom contour.

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Migration of silver American eels past a hydroelectric dam and through a coastal zone

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Abstract Twenty five silver American eels, *Anguilla rostrata* (Lesueur), were sonically tagged to determine the passage success at a recently reconstructed hydroelectric facility on the Magaguadavic River, and to explore the environmental correlates of surviving eel movements through the coastal zone to the Bay of Fundy. Downstream movements of many eels were delayed at the dam and tagged fish moved extensively in the reservoir, presumably searching for an exit. All 19 eels that entered the turbines died. Six eels survived by passing the dam either via a fish bypass chute (4), by spilling over the dam (1), or through a fish ladder for upstream migrating fish (1). Coastal zone movements of 20 control eels released downstream of the dam, and of the six survivors, were linked to environmental parameters (tides, luminosity). The efficiency of the downstream fish bypass at this site might be improved by altering water management strategies.

KEYWORDS: *Anguilla rostrata*, coastal zone, freshwater impoundment, migration, silver eel, telemetry.

Introduction

The two species of catadromous eel found in the North Atlantic Ocean are the American eel, *Anguilla rostrata* (Lesueur), and the European eel, *Anguilla anguilla* L. Populations of both the species have drastically declined in recent years largely because of the human impacts (Ritter, Stanfield & Peterson 1997; COSEWIC 2006).

Safe passage to and from rivers, and protection of freshwater habitats is critical for the conservation of eels. Hydroelectric dams have been constructed in many rivers that historically had eels, necessitating research to determine the best ways to provide safe passage around turbines for eels and other migrating diadromous fish. As new dams are constructed, or old ones refitted, there is potential to fit state-of-the-art bypass facilities to benefit diadromous fishes.

The Magaguadavic River in New Brunswick, Canada, supports three diadromous fishes: American eel, Atlantic salmon, *Salmo salar* L., and alewife, *Alosa pseudoharengus* (Wilson). A hydroelectric facility originally constructed in 1902 at the head of the tide was rebuilt in 2004 and fitted with a new downstream fish passage facility. Maturing eels (referred to as silver eels) migrating downstream from this river were sonically tagged to: (1) document the migration timing of eels as they moved from fresh to sea water; (2) assess movement rates and survival past the new hydroelectric dam; and (3) compare and contrast the movements in the coastal zone of eels that had successfully passed the dam with a control group of fish that were sonically tagged and released downstream of the dam.

Materials and methods

The Magaguadavic River is located in southwest New Brunswick and flows freely for 82 km before entering a 16-km reservoir that ends at a 13.4-m head-of-tide dam. The dam marks the start of the estuary, which extends for 8 km to sea entry at Passamaquoddy Bay (for details, see Carr, Whoriskey & O'Reilly 2004). The powerhouse is located 350 m downstream from the dam around an S-shaped bend in the river. The 2004 rebuilding of the facility upgraded the power generation capacity from 3.7 to 15 MW. At that time, four Francis turbines were replaced with two Kaplan variable pitch propeller turbines. A newly designed downstream fish bypass facility was installed replacing the old facility. The bypass entry gate (0.76 m wide by 2 m high) is centred 1.65 m between the two penstock intakes (5 m wide each) for the turbines. The existing fish ladder for upstream passage was not altered during the rebuilding of the hydro facility, and was not designed to attract downstream migrating fish.

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The estuary empties into Passamaquoddy Bay (an offshoot of the Bay of Fundy). This bay is about 8.7 km wide (east-west transect) and 26.7 km long (north-south transect) and depths generally range from 20 to 30 m, but can go to 70 + m (for details, see Trites & Garrett 1983).

American eels were captured and counted in the downstream fish bypass facility at the hydroelectric dam from 2004 to 2006 to determine the timing of seaward migration. An attraction flow ranging between 0.85 and 1.13 $m^3 s^{-1}$ passed through the fish gate into the collection facility during the monitoring periods. Eels passing through the entry gate dropped about 2 m into a plunge pool (6 m wide by 9 m long by 2 m water depth). A fine mesh (1 cm) inclined screen closed the downstream end of the collection facility to prevent the escape of eels. Eel enumeration typically took < 30 min and was conducted by closing the bypass entry gate to reduce the water depth in the collection pool. After enumeration, the inclined screen was raised to release eels downstream, and then the screen was lowered and the bypass entry gate reopened. In 2004, the bypass facility was monitored daily from 12 April to 23 November, after which flood conditions prevented access to the structure. In 2005, the bypass facility was monitored daily from 20 April to 8 October, after which flood conditions again prevented access to the structure. In 2006, the bypass collection facility was monitored daily from 10 April to 13 November, except from 14 to 18 September.

Hydroelectric operating regime, and spill during migration periods

In 2004 and 2005, power generation occurred both day and night throughout the autumn months (September–November). In 2006, from 1 September to 22 October, power generation occurred only on Mondays to Fridays from about 7:00 to 21:00 h, with the exception of no generation on 19 October. Power generation occurred continuously from 8:00 h on 22 October until 21:15 h on 24 October, and from 29 October until the end of the study. No water spilled over the dam during the entire eel migration period in 2004. In 2005, high flows resulted in spills from 9 October until the end of the study. In 2006, short periods of spill occurred from 21:00 h on 21 October to 9:15 h on 22 October and from 18:00 to 20:00 h on 29 October. In 2006, the maximum height of water cresting the dam during spill times was 13 cm. Water stopped spilling shortly after power generation started on both dates. During no-spill periods, the only exit routes for eels past the dam besides the downstream bypass facility were the turbines or the upstream entrance of the fish ladder. No known lighting at night occurred at the dam in 2004 and 2005. However, in 2006, the area around the dam face was illuminated at night from 1 to 12 October.

In October 2006, 45 healthy silver eels (> 75cm total length) were collected from the bypass facility for sonic tagging. The fish were kept for up to 3 days prior to tag insertion. All other eels and fish species were released unharmed into the bypass channel downstream of the collection facility immediately after enumeration. Twenty five eels (reservoir releases) were tagged with V9-6L-R64K coded pingers (20 mm length × 9 mm diameter; 2 g weight in water, produced by Amirix/Vemco Limited, Halifax, Nova Scotia). An additional 20 eels ('control' estuary releases) were tagged using V9P-6L-R64K-100 m depth coded pingers (38 mm length × 9 mm diameter; 2.2 g weight in water). All pingers had a frequency of 69 kHz, 15–45 s off times, power output of 137 dB, and a minimum life of 25 days.

Eels were anaesthetised using clove oil (100–120 mg L^{-1}) and measured for total length (mm) and total weight (g) prior to surgery. The pingers were surgically implanted into the peritoneal cavities. During surgery, eels were placed ventral side up in a v-shaped trough with soft foam to stabilise the body of the eel. Furacin was used to disinfect the surface of the fish and a 20-mm incision was made on the mid-ventral line, 50 mm anterior to the anus. The pinger was inserted through the incision. One to two sutures (3-0 Ethilon black monofilament nylon with FSL reverse cutting needle) were applied to close the incision. Eels recovered from anaesthesia in < 10 min and were released 6–18 h after surgery. Median total lengths, weights, and tag-to-body-weight ratios of all 45 sonically tagged eels in 2006 were 89.2 cm (range 76.0–102.0 cm), 1.45 kg (range 0.86–2.35 kg) and 0.28% (range 0.13–0.44%), respectively. There were no significant differences in total lengths or weights among the release groups.

Releases occurred on different dates to provide a temporal replication. The above-mentioned dam releases occurred at night, shortly after sunset (18:30 h). Thirteen and 12 sonically tagged eels were released into the reservoir 1 km upstream from the head-of-tide dam on 3 and 17 October 2006, respectively. The 20 sonically tagged control eels were released in the estuary 500 m downstream of the dam on 11 October (10 eels at 21:45 h on a low tide) and 12 October (10 eels at 04:00 h on a high tide).

Movements of tagged fish were monitored by mooring submersible receivers (VEMCO VR2 and

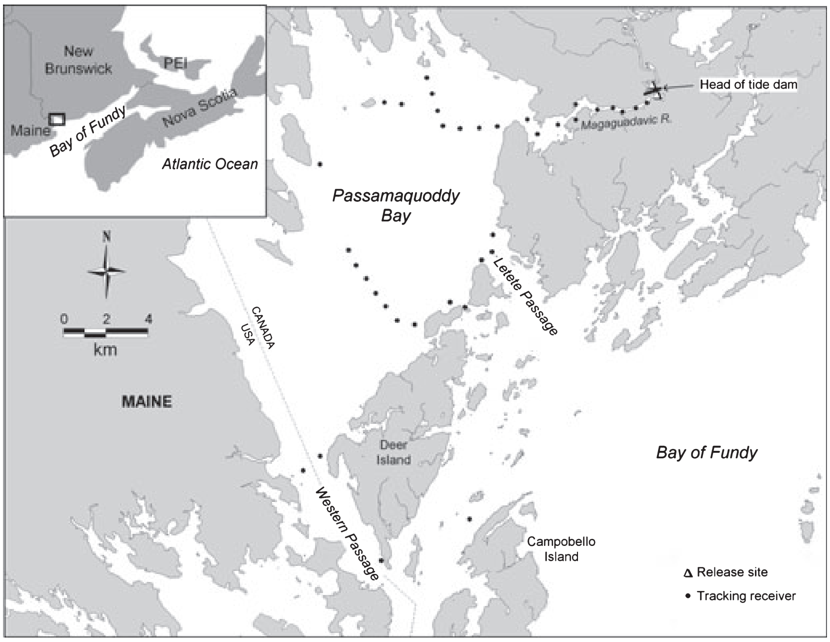

Figure 1. Study area showing release sites for sonically tagged eels. Tracking receiver locations are shown for Passamaquoddy Bay and in the Magaguadavic River estuary (except for receivers near the dam). Receivers are not shown for freshwater sites.

VR2W) at various fixed points above, at, and below the dam (16 in fresh water, 16 in the estuary, and 26 in the Passamaquoddy Bay; see Fig. 1). In fresh water, 10 receivers provided coverage from the dam face of the reservoir to 3 km upstream of the dam. Reception of these ranged from 5 m to 500 m, with the poorest reception at the dam face. Three additional receivers were placed in the reservoir from 5 to 8 km upstream of the dam. Reception range of these was ≥500 m. One receiver each was placed in the fish ladder, bypass facility, and in the underground tailrace about 20 m downstream from the turbines. Detection range was variable and generally low in the vicinity of the tailrace and fish ladder exits, so receivers were positioned 15–50 m apart to provide complete coverage. For all seawater sections, the receivers were organised spatially (≤ 1 km apart) as follows: a line along the axis of the Magaguadavic River estuary, a line across the northern portion of the Passamaquoddy Bay near the confluence of the estuary with the bay, a line that roughly bisected the Passamaquoddy Bay, and two or more hydrophones across the entrances to the various tidal passages (Fig. 1). The detection radius of the hydrophones in the estuary, bay and passages was typically ≥ 500 m. All receivers were deployed prior to tagged eel releases and remained in position until at least 6 November. Active searches for tagged fish were also performed using a boat equipped with a portable receiver (VEMCO VR60 and VR100) having either directional (VEMCO V10) or omni directional (VEMCO VH65) hydrophones.

Distances covered and daily movement rates of tagged eels were determined for the Magaguadavic River reservoir, the estuary, and the Passamaquoddy Bay. Straight line distances from the exit of the river's estuary to the Letete and the Western Passage exits of the Passamaquoddy Bay to the Bay of Fundy are about 8.7 and 20.7 km, respectively. An eel was defined as active if either sequential, regular detections of a tagged eel on different receivers were found, or the individual spent < 20 min within range of a single receiver. Localized movements covered short distances (generally much < 5 km), and were followed by a cessation of movements for > 5 h but generally for several days. Committed movements were unidirectional downstream displacements and were characteristic of tagged silver eels moving either to the dam face for the first time, or from the release site downstream of the dam through the estuary into the Passamaquoddy Bay. Approach (search) times at the dam face were calculated as the total amount of time the eels were in range of any of the receivers positioned on the dam face near the turbine intakes and bypass entrance. Sequential detections on more than one of these receivers over a short time were considered part of a single approach to the dam. An approach to the dam face was considered terminated when the signal from the tagged eel was detected either at a receiver ≥500 m upstream of the dam, or downstream of the dam face. After passing the dam, a tagged eel was considered dead if its signal was repeatedly detected in the same position in the underground tailrace or within 10 m of the tailrace exit over the rest of the study duration.

Water temperature (Vemco TR minilogs) was recorded in the Magaguadavic River from 2004 to 2006. Water discharge records were obtained from a gauging site 24 km upstream of the dam (Environment Canada, Station No. 01AQ002). Hydroelectric operational data (hours of turbine operation, power output, numbers of turbines operational, presence of spill) were obtained from the head-of-tide power dam. Other parameters recorded included moon phase, precipitation, tidal phase, sunrise and sunset times. Statistical comparisons were made by non-parametric tests (χ^2, Mann–Whitney *U*-test, Kruskal–Wallis ANOVA).

Results

More than 72% of eels were captured in the bypass facility from early September to early November in all years (Fig. 2). However, the total number of eels captured and times of peak migration varied among the 3 years. There was no evident relationship between water temperature and discharge rates with the number of eels captured; however, more eels were captured later in the season at cooler temperatures (Fig. 2). No relationship was evident between lunar phase and eel captures (Fig. 2), and a similar number of eels entered the bypass facility on days with or without rain events ($P > 0.05$). Most eels were captured at times of no water spillage at the dam. During the peak period of eel migration in 2006, a sudden decrease in the number of eels captured per day in the fish bypass facility from 1 to 12 October was observed (Fig. 2). This corresponded with a period when the area near the bypass entry gate and turbine intakes was illuminated.

Nineteen of the sonically tagged eels passed the dam via the turbines, four via the downstream bypass facility, one via the fish ladder, and one over the dam. Signals from all tagged eels that passed through the turbines were repeatedly detected in the immediate area downstream of the turbines during the study period, and they were judged to be dead. The six eels that used alternate routes survived bypassing dam passage.

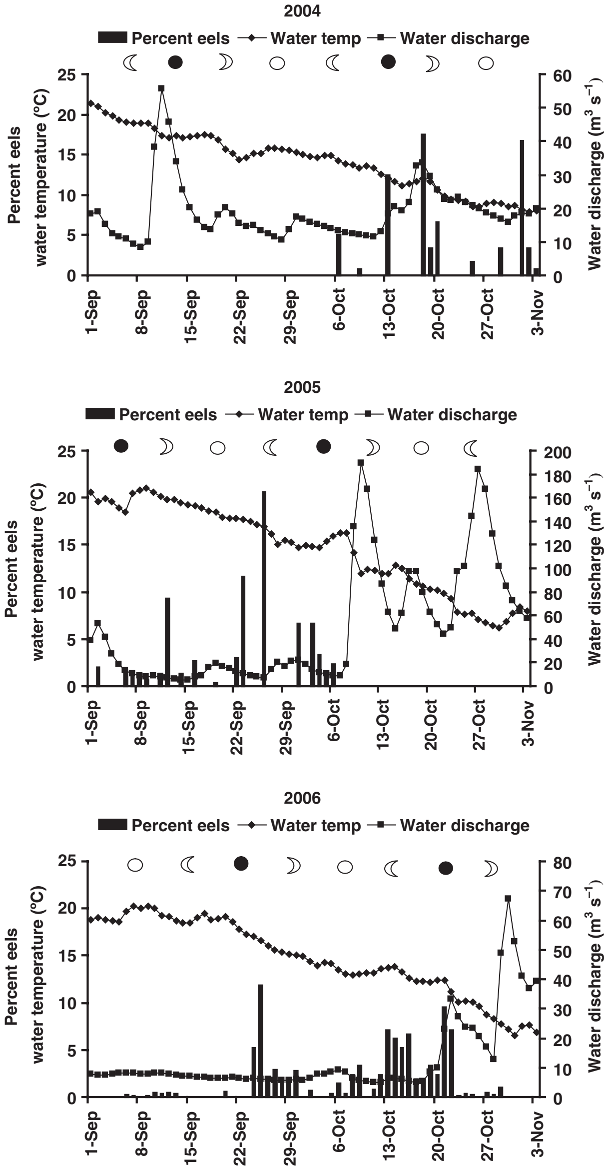

Figure 2. The percentage (fraction of annual total run) of eels captured within a given year, taken on a given date (histogram bars) in the head-of-tide downstream fish bypass facility in the Magaguadavic River from 2004 to 2006. The histograms cover the September to November period when most annual eel captures occurred (72% in 2004, 77% in 2005, and 96% in 2006). Water temperatures, water discharges, and moon phases are also presented.

After release in the reservoir (1 km above the dam), patterns of dispersal, total distances moved, number of dam approaches, and search times on the dam face for the fish released on different dates did not differ significantly between the two release dates, hence data were pooled for analysis. All 25 eels showed localised movements in at least one of five different reservoir zones: near the release site ($n = 5$), or upstream of the release site at 1 km ($n = 5$), 2 km ($n = 7$), 5 km ($n = 4$), and >5 km ($n = 4$). Eels took a median of 5 days (range 1–18 days) post-release before they made their first approach to the dam face. There was considerable variation among individuals in total distances moved, depending on how far upstream they initially distributed themselves and how many times they approached the dam before finding an exit route.

The time eels spent holding position prior to committing to move to the dam did differ significantly between the two release groups. Median times spent holding post-release was 9 days (range 5–18 days) for the 3 October release group and 3 days (range 1–4 days) for the 17 October release group ($P < 0.05$).

Sixteen eels (64%) passed the dam on their first approach, 14 during the night and one during the day. The remaining eel first approached the dam at night (3:00 h) where it remained, presumably searching for an exit until it passed through the turbines 4.2 days (6002 min) later (Table 1). Passage of this fish occurred at 7:10 h, just as daily power generation had begun. Nine other eels (36%) approached the dam face on multiple occasions before passage. Of these, five passed on their second approach (three during the day, two at night), two passed on the third approach (one each during day and night), and the final two eels passed during the day on their fourth approach.

Overall, 23 of 25 eels transited the dam either during or within 48 h of a rain event. Eighteen (72%) passed the dam at night, the majority (11 of 18 eels) within < 3 h after sunset. Twelve eels passing the dam at night moved on the last quarter moon phase, five during the new moon phase, and one during the full moon phase. All seven eels that passed the dam during daylight hours did so within 48 h of rain events. Of the seven eels that passed the dam during daylight hours, six had first approached the dam face on at least one previous occasion at night.

Table 1. Number of eels, the median amount of time spent searching at the dam face prior to final passage, and comparison of the time it took day versus night passing sonically tagged American eels to exit the reservoir using four different passage routes

Passage route	Number of eel	Median time on dam face (min.)	Range
Turbines			
Day	6	8.5	1–57
Night	12	1.0	1–91
*	1	6002	
Bypass			
Night	4	46.5	39–104
Fish ladder			
Night	1	1	
Over dam			
Night	1	1	
Final approach for all routes			
Day	6	8.5	1–57
Night	18	1.0	1–104
Total approaches for all routes			
Day	8	31.5	1–69
Night	31	21.0	1–216

* indicates an eel that spent more than four days by the dam face prior to passage via the turbines during the day.

No significant differences were observed in the time spent on the dam face between exploratory approaches and final passage approaches for all 25 eels ($P > 0.05$). The median search time spent on the dam face prior to passage for most tagged eels (24 out of 25) was 21 min (range 1–216 min). There was no significant difference in the median time spent on the dam face between day (31.5 min, $n = 8$) and night (21.0 min, $n = 31$) approaches (Table 1, Mann–Whitney *U*-test, $P > 0.05$). Active movements in the reservoir occurred mostly at night (79% night vs 21% day).

The six eels that avoided the turbines moved at night over a 23 h and 8-min period. Four passed during 12 h of spill: three via the downstream fish bypass and one over the dam. The median time spent searching on the dam face for eels that passed via the bypass was significantly longer than for eels that used alternate safe passage routes (Table 1; $P < 0.05$).

On 14 October, several dead eels were observed at low tide near the tailrace. The smallest dead eel measured 49 cm (total length). All eels sampled ($n = 8$) were severed in the posterior region (anus region). None of these could be confirmed as fish used in the telemetry work.

The six eels that survived by passing the dam were tracked to sea. These fish were of similar total lengths to the 20 sonically tagged fish in the control group released downstream of the dam, which were also tracked to the ocean. After release, the control eels either remained stationary near the release site ($n = 12$) or exhibited limited local movements of distances ≤ 5 km from the release site ($n = 8$). Overall, the 20 control fish spent 31.3 h (range 0.1–238.6 h) near the river bottom before committing to directed movements through the estuary into the Passamaquoddy Bay. The median time that the survivors of dam passage ($n = 6$) spent downstream of the dam before they initiated movements from the estuary to the Passamaquoddy Bay was 1.8 h (range 0.2–14.9 h). This was significantly less than the time the control group spent in the estuary before they committed to move towards the ocean (Mann–Whitney *U*-test, $P < 0.05$, Table 2).

All 26 tracked eels survived passage through the estuary to the sea. Once the fish committed to move from the start of the estuary to the Passamaquoddy Bay, it took a median of 5.6 h for the survivors to cover this 8 km distance, which was similar to the time of 3 h spent by the control group ($P > 0.05$, Table 2). Committed movements through the estuary began

Table 2. Time spent by silver American eels (1) in the basin near the downstream side of a hydro dam before beginning active seaward migration, (2) actively moving through the river's estuary to Passamaquoddy Bay, and (3) in Passamaquoddy Bay before exiting towards open ocean

	Control ($n = 20$)		Survivors ($n = 6$)	
Location	Median hours	Range	Median hours	Range
Basin	31.3*	0.1–238.6	1.8*	0.2–14.9
Estuary to Pass. Bay	3.0	1.9–6.1	5.6	2.3–7.8
Passamaquoddy Bay	51.0	2.5–668.1	57.7	2.4–167.8

Control fish were those released downstream of the dam after surgical implantation of tags. Survivors were sonically tagged fish released upstream of the dam, and which managed to pass the dam alive by using a route other than the turbines. * indicates $P < 0.05$.

during or within 24 h of a rain event. All detected eels moved through the estuary towards the bay at night, with 11 moving on an ebb tide, five on a flood tide, eight during a flood-ebb cycle, and two on an ebb-flood cycle.

The dam survivors and control eels spent similar amounts of time in the Passamaquoddy Bay before exiting to the Bay of Fundy. Median residence time was 57.7 h for the survivors, which was not significantly different from the 51.0 h spent by the control group ($P > 0.05$, Table 2). Most (99.5%) active movements in the Passamaquoddy Bay occurred at night. Twenty four eels exited the bay at night, 18 on an ebb tide and six on a flood tide. Sixteen eels (61.5%) left Passamaquoddy Bay to enter the Bay of Fundy via the Western Passage, whereas eight (30.8%) used the much smaller Letete Passage (Fig. 1). Two eels (7.7%) were detected last at night (29 October) near the mouth of the Magaguadavic River.

Discussion

Silver eels migrated from the Magaguadavic River to the sea within a 3 month autumn period. Peak movements measured by counting eel in the hydroelectric bypass facility occurred over a 4-week period, although the timing of this period and the pattern of runs varied among years. These results are consistent with other studies on this species (Haro, Castro-Santos & Boubée 2000; Verreault, Pettigrew, Tardif & Pouliot 2003).

Environmental variables may play a role as a stimulus for eel migration. In this study, water temperatures during migration varied substantially among the years. Other studies found that a sudden decrease in water temperature coincided with increases in downstream silver eel migration (Lowe 1952; Durif, Elie, Gosset, Rives & Travade 2003). No obvious trends between water discharge (river's flow) and eel capture rates were observed in this study, although the presence of the hydroelectric facility and its impacts on eel movements may have obscured any correlations. No relationship was found between lunar cycle and eel captures, in common with the observation by Smith & Saunders (1955) and Jellyman (1991). Lowe (1952) reported that the moon phase appeared to affect eels through the inhibiting action of light, rather than darkness or any periodic effect that may stimulate silver eel migration.

The hydroelectric facility at the study site was recently rebuilt, and a new fish passage facility was provided that was intended to bypass most fish safely around the turbines. This new facility did not perform as anticipated. Sonically tagged eels were frequently delayed during the migration at the dam face. Eels that initially approached the dam and could not find an exit would withdraw to return on a second, third or even fourth occasion before they found a way out of the reservoir. Hydroelectric generation patterns were intermittent, and spill rarely occurred over the dam during the eel migration. This may have contributed to delaying migration in some individuals, a pattern that has been observed at other dams (Durif *et al.* 2003; Watene, Boubée & Haro 2003).

The attraction flow in the bypass chute did not attract many of the eels. Even when no power was being generated, the attraction flow was not very effective in bringing eels into the bypass facility. The longest median delay times recorded were for eels that ultimately used the downstream bypass during times of no power generation. Criteria used by the Fish and Wildlife Service in the Northeast Region of the United States requires a flow into a bypass facility of up to 4% of turbine capacity (Ben Rizzo, US Fish & Wildlife Service, personal communication), three times greater than that provided at the new bypass in this study.

In addition to delaying migration and increasing migration distances, the hydro facility caused eel mortalities. Seventy-six percent of sonically tagged eels passed through the turbines, and all died. Large eels, presumably females, were tracked and these were particularly susceptible to turbine strikes. However, dead eels as small as 49 cm were collected in the tailrace downstream of the dam. Six tagged eels found their way downstream by routes other than through the turbines, all within a 24 h period of rain, which included 12 h when water spilled over the dam. All six fish survived to enter the sea, indicating alternate, safe passage routes were available if the eels could be attracted to them.

Thus, despite the accumulated knowledge of ways to provide safe passage around dams, the new facilities at the current site have failed to meet the needs of the fish. Assuming that rebuilding the dam is too costly, provision of higher attraction flows into the downstream fish bypass might improve the efficiency of this route but would be difficult to implement because of engineering constraints. Alternatively, a combination of highest possible attraction flows into the bypass facility at night during the eel migration, with no power generation or scaling hydropower generation intake such that the bypass flow is near 4% of the intake at all migration times, could pass more eels. It may also be possible to ensure spills occur during key migration periods. Exterior lights were on near the bypass entrance at night during part of the study and this might have delayed migration. Lowe (1952) reported that downstream eel migration may be delayed by artificial lights. Finally, it is recommended that designs for fish bypass facilities for new or refit hydroelectric dams be submitted to an extensive peer review by independent experts in the field prior to the beginning of construction, so that inadvertent design flaws harmful to fish can be corrected.

The use of sonic telemetry allowed the documentation of individual eel movement patterns. Most tagged eels made initial localised movements immediately after release prior to committing to directed downstream movements. Some remained still for periods ranging from hours to days. Durif *et al.* (2003) reported similar findings for freshwater releases of radio-tagged eels and found that no directed downstream movements occurred from 1 to 28 days after release. In this study, migration delays may have been in part a response to handling and surgeries, but also in part to dry periods (no precipitation). Most eels moved during or within 48 h of a rain event.

Magaguadavic eels preferred to make nocturnal freshwater movements and movements continued to be nocturnal as the fish entered the sea. In addition, 96% of the tagged eels first approached the dam face at night during or shortly after rain events. Durif *et al.* (2003) and Watene *et al.* (2003) reported similar findings with their eels making their first approach to a dam at night. Daytime passage of their eels occurred on overcast days. Upon commitment to move, all tagged eels migrated through the narrow river estuary quickly (median of 3 h to cover 8 km), during or a short time after rain events, at night, and with little or no relation to a tidal cycle. Silver eels generally took longer time to travel through the larger Passamaquoddy Bay, but passage times varied from 2.4 to 668 h. The shortest exit times for eels from Passamaquoddy Bay were for the individuals that used Letete Passage, which was much closer (about 8.7 km straight line distance) to the estuary of the river (Fig. 1). Most eels took the longer route (about 20.7 km straight line distance), which followed the dominant current system in the bay (Trites & Garrett 1983) and exited via the larger Western Passage. Atlantic salmon smolts leaving the Magaguadavic River also generally followed this pathway (Lacroix, McCurdy & Knox 2004), and it appears to be a critical migration corridor for diadromous species.

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Quantifying migratory delay: A new application of survival analysis methods

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Quantifying migratory delay: a new application of survival analysis methods

Theodore Castro-Santos and Alex Haro

Abstract: Statistical techniques commonly used in fish passage research fail to adequately quantify delays incurred at obstacles, or the effects of modifications to those obstacles on passage rates. Analyses of telemetry data describing these effects can be misleading, particularly when passage route of some individuals is not established (e.g., because of mortality, tag failure, passage through unmonitored or alternate routes, etc.). Here, we demonstrate how event-time analysis, better known as survival analysis, can be used to quantify passage rates for any study that allows tracking of individuals through time, even when some individuals fail to pass the route or obstacle in question. We review two of the primary methods of event-time analysis (parametric and Cox's proportional hazards regression analyses) and use them in combination with logistic regression to provide unbiased estimates of delay incurred at a hydroelectric facility, as well as insights on factors affecting both rates of passage and route selection. Passage rate increased with increased depth of a surface bypass sluice gate and, among fish that passed through the turbines, with turbine flow. The data further indicate that risk of turbine passage increased with both delay and turbine flow.

Résumé : Les techniques statistiques couramment utilisées pour étudier le passage des poissons ne réussissent pas à quantifier adéquatement les délais face aux obstacles, ni à évaluer les effets des modifications de ces obstacles sur les taux de passage. Les analyses de données de télémétrie qui décrivent ces effets peuvent être faussées, particulièrement lorsque la voie de passage de certains individus ne peut être déterminée (e.g., à cause de la mortalité, de la perte des étiquettes, du passage par des routes non surveillées ou des routes de rechange, etc.). Nous démontrons comment l'analyse temporelle des événements (« event-time analysis »), mieux connue sous le nom d'analyse de survie, peut servir à quantifier les taux de passage dans toute étude qui permet de suivre des individus dans le temps, même lorsque certains ne suivent pas la route ou ne traversent pas les obstacles en question. Nous examinons deux des principales méthodes de l'analyse temporelle des événements (l'analyse de régression paramétrique et l'analyse de régression aléatoire proportionnelle de Cox) et les utilisons en combinaison avec la régression logistique pour obtenir des estimations non biaisées des délais encourus à un ouvrage hydroélectrique, de même que des informations sur les facteurs qui affectent à la fois les taux de passage et le choix de route. Les taux de passage augmentent en relation avec la profondeur d'une vanne de dérivation de surface et, chez les poissons qui passent par les turbines, en relation avec le débit de la turbine. Nos données montrent, de plus, que le risque associé au passage dans la turbine augmente tant avec le délai devant l'obstacle qu'avec le débit de la turbine.

[Traduit par la Rédaction]

Introduction

A growing body of research shows that delays to the migrations of anadromous fishes can cause dramatic reductions in adult recruitment and spawning success. For many species, the ability of juveniles to osmoregulate in both salt and fresh water (a prerequisite for successful transition to the marine environment) can only be maintained for a brief period (McCormick et al. 1998; Whalen et al. 1999). Similar time-dependent effects have been shown for thermal tolerance (Zydlewski and McCormick 1997*a*) and predation risk (Hargreaves 1994; Venditti et al. 2000). Failure of migrants to reach the marine environment within the resulting "smolt window" reduces likelihood of survival. Adult migrants are also vulnerable to delay: freshwater spawning migrations are often powered exclusively by energy stores acquired at sea, and growing evidence suggests there is a trade-off between depletion of these stores and reproductive success (Glebe and Leggett 1981; Leonard and McCormick 1999; Hinch and Bratty 2000).

Concern over the delays incurred at dams and similar barriers has caused the U.S. National Marine Fisheries Service (NMFS) to call for operational changes at hydroelectric facilities to minimize this effect (NMFS 2000). How the effects of these changes should be quantified, however, remains unclear. Migration rate is affected by numerous environmental variables including flow, temperature, photoperiod, and previous experience of the migrants (Zabel and Anderson 1997; McCormick et al. 1998). Although data from various forms of telemetry can provide detailed information on migratory behavior near obstacles, current analytical methods fail to make full use of incomplete data, i.e., data from indi-

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viduals that experience any form of tag failure, suffer predation, pass via undetermined routes, routes other than the route of interest, or fail to pass during the course of the study. Typically, such "censored" individuals are removed from the study and only those that pass are compared; alternatively, some analyses might require an assumption that the fish passed through one route or the other. Either approach introduces bias into the analysis and casts suspicion on the results (e.g., Nettles and Gloss 1987; Wilson et al. 1991; Johnson et al. 2000).

A further deficiency in standard analytical techniques is that they often fail to adequately account for covariates that change over time. Many variables that could potentially influence passage rate may not be constant during the prepassage period, and depending on delay duration, fish may be exposed to multiple levels of these covariates.

In this paper, we demonstrate how the problems of incomplete data, alternate passage routes, and time-varying covariates can be overcome using statistical methods best known for their applications in biomedical research. Collectively known as "survival analysis" (Cox and Oakes 1984; Lee 1992; Hosmer and Lemeshow 1999), these methods were developed to describe the timing of events, incorporating data from individuals that are removed from studies or whose fate was not determined before ending a study. Although the name implies application to survival studies, we use it here to quantify passage rate with no direct inference on survival. Thus, to avoid confusion with actual survival studies, we will use the synonym "event-time analysis".

Although these methods have broad application in ecological studies (e.g., Chambers and Leggett 1989) and have been used to estimate survival during the course of migration (Skalski et al. 1993; Lowther and Skalski 1997), they have yet to be applied to behavioral components of fish passage. Here, we present a novel application of event-time analysis and demonstrate its usefulness by quantifying effects of modifications at a hydroelectric facility on rates of passage. Results of a radiotelemetry study of migrating Atlantic salmon (*Salmo salar*) smolts were selected to demonstrate this application.

Methods

Rationale and techniques of event-time analysis

Because the theory and application of event-time analysis are unfamiliar to most fish passage researchers, we present a brief overview of event-time analysis techniques in the following subsections. Interested readers should consult Lee (1992), Allison (1995), and Hosmer and Lemeshow (1999) for more complete details. In each technique, a binary variable, δ, is used to denote whether the individual's passage time was observed ($\delta = 1$) or not (censored observations; $\delta = 0$). This allows calculation of probability functions without attributing passage routes or times to censored individuals. The only limitation to the use of censored data is that censoring must not be informative, i.e., covariate effects should be the same for censored and uncensored observations.

In addition to censoring, the feature that best distinguishes event-time analysis from other parametric and nonparametric methods is its use of the hazard and survivorship functions ($h(t)$ and $S(t)$, respectively). The hazard function is the instantaneous rate of passage for those individuals that have not yet passed, i.e.,

$$(1)\qquad h(t) = \lim_{\Delta t \to 0} [P\{\text{an individual remaining at time } t \text{ passing in the interval } (t, t + \Delta t)\}/\Delta t]$$

The survivorship function is the complement of the cumulative distribution function and indicates the proportion of individuals remaining at time t.

$$(2)\qquad S(t) = P\{\text{an individual passing after time } t\}$$

These relate to the more familiar probability density function (PDF, or $f(t)$)

$$(3)\qquad f(t) = \lim_{\Delta t \to 0} [P\{\text{an individual passing in the interval } (t, t + \Delta t)\}/\Delta t]$$

in that $f(t) = h(t) \times S(t)$ (Lee 1992).

The utility of the hazard and survivorship functions becomes apparent when we consider the effects of censoring. Censored data preclude the usual approach of estimating mean and variance of passage times. The hazard function can be easily estimated, however, by dividing numbers of fish passing in an interval by the number of fish available to pass. Likewise, because censoring indicates that the fish has not yet passed, the last extant observation still contributes to the calculation of the survivorship function (see below).

The most straightforward approach to describing passage times and their associated probability functions is to construct a life table. This is done by breaking time down into meaningful, but not necessarily equal, intervals and calculating estimates of the above probability functions. Lee (1992) provides a clear and detailed description of this procedure. An alternative approach, developed by Kaplan and Meier (1958), is helpful when plotting data. Here, instead of being fixed, intervals are defined by the actual occurrence of events. At each time that a passage event occurs, the value of the survivorship function is estimated based on the cumulative product of the conditional proportion passing:

$$(4)\qquad \hat{S}'(t) = \prod_{i: t_i \le t} \left[1 - \left(\frac{p_i}{n_i}\right)\right]$$

where p_i of n_i available individuals pass at each time t_i. Note that the t_i's refer only to uncensored observations, but both censored and complete observations are included in the denominator (n_i, or the risk set) as long as they remain available to pass. Life-table methods are best for constructing tables, Kaplan–Meier curves are best for plotting data. Nonparametric tests and predictions can be generated based on either life-table or Kaplan–Meier methods with similar results. However, when multiple covariate effects are present, these tests may be inappropriate, so we avoid them here.

Parametric models for event-time data with censoring

The influence of covariates on passage rate, as well as the shape of probability functions, can often be described by fitting models to the data and testing for fit. Covariate effects can be readily expressed as a linear model. An intuitive form is the accelerated failure time (AFT) model (Allison 1995; Hosmer and Lemeshow 1999):

$$(5) \qquad \ln(T) = \beta_0 + \beta_1 x_1 + \ldots + \beta_k x_k + \varepsilon$$

where passage time T is a random variable conditional on covariates $x_{1,\ldots,k}$. The disturbance component ε determines the shape of the error distribution and thus the spread of the quantile estimates of passage time. Note that the "failure" term in AFT carries over from the survival analysis literature but refers here to passage events. Using this approach, covariate effects are multiplicative: a given quantile of passage time T changes by a factor of e^{β_j}, or increases by $100(e^{\beta_j} - 1)$ percent per unit increase in the covariate.

Most texts and computer programs fit these models using maximum likelihood (ML) estimation (Lee 1992). The likelihood, L, of a model is described by

$$(6) \qquad L = \prod_{i=1}^{n} [f_i(t_i)]^{\delta_i} [S_i(t_i)]^{1-\delta_i}$$

where $f_i(t_i)$ and $S_i(t_i)$ are, respectively, the estimated PDF and survivorship functions of the fitted distribution for each individual, contingent on model covariates, and $\delta_i = 0$ for censored and 1 for uncensored observations. The likelihood function is maximized with respect to model covariates and parameters, usually by applying some version of the Newton–Raphson algorithm (Lee 1992).

The presence of censored data complicates the evaluation of model fit. Although Allison (1995) and Hosmer and Lemeshow (1999) provide a method for calculating a generalized R^2 statistic, both texts caution against its use, because its value is affected by the proportion of censored observations in the data set; indeed, no statistic can be calculated that quantifies the proportion of variance in a data set that is accounted for by a given model, because the variance of censored observations is not known. Therefore, alternative methods must be used for evaluating model fit. The log-likelihoods of ML-generated models can be used to test for differences in fit between nested distributions and models: a significant likelihood ratio indicates superior fit of the model with the greater likelihood. Akaike's information criterion (AIC) can also be used to identify the best distribution or model and has the advantage of not requiring nesting (see Allison (1995) for a clear discussion of nested distributions and Burnham and Anderson (1998) for the theory and application of AIC).

A more general method for numerically testing whether the data follow a particular distribution was proposed by Hollander and Proschan (1979), summarized in Lee (1992): a statistic following the standard normal distribution is calculated by comparing predicted and observed values of the survivorship function. In a related approach, Cox–Snell residuals (defined as $e_i = -\ln(S(t_i|\boldsymbol{x}_i))$, where $\boldsymbol{x}_i$ is the vector of covariate values for individual i and $S(t_i)$ is the estimated probability of that individual remaining until time t, based on the fitted model) are plotted against $-\ln(S'(t_i))$, where $S'(t_i)$ is the Kaplan–Meier estimate of the survivorship function. If the model adequately describes the data, then this plot yields a straight line with a slope of unity (Allison 1995).

Cox's proportional hazards regression

More often than not, knowledge of underlying distributions is limited, or the complexity of the shape of those distributions precludes predictive modeling. Furthermore, the presence of covariates that change over time can complicate modeling efforts and may not even be possible using many standard software packages. The effect of various treatments on the hazard function can still be estimated, however, using a semiparametric, proportional hazards regression approach first described by Cox (1972). This model is based on the premise that the log of the hazard is a linear function of k covariates; the relationship between the hazard functions of two treatment groups i and j is described by

$$(7) \qquad \ln(h_i(t) - h_j(t)) = \beta_1(x_{i1} - x_{j1}) + \beta_2(x_{i2} - x_{j2}) + \ldots + \beta_k(x_{ik} - x_{jk})$$

The effects of covariates on the ratio of the two hazards are estimated by the coefficients $\beta_1,\ldots, \beta_k$. This approach is similar to that described above for the parametric regression model, with the important distinction that where the parametric regression uses ML to model the effects of covariates on the actual time of the event, Cox's proportional hazards regression uses partial likelihood (L_P) to describe their effects on the rate at which the event occurs. A general expression for the L_P of a proportional hazards model with fixed (i.e., not time-varying) covariates is

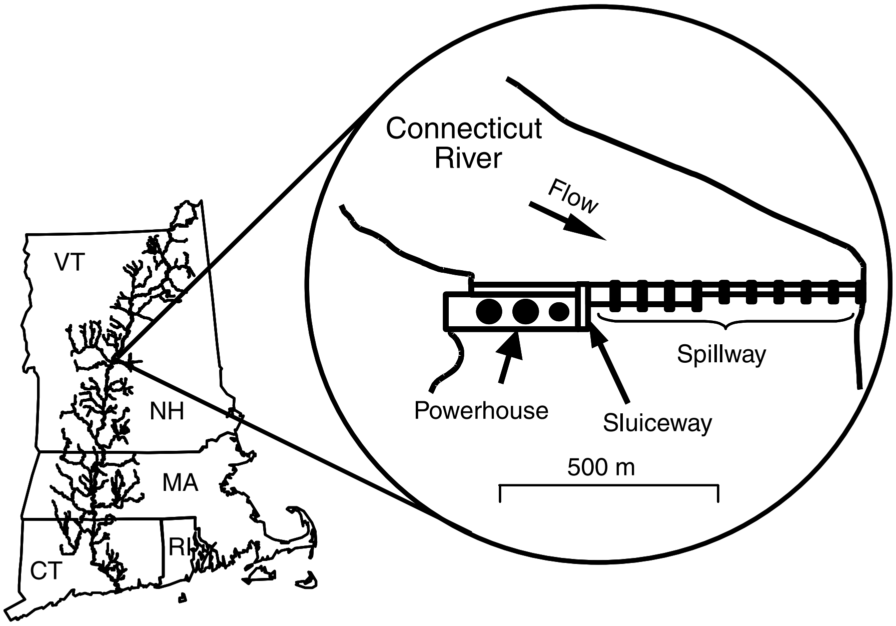
$$(8) \qquad L_P = \prod_{i=1}^{n} \left[\frac{e^{\boldsymbol{x}_i \boldsymbol{\beta}}}{\sum_{j \in R(t_i)} e^{\boldsymbol{x}_j \boldsymbol{\beta}}} \right]^{\delta_i}$$

where $R(t_i)$ constitutes the risk set, all individuals available to pass at time t_i (equivalent to n_i in eq. 4) and $\boldsymbol{x\beta}$ refers to the vector product of the covariates and their coefficients, either for the individual passing (i) or for each member of the risk set (j). As with ML estimation, the L_P is then maximized with respect to $\boldsymbol{\beta}$ using the Newton–Raphson algorithm. Note that by constructing the denominator in this way and including the censoring indicator δ_i, censored data are included in the analysis and contribute to the denominator until the last extant observation.

Because the L_P is based on the rank of time, rather than its actual value, combined, relative covariate effects on the hazard function can be tested without requiring the underlying probability to follow a particular distribution. In addition to significance, software packages may generate estimates of coefficients in eq. 7; their interpretation is simplified by using an alternate quantity, the hazard ratio, which equals e^{β_j} and indicates the proportional change in hazard per unit change in the covariate.

Another attractive feature of proportional hazards regression is that because it makes no assumptions about the underlying hazard function, inclusion of covariates that change over time is a simple process that is included as a standard feature in many software packages. Hazards are calculated at each event time based on the current risk set, regardless of whether individuals had previously been exposed to a different set of covariate values.

Fig. 1. Plan view of Wilder Station (42°40′N, 72°18′W) showing location of forebay, powerhouse, and sluiceway. Inset shows location of facility on the Connecticut River in northeastern U.S.A. (VT, Vermont; NH, New Hampshire; MA, Massachusetts; CT, Connecticut; RI, Rhode Island).



Although Cox's proportional hazards regression is not fully parametric, it still requires certain assumptions about the data, primarily that the effects of covariates on hazard are constant over time. Deviation from this assumption can cause misleading results. The proportional hazards assumption can be tested by plotting Shoenfeld residuals (Shoenfeld 1982; Grampsch and Therneau 1994) against the log of time (any significant slope indicates that proportionality is time-dependent). Another residual, called the score residual, is useful for identifying influential and poorly fit observations (see Hosmer and Lemeshow (1999) for a thorough discussion of fit evaluation for proportional hazards models).

Competing risks

The binary approach to censoring applied in the preceding sections is complicated by the availability of multiple passage routes, a common feature of downstream passage studies. Before passage, fish are available to pass through all routes, i.e., they are part of the risk set. Once a route is selected, however, they no longer contribute to the passage rate of any route. In other words, an individual passing via a given route is effectively censored with respect to the other routes. This constitutes a competing risks situation for which event-time analysis methods are particularly well suited (Allison 1995).

When confronted with a competing risks situation, a multistep approach is appropriate. First, all covariates except passage route are included in the model. This allows inclusion of individuals that were not observed passing through either route (i.e., censored observations) and provides the best estimate of overall passage rate. Next, separate models are developed for individuals passing through each route of interest by modifying the censoring variable. For each model, non-passers, as well as those that pass through alternate routes, are included, censored at time of passage; only those that pass through the route of interest are noncensored. The advantage of this step is that it evaluates separately variables that affect the rates at which fish pass through each route. Although different results for competing passage routes suggest some underlying difference in covariate effects on passage rate that may be of substantial biological interest, the competing risks approach does not test for these differences explicitly; it is simply a means of quantifying covariate effects on passage rates through a particular route. This means that researchers can use the entire risk set when analyzing passage rates through any route, or combination of routes, and the approach can be applied to any of the methods detailed above, a potent tool for evaluating the effects of facility operations on passage rate through a particular route. The only assumption required is that fish that have not yet passed the dam are equally available to pass via all routes and should be included when evaluating the effects of covariates on the groups.

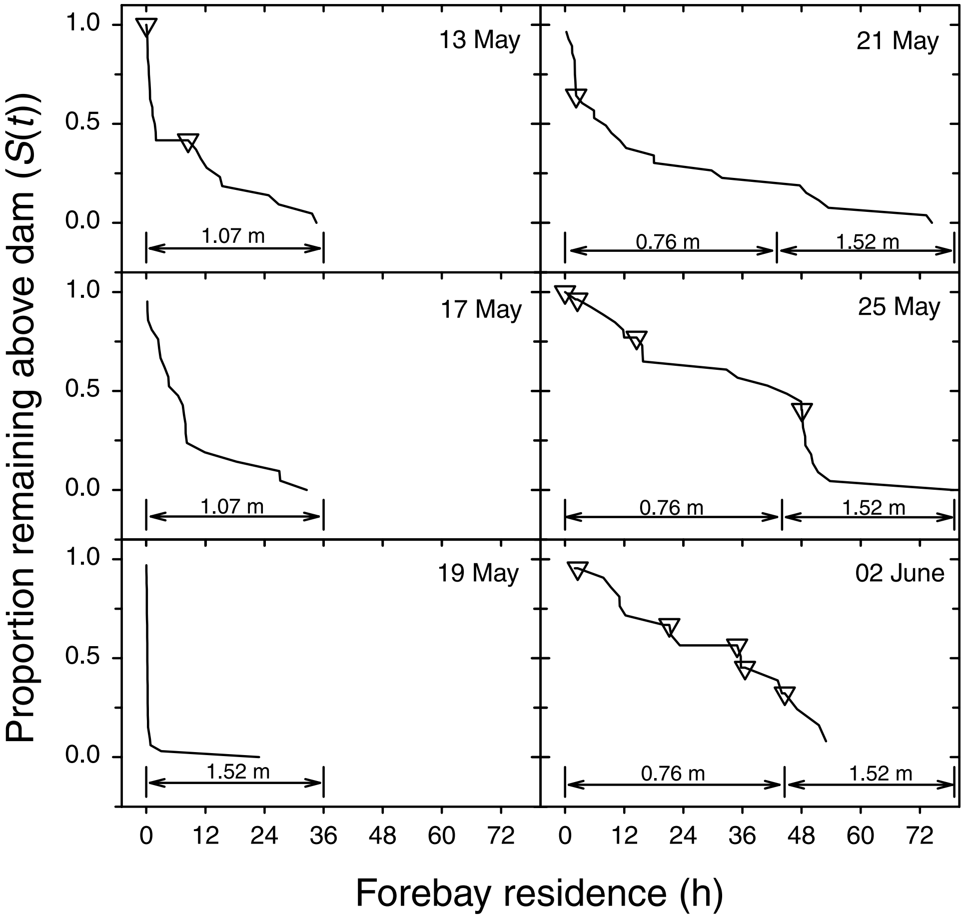
Logistic regression

Although the competing risks approach allows separate analysis of rates of passage through various routes, it does not directly quantify which variables most influence the likelihood of selecting one route over another. Logistic regression is a standard method for quantifying covariate effects on the likelihood of selecting one of two or more categorical variables (Hosmer and Lemeshow 1989). By including time as a covariate, the effect of delay on passage route selection can be tested directly. Censored individuals, because they are not observed to actually pass, are generally not included in such analyses (Allison 1995).

Data set

We selected a sample data set (RMC Environmental Services, Inc., currently Normandeau Associates, 917 Route 12,

Fig. 2. Kaplan–Meier curves describing forebay residence times of radio-tagged Atlantic salmon (*Salmo salar*) smolts passing Wilder Station during each of six releases. Open triangles indicate censored individuals. Gate depth is indicated in each panel, as well as shifts from 0.76 m to 1.52 m during the last three releases. Note the rapid passage on the 19 May release and increased passage rate following increased gate depth during the last three releases.

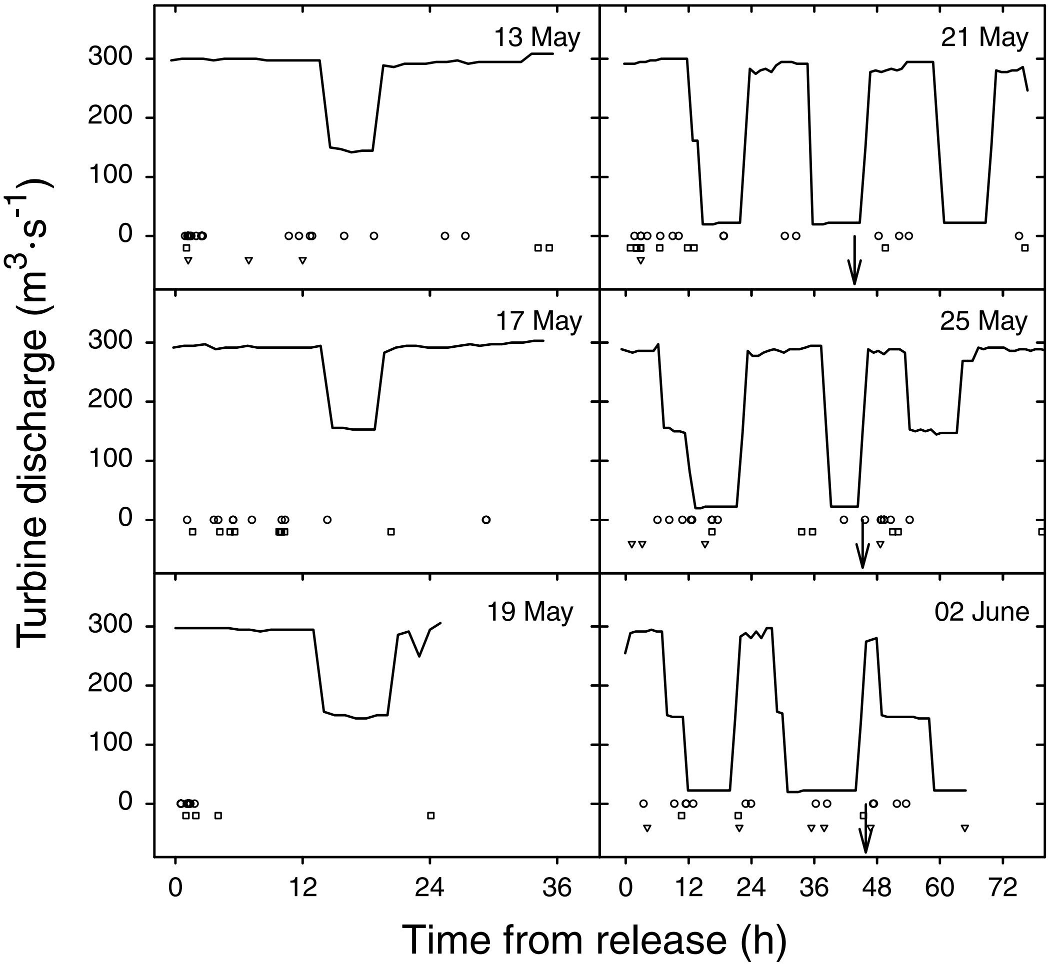


Suite 1, Westmoreland, NH 03467, U.S.A., unpublished data) to demonstrate the utility of each of the event-time analysis methods for describing the effects of operational modifications and other variables on delay. Here we define delay as the time elapsed between forebay entry and passage, i.e., forebay residence time. The study was conducted in 1994 at Wilder Station, a hydroelectric facility on the Connecticut River mainstem at river-kilometre 348, and was designed to test the effect of different depth settings of a bypass sluice gate on passage route selection. The ice–log sluice used for downstream smolt passage is located adjacent to the powerhouse (Fig. 1). A 3.0 m × 4.6 m skimmer gate regulates surface flows of 1.38 $m^3 \cdot s^{-1}$ at 0.305-m depth to 18.7 $m^3 \cdot s^{-1}$ at 1.83-m depth. When operated for smolt passage, the gate is normally set to 1.07-m depth, passing 8.77 $m^3 \cdot s^{-1}$ down an 18.3-m-long sluice into the station tailrace. The powerhouse contains two 19-MW Kaplan turbines and one 3.2-MW generator, protected by trashracks with 15 cm horizontal and 48 cm vertical spacing. No spill occurred during this study, so all flow passed through the turbines or over the bypass sluice. These were the only two passage routes available to the smolts during this period.

Atlantic salmon smolts were obtained from two sources: a bypass sampler on the Connecticut River mainstem ("wild" fish; n = 65, fork length (FL) = 137–235 mm, $\overline{FL}$ = 179 mm), and the White River National Fish Hatchery in Bethel, Vermont ("hatchery" fish; n = 93, FL = 152–218 mm, $\overline{FL}$ = 190 mm). Source of fish is referred to as "origin" and coded 0 and 1, respectively, for wild and hatchery smolts in analyses. Smolts were anaesthetized using MS-222 and radio-tagged using esophageal implants. Following a 24-h recovery period, fish were released 1 km upstream of Wilder Station. This was considered sufficient distance to prevent any predisposition on the part of the smolts to pass through one route over the other. Smolts were released in six groups of 21–33 individuals on 13, 17, 19, 21, and 25 May and 2 June 1994 (Fig. 2). Wild fish made up 48% (33–58%) of the first five releases; the last release consisted of hatchery fish only. Telemetry receivers were placed in such a way that smolts were detected when they entered the forebay of the project (which extended about 200 m upstream of the dam) and were monitored continuously during their forebay residence. A four-element YAGI antenna situated halfway down the sluice identified sluice passers, and antennas submerged at the entrance of each intake identified turbine passers. Time to passage (delay or residence time) was calculated from the time that fish first entered the forebay to ensure that only data from actively migrating fish were used.

Turbine flow was logged each hour and ranged from 19.8 to 308 $m^3 \cdot s^{-1}$, $\bar{X}$ = 253 $m^3 \cdot s^{-1}$. Because fish that did not im-

Fig. 3. Turbine flow at Wilder Station during each of six releases. Data are presented as hours from each release. Points indicate passage time for sluice passers (circles) and turbine passers (squares); censored times are indicated by triangles. Arrows indicate time to increased gate depth in each of the last three releases.



mediately pass the station were often subjected to more than one level of turbine flow (Fig. 3), this was not included in the parametric models but was included as a time-dependent covariate in the proportional hazards models.

A similar complication arose with respect to sluice gate depth. This was set to 1.07 m for the first two releases, 1.52 m for the third release, and 0.76 m for the last three releases. For each of the last three releases, the sluice gate depth was increased to 1.52 m after 44–46 h (Fig. 2). To prevent the artificial association of greater gate depth with long passage delays, we censored data from these releases at the corresponding residence times for the parametric regression analysis but included the data with gate depth as a time-dependent covariate in the proportional hazards analysis.

Because fish could only pass through the turbines or over the sluice, passage route constituted a competing risks variable in this study. Separate models were generated for each route as well as for the combined data using fully parametric and Cox's proportional hazards techniques. To test for differences between routes in covariate effects on passage rate, we ran the above tests, including route and its interactions with the other covariates.

We used SAS software (SAS 1999) to estimate covariate effects on passage time. We selected from among exponential, Weibull, lognormal, and generalized gamma distributions, testing for the best fit using likelihood ratio statistics and AIC. Where nested models were not significantly different from each other, we selected the most parsimonious model, i.e., the one with the fewest parameters. Adequacy of the parametric models was evaluated both numerically and graphically, using each of the methods detailed above. We evaluated adequacy of proportional hazards models using Schoenfeld and score residuals.

In addition to the above tests, we used logistic regression to test for covariate effects on likelihood of passing through the bypass sluice, including log of delay time as a covariate. All analyses were conducted using SAS software (SAS 1999).

Results

Of the 158 Atlantic salmon smolts used in this study, 14 (eight hatchery and six wild) had undetermined passage routes or failed to pass; these were included in the analyses, censored at their last extant observation. In all, 144 smolts passed the station by known routes: 106 over the bypass sluice and 38 through the turbines. Most fish entered the forebay shortly after release; mean ± standard deviation (SD) of postrelease delay was 1.27 ± 1.70 h, with all but three in-

Table 1. Results from parametric and Cox's proportional hazards regression.

		Parametric			Proportional hazards		
Data set	Variables	*N*	$\hat{\beta}$	*P* value	*N*	$\hat{\beta}$	*P* value
Combined data	Number passed	121			144		
	Number censored	37			14		
	Intercept		–787.190	0.018		—	—
	Origin		–0.007	0.981		0.042	0.810
	Release date		0.063	0.017		–0.019	0.323
	Turbine flow (m^3·s^{-1})		—	—		0.001	0.611
	Gate depth (m)		–5.459	<0.001		3.528	<0.001
	Scale		1.589			—	—
Sluice passers	Number passed	89			106		
	Number censored	69			52		
	Intercept		–499.125	0.183		—	—
	Origin		0.163	0.609		–0.055	0.787
	Release date		0.040	0.175		–0.025	0.289
	Turbine flow (m^3·s^{-1})		—	—		–0.003	0.210
	Gate depth (m)		–6.316	<0.001		3.972	<0.001
	Scale		1.695			—	—
Turbine passers	Number passed	32			38		
	Number censored	126			120		
	Intercept		–1460.460	0.034		—	—
	Origin		–0.462	0.406		0.316	0.378
	Release date		0.117	0.033		–0.029	0.433
	Turbine flow (m^3·s^{-1})		—	—		0.009	0.024
	Gate depth (m)		–2.562	0.039		1.820	0.068
	Scale		2.185			—	—

Note: The competing risks approach was applied to each passage route, where censored individuals include those passing through the alternate route. Parametric models are based on the lognormal distribution: coefficients ($\hat{\beta}$) indicate effect of each variable on the log of delay ($\ln(T)$); scale refers to the error term. Coefficients for the proportional hazards models indicate their effect on the log of the hazard ($\ln(h(t))$). Origin is coded 1 (hatchery) and 0 (wild).

dividuals entering within 4 h. Kaplan–Meier survivorship curves of residence time ($S'(t)$; Kaplan and Meier 1958) are presented for each release group (Fig. 2).

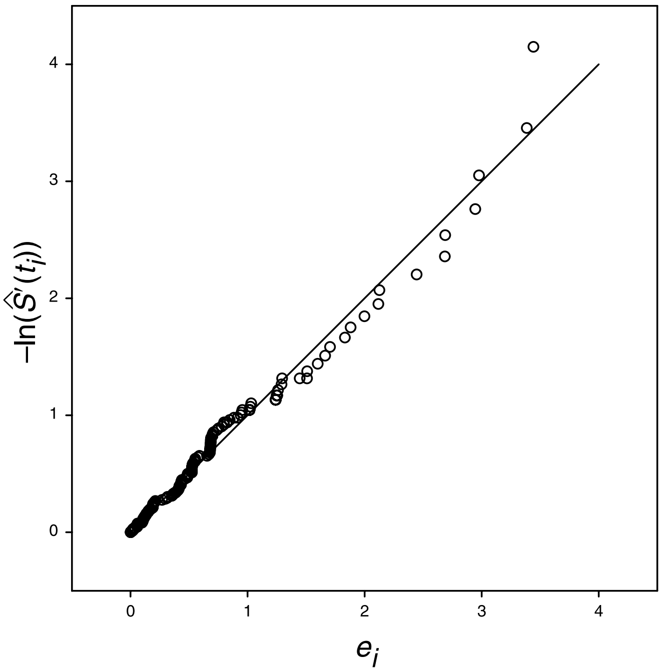
Results from both parametric and proportional hazards models (Table 1) should be interpreted with some caution, as there was significant collinearity among covariates. This effect was greatest between release date and both gate depth and turbine flow. Correlation coefficients were in all cases less than 0.45, however, and the effect of collinearity on the models should be small.

Among the parametric models, the generalized gamma distribution provided a better fit to the combined data than did the Weibull or exponential distributions (χ^2; 1 and 2 df, respectively; $P < 0.001$). However, the lognormal and gamma distributions provided nearly identical fits (χ^2; 1 df; $P = 0.81$). Based on these results, combined with the AIC values, the Hollander and Proschan test ($P = 0.20$), and analysis of Cox–Snell residuals (Fig. 4), we concluded that the lognormal distribution was the most appropriate and parsimonious of the distributions tested and that it adequately described the data. Under this parameterization, the scale variable is analogous to the error term under the standard normal distribution; location is estimated by $\boldsymbol{x\beta}$. The estimated scale values of 1.6–2.2 indicate that passage rate (i.e., hazard) follows an inverted U shape: initially low, it rapidly increases to a maximum value and then declines gradually over time (Meeker and Escobar 1998).

Coefficients of the parametric models indicate covariate effects on $\ln(T)$. These describe reduced delay with increased gate depth for combined passage data and for both passage routes, as indicated by a significant negative coefficient. The interpretation of this for the combined data, adjusting to centimetres, is $T = 100(e^{-0.055} - 1)$, or mean delay decreases by 5.4% for every centimetre of increased gate depth. The same transformation for sluice and turbine passers shows a 6.1% and 2.6% decrease in delay time per centimetre increase of gate depth, respectively. For the current data set, this implies that by increasing gate depth from the shallowest to deepest settings and setting all other covariates to their mean values, median delay declines from 19.9 to 0.3 h when both passage routes are available, and delay of the 90th percentile declines from 152.8 to 2.4 h (Table 2). The parametric approach also suggests that fish released later in the study passed more slowly than earlier releases (positive β), regardless of passage route.

Results of proportional hazards regression indicate that gate depth affected passage rate, particularly among sluice passers (Table 1). Faster passage rate at greater depths is indicated by a significant positive coefficient (greater hazard; note the contrast with the parametric approach). Adjusting to centimetres and transforming the data to risk ratios, we find here a 3.6% increase in passage rate associated with each centimetre of gate depth for combined data and a 4.0% increase for sluice passers.

Fig. 4. Graphical assessment for goodness-of-fit of the lognormal parametric regression model using the total data set (sluice and turbine passers combined). Cox–Snell residuals (e_i) are plotted against the negative log survivorship function (–ln(S′(*t*)), calculated using the Kaplan–Meier method (circles). A line with slope of unity, an indicator that the model provides a reasonable fit, has been included for reference.



Turbine flow also affected passage rate, but only among fish that passed through the turbines. Here, the positive coefficient indicates a hazard ratio of 1.009. Over the range of turbine flows encountered in this study, this means that the rate of fish passage through the turbines was 2.6 times greater under the highest flow than under the lowest flow.

Residual analysis confirmed that the assumption of proportionality was met in these models. However, analysis of the score residuals suggested that the last fish to pass from the 19 May release was influencing the results, particularly with respect to the coefficients for gate depth. When this individual was removed from the analysis, the coefficients for total data and sluice passers increased to 4.7% and 5.3% per centimetre increase in gate depth, respectively. The only other notable effect of removing this data point was to increase the *P* value for the coefficient describing the gate depth effect on turbine passers to 0.121, casting further doubt on its importance.

Significance tests for different effects of gate depth and turbine flow on rates of sluice and turbine passage suggested that covariate effects did differ between the two routes, although neither the parametric nor the proportional hazards approach found a significant difference in passage rate by route (main effect), the interaction of route × gate depth was strongly significant for both models ($P < 0.001$). In contrast, the interaction of route × turbine flow was only marginally significant ($P = 0.063$; proportional hazards model only). It is important to emphasize here that these tests, because they include passage route, exclude the censored observations and should therefore be considered biased approximations.

Table 2. Predicted delay times from the parametric regression model (Table 1) for median and first and last deciles of Atlantic salmon smolts passing a hydroelectric facility by either route (“combined data”) under three different sluice gate settings.

			95% bounds	
Gate depth	Percent passed	Predicted delay (h)	Lower	Upper
0.76	10	2.60	1.83	3.71
	50	19.94	14.56	27.32
	90	152.78	101.16	230.77
1.07	10	0.49	0.37	0.66
	50	3.78	3.02	4.73
	90	28.93	20.67	40.50
1.52	10	0.04	0.02	0.07
	50	0.31	0.19	0.50
	90	2.38	1.41	4.03

Table 3. Logistic regression results describing the effect of covariates on passage route selection.

Source	$\hat{\beta}$	*P* value
Intercept	–4.0930	0.9936
Origin	–0.3732	0.3931
Release date	0.0004	0.9904
Turbine flow ($m^3 \cdot s^{-1}$)	–0.0077	0.0158
Gate depth (m)	0.4549	0.0339
ln(hours)	–0.2651	0.0234
N	144	

Note: Positive coefficients ($\hat{\beta}$) mean increased probability of sluice passage; origin was denoted as 1 (hatchery) and 0 (wild); gate depth and turbine flow indicate settings at time of passage. Fish that failed to pass were excluded from this analysis; sample size (*N*) refers to passers only.

The logistic model (Table 3) demonstrates that the probability of turbine passage increases with delay (note that data from censored individuals are omitted). Probability of turbine passage was likewise increased by greater turbine flow but was reduced by greater sluice gate depth.

Discussion

Although the results of this study inevitably have management implications, readers should bear in mind that our objective is to introduce a new technique and demonstrate its use. RMC (1994) correctly concluded that the gate depth setting of 1.52 m afforded more expedient downstream passage to emigrating smolts and the sluice is currently operated at this depth for fish passage. Therefore, the intent of this paper is not to call into question existing management decisions, but to present techniques that will lead to further and more complete investigations of bypass configurations and other passage evaluations. Moreover, it is important to recognize that although we define delay here as forebay residence time, the term implies a change relative to some minimum transit time. This value is unknown, but its identification and characterization should be an objective of future studies.

Our results do highlight some important differences among the event-time analysis techniques, as well as their relative strengths. Survivorship curves are simple to construct, and their significance is readily interpreted. Here, they show that although the first half of the fish pass fairly quickly, passage rate declines with time and the slowest fish take several days to pass the project. Reporting only median passage time masks this important feature (Venditti et al. 2000), a fact that should be of some concern to restoration efforts. The survivorship curves also provide graphical representation of the effects of sluice gate depth setting. Delay times were least for the 19 May release (gate depth of 1.52 m) and greatest for the last three releases (gate depth of 0.76 m). Those fish that delayed passage until after the gate was set to a greater depth show a correspondingly greater rate of passage at that time.

More censoring was observed for later release dates and shallower gate settings. However, there is no reason to believe that censored and complete observations differed with respect to their response to covariate values. With the exception of the competing risks models, the exact cause of censoring was not determined and may have been due to mortality, tag failure or expulsion, or undetected passage.

Although nonparametric methods for analyzing data with censored observations exist, adequate sample size and the presence of multiple, nonorthogonal covariates indicated the use of a regression approach in this analysis. Both the parametric and Cox's proportional hazard models simultaneously account for all covariate effects (Type III hypotheses; SAS 1999), but they differ in important ways. First, because the parametric models describe covariate effects on forebay residence time, we are able to use them to estimate time to passage of specific proportions of the population at defined covariate levels, a useful tool for managers and those interested in understanding population-level implications of delay. The proportional hazards models, by contrast, describe covariate effects on passage rate (hence the reversed sign of the regression coefficients: passage rate is inversely related to residence time) and cannot be used to directly estimate delay time (but see Hosmer and Lemeshow (1999) for methods by which indirect estimates can be extrapolated).

The parametric models can also be used to draw important inferences on the shape of the hazard and consequently the distribution of quantiles. Here, the numerical and graphical goodness-of-fit analyses both suggest that the lognormal models provide a reasonable fit to the data. This implies that the log delay times are approximately normally distributed (Allison 1995). The inverted U-shaped hazard function described by these models may have some biological significance, as it implies both initial delay (as might be expected from time required to locate the passage route) and reduced passage rates for fish that do not subsequently pass quickly. Thus, the risks of delay include increased likelihood of further delay, a pattern that one would predict if delay resulted in loss of migratory motivation (Meeker and Escobar 1998).

Although the Cox's proportional hazards models do not yield parametric descriptions of the hazard, their independence from specific distributions make them robust against fluctuating passage rate, e.g., resulting from diel migratory patterns, requiring only that covariate effects on hazard ratios be consistent across these patterns. With either approach, we advise against extrapolating far beyond observed values of covariates or delay times.

A further distinction is that proportional hazards models allow ready computation of effects of covariates that change over time, whereas the parametric models assume that each fish is exposed to fixed covariate conditions. This feature has important implications for the interpretation of the effects of release date and gate depth, as well as turbine flow. Because the shallowest gate settings were applied at the end of the experiment, the experimental design was unbalanced, and variability caused by this factor is wrongly apportioned between gate depth and release date under the fully parametric model. By including gate depth as a time-dependent covariate, potentially confounding effects between the two variables are reduced. Under this less biased interpretation of the data, delay is appropriately attributed to gate depth and not to date.

Similarly, fluctuations in turbine flow precluded its inclusion in parametric models — any quantity used (e.g., initial, mean, or final turbine flow) would yield misleading results, because it would not reflect the range of flows to which individual fish were exposed with their associated passage rates. As with the sluice gate settings, minimum daily discharge was less for later releases, and the significant date factor in the parametric models may have arisen in part from the covariance between date and discharge. When turbine flow was included in the proportional hazards models as a time-dependent covariate, it was found to significantly increase the rate of turbine passage, but not sluice passage, and was nonsignificant for the total data.

Both turbine passage models, with their weak effects of turbine flow and gate depth, also illustrate one of the limitations of event-time analysis: because only 38 fish passed through the turbines, over 75% of the observations are censored. This increases the standard error of the estimates, and the power of these models decreases with increased censoring. Thus, the weak significance values for gate depth should be viewed with caution: it may be that increased gate depth does increase passage rate for fish that pass through the turbines (as indicated by the parametric model), but the proportional hazards model lacked sufficient power to detect this effect.

The competing risks condition of this study illustrates the utility of event-time analysis techniques in analyzing passage data. The experimental design reasonably allows one to ask the question: does gate depth affect the passage rate through the bypass sluice? Including turbine passers among the censored data provides the clearest, least-biased answer to this question. Conversely, by censoring sluice passers, we were able to detect the effect of turbine flow on rate of passage through the turbines, which was not apparent from the analysis of the total data. This approach, however, does not directly test for the significance of differences in passage time between the two routes. Indeed, in the presence of censored data, there is no available way to conduct such a test objectively. Here, we constructed a model on the noncensored data that included passage route and the interactions of route × gate and route × turbine flow. The results showed strong significance for the former test but marginal significance for the latter. Because data from censored fish are omitted, these results are of necessity biased; at best, the resulting models tend to underestimate time to passage (eq. 6). They do, however, describe the data from those fish with observed passage times and routes and further support the results of the competing risks analysis.

Whether the increased passage rate with increased gate depth resulted directly from the associated increase in flow or whether it reflects behavioral avoidance of the surface is unclear from this study. Surprisingly, greater gate depth was also associated with increased turbine passage rate. This may be because increased flow over the sluice attracted fish to the powerhouse area or otherwise altered forebay hydraulics such that fish passed more quickly through both routes. The effect of turbine flow supports the view that hydraulics affect passage rate, but the results for sluice passers remain ambiguous. Future studies should attempt to simultaneously control for volume and depth of bypass flows, as well as the ratio of bypass flow to turbine flow, to improve orthogonality of these factors.

Each of the techniques described above has specific advantages and can provide unique information on passage rate. The most appropriate approach will depend on time resolution (Cox's proportional hazards regression can incorporate ordinal time data, whereas parametric regression requires a continuous time variable), shape of the hazard function, and research objectives. None of the above techniques, however, quantifies covariate effects on route selection as such. Logistic regression does just this and thereby complements event-time analysis. Bearing in mind that data from 14 individuals are missing, logistic regression reveals a significant time effect, with greater delays associated with increased risk of turbine passage. This result alone can be a powerful argument for trying to maximize passage rates. Also significant in the logistic model are effects of turbine flow and gate depth, with greater flow and shallower gate settings associated with increased risk of turbine passage.

Combining the logistic and event-time approach, we conclude that shallower gate depth not only increased delay, but also simultaneously (in part, because of the delay) increased the likelihood of turbine passage, particularly in the presence of high turbine flows. These results illustrate the complementary nature of event-time and logistic regression approaches: shallower sluice gate settings reduced passage rates, especially through the sluice. This, in turn, increased the time during which fish were exposed to the possibility of passing through both routes, thereby increasing the likelihood of turbine passage. Thus, by modifying operations to maximize the rate at which fish pass over the sluice, both delay and likelihood of turbine passage could be minimized simultaneously.

The bulk of current fish passage research work focuses on proportions of fish passing through various routes, primarily because this is thought to have the greatest relevance to survival and recruitment (Burnham et al. 1987; Skalski 1998; Skalski et al. 1998). Although the importance of delays to migration is not well understood, it is bound to vary by species, river system, and life history (McCormick et al. 1996; Zydlewski and McCormick 1997*b*; Zabel et al. 1998). Our understanding of the effect of delay is limited at the outset by our ability to quantify it. Event-time analysis provides a powerful set of tools for developing just such descriptions, as well as for evaluating effects of structural and operational modifications on passage rates. Because they afford continuous monitoring of individuals, radiotelemetry and acoustic telemetry are particularly well suited to these analyses. Other forms of telemetry and monitoring (e.g., from passive integrated transponder (PIT) tags) may also be useful; however, it is important that time to passage or censoring is known. Because PIT tags tend to have relatively short read ranges (Prentice et al. 1990; Castro-Santos et al. 1996), it may not always be possible to identify censoring times using this technology, although the competing risks approach could still be applied to some data.

Although this paper focuses on the application of event-time analysis to a radiotelemetry study of downstream fish passage, the techniques have much broader potential. Analogous applications include quantifying attraction of upstream migrants to fishway entrances, monitoring progress up fishways (where height can be substituted for time as the dependent variable and successfully exiting the top of the fishway constitutes censoring), and quantifying timing of movements of other migratory species: in short, any application may be appropriate where censoring and competing risks confound the use of standard techniques.

Acknowledgments

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PETITION TO LIST THE AMERICAN EEL *(Anguilla rostrata)* AS A THREATENED SPECIES UNDER THE ENDANGERED SPECIES ACT

Eel Mortality from Turbines -AD Latornell Symposium 2008

Submitted To: U. S. Fish and Wildlife Service, Washington D.C. and Sacramento Field Office, California

Submitted By: Council for Endangered Species Act Reliability

Date: April 30, 2010

Council for Endangered Species Act Reliability

CESAR

April 30, 2010

VIA FACSIMILE & CERTIFIED MAIL

Secretary Kenneth Salazar
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Re: American Eel Petition

Dear Sirs and Madame:

The Council for Endangered Species Act Reliability ("CESAR") hereby petitions the Departments of Interior and Commerce to list the American eel (*Anguilla rostrata*) as threatened pursuant to the federal Endangered Species Act ("ESA"), 16 U.S.C. §§ 1531, et seq. This petition is filed under 5 U.S.C. § 553(3) and 50 C.F.R. § 424.14 and includes new information that became available subsequent to the 'not warranted' 12-month finding published by the Fish and Wildlife Service ("FWS") on February 2, 2007[1] (herein "2007 Final Determination").

The petition includes this cover letter and the attached petition consisting of Parts I through IV, as well as all documents cited herein which are hereby specifically incorporated by reference.

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[1] Fed. Reg. 22, 4967, 22, 4997 (Feb. 2, 2007).

Please do not hesitate to contact me at (916) 341-7407 if you need more information. My address appears above.

Sincerely,

Craig Manson

Craig Manson
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cc: Gary Frazer
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Enclosures

Petition to List the American Eel as an Endangered Species Pursuant to the United States Endangered Species Act 16 U.S.C. §§ 1531, et seq.

PETITIONERS:

Petitioner the Council for Endangered Species Act Reliability ("CESAR") requests the United States Fish and Wildlife Service ("FWS") and the National Marine Fisheries Service ("NMFS") list the American eel (*Anguilla rostrata*) as threatened under the United States Endangered Species Act, 16 U.S.C. §§ 1531, et seq. This petition is filed under 5 U.S.C. § 553(e) and 50 C.F.R. § 424.14.

I. STATUS OF THE AMERICAN EEL

The American eel (herein "American eel" and "eel" interchangeably) is in steep decline across its range. This decline commenced in the mid-1980s and has continued to present populations which demonstrate a decrease of several orders of magnitude from the near past. The decline is based on the following factors:

- **Loss of habitat** – American eels have lost an estimated 84 percent[2] of their habitat; much of it due to the operation of dams which impede or completely block migration, reducing or removing habitats available for spawning, feeding, and growth. Further, dams have fragmented river habitats and changed upstream habitat by slowing water flow and changing temperatures. In addition, river habitat has been altered by changes in streambeds and banks and streamside vegetation, all of which are affected by the operation of dams.
- **Overutilization** – commercial and recreational fisheries that harvest virtually every life stage of the American eel throughout its habitat do so with little to no regard for population status.
- **Disease** – The spread of the invasive swim bladder parasite, *Anguillicola crassus*, has disrupted the eel's swim bladder function. This parasite, introduced to the immunologically naïve American eel, has quickly colonized populations and created what appears to be a potentially catastrophic epizootic.
- **Inadequacy of existing regulatory mechanisms** – The only regulatory authority currently exercised is that of Atlantic States Marine Fisheries Commission ("ASMFC"). That organization has done little over the past decade to effectively reverse the declines in eel recruitment, halt commercial and recreational take of American eels, or

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[2] Busch et. al 1998; Using spatial data from the EPA, dam locations from the U.S. Army Corps of Engineers, and eel presence/absence data from the State of Maine and the U.S. Fish and Wildlife Service, the authors found a reduction of 84% of the stream habitat available. This estimate is conservative as it only tallies losses on the American portion of the eel habitat, and not addressing habitat loss on the Canadian side.

implement consistent methods to accurately assess their population size. Other available authorities, such as that of the states and the Federal Energy Regulatory Commission ("FERC"), have been exercised only sporadically and are clearly insufficient to halt the decline of the species.

- **Other factors** – Other documented factors adversely affecting American eel populations include climate change, mortality and morbidity from acidification of stream flows, mortality and injury in hydroelectric turbines when mature eels are migrating downstream, and contaminants ranging from Mercury to PCBs.

This petition summarizes the natural history of the American eel, population information, and a description of existing threats to the species and its habitat. Petitioners are seeking listing of the species as threatened under the ESA. This petition is based on information developed by the FWS in its 2007 Final Determination that listing was "not warranted", new information published since that final agency action, as well as information not considered in that review.

A. Background

On May 27, 2004, the ASMFC requested that the FWS and the NMFS conduct a status review of the American eel based on extreme declines in the Saint Lawrence River/Lake Ontario portion of the species' range. The ASMFC also requested an evaluation of the appropriateness of a Distinct Population Segment ("DPS") listing under the ESA as well as an evaluation of the entire Atlantic coast American eel population. The FWS responded that the American eel was not likely to meet the discreteness element of the policy requirements due to lack of population subdivision. However, the FWS did undertake a range wide status review of the American eel in coordination with NMFS and ASMFC.[3]

On November 18, 2004, the FWS and the NMFS received a petition requesting the listing of the American Eel as a threatened species under the ESA. The petitioners cited destruction and modification of habitat, overutilization, inadequacy of existing regulatory mechanisms, and other natural and man-made factors (such as contaminants and hydroelectric turbines) as the threats to the species. After initially finding that the petition presented substantial information indicating that listing the American eel may be warranted, the FWS made a final determination that listing of the eel under the ESA was "not warranted". The final rule contains the following findings:

- The species has been extirpated from some portions of its historical freshwater habitat over the last 100 years or so, mostly as a result of dam construction which blocked access;

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[3] U.S. Fish and Wildlife Service, 12-Month Finding on a Petition To List the American Eel as Threatened or Endangered, 72 Fed. Reg. 4967-4997 (Feb. 2, 2007).

- There is also evidence that the species' abundance within freshwater habitats, and to some degree estuarine habitats, has declined in some areas likely a result of harvest or turbine mortality, or a combination of factors;
- The species remains widely distributed over the majority of its historical range;
- An indication of decline exists in yellow eel abundance, but recent glass eel recruitment trends, although variable from year to year, appear stable over the past 15 years;
- The American eel is a highly resilient species, with the ability to occupy the broadest range of habitats within freshwater, as well as estuarine and marine waters, and it remains a widely distributed fish species.
- Although roughly 25 percent of the American eel's historical freshwater habitat is now inaccessible due to dams, the loss of this habitat does not threaten the species' long-term persistence;
- A large amount of freshwater habitat still remains (roughly 75 percent of historic freshwater habitat in the United States remains available and occupied by the American eel);
- Although the significance of the estuarine and marine eel contribution to reproduction is considered speculative by some there is no doubt that substantial amounts of estuarine and marine waters remain available to, and are occupied by, the American eel throughout its range;
- Recreational and commercial eel harvests are no longer factors of concern at a population level due to economics, the species' resilience, and existing regulatory mechanisms;
- Although mortality during outmigration due to parasites and contaminants, and the potential effects of contaminants on early life stages, remain a concern, there is no information indicating that these threats are currently causing or are likely to cause population level effects to the American eel;
- There is no information indicating that predation or competition with non-native species or mortality from turbines is causing population-level effects;
- Recruitment success of the American eel is dependent on ocean conditions, and variation in ocean conditions cause fluctuations in recruitment. However, because the available information indicates that the species remains widely distributed and glass eel recruitment trends appear stable over the past 15 years: observed ocean conditions do not threaten the current population status of the American eel;
- There is no information to indicate that ocean conditions are likely to threaten the American eel at a population level in the future.

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1. FWS 2005 Status Review Information

The FWS reports that its 2007 12-month finding is based on the contents of the petition, existing literature, and information gathered during the status review that preceded the finding. The FWS identifies the documents most relevant to the status review as the stock assessments for the Atlantic coast, the American eel data assembled for the Canadian stock assessment and specific published research on life history and potential threats to the American eel.

The 12-month finding stated that the status review focused on available data within the North American Continent. The FWS was unable to identify data on eel distribution, habitat use, habitat degradation or loss, or other threats (other than international harvest data) from Central or South America, although some Caribbean Islands provided distribution information.

The 12-month finding referenced two scientific workshops in which over 25 scientific experts participated. The expert panelists represented a broad and diverse range of scientific perspectives relevant to the status review of the American eel. Participating individuals had expertise on threats, or life history characteristics associated with threats, to the American eel. Each of the participating experts was asked a series of questions. These questions asked the experts to assess the information used by the FWS in their status review as to its completeness, relevance, and quality. The FWS recorded each expert's individual assessment of the likelihood of eel extinction based on the information presented.[4]

B. Evolution And Population Structure

The American eel evolved approximately 52 million years ago and is among the longest-living animals in North America and one of longest-lived fishes of North America. A record exists of an American eel living 88 years in captivity and Swedish television carried the story of Håkan Wickström, who pulled a 130+ year old eel out of a well! American eels are both catadromous[5] and diadromous[6]. Eels spawn in ocean waters, migrate to coastal and inland continental waters to grow, and then return to ocean spawning areas to reproduce and die. Female American eels in northern latitudes reach ages of 20-50 years old before their one-way spawning migration to the Sargasso Sea.

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[4] *Draft Minutes*, American Eel Great Lakes/Canada Threats and Population Dynamics Workshop U.S. Fish and Wildlife Service in cooperation with National Marine Fisheries Service January 31 — February 2, 2006 Buffalo, New York and FWS. 2006. *Draft* Minutes from the American eel status review Workshop 2: Great Lakes/Canada threats and population dynamics. Buffalo, NY, January 31-February 2, 2006.

[5] Meaning that they live in fresh water, and breed in the ocean.

[6] Meaning that they travel between salt and fresh water.

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There are two closely related recognized species of eel found in the North Atlantic -- the American eel and the European eel. Genetic research indicates that the American eels are one, well-mixed, single breeding population. This is in contrast to many anadromous species (which, even though they have an oceanic phase, return to their rivers of origin to spawn), where mating is within separate populations that are geographically or temporally isolated. Similarities between the American and European Eel are remarkable.[7]

C. Life History

American eel eggs hatch in the Sargasso Sea. Ocean currents are hypothesized to transport the eels to the Atlantic coasts of North America and northern portions of South America and the Gulf of Mexico. The Gulf was a significant route when they populated the Mississippi. European reporters of the Illinois Nation document heavy use of eels in the upper Mississippi River. They enter coastal waters, where they may stay, may move into estuarine waters, or migrate up freshwater rivers. Upon nearing sexual maturity, these eels begin migration toward the Sargasso Sea, completing sexual maturation en-route. Spawning occurs in the Sargasso Sea. After spawning, they are believed to die.

Eels usually live on the muddy bottoms of freshwater streams or in freshwater stream-fed ponds. They generally seek deep streams and often work their way up brooks along the coast, but can penetrate hundreds to thousands of kilometers inland via major waterways. For example, historically, the American eel was found as far inland as Iowa (via the Mississippi River). Eels are able to leave the water and hide under muddy stones in swampy ground a few feet from the shores and can forage on the sand along streamsides. Eels are omnivorous.

Much of the information surrounding the ecology of eels is the result of speculation, hypothesis, or inference as there is inadequate knowledge to more specifically identify their life cycle requirements.

1. Egg and Larval Life History Stage

American eel eggs are believed to hatch into a leaf-like, laterally compressed larval stage known as "leptocephalus" in the Sargasso Sea. American eel spawning and eggs have never been observed. However, as leptocephali are found primarily in the Sargasso Sea, biologists infer the location of spawning and egg distribution. While there have been leptocephali found drifting outside the Sargasso sea, to date, that behavior has been treated as an anomaly. Leptocephali distribute in the upper 300 meters (m) of the ocean and are subject to transport from surface currents. The Sargasso Sea is bounded by a powerful western boundary current, the Florida Current and Gulf Stream, which flows to the north and northeast along the Atlantic coast of

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[7] See page 40 of 42 from the February 2007, 2006 FWS Workshop on the American Eel Great Lakes/Canada Threats and Population Dynamics Workshop: In response to a question of how much the European Eel should be used as a surrogate "*John Casselman noted that the similarities are rather remarkable. He noted that, because of currents, the European species may take longer to get there, but there is remarkable synchrony.*"

North America. The Florida Current transports water from the Caribbean, Gulf of Mexico, and more distant regions through the Straits of Florida.

The transport mechanism for the leptocephali is largely unknown, although it is believed that a majority of the leptocephali enter the Florida Current and Gulf Stream from the Sargasso Sea. Although there are several theories, the path taken by the remainder is unknown. Other than likely current transport, very little is known about the leptocephali. It can be inferred, based on recent studies on other species, that they may feed on marine snow or detrital particles such as zooplankton fecal pellets.

The American eel undergoes significant morphological and physiological changes twice during its life cycle. The first occurs when the leptocephali enter the Continental Shelf waters; the second is during sexual maturation. The leptocephali's leaf-like, laterally compressed shape transforms during metamorphosis into a reduced, characteristically eel-like shape, as they become transparent "glass eels." Leptocephali are unusual fish larvae that are filled with a transparent gelatinous energy storage material, and they can swim either forwards or backwards equally well. This may be an important aspect in leaving the Gulf Stream. This directional swimming is the only way that leptocephali can cross and detrain from the Gulf Stream system and cross the Continental Shelf waters, due to the lack of any persistent oceanic transport mechanism that can account for the large-scale transport of millions of larvae across the current.

2. Juvenile Life History Stage

There is considerable annual variation in the number of juvenile eels in coastal waters as either unpigmented "glass eels" and pigmented "elvers." The variation in recruitment between years can be quite significant. Some of the young eels remain in brackish or salt waters, others migrate up rivers to a variety of fresh water habitats, and others develop movement patterns between these habitats.

Information on mortality rates for all of the eel's life stages is limited. However, the available data from mark-capture studies indicate juvenile mortality rates of 99 percent and elver mortality in fresh waters may be density-dependent.[8] There is uncertainty regarding early juvenile mortality. Surviving elvers mature into fully pigmented "yellow eels."

a) Mortality rates may decrease with size.

One study in Prince Edward Island, Canada, calculated loss from the population due to mortality and emigration. Estimates of loss in American yellow eels from the Prince Edward Island study are reported at 22 percent, with mortality rates decreasing to 12 to 15 percent as the juvenile yellow eels age (Anonymous 2001 in Morrison and Secor 2003, p. 1498), the reasons for this are unknown, but may be due to lower mortality from predation and starvation as size increases.

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[8] Jessop (2000), p. 514, Vøllestad, L.A. and B. Jonsson (1988).

b) Juvenile diet.

Because they are omnivorous, eels can adapt to changes in prey species and abundance. Yellow eels are opportunistic, consuming nearly any live prey that can be captured. Smaller eels eat benthic invertebrates; larger eels include mussels, fish, and even other eels in their diet. Yellow eels also adapt to seasonal changes, decreasing intake or ceasing to eat during the winter. Eels can also respond to local abundances of appropriately sized prey through the seasons (Tesch 2003, pp. 152-163). This adaptability with respect to diet allows for resource partitioning as well as the ability to withstand sudden changes in local environmental conditions and the ability to occupy a geographically wide variety of habitats.

c) Density-dependent dispersal.

As young eels begin to grow, density-dependent competition promotes eels to disperse into less crowded areas (Feunteun et al. 2003, pp. 201-204; Ibbotson et al. 2002 in Knights et al. 2006, p. 10). Aggressive interactions at high density can inhibit feeding and growth, but stimulate dispersive swimming activity in smaller eels (Knights 1987 in Knights et al. 2006, p. 10), the latter likely as a defense against predation. As size differences in these juveniles increase, cannibalism can also be an important cause of mortality (Knights 1987 in Knights et al. 2006, p. 10). Density dependent dispersion ensures wider distributions, further minimizing intra-specific competition. Benefits of density dependent dispersion include selection of optimal habitat productivity and temperature, lower predation risks, rapid colonization or re-colonization of habitats, and avoidance of inter-specific competition. Upstream, larger females predominate, densities of eels decline, and individuals tend to become more sedentary and occupy territories (Feunteun et al. 2003, p. 201).

Generally, density dependence is a function of population, the higher the population of eels, the more likely mortality results from cannibalism. Logically, one would not expect this to occur when a species is experiencing the kind of catastrophic declines exhibited over the past two decades by eels. However, it is possible that anthropogenic barriers could create artificial density conditions where the few eels are found congregated at some barrier such as a dam. At that point, density dependent cannibalism would occur situationally with potentially catastrophic effects for the already small eel population.

d) Distribution clines.

There is a theory that there are latitudinal clines in eel distribution related to river typologies. For example, the American eel tends to extend farther inland in southerly lowland drainages compared to distributions in the shorter and steeper post-glacial stream systems in the Northeast (Jessop et al. 2004 in Knights et al. 2006, p. 11). Smogor et al. (1995, p. 799) and Knights (2001 in Knights et al. 2006, p. 8) have documented decreases in densities with increasing distance from the Continental Shelf in a predictable pattern. Although mean watershed densities decrease by an order of magnitude with distance inland from the

Continental Shelf, mean biomass only declines by about 50 percent because mean body weight and eel length increase (and hence increased fecundity). This, according to Knights et al. (2006, p. 10), helps maintain biomass relative to carrying capacity. Machut (2006, p. 13) indicates that as barrier intensity increases, so does eel growth above the barrier. It is well documented that as eel density decreases, the proportion of females increases, which, assuming females are the limiting sex, would be, according to Knights et al. (2006, p. 13), a compensatory mechanism during times or in areas of low density. However, as discussed later in this petition this mechanism can have unintended consequences if all the of the highly fecund females living above turbines will be destroyed or mortally injured before spawning.

3. Sexually Maturing Life History Stage

a) Sex determination.

There are no morphologically differentiated sex chromosomes in the American eel and, prior to sexual differentiation, eels are intersexual, meaning they can develop into either sex. When yellow eels reach a length of about 20-35 cm it is possible visually to distinguish males from females (by post-mortem inspection of their gonads). There is significant variation in age and size at differentiation. Biologists speculate that sex determination is influenced by environmental factors, including eel densities. Studies indicate that increasing eel density increases the incidence of male eels and decreasing density produces more females. It has been hypothesized that his life history strategy results in population responses that are beneficial to eel conservation. For example, with this strategy, when recruitment declines, so will density and tendencies to migrate far upstream in rivers, which leads to relative increases in the number of larger females and compensatory increases in fecundity. The results of this strategy may take a number of generations (and hence decades) to manifest itself, but it can produce benefits in the face of threats, past, present and future, such as changes in ocean currents and climate. However, the strategy is ineffective when large, fecund females are destroyed by anthropogenic factors (such as turbines) before they can spawn.

b) Silvering.

Beginning at 3 years old in the southern portion of the range, and up to 24 years in the northern portion, the yellow eels begin metamorphosis. The actual age of silvering increases with increasing latitude. The metamorphosis from bottom-oriented yellow eels to silver eels is important physiologically as it prepares the eels for oceanic migration and eventual spawning. It is unknown what actually triggers silvering. It is speculated that environmental factors may contribute. Habitat conditions, such as food availability and temperature, will influence the size and age of silvering eels. Thus, variation in length and age at maturity can occur in different habitats.

Growing season length and temperature vary by latitude and thus, age at maturity also varies with latitude. Characteristics of silver eels vary across the species' range. Eels from northern areas, where migration distances are great, show slower growth and greater length, weight, and age at migration, resulting in their better preparation for a longer migration. This also leads to a higher contribution to recruitment from larger, more fecund females.[9]

It appears that favorable growth conditions cause eels to silver more rapidly, such as is the case in aquaculture, under experimental conditions, or in brackish water and at low latitudes. For example, Morrison et al. (2003, p. 95-96) found annual growth rates in brackish water were two times higher than growth rates of eels that resided entirely in fresh water. Also American eels in the warmer, more stable water conditions of the U.S. southern Atlantic coast waters develop into silver eels about 5 years sooner than northern populations.

Variation in maturation age benefits the population by allowing different individuals of a given year class to reproduce over a period of many years, which increases the chances of encountering environmental conditions favorable to spawning success and offspring survival. For example, variability in the maturation age of eels born in 2006 may result in spawners throughout 2010-2030, during which time favorable environmental conditions are likely to be encountered at least during some periods. However, disproportionate loss of older, more fecund eels to anthropogenic causes can remove that mechanism from contributing to the continued existence of the eel.

Males and females differ in the size at which they begin to silver. Eels appear to need to reach a certain size to begin the silvering process, with this size increasing with age (thus, rapidly growing eels will silver at smaller sizes than slow-growing eels). In males, silvering happens at a very early stage, at a size typically greater than 35 centimeters (cm). In females, silvering happens at a size greater than 40 to 50 cm.

Metamorphosis occurs gradually beginning in summer, and in the fall eels metamorphosing in preparation for migration back to the spawning grounds have a silvery body color, enlarged eyes and nostrils, and a more visible lateral line. During metamorphosis, the structure and metabolism of the liver changes and the swim-bladder also changes.

It is believed that, generally, a drop in temperature signals final metamorphosis characterized by gut regression and cessation of feeding. Once metamorphosis is complete and the appropriate environmental conditions exist, emigration occurs. Biologists theorize that responding to a drop in temperature synchronizes emigrating eels, and increases their chances of reaching the Sargasso Sea simultaneously. The specifics of the important environmental variables are unknown and the subject of much speculation; among the variables considered to have the potential to affect migration are increasing temperatures, delays in migration, or possibly low fat content. It has, however, been observed that even after eggs and sperm have developed, eels are capable of gut regeneration and feeding. This leads to the conclusion that silvering may occur more than once in the lifetime of an eel under specific (and as yet unknown) circumstances. If so, this phenomenon could explain the extreme variability in age and size of silver eels. Biologists are uncertain as to the cause of final sexual maturation of eels,

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[9] McCleave 2001a, p. 803, MacGregor 2008, Knights et al. 2006.

but hypothesize that high pressure they experience during migration in the open ocean may be the trigger.

4. Life History Stage at Emigration

a) Energy requirements.

To successfully complete their emigration from the continent to the Sargasso Sea, great endurance and an extensive fat reserve are required. Larger, fatter eels have an advantage over smaller eels in reaching the Sargasso Sea and having sufficient energy stores to reproduce. Eels are very efficient swimmers and larger eels appear more efficient than smaller eels. Also, larger eels usually have larger fat stores per body weight. Silver eels have ceased feeding, and use their stored fat for energy during their migration and for completing gonadal growth. In a study conducted on European eels, the most recent estimate of necessary energy (fat) needed to successfully complete the migration to the Sargasso Sea from Europe and spawn is 20 percent fat reserves, of which 13 percent is for transport, and an additional 7 percent for completing gonadal growth. In European silver eel, about 50 percent of the eels studied had a fat percentage of 20.

It is unknown if American eels require the same fat reserves as European eels because American eels travel a shorter distance to reach the Sargasso Sea than do European eels. Actual distances, routes, and depths of migration for adult eels are unknown. Distances traveled by migrating silver American eels likely vary from under 1,500 km to over 4,500 km, shorter than the 5,000 km to 7,000 km likely traveled by European eels. It is not known whether American eels follow the Deep Western Boundary Current or the upper portions of the ocean to return to the Sargasso Sea.

b) Fecundity.

Fecundity varies with size and increases exponentially with length, ranging from about 0.6 million to almost 30 million eggs depending on the size of the female. Fecundity is also linked to the habitat which the eel occupies. In an eel farm growth experiment, favorable nutrition was one of two factors identified as producing eels with a high reproductive capacity. This high fecundity is thought to compensate for larval mortality which is believed to be well in excess of 99 percent. Loss of fecundity related to the Saint Lawrence River stock is staggering (Casselman and Marcogliese 2007). That stock was all female, and as far as the upper Saint Lawrence River Lake Ontario was concerned, large, and highly fecund (Casselman 2006).

c) Spawning.

Spawning is believed to take place in the Sargasso Sea. Some biologists have hypothesized that there is some (as yet unidentified) feature of the surface water that serves as a cue for migrating adults to cease migration and begin spawning. While spawning has never been documented, the 2007 Final Determination assumes that adult eels die after spawning.

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5. Range

The range of the American eel includes all accessible river systems and coastal areas to which western North Atlantic Ocean oceanic currents provide transport. These drainages and coastal areas range from Northern Brazil/Venezuela to southern Greenland and include most Caribbean Islands and Bermuda.

Currently, the majority of the American eel population is located along the Atlantic seaboard of the United States and Canada. The historic distribution of the American eel within its extensive continental range is well documented along the United States and Canadian Atlantic coast and inland. The FWS reports that the distribution is less well documented in the Gulf of Mexico, Mississippi watershed, and Caribbean Islands, and least understood in Central and South America if indeed they exist there in any abundance at all. We were unable to document any populations of any size or any harvest data that would support a statement that there are American eels in any meaningful numbers. Further, the documentation in the Mississippi watershed confirms that eel populations have nearly disappeared. Some commercial catch data for Mexico, Cuba, and the Dominican Republic may exist[10].

D. Population Status

Eel populations throughout the world are declining catastrophically. The European eel fishery has collapsed and the European eel is likely to become extinct in the foreseeable future. As a result, there is additional harvest pressure on the American eel. However, American eel populations are and have been declining nearly as precipitously for two decades across its range in Canada and the United States. Recruitment of both the European eel and American eel have fallen to levels possibly as low as 1 percent of their highest levels.[11]

Juvenile recruitment to the Saint Lawrence River system and Lake Ontario virtually ceased during the 1990s, recruitment in the mid-to-late 2000s was 4 orders of magnitude smaller than that of the 1970s, and there is no information that demonstrates any reversal to this downward trend. In 1985, nearly a million juvenile eels migrated into the Saint Lawrence River; that number had fallen to levels approaching zero by 2000, and there is no data which demonstrates any reversal to this downward trend. Recruitment of the European eel and American eel have fallen to levels possibly as low as 1 percent of their highest levels.[12]

Ontario's Ministry of Natural Resources indicated that Ontario's commercial eel harvest peaked at more than 500,000 pounds in 1978 and had declined to 30,000 pounds in the first decade of the 21st century. Ontario officials blame the eel's plight on overharvesting, migration barriers, climate conditions and hydro-electric turbines. Studies on the St. Lawrence River hydro-electric

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[10] MacGregor 2008.
[11] Aoyama 2009.
[12] Aoyama 2009.

dams reveal cumulative mortality of 40 percent of emigrating silvering eels.[13] Ontario closed its fishery in 2004.

In 1974, the number of juvenile eels counted annually at the Conowingo Dam on the Susquehanna River was 126,543 and was nearly zero in by the turn of the 21st century. At the November 18, 2002 meeting of the ASMFC Eel Management Board, Mr. Richard Snyder, ASMFC representative for Pennsylvania, stated: "No American eels really pass the Conowingo Fish Lift, based on the annual samplings there lately."

U.S. harvests of American eels on the Atlantic Coast have declined 64 percent of the long-term average since 1950; almost 44 percent below the 20-year average; and about 30 percent below the five year average, based on 2002 harvest reports collected by the ASMFC, Geer (2004).[14]. Limburg and Waldman (2009) report an even steeper decline of 72.2 percent in recent decades.

On August 14, 2003, eel biologists from 18 countries meeting in Quebec, Canada, drafted and unanimously approved a declaration titled: *The Quebec Declaration of Concern: Worldwide Decline of Eels Necessitates Immediate Action.*

The Declaration states:

> *"The steep decline in populations of eels endangers the future of these legendary fish. With less than 1 percent of major juvenile resources remaining, precautionary efforts must be taken immediately to sustain these stocks. In recent decades, juvenile abundance has declined dramatically; by 99 percent for the European eel (Anguilla anguilla) and by 80 percent for the Japanese eel (Anguilla japonica). Recruitment of American eel (Anguilla rostrata) to Lake Ontario, near the species' northern limit, has virtually ceased.*
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> *"Eels, which depend on freshwater and estuarine habitats for their juvenile growth phase, anthropogenic impacts (e.g. pollution, habitat loss and migration barriers, fisheries) are considerable and may well have been instrumental in prompting these declines. Loss of eel resources will represent a loss of biodiversity but will also have considerable impact on socioeconomics of rural areas, where eel fishing still constitutes a cultural tradition. Research is underway to develop a comprehensive and effective restoration plan. This, however, will require time. The urgent concern is that the rate of decline necessitates swifter protective measures. As scientists in eel biology from 18 countries assembled at the International Eel Symposium 2003 organized in conjunction with the 2003 American Fisheries Society Annual Meeting in Quebec, Canada, we unanimously agree that we must raise an urgent alarm now. With less than 1 percent of juvenile resources remaining for major populations, time is running out. Precautionary*

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[13] Verreault et. al 2004.

[14] Geer, Patrick. Minutes of March 29, 2004 Atlantic State Marine Fisheries Commission meeting. Alexandria, Virginia.

> *action (e.g., curtailing exploitation, safeguarding migration routes and wetlands, improving access to lost habitats) can and must be taken immediately by all parties involved and, if necessary, independently of each other. Otherwise opportunities to protect these species and study their biology and the cause of their decline will fade along with the stocks."*

According to official minutes of the March 29, 2004 meeting of the American Eel Management Board of the ASMFC in Alexandria, Virginia, Mr. Patrick Geer, Technical Committee chairman of the American Eel Management Board, stated:

> *"You can see, basically, they've had very little or no recruitment for the last nearly ten years at this point [in the St. Lawrence River system]. Typically, when the eels get to this area on the St. Lawrence River, they're five to seven years old. They're noticing in the last few years they're getting much older than that, so they're speculating they're having a failure of recruitment to the St. Lawrence system."*
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> *U.S. landings on the Atlantic Coast are down about 64 percent of the long-term average back to 1950, almost 44 percent below the 20-year average and about 30 percent below the five year average. This is from 2002 landings reports."*

On November 10, 2004 a public information document for potential changes to the interstate fishery management plan was provided by the ASMFC and contains the following information:

Dr. John Casselman presented findings of a continued decline in the abundance of eel in the Saint Lawrence River. This decline in the northern portion of the population is of concern because this segment of the population consists mainly of large, fecund females which are believed to have a direct effect on recruitment of eels along the coast. Dr. Casselman has since noted that this phenomenon is occurring everywhere.[15]

In 2006, Canada nominated the American eel as a species of special concern based on declines in populations. A 2006 report on the American eel prepared by the Committee on the Status of Endangered Wildlife in Canada 2006 includes the following information:

- Indices of abundance in the upper St. Lawrence River and Lake Ontario have declined by 99 percent since the 1970s.[16]
- The only other data series of comparable length is from the lower Saint Lawrence River and Gulf of St. Lawrence, where four of five time series showed declines.

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[15] Dr. Casselman pers comm.

[16] Page vii. Changes in the data series for the eel were evaluated between years prior to 1980 and 2000-2005. The interval between these periods represents about 3 times the approximate generation time of female American eels. Percent change between early and recent ranged from -99.5-+74.8 percent. All four landing series and five of the six survey indices were negatives. The sole U.S. series was -67.5.

- These eels were a substantial portion of the breeding population of the species (estimates range from 59.2 percent to 48.8 percent of the spawn output[17]).
- The collapse of the Lake Ontario-Upper St. Lawrence components may have significantly affected total reproductive output.
- Positive trends in some indicators are too short to provide strong evidence that this component is increasing.
- Possible causes of the decline include: habitat alteration, dams, turbine mortality, harvest, ocean conditions, acid rain and contaminants.
- The report did not indicate that the species was data deficient.[18]
- The ASMFC noted that since the fishery's peak in the mid 1970s at 3.5 million pounds, commercial landings have declined significantly to a near record low of 868,215 pounds in 2001[19] , and the most recent data record a catch of even less, 714,723 in 2008. Recreational data concerning eel harvest appears to indicate a decline in abundance. According to the NMFS Marine Recreational Fisheries Statistics Survey, recreational harvest in 2001 was 10,805 eel, a significant decrease from the peak of 106,968 eel in 1982."[20]
- Environment Canada notes that annual landings of eels have declined from 1,000 tonnes per year in the 1930s and did not reach 100 tonnes in 2007, despite the fact that the value was $7.65/kg, making it one of the most valuable species in Canada's commercial freshwater fishery.[21]
- ASMFC notes that harvest and habitat loss are the primary causes of any decline in abundance of American eel. Harvest risk is based on the fact that eels are slow maturing, early life stages gather seasonally for migration and are thus more easily caught, yellow eel harvest accrues cumulative stress over multiple years, and all eel mortality is pre-spawning mortality.

A final report, issued January 2005, Estimation of Reproductive Capacity of the American Eel found:

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[17] Page vii.

[18] Page x.

[19] Addendum I To The Interstate Fishery Management Plan For American Eel Approved for Public Comment October 31, 2005; page 4.

[20] Addendum I To The Interstate Fishery Management Plan For American Eel Approved for Public Comment October 31, 2005; page 5.

21 The American Eel of the St. Lawrence: A Species In Decline for the Past 40 Years: http://www.ec.gc.ca/default.asp?lang=En&n=EEB1B2FF-1&printerversion=true.

- Study results strongly suggest that the larger eels arrive at the spawning site with much higher remaining fat reserves for their gonad production during final maturation, resulting in higher quality and quantity of offspring.
- Successful fertilization was obtained in the bigger eels and no difference between locations was observed.
- Larger and fatter eels have a much higher chance of success to reach the spawning site and when they reach it, they also have a higher percentage of energy stores left for reproduction. Obviously, those eels are the best future genitors for the eel population.

A draft addendum to The Interstate Fishery Management Plan for American Eel, ASMFC, dated October 31, 2005 contains the following information:

- Available data points to decreasing recruitment.
- Available data points to localized declines in abundance.

In 2007, the FWS determined that listing of the American Eel was not warranted. As part of their determination the FWS held two workshops regarding American Eel Great Lakes/Canada threats and Population Dynamics Workshop (FWS in cooperation with NMFS), (expert opinions paraphrased) draft minutes, Jan/Feb 2006:[22]

- Karin Limburg: felt it **could go extinct** on the North American continent;
- Guy Verreault felt it **could go extinct** regionally;
- Catherine Couillard felt there was a **good likelihood of local extinction**, and a spiraling effect could be created to plunge the species into further decline;
- John Casselman felt **local extinctions were already occurring;**
- Bob Graham did not feel extinction was likely but that **local or regional extinction could occur;**
- Ken Oliviera did not believe it could go extinct, but thought that there might be some threshold he does not know about that would change that opinion;
- Pete Hodson thought **local extinctions were a fact** and predicted the Lake Ontario and Saint Lawrence River declines;
- Len Machut felt there was a **good possibility of extinction;**

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[22] FWS. 2006. *Draft* Minutes from the American eel status review Workshop 2: Great Lakes/Canada threats and population dynamics. Buffalo, NY, January 31-February 2, 2006.

- Rob Macgregor felt **regional extinctions at the range extremities likely**, they were a grave warning, and we were on track for bigger problems;
- Paul Angermeier did not believe rangewide extinction was imminent but it was approaching some unknown level at which it would be threatened;
- Alastair Mathers felt **local extinctions were likely**, we needed to take action across the range, and the only way to address the decline was to take broader scale action;
- John Dettmers thought the likelihood of extinction throughout the range was low, but that range contraction and **local extinction was high**. He felt we could not rule out the possibility of extinction;
- Joe Hightower felt the possibility of extinction was low; however, he noted particular data indications over time that would change his opinion.

Nevertheless, the FWS determined that listing of the eel for protection under the ESA was "not warranted."

E. Petition to List

As a result of the data indicating continued population decline, CESAR hereby presents a second petition to the FWS to list the American eel under the ESA. The American eel is currently threatened with extinction due to the present or threatened destruction, modification, or curtailment of its habitat or range; overutilization for commercial and recreational purposes, disease and possibly predation, the inadequacy of existing regulatory mechanisms, as well as global warming, and anthropogenic factors related to generation of hydroelectric power and the spread of swim bladder parasites from ship ballast water.

Following is a detailed and specific recitation of the threats to the species.

II. CRITERIA FOR ENDANGERED SPECIES ACT LISTING

FWS and NMFS are required to determine, based solely on the basis of the best scientific and commercial data available, whether a species is endangered or threatened because of any of the following factors: (1) the present or threatened destruction, modification, or curtailment of its habitat or range; (2) overutilization for commercial, recreational, scientific or educational purposes; (3) disease or predation; (4) the inadequacy of existing regulatory mechanisms; or (5) other natural or manmade factors affecting its continued existence. 16 U.S.C. §§1533(a)(1) and 1533(b).

Petitioners provide evidence below showing that all of these factors are acting in concert to cause the precipitous decline of American eel in the United States of America, thus warranting the species' protection under the ESA.

A. The Present Or Threatened Destruction, Modification Or Curtailment Of The Species' Habitat And Range

1. Loss Of Habitat Or Range

In their 2007 Final Determination the FWS estimated that the available coastline (including barrier islands) from Maine to Texas (Atlantic and Gulf coasts) is 29,612 km and noted that this extensive range should provide the American eel with a buffer against adverse conditions. However, significant anthropogenic changes within the range have reduced the accessible habitat by percentages perilously close to 100 percent in some places. Access to the Atlantic coastal tributaries has been lost or restricted by 84 percent. Habitat loss is greatest from Maine to Connecticut at 91 percent. States from New York to Virginia have seen stream habitat reduced by 88 percent, and from North Carolina to Florida stream habitat has been reduced by 77 percent.[23] Drainages in the Gulf of Mexico excluding the Mississippi River Basin have seen significant habitat and population reductions.[24] Eels are still found in limited areas in Alabama and Mississippi, however, eels native to much of Texas have been eliminated from most central and western areas of the state. Statewide eel populations have declined drastically on virtually all of Arkansas's major rivers. Also, Eels have been eliminated from large areas of the Missouri Ozarks. Eels have been eliminated from Southern Kansas and only occur sometimes in streams in northeastern and central Kansas.

These reductions in habitat and their causes can have cascading adverse effects on eel populations.

Female American eels spend most of their lives in freshwater habitat along the Atlantic seaboard prior to returning to the Sargasso Sea to spawn. Safe and efficient access to and from their freshwater habitat is essential to the survival of the American eel. Coastal river systems along the Atlantic seaboard are the sole migratory pathways for female American eels to gain access to their required freshwater habitat. While it is possible that some eels spend their entire life cycle in salt water, oceanic research indicates such behavior is rare and virtually nonexistent; catch data from commercial trawling confirms empirically that this is rare. Certainly the marine component is small and at best an unknown and unquantified life strategy which provides little foundation for reliance on it as a basis for sustaining the American eel production.

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[23] Atlantic States Marine Fisheries Commission Atlantic Coast Diadromous Fish Habitat: A Review of Utilization, Threats, Recommendations for Conservation, and Research Needs; Habitat Management Series #9 January 2009; Chapter 10.

[24] NatureServe. 2004. Downloadable animal datasets. NatureServe Central Databases. Available from: www.natureserve.org/getData/dataSets/watershedHucs/index.jsp ([access date:3/19/10] The U.S. Geological Survery and National Park Service are partners with Natureserve.

The ASFMC states:

> *"By region, the potential habitat loss [for American eels] is greatest (91 percent) in the North Atlantic region (Maine to Connecticut) where stream access is estimated to have been reduced from 111,482 kilometers to 10,349 kilometers of stream length. Stream habitat in the Mid Atlantic region (New York through Virginia) is estimated to have been reduced from 199,312 km to 24,534 km of unobstructed stream length (88 percent loss). The stream habitat in the South Atlantic region (North Carolina to Florida) is estimated to have decreased from 246,007 km to 55,872 km of unobstructed stream access, a 77 percent loss".*
>
> *Of 15,570 dams blocking American eel habitat*[25] *in the United States, Busch et al. (1998) reported that 1,100 of these dams are used for hydro-electric power. Virtually none of these 1,100 hydro-electric dams provide, or are required to provide, safe and efficient upstream and downstream passage for American eels to utilize their historic freshwater habitat. Virtually none of the 14,470 non-hydroelectric dams reported by Busch et al. (1998) provide, or are required to provide, safe and efficient upstream and downstream passage for American eels to utilize their historic freshwater habitat.*

The Maryland Department of Natural Resources, MBSS Newsletter March 1999, Volume 6, Number 1 states:

> *"The most dramatic example of the decline of American eel abundance is dam construction on the Susquehanna River. Prior to the completion of Conowingo and three other mainstem dams in the 1920's, eels were common throughout the Susquehanna basin and were popular with anglers. To estimate the number of eels lost as a result of construction of Conowingo Dam, we used MBSS data on American eels from the Lower Susquehanna basin and extrapolated it to the rest of the basin above the dam. Our best conservative guess is that there are on the order of 11 million fewer eels in the Susquehanna basin today than in the 1920s.*
>
> *"The magnitude of this loss is corroborated by the decline in the eel weir fishery in the Pennsylvania portion of the Susquehanna River. Before the mainstem dams were constructed, the annual harvest of eels in the river was nearly 1 million pounds. Since then, the annual harvest has been zero. Given the longevity of eels in streams (up to 20 years or more) and their large size, the loss of this species from streams above Conowingo Dam represents a significant ecosystem-level impact. Because adult eels migrate to the Sargasso Sea to spawn and die -- transporting their accumulated biomass*

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[25] Some argue that this is a conservative estimate. K. Limburg pers comm.

> *and nutrient load out of Chesapeake Bay -- the loss of eels has increased nutrient loads in the basin and reduced them in the open ocean where they are more appreciated."*

The number of juvenile eels counted annually at the Conowingo Dam on the Susquehanna River has declined from a peak of 126,543 in 1974 to nearly zero in recent years (ASMFC 2000). At the November 18, 2002 meeting of the ASMFC Eel Management Board, Mr. Richard Snyder, ASMFC representative for Pennsylvania, stated: "No American eels really pass the Conowingo Fish Lift, based on the annual samplings there lately."

Dohne (2004) states:

> *"As for elvers, the local evidence is equally thin but just as bleak. At York Haven's dam -- whose fish ladder is the only one on the lower Susquehanna to specifically monitor eel traffic -- no elvers appeared during this spring's shad run (April through mid-June)."*

A recently published work by the American Fisheries Society 'Eels at the Edge' (2009) is a compilation of papers addressing the plight of the American eel.[26] Papers included in the volume address overfishing in the Chesapeake Bay. The papers document that this area, once considered one of the biggest eel producers along the Atlantic Seaboard, may now depend on recruitment from areas where fishing mortality is low.

Dams limit the amount of habitat available to eels and disproportionately eliminate larger more fecund females prior to spawning. This particular type of habitat reduction and limitation logically leads to reduced eel productivity and abundance.

NatureServe catalogues the incidence and effects of dams in specific geographic regions of the United States:[27]

Geographic Area	
Atlantic coastal streams from Maine to Florida	15,115 dams that can hinder or prevent upstream and downstream fish movement. Loss of access to 84 percent of the stream habitat. A reduction from 556,801 kilometers to 90,755 kilometers of stream habitat available. Only 7 percent of these dams are covered by regulatory programs that could provide fish passage.

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[26] Eels at the edge : science, status, and conservation concerns edited by John M. Casselman and David K. Cairns. Published Bethesda, Md. : American Fisheries Society, 2009.

[27] NatureServe. 2004. Downloadable animal datasets. NatureServe Central Databases. Available from: www.natureserve.org/getData/dataSets/watershedHucs/index.jsp {[access date:3/19/10]; The U.S. Geological Survery and National Park Service are partners with Natureserve.

Eastern Canada	Obstruction by hydroelectric dams may contribute to reduced eel abundance (Jessop 2000). Silver eel declines in the St. Lawrence River basin may be due to escapement reductions from upper St. Lawrence dams and water flow control, rather than fisheries (Richkus and Whalen 1999). In Canadian lakes, Smith and Saunders (1955) found smaller standing stocks of eels in lakes that were farther from the ocean and that had obstructions such as dams, falls, and lakes.
New Brunswick	Impoundment of the upper estuary of the Petitcodiac River resulted in reduced abundance of American eels (Locke et al. 2003).
New York	The U.S. Geological Survey examined historic records and literature and compared the information with recent fisheries surveys contained in the Statewide Fisheries Database and other sources. Results indicate dramatically reduced numbers of eels statewide. Eel are thought to be extirpated from the New York portions of the Susquehanna watershed.[28]
Maine	Within a year after the removal of Edwards Dam on the Kennebec River, large numbers of American eels were observed in upstream habitats that had been inaccessible for more than 150 years (O'Donnell et al. 2001).
Rhode Island	In Rhode Island, eels were commonly collected throughout the state but were not well represented in the upper reaches of the Blackstone and Pawtuxet River watersheds, undoubtedly due to the many dams that impede upstream migration (Libby 2004).
Connecticut	Eel densities are much lower in headwater regions of streams that have many, or high, dams or falls. Movement upstream appears to be affected by both the number and height of obstructions (Levesque and Whitworth 1987, Whitworth 1996, Hagstrom et al. 1996).
Pennsylvania	American eel passage has been blocked for many years on the Susquehanna River by four large hydroelectric projects on the lower river (Conowingo, Holtwood, Safe Harbor, and York Haven). Fish passage facilities designed for American shad were installed in each of these facilities within the past 15 years, but eel passage may be limited. No eels were passed at

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[28] State of New York, Department of Environmental Conservation, Bureau of Fisheries 2008-2009 Annual Report.

	these fishways in 2005. In the Delaware River basin, elvers use fishways at the Easton, Chain, and Hamilton Street dams, but quantification of eel passage is not possible. Cementon Dam, upstream of the Hamilton Street Dam, lacks a fishway, but at least some elvers successfully pass this dam. The Schuylkill River, a major tributary of the Delaware River in Pennsylvania, has nine dams, some of which have fish passage facilities, are breached or partially breached, or are scheduled for fishway installation within the next few years. A dam upstream of the Felix Dam was exposed when the Felix Dam was breached and is currently an impediment to fish passage. There are no plans to remove the two uppermost dams (New Kernsville and Auburn) on the Schuylkill River. Some eels pass the dams downstream of New Kernsville, but the efficiency of passage is unknown.
Maryland	More than 1,000 human-made barriers to migratory fish (Leasner, DNR, pers. comm.) reduce access of American eels and other fishes to their historical habitats. Stream survey data suggest that mainstem dams have been a major factor in this decline by blocking the upstream migration of juvenile eels.
South Carolina	Populations of diadromous fishes (eels are diadromous) in the Santee-Cooper Basin are significantly depressed relative to historical levels, primarily as a result of the more than 50 dams in the basin.
Mississippi	Upstream movement of eels could be impeded by dams.
Alabama	Dams on major rivers impede eel progress to far upstream reaches.
Kansas	Much formerly occupied habitat is now inaccessible as a result of dams and flow diversions.
Iowa	Construction of impassable flood control dams on the Des Moines, Iowa, and Chariton Rivers undoubtedly has restricted the migration of eels in these drainages.

2. Overutilization For Commercial, Recreational, Scientific Or Educational Purposes

a) Commercial

American eels are commercially harvested at all of their life stages except the larval state. They are harvested in all of their habitats as well; freshwater lakes and rivers, estuaries, the Atlantic Ocean, the Gulf of Mexico and the Caribbean Sea. Commercial American eel fisheries are found from Maine to the Gulf of Mexico as well as inland in every state of their range excepting Alabama and Mississippi. Eels have been largely extirpated from the Mississippi Basin although they were once found in substantial numbers throughout the drainage.[29] Commercial fisheries for glass eels, elvers, silver and yellow exist in Asia and Europe, Mexico, Cuba and the Dominican Republic. Eels are variously harvested for aquaculture, bait and food.

It is undisputed that overutilization of American eel is now occurring across the species' range in the United States of America. ASMFC (2000) states: "Harvest pressure and habitat loss are listed as the primary causes of decline in abundance of American eel (Castonguay et al. 1994a and 1994b). Several factors contribute to the risk that heavy harvest may adversely affect eel populations: (1) American eels mature slowly, requiring 7 to 30+ years to attain sexual maturity; (2) glass eel aggregate seasonally to migrate; (3) yellow eel harvest is cumulative stress, over multiple years, on the same year class; and (4) all eel mortality is pre-spawning mortality.

ASMFC (2000) further states: "Since the fishery's peak in the mid 1970s at 3.5 million pounds, commercial landings have declined significantly to a near record low of 868,215 pounds in 2001. Recreational data concerning eel harvest appears to indicate a decline in abundance. According to the NMFS Marine Recreational Fisheries Statistics Survey, recreational harvest in 2001 was 10,805 eel, a significant decrease from the peak of 106,968 eel in 1982."

Geer (2004) states: "U.S. landings on the Atlantic Coast are down about 64 percent of the long-term average back to 1950, almost 44 percent below the 20-year average and about 30 percent below the five year average. This is based on 2002 landings reports."

In its 2008 Addendum II the ASMFC chronicles the continued decline of the American Eel.[30] The Commission notes that American eels continue to support both recreational and important commercial fisheries throughout their range and fisheries are executed in rivers, estuaries, and the ocean.

The addendum states that commercial glass eel harvest is legal in Maine and South Carolina, although reported landings are minimal in South Carolina. Yellow and silver eel fisheries exist in all states and jurisdictions with the exception of Pennsylvania and the District of Columbia. South Carolina and Georgia recorded no commercial yellow or silver eel landings in 2007.

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[29] MacGregor et.al. 2008.

[30] Atlantic States Marine Fisheries Commission ADDENDUM II TO THE FISHERY MANAGEMENT PLAN FOR AMERICAN EEL Approved October 23, 2008.

The addendum also records a decline in commercial landings from a high of 1.8 million pounds in 1985 to a low of 641,000 pounds in 2002. Landings of yellow and silver eel in 2007 totaled 834,500 pounds.[31] New Jersey and Delaware and Maryland account for 73 percent of the coast-wide commercial landings. Each state reported landings of over 100,000 pounds of eel and Maryland reported landings of over 300,000 pounds in 2007.

Massachusetts, Pennsylvania, Georgia, Florida, and the District of Columbia were granted *de minimis* status for the 2007 commercial fishing year. *De minimis* is approved if a member states' commercial landings of yellow and silver eel for the previous year are less than 1 percent of the coast-wide landings for the same year.

Records of the ASMFC show the Commission has failed to undertake protective measures for the remaining American eels living along the Atlantic seaboard of the United States; nor has the ASMFC taken any action to restrict or prohibit the ongoing harvest of American eels along the Atlantic Seaboard during the past five years as they wait to confirm these already obvious declines in abundance.

b) Recreational

The addendum notes that few recreational anglers directly target eels and most landings are incidental when anglers are fishing for other species. There is a commercial fishery for human consumption of eels, but there is also a commercial fishery for eels that are used and sold as bait for larger sport fish such as striped bass. Finally, some recreational fishermen may catch their own eels to utilize as bait.

This petition presents new information, not available at the time of the 2007 Final Determination which documents continued widespread population decline, and little information that shows any reverse in this decline. We believe this new information is substantial and coupled with the information already before the FWS, warrants a 12-month status review.

3. Disease Or Predation

a) Threats To The American Eel From The Swim Bladder Parasite: *Anguillicola Crassus*

The spread of the swim bladder parasite Anguillicola crassus to American and European eels can be attributed to an expanding eel trade between East Asian countries, Europe, and the U.S. in the 1980s, as well as eel aquaculture (Kirk 2003). Similar to other epizootics that have had severe consequences for new host species, American and European eels were immunologically naïve, allowing this parasite to quickly colonize populations of new hosts.

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[31] Harvest data for 2007 comes from the 2008 State Compliance Reports. The landings are preliminary and some are incomplete.

It is recognized that several factors contributed to the decline of the American eel, including: dams, hydro-electric turbines, habitat modification, commercial and recreational fishing, and oceanic changes (Castonguay et al. 1994, Haro et al. 2000, Casselman 2003, Friedland et al. 2007). However, the recent introduction of *A. crassus* into North American eel populations is a threat that was not fully appreciated until the last several years. In its 2007 final listing determination the FWS noted that:

> *"We remain cautious in extrapolation of these preliminary laboratory studies with regard to rangewide implications given the absence of evidence for population-level effects, such as reduced recruitment of glass eels (which would be an indicator of decreased outmigration survival). This being said, we acknowledge the statement by the International Council for the Exploration of the Sea (ICES 2001, p. 6) that due to the fairly recent invasion of the U.S. by A. crassus and the long-lived nature of at least a portion of the American eel population, the impact of A. crassus on American eel may not yet have been fully realized."*

The following paragraphs provide additional information that was not considered by the FWS in its 2007 Final Determination.

b) The threat of *A. crassus* is spreading

A study by Aieta and Oliveira (2009) sampled yellow phase American eels from 38 locations, ranging from Rhode Island's Pawcatuck River in the south, to the St. Lawrence River and Newfoundland in the north, in the years 2005-2007. The swim bladder parasite A. crassus was found in all locations within New England, with infection rates of 7 to 76 percent per location. Locations in New Brunswick and northern Nova Scotia which had infected eels had rates of infection ranging from 3 to 30 percent. No infected eels were found in Southern Nova Scotia or the St. Lawrence River. The authors therefore reported that there was no significant correlation of the parasite with latitude. This suggests that there may be more than one transport mechanism for the parasites. The authors also concluded that it was only a matter of time before the parasite reached the St. Lawrence River system. A similar study (Rockwell et al. 2009) found additional infestations on Cape Breton Island, Nova Scotia; more than half of the eels sampled in the Mira River were infested with the parasite.

c) The extent of damage to American eels caused by *A. crassus* parasitism was previously unrealized

It was not until 2006 that the results of the first pathogenesis study of A. crassus infections in wild American eels was reported (Sokolowski and Dove 2006). That study documented the seriousness of Anguillicola crassus infestations in American eels:

> *"All of the examined American eels showed gross signs of previous or ongoing infections with A. crassus; the normally translucent swim bladder appeared opaque and blood vessels were dilated. When present, A. crassus worms were visible in the lumen of the*

> *swim bladder; in one case, A. crassus eggs and L2 larvae were associated with a blood-filled swim bladder. Histologically, the swim bladders of infected eels showed focal, multifocal, and diffuse changes, including abnormal papillose appearance of the mucosa; hyperplasia of the lamina propria, muscularis mucosa, and submucosa; edema of the mucosa and muscularis mucosa; dilation of the blood vessels in the lamina propria; L2 larvae in the lumen of the swim bladder (Figure 4); and damage in the submucosa attributable to migrating L3 and L4 larvae (Figures 2, 3). Pathologies included fibrosis or lymphocytic infiltration (or both) around L3 and L4 larvae in the submucosa (Figure 3); destruction of the mucosa, which in some cases completely exposed the mucosal blood vessels (Figure 4); L2 larval penetration of the tissues of the swim bladder and bacterial infections in the submucosa (Figure 5) and muscularis mucosa (Figure 6); and migration of an L4 larva through the rete mirabile (Figure 7). In one case, the submucosa was infiltrated with an adult worm and L2 larvae. In two American eels, a total of three A. crassus L3 larvae were found free in the intestine."*

In a 2004 letter to the journal *Science*, Sures and Knopf (2004) raised the worldwide alarm that *A. crassus* is an unappreciated threat to eels on both sides of the Atlantic ocean. This is due to the fact that they inhibit the ability of eels to reach spawning grounds in the Sargasso Sea:

> *"The parasites migrate into the swim bladder and suck the eel's blood. Pathological effects include thickening, inflammation, fibrosis, and changes in the epithelial cells of the swim bladder wall (6), as well as alterations of the gas secretion into the swim bladder (7). In severe cases, pathologic alterations even lead to a complete loss of the swim bladder lumen, or the lumen becomes totally filled with worms (see figure). From these massive alterations of the swim bladder, one may expect a loss of its function. Although eels are benthic when living in freshwater habitats, a functional hydrostatic organ is essential for their spawning migration through the Atlantic, where eels perform diurnal vertical migrations ranging between 40 to 600 m (8). Thus, eels with a damaged swim bladder are unlikely to reach their spawning grounds in the Sargasso Sea. "It is clear that A. crassus was not the initial reason for this negative trend, because the decline of European eel fry began before the parasites' appearance in Europe. However, A. crassus may now be contributing to the rapidly decreasing numbers of the Atlantic freshwater eels and may be a crucial factor among an array of threats."*

Clearly, the cumulative effects of *A. crassus* parasitism in combination with injury from hydroelectric turbines, accumulation of contaminants, and low fat stores, further lowers the ability of American eels to successfully reach their spawning grounds and reproduce. When coupled with the take of reproductive aged silver eels by commercial fisheries and recreational fisheries (who frequently waste this resource by using them for fishing bait), and mortality when passing hydroelectric dams (>50 percent mortality per dam[32]), it is reasonable to conclude that fewer American eels are reaching their spawning grounds in the Sargasso Sea than ever before.

d) Experimental evidence shows that parasitism by *A. crassus* makes eels more vulnerable to mortality from hypoxia

Experiments conducted by Gollock et al. (2005) tested the hypotheses that eels infected with *A. crassus* would be vulnerable to mortality when exposed to hypoxic conditions, and that an increased oxygen demand due to parasitism would alter their physiological response to hypoxic stress. Their results were consistent with the hypotheses: parasitism by *A. crassus* exacerbated the corticosteroid stress response associated with exposure to severe hypoxia. Their hypothesis was also consistent with field observations of eels parasitized by *A. crassus* in lakes in eastern Europe. The eels suffered high mortalities while living in alternating conditions of nighttime hypoxia and higher than normal daytime water temperatures (28°C). These experimental results and field observations led the authors to conclude that parasitism by *A. crassus* increases an eel's *"exposure to the compounding effects of sequential periods of high temperatures during the day and hypoxia during darkness [when photosynthesis from phytoplankton ceases] and will be more stressful and could ultimately result in mortalities." Thus, eels infected with A. crassus fare poorly in hypoxic conditions and appear to suffer high mortalities* when additional stressors are added. Such experiments have not been conducted on American eels; however, because American and European eels are closely related and have similar life histories, the same cause and effect mechanisms can be expected to affect their physiology.

e) Experimental evidence shows that parasitism by *A. crassus* can lead to failure of migrating eels to reach their spawning area, and compromise the reproduction of those eels that do reach their spawning area

The hypothesis that eels burdened with *A. crassus* infections could be experiencing migratory failure was experimentally tested by Palstra et al. (2007a). These authors used experimental data to test the hypotheses that: 1) parasitic sanguivorous *activities* – related to parasite weight – reduce swimming endurance; and 2) mechanical damage of the swim bladder impairs buoyancy control. The experiment consisted of placing eighty silver eels suffering various degrees of infection into swim-tunnels designed to simulate long-distance migration and measuring their swimming capacity, performance, and physiological parameters. The authors reported:

> *"Infected eels had lower cruising speeds and a higher cost of transport. Eels without parasites, but with a damaged swim-bladder, showed similar effects. Almost half of the eels that contained damaged swim-bladders (43 percent)*

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[32] On the downstream journey, eels may have to pass through turbines at hydroelectric dams. Mortality may be 50% or more for some types of turbines, with 80-100% being injured (Haro et al. 2003).

> *stopped swimming at low aerobic swimming speeds (<0.7 m/s). Simulated migration trials in a recent related study* [(Palstra et al. 2007b)] *have confirmed that eels with a high parasite level or damaged swim-bladder show early migration failure (<1000-km)."*

In other words, the eels with high parasite levels or damaged swim bladders simply do not have the endurance or ability to complete their migration. In the wild, eels with heavy parasite loads or damaged swim bladders will spend more energy on migration, leaving less fat for egg or sperm production.

It was previously recognized that eels migrated at varying depths (Tesch 1999). The importance of swim bladder function to migrating and spawning eels, however, was underscored by the first live captures of pre-spawning eels at spawning grounds by Chow et al. (2009) These authors captured giant mottled eels (*A. marmorata*) and two Japanese eels (*A. japonica*) approximately 130 km south of the Suruga Seamount using trawl nets at depths >230m. Without a functioning swim bladder, such as those damaged by *A. crassus*, eels cannot make vertical migrations into or out of such depths.

An additional discovery from these eels - that is relevant to understanding the energetic requirements for reproduction of American eels - is the tremendous energetic investment that eels make in reproductive tissues. The gonad-somatic index (relative weight of gonad to body weight) in one of the *A. japonica* eels captured by Chow et al (2009) was 18.8 percent, and an *A. marmorata* eel specimen had a gonad-somatic index of 13.4 percent. These are much higher percentages than those typically found in wild male silver-stage eels from coastal areas (~1.0 percent). These observations show that there is a tremendous investment in reproductive tissues prior to spawning and that fat reserves are essential for high fecundity.

In their paper titled *"Decreasing eel stocks: survival of the fattest?"* Belpaire et al. (2009) used data on fat reserves in eels (from wild-caught eels) to model (based on previous data from swim tank endurance tests) the conditions under which migration and spawning could be completed by European eels. They reported that eels from Belgium and the Netherlands sampled from 1975 to 2005 had reduced fat content (approximately one third less), and thus had a reduced chance of completing their migration *and* having adequate reserves available for spawning. Although American eels have approximately half the distance to migrate to the Sargasso Sea than their European counterparts, similar energetic constraints could prevent them from reaching their spawning grounds or from being in strong enough condition upon arrival to spawn.

f) The cumulative effects of *A. crassus* parasitism and other factors will lead to such lost density in spawning areas that eels will be unable to reproduce

The probability of American eels finding a mate under conditions of low recruitment, increased migratory failure, and a skewing of the sex ratio towards males (discussed elsewhere) leads inevitably to the conclusion that eels swimming in the vast Sargasso Sea will have increasing difficulty finding mates. This "Allee effect" will further drive down productivity and edge the species closer to extinction (Allee 1931). This same mechanism has been recently invoked in the extinction risk of the polar bear (Molnár et al. 2008).

4. Inadequacy Of Existing Regulatory Mechanisms

In the document, Status of Fishery Resources off the Northeastern U.S. NEFSC - Resource Evaluation and Assessment Division Dec 2006, the authors state: *"A preliminary analysis of the suite of indices indicates a strong downward trend in abundance (ASMFC 2006a and 2006b); however an analytical assessment of Atlantic coast eel stocks has not been completed."*[33] Three years later, such a stock assessment still has not been completed.

Similarly, the U.S. Fish and Wildlife Service Strategic Plan Fiscal Year 2007 to 2011 Region 5 - only suggests limited actions such as a *"focus on restoration of diadromous fish passage through dam removal, or installation of fish passage structures."* Little is offered in the way of a detailed plan, proposed budget, or biological means of measuring the effectiveness of results relative to increasing eel numbers.

NMFS gives scant mention to the American eel in its recent and influential policy document: *Our Living Oceans: Habitat. Status of the Habitat of U.S. Living Marine Resources. Policymakers' Summary* (NMFS 2009). Below is all the mention of eels in this document, with no specific mention of the American eel, the only catadromous fish in the United States:

> ***Habitat Impacts***
>
> *"Dams fragment river habitats and present impediments to catadromous fishes such as eels (which spawn in the ocean and grow to maturity in fresh water) and anadromous fishes such as salmon, sturgeon, striped bass, shad, and river herring (Roni, 2005; NMFS, 2008). Dams also change upstream habitat by creating reservoirs that slow water velocities and alter temperatures. Reduced freshwater flows resulting from water removals for domestic and commercial use can affect river and downstream estuarine habitats as well. Altering natural flows and the processes associated with flow rates (such as nutrient and sediment transport) impact in-stream habitats, shoreline riparian habitats, and prey bases. Water quality may also be reduced from water withdrawals: temperature, salinity, and concentrations of toxic chemicals all increase as water volumes shrink; dissolved oxygen decreases; and pathogens may be introduced.*
>
> *Changes to stream beds and banks and streamside vegetation can have major impacts on adjacent aquatic habitats. Hydrologic characteristics such as temperature and*

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[33] http://www.nefsc.noaa.gov/sos/spsyn/op/eel/index.html.

> *dissolved oxygen can be altered, and habitat complexity can be reduced by lowering the availability of large woody debris. Changing flow and channel structure, increasing stream bank instability and erosion, and altering nutrient and prey sources also degrade riverine habitat.*
>
> ***Impacts To Living Marine Resources***
>
> *By blocking upstream access for migrating species, dams greatly reduce the amount of habitat available for spawning, feeding, growth, and migration. Adequate freshwater flow is critical to all life stages, from eggs to spawning adults, so reduced flow can have a negative effect on anadromous and catadromous fish populations. As an example, a drought extending from 2001 through 2005 in the Klamath River basin of California and Oregon, combined with above-average withdrawals for agricultural use during the drought, allowed for the proliferation of endemic diseases, causing large fish kills. As a result, the Klamath River Chinook salmon stock fell below conservation objectives. This triggered the declaration of a commercial fishery failure by the Secretary of Commerce in 2006, which authorized a total of $60.4 million for distribution to eligible participants in the West Coast salmon fishery (DOC, 2006)."*

There has been little in the way of systematic effort to alleviate the threat of dams to eels despite the documented benefits of dam removal to eels and other native fish (Conyngham et al. 2006). An additional problem is that the actual number of structures is greatly under-reported.

a) Regulation Promulgated Through The Atlantic States Marine Fisheries Commission Has Failed To Protect The American Eel From Decline

The "mission" of the 16 state ASMFC is: *"To promote the better utilization of the fisheries, marine, shell and anadromous, of the Atlantic seaboard by the development of a joint program for the promotion and protection of such fisheries, and by the prevention of physical waste of the fisheries from any cause."* And while the stated "vision" of the ASMFC is to promote: *"Healthy, self-sustaining populations for all Atlantic coast fish species or successful restoration well in progress by the year 2015,"* this agency has a consumptive mission and has been in constant denial of the indicators of decline. For example, it is well known that freshwater habitat for the American eel has been adversely modified in many ways by human actions, yet the ASMFC (2009) persists in its optimism. In their summary on Present Conditions of Habitat and Habitat Areas of Particular Concern for American Eel they state: *"Much of American eel habitat has not been quantified."* And, *"Fortunately, American eel are habitat generalists, and therefore may be somewhat resilient to impacts on habitat availability."*

ASMFC's regulation and reporting on eel landings are also clearly inadequate to regulate eel harvest. The Monterey Bay Aquarium's program, "Seafood Watch" reported in 2007 (Halpin 2007) that *"Export data from the U.S. underscore the unreliability of capture data for eels. As* *mentioned above it has been estimated that reported landings of European eels within the EU are 50 percent of true landings. It appears that a similar problem occurs within the US, as exports of eels greatly exceed domestic landings. Looking only at fresh and frozen eels, exports have exceeded reported landings by as much as 2,760 mt (949 percent) (Figure 16).The lumping of multiple eel and eel-like species is not enough to explain discrepancy."* Seafood Watch urges consumers to avoid eel consumption.

The stock status of American eels is currently classified as *"Unknown"* and a contributing factor to this apparent lack of information is the fact that the 2005 ASMFC Stock Assessment *failed peer review*. However, the Peer Review Panel on that report concurred that yellow eel abundance was at an historic low. The fact that the ASMFC can state that the status of the stock is unknown in the face of declines of up to 4 orders of magnitude in a panmictic population is incredible. The next benchmark assessment has been scheduled for late 2010, although there are numerous indicators that eel abundance is declining and that threats to the American eel are not abating.

The ASMFC has done little over the past decade effectively to reverse the declines in eel recruitment, halt commercial and commercial take of American eels for recreational use as bait, or implement consistent methods to accurately assess their population size (ASMFC 2008; Taylor et al. 2008). The data reporting in the U.S. contrasts sharply with that in Canada (Cairns et al. 2008). In June of 2008, Table C of the NMFS - Status of U.S. Fisheries, listed the American eel as within the Jurisdiction of the ASMFC and summarized stock status as "unknown" for the following categories: 1) "Overfishing? (Is Fishing Mortality above Threshold?), 2); "Overfished? (Is Biomass below Threshold?)"; and 3) "Approaching Overfished Condition?" The Revised September 2009 ASMFC Stock Status Overview also lists the species status and trends as "unknown" and their stock status and rebuilding schedule as *"No rebuilding schedule."* It is unconscionable that a resource agency charged with regulatory responsibility for a resource could exist in near total ignorance of the status of that resource for *decades*. And yet, when faced with quantitative evidence that the species is in decline, this agency continues to claim that its status is "unknown" and continues to oversee the harvest of silver eels that are necessary to replenish the stocks (ASMFC 2008). This is why it is imperative that the inadequate regulatory authority of the ASMFC yield to the primacy of federal law under the ESA to prevent this species from becoming endangered in the reasonably foreseeable future.

b) International Experts On The American Eel Concur On The Inadequacy Of Current Regulatory Mechanisms To Protect This Species

The conclusions of a bi-national team of experts on the American eel have further detailed the inadequacy of current regulatory mechanisms in the United States and Canada to halt the decline (MacGregor et al. 2008):

> *"Management actions aimed at protecting American eel and European eel have largely been unsuccessful in halting declines and rebuilding stocks as, until*

> *recently, actions have not been well coordinated at the appropriate scale in recognition of the unique life cycle of these species."*

The primary reasons for failure of the FWS to list the eel as threatened or endangered under the ESA were also noted by MacGregor et al. (2008):

> *"In the meantime, the FWS has determined that listing American eel as threatened or endangered is not warranted (Federal Register 2007). The finding was constrained by the need to demonstrate that American eel is in danger of extirpation within a significant component of its range, or likely to become an endangered species within the foreseeable future. Because of a lack of scientific information relating to population-level status of American eel and the best genetic information that the species is panmictic, the FWS concluded that range-wide persistence of American eel was not in doubt. The finding appeared to rely heavily on information suggesting that some American eel complete their life cycle in marine environments and on two short-term data series relating to glass eel abundance. The finding placed less emphasis on longer-term data series that illustrated declining trends in abundance of yellow and silver American eel. Unlike the Species at Risk Act in Canada, the Endangered Species Act does not provide for designation and protection measure for species of special concern.*
>
> *It is not within the scope of this paper to comment substantially on the official designation of American eel under species at risk legislation; however, we are compelled to comment that (1) numerous data series suggest American eel is in decline in significant components of its range, (2) substantial habitat has been lost, (3) numerous and significant sources of anthropogenic mortalities exist for American eel, and (4) American eel is semelparous with late onset of maturity, particularly for the northern, more fecund segment of the population. Numerous threats have been identified, and their cumulative effects were not addressed in detail within the 12-month finding, but they apparently have been substantial. The precipitous (99%) loss of recruitment to Lake Ontario and the Susquehanna River, the major declines in silver American eel landings in Québec fisheries, the fact that yellow American eel are at or near historic lows within the ASMFC jurisdictions, and the 50% decline in the Chesapeake Bay VIMS index all point to significant cause for concern, regardless of designation as a species at risk. The lack of designation under the Endangered Species Act should not be perceived as a reason for inaction. Waiting to take appropriate action until a species is threatened with extinction is not in the best interests of agencies, ecosystems, or stakeholders. Strong, coordinated management actions are required to reverse the decline in American eel, actions that include habitat as well as fisheries management. Managers must also be mindful of the parallels between the experiences managing European eel and those of American eel. We certainly do not wish to be faced with the even more dire circumstances of European eel (ICES 2006; Dekker 2008, this volume)."*

Despite this call to action, regulatory mechanisms remain wholly inadequate to reverse the decline of the American eel.

Contrary to the positive forecasts promoted by the FWS 2007 Final Determination in their single-author, qualitative "analysis," the risk of endangerment and extinction facing the American eel and other eel species is real and has been clearly recognized by the top scientists in the eel research community for years. In 2003, the *Québec Declaration of Concern* was signed at the International Eel Symposium of the American Fisheries Society. It stated:

> *"The steep decline in populations of eels (Anguilla spp.) endangers the immediate future of these legendary fish. With less than 1% of major juvenile resources remaining, precautionary action must be taken immediately to sustain the stocks."*

This declaration underscores the inadequacy of regulatory mechanisms that have clearly failed to reverse this decline over the past two decades:

> *"As scientists in eel biology from 18 countries assembled at the International Eel Symposium ... we unanimously agree that we must raise an urgent alarm now. With less than 1% of juvenile resources remaining for major populations, time is running out. Precautionary action (e.g., curtailing exploitation, safeguarding migration routes and wetlands, improving access to lost habitats) can and must be taken immediately by all parties involved and, if necessary, independently of each other. Otherwise, opportunities to protect these species and study their biology and the cause of their decline will fade along with the stocks."*

This declaration (AFS 2009) was signed by eel scientists representing Aboriginal Nations, Belgium, Canada, Denmark, France, Germany, Great Lakes Fishery Commission, Ireland, Italy, Japan, Korea, Morocco, Netherlands, New Zealand, Sweden, Taiwan, United Kingdom, and the United States. It appears as the final chapter of the 2009 publication "Eels at the Edge" (American Fisheries Society 2009).

There are currently no regulatory mechanisms in the United States of America which adequately protect the American eel from extinction.

In its Addendum II, the ASMFC stated: "While the status of the American eel stock is uncertain, the latest stock assessment information indicates that the abundance of yellow eel (a juvenile life stage) has declined in the last two decades and the stock is at or near low levels. Further, relative abundance is likely to continue to decline unless mortality decreases and recruitment increases."

The Addendum then went on to state in the wake of the FWS 2007 Final Determination that listing of the American Eel was 'not warranted':

"The primary objective of this document is to recommend stronger regulatory language to improve upstream and downstream passage of American eel to state and federal regulatory agencies."

c) The United States Fish and Wildlife Service (FWS)

Pursuant to Section 18 of the Federal Power Act[34] ("FPA"), the FWS has the legal authority to require the licensees of private hydro-electric dams to provide safe and efficient upstream and downstream passage for American eels at hydro-electric dams in the historic range of American eel in the United States.

To date, the FWS has declined to exercise this legal authority in order to conserve the remaining American eels of the Atlantic seaboard of the United States of America. Instead, using weak inferences in their 2007 Final Determination to find that the eel's listing was 'not warranted'.

d) The National Marine Fisheries Service (NMFS)

Pursuant to the FPA the NMFS has the legal authority to require the licensees of private hydro-electric dams to provide safe and efficient upstream and downstream passage for American eel at hydro-electric dams in its historic range.

To date, NMFS has declined to exercise this legal authority in order to conserve the remaining American eels of the Atlantic seaboard.

e) The Federal Energy Regulatory Commission (FERC)

Pursuant to The FPA, FERC has the legal authority to require licensees of private hydro-electric dams to provide safe and efficient upstream and downstream passage for American eels in the historic range of American eel in the United States.

To date, FERC has declined to exercise this legal authority in order to conserve the remaining American eel stocks of the Atlantic seaboard of the United States.

f) The United States Environmental Protection Agency (EPA)

Pursuant to the federal Clean Water Act ("CWA"), [35] the EPA has the legal authority to require the licensees of private hydro-electric dams to provide safe and efficient upstream and downstream passage for American eels at hydro-electric dams to allow these waters to meet their designated uses for fishing and habitat for aquatic species as required under the CWA. Further, the CWA provides the authority to regulate the disposition of ballast water. To date,

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[34] 16 U.S.C. § 797(e).

[35] 33 U.S.C.§§1251 *et seq.*

the EPA has declined to exercise this legal authority in order to conserve the remaining American eels of the Atlantic seaboard.

g) The Spread Of *A. Crassus* Is Due To Inadequate Regulation Of Ship Ballast Water Discharge.

Numerous authors, as well as panelists in the 2004 FWS sponsored workshop, pointed out that ballast water of ships is the most likely mechanism for the rapid spread of the parasite from one location to another, through the dispersal of its intermediate hosts. This hypothesis is consistent with observations that busy deep water ports receiving empty ships (therefore laden with ballast waster), such as Boston Harbor and Hudson River, had some of the highest prevalence rates (76 percent and 60 percent respectively), while the parasite either did not occur or was at low prevalence in some coastal locations that had little or no shipping traffic (Morrison and Secor 2003; Rockwell et al. 2009). In infested areas, the frequency of infected eels, and the average number of *A. crassus* within their swim bladders, has been increasing (FWS Workshop 2004; Machut and Limburg 2008).

Ballast water is used to ensure the stability of ships when transiting the ocean with less than a full load, and is discharged in ports of call during the process of loading and unloading cargo (Bright,1998). The ability of thousands of invasive species of zooplankton, copepods, clams, and other invertebrates to invade new locations is made possible through the discharge of ship ballast water (Mooney and Hobbs 2000). Ballast water containing invasive species from riverine or coastal waters is regularly transported around the globe through international shipping. The further transport of invasive species along coastlines and inland waterways can occur through discharge from smaller commercial and recreational vessels, or aquaculture. The spread of zebra mussels is an excellent example of this ongoing problem.

A 2005 report by the Congressional Research Service (Buck 2005) documented extensively the inadequacy of regulations controlling invasive species transported in ballast water. The ongoing threat of parasites carried in the ballast water of ships was not mentioned anywhere in the 2007 Final Determination to not list the American eel.

h) Atlantic States Marine Fisheries Commission (ASMFC).

Pursuant to the federal Magnuson-Stevens Fisheries Conservation Act, the ASMFC has the legal authority to limit or prohibit the harvest of American eel along the Atlantic seaboard of the United States. To date, the ASMFC has declined to exercise this legal authority.

On March 10, 2004 the American Eel Management Board of ASMFC issued a press release recommending the protection of American eels under the ESA. The statement reads in part:

> *"Canadian and U.S. data show 2003 commercial landings are the lowest on record since 1945 and there are indications of localized recruitment failure in the Lake Ontario/St.*

> *Lawrence River system. The International Eel Symposium at the 2003 American Fisheries Society Annual Meeting reported a worldwide decline of eel populations, including the Atlantic coast stock of American eel ... The Commission also recommended that the U.S. Fish and Wildlife Service (FWS) and the National Marine Fisheries Service (NMFS) consider American eel in the Lake Ontario/St. Lawrence River/Lake Champlain/Richelieu River system as a candidate for listing as a Distinct Population Segment under the Endangered Species Act. The Board also recommended that the FWS and NMFS consider designating the entire coastwide stock as a candidate for listing under the ESA."*

Despite this statement in March 2004, ASMFC has not reduced or prohibited the ongoing harvest of all life stages of American eel from the waters of the Atlantic seaboard.

5. Other Natural Or Manmade Factors Affecting Its Continued Existence

a) Anthropogenic Impacts on American Eel

(1) Upstream Passage at Dams. Female American eels spend most of their lives in freshwater habitat along the Atlantic seaboard prior to returning to the Sargasso Sea to spawn. Safe and efficient access for juvenile eels to their freshwater habitat is essential to the survival of the American eel. Coastal river systems along the Atlantic seaboard are the sole migratory pathways for female American eels to gain access to their required freshwater habitat.

ASMFC (2000) states:

> *"By region, the potential habitat loss [for American eel] is greatest (91 percent) in the North Atlantic region (Maine to Connecticut) where stream access is estimated to have been reduced from 111,482 kilometers to 10,349 kilometers of stream length. Stream habitat in the Mid Atlantic region (New York through Virginia) is estimated to have been reduced from 199,312 km to 24,534 km of unobstructed stream length (88 percent loss). The stream habitat in the South Atlantic region (North Carolina to Florida) is estimated to have decreased from 246,007 km to 55,872 km of unobstructed stream access, a 77 percent loss."*

The Maryland Department of Natural Resources, MBSS Newsletter March 1999, Volume 6, Number 1 states:

> *"The most dramatic example of the decline of American eel abundance is dam construction on the Susquehanna River. Prior to the completion of Conowingo and three other mainstem dams in the 1920's, eels were common throughout the Susquehanna basin and were popular with anglers. To estimate the number of eels lost as a result of construction of Conowingo Dam, we used MBSS data on American eels from the Lower Susquehanna basin and extrapolated it to the rest of the basin above the dam. Our best*

> *conservative guess is that there are on the order of 11 million fewer eels in the Susquehanna basin today than in the 1920s.*
>
> *"The magnitude of this loss is corroborated by the decline in the eel weir fishery in the Pennsylvania portion of the Susquehanna River. Before the mainstem dams were constructed, the annual harvest of eels in the river was nearly 1 million pounds. Since then, the annual harvest has been zero. Given the longevity of eels in streams (up to 20 years or more) and their large size, the loss of this species from streams above Conowingo Dam represents a significant ecosystem-level impact. Because adult eels migrate to the Sargasso Sea to spawn and die -- transporting their accumulated biomass and nutrient load out of Chesapeake Bay -- the loss of eels has increased nutrient loads in the basin and reduced them in the open ocean where they are more appreciated."*

(2) **Downstream Passage at Dams.** Depending on their geographic location, female American eels spend 20 to 50 years in freshwater habitat along the Atlantic seaboard before returning to the Sargasso Sea to spawn. Safe and efficient access for maturing female American eels from their freshwater habitat to the Atlantic Ocean is essential for female American eel to spawn in the Sargasso Sea. Coastal river systems along the Atlantic seaboard are the sole migratory pathways for female American eels to gain access to their oceanic spawning grounds.

Records of severe kills of female American eels by the turbines of hydro-mechanical and hydroelectric dams have existed since as early as the 1880s. A corporate history of the S.D. Warren Paper Company describes severe kills of female American eels at the company's dam at Ammoncongin Falls on the Presumpscot River, Maine during the 1880s. The Presumpscot River is the outlet of Sebago Lake, the second largest lake in Maine. The dam at the outlet of Sebago Lake has long been called the Eel Weir Dam. The S.D. Warren corporate history states at page 46:

> *"Water power had its peculiar troubles: every cold winter morning anchor ice would clog in the intakes, and the mill would be down. Then when warm weather came, the water would be full of eels and eels are fish with tough hides. The blades of the water wheels would not chew them up and there are frequent entries in the record stating the water supply had failed and the mill was down, because the eels had stopped the wheels."*

One hundred years later, a similar report was made in 1996 by the operator of the Damariscotta Mills hydro-electric dam on the Damariscotta River in Newcastle, Maine to Lewis Flagg of the Maine Department of Marine Resources ("MDMR"). [36]

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[36] November 12, 2004 Petition from Timothy A. Watts and Douglas H. Watts, requesting that the FWS and NMFS list the American eel as an endangered species under the Endangered Species Act.

Hydro-electric dams located on the coastal watersheds of the Atlantic seaboard are a major source of mortality for female American eels as they attempt to migrate from freshwater to the Sargasso Sea to spawn. Of 15,570 dams blocking American eel habitat in the United States, Busch et al. (1998) reported that 1,100 of these dams are used for hydroelectric power. Few of these 1,100 dams provide safe passage for migrating female American eels. As a result, downstream passage by female American eels at these dams is via the project turbines, which results in the death of virtually all female eels attempting to migrate (see Appendix for graphic photographs of the effects of turbines on migrating eels[37]).

Radio tagging studies of migrating female American eels by the MDMR at two hydro-electric dams in Maine indicate nearly 100 percent of adult female eels entering project turbines are killed or severely injured and, therefore, are unable to complete their spawning migration (MDMR 2002).

Despite clear evidence of the deadly effects of turbines on large eels, ASMFC (2000) states:

> *"Downstream passage to the American eel's historic habitat is just as important as successful upstream access. Therefore, turbine induced mortality during downstream passage needs to be resolved since it impacts prespawning adult silver eel."*

A December, 1994 memorandum written by State of Maine fisheries biologist Frederick W. Kircheis states: *"Apparently eels are attracted to the current drawn by the turbines while migrating at night. "*

Radio-tracking of adult American eels by the MDMR just above the Lockwood hydro-electric project on the Kennebec River during fall 2002 indicates that 40 percent or more of the adult American eel attempting to migrate past the Lockwood Project each fall are entrained and killed in the Lockwood Dam turbines, despite the availability of the project spillway for passage (MDMR 2003).

Radio-tracking of adult female American eels by the Maine Department of Marine Resources (MDMR) at the Benton Falls Project in 2000 and 2001 indicate more than 50 percent of the migrating eels attempting to pass the Benton Falls Project are entrained and killed in the project turbines. The studies also found that 100 percent of the eels entrained in the Benton Falls Project turbines were killed by them. In fall 2001, MDMR staff used an underwater video camera at this project turbine outfall to attempt to locate two radio-tagged eels which had passed through. The video camera revealed large numbers of dead eels and eel carcasses resting on the river bottom at the turbine outfall. MDMR's 2001 study reported stated: *"Based on two years data, the surface bypass at Benton Falls is not efficient at passing eels."*

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[37] Powerpoint presented at A.D Latornell Conservation Symposium 2008, Barriers Management Session, Alliston, Ontario, November 20, 2008

The State of Maine states it has no legal authority to stop the ongoing killing of female American eel at the Benton Falls Project.

The FWS has documented large kills of migrating female American eels at the Holyoke Dam, the lowermost hydro-electric dam on the Connecticut River. [38]The Connecticut River is the largest watershed in New England. To date, no provision for safe passage of migrating female eels is provided at the Holyoke Dam or any other hydro-electric dam in the Connecticut River watershed.

There is no question that hydro-electric dam turbines inflict carnage on migrating eels. On the downstream journey, eels may have to pass through those turbines. Mortality may be 50 percent or more for some types of turbines, with 80-100 percent being injured (Haro *et al.* 2003). This is particularly important because the largest eels migrating downstream will be caught by the turbines. Upstream passage past dams is difficult. As a result, few eels live upstream of dams. However, those that do successfully migrate upstream of dams, grow larger than eels that remain downstream of the dam. This is because there is more competition for food and space downstream of the dam. These larger eels found upstream of dams are disproportionately female. The larger the eel, the more fecund. So while the eels that are able to migrate upstream from a dam grow larger and more fecund, sadly, because of their large size, they are more likely to be killed by turbines. The existing configuration results in turbines killing a higher percentage of large, fecund females on their way to spawn. This has a disproportionately detrimental effect on recruitment.

b) New Scientific Information Supports The Conclusion That The Effects Of Global Warming On Oceanic Conditions Have Contributed To The Decline Of The American Eel.

According to several recent authors (including several available but not cited in the FWS literature used in support of their decision), changes in oceanic conditions are currently contributing to the dramatic decline of anguillid eels worldwide (ICES 2006; IPCC 2007; Bonhommeau et al. 2008; Friedland et al. 2009; Tsukamoto 2009). The mechanisms by which this is occurring are primarily: sea surface temperature changes affecting depth of the mixed layer which disrupts the primary productivity in eel's spawning areas, as well as changes in latitude of the 22.5°C isotherm (affecting northern extent of spawning area), the transport and survival of their larvae (leptocephali).

Friedland et al. (2009) reported changes in the sea surface temperature in the Sargasso Sea and its effect on European eels. These changes included shifts in the winds in the northern Sargasso Sea, reducing southward Ekman transports and larval retention in the Sargasso Sea gyre. Such changes could also affect recruitment of the American eel because of the proximity of its

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[38] November 12, 2004 Petition from Timothy A. Watts and Douglas H. Watts, requesting that the FWS and NMFS list the American eel as an endangered species under the Endangered Species Act.

spawning area to that of the European eel in the southern Sargasso Sea, and similarities in the early migration pathways of both species out of the Sargasso Sea (Friedland et al. 2009; Tsukamoto 2009). Similarly, increased temperature negatively affects cross-shelf transport of eels and the condition and size of eels upon arrival at the coast (Wuenschel and Able 2008).

Little attention has been paid to the influence of changing ocean conditions contributing to the decline of eels until recent years. Bonhommeau et al. (2008) analyzed the relationships between oceanic conditions in eel spawning areas and glass eel recruitment success of three species of Anguilla: *A. anguilla*, *A. rostrata*, and *A. japonica*. They report that global warming since the 1970s appears to have hastened the decline of these species due to regime changes that decreased primary productivity and therefore eel recruitment.

Climate models based on historical temperature data and simulations for the 1870–2000 period were developed by Donner et al. (2005) and revealed that *"observed warming in the region* [including the sea surface temperature anomaly in the tropical North Atlantic] *is unlikely to be due to unforced climate variability alone."* The authors further show that under different scenarios of future greenhouse gas emissions, temperature rise that would both affect coral and primary productivity of the mass coral bleaching in the Eastern Caribbean (and hence the Antilles and North Equatorial currents). These effects can be expected even after stabilization of atmospheric CO^2 levels, adding to the long term threat to the American eel.

The worldwide recruitment decline in freshwater anguillid populations began almost simultaneously in the 1980s. While there are many factors that have contributed to this decline, recent analyses point to oceanic changes as being the most likely factor driving this trend (Bonhommeau et al. 2008; Friedland et al. 2007). Although the American eel has an evolutionary history (measured in tens of millions of years) and has survived climate changes before, this species has not previously faced the rapid, cumulative effects of anthropogeneticlly-exacerbated environmental climate change (IPCC 2007), coupled with other deleterious anthropogenetic environmental changes. These include: loss of habitat due to dams blocking migration, morbidity and mortality resulting from low stream pH (Jessop 2000; Vélez-Espino and Koops 2009), mortality and injury in hydroelectric turbines when mature eels are migrate downstream, commercial and recreational fishing harvest, contaminants ranging from mercury to PCBs (Ashley et al. 2007), and an invasive swim bladder nematode infestation (*A. crauss*) that disrupts swim bladder function and is especially debilitating during the long migration of eels to spawning areas in the Sargasso Sea. The life history traits and resilience exhibited by American eels, while allowing them to colonize diverse habitats in North America, is inadequate in the face the cumulative effects of these simultaneous threats.

c) Toxic Contaminants.

ASMFC (2000) states:

> *"American eel are benthic, long-lived and lipid rich. Therefore, American eel can accumulate high concentrations of contaminants, potentially causing an increased*

> *incidence of disease and reproductive impairment as is found in other fish species (Couillard et al. 1997). An analysis of the contaminants in migrating silver eel in the St. Lawrence River showed that the highest concentrations of chemicals were in the gonads. Concentrations of PCB and DDT were found to be 17 percent and 28 percent higher in the gonads than in the carcasses. The chemical levels in the eggs could exceed the thresholds of toxicity for larvae. Also, since the migrating females are not feeding, the chemical levels in the eggs could be even higher at hatching, increasing the likelihood of toxicity to the larvae (Hodson et al. 1994)."*

Because of mercury contamination, in 2008 the Vermont Department Of Health issued a Fish Consumption Advisory suggesting that no more than three meals per month of American eels be consumed (Vermont Fish & Wildlife Department 2008). Women of childbearing age, pregnant women, and breastfeeding mothers were advised to restrict their consumption to no more than one meal per month. Elevated levels of mercury in streams of the eastern United States are primarily the result of the burning of coal for electrical power generation. That has also resulted in acid rain that has altered stream chemistry and ecology, and has killed fish in acidified streams for many decades.

6. Emerging threats to the American eel

Emerging threats to American eels include: electro-magnetic fields from submarine cables, acoustic disturbance from offshore wind development (Oham et al. 2007), and the potential for biofuel production from floating biomass (including sargassum) harvested from gyres in the open ocean (Markels 2009).

III. CONCLUSION

American eels are virtually unique from other animals in that they spawn only once in their lives, in the Sargasso Sea. Eels are harvested at virtually every stage of their lives. They have lost access to 90 percent of their habitat due to anthropogenic activities. The remaining habitat is severely degraded due to a number of anthropogenic threats, including contaminants, urbanization and acid rain. Those eels which successfully evade harvest and find suitable habitat to mature are at risk of death or mortal injury as a result of contact with hydro-electric dam turbines during their downstream spawning migration. Even then, if they survive the turbines, the invasive parasite *a. crassus* can still kill them before they can complete their spawning migration.

The government has the legal authority to eliminate all mortality to American eels caused by human harvest and hydro-electric turbine mortality using the authorities of FERC and the NMFS. Those authorities have not been exercised.

The FWS has the ultimate authority to protect the American eel under the ESA. In 2007, ignoring the advice of experts and the data demonstrating catastrophic declines, the FWS chose not to list this species. Instead, the FWS 2007 Final Determination not to list the American eel was based on weak inferences. Ignoring the information presented the FWS painted an overly optimistic picture of the American eel's status and was dismissive of threats to the species, even in the face of contrary data and analysis. The decision not to list the American eel as threatened or endangered was based upon weak epistemological grounds.

The new information presented herein, including data and research not used or not available to the FWS at the time of their 2007 Final Determination demonstrates continued declines in American eel populations and more specific evidence of significant threats. The American eel is threatened and declining throughout all or a significant portion of its range and, therefore, qualifies for listing as threatened based on the provisions of the ESA.

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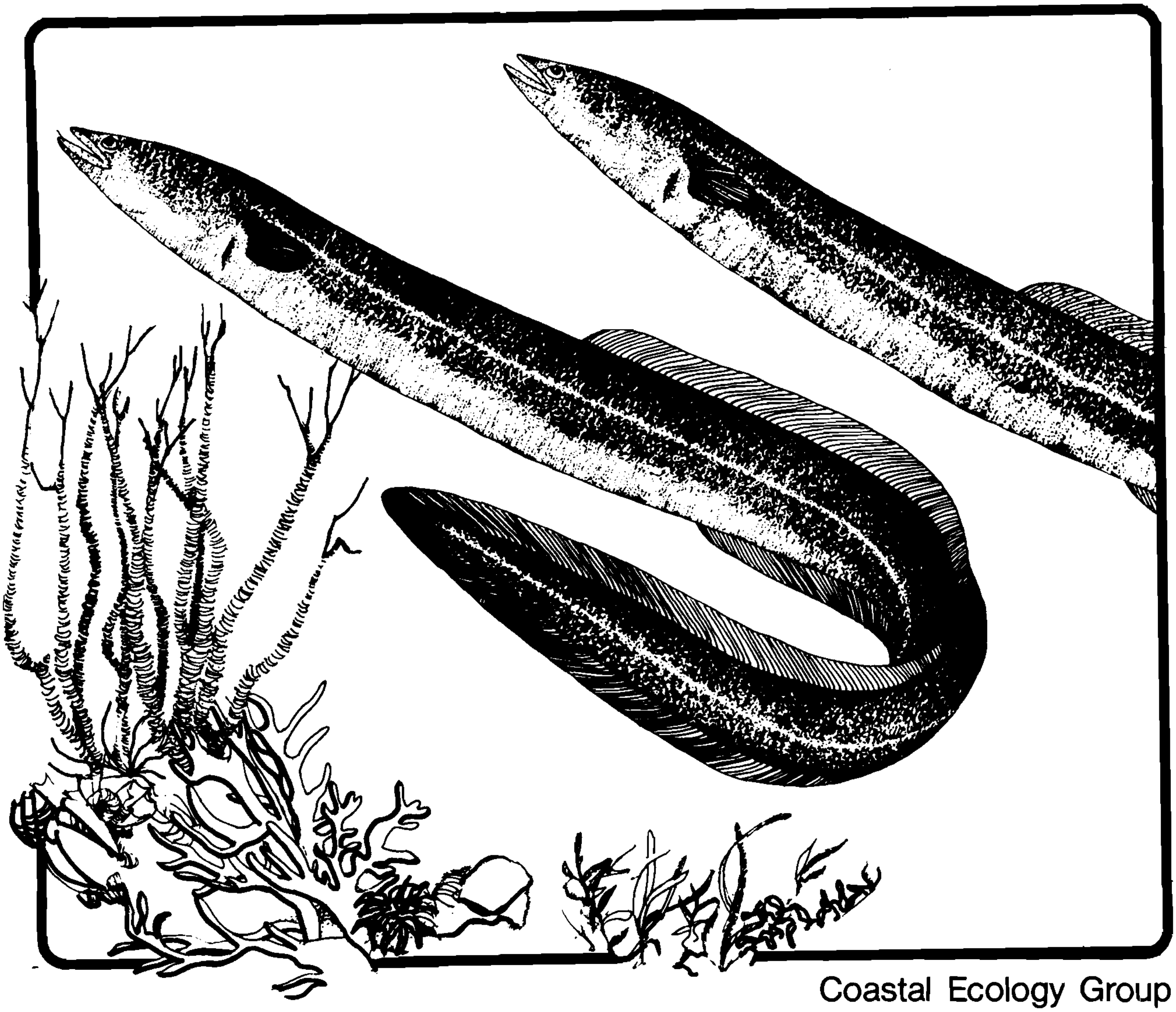
Biological Report 82 (11.74)
August 1987

TR EL-82-4

Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (North Atlantic)

AMERICAN EEL



Fish and Wildlife Service
U.S. Department of the Interior

Coastal Ecology Group
Waterways Experiment Station
U.S. Army Corps of Engineers

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Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (North Atlantic)

AMERICAN EEL

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Facey, D.E., and M.J. Van Den Avyle. 1987. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic)--American eel. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.74). U.S. Army Corps of Engineers, TR EL-82-4. 28 pp.

PREFACE

This species profile is one of a series on coastal aquatic organisms, principally fish, of sport, commercial, or ecological importance. The profiles are designed to provide coastal managers, engineers, and biologists with a brief comprehensive sketch of the biological characteristics and environmental requirements of the species and to describe how populations of the species may be expected to react to environmental changes caused by coastal development. Each profile has sections on taxonomy, life history, ecological role, environmental requirements, and economic importance, if applicable. A three-ring binder is used for this series so that new profiles can be added as they are prepared. This project is jointly planned and financed by the U.S. Army Corps of Engineers and the U.S. Fish and Wildlife Service.

Suggestions or questions regarding this report should be directed to one of the following addresses.

Information Transfer Specialist
National Coastal Ecosystems Team
U.S. Fish and Wildlife Service
NASA-Slidell Computer Complex
1010 Gause Boulevard
Slidell, LA 70458

or

U.S. Army Engineer Waterways Experiment Station
Attention: WESER-C
Post Office Box 631
Vicksburg, MS 39180

CONVERSION TABLE

Metric to U.S. Customary

Multiply	By	To Obtain
millimeters (mm)	0.03937	inches
centimeters (cm)	0.3937	inches
meters (m)	3.281	feet
meters (m)	0.5468	fathoms
kilometers (km)	0.6214	statute miles
kilometers (km)	0.5396	nautical miles
square meters (m^2)	10.76	square feet
square kilometers (km^2)	0.3861	square miles
hectares (ha)	2.471	acres
liters (l)	0.2642	gallons
cubic meters (m^3)	35.31	cubic feet
cubic meters (m^3)	0.0008110	acre-feet
milligrams (mg)	0.00003527	ounces
grams (g)	0.03527	ounces
kilograms (kg)	2.205	pounds
metric tons (t)	2205.0	pounds
metric tons (t)	1.102	short tons
kilocalories (kcal)	3.968	British thermal units
Celsius degrees (°C)	1.8(°C) + 32	Fahrenheit degrees

U.S. Customary to Metric

Multiply	By	To Obtain
inches	25.40	millimeters
inches	2.54	centimeters
feet (ft)	0.3048	meters
fathoms	1.829	meters
statute miles (mi)	1.609	kilometers
nautical miles (nmi)	1.852	kilometers
square feet (ft^2)	0.0929	square meters
square miles (mi^2)	2.590	square kilometers
acres	0.4047	hectares
gallons (gal)	3.785	liters
cubic feet (ft^3)	0.02831	cubic meters
acre-feet	1233.0	cubic meters
ounces (oz)	28350.0	milligrams
ounces (oz)	28.35	grams
pounds (lb)	0.4536	kilograms
pounds (lb)	0.00045	metric tons
short tons (ton)	0.9072	metric tons
British thermal units (Btu)	0.2520	kilocalories
Fahrenheit degrees (°F)	0.5556 (°F - 32)	Celsius degrees

CONTENTS

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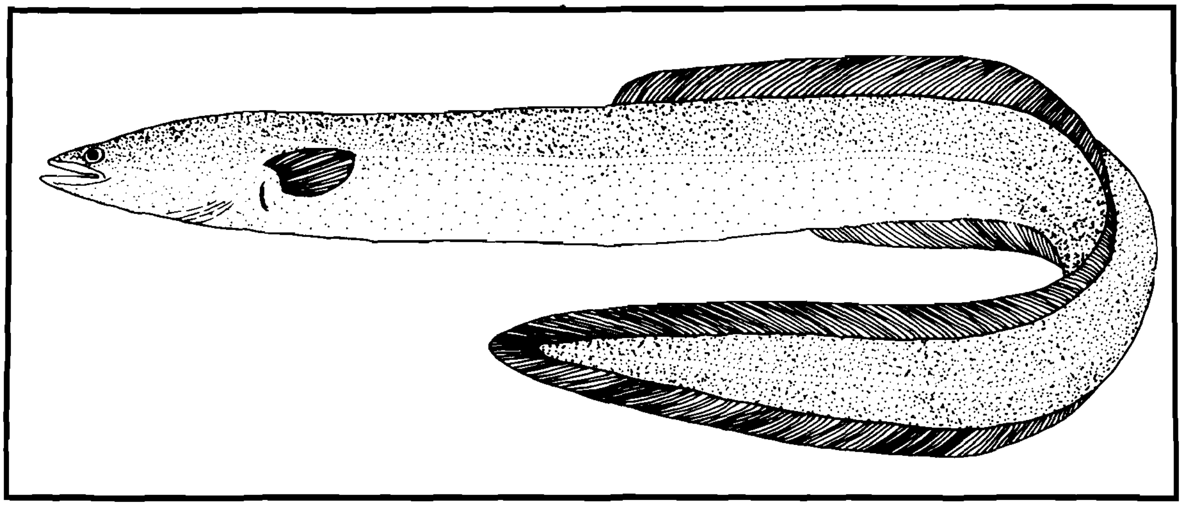

Figure 1. American eel.

AMERICAN EEL

NOMENCLATURE/TAXONOMY/RANGE

Scientific name......Anguilla rostrata
Preferred common name.........American eel (Figure 1)
Other common names...........Anguille, yellow eel, green eel, black eel, little eel, bronze eel, glass eel, silver eel, river eel
Class.....................Osteichthyes
Order...................Anguilliformes
Family....................Anguillidae

Geographic range: Adults or various developmental stages commonly occur in freshwater, coastal waters, and the open ocean from the southern tip of Greenland, Labrador, and Newfoundland southward along the Atlantic coast of North America, into the Gulf of Mexico as far as Tampico, Mexico, and in Panama, the Greater and Lesser Antilles, and southward to the northern portion of the east coast of South America (Tesch 1977). The species is abundant in the North Atlantic states (Figure 2), the eastern Canadian provinces, and southward to Mexico; it is resident in the Mississippi Valley, and occurs in the West Indies and Bermuda. Bertin (1956) reported the latitudinal range for the American eel as 5° to 62° N. It occurs in warm brackish and freshwater streams, estuaries, and coastal rivers, and sometimes in cold freshwater trout streams in mountainous regions. Its distribution has increased because of its

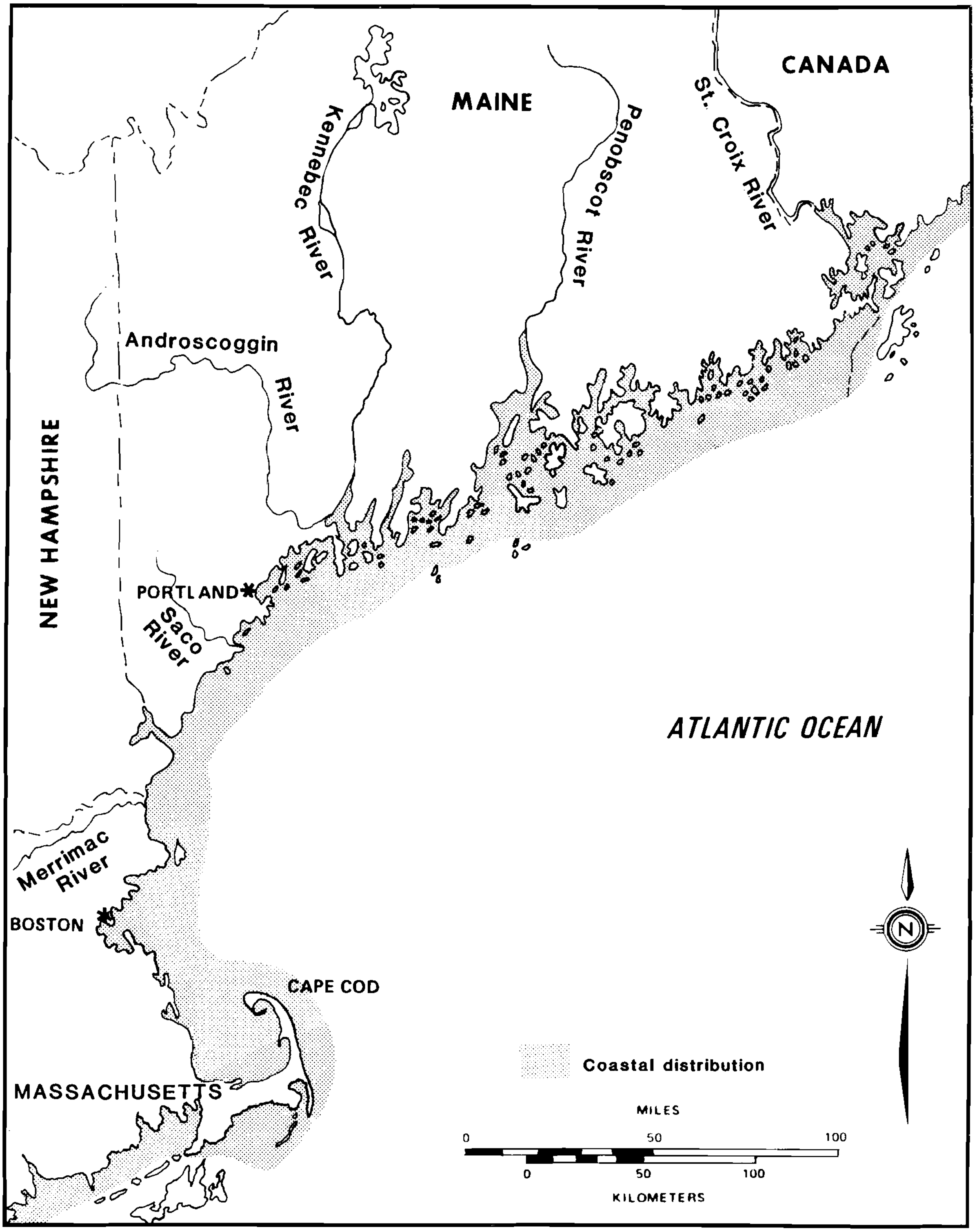

Figure 2. Major rivers that support the American eel in the North Atlantic United States. Eels also are common in other freshwater tributaries and in bays and estuaries.

hardiness (as shown by the range of habitats it occupies, including polluted areas), the ease with which it can be transplanted, and its ability to travel across damp ground and wet vertical surfaces such as dams. Adult eels are occasionally found in landlocked lakes, primarily in the northeastern United States.

MORPHOLOGY AND IDENTIFICATION AIDS

The American eel undergoes a series of morphological changes in its life cycle, which are described in the later section on LIFE HISTORY. The following information was summarized primarily from Fahay (1978) and Tesch (1977).

The body is elongate (Figure 1). The dorsal and anal fins are confluent with the rudimentary caudal fin. Pectoral fins are present, but ventral (pelvic) fins are absent. Scales form at about 3 to 5 years of age, but are minute and embedded, causing eels to appear scaleless. The lateral line is well developed. The mouth is terminal; the jaws have bands of small, pectinate, or setiform teeth, and the vomer has a long tooth patch. The number of vertebrae ranges from 103 to 111 but usually is 106 to 108 (Schmidt 1913). Ege (1939) presented comprehensive morphological data for A. rostrata.

No other anguillid eels occur in North American coastal waters, but the American eel's spawning area apparently overlaps with that of the European eel (Anguilla anguilla) (McCleave et al. 1986). Mean myomere counts for American and European eel larvae are 106.84 ± 0.032 S.E. and 114.52 ± 0.047 S.E. (Kleckner and McCleave 1985). Externally visible traits of adults are similar, but the European eel has more vertebrae (111-119; mean,115). Some authors have argued that European and American eels should be regarded as geographical variants of the same species (Williams and Koehn 1984). Recent analysis of mitochrondrial DNA indicates that American and European eels belong to separate breeding populations (Avise et al. 1986). The lack of interbreeding even though the spawning areas overlap supports the belief that American and European eels are different species. No available data conclusively point to geographic variations in morphology, and no subpopulations have been distinguished. Koehn and Williams (1978) noted protein differences among juvenile eels collected from different locations along the Atlantic seaboard, but concluded that the differences were due to variation in selective pressures among the environments in which the eels grew. Avise et al. (1986) reported no significant geographic differentiation in the mitochondrial DNA of 108 eels collected from Maine to Louisiana. This evidence strongly supports the conclusion that American eels are a single, panmictic breeding population.

REASON FOR INCLUSION IN SERIES

The American eel supports commercial and limited recreational fisheries throughout most of its range. In the United States eels are marketed for human consumption and as bait for crabs and game fishes, including striped bass (Morone saxatilis), cobia (Rachycentron canadum), and largemouth bass (Micropterus salmoides). Adult eels often are shipped alive or frozen to Europe where they frequently are smoked before marketing. Elvers (immature eels typically < 60 mm long) have been harvested in Maine and shipped to Japan where they were cultured in ponds. Pond rearing of eels is being developed in the United States, and there is a potential for development and expansion of an eel culture industry.

The American eel is an important food of larger marine and freshwater fishes. It preys on a variety of other animals including commercially important crabs and clams. Eels contribute to the loss of nutrients from freshwater rivers and lakes because of their high organic intake, large numbers, lengthy stay in freshwater, and subsequent migration to sea (Smith and Saunders 1955).

LIFE HISTORY

The life cycle of the American eel includes oceanic, estuarine, and riverine phases (Figure 3). Many details of its life history are only generally understood or have been inferred from knowledge of the European eel. Much of what is known has been derived from studies in the Middle and North Atlantic regions of the United States and the eastern provinces of Canada.

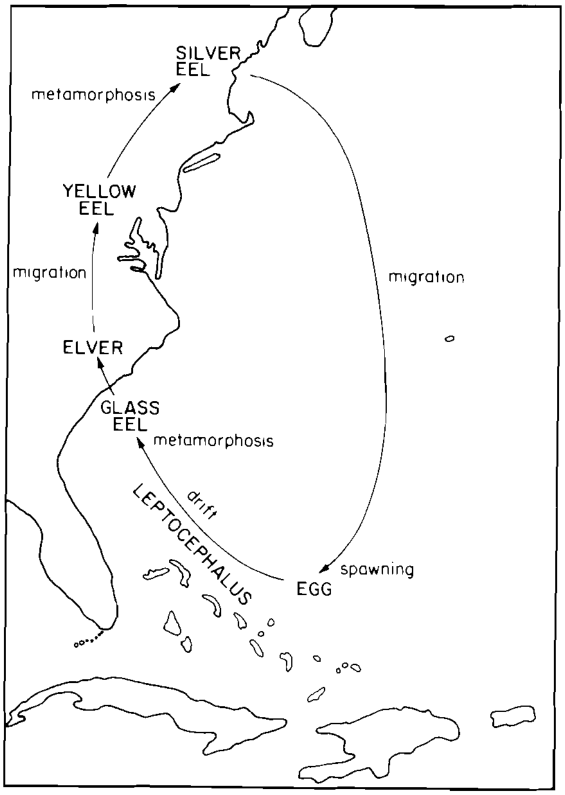

Figure 3. Diagrammatic representation of the life cycle of the American eel.

Different stages of the eel's life cycle are known by a variety of common names that are used throughout the scientific literature. The larva (leptocephalus) metamorphoses into an unpigmented glass eel which migrates into freshwater and gradually develops pigmentation. The young eel is now called an elver. Elvers may remain in coastal rivers or may continue to move upstream. The following growth phase, called the yellow eel, may last many years. Yellow eels may be sexually undifferentiated (gonads contain no definable gametes), hermaphroditic (oogonia and spermatogonia present), or sexually differentiated (females with oogonia; males with spermatogonia). Because none of these stages are capable of reproduction, all yellow eels are immature. Maturation is accompanied by changes in body color and morphology; maturing eels that migrate downriver and through the ocean to the spawning grounds are known as bronze eels or silver eels.

Spawning

The American eel is catadromous. It spends most of its life in rivers, freshwater lakes, and estuaries, but returns to the sea to spawn (Figure 3). The age at maturity has not been well defined; Fahay (1978) reported that maturation occurred after age III for males and at ages IV-VII for females from northerly populations, although females more than 15 years old have been reported in large inland lakes (Hurley 1972; Facey and LaBar 1981). Eels mature at younger ages in the southeastern United States than in New England (Helfman et al. 1984a; Hansen and Eversole 1984; Facey and Helfman, in press).

Before seaward migration in the fall, maturing eels begin metamorphosis into the silver eel stage.

(This metamorphosis and the timing of the reproductive migration are described later.)

Spawning by American eels has never been directly observed, and spawning areas have been inferred on the basis of collections of larvae. Spawning seemingly occurs in the Sargasso Sea as early as February and may continue until at least April (Kleckner et al. 1983; McCleave et al. 1986). Tesch (1977), who summarized work by Schmidt (1923), Vladykov (1964), Smith (1968), and Vladykov and March (1975), showed a spawning zone south of Bermuda and north of the Bahamas that is centered at about 25° N. and 69° W. McCleave et al. (1986) reported that American eels spawn in the area from 19.5° to 29.0° N. and 52° to 79° W., and that European eels spawn from 23° to 30° N. and 48° to 74° W. The youngest stages of American eel larvae may coexist with European eel larvae, but American eel larvae predominate west of 62° W. and south of 25° N. (Kleckner and McCleave 1985). The large overlap of spawning areas between American and European eels is evidenced by the capture of leptocephali of both species in the same trawl (McCleave et al. 1986). Thermal fronts that separate the northern and southern water masses of the Sargasso Sea are believed to form the northern limit of American eel spawning (Kleckner et al. 1983). The smallest American eel leptocephali that have been found (3.9-5.5 mm) were taken along the warm side of these fronts.

The depth at which spawning occurs is not known, but morphological and physiological evidence suggests that eels may migrate and spawn in the upper few hundred meters of the water column (Kleckner et al. 1983; McCleave and Kleckner 1985). The smallest leptocephali yet reported were taken in trawls fished at a maximum depth of about 300 m (Kleckner et al. 1983). Egg diameter of _A. rostrata_ is about 1.1 mm (Tesch 1977). Incubation periods of American eel eggs are not known, but the eggs of artificially spawned Japanese eels (_A. japonica_) are known to hatch in 38-45 hours at 23 °C (Yamamoto and Yamauchi 1974).

Relationships between eel size and fecundity for 21 eels (418-845 mm TL) were reported by Wenner and Musick (1974) as $\log F = -4.29514 + 3.74418 \log TL$, $\log F = 3.2290 + 1.1157 \log W$, where F = number of eggs per female, TL = total length (mm), and W = total weight (g). Therefore, fecundity for many American eels is between about 0.5 and 4.0 million eggs, with very large individuals (1,000 mm) producing perhaps as many as 8.5 million eggs. The European eel has fecundity estimates of 0.7 to 2.6 million eggs for individuals 630-920 mm TL (Boetius and Boetius 1980).

Adult eels presumably die after spawning. None have been observed to migrate up rivers, and spent eels have not been reported.

Larval (Leptocephalus) Stage

Hatching probably begins and peaks in February, but may continue through April (Kleckner et al. 1983; Kleckner and McCleave 1985; McCleave et al. 1986). The larval stage lasts up to about 1 year. The body is lanceolate, sharply pointed at both ends, and deepest at the middle; illustrations were published by Tesch (1977) and Fahay (1978). The length at hatching has not been described for the American eel; however, the Japanese eel is about 2.7 mm long at hatching and about 6.2 mm long 5 days after hatching (Yamamoto and Yamauchi 1974). Kleckner et al. (1983) caught larval American eels less than 5.5 mm long (perhaps less than 1 week old) from mid-February to early March. Schmidt (1925) collected larvae 7 to 8 mm long in February. The smallest larvae collected by Vladykov and March (1975) and Smith (1968) were 12 mm and

17 mm, respectively, and were caught in the summer.

American eel larvae grow as they are transported by ocean currents. Total lengths of larvae collected by Schmidt (1925) were 7 to 8 mm in February, 20 to 25 mm in April, 30 to 35 mm in June, 40 mm in July, 50 to 55 mm in September, and 60 to 65 mm by the end of the first year of life. The largest leptocephalus collected by Vladykov and March (1975) was 69 mm long. A thorough analysis of available data from 4473 larval and postmetamorphic American eels showed that the relationship between length (Y: mm TL) and collection date (X: Julian date) for 0-group leptocephali collected between 13 February and 15 October was $Y=0.238\ X - 6.569$ (Kleckner and McCleave 1985).

Leptocephali grow rapidly until October when growth slows or stops, and many metamorphose into glass eels (Kleckner and McCleave 1985). Most leptocephali undergo metamorphosis at 55-65 mm TL and 8-12 months of age. Limited evidence suggests that some eels may remain in the leptocephalus stage for more than 1 year. Smith (1968) reported a leptocephalus 50 mm long near the spawning grounds during April; it was thus too long to have been spawned in the immediate season (Fahay 1978). Vladykov and March (1975) also suggested that larval *A. rostrata* may spend more than 1 year in the sea.

Larvae are transported from the spawning grounds to the eastern seaboard of North America by the Antilles Current, the Florida Current, and the Gulf Stream. Power and McCleave (1983) developed a model of surface current drift to simulate the dispersal of eel leptocephali from the Sargasso Sea. Sampling has shown that larvae are abundant in the Florida Straits and in the area between Bermuda and the Bahamas from April through August (Smith 1968). Most leptocephali probably enter the Gulf Stream directly from the Sargasso Sea, rather than by a more southerly route through the Bahama Islands (Kleckner and McCleave 1982). Eldred (1971) found larval *A. rostrata* in the Gulf of Mexico and Yucatan Straits, but mechanisms by which they are dispersed into the Gulf of Mexico and southward to the coast of South America have not been determined.

Glass Eel and Elver Stages

During the pelagic phase, leptocephali reach the size and physiological state at which they begin to metamorphose. The early stages of this transition involve a decrease in length and weight due to a reduction in water content, changes in the configuration of the head and jaws, and accelerated development of the digestive system (Fahay 1978). After these changes occur, the eels are similar in overall morphology to yellow eels, but lack external pigmentation and are therefore called "glass eels." Glass eels actively migrate toward land and freshwater, and develop external pigmentation as they enter coastal areas. These small, pigmented eels are called "elvers."

The young eels begin migrating upstream before pigmentation is complete. Initially they are active at night and burrow or rest in deep water during the day (Deelder 1958). They typically move up into the water column on flood tides and return to the bottom during ebb tides (McCleave and Kleckner 1982; McCleave and Wippelhauser 1986). Similar behavior was reported for elvers at the mouth of the Indian River, Delaware, by Pacheco and Grant (1973), and for elvers of the European eel by Tesch (1977). The cues that trigger the change in behavior are not known, though Creutzberg (1959, 1961) showed that European glass eels were able to detect the odor of fresh water and alter their behavior accordingly. Sorensen (1986) showed that American eel elvers were strongly attracted to the odor of brook water and the odor of decaying leaf detritus and its associated microorganisms. Temperature gradient may also aid in the upstream orientation of glass eels (Tongiorgi et al. 1986). Glass eels and elvers may delay upstream migration at the freshwater-saltwater interface while behaviorally and physiologically adjusting to the new environment (Sorensen and Bianchini 1986).

Most glass eels and elvers move into coastal areas, estuaries, and up freshwater rivers in late winter or early spring. Vladykov (1966) suggested that elvers generally arrive in southern estuaries earlier and at smaller sizes than in the north, but records indicate considerable overlap in the timing of shoreward movements along the Atlantic coast. In the Southeastern and Middle Atlantic States, migrating glass eels and elvers have been collected from January through May (Jeffries 1960; Smith 1968; Fahay 1978; Hornberger 1978, cited by Sykes 1981; Sykes 1981; Helfman et al. 1984a).

Glass eels and elvers may reach New England estuaries as early as late winter (Jeffries 1960), but the main upstream migration is in spring. Glass eels have arrived at the coast of Maine from the end of March to about the third week of May (Dr. J. D. McCleave, University of Maine at Orono; pers. comm.). In Rhode Island the elver migration peaks during April and May (Haro 1986; Sorensen and Bianchini 1986), whereas in Maine the run is primarily from late April to June (Ricker and Squiers 1974; Sheldon 1974). Most upstream migrating eels arriving in May at the freshwater interface in a Rhode Island brook were not completely pigmented, but most were fully pigmented by July (Sorensen and Bianchini 1986). In 1974 the run along the southern and central portions of the Maine coast was composed primarily of unpigmented glass eels for the first few weeks and almost entirely of pigmented elvers by the eighth week. In northern coastal Maine the entire run was composed of glass eels. Smith and Saunders (1955) reported the arrival of elvers in Passamaquoddy Bay, New Brunswick, in late April.

Small numbers of elvers regularly arrive in estuaries in the fall, and Fahay (1978) suggested that these "early" arrivals may be the earliest spawned individuals or a segment of the main body of leptocephali that is moved northward more quickly than most by localized water currents. Alternatively, these elvers may be "late" arrivals produced from leptocephali that did not metamorphose during the previous winter and spring.

Elvers eventually begin swimming upstream and become most active during the day (Sorensen and Bianchini 1986). The onset of this active upstream migration may be triggered by changes in water chemistry caused by intrusion of estuarine water during high spring tides (Sorensen and Bianchini 1986). Tesch (1977) indicated that elvers of A. anguilla orient to river currents for upstream movement; if the current becomes too weak or too strong (velocities not specified), the fish may move into backwater areas, severely delaying upstream progress. Basic similarities in behavior of European and American eel elvers suggest that those of American eels would be similarly affected by fast or slow river currents.

Haro (1986) indicated that the main concentration of elvers in a coastal Rhode Island stream required about 1 month to move a distance of 200 m above the tidal zone, and that some American eels may continue migrating upstream as yellow eels of age II or older. The scarcity of small, young eels in lakes that are far inland supports the idea of continued upstream migration by yellow eels (Hurley 1972; Facey and LaBar

1981; Kolenosky and Hendry 1982). Eels ascending the eel ladder at the Moses-Saunders Dam on the St. Lawrence River at Cornwall, Ontario (approximately 1600 km from the ocean), were generally 3 to 8 years old (Liew 1982).

Yellow and Silver Eels

Many investigators (e.g., Bigelow and Schroeder 1953; Vladykov 1966) have stated that female yellow eels occur primarily in freshwater, and males generally in saltwater or brackish water. Dolan and Power (1977), however, after an extensive review of literature, concluded that this "female-freshwater, male-saltwater" theory was not supported. In a Georgia river, the percentage of sexually differentiated yellow eels that were males was 36 in the estuary and 6 in freshwater (Helfman et al. 1984a). In the Cooper River system in South Carolina the percentages of males were 7 in saltwater (Michener 1980), 5 in brackish water (Hansen and Eversole 1984), and 3 in freshwater (Harrell and Loyacano 1980). Winn et al. (1975) reported higher percentages of males in freshwater and females in saltwater in Rhode Island streams and estuaries, but did not explain the methods used to determine sex. Dolan and Power (1977) indicated that histological examination of the gonads is necessary to determine sex in eels.

Sexual differentiation does not occur until eels are about 200-250 mm long (Dolan and Power 1977). Before completion of the differentiation process some eels have gonads containing male and female gametes (juvenile hemaphroditism; Tesch 1977), but after gender is established, it does not change (Fahay 1978). Differentiated and undifferentiated yellow eels may overlap considerably in size and age (Gray and Andrews 1970; Dolan and Power 1977; Hansen and Eversole 1984; Helfman et al. 1984a).

In addition to the possible freshwater-saltwater variation in the sex ratio, there seems to be geographic variation in the distribution of the sexes. Vladykov (1966) wrote that males predominate from New Jersey to Florida, whereas females predominate from New York to Newfoundland. Although work in South Carolina and Georgia did not support the idea that southern stocks are predominantly male, the percentage of males was higher than that reported in northern areas. Vladykov believed that a latitudinal change in sex composition was related to the size differences in elvers along the coast, and supposed that the smaller elvers entering southern streams become males and the larger elvers entering northern systems develop into females. The presumed geographic distribution of sex in the American eel may be a result of selectivity of sampling gear and the possible exclusion of smaller males in northern studies, plus the assumption that the geographic distribution of sex in the American eel would parallel that demonstrated for the European eel (Dolan and Power 1977).

Limited evidence suggests that the gender of American eels is determined to some extent by environmental factors. Fahay (1978) wrote that the sex of the European eel can be environmentally influenced, but indicated that the factors responsible could only be speculated about. The long developmental period in freshwater or brackish water in combination with juvenile hermaphroditism provides a setting in which environmental factors could regulate the gender of eels.

Male American eels tend to be more abundant in estuaries than in upriver sites, and more males have been found in Southeastern States than in northern locations. One possible explanation is that male leptocephali and elvers do not migrate as far as females, and hence remain in southerly or downstream areas. It is also possible that male eels prefer higher salinities than females and move downstream to coastal areas after they are differentiated, but this behavioral pattern has not been observed and it would not explain the latitudinal trend. Even where males have been found to be most abundant, in Georgia estuaries (Helfman et al. 1984a), they are still outnumbered by females.

The fact that American eels appear to be a single, panmictic population suggests that latitudinal variations in the sex ratio are not genetically determined but could be due to variations of environmental factors, such as food quality and population density (Fahay 1978). Parsons et al. (1977) believed that stocking of European eel elvers into Lough Neagh, Northern Ireland, led to a higher population density and a marked increase in the proportion of male eels that subsequently emigrated from the lake. Similarly, Egusa (1979) indicated that elvers of A. anguilla and A. japonica grown in Japanese ponds under crowded conditions produced higher percentages of males than are found in wild populations, suggesting that variations in the sex ratio of anguillid eel populations may be related to population density. Salinity apparently is not an important sex determinant; sex ratios were similar in the freshwater and brackish water culture ponds studied by Egusa.

Growth rate, which is affected by temperature, food availability, and length of the growing season, might also be a factor in determining sex. This could result in different life history strategies for males and females (Helfman et al., in press). Eels that grow rapidly, such as those in highly productive southern estuaries, may have greater reproductive fitness if they are males. This is especially true if rapid growth results in earlier maturation (see Stearns and Crandall 1984). Large size would not be beneficial to male eels because small mature males can produce an abundance of gametes. However, the fecundity of female eels is highly dependent on size. Therefore, females that grow slower but reach larger sizes, such as those in northern and upriver locations, probably contribute more eggs to the next generation than do females that grow rapidly but mature at younger ages and smaller sizes, such as those in the southeastern United States. Natural selection would perpetuate such a system where the fastest growing eels tend to be males whereas eels that grow slower but get larger are females (Helfman et al., in press).

Eels are more active at night than during the day. Direct observation of yellow eels in a north Florida cave-spring indicated that eels changed behavior at dawn and dusk, when light levels were generally 10-100 lux (Helfman 1986). Laboratory studies have shown that silver eels are also more active in darkness than in light, and that activity peaks during light-dark transition (Edel 1975, 1979). Telemetry showed that yellow eels in a tidal creek were generally inactive during the day and active at night (Helfman et al. 1983). Activity was, however, influenced by tidal cycles with eels exhibiting greater activity during high tide. In a tidal cove studied in Maine, eels were moderately abundant in seine hauls at night but were never captured during the day (McCleave and Fried 1975). Commercial harvest information also indicates that eels are more active at night (see Eales 1968; Tesch 1977).

Estimates of the home range of eels extend to 3.4 ha in small streams, tidal rivers, and tidal creeks (Gunning and Shoop 1962; Bianchini et al. 1982; Bozeman et al. 1985); from 2.4 to 65.4 ha in a large lake (LaBar and Facey 1983); and < 100 m along a tidal creek in summer in a Massachusetts salt marsh (Ford and Mercer 1986). Ford and Mercer suggested that large eels may establish territories in the wider marsh creeks, thus restricting small eels to narrower creeks at the back of the marsh. Agonistic interactions in which large eels displace smaller eels have been reported elsewhere (Helfman 1986).

Eels begin the spawning migration in late summer and fall throughout much of New England and eastern Canada. Migration from lakes that are well inland may begin earlier. Catches of eels leaving Lake Champlain by way of the Richileau River were heaviest from June to August (R. Thuot, commerical fisherman, Iberville, Quebec; pers. comm.). Eels seem to leave later in the Southeastern and Middle Atlantic United States than in New England States. This delay may function to synchronize arrival at the spawning grounds in the Sargasso Sea (Wenner 1973; Facey and Helfman, in press). Many downstream migrating eels may not yet have developed the external characteristics associated with the migratory silver eel stage. Northern eels may begin migration at an earlier developmental stage, perhaps to compensate for the longer time required to reach the spawning grounds (Wenner 1973).

The metamorphosis from yellow eel to silver eel includes several physiological changes: (1) color change (to a metallic, bronze-black sheen; pectoral fins change from yellow-green to black); (2) fattening of the body; (3) thickening of the skin; (4) enlargement of the eyes and changes in visual pigments in the eye in preparation for migrating at greater ocean depths (Vladykov 1973; Beatty 1975); (5) increased length of capillaries in the rete of the swim bladder, which also may be an indication of migration at greater depths (Kleckner and Kruger 1981); and (6) degeneration of the digestive tract. Silver (metamorphosed) eels appear to be better adapted to swimming than yellow eels (Holmberg and Saunders 1979).

Few details are known about the oceanic spawning migration of the American eel. The first collections of adults in offshore waters were reported by Wenner (1973) in the open ocean southeast of Cape Cod; east of Assateague Island, North Carolina; and southeast of Chesapeake Bay. The means by which eels locate the spawning grounds are poorly understood. Miles (1968) concluded that eels were capable of noncelestial orientation (southward), and Rommel and Stasko (1973) indicated that eels may use geoelectric fields generated by ocean currents for orientation. Robins et al. (1979) photographed two adult _Anguilla_ eels on the floor of the Atlantic Ocean in the Bahamas at depths of about 2000 m, and although it was impossible to identify the species, the authors believed the specimens to be prespawning _A. rostrata_.

Stasko and Rommel (1977), who tracked five migrating eels in the lower St. Croix River estuary, New Brunswick, Canada, reported that one eel moved 25 km in 20 h and another moved 38 km in 40 h. The eels they studied showed considerable vertical movements in the water column; behavior did not change with diel or tidal cycles. Edel (1976) believed that the depth at which American eels migrate in the ocean varied with light intensity, and that swimming depth varied with turbidity of the water.

GROWTH CHARACTERISTICS

For the American eel the length at hatching is not known; however, the Japanese eel hatches at about 2.7 mm (Yamamoto and Yamauchi 1974). Growth rate of American eel leptocephali has been estimated to be 0.243 mm/day (Wippelhauser et al. 1985). Larvae typically reach 40 to 70 mm after 1 year. The metamorphosis from planktonic larva to the upstream migrating form is accompanied by a decrease in length and weight due to reduction in water content of the body. Glass eels captured while migrating upstream in late February in Georgia were 49-56 mm long and 250-300 days old (Helfman et al. 1984a). The length of glass eels collected from January through April in South Carolina averaged 55 mm long and ranged from 45 to 65 mm (Hornberger et al. 1978). Ricker and Squiers (1974) reported that glass eels and elvers caught along the coast of Maine from late April through the end of June averaged 59.2 mm (95% confidence interval, 57.5-60.8 mm). Elvers grow slowly, reaching about 127 mm after the first year in freshwater (Bigelow and Schroeder 1953). Yellow eels typically grow slowly but reach weights up to 6.8 kg; females caught from the St. Lawrence River were 960 to 1,270 mm in length and weighed 0.9 to 4.5 kg (Fahay 1978). Females grow to a larger size than males.

Eels have been aged from otoliths and scales. Otoliths in eels consist of a translucent nucleus (formed at sea), surrounded by broad opaque summer zones and narrow translucent winter zones (Gray and Andrews 1971). Eels in Canadian waters formed their first scales at 160 to 200 mm during their third to fifth year of life, and annual rings were formed on the scales in subsequent winters (Smith and Saunders 1955). Thus, in northerly areas, age in years generally is the number of scale rings plus three. However, because scales continue to form as the eel grows, different scales from the same fish yield different ages (Smith and Saunders 1955). Although otoliths may show more than one opaque ring in a year (Deelder 1976), they are preferred for estimating the age of eels.

Growth rates within year classes are highly variable, leading to considerable variation in length at age and poor predictability of age from size. Lengths of eels at various ages in northern locales are summarized in Table 1. Eels in the Southeastern United States seem to mature at younger ages and smaller sizes and therefore may not get as large as northern eels (Helfman et al. 1984a).

The great variability in length within an age class makes it virtually impossible to accurately estimate eel growth rates from length-age regressions. Perhaps the best way to determine growth rates is to monitor individuals during long-term tagging studies. Helfman et al. (1984b) compared growth rates estimated from length-age analysis to measured growth rates of tagged eels (initial size: 275-475 mm) in a Georgia estuary. On the basis of indirect measurements (length-age regression and mean-length-at-age analysis), estimated annual growth rates were 44 mm/year, whereas independent direct measurements (seasonal summation and long-term recaptures) yielded values of 57 and 62 mm/year. Gunning and Shoop (1962) reported that four recaptured eels (initial lengths, 255-915 mm) in Louisiana streams grew an average of 140 mm/year (range, 46-325 mm/year). In Massachusetts salt marshes, Haedrich and Polloni (1978) showed that eels averaging 52 cm long grew about 4% per year, and Polloni et al. (1980) reported that eels 500-700 mm long grew about 6% (range, 4.1-8.4%). The lengths of 10 eels tagged in 1979 and recaptured in 1986 in Vermont waters of Lake Champlain increased an average of 9.7 cm over the 7-year period (Dr. G. W. LaBar, University of Vermont, Burlington; pers. comm.).

COMMERCIAL AND SPORT FISHERIES

The European market has been the major outlet for U.S. landings of yellow and silver eels (Fahay 1978). Eels are hardy and can be densely packed and shipped alive if they are

Table 1. Total lengths (cm) of American eels at various ages in different localities.

	Locality							
Age group	Newfoundland[a]	New Brunswick[b]	Ontario[c]	Vermont[d]	Rhode Island[e]	New Jersey[f]	Delaware River[g]	South Carolina[h]
I							12-16	
II	16-19		19-20				14-25	26-33
III	21-23	20-32	20-23				18-28	29-45
IV	23-30	22-40	22-32		27-46	29-32	24-32	30-59
V	25-40	26-50	29		28-51		26-34	33-62
VI	29-46	22-56	22-67		28-51	41-67	28-42	32-63
VII	36-50	30-62	29-67		29-58	36-67	29-43	42-66
VIII	43-59	32-62	39-70	43	33-64	44-70	35-47	48-69
IX	49-66	38-66	33-74	57	38-62	37-74	35-50	46-55
X	60-78	48-66	44-86	45-71	37-65	44-86	40-52	52-66
XI	66-84		63-90	50-79	46-65	63-90	45-54	
XII	75-77		67-94	48-80		67-94	43-64	
XIII			68-98	45-72		68-98		55
XIV			78-97	43-80		78-97	56-59	
XV			78-104	53-78		78-104		
XVI			78-100	53-85		77-100		
XVII			96-99	49-83		95-99		
XVIII			91	58-90				
XIX				51-82				
XX				66-86				
XXI				52-85				
XXII				58-85				
XXIII				80				

[a] Gray and Andrews 1971.
[b] Smith and Saunders 1955. Ages estimated by adding 3 years to the number of scale rings counted by authors.
[c] Hurley 1972.
[d] Facey 1980.
[e] Bieder 1971.
[f] Ogden 1970.
[g] Johnson 1974.
[h] Hansen and Eversole 1984. Ages estimated by adding 1 year to the number of inland years reported by authors.

kept moist, cool, and supplied with oxygen. Although live eels are preferred in Europe, many are shipped frozen.

Commercial fishermen use a variety of methods to catch eels, including lift nets, drift nets, traps, weirs, otter trawls, pound nets, fyke nets, spears, handlines, eel pots, and haul seines (Fahay 1978). Yellow eels in freshwater or brackish water are taken primarily with baited traps or eel pots.

A summary of catch statistics along the Atlantic coast from 1955 to 1973 showed that landings from the Middle Atlantic (New Jersey to Virginia) consistently exceeded those from the North Atlantic (Maine to New York) and South Atlantic (North Carolina to Florida) (Fahay 1978). From 1970 to 1973, the annual North Atlantic harvest averaged 125,418 kg, with an average value of $84,000. In 1977 the eel landings for Maine, New Hampshire, and Massachusetts were about 79,700, 2,700, and 143,300 kg, valued at $263,000, $5,000, and $173,000, respectively (U.S. Department of Commerce 1984). Massachusetts landings were about 100,300 kg in 1978 and 81,800 kg in 1979 (U.S. Department of Commerce 1980a), and Maine landings were about 60,500 kg in 1978 and 50,400 kg in 1979 (U.S. Department of Commerce 1980b). By 1985 the Massachusetts catch was less than 3,800 kg (E.D. Hubbard, Massachusetts Division of Marine Fisheries; pers. comm.). Landings in Maine and Massachusetts in 1980-85 are shown in Table 2. Some of the landing statistics may be inaccurate.

Although U.S. eel harvests seemed to be increasing through the 1970's, eel fishing in New England has declined drastically in recent years. The situation may be due to reasons cited by E. D. Hubbard, in her assessment of the Massachusetts eel fishery (pers. comm., June 1986).

Table 2. Preliminary commercial fishery landings of eels in Maine and Massachusetts, 1980-1985[a]. (Information provided by R. Schultz, Resource Statistics Division, National Marine Fisheries Service).

	Maine		Massachusetts	
Year	weight (kg)	value	weight (kg)	value
1980	47,938	$111,061	841	$219
1981	25,057	45,308	-	-
1982	20,478	36,637	205	23
1983	5,409	8,925	80	26
1984	-	-	2,148	1,679
1985	10,955	18,288	-	-

[a] Does not include 9 kg reported in New Hampshire in 1981.

"During the years from roughly 1975 to 1980 the estuarine eel fishery grew considerably in Massachusetts, principally on Cape Cod, south of Boston and in southeastern Massachusetts coastal towns. Numbers of men fishing increased as well as the total landings, although accurate statistics are lacking. This was due to the high ex-vessel prices paid to fishermen, the result of renewed interest and an ever-increasing European eel demand. Whereas nearly every European country consumes eels, apparently local supplies could not meet the total demand and so North American exports began to fill this gap.

"Somewhat abruptly in 1981 most of these U.S. export markets plummeted due to a number of factors, but principally due to the very tight economic situation in the U.S. as well as abroad. Other contributing factors were contaminated shipments of eels from Canada and grading (live eels) problems. Exports of all finfish have slumped over the last several years due to an inflationary U.S. dollar. During this time, the Europeans imported eels from new sources across the Pacific.

"Several well established eel buyers along the American East Coast closed their doors during 1982, [primarily due] to high shipping costs and inflated exchange rates. Because buyers were not interested in eels, or at much lower prices, very few persons fished during 1982, continuing through to the present. The last major buyer/ exporter in Massachusetts ceased his eel operations in 1985. With unfavorable market conditions continuing in Europe over the last 4 to 5 years, the coastal eel fishery here in Massachusetts has been practically nonexistent. In the fall months, the traditional Christmas eel demand in the larger U.S. cities means a short-term, high priced market for fishermen. But other than scattered and seasonally limited sales demand, fishermen have not set their pots, although the interest is very high. One buyer in Maine is doing business with some of the local fishermen and another company in New Hampshire has very recently expressed interest in exporting eels."

It is possible, however, that European demand for American eels may increase in the late 1980's because of the accidental release of toxic chemicals into the upper Rhine River in fall 1986; hundreds of thousands of European eels were killed. If the accident significantly affects European eel fisheries for many years, an increased demand for American eels might extend into the 1990's.

A fishery for European eel elvers began in Europe during the late 1960's to supply Japan's demand for young eels to use in pond culture. Elvers were packed live in boxes and shipped to Japan, where prices paid for local A. japonica elvers were $7/kg in 1965-68, $300/kg in 1969, and $330 to $925/kg in 1971-73 (Fahay 1978; Egusa 1979). Prices paid for European eel elvers in Japan initially were equivalent to those paid for local elvers, but European eels were inferior in the pond culture systems because of poor growth and disease problems; in 1973, the Japanese paid only $30 to $50/kg for European elvers (Egusa 1979).

Reports of $100 to $2,000 per kg attracted some Maine fishermen into the elver market, but they found that these reports were inflated over the actual value of a successful shipment (Ricker and Squiers 1974). Elvers vary widely in size, and the number per kilogram may range from about 2,200 to more than 12,000 (Ricker and Squiers 1974). Sheldon (1974) reported locations and techniques for catching, holding, and transporting elvers in Maine. In Maine, elver landings were 10 metric tons in 1977 and 7.6 in 1978, valued at $110,000 and $63,251 (Dow 1982). Massachusetts prohibits harvesting of elvers except for aquaculture purposes, for which a permit is required. From 1978 to 1986 only one such permit was requested and issued (E.D. Hubbard; pers. comm.). The Japanese Elver Culture Association began assessing the performance of Maine elvers in the mid 1970's. There have been reports that the elvers of the American eel did not thrive and that the Japanese eel culture industry began buying A. japonica elvers from China (L. Flagg, Maine Department of Marine Resources; pers. comm.).

The feasibility of commercial "grow-out" operations in North Carolina was assessed by Easley and Freund (1977). Interest in culturing was stimulated by rising prices during the late 1960's and early 1970's, but considerable refinement of techniques was needed. Development of eel aquaculture has focused on methods for collecting elvers and on physical features of grow-out systems. Hormone injections can be used to induce maturation of female American eels (Edel 1976), but proper spawning conditions are unknown, and eel culture remains dependent on capturing wild elvers. Hinton and Eversole (1978, 1979, 1980) evaluated the toxic effects of chemicals commonly used in aquaculture on glass eels (mean length, 55 mm), elvers (mean length, 97 mm), and yellow eels collected from South Carolina rivers. Lower temperatures and the shorter growing season might make commercial culturing of eels less practical at northern latitudes.

Restrictions on eel harvest vary among the North Atlantic states. In Maine the size of catch is not regulated, but certain permits and regulations pertain to some towns and rivers (Ricker 1976). Commercial fishing licenses are issued by the Department of Marine Resources, or by the Department of Inland Fisheries and Wildlife (for inland waters). The Department of Marine Resources also issues licenses for anyone buying or selling eels in the wholesale trade. In Massachusetts, coastal towns regulate commercial eel fishing in saltwater and estuaries (Amaral 1982; E.D. Hubbard; pers. comm.). Only eels 102 mm (4 inches) long or longer may be harvested, and only by nets, pots, spears and angling. Commercial fishing for eels is permitted in inland waters, but a permit and fishing license are required. Only eel pots with a mesh no less than 13 mm (0.5 inch) and a funnel opening not greater than 51 mm (2 inches) may be used. Fishermen are required to keep daily logs, and no eels less than 102 mm long may be taken. The Division of Marine Fisheries issues the licenses required to sell eels. New Hampshire also prohibits the taking of eels less than 102 mm long (T. Spurr, New Hampshire Fish and Game Department, Concord; pers. comm.).

Population size and biomass estimates of American eels are scarce and vary widely. Bianchini et al. (1982, cited by Bozeman et al. 1985) estimated eel biomass at 75 kg/ha in the tidal section of a Rhode Island river. Bozeman et al. (1985) reported about 13 kg/ha in a Georgia tidal creek. A 600-m section of a marsh creek in Massachusetts was estimated to contain about 350 yellow eels, a stock density equivalent to 875 eels/ha (Ford and Mercer 1986). Standing crops up to about 80 kg/ha were reported in lakes in New Brunswick, Nova Scotia, and Prince Edward Island (Smith and Saunders 1955). The eel biomass in Coleback Lake, Maine, was about 50 kg/ha (Rupp and DeRoche 1965), whereas estimates in shallow (<2 m) portions of Lake Champlain, Vermont, were 161 to 421 kg/ha (LaBar and Facey 1983). The biomass estimates in Lake Champlain may have been high because there had been no commercial eel fishery on the lake before the study.

Estimates of mortality or other vital statistics of eel stocks generally have not been reported, and factors regulating survival or stock size have not been evaluated. Helfman (unpubl. MS.[1]) suggested that the eel's long life in freshwater may make the stocks prone to local overharvest. Keefe (1982) suggested that declines in catch of eels per unit of fishing effort in North Carolina indicated overharvest. Because all American eels spawn in the Sargasso Sea, and there are apparently no genetically distinct stocks or subpopulations (Koehn and Williams 1978; Avise et al. 1986), overharvest in one region could affect recruitment in other regions. Kolenosky and Hendry (1982) suggested taking a conservative approach to the harvesting of eels in Canadian waters

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[1] Development and expansion of the fishery for American eels in Georgia. G.S. Helfman, Department of Zoology, University of Georgia, Athens, GA 30602. Project summary, University of Georgia Sea Grant Program, 1983.

of Lake Ontario, partly because of the declining catch per unit of effort. Nevertheless, some management policies allow or encourage locally heavy exploitation of migrating silver eels or elvers under the assumption that the numbers of elvers returning in later years will be maintained by escapement of spawning stock from other areas.

American eels are caught by sport fishermen along the entire east coast of the United States. The estimated catch in 1979 by marine and estuarine recreational fishermen was 113,000 eels in the North Atlantic States, 172,000 in the Mid Atlantic, 47,000 in the South Atlantic, and 43,000 in the Gulf coast region (U.S. Department of Commerce 1981).

ECOLOGICAL ROLE

Yellow eels are nocturnal, and a significant amount of their feeding is at night (Helfman 1986). They probably depend more on scent than on sight to locate food (Fahay 1978). The diet is diverse and generally includes nearly all types of aquatic fauna that occupy the same habitats. Eels swallow some types of prey whole, but also can tear pieces from large dead fish, crabs, or other items. Helfman and Clark (1986) documented the ability of eels to grasp large food items and spin rapidly to tear away pieces. Eels in freshwater feed on insects, worms, crayfish and other crustaceans, frogs, and fishes. Elvers collected from the Cooper River, South Carolina, ate aquatic insects (mainly larval and adult chironomids), cladocerans, amphipods, and fish parts (McCord 1977). The diet of yellow eels from the Cooper River varied with eel size and season. More types of food were eaten by intermediate-sized eels than by elvers or maturing eels; fish occurred in the diet primarily in winter and spring, whereas insects and mollusks were eaten from spring through fall. Crustaceans, bivalves, and polychaetes were the major prey of eels in lower Chesapeake Bay; blue crabs (Callinectes sapidus) and soft-shell clams (Mya arenaria) were significant prey (Wenner and Musick 1975). Eels shorter than 40 cm in New Jersey streams ate mainly aquatic insects whereas larger eels fed mostly on fishes and crustaceans (Ogden 1970). Most fishes eaten were bottom dwellers, reflecting the tendency of eels to feed near the bottom. In Vermont waters of Lake Champlain, eels ate primarily insects, crayfish, and fishes; larger eels (≥ 58 cm) ate more crayfish and fishes than did smaller eels (Facey and LaBar 1981). Eels have been considered significant predators on young salmonids, but this is not well supported by the literature. In New Brunswick streams, only 6 of 300 eels with food in their stomachs had eaten salmonids (Godfrey 1957). Of 4,340 European eels examined from six Welsh rivers, Sinha and Jones (1967) found only 10 that had eaten salmonids.

Little has been published about predation on eels. Hornberger et al. (1978) reported that elvers and small yellow eels were eaten by largemouth bass and striped bass in the Cooper River, South Carolina, but that eels were never a major component of these predators' diets. Leptocephali, glass eels, elvers, and small yellow eels probably are eaten by a variety of predatory fishes. Sorensen and Bianchini (1986) stated that older eels eat incoming glass eels and elvers. Grown eels are eaten by species of eels other than anguillids and by gulls, bald eagles (Haliaeetus leucocephalus), and other fish-eating birds (Sinha and Jones 1967; Seymour 1974).

Crane and Eversole (1980) found no parasites on glass eels migrating into the Cooper River, South Carolina, but examinations of elvers yielded four genera of protozoans (Trichodina, Ichthyophthirius, Myxidium, and Myxobolus) and one species of monogenetic trematode (Gyrodactylus anguillae). Crane and Eversole (1981) reported that 214 of 218 yellow eels collected from brackish waters of the Cooper River, South Carolina, were parasitized by 1 or more of 22 helminth species. About 48% of yellow eels collected from brackish portions of the Cooper River were infested with one or more ectoparasitic species from the classes Monogenea and Crustacea (Crane and Eversole, in press). Levels of parasitism by Ergasilus cerastes and E. celestis varied seasonally and with size and age of the host. Parasites of American eels in Quebec included protozoans, trematodes, nematodes, cestodes, and copepods (Hanek and Molnar 1974). The myxosporidian protozoan Myxidium zelandicum has been found in the kidneys and on the gills of the American eel (Komourdjian et al. 1977).

ENVIRONMENTAL REQUIREMENTS

Temperature

The eel's broad geographic range and diverse habitats suggest flexible temperature requirements. Elvers and yellow eels live in waters ranging from cold, high-elevation or high-latitude freshwater streams and lakes to warm, brackish coastal bays and estuaries in the Gulf of Mexico. Jeffries (1960) found elvers at temperatures as low as -0.8 °C.

Barila and Stauffer (1980) acclimated yellow eels to a range of temperatures between 6 and 30 °C and then measured preferred temperatures. Although preferred temperatures tended to increase with increased acclimation temperature, group differences were not significant, and the authors reported a final mean temperature preference of 16.7 °C. Karlsson et al. (1984) disagreed with the techniques and interpretation of Barila and Stauffer (1980), and claimed that acclimation temperature does influence preferred temperature. They found a final temperature preferendum of 17.4 ± 2.0 °C (95% confidence interval). Marcy (1973) reported that American eels survived passage through the cooling system of a nuclear power plant, during which they were exposed to elevated temperatures for 1-1.5 hr. Poluhowich (1972) suggested that the American eel's multiple types of hemoglobins serve to maintain a nearly constant blood oxygen affinity when the eel is exposed to temperature changes. American eels acclimated at 10 to 20 °C fed regularly and exhibited compensatory adjustments in oxygen consumption characteristic of many ectotherms (Walsh et al. 1983). However, acclimation to temperatures $\leq$ 5 °C for over 5 weeks resulted in cessation of feeding and a dramatic decrease in oxygen consumption.

Salinity

The mechanisms by which glass eels or elvers orient during their shoreward migration have not been described. Eels are known for their extremely sensitive sense of smell, and olfaction may play a role in the ability of elvers to locate freshwater (Sheldon 1974; Sorensen and Bianchini 1986; Sorensen, 1986). European glass eels and elvers become positively rheotactic when they first encounter freshwater that is mixed with seawater (Tesch 1977). Alterations of patterns or magnitudes of freshwater inflows to bays or estuaries could alter flow regimes and thereby affect the size, timing, and spatial patterns of upstream migrations by elvers.

Like temperature requirements, salinity requirements of postlarval eels can be inferred as being broad from the fact that the postlarval eels occur throughout a gradient of strictly fresh to brackish waters. Elvers do appear to delay upstream migration at the freshwater interface, however, perhaps to permit physiological adaptation to the new environment (Sorensen and Bianchini 1986). Leptocephali are in near-ionic equilibrium with sea water (Hulet et al. 1972), but the osmolality of glass eels and elvers has not been reported.

Dissolved Oxygen

Dissolved oxygen requirements have not been thoroughly documented, but eels generally select water with high oxygen tension (Hill 1969). Elvers are sensitive to low oxygen, and should be held and transported in water with an oxygen concentration of at least 11 ppm (Sheldon 1974). Because elvers can absorb oxygen through the skin, they can better be transported damp and in air than in poorly oxygenated water. Evidently this is also true of adult eels. Tesch (1977) wrote that, "The capacity of the adult eel to survive in both air and water is associated with its ability to use both branchial and cutaneous modes of respiratory gas exchange. The eel survives better in air than in poorly oxygenated or polluted water...."

Habitat Structure

Postlarval eels tend to be bottom dwellers and hide in burrows, tubes, snags, plant masses, other types of shelter, or the substrate itself (Fahay 1978). This behavior is reflected in their food habits, protects them from predators, and influences commercial fishing techniques. Few other freshwater fishes display similar habitat use; interspecific competition for living space may therefore be limited. The presence of soft, undisturbed bottom sediments is important to migrating elvers as shelter. Edel (1979) indicated that eels in his experimental systems were less active when shelter was present than when it was lacking. Vladykov (1955, cited by Fahay 1978) reported that adult eels in northern habitats lie dormant in the bottom mud during winter.

River and Tidal Currents

The glass eel's and elver's nocturnal activity and reliance on tides for upstream movement have already been mentioned. Flow alteration in estuaries might affect upstream migration of small eels. Dams and other obstructions probably inhibit migrating elvers (Tesch 1977), and limit recruitment to upstream sites; however, eels can travel over wet vertical surfaces such as dams.

Tides and the time of day affected movements of yellow eels in a tidal creek in Georgia (Helfman et al. 1983). Movements of eight telemetered eels were restricted to the main creek channel during the day, but at night the fish were near the mouths of feeder creeks at low tide or in flooded marsh areas during high tide. Helfman et al. (1983) termed this movement "a nocturnal activity pattern modified by tidal flow," and suggested that such movements were foraging trips.

Contaminants

Little work has been done on toxic effects of pollutants or the tolerance limits in American eels. Tolerance would be expected to vary with developmental phase, and the eel's long residence in freshwater rivers could lead to repeated doses of toxicants and accumulation of toxic levels (Holmberg and Saunders 1979). Work done by Hinton and Eversole (1978, 1979, 1980) on toxicity of aquacultural chemicals to various life stages of eels suggested that tolerance to chemicals increases with size or age.

In September 1976 the New York State Department of Environmental Conservation and the Department of

Health banned the possession and sale of eels taken from the Hudson River and Lake Ontario because levels of polychlorobiphenyls (PCBs) exceeded the U.S. legal maximum level of 2 ppm: they were 50-75 ppm in Hudson River eels and 2.5-4.5 ppm in Lake Ontario eels (Blake 1982). This ended the Hudson River fishery for eels. In 1978 the restrictions were modified to allow sales of Lake Ontario eels to foreign markets, which apparently permit higher PCB concentrations than are allowed in the United States.

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Species profiles are literature summaries of taxonomy, life history, and environmental requirements of coastal fishes and aquatic invertebrates. They are prepared to assist with impact assessments. The American eel is an ecologically and economically important catadromous species that occupies freshwater streams, rivers, brackish estuaries, and the open ocean during various phases of its life cycle. Adult eels apparently spawn in the Sargasso Sea, and ocean currents transport the developing larvae northward until the young metamorphose into juveniles capable of swimming shoreward and moving upstream into coastal areas, estuaries, and rivers. Developing eels commonly remain in freshwater or brackish areas for 10-12 years before migrating to spawn. American eels tend to be bottom-dwellers and feed on a variety of fauna that occupy the same habitats. Eels occupy areas having wide ranges of temperature, salinity, and other environmental factors, suggesting broad tolerance limits, but few studies of requirements have been reported. Salinity patterns and water currents created by river discharges into coastal areas apparently provide the gradient that cues shoreward migration of juvenile eels. Alteration of patterns of freshwater inflows to estuaries and bays could affect upstream migrations.

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National
Engineering
Laboratory

Development of Environmentally Advanced Hydropower Turbine System Design Concepts

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LOCKHEED MARTIN

DISCLAIMER

This report was prepared by Voith Hydro, Normandeau Associates, TVA, Harza Engineering Company and the School of Civil and Environmental Engineering of Georgia Institute of Technology for the purpose of documenting work undertaken in connection with the Advanced Hydropower Turbine System Program sponsored by the U.S. Department of Energy (DOE). The information contained in this report represents our opinion about the issues it addresses within the context of its intended purpose. However, the DOE, any co-sponsor of this program, Voith Hydro, Normandeau Associates, TVA, Harza Engineering Company and the School of Civil and Environmental Engineering of Georgia Institute of Technology, and any of their representatives make no warranty or representation of any kind (whether express or implied) as to the accuracy, completeness, or use of such information, and expressly disclaim any and all liability for the use of, or reliance on, such information beyond its stated purpose.

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Development of Environmentally Advanced Hydropower Turbine System Design Concepts

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Pacific Gas & Electric
Tennessee Valley Authority
Washington Water Power Company
National Hydropower Association (NHA)

The Technical Review Committee consisted of representatives of these participants and representatives from:

U.S. Department of Energy
U.S. Army Corps of Engineers
Bonneville Power Administration
National Marine Fisheries Service
Department of Interior
Bureau of Reclamation
Northwest Power Planning Council
Native American Tribes
Oak Ridge National Laboratory
Idaho National Engineering Laboratory
Other utilities and private companies

The Technical Review Committee provided valuable input toward development of Advanced Hydropower Turbine System Design Concepts and to this report.

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1.0 EXECUTIVE SUMMARY

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The team consisting of Voith Hydro, Normandeau Associates, TVA, Harza Engineering Company and the School of Civil and Environmental Engineering of Georgia Institute of Technology worked together on the Development of Environmentally Advanced Hydro Turbine Design Concepts to reduce hydropower's impact on the environment, and to improve the understanding of the technical and environmental issues involved, in particular, with fish survival as a result of their passage through hydro power sites. Through a combination of advanced technology and engineering analyses, innovative design concepts for this Phase I project were developed. In line with the request of the DOE, the solutions explored are adaptable to both new and existing hydro facilities.

The approach teamed a turbine design and manufacturing company, biologists, a utility, a consulting engineering firm and a university research facility in order to benefit from the synergy of diverse disciplines. One of the primary objectives of the project was to advance the understanding of the issues involved to effectively improve the environmental compatibility of hydro plant equipment designs.

The approach was divided into four tasks. Task 1 investigated a broad range of environmental issues and how the issues differ throughout the country. From this overview, the team looked for common elements which characterize the problems and chose three families of design concepts addressing the groups of most significant problem elements for further investigation. The concept families address environmentally advanced Kaplan turbines designed for improved fish survivability; environmentally advanced Francis turbines designed for improved fish survivability; and aerating Francis turbines designed for increasing dissolved oxygen content in turbine discharges. Of the families chosen, Kaplan units are the most important for considerations of fish passage. However, low head Francis units are also important for fish passage at older projects in the eastern states and in the upper mid west. Designs to enhance dissolved oxygen in turbine discharges require consideration of medium-head Francis units in addition to low-head Francis, propeller and Kaplan turbines.

Task 2 addressed fish physiology and turbine physics. In this task, the team studied the state of available information, the mechanisms for injury and methods to predict injury and defined which design elements to address to improve fish survival at hydro sites. Characteristics of turbine types are defined. The importance of a turbine's geometry and operation on fish passage survival is presented. Misconceptions present in the literature derived from interpreting past experiments are pointed out. The concept of the zonal effectiveness of fish passage survival in turbines is introduced. The need for additional controlled experiments to further clarify the effects of turbine geometry and the associated flow conditions on injury mechanisms is discussed.

Task 3 investigated individual design elements needed for the refinement of the three families of design concepts defined in Task 1. Advanced computational fluid dynamic (CFD) tools for numerical flow simulation in turbines were used to quantify characteristics of flow and pressure fields within turbine water passageways. Improvements of the simulation tools are discussed and evaluated in light of their utility in improving the environmental design of hydraulic turbines. The issues associated with dissolved oxygen enhancement using turbine aeration are defined. The state of the art and recent advancements of this technology are reviewed. Key elements for applying turbine aeration to projects to improve aquatic habitat are discussed. A review of the procedures for testing of aerating turbines is presented.

Tasks 2 and 3 activities brought forth several conclusions. Turbine operation has a significant effect on fish survival during turbine passage. Controlled field test experiments and CFD calculations demonstrate that different zones of the turbine have significantly different effects on fish during passage. Zonal geometry and associated flow conditions are important. In planning tests to evaluate fish passage, zonal effect determination must be considered to adequately develop a survival estimate for the turbine. Advanced turbulence models in CFD investigations were demonstrated to more accurately correlate with measured flow fields. In the absence of cavitation, pressure effects on fish during turbine passage are not significant. Effects related to the state of pressure acclimation are significant. These effects relate more to project planning than to turbine design or operation. Incorporating capabilities for aeration into the turbine design can alleviate water quality problems stemming from low dissolved oxygen in hydropower releases. Depending on design conditions aerating turbines can increase the level of dissolved oxygen by over 5 mg/L.

Multiple areas for additional investigation were identified. Fish paths within intakes and turbines are not well understood. Additional testing is required to develop accurate indices of forces, pressure differentials, or other deterministic quantities that can be related to fish damage mechanisms in more detail. Calculation of flow fields can be performed. However, a means of calculating the resulting forces on the fish and the effect of the loads on fish survival is needed to advance the state of the art.

Task 4 assembled the results of Task 2 and Task 3 into three families of design concepts to address the most significant issues defined in Task 1. Significantly, the team pointed out that improvements in fish passage survival are achievable. The team provided design concepts which can be, and in some instances are being, implemented at today's existing hydro projects.

Finally, the team developed a set of recommendations for future work needed to improve the knowledge of the processes involved in inflicting injury to fish and pointed out the need for additional testing in controlled laboratory experiments and at existing hydro plants and at those currently being rehabilitated. They pointed out that none of the passage routes is 100% safe for fish and that recent experimental data and reanalysis of historical data **do not** support certain historical hypotheses. Instead, they show that (1) survival is not necessarily maximized at peak turbine operating efficiency, (2) survival is not necessarily higher for fish entrained near the hub, and (3) survival is not necessarily lower for unguided fish at turbines equipped with fish guidance screens. The report demonstrates that complex interacting mechanisms occur within the turbine and that fish passage survival depends on the turbine geometry, its operation and the location of the fish in the water column. In addition, they concluded that the effectiveness of turbine designs should be evaluated against "best of class" benchmarks. This would help in setting realistic, achievable goals in fish survival improvement for each turbine type. Effects of turbine modifications on fish survival can be evaluated using consistent test protocols and "comparative" benchmarking.

While the fundamental focus of the solutions developed is in the environmental arena, many of the issues addressed to improve the environmental compatibility also can improve plant efficiency thereby improving project economics and reducing the need for replacement energy generation from non-renewable sources. In addition, improvements reducing cavitation and vibration will result in lowered maintenance requirements for operators implementing the designs.

2.0 INTRODUCTION

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2.1 GOALS AND OBJECTIVES

In the spirit of the DOE's mission for the Advanced Hydro Turbine System Program, Voith Hydro, Normandeau Associates, TVA, Harza Engineering Company and the School of Civil and Environmental Engineering of Georgia Institute of Technology worked together on a goal to define a family of environmentally advanced hydro turbine design concepts to meet the objective of improving hydropower's impact on the environment. Another goal of the project was to separate fact from fiction in understanding the issues involved to effectively improve the environmental compatibility of hydro plant equipment. To meet this goal a project objective to improve the understanding of the technical and environmental issues involved was established, in particular relating to fish survival as a result of their passage through hydro power sites. In addition, an objective to point out needs and provide recommendations for further research was defined.

While the fundamental objectives of the solutions sought were in the environmental arena, it was envisioned that many of the issues addressed to improve the environmental compatibility could also improve plant efficiency thereby improving project economics and reduce the need for replacement energy generation from non-renewable sources. In addition, improvements sought were expected to reduce cavitation and vibration which would result in lowered maintenance requirements for operating utilities implementing the designs.

In developing the design concepts, the team remained cognizant of the following:

1. Design features of existing turbines can be modified to make significant improvements in their environmental compatibility. Hydro turbine plants contain more than 92,000 MW of installed capacity at over 2300 sites in the US alone. This large installed base creates an opportunity to significantly address the environmental improvement issues through upgrade and rehabilitation of existing units.

2. Design features of new turbines can be chosen to make them more environmentally compatible. However, few new hydro installations are currently envisioned.

3. Conventional thinking, with respect to turbine design economics, was not used as a limitation. While turbine performance is still a very important factor in evaluating the benefits of different designs, the focus for design concept development was on environmental enhancement. When environmental cost/benefit values are used in the economic evaluation of the project, unconventional environmentally enhanced design solutions will be seen as cost effective.

4. Understanding of the behavior of fish in turbine flow fields and of the fluid and mechanical mechanisms involved in injuring fish in their passage through turbines is key to the development of design concepts for producing environmentally enhanced designs. This understanding will come from investigations using advanced technology for simulation of flow fields within turbines and from analysis of carefully designed field and laboratory testing.

2.2 APPROACH TO THE PROJECT

The approach teamed a turbine design and manufacturing company, biologists, a utility, a consulting engineering firm and a university research facility, in order to benefit from the synergy of diverse disciplines. The knowledge of the contributors based on work done on related projects funded elsewhere (previous and concurrent activities) was combined with that developed from work done on this project to formulate the background, interpret experiments and conduct specific analyses. The design concepts presented are a combination of concepts developed on related projects funded elsewhere (both previous and concurrent) and those developed as a result of the work done on this contract. Some of the design concepts which were derived based on non DOE funding are covered by patents or are the subject of pending patent applications.

The approach was divided into four tasks. Task 1 investigated a broad range of environmental issues and how the issues differ throughout the country. From this overview, the team looked for common elements which characterize the problems and chose three families of design concepts addressing the groups of most significant problem elements for further investigation. Task 2 addressed fish physiology and turbine physics. During this task, the team studied the state of available information, the mechanisms for injury, injury prediction methods and defined what design elements to address to improve fish survival at hydro sites. Task 3 investigated individual design elements needed for the refinement of the three concepts defined in Task 1. Task 4 then assembled the results of Task 2 and Task 3 into three design concept families to address the most significant issues defined in Task 1. Details of the four tasks were as follows:

Task 1: Categorization of Environmental Issues and Selection of Concepts for Further Detailed Conceptual Design

Biological issues related to environmental compatibility (EC) improvements are geographically dependent. In the Pacific Northwest, the EC issues are dominated by migratory fish and their survival in passing through turbines. In the Southeast, the EC issues are driven by resident fish and dissolved gas content. In the Northeast, migratory fish and resident fish survivability when passing through turbines are the principal factors. In all regions, maintaining minimum stream flows and reducing oil and grease pollution play a role.

In Task 1, a broad summary of the principal issues addressing the environmental compatibility of turbines and power plants in all regions of the United States was made. The principal issues were related to fish passage survival through hydropower sites and the effect of hydropower sites on aquatic habitat. In the Northwest region of the country, Kaplan turbines and fish passage survival predominated. In the upper mid west region and the northern Atlantic coast region, Francis turbines and fish passage survival issues were dominant. In the Southeast region, issues associated with low levels of dissolved oxygen in turbine releases were dominant. In all areas, issues with respect to minimum stream flows existed.

The above, as well as additional concepts developed from the Task 1 activities, were evaluated with the help of the DOE AHT's project review committee. From those considered, three families of design concepts best addressing the hydropower industry's needs were selected for further design element development in Task 3. They were an advanced Kaplan turbine focused on fish passage survival improvements; an advanced Francis turbine focused on fish passage survival improvements; and an advanced Francis turbine stressing improvements in the levels of dissolved oxygen in the discharge water.

Task 2: Fish Physiology and Hydropower Physics

The team reviewed the data available in the literature and screened selected data from the data set for detailed investigation. Injury mechanisms related to fish passage through hydro sites were quantified. Fish passage survival models were developed based on simplified models of the turbine geometry and performance characteristics and further evaluated with the help of sophisticated flow analysis tools. The mechanisms and models were then evaluated in light of the screened data.

In the process of the above, the opportunity to gather some additional data in conjunction with a planned site test at Wanapum dam was used and the test plan was expanded to gather additional data. Analysis of the test results provided further insight into importance of the zonal characteristics of the turbine geometry and associated flow fields.

Data associated with fish passage survival in turbine bypasses were also reviewed to benchmark these alternative routes.

A key facet in developing an environmentally compatible design relating to fish survivability involves the development of a clear understanding of the physiology of the fish and how the fish behave as they enter a hydro project. More specifically, the following questions, among others, were addressed:

1. What pressure, velocity and acoustic gradients influence fish behavior?
2. What physiological stresses and turbine features are responsible for injuring and killing fish?
    - Decompression
    - Strike
    - Gas supersaturation (bends)
    - Velocity shear/turbulence
    - Cavitation
3. How are different species and size of fish affected?
4. How do plant civil design, head and flow impact fish behavior and mortality?

A survey of available data and discussions between team members to share insights were used to discover features and operations that have proven to be relatively fish friendly. Additional laboratory and field tests were identified that will enable the designer to formalize features that will produce a hydraulic environment compatible with high fish survival.

Task 3: In-depth Investigation of Selected Design Elements

Based on the results of Task 1 and Task 2, selected design studies were conducted to gain a technical understanding of the issues required to achieve the design objectives of the three selected families of design concepts.

Advanced methods of Computational Fluid Dynamics (CFD) were used by Voith Hydro to analyze and evaluate elements of existing designs to provide insight leading to the conceptual designs. The methods were used to calculate velocity and pressure fields to: i) calculate the pressure gradients experienced by fish passing through the turbine; ii) identify regions where cavitation would occur; iii) identify the presence of vortices, regions of flow reversal, and regions of large velocity gradients; iv) identify loss zones; and v) illustrate flow streamlines, among others. Investigations of limitations of existing CFD turbulence modeling were conducted by Georgia Tech. Improved methods were tested numerically to evaluate their impact.

A definition of critical issues relating to turbine aeration was developed by TVA. The issues associated with dissolved oxygen enhancement using turbine aeration are defined. The state of the art and recent advancements of this technology are reviewed. Key elements for applying turbine aeration to projects to improve aquatic habitat are discussed. A review of the procedures for testing of aerating turbines is presented.

Task 4: Development of Conceptual Designs

Because each hydro plant is custom designed to adapt to its unique site and operational requirements, a single design for each of the three topics selected in Task 1 was not addressed. Instead, based on the results of Tasks 1, 2 and 3, three sets of design concepts were developed which can be implemented in the context of the unique requirements of each hydropower plant. The sets of concepts address:

Advanced environmentally friendly Kaplan turbines.
Advanced environmentally friendly Francis turbines.
Advanced environmentally friendly aerating Francis turbines.

2.3 TEAM STRUCTURE

To accomplish the proposed tasks, a multi-disciplinary team was formed to address the issues. The team consisted of the following organizations:

1. Voith Hydro, Inc.
2. Tennessee Valley Authority (TVA)
3. Harza Engineering Company
4. Normandeau Associates
5. The School of Environmental and Civil Engineering of Georgia Institute of Technology

The team brought to the project tremendous synergy benefits from the diverse background of each. It is important to note that the team consisted of a manufacturer, utility, consulting engineers, an environmental service group and a university. Voith Hydro served as the prime contractor and team leader. The other organizations served as subcontractors.

2.4 ORGANIZATION OF THE REPORT

This report is organized following the Tasks of the Project. Section 3.0 discusses Task 1 activities. Section 4.0 discusses Task 2 activities. Section 5.0 presents the results of Task 3 studies including those associated with CFD investigations of turbine components, those associated with development of advanced CFD capabilities, and those associated with the enhancement of dissolved oxygen levels in water passing through turbines. Section 6 presents design concept families. Section 6.4 reporting on a third concept family related to aerating Francis turbines will be supplied as a report supplement. Section 7 presents a summary of conclusions derived from the work of all Sections. Section 8 presents recommendations for future work. Section 9 is reserved for a future supplement which will report on the use of advanced CFD and a "virtual fish" to evaluate the 4 conditions tested experimentally by fish injection at Wanapum dam (described in Section 4.4.6). An appendix (Section 10) contains background material from all sections.

3.0 TASK 1 REPORT -- REVIEW OF THE ISSUES

3.0 TASK 1 REPORT– REVIEW OF THE ISSUES

3.1 INTRODUCTION

The objective of Task 1 was to review environmental issues and select concepts having the greatest impact on improving the environmental compatibility of turbines for further detailed conceptual design. Issues identified as addressable through turbine design included fish passage through turbines, dissolved gasses in turbine discharge, and minimum flow downstream of hydroelectric stations. Experience of the Voith team was used to define the geographic distribution of concerns about these issues. The cumulative experience of the Technical Committee was solicited by mail. Design characteristics of turbines associated with identified issues and regions were established by queries of an extensive database. Initial results were presented at the Design Review Meeting on March 6-7, 1996. Additional queries followed receipt of comments from the Technical Committee.

Harza Engineering Company compiled a database from the National Inventory of Dams of the U. S. Army Corps of Engineers and the Federal Energy Regulatory Commission database of licensed hydroelectric projects. The database includes information on 2555 dams associated with hydroelectric projects. The dam database includes:

Name
Nearest City
Height
Owner
State
Max. Discharge
Latitude
Purpose
Max. Storage
Longitude
Age
Normal Storage
River
Length

Harza also has a series of manufacturers' turbine databases that includes more than 6,000 entries, including some overseas. The Voith-Allis Chalmers dataset is the largest, with more than 850 entries in the U.S. Turbine data were also provided by Neyrpic, GE Canada, Mitsubishi, Fuji, Hitachi, Toshiba, Kvaerner, Sulzer and Voest Alpine. Turbines were designated by the following parameters:

Hydraulic Type
Axial (Kaplan, propeller)
Diagonal (Deriaz)
Radial (Francis)
Impulse (Pelton)
Cross-Flow (Ossberger, Banzi)

Hydromachine Control
Axial (Kaplan, propeller runner blades only)
Dual control
No control

Hydromachine
Turbine
Pump Turbine

No. of Jets
(Pelton only)

Runner
Single
Tandem
Multistage

No. of Stages
(or no. of Pelton runners)

Arrangement
Bulb
Pit
S-type
Straflo (Harza)
Tube
Conventional

Drive
Direct
Spur gear
Bevel gear
Chain
Belt
Plenetary gear

Orientation
Horizontal
Vertical
Inclined

Spiral Case
Concrete, semispiral
Steel
Flume

Each project was identified by name and units were specified by:

Year	No. Units	Rpm	Diameter	Turbine	Pump
				Head (m)	Head (m)
				Discharge (m^3/s)	Discharge (m^3/s)
				Power (kW)	Power (kW)

Original goals of the project team were to develop turbine design concepts to address questions of fish passage, dissolved gases and minimum flow. Filters and sorting on various fields were applied to the database to screen for key issues. The filters were used to associate turbine and plant parameters with fish related problems such as fish passage, dissolved gases, and minimum streamflow. The turbine and plant parameters that were identified as related to these problems and could be obtained from the database were: location (state and region) of the plants, turbine type, head, turbine and plant discharges, turbine output, turbine size, plant output, and plant factor. Filters were developed in the form of specifying states composing regions that defined the geographic extent of a question or the regionally specific manner in which a question was addressed. After presentation of the results of filtering the database at the first Design Review Meeting, the team was directed to drop the minimum flow objective and develop design concepts for a fish-friendly Francis turbine. Additional queries were made of the turbine database to determine the size distribution of Francis turbines by region. In most cases, turbine diameter was missing from the database, but rated discharge was nearly always listed. From the cases where both parameters were quantified, the equation

$$D = (Q/6.5)^{0.5}$$

where D = runner diameter (m) and
Q = discharge (m^3/s)

was fit and applied to the cases where diameter was missing for a conventional Francis turbine.

The dissolved oxygen issue was researched by the Tennessee Valley Authority. Depending on site-specific conditions, one or more of the factors listed in Table 3.1 (Ruane and Hauser 1991) may affect DO in turbine discharges.

Category	Items
Physical Design Factors	Reservoir volume Reservoir surface area Reservoir depth Discharge capacity of turbines Location and depth of outlets
Environmental Factors	Meteorology (e.g. air temperature and rainfall), Hydrology (e.g., mean annual flow rate) Inflow water temperature
Watershed Factors	Size Type of land use Point and nonpoint wastewater discharges Natural loadings of organic substances and nutrients
Operational Factors	Schedule for hydropower releases Schedule of releases for upstream projects

Table 3.1 Factors Affecting Tailrace Dissolved Oxygen Concentrations

To determine the dominant factors, correlation studies are required using data from a large sample of projects. The data requirements include tailwater DO as well as the items given in Table 3.1. The results for one such study were presented by EPRI (1990). Reservoirs were identified as more than likely to encounter periods when the discharge contains less than 5 mg/L DO if they had the following characteristics:

Depth at dam	> 50 feet,
Power capacity	> 10 MW,
Reservoir volume	> 50,000 acre-feet,
Densimetric Froude Number F_d	< 7, and
Retention Time V_A,Q	> 10 days,

where $$F_d = \frac{1.952 LQ(V_1 - V_A)}{V_A^{2}}$$ and

L = Reservoir length (miles),
Q = Average annual inflow (CFS),
V_A = Average annual volume (1000 CFS-days), and
V_1 = 1-foot above average storage (1000 CFS-days).

and 1 ft = 0.305m, 1 acre-ft = 1,233 m^3, and 1 cfs = 2.830 x 10^{-2} m^3/s. At this time, these conditions represent the only filter for identifying projects that are likely to encounter low DO. It should be emphasized that EPRI (1990) does not provide a reference from which these conditions are recommended. Hence, the "accuracy" of this filter for identifying projects with low DO is unknown.

A measure of the extent of low DO throughout the country can currently be obtained from two sources of information: statistical analyses by others who have obtained DO data for many US hydro projects, and water quality summaries for dams managed by the US Army Corps of Engineers (USACE), Tennessee Valley Authority (TVA), and US Bureau of Reclamation (USBR).

3.2 **DISCUSSION**

The Harza dataset included 64 GW of capacity among 1600 projects identified by head, discharge and turbine type. FERC (1992) reported a total of 92 GW capacity in the private and public utilities of the U.S. Known projects that were not included in the database because of incomplete data were less than 1 MW capacity. Examination of the entire data set showed that most generating capacity was installed in Francis turbines at medium and low head (Tables 3.2 and 3.3). However, most flow passed through axial turbines at low and very low head.

	Axial	Francis	Pelton	Ossberger
Capacity (MW)	20,561	43,859	2,959	124
Design discharge (m^3/s)	123,998	95,829	962	886

Table 3.2 Distribution of Capacity and Design Flow by Turbine Type

	very low <10 m	low 10-50 m	medium 50-150 m	high > 150 m
Axial units	24,475	99,277	247	0
Francis units	5,523	50,995	35,721	3,590

Table 3.3 Total Design Discharge (m^3/s) as a Function of Head (m)

The fish passage issue has been the impetus for studies dealing with anadromous salmon species on the West Coast, anadromous Atlantic salmon and American shad on the East Coast, and freshwater resident species in the Upper Midwest and other inland sites (Eicher and Associates 1987, Stone & Webster 1992). Concerns about the effects of dams on anadromous fishes date back to the Industrial Revolution, but most studies on resident species have been conducted since 1990. Low head Axial units typify turbines associated with Pacific salmon in California, Oregon, Washington and Idaho (Tables 3.4 and 3.5). Francis units at low and medium head were also important on the East Coast states from Maine to Georgia and in the New York and the Upper Midwest (Michigan, Wisconsin and Minnesota).

	Turbine Type			
	Axial	Francis	Pelton	Ossberger
West Coast				
Total MW	11,333	22,581	2,624	7
Total m^3/s	52,234	38,827	842	53
East Coast				
Total MW	1,950	5,619	33	30
Total m^3/s	11,908	19,975	12	199
Upper Midwest				
Total MW	1,386	3,229	0	29
Total m^3/s	10,307	9,888	0	395

Table 3.4 Total Turbine Capacity (MW) and Design Discharges (m^3/s) at Hydroelectric Projects in Regions with Notable Fish Passage Issues

		very low <10 m	low 10-50 m	medium 50-150 m	high > 150 m
West Coast	Axial units	1,950	50,046	237	0
	Francis units	27	16,527	20,522	1,750
East Coast	Axial units	3,046	8,859	3	0
	Francis units	2,556	13,891	3,394	124
NY & Upper Midwest	Axial units	4,149	6,151	7	0
	Francis units	2,338	4,877	2,657	17

Table 3.5 Total Design Discharge (m^3/s) as a Function of Head (m) at Hydroelectric Projects in Regions with Notable Fish Passage Issues

Axial turbines accounted for 31 percent of the total hydro generation capacity and 57% of the design discharge for the West Coast. Ninety-six percent of the West Coast Axial design discharge was through low (10-50 m (33-164 ft.)) head units. Low head Axial units accounted for 28 percent and 30 percent of the design discharge on the East Coast and in New York and the Upper Midwest, respectively. The hydro generation capacity for the East Coast and Upper Midwest was mostly (74 and 70 percent, respectively) by Francis turbines. Low head Francis units were 43 percent of the design discharge database for the East Coast and 24 percent for the Upper Midwest.

Francis turbine size data were sorted for Pacific Northwest and New York and Upper Midwest States to address the fish passage issue (Table 3.6). There was a nearly even distribution of turbines across size categories in the Pacific Northwest. Size categories of less than 2 m (6.6 ft.), 2 m to 4 m (13.1 ft.), and greater than 6 m (19.7 ft.) each accounted for 27 to 29 percent of the number of turbines. Turbines tended to be smaller in the Upper Midwest. Most (55 percent) of those turbines had diameters of 2 to 4 m, and 23 percent were smaller than 2 m.

Diameter (m)	Pacific Northwest States	New York and Upper Midwest States
<2	91	73
2-4	97	174
4-6	51	43
>6	93	27
no data	3	1

Table 3.6 Size Distribution of Francis Turbines in Pacific Northwest and Upper Midwest States

Statistical analyses of the water quality of hydro releases were summarized by Cada et al. (1981, 1983) and EPA (1989). Cada et al. (1981, 1983) compiled information for water quality downstream of hydro projects from two databases, the USACE National Hydropower Study (NHS), and the National Water Data Storage and Retrieval System (WATSTORE). At the time of the study, NHS contained descriptive information for about 15,300 dams. WATSTORE contained water quality measurements for about 220,000 stations. Searches of the databases paired hydro projects with water quality stations that were located within three miles downstream of the dam. All projects in the NHS database having more than two DO measurements from a tailrace station in the WATSTORE database were evaluated statistically to determine the probability of noncompliance (PNC). The PNC was defined as the probability that concentration of dissolved oxygen downstream of the project will be less than 5 mg/L.

The data were evaluated based on a regional division of the 48 contiguous states (Table 3.7). The analyses included two groups of hydro projects and two seasons, those with capacity less than 30 MW and those with capacity greater than 30 MW, for summer (July-October) and winter (other months). The frequency of occurrence of low DO, and hence the mean PNC, is generally greater for the summer. This is due to warmer temperatures, which cause thermal stratification in the reservoirs. This process inhibits reservoir mixing and causes hypolimnetic oxygen depletion. Since these are the conditions that usually create the need for low DO improvements, the results summarized herein will focus only on the summer. For these months, the mean PNC for each group of hydro projects is given in Tables 3.8 and 3.9, respectively.

Region	States
Great Basin States	Arizona, Nevada, New Mexico, Utah
Great Plains States	Iowa, Kansas, Nebraska, North Dakota, Oklahoma, South Dakota, Texas
Lake States	Michigan, Minnesota, Wisconsin
Northeast States	Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont
Ohio Valley States	Delaware, Illinois, Indiana, Kentucky, Maryland, Missouri, Ohio, Tennessee, Virginia, West Virginia
Pacific Coast States	California, Oregon, Washington
Rocky Mountain States	Colorado, Idaho, Montana, Wyoming
Southeast States	Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, South Carolina

Table 3.7 Regional Division of US for Analysis of Low DO

EPA (1989) repeated the study of Cada et al. (1981, 1983) using available data from a sample of 40 hydro projects randomly selected from the USACE National Inventory of Dams Database. At the time of the study this database included 68,155 dams. In the EPA work, the 40 projects were selected from a subset of the USACE database defined as those sites with over 100 kilowatts of installed power and over 12,340,000 m^3 (10,000 acre-feet) of reservoir volume. This subset included 424 hydro projects. Dissolved Oxygen data for each of the 40 sites, if any, were obtained from EPA's STORET data repository. Results of the EPA study, again for the summer months, are also shown in Tables 3.8 and 3.9.

Region	Cada et al. (1981, 1983)		EPA (1989)		Observed
	No. Sites	Mean PNC (%)	No. Sites	Mean PNC (%)	Range (%)
Great Basin	3	37.3	no data	-	no data - 37.3
Great Plains	1	0.0	1	0.0	0.0 - ?
Lake	5	4.3	4	12.3	4.3 - 12.3
Northeast	15	6.6	no data	-	no data - 6.6
Ohio Valley	3	11.1	3	22.0	11.1 - 22.0
Pacific Coast	7	0.3	no data	-	no data - 0.3
Rocky Mountain	9	2.7	no data	-	no data - 2.7
Southeast	17	13.1	2	19.0	13.1 - 19.0

Table 3.8 Mean Summer PNC for Projects < 30 MW

Region	Cada et al. (1981, 1983)		EPA (1989)		Observed
	No. Sites	Mean PNC (%)	No. Sites	Mean PNC (%)	Range (%)
Great Basin	3	0.4	no data	-	no data - 0.4
Great Plains	6	18.2	no data	-	no data - 18.2
Lake	no data	-	no data	-	no data
Northeast	3	14.4	no data	-	no data - 14.4
Ohio Valley	16	40.4	5	56.0	40.4 - 56.0
Pacific Coast	19	3.9	4	5.3	3.9 - 5.3
Rocky Mountain	6	5.2	2	0.0	0.0 - 5.2
Southeast	18	30.8	2	17.0	17.0 - 30.8

Table 3.9 Mean Summer PNC for Projects > 30 MW

No region is free of low DO episodes. PNC had some value above zero in all regions among large or small projects. Mean summer PNC's tend to be higher for large scale facilities (> 30 MW), indicating that low DO occurs more frequently for these sites. The mean summer PNC tends to be higher for the Southeast and Ohio Valley, indicating that low DO occurs more frequently in these areas. For small scale facilities (< 30 MW), the same was true for the Great Basin, but it was represented by only three projects. High PNC occurs in these regions because summers are longer and hotter, and therefore the magnitude and duration of thermal stratification in reservoirs are higher. For large scale facilities (> 30 MW), the mean summer PNC's for the Great Plains and Northeast are less than half of that for the Ohio Valley and Southeast, respectively, but are still considered significant.

Cada et al. (1981, 1983) urged caution in reviewing the results summarized in Tables 3.8 and 3.9. For some regions, not enough hydro projects have data for tailwater DO to obtain reliable statistics. At some sites the measurements are infrequent. Watershed and meteorological data that affect DO are very limited. Improved predictions of low DO may result by including not only power capacity but also retention time, reservoir depth, outlet location, inflow temperature, and size and character of watershed in the analyses.

Kennedy and Gaugush (1987) summarized the results of an analysis of USACE hydropower projects in an exhibit that showed sites in the Southeast, Ohio Valley, Great Plains, Rocky Mountain, and Pacific Coast regions (Table 3.10). Their data generally supported the previous indication that low DO occurs more frequently in the Ohio Valley and Southeast, and less frequently in the Great Plains, Pacific Coast, and Rocky Mountains. This exhibit was also presented by EPA (1989) and EPRI (1990). For the Ohio Valley, the overall fraction of sites with DO problems was slightly higher than that suggested by the mean summer PNC in Table 3.9. For the Southeast, the overall fraction of sites with DO problems is more than twice that suggested by the mean summer PNC in Table 3.9. However, data by Kennedy and Gaugush (1987) were not selected at random and may contain bias towards USACE projects with DO problems.

Detailed DO measurements for TVA hydro projects were reported by TVA (1990). Nineteen out of 29 projects located in the Ohio Valley and Southeast were reported as having tailwater DO < 5 mg/L on the average of at least 3 weeks per year. Table 3.11 gives the percent of low DO projects based on the total number of TVA hydropower sites in each of these regions. For both regions, the overall fraction of sites with low DO is higher than that suggested by the mean summer PNC's in Table 3.9. The data in Table 3.11 obviously is biased towards TVA projects, and probably toward larger projects and reservoirs as well, but again supports the previous indication that low DO occurs more frequently in the Ohio Valley and Southeast.

Region	No. Of Sites	Sites With Low DO	
		No.	Percent
Great Plains	5	0	0.0
Ohio Valley	14	9	64.3
Pacific Coast	18	0	0.0
Rocky Mountain	4	0	0.0
Southeast	20	15	75.0
Total	61	24	39.3

Table 3.10 USACE Hydropower Projects Having at Least Minor DO Problems

Region	Total No. Sites	Sites With Low DO	
		No.	Percent
Ohio Valley	23	16	69.6
Southeast	6	3	50.0
Total	29	19	55.2

Table 3.11 TVA Hydropower Projects with
DO < 5 mg/L at Least 3 Weeks Per Year on Average

EPA (1989) presented the results of a water quality survey for 250 of 349 USBR power and nonpower water resources projects. These include sites in the Great Plains, Rocky Mountains, Great Basin, and Pacific Coast states. About 54% of the surveys reported that no data were available to assess low DO problems (i.e., 134 of 250 projects). Assuming there are no DO problems at the "no data" sites (i.e., data are collected only when a problem exists), only about 4% of the reported USBR projects would contain low tailwater DO as at least an intermittent problem. At this time, 4% is the best estimate available for the fraction of USBR hydropower projects that contain low tailwater DO. This is based on the unsubstantiated assumption that sites with low DO are uniformly distributed among all the different project types. These results, however, support the general indication in Tables 3.8 and 3.9 that low DO is not as frequently observed in the western regions of the US, especially the Pacific Coast and Rocky Mountain states.

The Harza database was sorted by the same regions used in the dissolved oxygen analyses and queried to determine the turbine types that have been most commonly associated with low DO. Low head Axial units pass most of the flow in Southeast and Ohio Valley states where DO problems are well documented (Tables 3.12 and 3.13).

	Axial	Francis	Pelton	Ossberger
Southeast				
Total MW	4,984	6,206	23	98
Total m^3/s	37,618	18,554	7	545
Ohio Valley				
Total MW	794	2,066	10	0
Total m^3/s	7,926	6,942	5	0
Entire Data Set				
Total MW	20,561	43,859	2,959	124
Total m^3/s	123,998	95,829	962	886

Table 3.12 Total Turbine Capacity (MW) and Design Discharge (m^3/s) at Hydroelectric Projects in Regions with Notable Dissolved Oxygen Issues

	Very Low <10 m	Low 10-50 m	Medium 50-150 m	High > 150 m
Southeast				
Axial units	10,639	26,972	7	0
Francis units	1,187	12,689	4,555	124
Ohio Valley				
Axial units	3,376	4,550	0	0
Francis units	257	4,727	1,958	0
Entire Data Set				
Axial units	24,475	99,277	247	0
Francis units	5,523	50,995	35,721	3,590

Table 3.13 Total Design Discharge (m^3/s) as a Function of Head at Hydroelectric Projects in Regions with Notable Dissolved Oxygen Issues

Low head Axial units accounted for 48 percent of the installed design discharge in the Southeast: very low head Axials, 19 percent; and low head Francis units 22 percent. In the Ohio Valley, low head Axials accounted for 31 percent of the installed design discharge; very low head Axials, 23 percent; and low head Francis units 32 percent.

Storage projects are more likely than run-of-river projects to suffer DO problems. Storage reservoirs are more likely to stratify in the summer and have low DO in their deeper layers because storage reservoirs tend to be larger and deeper and have much longer hydraulic residence times than run-of-river facilities. Data for the Southeast and Ohio Valley regions were also sorted by plant factor, where

$$plant\ factor = \frac{yearly\ kWh\ produced}{plant\ capacity \times hours\ per\ year}$$

Plant factor may have values from 0, representing no generation, to slightly greater than 1, representing continuous operation of all units with actual output slightly above nameplate capacity. Run-of-river projects tend to have high plant factors. At these facilities, dissolved oxygen problems, when they occur, tend to be due to causes unrelated to hydro operation. Storage projects tend to have lower plant factors because discharge and generation vary over daily or seasonal scales. Dissolved oxygen problems at these sites may be due to thermal stratification of the reservoir and oxygen consumption by the deep (hypolimnetic) aquatic community as well as the internal factors affecting run-of-river projects.

Most of the flow at projects with low plant factors (where there is some idle capacity most of the time) passes through Francis units. Nationally, the distribution of plant factors was:

	< 0.33	0.33-0.66	> 0.66
Total MW	13,580	43,306	10,707
Total m^3/s	37,041	137,323	47,311

Most of the generating capacity and design flow through plants with plant factors less than 0.33 was through Francis units:

	Axial	Francis	Pelton	Ossberger
Total MW	2,852	10,150	564	15
Total m^3/s	15,394	21,353	188	106

In the Southeast and Ohio Valley states, low plant factors were associated with a tendency toward larger turbines (Table 3.15). In the Southeast, 53% of the turbines with low plant factors were larger than 6 m in diameter; in the Ohio Valley, 68%. This is consistent with a peaking mode of operation that would discharge large volumes of water in a short period of time. Most turbines with plant factors of 0.33 to 0.66 were 2 to 6 m in diameter.

Diameter (m)	Number of Turbines in Southeast		Number of Turbines in Ohio Valley States	
	Plant Factor (kWh produced /(plant capacity x hours per year))			
	0 - 0.33	0.34 - 0.66	0 - 0.33	0.34 - 0.66
<2	8	28	2	2
2-4	18	69	4	18
4-6	38	62	4	13
>6	71	25	21	14

Table 3.15 Turbines in Southeast and Ohio Valley States Sorted by Diameter and Plant Factor

3.3 SUMMARY OF TASK 1 FINDINGS

Most hydroelectric generating capacity in the United States is in low and medium head Francis units, but most flow (and therefore possibly fish) passes through low head Axial units. Low head Francis accounted for 23 percent of the installed design discharge. Because most early hydropower development was in the East, there are more low head Francis units in the eastern and central states than there are in western states.

The fish passage issue has been the impetus for studies dealing with anadromous salmon species on the West Coast, anadromous Atlantic salmon and American shad on the East Coast, and freshwater resident species in the Upper Midwest and other inland sites. Low head Axial units typify turbines associated with Pacific salmon in California, Oregon, Washington and Idaho. Francis units at low and medium head are also important on the East Coast and Upper Midwest. There is a nearly even size distribution of turbines in the Pacific Northwest. Size categories of less than 2 m, 2 to 4 m and greater than 6 m each account for 27 to 29 percent of the number of turbines. Turbines tend to be smaller in the Upper Midwest. Most (55 percent) of those turbines have diameters of 2 to 4 m, and 23 percent are smaller than 2 m.

Low levels of dissolved oxygen in hydropower discharges are most common the Southeast and Ohio Valley states. Probabilities of low DO episodes for the Great Plains and Northeast are less than half of that for the Ohio Valley and Southeast, but are still considered significant. Low DO occurs less frequently in the Great Plains, Pacific Coast, and Rocky Mountains. Low head Axial units accounted for 48 percent of the installed design discharge in the Southeast: very low head Axials, 19 percent; and low head Francis units 22 percent. In the Ohio Valley, low head Axials accounted for 31 percent of the installed design discharge; very low head Axials, 23 percent; and low head Francis units 32 percent.

TVA demonstrated that minimum flow and dissolved oxygen problems are most common at projects with plant factors below 0.35. About 80 percent of the capacity and 2/3 of the flow through projects with low plant factor is through Francis units. In the Southeast, 53% of the turbines with low plant factors were larger than 6 m in diameter; in the Ohio Valley, 68%. This is consistent with a peaking mode of operation that would discharge large volumes of water in a short period of time. Most turbines with plant factors of 0.33 to 0.66 were 2 to 6 m in diameter.

Based on interaction with the Technical Committee, these issues were selected for further study and for development of design concepts for environmental compatibility enhancement. The three concepts were:

1. Large Axial turbines characteristic of those on the Columbia River in the Pacific Northwest where fish passage survival is the dominant issue.
2. Medium size Francis turbines characteristic of the Upper Midwest and Atlantic coast where fish passage survival is of dominant interest.
3. Medium to large size Francis turbines in Southeast and Ohio when low D.O. in turbine discharges in summer months is of dominant interest.

3.4 REFERENCES

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4.0 TASK 2 REPORT -- BIOLOGICAL ISSUES & TURBINE DESIGN AND OPERATIONAL CONSIDERATIONS

4.0 TASK 2 REPORT -- BIOLOGICAL ISSUES & TURBINE DESIGN AND OPERATIONAL CONSIDERATIONS

4.1 INTRODUCTION

Fish traveling downstream encounter three major exit routes at hydro dams: turbines, spillways/sluices, or bypasses. A successful passage through any of these routes is of importance, particularly for emigrating juveniles of migratory fish, for maintenance and enhancement of adult populations. Fish passage through sluices, spillways, and bypasses generally has been considered a benign process; survival rates have been assumed to be 98% in the Pacific Northwest (EPRI 1992). A survival rate of 85 to 89% in passage through Kaplan type turbines has been generally assumed for juvenile salmonids in the Pacific Northwest (EPRI 1992). However, turbine passage survival rates have been reported to be as low as 18% for young clupeids in passage through Kaplan type turbines (Taylor and Kynard 1985). It is not clear, however, whether the reported survival rates represent immediate (direct) effects of turbine passage or include the indirect effects as well. Thus, it is imperative that results of studies that may be useable for developing biological criteria for turbine design modifications be separated for identification of important biological issues. Where information is lacking from field studies laboratory data may be gleaned to increase our understanding of threshold values of factors that affect injury/mortality rates. Mathur *et al.* (1996a) have suggested that quantification of direct effects of passage has practical importance in improving turbine design, as they reflect the effects embodied in turbine geometries and hydraulics. As an example, the turbine replacement program (design, model testing, and installation of structural modifications) undertaken by the Public Utility District No. 1 of Chelan County at Rocky Reach Dam on the Columbia River to improve fish passage survival through the new turbines, utilized the data on direct effects (RMC 1994a; RMC and Skalski 1994a,b). These data guided a design effort to improve fish passage survival for a turbine rehabilitation project. The modified design included elimination of the gaps between the hub and leading edge of the runner blades, an area which was believed to inflict higher rate of injury/mortality to entrained fish.

This section provides (1) a brief review of historical literature with some statistical analysis of those data, detailed reviews have been provided elsewhere (Bell 1981; Monten 1985; Eicher Associates 1987; EPRI 1992); (2) a summary of some of the most recent data on fish survival as a function of physical and hydraulic characteristics of turbines, operating efficiency, and fish size; (3) a review of sources of injury/mortality in passage through Kaplan, propeller (fixed blade tilt), and Francis turbines; (4) the development of new leading blade edge strike prediction method; (5) a description of mechanisms of fish injury due to mechanical, fluid induced, and pressure reduction; and (6) in-depth analysis of controlled experiments conducted recently at large turbine in the Pacific northwest. Most of our emphasis is placed on the above types of turbines (Figures 10.2-1 through 10.2-6) because some recent studies provide reliable estimates of direct effects of turbine passage and also these turbines are dominant in the United States (see Section 3.0). However, to provide a benchmark for survival through turbines, available data from sluices, bypasses, or spillways are also presented. The latter structures, though devoid of moving parts, may expose fish to similar type of fluid-induced risks, thus provide some idea on quantification of their effects on fish survivability.

The ultimate objectives of summarizing the available data are to (1) identify turbine characteristics that enhance survival so that biological criteria can be incorporated into a new turbine design; (2) evaluate the importance of factors that affect survival; (3) provide fish survivability in passage through other exit routes without moving parts such as spillways and sluices; (4) provide some perspective on the magnitude of improvement in survival that can be achieved given the observed survival rates; (5) discern avenues wherein the turbine environment improvements should or could be made, and (6) point out significant data deficiencies and need for conducting controlled experiments with the objective of enhancing the application of the Advanced Hydropower Turbine System Program.

4.2 GENERAL REVIEW

Summary

Historical studies primarily focused on juvenile salmonids of limited size range. However, in recent years turbine passage survivability of other species has also been reported allowing for examination of species differences. Most studies did not provide turbine operating data and where exactly within the turbine test fish were released. Many prevailing hypotheses and "rules of thumb" relative to turbine fish passage were developed based on system configuration and operating conditions that are vastly different from those of the present and thus may not be applicable at many sites. There is a need to test some of the hypotheses over a range of operating conditions with fish introductions at multiple locations in a turbine to improve the knowledge on injury mechanisms and their relationship to biological factors so that it can be incorporated into the advanced turbine design.

Discussion

Several reviews of turbine passage survival (Ruggles 1980; Bell 1981; Turbak *et al.* 1981; Monten 1985; Eicher Associates 1987; Ruggles and Palmeter 1989; Cada 1990; Ruggles *et al.* 1990; EPRI 1992) indicated that most efforts on estimating turbine passage survival were initially focused on Pacific salmonids (e.g., steelhead trout, coho salmon, chinook salmon). However, in recent years survival rates of other species (e.g., resident fish, clupeids such as American shad, river herrings, and Atlantic salmon) have also appeared, primarily as a result of relicensing of hydro dams and interest in restoration and enhancement of migratory fish on the East Coast. These data provide a perspective on fish species/size-related interaction with a turbine type. This perspective is important from the standpoint of developing design features for an advanced hydro turbine to protect the greatest number of species encompassing wide size ranges.

Eicher Associates (1987) concluded that despite decades of research on salmonids, much uncertainty remained in estimating turbine passage survival. A variety of factors may cause this uncertainty; namely, the variability and lack of details relating to the design of turbine, wicket gate setting, head, species, size of fish, trajectory of entrained fish, rotational speed of runner blades, runner blade angle, number of blades, discharge, etc. These factors in combination with uncertainties associated with the prevailing tag-recapture methodologies used to estimate survival have made results of some early studies difficult to interpret. Eicher Associates (1987) also concluded that there was no turbine operating mode or design that can result in fish survival of greater than 90%. Some recent studies, however, have reported survival higher than 95% (Heisey *et al.* 1992, 1995, 1996; RMC 1994c,d; Mathur *et al.* 1994; Normandeau Associates 1996a,b).

In general, survival of fish was and still is deemed higher in passage through Kaplan type turbines than through Francis type turbines (Eicher Associates 1987; EPRI 1992). The survival was also hypothesized to be higher when turbines operate at maximum efficiency (Bell 1981). As a consequence, many large Kaplan turbines on the Columbia River Basin are operated within 1% of maximum efficiency for the head. However, a statistical analysis of the data presented in Bell (1981) was recently performed by Dr. John R. Skalski, Professor of Biostatistics at University of Washington, to evaluate the effects of turbine efficiency, wicket gate openings, fish length, specific speed, and head on passage survival. His analysis showed that the survival was more a function of percent wicket gate opening, fish length, and runner blade speed. However, the latter three variables explained only about 40% of the variation in survival. No single variable was significantly correlated to fish survival. While the effects of peak turbine efficiency on survival were not statistically correlated in this analysis, subsequent analysis of recent data from Wanapum Dam on the Columbia River show that the point of turbine operation can have significant influence on fish survival (Fisher *et al.* 1997). Higher survival coincided with turbine discharges greater than the discharges in the range of 1% efficiency below peak efficiency.

These reviews indicated the same factors that were considered as the underlying sources of variability are also deemed critical factors in affecting fish survival. These include: turbine type, fish size, trajectory of entrained fish relative to flow streams, clearance between structural components (*i.e.*, spacing between runner blades or buckets, wicket gates, and turbine housing), number of runner blades or buckets, runner blade speed, flow, and angle of water flow through turbines (Bell 1981; Eicher Associates 1987). A mathematical equation, attributed to Von Raben (Bell 1981; Ruggles and Palmeter 1989; Cada 1990), has been developed for axial flow turbines incorporating some of the above variables to predict the probability of contact with runner blades or buckets. However, the equation tends to underestimate turbine passage survival when compared to site test estimates (Bell 1981; Ruggles and Palmeter 1989; RMC 1994b). It should be noted, however, that fish mortality depending upon the site may occur from other sources as well. The equation predicts only the strike probability, which has been used to estimate potential fish mortality in some investigations. In one study, the mathematical equation predicted fish survival rates that were 3.7 to 13.9% lower than through testing (RMC 1994b). All observed fish mortality at this low head project (6 m or 21 ft) was attributed to blade strikes. The existing equation has been modified by our team to improve predictability (see Section 4.3).

It should be emphasized that many earlier hypotheses have not been widely tested over a range of operating conditions with spatially distributed fish introduction locations within turbines, particularly using newer mark-recapture techniques (balloon tag, Passive Integrated Transponder (PIT) tag, radio tags, etc.). Also, most of the earlier survival estimates, particularly in the Pacific northwest, were derived under system configurations and operational conditions that were significantly different from those presently used. For example, estimates of survival of fish entering different intake bays and depths at differing turbine operating efficiencies had been largely lacking. Similarly, the effects of turbine intake screens on unguided entrained fish were not well defined. Most of the earlier studies had involved fish releases at a single depth within an intake bay when the turbine was operating over a narrow range of operating conditions or in the absence of intake screens. Even now, many experiments are limited to tests over a narrow range of operating conditions and only to obtain estimates of survival within pre-specified variation. Experiments to determine the actual path an entrained fish traverses for quantification of the mechanisms of injury/mortality are lacking. Thus, there are large gaps in our knowledge of which factors affect fish survival in passage through turbines. It may be further compounded at many sites by the observed high survival rates, leaving little room for significant improvements.

The discussion presented in this Task 2 will address the information gaps and attempt to shed more light on the causes of fish mortality. Only through recognition of the causes can design and operational methods be changed to improve fish passage survival at hydro plants.

4.2.1 COMPILATION OF SURVIVAL DATA

Summary
Available fish survival and injury data were assembled and sorted by turbine type [axial flow-(Kaplan and propeller) and Francis], head, turbine operating status, depth of entrainment, fish size, tag-recapture methodology, recapture rates, etc. for examining trends and evaluation of variables in affecting survival. Although many of the literature data were collected to obtain only point estimates of passage survival, available information provided some general trends which can be utilized for turbine enhancement. These were: species *per se* is not important, larger sized fish (>200 mm) suffer greater mortality in passage through smaller turbines (as indexed by runner diameter (<2.5 m or 100 in), runner speed (>100 rpm), lower discharge (<71 cms or 2,500 cfs), and when wicket gate settings are narrow. The survival of smaller sized fish (<200 mm) are affected less by the above parameters. However, detailed experiments, conducted specifically to modify turbine design, showed that the survival of small sized fish (<200 mm) varied between sites having similar type turbines, entrainment depth, turbine operating status, and presence or absence of fish guidance screens or protective devices. The assembled data provided a basis for further analysis and data needs for turbine design modifications.

Discussion
The available data were separated by turbine types because earlier reviews had indicated that differences in survival may be due to whether the tested turbine was Francis or axial flow type (Kaplan and propeller). Additional information extracted from each study included the following: site name and location; turbine characteristics; head; species/size; estimation of direct effects (use of full discharge netting, radio telemetry, balloon tags) versus total effects (PIT tags, coded wire tags, branding, etc.); tag-recapture methodology; sample size employed; turbine operating conditions; recapture rate; control survival rate; statement of assumptions and tests for their validity; and precision of survival estimates. All data listings and projects at which survival was estimated are provided in Appendix Section 10.1.

Studied axial flow turbines had 3 to 7 runner blades, runner blade speeds of 75 to 241 rpm, runner diameters of 1.8 to 7.9 m (69 to 312 inches), heads ranging from 5 to 30 m (16 to 98 ft), discharges of 6 to 600 cms (200 to 21,000 cfs), and operated in efficient or inefficient mode. The latter designation is that of the dam operator at the time each study was conducted.

The Francis turbines had 12 to 19 buckets; single, double, or quad runners with runner blade speed of 72 to 510 rpm; head 4 to 120 m (13 to 387 ft); and discharge of 8 to 200 cms (275 to 7,000 cfs). Relative to the axial flow turbines, Francis turbines were generally smaller, with lower discharges and higher runner speeds.

Species tested include some of the most sensitive ones such as the juvenile clupeids (e.g., American shad, blueback herring) to the more hardier ones like the salmon, sunfish, and catfish. However, most emphasis has been on juvenile salmonid survival.

The dataset includes three basic fish body forms: generally cylindrical (e.g., salmon, most clupeids, catfish, sucker), compressed (e.g., bluegill), and ribbon-like (eels). Fish size ranged from about 55 to 881 mm; most data is for fish less than 200 mm because they are more likely to be entrained (EPRI 1992). Consequently, perhaps, data on survival of larger sized adult fish are limited. The results reviewed by Eicher Associates (1987) pertain mostly to salmonids while those given in EPRI (1992) show a greater diversity of species/size.

4.2.2 COMPONENTS OF MORTALITY

Summary

Definitions of the two primary components of fish passage mortality are given so that relevant information for turbine design modifications can be extracted from the available data or from any future planned experiments. In most cases the literature data do not provide a clear separation between the direct and indirect effects. Quantification of direct effects is deemed important from the standpoint of turbine design modifications.

Discussion

There are two primary components of total mortality of fish entrained in hydro turbines or other passage routes: direct and indirect effects. The direct effects (e.g., mechanically-induced, pressure, cavitation, or shear-related) are manifested immediately after passage as instantaneous mortality, injury, and loss of equilibrium; the indirect effects (e.g., predation, disease, physiological stress, etc.) may occur over an extended period and distance individually or synergistically. The direct effects are easier to quantify and isolate than those due to indirect sources. Tables 4.2-1, 4.2-2, and 4.2-4 provide fish survival data based on estimating direct effects while Table 4.2-3 shows data depicting effects of both direct and indirect sources. The latter studies cover exclusively large hydro dams in the Pacific Northwest.

4.2.3 POTENTIAL SOURCES OF INJURY/MORTALITY

Summary

The risks encountered by turbine entrained fish are defined (e.g., mechanical, shear-turbulence, pressure, and cavitation) and the difficulty of quantifying each risk is explained, particularly in light of intake modifications (installation of fish guidance screens, surface bypass collection system, etc.) at several hydro plants. These modifications alter the turbine hydraulics resulting in deflection of unguided fish to areas through which fish may not have been transported if modifications had not been made. This points out the need for considering turbine design modifications separately for turbines equipped with intake fish guidance screens and those without them.

Discussion

Entrained fish face three primary risks associated with turbine environment:

Mechanical mechanisms:	forces on fish body resulting from direct contact with turbine structural components such as rotating runner blades, wicket gates, stay vanes, discharge ring, draft tube, passage through gaps between the blades and the hub or at the distal end of blades, and other structures inserted into the water passageway (e.g., trash racks, intake fish guidance screens, etc.). The probability of mechanical contact depends on the distances between blades, number of blades, and fish length;
Fluid mechanisms:	*Shear-turbulence* - the effect on fish of encountering hydraulic forces due to rapidly changing water velocities; forces on fish body resulting from strong velocity gradients relative to fish length are significant. *Cavitation* - injury resulting from forces on fish body due to vapor pockets imploding near fish tissue. Under certain hydraulic conditions implosions can cause formation of velocity jets, high levels of turbulence, and high pressure shock waves. It has been assumed that if these implosions can erode metal they could damage fish tissue as well. The probability of a fish encountering these fluid mechanisms also depends to a large extent on the distances between blades, number of blades, and fish length.
Pressure:	injuries resulting from the inability of fish to adjust from the regions of high pressure immediately upstream of the turbine to regions of low pressure downstream of the turbine; the pressure change in the turbine environment itself may not be of sufficient magnitude and duration to be significant.

The above risks are related to details of the turbine, plant design, and operation as well as to the location of fish in the water column. These risks, however, are not universally applicable to all species and their life stages at all turbines; only a small proportion of the entrained fish population may be exposed to any of these risks at a site (Heisey *et al.* 1992; RMC 1994b,c,d,e; RMC and Skalski 1994a,b; RMC *et al.* 1994). Unless an individual fish is physically retrieved immediately after passage and/or somehow visually observed during its passage through a turbine, it is difficult to quantify these risks when evaluating at the level of fish size relative to the magnitude of the above forces. Thus, only probable causal sources of injury, mortality can be attributed. Also, if a fish suffers multiple injuries it may be difficult to pinpoint the causative factor. Although fish recaptured after exiting turbines have shown manifestations of injury/mortality suspected to be due to mechanical, pressure, or shear forces such categorizations were not based on direct observations (Eicher Associates 1987).

4.2.3.1 **Effect of Intake Modifications**

Summary

Turbine intake modifications such as installation of fish guidance screens can alter hydraulic conditions such that entrained fish may encounter areas of higher mortality. The survival of unguided fish could vary with the magnitude of alteration in the intake hydraulics. The redistribution, deflection, and acceleration of intake flows toward the bottom may transport unguided fish near the blade tips potentially resulting in lower survival. The Advanced Hydropower Turbine System design needs to account for the presence of intake structural modifications.

Discussion

Turbine passage survival can be influenced by turbine geometries and the point of operation, but can also be significantly altered by installation of new structures in the waterway. As an example, extended length fish guidance screens (Figure 4.2-1) installed at intakes to exclude fish from entering turbines may drastically alter hydraulic conditions such that entrained fish may encounter areas of higher mortality (Turner *et al.* 1993). The screening devices may also result in non-uniform distribution of intake flow, formation of eddies, and turbulence in some areas (Turner *et al.* 1993). Figure 4.2-2 shows a schematic of modeled velocity distribution within a turbine intake with and without an extended length screen. Intake screens cause acceleration of velocities downward. The redistribution and acceleration of flow may increase the level, incidence, and effect shear has on unguided fish passing through the turbine. Thus, the potential effect on survival of unguided fish could vary by the magnitude of alteration in the intake hydraulics. An effect of installing intake screens is a head loss which in turn can alter the hydraulics of hydro turbines.

The redistribution, deflection, and acceleration of intake flows toward the bottom of the intake may also transport unguided fish near the blade tips. Although the actual effects are not yet fully understood it has been hypothesized that fish may suffer different mechanically-induced mortality in passage near the blade tips or hub than at the mid region of the blade (Ferguson 1993, Fisher *et al.* 1997). Survival of fish entering different depths within a turbine operating at various efficiencies is largely unknown. However, a few recent studies shed more light on this issue, specifically, studies at Wanapum and Rocky Reach Dams on the Columbia River have provided impetus for turbine design improvements (RMC 1994b; RMC and Skalski 1994a,b, 1996; Ledgerwood *et al.* 1990; Normandeau Associates *et al.* 1995, 1996a; Fisher *et al.* 1997).

4.2.4 QUANTIFICATION OF PROBABLE CAUSES OF INJURY/MORTALITY

Summary

Quantification of exact sources of injury/mortality on fish transported through turbines is difficult in the field due to a lack of controlled experiments and that the observed symptoms could be manifested by two different sources. However, some mechanically-related injuries, as evident by sliced bodies and pinched bodies, may be quantified with greater certainty. Results from most studies indicate that mechanically related injuries are a dominant source of mortality, particularly for fish transported near the hub where gaps exist (Kaplan adjustable blades). Pressure-related injuries appear to be more a function of acclimation history of fish upstream of turbine than passage through turbines *per se*. At dams (>30 m or 100 ft head), without hydro turbines, fish transported through bottom sluices or openings suffer decompression trauma (as evident by rupture of air bladder and other internal organs) when rapidly exposed to shallow tailrace conditions. Though evidence of injuries due to fluid shear forces exist, relative to other sources, it is not a dominant source of fish injury/mortality in passage through turbines.

Discussion

A lack of controlled experiments to replicate and correlate each injury type/characteristic to a specific causative mechanism (in combination with the meager knowledge of the actual path fish traverse within a turbine) precludes definitive classification of observed injuries in the field. Literature suggests that observed injury symptoms could be manifested by two different sources and accurate delineation of a cause and effect relationship may be difficult (Eicher Associates 1987). Consequently, only probable causal mechanisms of injury can be assigned. However, some mechanically related injuries (e.g., sliced or pinched bodies) may be assigned with greater certainty (Figure 4.2-3). Injuries likely associated with direct contact with turbine runner blades or impacting structural components are classified as mechanical and include: bruises/hemorrhaging, lacerations, and severed/sliced body (Dadswell *et al.* 1986; Eicher Associates 1987; RMC and Skalski 1994a,b). Injuries likely attributed to fluid shear forces are decapitation (with the isthmus attached to the body and a slanted wound), torn or flared opercula, and inverted or broken gill arches (Dadswell *et al.* 1986). The effects of pressure changes are manifested as bloody eyes, popped eyes, air bladder rupture, and embolism (Figure 4.2-4).

In general, turbine-passage experiments conducted in the Pacific Northwest have provided most of the empirical field evidence (Oligher and Donaldson 1966; RMC 1994a; RMC and Skalski 1994a,b; RMC *et al.* 1994; Normandeau Associates *et al.* 1995; 1996a; Normandeau Associates and Skalski 1996) of probable sources of injury while laboratory experiments (Muir 1959; Lucas 1962; Harvey 1963; Groves 1972; Feathers and Knable 1983; Turnpenny *et al.* 1992) have provided data on effects of individual factors such as pressure changes, cavitation, velocity, shear, turbulence, etc. Some of the former experiments were conducted over a narrow range of turbine operation efficiencies, entrainment depths, head, fish size, etc. and could only speculate where in the turbine environment the observed injuries may have been inflicted.

4.2.4.1 **Mechanical Related Injuries**

Summary

Direct contact with the turbine runner blades and passage through the gaps between the blades and hub are prime suspect areas of mechanical fish damage. Injury rates increase with fish size. However, the rate of mechanical related injury can also vary with the fish entrainment depth, turbine operating status, and whether the intakes are equipped with fish guidance screens. Mechanical related causes have been reported as dominant cause of fish mortality at low head (<30 m or 100 ft) projects.

Discussion

Mechanically related injuries have been posited as a major source of fish mortality in field investigations. Oligher and Donaldson (1966) conducted a series of relatively controlled experiments to evaluate the effects of turbine operating efficiency, head, and runner blade angles on fish survival at Big Cliff Dam and provided some early empirical evidence of probable sources of injury/mortality. Data presented by these authors for the 1966 experiments showed that about 70 to 93% of the injuries (based on number of recaptured injured fish) were due to probable mechanical causes and less than 2% due to probable pressure related causes. In the 1964 experiments by Oligher and Donaldson (1996) probable pressure-related injuries accounted for 9 to 16% while mechanically related causes accounted for 77 to 84%. The relative importance of pressure related injuries as a function of head (22 to 28 m or 71 to 91 ft) or blade angles could not be observed.

Turbine configuration can influence the rate of mechanically related injuries. RMC (1994a) noted a predominance of probable mechanically related injuries (4.4% as indicated by severed body, bruises/hemorrhaging) on chinook salmon smolts in passage through Unit 7 (adjustable blade Kaplan) of Rocky Reach Dam. In contrast, only 1.3%, though nonsignificant, of fish showed similar injury types in passage through Unit 8 (fixed blade). At Unit 3 (adjustable blade Kaplan), mechanically induced injuries were observed on 5.7% of recaptured fish after passage at 3 m (10 ft) depth; at 9 m (30 ft) depth injury rate was estimated at 4.1% (RMC 1994a). It was concluded that the higher injury rate, though nonsignificant, on fish entrained at 3m (10 ft) depth of Units 3 and 7 was most likely due to passage through the gaps between the runner blades and the hub; Unit 8 is a fixed blade turbine and gaps are absent. At Wilder Dam on the Connecticut River, RMC (1994c) reported that 3.2% of recaptured Atlantic salmon smolts had severed bodies and on additional 1.6% showed external bruises/hemorrhaging and internal hemorrhaging.

Injury types and rates can differ between entrainment depth, turbine operating status, and whether the intake is equipped with fish guidance screens. At Lower Granite Dam turbine Unit 4 injuries on recaptured/injured chinook salmon smolts introduced at the depth of standard length screens, with the turbine operating at normal efficiency, were attributed to probable sources as follows: 67% mechanical (Figure 4.2-3), 21% shear and pressure (Figure 4.2-4), and the remainder to multiple causes (RMC *et al.* 1994). In a 1995 study at the same turbine the overall injury distribution for chinook smolts introduced about 3m deeper (at the depth of extended length screens) than in 1994 was as follows: 50% mechanical, 18.8% pressure, 14.1% to shear, and remainder to multiple causes. Fish introduced at upper elevation (about 3 m below the intake ceiling) appeared to suffer greater rate of mechanical injuries (70.6%) than at greater depths (< 45%); the difference in mechanically-related injury rate was attributed to the presence of gaps between the runner blades and the hub through which the upper released fish were transported (Normandeau Associates *et al.* 1995). Gap related injuries were characterized by pinching types (Figure 4.2-3). At Wanapum Dam, probable mechanically related injuries were also common (43%) on injured coho salmon smolts; pressure related injuries accounted for 23%, and shear 10%; the remainder to multiple causes (Normandeau Associates *et al.* 1996c). Only 30 of the 1,202 turbine passed, recaptured fish were injured and a shift of one or two fish into any injury category can make substantial changes in the indicated percentages. As mentioned earlier, a lack of controlled experiments precludes definitive assignments of exact source of injury mechanism.

4.2.4.2 Pressure Related Injuries

Summary

Fish are more sensitive to exposure to sudden pressure reduction than an increase in pressure. However, the magnitude of pressure change and fish acclimation history are important factors. Fish with a pneumatic duct (physostomes) attached to air bladders are able to adjust to pressure changes quicker than those without the duct (physoclist). Physostomes can vent excess gas quicker, but if access to free air is not available, they will not be able to adjust to increasing pressures except through the gas gland. Laboratory studies have shown variable effects of pressure reduction and most studies did not simulate pressure regimes that fish encounter in the field. Thus, a blanket application of these results to the design of a turbine may be risky; the effects of pressure reduction are related to the difference in pressure between the acclimation depth (a function of head) and duration of time and the pressure at the exposure depth. A true acclimation history of fish and the time it takes to become fully acclimated to a depth are unknown. Recent studies at low head projects (<30 m or 100 ft) have shown only a secondary importance of pressure-related damage. However, at higher head dams whether equipped with hydro turbines or not, exposure to pressure reduction is a significant source of fish mortality; at projects with head less than 18 m (60 ft) probable pressure-related injuries have not been observed. Also, even relatively shallow intakes (<9 m or 30 ft) leading into long pipes or penstocks (>303 m or 1,000 ft) at high head projects (generally discharge <23 cms or 800 cfs) pose significant risk of fish mortality because of low relative velocities allowing time for entrained fish to become acclimated to deeper depth prior to passage through the turbines. The proportional contribution of pressure-related injuries to the total fish mortality may increase at large turbine intakes equipped with fish guidance screens compared to those without them (Normandeau Associates *et al.* 1995).

Discussion

Fish are more tolerant of increases in pressure than sudden reduction in pressure. The latter is of more relevance to fish passage at hydro dams. Fish are more tolerant of gradual reduction in pressure than to sudden exposures. In this section pressure is expressed as pound per inch (psi) in the English system and as pascal (Pa) in the International (SI) System (Cada *et al.* 1997). One pascal equals one N/m^2; water pressure at one atmosphere equals 101.3 (kilo pascal or kPa) or 14.7 psi.

The laboratory-derived relationship between fish acclimation pressure and subsequent rapid exposure to pressure reductions may not adequately simulate fish responses within the turbine environment of concern (e.g., exposure to limited or small area of low pressure on the lower side of runner blades). The results from laboratory-derived data need to be carefully applied to the turbine rehabilitation program. In addition, some field tests may prove more instructive in determining the tolerance of fish to pressure regime experienced in lakes or reservoirs; these studies are described below. Little guidance exists to estimate the time needed to acclimate fish to a given pressure. In addition, Cada *et al.* (1997) have pointed out several shortcomings of some of the past laboratory experiments including poor documentation, inadequate or no controls, use of small numbers of fish, and measurement of fish responses to reduction in pressure only from atmospheric levels (101 kPa or 14.7 psi) to sub-atmosphere levels. The latter factor is, perhaps, of most relevance to hydroelectric turbines because fish are subjected to pressures higher than atmospheric levels prior to entering the turbine environment and then rapidly traversing a low pressure region on the downstream side of the runner blade and then finally becoming exposed to near atmospheric levels in exiting the turbine draft tube. Thus, the eventual fate of entrained fish would be dictated by its previous acclimation history (depth and time) prior to entering the turbine and its end pressure in the tailrace rather than a split second passage through a zone of pressure differential at the runner blades. It is unlikely that this turbine passage time is sufficient for acclimation to changing pressures. At many sites surface-oriented fish have to sound to greater depths (12 to 20 m or 40 to 65 ft) to exit, such as bottom opening tainter gates at spillways in the Pacific Northwest, and are undoubtedly subjected to rapid pressure reductions presumably with little adverse effect; spill is routinely used to minimize fish passage through turbines. Also, fish intercepted by extended length screens from deeper depths and collected in gatewells or surface bypass structures have not shown adverse effects of pressure reductions.

The tolerance to pressure reduction appears to be dependent on whether a species is physostome or physoclist and their acclimation history (depth and time). Physostomous species (those having pneumatic duct, connecting air bladder to the esophagus, for venting air bladder gas) are more tolerant than physoclists (those without pneumatic duct). The presence of pneumatic duct allows the physostomous species (e.g., salmon, minnows, catfish, etc.) to rapidly take in or vent gases from the swim bladder (within seconds) through the mouth so that adjustment to changing water pressures can be made quickly. However, the time to reach full acclimation to a given depth is uncertain. In physoclists (e.g., basses, sunfish, perch, walleye, etc.) contents and pressures within the swim bladder must be adjusted by diffusion into the blood, a process taking hours (Cada *et al.* 1997).

Laboratory experiments by Harvey (1963) and Turnpenny *et al.* (1992) suggest that physostomes (primarily salmonids) can tolerate reductions in absolute pressures. When he rapidly exposed sockeye smolts presumably acclimated at 2,064 kPa (20.4 atmosphere or 300 psi) to atmospheric pressure, Harvey (1963) observed mortalities of less than 1% per week. Mortalities among the treatment groups were indistinguishable from those among the controls. Smolts acclimated to 350 kPa (50 psi) for 24 h then rapidly returned to atmospheric conditions exhibited less than 0.5% mortality per week. Similarly, Tsvetkov *et al.* (1972), cited by Cada *et al.* (1997), reported resistance of two species of sturgeon to reduction in absolute pressure. These fish presumably acclimated to 608 kPa (88 psi) and then rapidly exposed to decompression did not exhibit lethality. However, physoclists (primarily largemouth bass) when acclimated at 280 to 369 kPa and rapidly exposed to 101 kPa suffered significant mortalities (Feathers and Knable 1983). Muir (1959) rapidly decompressed (from 22 psi to vapor pressure) 20 coho salmon fingerlings (66 mm) for 0.4 sec. A 60% mortality was observed which was attributed to the rapid high pressure shock waves associated with the collapse of the cavitation bubble. In another experiment 10 coho fingerlings were exposed for 1.6 sec. to similar decompression (from 22 psi to vapor pressure) no mortality was observed. Laboratory experiments by Turnpenny *et al.* (1992) indicated low mortality (0 to 10%) with sudden pressure reductions of up to 90% exposure (from a pressure of 343 kPa to 30 kPa); these low mortalities were supported by mathematical equations developed to predict pressure-related fish damage in a reference turbine. No external damage (e.g., popped eyes, hemorrhaging) was observed; a small proportion of fish (10%), however, showed air bladder rupture. They attributed the high tolerance of tested species to the ability to rapidly vent gases from their swim bladders under decompression conditions. In contrast, physoclists (e.g., seabass) suffered a higher rate of air bladder rupture and mortality. Under sustained decompression conditions swim bladder rupture of physoclists occurred at about doubling of the swim bladder volume.

Field studies provide important perspective on fish tolerance to pressure reduction (differences in absolute pressure values from high to low) manifested through intake configuration and depth, acclimation depth/time, and transit time through penstocks. Studies at Bond Falls Station, MI (head 64 m or 210 ft, discharge 11 cms or 385 cfs) and McClure, MI (head 129 m or 425 ft, discharge 9 cms or 309 cfs), each equipped with a 2 to 4 km (1.3 to 2.5 mi) long penstock showed that most fish suffered decompression trauma (as evidenced by ruptured or extruded air bladder, ruptured heart and kidney, and broken bones) upon exiting the turbines. The intakes for these plants are located in the upper water level (about 3 to 9 m or 10 to 30 ft, below reservoir surface) and lead into long sloping 3 m diameter pipes (penstocks) which may allow fish to gradually acclimate to deeper depths (absolute pressures exceeding 9 atmospheres or 142 psi); the estimated velocity through the pipes (penstocks) were less than 0.5 m/s (1.5 ft/s), probably insufficient to move fish rapidly through the system (RMC 1993b, 1996).

At Berlin Lake in Ohio, walleye passing through bottom sluice gates (depth approximately 35 m or 115 ft) died of decompression trauma (air bladder rupture) while those passing over the tainter gates survived (Smith and Anderson 1984). Fish acclimated to higher pressure at greater depths suffered decompression trauma when discharged into shallow tailrace depths. Decompressed fish were more common during the winter months, perhaps walleyes moving to deeper waters to over-winter. Similar high levels of decompression-related fish mortalities (74%) were observed in Allegheny Reservoir in the winter months.

Fish migrating through bottom opening sluices (depth about 30 m or 100 ft) at Tygart Dam, West Virginia exhibited symptoms of decompression trauma when discharged into a stilling basin (depth 9 m or 30 ft; Jernezcic 1986). RMC (1992a) reported that a rapid exposure of fish acclimated to pressures of about 376.3 kPa (54.6 psi) or in a long tunnel to pressures of 12.3 kPa (1.8 psi) or in the tailrace, proved lethal in passage through wheel gates (without hydroelectric plant) at the Youghiogheny Dam, PA (operating head of about 36 m or 120 ft). The estimated mortality due to decompression trauma exceeded 90% for small fish (<100 mm) and about 49% for large fish (221-531 mm). Most common fish depicting symptoms of decompression were physoclists such as walleye, alewife, crappies, and yellow perch. These fish had entered the long intake tunnel and became acclimated to deeper depth prior to being suddenly discharged into the shallow tailrace. Decompression trauma was most common in winter, presumably most fish moved deeper in the water column to over-winter or to follow the forage fish.

4.2.4.3 **Shear Related Injuries**

Summary

Effects of shear induced forces have been studied primarily under laboratory conditions which are not representative of internal hydraulic conditions of a turbine. However, these studies indicate that shear effects may be species and size specific and are related to the orientation of fish in the shear zone. Larger sized fish and those facing a water jet appear to suffer less injuries; water jet experiments involved velocities of 9 to 36 m/s (30 to 120 ft/s) with a velocity of 18 m/s (58 ft/s) having little effect on fish.

Discussion

Effects of shear induced forces have been primarily studied under laboratory conditions (Groves 1972; Johnson 1972; Turnpenny *et al.* 1992) where fish were exposed to high velocity discharge in a static water tank rather than interaction between moving flows as encountered in a turbine. These studies indicate that shear effects may be dependent on species and size and related to the manner of contact rather than to a particular velocity difference. Johnson (1972) observed no mortality of juvenile coho salmon, chinook salmon, and steelhead when these fish were transported through a 102 to 152 mm (4 to 6 inches) submerged nozzle at velocity of about 18 m/s (58 ft/s; Figure 4.2-5). The location of fish in the jet, orientation of fish as they exited the nozzle, and location where they exited the jet into the static water could not be controlled (Cada *et al.* 1997). Groves (1972) flushed juvenile coho, chinook, and steelhead into the water tank through an angled tube so that they would strike the jet within 76 mm (3 inches) of its emergence from the nozzle. Jet velocities ranged from 9 to 36 m/s (30 to 120 ft/s), however, actual shear forces and velocities experienced by fish relative to its length were not measured. He concluded that fish could be injured in any high energy flow situation that creates momentary localized points of sharp velocity change. Smaller salmon (<30 mm long) suffered greater injury and mortality rates than larger salmon (135 mm long), probably because of lesser tissue strength and exposure of a greater proportion of the body to initial contact with the water jet (Groves 1972). Greatest injuries occurred when the water jet contacted the head region and it was moving from the rear toward the head of the fish (Figure 4.2-6). Less injurious was when the fish faced into the jet. Turnpenny *et al.* (1992) noted that water velocities of up to 15 m/s (50 ft/s) and associated shear stresses caused little mortality in young clupeids and American eel while the salmonids suffered no mortality at velocities up to 9 m/s (30 ft/s); velocities of 15 m/s (50 ft/s) inflicted variable mortality rates. None of the studies considered the interrelationships of force, inertia, and shear relative to the fish size. Section 4.3 provides a detailed background of effects of shear forces for greater understanding so that better experiments can be run.

4.2.5 SCREENING CRITERIA FOR SELECTING SURVIVAL TESTS

Summary

Although literature reviews and recent studies provide estimates of fish survivability, not all estimates are useful from the standpoint of understanding the mechanisms leading to the observed mortality so that the knowledge can be used for potential turbine design modifications; mechanistic types of studies are generally lacking. Most studies were not conducted with the objective of developing information for turbine modifications However, a careful review of the studies from the existing database was considered necessary to select those which may contribute to our understanding of fish reactions within turbines. Therefore, screening criteria which when applied in aggregate rather than only one parameter were developed to select studies to provide some insight into the mechanisms of turbine related mortality. These included: high recapture rates, low control mortality, acceptable tag-recapture methodology, description of injury types, direct or indirect effects, turbine operating status, turbine characteristics, and adequate sample size. Also, because reporting of survival estimates vary (1 h, 24 h, 48 h, 72 h, 120 h, etc.) only 1 h estimates were extracted to delineate the direct effects of turbine passage. This selection criteria is similar to that recommended recently by EPRI (1992) for conducting survival studies.

Discussion

Major obstacles to obtaining reliable survival estimates of fish in transport through a hydro turbine, particularly those with higher discharges (>60 cms or 2,000 cfs), have been the inability of investigators to recapture a high proportion of released fish and to maximize control group survival (Heisey *et al.* 1992). Burnham *et al.* (1987) and Ruggles *et al.* (1990) indicated that the reliability of a survival estimate is enhanced when investigators can recapture a high proportion of fish after turbine passage, recapture rates of treatment and control fish are similar, and survival of control fish is high. Mathur *et al.* (1994) noted that reliability of survival estimates (direct passage effects) generally increases when fish recapture rates exceed 70%. The superiority of high recapture of fish in release-recapture survival experiments in simplifying assumptions is well known (Burnham *et al.* 1987).

Although literature reviews contain numerous field estimates of fish survivability not all estimates can be considered useful from the standpoint of potential turbine design modifications. Most of the studies, particularly related to relicensing of hydroelectric projects in the 1990's, were not conducted with a specific objective of obtaining information usable for advanced turbine design development; in many cases the objective was simply to provide an estimate of fish survivability. Additionally, many studies suffered from a lack of standardized experimental protocols, lack of replication for estimating between turbine variability, poor documentation of results, low precision, lack of adequate controls or high control mortality (>30%), limited turbine operating data, accurate species/size identification, and sampling gear deemed suitable for survival estimation (e.g., tailrace netting for estimating survival of herring like fish is not deemed suitable by EPRI (1992, 1997). Therefore, a set of criteria was developed, to be used in aggregate rather than based on a single parameter, to select data that may yield useful information for turbine designers and narrow down the list of important candidate variables which may be amenable for possible modifications. Undoubtedly, the selection process requires some professional judgment. Although the database in this report contains almost all the studies reported in the literature reviews (Bell 1981; Eicher Associates 1987; EPRI 1992) usable information was extracted from studies which also provided the following: estimation of direct effects; acceptable sample size (≥100); high control survival rate (preferably ≥90%); high recapture rates (generally ≥70%); description and quantification of injury types; turbine characteristics (e.g., discharge, number of blades or buckets, runner diameter, head, etc.); turbine operating data; and species and size. Tables 4.2-1 to 4.2-4 show the data used in further analysis.

Recently, EPRI (1997) has provided detailed guidelines for obtaining acceptable survival estimates emphasizing many of the criteria components cited above: need for detailed turbine physical and hydraulic characteristics; detailed information on species origin (hatchery-reared or river run), condition, number used, and size; accurate location of fish introduction within a turbine; turbine operating status; minimal control mortality (preferably less than 20%); number of fish recaptured; replication of tests; survival estimates (1 h, 48 h, or 120 h, etc.) accompanied with confidence limits (precision); and assumptions used to derive estimates.

The following Section 4.3 presents background and quantitative characteristics of flow through turbines so that a better understanding of the processes occurring in turbine leading to fish mortality can be obtained. As a result, better turbine design features can be planned and implemented to achieve lower turbine passage mortality.

Table 4.2-1

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data were deemed usable (direct effects data used in statistical analyses).

Station	Sampling Method	Species Tested	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	Percent Survival 1 Hr.
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	100	52.5	6	163.6	27.7	3.76	32.2	91.1
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	100	71.1	6	163.6	24.7	3.76	32.2	94.5
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	100	71.1	6	163.6	21.6	3.76	32.2	89.7
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	100	52.5	6	163.6	27.7	3.76	32.2	92.2
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	100	71.1	6	163.6	24.7	3.76	32.2	89.8
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	100	71.1	6	163.6	21.6	3.76	32.2	90.6
Big Cliff, OR (1967)	Full discharge netting	Steelhead	152	71.1	6	163.6	21.6	3.76	32.2	90.4
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	Bluegill	103	37.7	4	150	8.8	2.59	20.3	97.0
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	Bluegill	153	37.7	4	150	8.8	2.59	20.3	98.0
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	W. Sucker/R. Trout	119	37.7	4	150	8.8	2.59	20.3	91.0
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	W. Sucker/R. Trout	261	37.7	4	150	8.8	2.59	20.3	97.0
Conowingo, MD	Hi-Z Turb'N Tag	American Shad	125	226.6	6	120	27.4	5.72	35.9	94.9
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	180	17.0	4	229	6.4	1.75	21.0	93.0
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	180	5.7	4	229	6.4	1.75	21.0	90.0
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	277	5.7	4	229	6.4	1.75	21.0	81.0
Craggy Dam, NC	Hi-Z Turb'N Tag	Bluegill	100	5.7	4	229	6.4	1.75	21.0	96.0
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	277	17.0	4	229	6.4	1.75	21.0	93.0
Craggy Dam, NC	Hi-Z Turb'N Tag	Bluegill	155	5.7	4	229	6.4	1.75	21.0	86.0
Crescent, NY	Hi-Z Turb'N Tag	Blueback Herring	91	43.0	5	144	8.2	2.74	20.7	96.0
Essex, MA (bulb turbine)	Radio telemetry	Atlantic Salmon	288	124.6	3	128.6	8.8	4.00	26.9	98.0
Foster, OR (tests combined)	Full discharge netting	Chinook Salmon	120	22.7	6	257	26.2	2.54	34.2	82.1
Foster, OR (tests combined)	Fyke netting	Chinook Salmon	130	22.7	6	257	30.8	2.54	34.2	92.7
Foster, OR (tests combined)	Full discharge netting	Chinook Salmon	120	22.7	6	257	33.5	2.54	34.2	91.2
Feeder Dam, NY	Full discharge netting	Bluegill	92	29.5	6	120	4.7	2.92	18.3	97.3
Feeder Dam, NY	Full discharge netting	Bluegill	129	29.5	6	120	5.2	2.92	18.3	92.3
Feeder Dam, NY	Full discharge netting	Largemouth bass	88	29.5	6	120	5.5	2.92	18.3	98.0
Feeder Dam, NY	Full discharge netting	Largemouth bass	190	29.5	6	120	5.8	2.92	18.3	90.0
Feeder Dam, NY	Full discharge netting	Largemouth bass	292	29.5	6	120	6.1	2.92	18.3	86.8
Feeder Dam, NY	Full discharge netting	Brown trout	206	29.5	6	120	6.4	2.92	18.3	86.4
Feeder Dam, NY	Full discharge netting	Golden shiner	88	29.5	6	120	6.7	2.92	18.3	96.8
Greenup Dam, OH (Vanceburg)	Radio telemetry	Sauger	231	336.1	5	90	9.1	6.10	28.7	85.4
Hadley Falls, MA	Radio telemetry	American Shad	560	118.9	5	128	15.8	4.32	28.9	78.2
Hadley Falls, MA	Hi-Z Turb'N Tag	American Shad	82	118.9	5	128	15.8	4.32	28.9	97.3
Hadley Falls, MA	Radio telemetry	Atlantic Salmon	285	118.9	5	128	15.8	4.32	28.9	93.7
Hadley Falls, MA	Hi-Z Turb'N Tag	American Shad	82	43.9	5	128	15.8	4.32	28.9	100.0

Table 4.2-1

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data were deemed usable (direct effects data used in statistical analyses).

Station	Sampling Method	Species Tested	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	Percent Survival 1 Hr.
Herrings, NY	Full discharge netting	Centrarchid	100	34.0	4	138	5.8	2.87	20.7	98.3
Herrings, NY	Full discharge netting	Centrarchid	175	34.0	4	138	5.8	2.87	20.7	97.3
Herrings, NY	Full discharge netting	Centrarchid	250	34.0	4	138	5.8	2.87	20.7	93.2
Herrings, NY	Full discharge netting	Percid	100	34.0	4	138	5.8	2.87	20.7	91.1
Herrings, NY	Full discharge netting	Salmonids	100	34.0	4	138	5.8	2.87	20.7	90.0
Herrings, NY	Full discharge netting	Salmonids	175	34.0	4	138	5.8	2.87	20.7	87.5
Herrings, NY	Full discharge netting	Salmonids	250	34.0	4	138	5.8	2.87	20.7	96.2
Herrings, NY	Full discharge netting	Centrarchid	100	34.0	4	138	5.8	2.87	20.7	95.0
Herrings, NY	Full discharge netting	Centrarchid	175	34.0	4	138	5.8	2.87	20.7	96.4
Herrings, NY	Full discharge netting	Centrarchid	250	34.0	4	138	5.8	2.87	20.7	92.5
Herrings, NY	Full discharge netting	Percid	100	34.0	4	138	5.8	2.87	20.7	94.9
Herrings, NY	Full discharge netting	Percid	175	34.0	4	138	5.8	2.87	20.7	98.2
Herrings, NY	Full discharge netting	Percid	250	34.0	4	138	5.8	2.87	20.7	96.2
Herrings, NY	Full discharge netting	Salmonids	100	34.0	4	138	5.8	2.87	20.7	95.5
Herrings, NY	Full discharge netting	Salmonids	175	34.0	4	138	5.8	2.87	20.7	98.7
Herrings, NY	Full discharge netting	Salmonids	250	34.0	4	138	5.8	2.87	20.7	98.6
Herrings, NY	Full discharge netting	Soft ray	100	34.0	4	138	5.8	2.87	20.7	97.5
Herrings, NY	Full discharge netting	Soft ray	175	34.0	4	138	5.8	2.87	20.7	91.7
Herrings, NY	Full discharge netting	Soft ray	250	34.0	4	138	5.8	2.87	20.7	85.1
Herrings, NY	Full discharge netting	Clupeids	100	34.0	4	138	5.8	2.87	20.7	92.8
la centrale de Beauharnois,	Float tag	American eel	881	262.7	6	94.7	24.1	6.32	31.3	76.1
Lowell, MA	Radio telemetry	Atlantic Salmon	265	127.4	5	120	11.9	3.86	24.2	88.5
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	134	594.7	6	90	29.9	7.92	37.3	94.6
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	151	509.8	6	90	29.9	7.92	37.3	94.9
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	150	509.8	6	90	29.9	7.92	37.3	95.3
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	148	382.3	6	90	29.9	7.92	37.3	97.2
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	148	538.1	6	90	29.9	7.92	37.3	94.6
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	151	509.8	6	90	29.9	7.92	37.3	97.5
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	150	509.8	6	90	29.9	7.92	37.3	97.5
Raymondville, NY	Full discharge netting	Eel	625	46.4	6	120	6.4	3.33	20.9	63.0
Rock Island, WA (bulb turbine)	HI-Z Turb'N Tag	Chinook Salmon	179	481.5	4	85.7	12.2	7.01	30.5	96.1
Rock Island, WA (PH 1, U 4)	HI-Z Turb'N Tag	Chinook Salmon	179	481.5	6	100	13.7	5.74	30.5	95.0
Rock Island, WA (PH 1, U 5)	HI-Z Turb'N Tag	Chinook Salmon	179	481.5	6	100	13.7	5.74	30.5	96.1
Rocky Reach, WA (30',U. 3)	HI-Z Turb'N Tag	Chinook Salmon	161	453.1	6	90	28.0	7.11	33.5	94.7
Rocky Reach, WA (10',U. 3)	HI-Z Turb'N Tag	Chinook Salmon	161	453.1	6	90	28.0	7.11	33.5	93.9
Rocky Reach, WA (10',U. 5)	HI-Z Turb'N Tag	Chinook Salmon	184	396.5	6	90	28.0	7.11	33.5	97.3

Table 4.2-1

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data were deemed usable (direct effects data used in statistical analyses).

Station	Sampling Method	Species Tested	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	Percent Survival 1 Hr.
Rocky Reach, WA (30',U. 5)	Hi-Z Turb'N Tag	Chinook Salmon	184	396.5	6	90	28.0	7.11	33.5	94.4
Rocky Reach, WA (10',U. 6)	Hi-Z Turb'N Tag	Chinook Salmon	184	396.5	6	90	28.0	7.11	33.5	94.2
Rocky Reach, WA (30',U. 6)	Hi-Z Turb'N Tag	Chinook Salmon	184	396.5	6	90	28.0	7.11	33.5	95.8
Safe Harbor, PA (Unit 7)	Hi-Z Turb'N Tag	American Shad	118	235.1	5	109	16.8	5.64	32.2	98.0
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Largemouth Bass	217	42.5	3	152	4.9	2.87	22.8	96.8
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Rainbow Trout	139	22.7	3	152	4.9	2.87	22.8	94.4
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Rainbow Trout	344	22.7	3	152	4.9	2.87	22.8	86.5
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Largemouth Bass	102	22.7	3	152	4.9	2.87	22.8	100.0
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Largemouth Bass	217	22.7	3	152	4.9	2.87	22.8	86.0
Townsend Dam, PA (bulb turbine)	Hi-Z Turb'N Tag	Rainbow Trout	139	42.5	3	152	4.9	2.87	22.8	100.0
Wanapum, WA (10ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	254.9	5	85.7	22.9	7.24	32.5	89.7
Wanapum, WA (10ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	311.5	5	85.7	22.9	7.24	32.5	92.4
Wanapum, WA (10ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	424.8	5	85.7	22.9	7.24	32.5	94.8
Wanapum, WA (10ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	481.5	5	85.7	22.9	7.24	32.5	88.5
Wanapum, WA (30ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	254.9	5	85.7	22.9	7.24	32.5	94.9
Wanapum, WA (30ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	311.5	5	85.7	22.9	7.24	32.5	96.8
Wanapum, WA (30ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	424.8	5	85.7	22.9	7.24	32.5	100.0
Wanapum, WA (30ft, Unit 9)	Hi-Z Turb'N Tag	Coho Salmon	154	481.5	5	85.7	22.9	7.24	32.5	96.8
West Enfield, ME	Radio telemetry	Atlantic Salmon	212	150.1	3	89	6.4	4.88	22.7	96.0
Wilder, VT-NH	Hi-Z Turb'N Tag	Atlantic Salmon	191	127.4	5	112.5	15.5	4.57	26.9	96.0

Table 4.2-2

Physical and hydraulic characteristics of all hydroelectric dams equipped with propeller type turbines for which survival data were deemed usable (direct effects data used in statistical analyses).

Station	Sampling Method	Species Tested	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	Percent Survival 1 Hr.
Hadley Falls, MA	HI-Z Turb'N Tag	American Shad	82	118.9	5	150	15.8	3.96	31.1	89.1
Rocky Reach, WA (10',U. 8)	HI-Z Turb'N Tag	Chinook Salmon	114	566.4	5	85.7	26.4	7.90	35.4	96.9
Safe Harbor, PA (Unit 8)	HI-Z Turb'N Tag	American Shad	118	260.5	7	75	16.8	6.15	24.1	97.8
Safe Harbor, PA (Unit 8)	HI-Z Turb'N Tag	American Shad	118	260.5	7	75	16.8	6.15	24.1	98.9

Table 4.2-3

Survival estimates (containing both direct and indirect effects) in passage through Kaplan type turbines.

Station	Sampling Method	Species Tested	Runner Dia. (m)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Peripheral Velocity (m/s)	Percent Survival 1 Hr.
Bonneville, OR/WA	Branding/CWT	Chinook salmon	7.62	498.4	5	69.2	18.3	27.6	97.5
Little Goose, WA	PIT tag	Chinook salmon	7.92	509.8	6	90.0	28.3	37.3	92.0
Lower Granite, WA	PIT tag	Chinook salmon	7.92	509.8	6	90.0	29.9	37.3	92.7
Rock Island, WA	Brand/partial netting	Coho salmon	7.01	509.8	4	85.7	12.2	31.4	93.0
Rock Island, WA	Brand/partial netting	Steelhead	7.01	509.8	4	85.7	12.2	31.4	96.9
Wells, WA	Brand/partial netting	Steelhead	7.43	566.4	6	85.7	19.8	33.3	84.0
Lower Monumental, WA	PIT tag	Chinook salmon	7.92	509.8	6	90.0	28.7	37.3	86.5
McNary, WA	Brand/partial netting	Chinook salmon	7.11	348.3	6	87.5	24.4	32.6	89.0

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Alcona, MI	Full dschrg netting	Bluegill	97	-	118	46.9	16	90	13.1	2.54	12.0	90.2
Alcona, MI	Full dschrg netting	Bluegill	102	-	170	47.0	16	90	13.1	2.54	12.0	84.1
Alcona, MI	Full dschrg netting	Gold./Common Shiner	51	-	114	47.0	16	90	13.1	2.54	12.0	80.9
Alcona, MI	Full dschrg netting	Gold./Common Shiner	58	-	154	47.0	16	90	13.1	2.54	12.0	84.7
Alcona, MI	Full dschrg netting	Northern Pike	44	-	352	47.1	16	90	13.1	2.54	12.0	51.2
Alcona, MI	Full dschrg netting	Rainbow Trout	40	-	108	47.1	16	90	13.1	2.54	12.0	100
Alcona, MI	Full dschrg netting	Rainbow Trout	40	-	317	47.1	16	90	13.1	2.54	12.0	89.4
Alcona, MI	Full dschrg netting	Walleye	45	-	385	47.2	16	90	13.1	2.54	12.0	38.7
Alcona, MI	Full dschrg netting	White Sucker	60	-	180	47.2	16	90	13.1	2.54	12.0	94.4
Alcona, MI	Full dschrg netting	White Sucker	54	-	290	47.3	16	90	13.1	2.54	12.0	90.4
Buchanan, MI	Full dschrg netting	Chinook salmon	600	400	420	2.8	-	-	-	-	-	79.6
Buchanan, MI	Full dschrg netting	Steelhead trout	600	400	420	6.2	-	-	-	-	-	79.4
Bond Falls, MI	Full dschrg netting	Rainbow Trout	350	225	210	12.7	-	300	64.0	-	-	83.8
Bond Falls, MI	Full dschrg netting	Yellow Perch	360	225	102	12.7	-	300	64.0	-	-	79.5
Bond Falls, MI	Full dschrg netting	Golden Shiner	405	225	70	12.7	-	300	64.0	-	-	77.9
Bond Falls, MI	Full dschrg netting	Bluegill	660	450	115	12.7	-	300	64.0	-	-	81.7
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	144	94	76	18.4	15	226	24.4	1.83	21.6	100.0
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	141	90	127	18.4	15	226	24.4	1.83	21.6	98.2
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	76	35	178	18.4	15	226	24.4	1.83	21.6	86.8
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	145	86	76	18.4	15	226	24.4	1.83	21.6	80.3
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	139	92	127	18.4	15	226	24.4	1.83	21.6	84.8
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	125	58	178	18.4	15	226	24.4	1.83	21.6	70.3
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	136	63	229	18.4	15	226	24.4	1.83	21.6	64.3
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	146	94	292	18.4	15	226	24.4	1.83	21.6	59.5
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	153	76	>292	18.4	15	226	24.4	1.83	21.6	35.5
Centralia, WI (Unit 2)	Full dschrg netting	White Sucker	-	-	125	14.4	15	90	6.1	0.71	3.3	97.9
Centralia, WI (Unit 1)	Full dschrg netting	Bluegill	-	-	125	14.4	15	90	6.1	0.71	3.3	98.2
Centralia, WI (Unit 1)	Full dschrg netting	Bluegill	-	-	175	14.4	15	90	6.1	0.71	3.3	86.8
Crown Zellerback, OR (Unit 20)	Full dschrg netting	Steelhead trout	1,777	500	-	11.6	-	277	11.9	-	-	69.4
Crown Zellerback, OR (Unit 20)	Full dschrg netting	Chinook salmon	1,800	500	-	11.6	-	277	11.9	-	-	71.6
Crown Zellerback, OR (Unit 21)	Full dschrg netting	Steelhead trout	17,999	500	-	14.7	-	255	13.0	-	-	80.0
Crown Zellerback, OR (Unit 21)	Full dschrg netting	Chinook salmon	1,798	500	-	14.7	-	255	13.0	-	-	81.2
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	22.6	17	300	137.2	2.11	33.1	61.0
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	40% wicket					-	59.0
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	40% wicket					-	44.6
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	40% wicket					-	52.2
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	65% wicket					-	77.3
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	65% wicket					-	70.9

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	65% wicket					-	65.5
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	80% wicket					-	75.0
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	80% wicket					-	73.7
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	80% wicket					-	55.1
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	100% wicket					-	73.5
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	100% wicket					-	69.1
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	100% wicket					-	63.8
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	22.6	17	300	137.2	2.11	33.1	53.3
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	22.6	17	300	137.2	2.11	33.1	42.9
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	40% wicket					-	34.5
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	50% wicket					-	50.9
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	50% wicket					-	51.9
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	60% wicket					-	59.9
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	60% wicket					-	38.6
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	68% wicket					-	60.5
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	68% wicket					-	42.3
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	76% wicket					-	72.0
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	76% wicket					-	50.0
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	84% wicket					-	68.4
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	84% wicket					-	33.8
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	90% wicket					-	77.7
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	100% wicket					-	64.9
E. J. West, NY	Full dschrg netting	Centrarchid	320	320	< 100	76.3	15	113	19.2	3.33	19.7	71.7
E. J. West, NY	Full dschrg netting	Centrarchid	159	160	175	76.3	15	113	19.2	3.33	19.7	85.5
E. J. West, NY	Full dschrg netting	Centrarchid	128	128	> 250	76.3	15	113	19.2	3.33	19.7	59.8
E. J. West, NY	Full dschrg netting	Percid	240	240	< 100	76.3	15	113	19.2	3.33	19.7	56.1
E. J. West, NY	Full dschrg netting	Soft Ray	157	159	< 100	76.3	15	113	19.2	3.33	19.7	32.3
E. J. West, NY	Full dschrg netting	Soft Ray	160	159	175	76.3	15	113	19.2	3.33	19.7	71.3
E. J. West, NY	Full dschrg netting	Soft Ray	160	160	> 250	76.3	15	113	19.2	3.33	19.7	67.5
E. J. West, NY	Full dschrg netting	Salmonid	280	280	< 100	76.3	15	113	19.2	3.33	19.7	65.2
E. J. West, NY	Full dschrg netting	Salmonid	160	160	175	76.3	15	113	19.2	3.33	19.7	90.6
E. J. West, NY	Full dschrg netting	Salmonid	160	160	> 250	76.3	15	113	19.2	3.33	19.7	95.6
Elwha, WA	Partial netting	Chinook salmon	42,168	20,030	-	14.1	-	300	31.7	1.49	23.4	100.0
Finch Pruyn, NY (Unit 4)	Balloon tag	Smallmouth Bass	61	44	191	20.0	15	225	14.0	0.91	10.8	95.0
Finch Pruyn, NY (Unit 4)	Balloon tag	Smallmouth Bass	49	37	210	20.0	15	225	14.0	0.91	10.8	91.0
Finch Pruyn, NY (Unit 5)	Balloon tag	Smallmouth Bass	32	37	210	23.6	15	225	14.0	0.91	10.8	91.0
Finch Pruyn, NY (Unit 5)	Balloon tag	Smallmouth Bass	43	44	271	23.6	15	225	14.0	0.91	10.8	71.0

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Five Channels, MI	Full dschrg netting	Bluegill	95	-	118	33.0	16	150	11.0	1.40	11.0	93.6
Five Channels, MI	Full dschrg netting	Bluegill	91	-	170	33.0	16	150	11.0	1.40	11.0	89.2
Five Channels, MI	Full dschrg netting	Gold./Common Shiner	59	-	114	33.0	16	150	11.0	1.40	11.0	81.8
Five Channels, MI	Full dschrg netting	Gold./Common Shiner	60	-	154	33.0	16	150	11.0	1.40	11.0	85.5
Five Channels, MI	Full dschrg netting	Northern Pike	31	-	352	33.0	16	150	11.0	1.40	11.0	91.3
Five Channels, MI	Full dschrg netting	Rainbow Trout	40	-	108	33.0	16	150	11.0	1.40	11.0	95.8
Five Channels, MI	Full dschrg netting	Rainbow Trout	46	-	317	33.0	16	150	11.0	1.40	11.0	70.0
Five Channels, MI	Full dschrg netting	Walleye	55	-	162	33.0	16	150	11.0	1.40	11.0	71.2
Five Channels, MI	Full dschrg netting	Walleye	60	-	385	33.0	16	150	11.0	1.40	11.0	76.7
Five Channels, MI	Full dschrg netting	White Sucker	56	-	180	33.0	16	150	11.0	1.40	11.0	88.6
Five Channels, MI	Full dschrg netting	White Sucker	60	-	290	33.0	16	150	11.0	1.40	11.0	71.4
Five Channels, MI	Full dschrg netting	Yellow Perch	30	-	186	33.0	16	150	11.0	1.40	11.0	77.1
Glines, WA	Partial netting	Silver salmon	31,256	23,442	-	42.4	-	225	59.1	2.35	27.6	69.6
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	76	18.2	15	90	8.5	1.47	6.9	96.7
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	127	18.2	15	90	8.5	1.47	6.9	100.0
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	178	18.2	15	90	8.5	1.47	6.9	94.9
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	76	18.2	15	90	8.5	1.47	6.9	100.0
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	127	18.2	15	90	8.5	1.47	6.9	100.0
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	178	18.2	15	90	8.5	1.47	6.9	94.9
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	229	18.2	15	90	8.5	1.47	6.9	93.7
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	292	18.2	15	90	8.5	1.47	6.9	90.4
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	>292	18.2	15	90	8.5	1.47	6.9	80.5
Hardy, MI (Unit 2)	Full dschrg netting	Bluegill	63	-	118	14.4	16	163.6	30.5	2.13	18.2	89.5
Hardy, MI (Unit 2)	Full dschrg netting	Bluegill	30	-	170	14.4	16	163.6	30.5	2.13	18.2	91.5
Hardy, MI (Unit 2)	Full dschrg netting	Gold./Common Shiner	30	-	114	14.4	16	163.6	30.5	2.13	18.2	85.5
Hardy, MI (Unit 2)	Full dschrg netting	Gold./Common Shiner	59	-	154	14.4	16	163.6	30.5	2.13	18.2	88.7
Hardy, MI (Unit 2)	Full dschrg netting	Largemouth Bass	60	-	118	14.4	16	163.6	30.5	2.13	18.2	76.2
Hardy, MI (Unit 2)	Full dschrg netting	Northern Pike	58	-	352	14.4	16	163.6	30.5	2.13	18.2	76.0
Hardy, MI (Unit 2)	Full dschrg netting	Rainbow Trout	59	-	108	14.4	16	163.6	30.5	2.13	18.2	71.4
Hardy, MI (Unit 2)	Full dschrg netting	Rainbow Trout	60	-	317	14.4	16	163.6	30.5	2.13	18.2	68.6
Hardy, MI (Unit 2)	Full dschrg netting	Walleye	60	-	385	14.4	16	163.6	30.5	2.13	18.2	77.3
Hardy, MI (Unit 2)	Full dschrg netting	White Sucker	59	-	180	14.4	16	163.6	30.5	2.13	18.2	76.9
Hardy, MI (Unit 2)	Full dschrg netting	White Sucker	60	-	290	14.4	16	163.6	30.5	2.13	18.2	64.5
Hardy, MI (Unit 2)	Full dschrg netting	Yellow Perch	60	-	107	14.4	16	163.6	30.5	2.13	18.2	83.1
Hardy, MI (Unit 2)	Full dschrg netting	Yellow Perch	-	-	186	14.4	16	163.6	30.5	2.13	18.2	95.5

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	154	88	76	7.8	12	358	25.3	0.99	18.6	85.5
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	90	48	127	7.8	12	358	25.3	0.99	18.6	78.1
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	111	70	178	7.8	12	358	25.3	0.99	18.6	58.9
High Falls (Unit 5)	Full dschrg netting	Fusiforms	146	95	76	7.8	12	358	25.3	0.99	18.6	87.8
High Falls (Unit 5)	Full dschrg netting	Fusiforms	81	49	127	7.8	12	358	25.3	0.99	18.6	67.9
High Falls (Unit 5)	Full dschrg netting	Fusiforms	184	79	178	7.8	12	358	25.3	0.99	18.6	48.4
High Falls (Unit 5)	Full dschrg netting	Fusiforms	96	66	229	7.8	12	358	25.3	0.99	18.6	46.2
High Falls (Unit 5)	Full dschrg netting	Fusiforms	160	58	292	7.8	12	358	25.3	0.99	18.6	20.1
High Falls (Unit 5)	Full dschrg netting	Fusiforms	71	41	>292	7.8	12	358	25.3	0.99	18.6	2.7
Holst, MI	Full dschrg netting	Brown Trout	150	150	85	8.5	-	360	43.3	-	-	45.1
Holst, MI	Full dschrg netting	Brook Trout	150	150	135	8.5	-	360	43.3	-	-	43.0
Holst, MI	Full dschrg netting	Brown Trout	150	150	220	8.5	-	360	43.3	-	-	22.8
Holst, MI	Full dschrg netting	Bluegill	150	150	65	8.5	-	360	43.3	-	-	19.7
Holst, MI	Full dschrg netting	Bluegill	150	150	115	8.5	-	360	43.3	-	-	75.0
Holtwood, PA(U10/single runner)	Balloon tag	American Shad	100	100	125	99.0	16	94.7	18.9	3.80	18.8	89.4
Holtwood, PA (U3/double runner)	Balloon tag	American Shad	100	80	125	99.0	17	102.8	18.9	2.84	15.3	83.5
la centrale Beaurharnois, QE	Float tag	American eel	100	-	888	197.9	13	75	24.1	5.38	21.1	84.2
Leaburg, OR	Full dschrg netting	Rainbow trout	1,249	624	-	31.1	-	225	27.1	2.29	26.9	95.2
Lequille, NS	Full dschrg netting	Atlantic salmon	-	-	-	9.9	13	519	118.0	1.37	37.3	52.0
Luray, VA	Full dschrg netting	American Eel	393	-	853	10.4	12	164	4.9	1.59	13.7	99.0
Minetto, NY	Full dschrg netting	Centrarchid	164	104	< 100	42.4	16	72	5.2	3.53	13.3	62.0
Minetto, NY	Full dschrg netting	Centrarchid	236	110	175	42.4	16	72	5.2	3.53	13.3	83.0
Minetto, NY	Full dschrg netting	Centrarchid	165	120	> 250	42.4	16	72	5.2	3.53	13.3	84.0
Minetto, NY	Full dschrg netting	Percid	133	117	< 100	42.4	16	72	5.2	3.53	13.3	80.0
Minetto, NY	Full dschrg netting	Percid	243	142	175	42.4	16	72	5.2	3.53	13.3	86.0
Minetto, NY	Full dschrg netting	Soft Ray	348	220	< 100	42.4	16	72	5.2	3.53	13.3	82.0
Minetto, NY	Full dschrg netting	Soft Ray	214	133	175	42.4	16	72	5.2	3.53	13.3	94.0
Minetto, NY	Full dschrg netting	Soft Ray	177	160	> 250	42.4	16	72	5.2	3.53	13.3	84.0
Minetto, NY	Full dschrg netting	Salmonids	237	160	< 100	42.4	16	72	5.2	3.53	13.3	92.0
Minetto, NY	Full dschrg netting	Salmonids	184	107	175	42.4	16	72	5.2	3.53	13.3	91.0
Minetto, NY	Full dschrg netting	Salmonids	178	159	> 250	42.4	16	72	5.2	3.53	13.3	92.0
Minetto, NY	Full dschrg netting	American Eel	107	92	625	42.4	16	72	5.2	3.53	13.3	94.0
Minetto, NY	Full dschrg netting	Alewife	189	140	<100	-	-	-	-	-	-	80.0
North Fork, OR	Partial netting	Coho salmon	4,076	5,158	-	70.7	-	139	41.5	2.95	21.4	74.0

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	146	84	76	13.0	15	100	4.0	2.03	10.6	100.0
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	140	77	127	13.0	15	100	4.0	2.03	10.6	98.9
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	121	75	178	13.0	15	100	4.0	2.03	10.6	100.0
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	158	103	76	13.0	15	100	4.0	2.03	10.6	94.0
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	141	90	127	13.0	15	100	4.0	2.03	10.6	93.7
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	166	109	178	13.0	15	100	4.0	2.03	10.6	96.6
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	158	93	229	13.0	15	100	4.0	2.03	10.6	95.4
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	166	105	292	13.0	15	100	4.0	2.03	10.6	85.5
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	128	79	>292	13.0	15	100	4.0	2.03	10.6	82.8
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	134	94	76	14.1	15	123	5.2	2.13	13.7	100.0
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	154	93	127	14.1	15	123	5.2	2.13	13.7	84.7
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	111	70	178	14.1	15	123	5.2	2.13	13.7	83.0
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	168	104	76	14.1	15	123	5.2	2.13	13.7	89.2
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	104	69	127	14.1	15	123	5.2	2.13	13.7	76.5
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	150	91	178	14.1	15	123	5.2	2.13	13.7	68.4
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	160	96	229	14.1	15	123	5.2	2.13	13.7	61.1
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	136	83	292	14.1	15	123	5.2	2.13	13.7	53.3
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	145	112	>292	14.1	15	123	5.2	2.13	13.7	34.5
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	166	105	76	14.1	15	123	5.2	2.13	13.7	93.4
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	137	104	127	12.4	15	135	5.2	2.03	14.4	83.7
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	58	28	178	12.4	15	135	5.2	2.03	14.4	91.4
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	179	123	76	12.4	15	135	5.2	2.03	14.4	84.5
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	134	93	127	12.4	15	135	5.2	2.03	14.4	61.7
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	138	92	178	12.4	15	135	5.2	2.03	14.4	75.1
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	158	98	229	12.4	15	135	5.2	2.03	14.4	61.0
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	156	91	292	12.4	15	135	5.2	2.03	14.4	57.8
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	149	85	>292	12.4	15	135	5.2	2.03	14.4	48.2
Pricket, MI	Full dschrg netting	Bluegill	256	150	52	9.2	15	257	16.5	1.36	18.3	97.7
Pricket, MI	Full dschrg netting	Golden Shiner	182	120	< 100	9.2	15	257	16.5	1.36	18.3	93.9
Pricket, MI	Full dschrg netting	Bluegill	131	90	102	9.2	15	257	16.5	1.36	18.3	92.5
Pricket, MI	Full dschrg netting	Bluegill	21	21	> 127	9.2	15	257	16.5	1.36	18.3	85.7
Pricket, MI	Full dschrg netting	White Sucker	201	119	165	9.2	15	257	16.5	1.36	18.3	70.8
Publishers, OR (1960)	Full dschrg netting	Steelhead trout	1,768	500	-	7.8	-	255	12.2	-	-	87.9
Publishers, OR (1960)	Full dschrg netting	Chinook salmon	1,798	503	-	7.8	-	255	12.2	-	-	87.4
Publishers, OR (1961)	Full dschrg netting	Steelhead trout	1,800	500	-	7.8	-	255	12.2	-	-	84.5
Publishers, OR (1961)	Full dschrg netting	Chinook salmon	1,800	500	-	7.8	-	255	12.2	-	-	87.1
Puntledge, BC	Floating net	Steelhead trout	1,500	-	124	-	-	277	103.6	2.16	31.4	58.1
Puntledge, BC	Floating net	Kamloops	1,500	-	69	-	-	277	103.6	2.16	31.4	72.5
Puntledge, BC	Floating net	Kamloops	1,500	-	46	-	-	277	103.6	2.16	31.4	71.2
Puntledge, BC	Floating net	Salmon	1,500	-	36	-	-	277	103.6	2.16	31.4	67.4

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Rogers, MI (Units 1 & 2)	Full dschrg netting	Bluegill	90	-	118	10.8	15	150	11.9	1.52	12.0	96.0
Rogers, MI (Units 1 & 2)	Full dschrg netting	Bluegill	92	-	170	10.8	15	150	11.9	1.52	12.0	85.2
Rogers, MI (Units 1 & 2)	Full dschrg netting	Gold./Common Shiner	60	-	114	10.8	15	150	11.9	1.52	12.0	
Rogers, MI (Units 1 & 2)	Full dschrg netting	Gold./Common Shiner	34	-	154	10.8	15	150	11.9	1.52	12.0	92.5
Rogers, MI (Units 1 & 2)	Full dschrg netting	Largemouth Bass	60	-	118	10.8	15	150	11.9	1.52	12.0	77.4
Rogers, MI (Units 1 & 2)	Full dschrg netting	Northern Pike	47	-	352	10.8	15	150	11.9	1.52	12.0	83.4
Rogers, MI (Units 1 & 2)	Full dschrg netting	Spottail Shiner	31	-	116	10.8	15	150	11.9	1.52	12.0	73.5
Rogers, MI (Units 1 & 2)	Full dschrg netting	Walleye	40	-	385	10.8	15	150	11.9	1.52	12.0	86.2
Rogers, MI (Units 1 & 2)	Full dschrg netting	White Sucker	55	-	180	10.8	15	150	11.9	1.52	12.0	91.2
Rogers, MI (Units 1 & 2)	Full dschrg netting	White Sucker	57	-	290	10.8	15	150	11.9	1.52	12.0	88.1
Rogers, MI (Units 1 & 2)	Full dschrg netting	Yellow Perch	78	-	107	10.8	15	150	11.9	1.52	12.0	91.8
Ruskin, BC	Fyke netting dwnstrm	Sockeye Salmon	12,125	12,159	86	113.1	-	120	39.6	3.78	23.8	89.5
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	165	99	76	18.4	15	150	12.8	2.21	17.3	97.0
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	141	90	127	18.4	15	150	12.8	2.21	17.3	80.7
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	61	53	178	18.4	15	150	12.8	2.21	17.3	79.9
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	169	100	76	18.4	15	150	12.8	2.21	17.3	64.9
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	132	96	127	18.4	15	150	12.8	2.21	17.3	75.0
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	145	97	178	18.4	15	150	12.8	2.21	17.3	76.0
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	127	78	229	18.4	15	150	12.8	2.21	17.3	69.8
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	119	71	292	18.4	15	150	12.8	2.21	17.3	58.4
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	144	92	>292	18.4	15	150	12.8	2.21	17.3	47.1
Schaghticoke, NY	Full dschrg netting	Centrarchid	160	160	175	11.6	17	300	43.6	2.03	31.9	59.0
Schaghticoke, NY	Full dschrg netting	Percid	239	237	<100	11.6	17	300	43.6	2.03	31.9	68.0
Schaghticoke, NY	Full dschrg netting	Soft ray	160	160	<100	11.6	17	300	43.6	2.03	31.9	60.0
Schaghticoke, NY	Full dschrg netting	Soft ray	149	150	>250	11.6	17	300	43.6	2.03	31.9	22.0
Schaghticoke, NY	Full dschrg netting	Salmonid	159	160	<100	11.6	17	300	43.6	2.03	31.9	56.0
Seton Creek, BC	Fyke net in tailrace	Sockeye Salmon	-	-	86	127.2	-	120	43.3	3.66	23.0	90.8
Shasta, CA (January)	Full dschrg netting	Chinook Salmon	4,800	•	102	41% wicket	15	138.5	115.8	4.67	33.9	57.6
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	•	254	41% wicket	15	138.5	115.8	4.67	33.9	58.6
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	•	152	41% wicket	15	138.5	115.8	4.67	33.9	79.0
Shasta, CA (January)	Full dschrg netting	Chinook Salmon	4,800	•	102	50% wicket	15	138.5	115.8	4.67	33.9	60.4
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	•	254	50% wicket	15	138.5	115.8	4.67	33.9	53.1
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	•	152	50% wicket	15	138.5	115.8	4.67	33.9	75.4
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	•	254	55% wicket	15	138.5	115.8	4.67	33.9	58.8
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	•	152	55% wicket	15	138.5	115.8	4.67	33.9	81.6
Shasta, CA (January)	Full dschrg netting	Chinook Salmon	4,800	•	102	60% wicket	15	138.5	115.8	4.67	33.9	72.1
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	•	254	60% wicket	15	138.5	115.8	4.67	33.9	66.2

Table 4.2-4

Physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines for which survival data were deemed usable for direct effects (analysis utilized these data).

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Peripheral Velocity (m/s)	% Survival 1 hr
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	*	152	60% wicket	15	138.5	115.8	4.67	33.9	78.6
Shasta, CA (January)	Full dschrg netting	Chinook Salmon	4,800	*	102	65% wicket	15	138.5	115.8	4.67	33.9	54.8
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	*	254	65% wicket	15	138.5	115.8	4.67	33.9	71.2
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	*	152	65% wicket	15	138.5	115.8	4.67	33.9	89.3
Shasta, CA (November)	Full dschrg netting	Chinook Salmon	11,500	*	102	40% wicket	15	138.5	115.8	4.67	33.9	61.7
Shasta, CA (November)	Full dschrg netting	Steelhead	4,400	*	152	40% wicket	15	138.5	115.8	4.67	33.9	39.6
Shasta, CA (November)	Full dschrg netting	Rainbow Trout	1,025	*	254	40% wicket	15	138.5	115.8	4.67	33.9	50.5
Shasta, CA (November)	Full dschrg netting	Chinook Salmon	11,500	*	102	50% wicket	15	138.5	115.8	4.67	33.9	69.7
Shasta, CA (November)	Full dschrg netting	Steelhead	4,400	*	152	50% wicket	15	138.5	115.8	4.67	33.9	72.3
Shasta, CA (November)	Full dschrg netting	Rainbow Trout	1,025	*	254	50% wicket	15	138.5	115.8	4.67	33.9	59.8
Shasta, CA (November)	Full dschrg netting	Chinook Salmon	11,500	*	102	55% wicket	15	138.5	115.8	4.67	33.9	77.9
Shasta, CA (November)	Full dschrg netting	Steelhead	4,400	*	152	55% wicket	15	138.5	115.8	4.67	33.9	90.5
Shasta, CA (November)	Full dschrg netting	Rainbow Trout	1,025	*	254	55% wicket	15	138.5	115.8	4.67	33.9	69.2
Shasta, CA (November)	Full dschrg netting	Chinook Salmon	11,500	*	102	61% wicket	15	138.5	115.8	4.67	33.9	84.5
Shasta, CA (November)	Full dschrg netting	Steelhead	4,400	*	152	61% wicket	15	138.5	115.8	4.67	33.9	59.8
Shasta, CA (November)	Full dschrg netting	Rainbow Trout	1,025	*	254	61% wicket	15	138.5	115.8	4.67	33.9	68.1
Stevens Creek, SC	Balloon tag	Bluegill	110	110	122	28.3	14	75	8.5	3.43	13.5	95.4
Stevens Creek, SC	Balloon tag	Blueback Herring	131	120	203	28.3	14	75	8.5	3.43	13.5	95.3
Stevens Creek, SC	Balloon tag	Spotted Sucker/Y. Perch	120	120	165	28.3	14	75	8.5	3.43	13.5	98.3
T. W. Sullivan, OR	Discharge netting	Steelhead trout	-	-	-	-	-	242	12.5	-	-	74.1
T. W. Sullivan, OR	Discharge netting	Chinook salmon	-	-	-	7.4	-	242	12.5	-	-	85.7
Vernon, VT/NH	Balloon tag	American Shad	153	150	95	51.9	15	74	10.4	3.96	15.3	94.7
White Rapids, WI	Balloon tag	White Sucker	42	36	204	25.4	14	100	8.8	3.40	17.8	93.0
White Rapids, WI	Balloon tag	White Sucker	58	64	112	25.4	14	100	8.8	3.40	17.8	100.0
White Rapids, WI	Balloon tag	Bluegill	56	62	90	25.4	14	100	8.8	3.40	17.8	95.0
White Rapids, WI	Balloon tag	Bluegill	44	38	155	25.4	14	100	8.8	3.40	17.8	100.0
Youghiogheny, PA	Full dschrg netting	Alewife	Naturally entrained		51	21.2	-	-	36.6	-	-	0.1
Youghiogheny, PA	Full dschrg netting	Walleye	Naturally entrained		376	21.2	-	-	36.6	-	-	39.5
Youghiogheny, PA	Full dschrg netting	Rock bass	Naturally entrained		-	21.2	-	-	36.6	-	-	4
Youghiogheny, PA	Full dschrg netting	Yellow perch	Naturally entrained		-	21.2	-	-	36.6	-	-	7
Youghiogheny, PA	Full dschrg netting	Crapples	Naturally entrained		-	21.2	-	-	36.6	-	-	0.2
Youghiogheny, PA	Full dschrg netting	White sucker	Naturally entrained		-	21.2	-	-	36.6	-	-	9.5

* Composite number of fish introduced and their recapture rates; November tests - test=91.0% and control=73.8%, January tests - test=72% and control=66%.

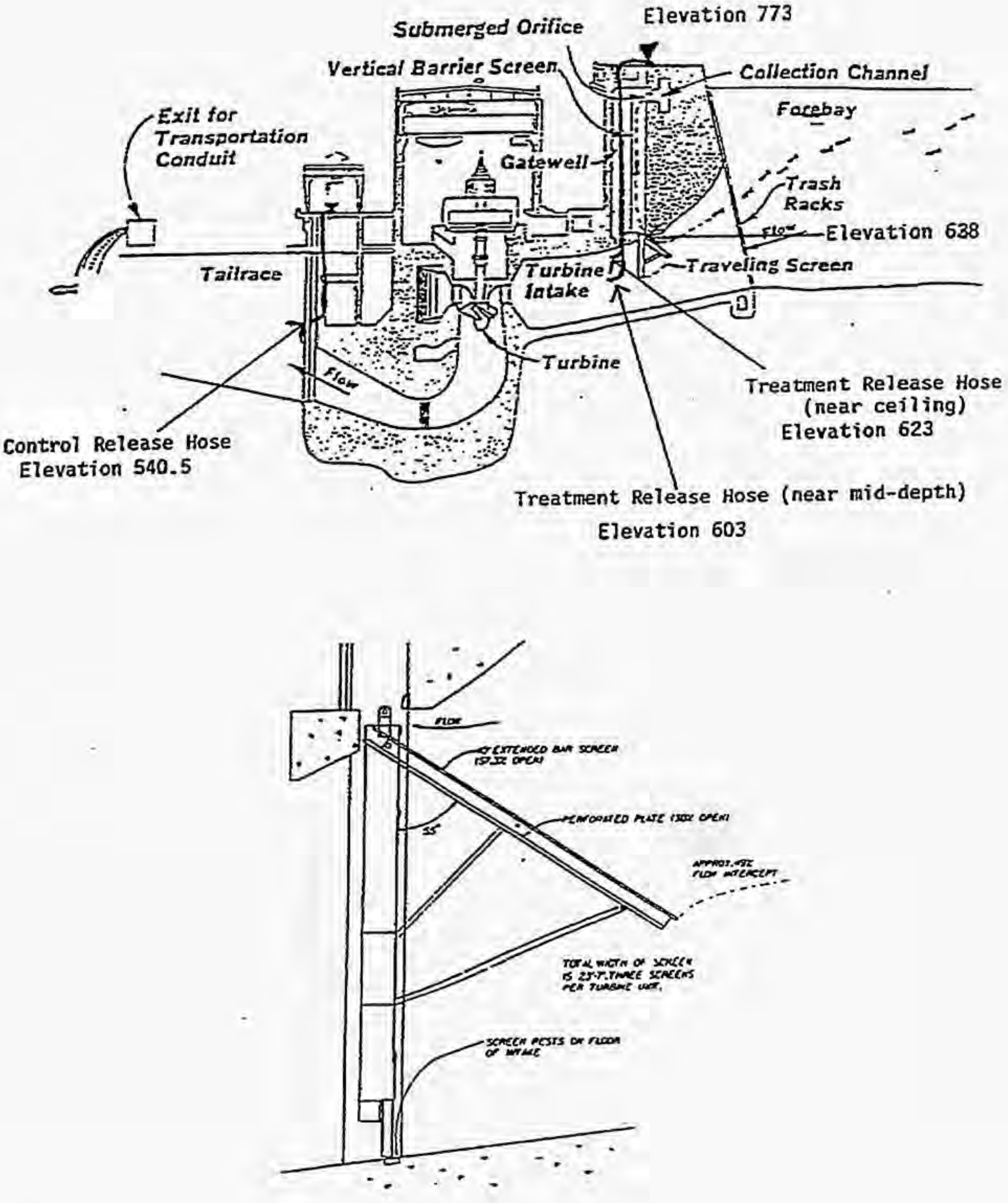
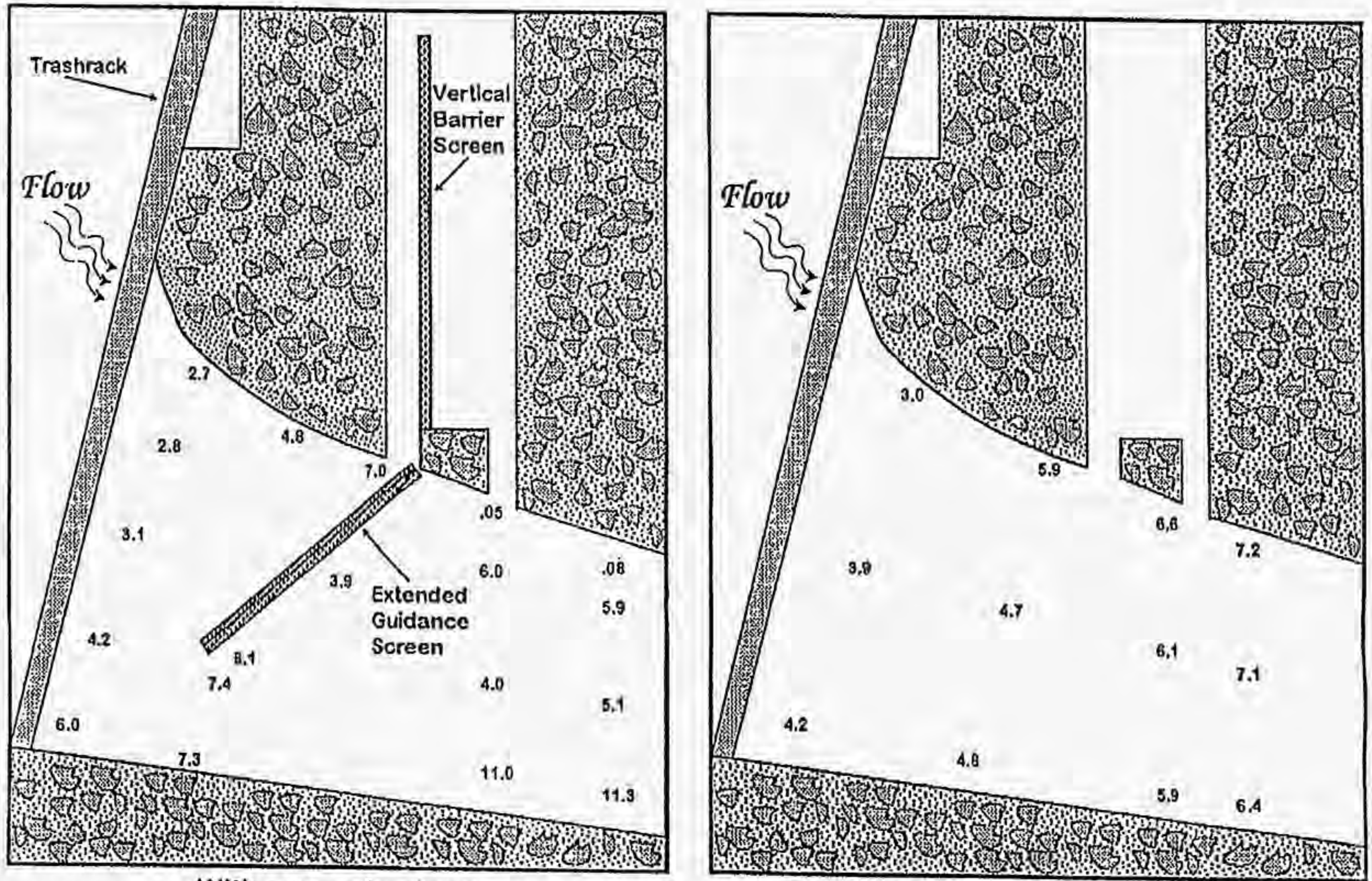


Figure 4.2-1

Turbine intake equipped with fish guidance extended length screens at Lower Granite Dam. From Normandeau Associates *et al.* (1995).

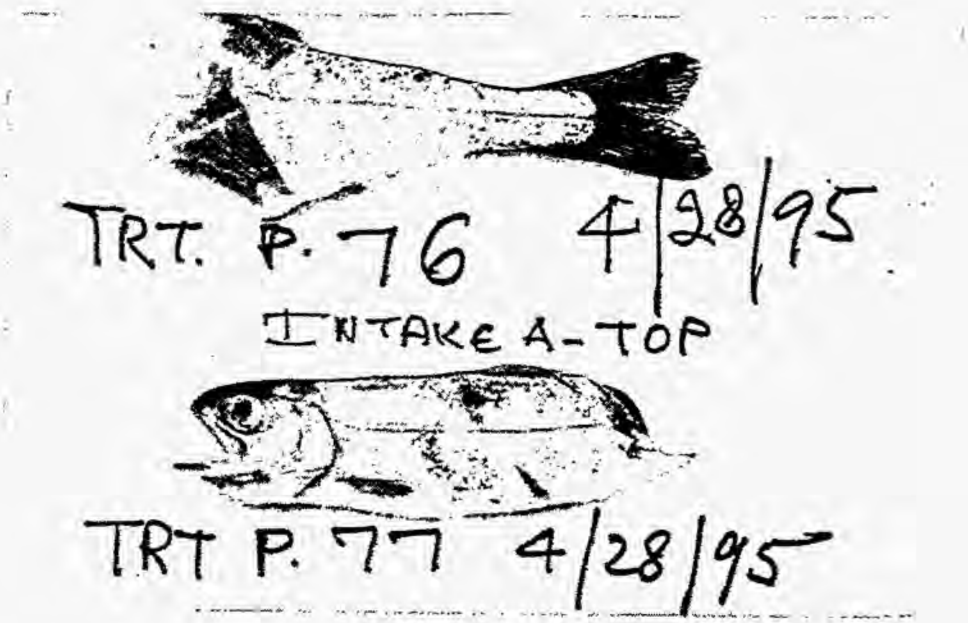


With screen in place.

Without screen.

Figure 4.2-2

Effect of extended length screens on velocity distribution. From Turner *et al.* (1993).

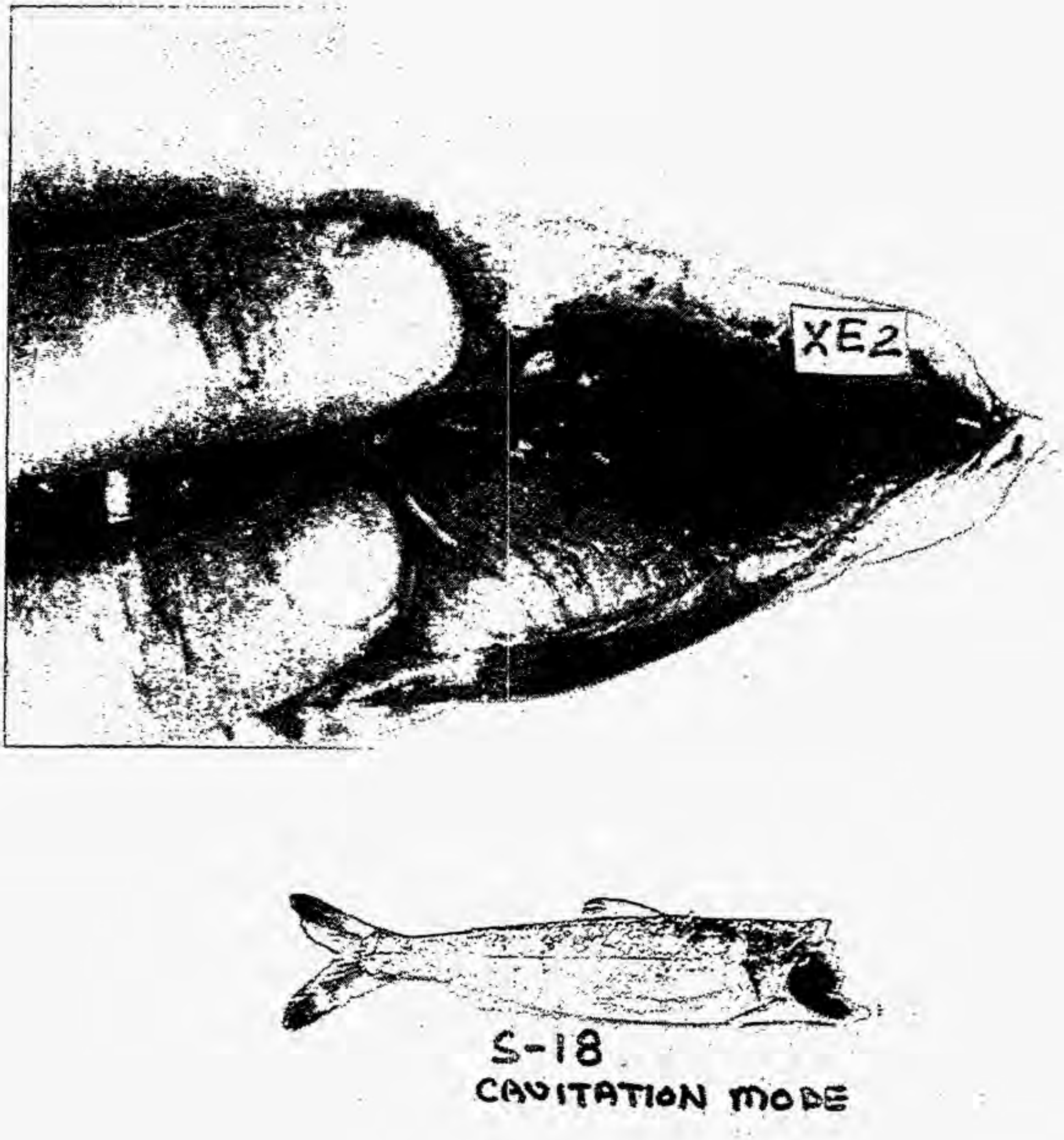


Blade strike

Pinched -
gap related

Figure 4.2-3

Example of mechanical injuries, blade and gap related. From Normandeau Associates *et al.* (1995).



Shear related

Figure 4.2-4

Examples of pressure (top photo) and shear related (bottom photo) injuries due to pressure. Note the hemorrhaged/ruptured kidneys in bottom photo. From Normandeau Associates and Skalski (1996).

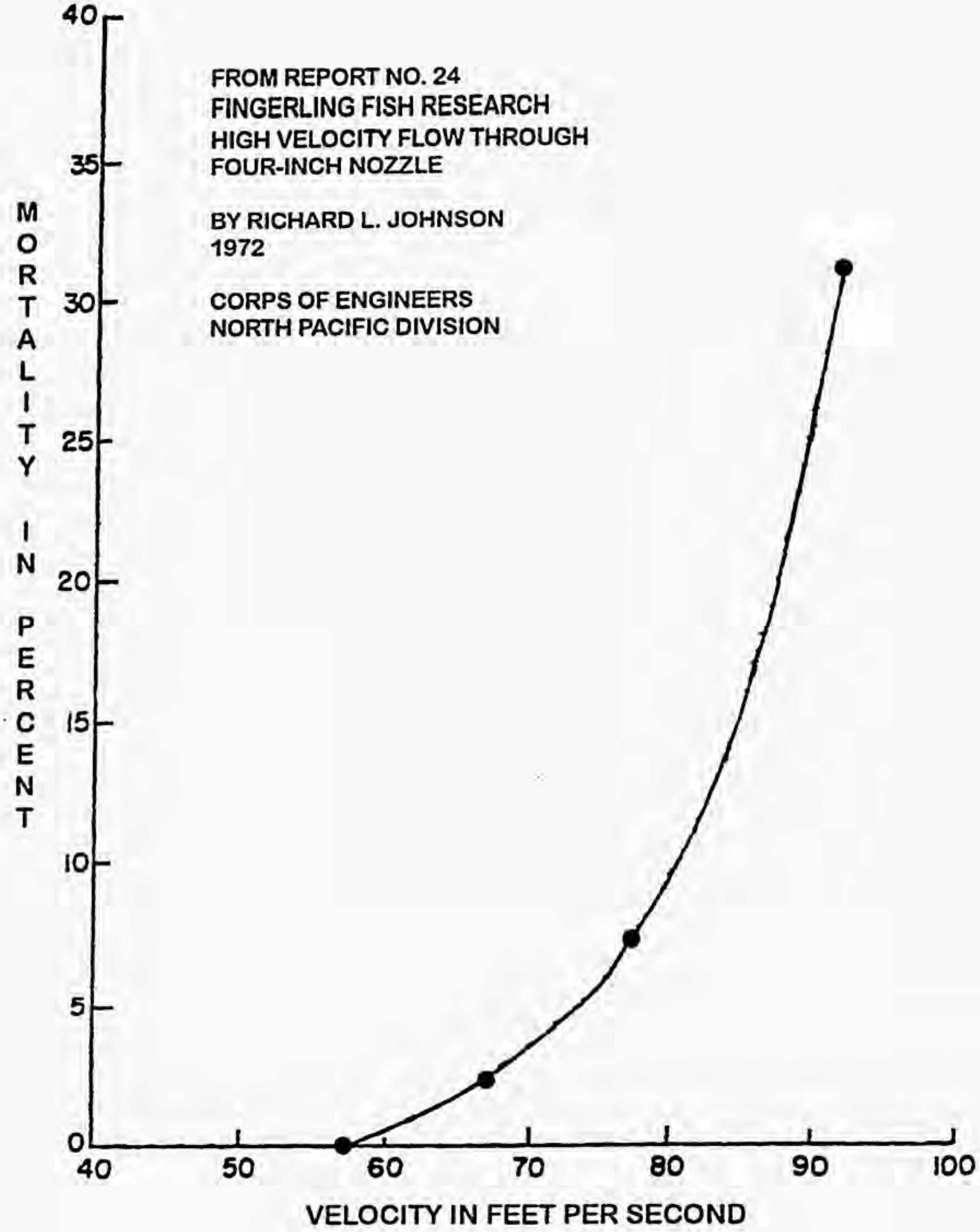


Figure 4.2-5

Effect of shear velocity on fingerling salmonid mortality. From Bell (1984).

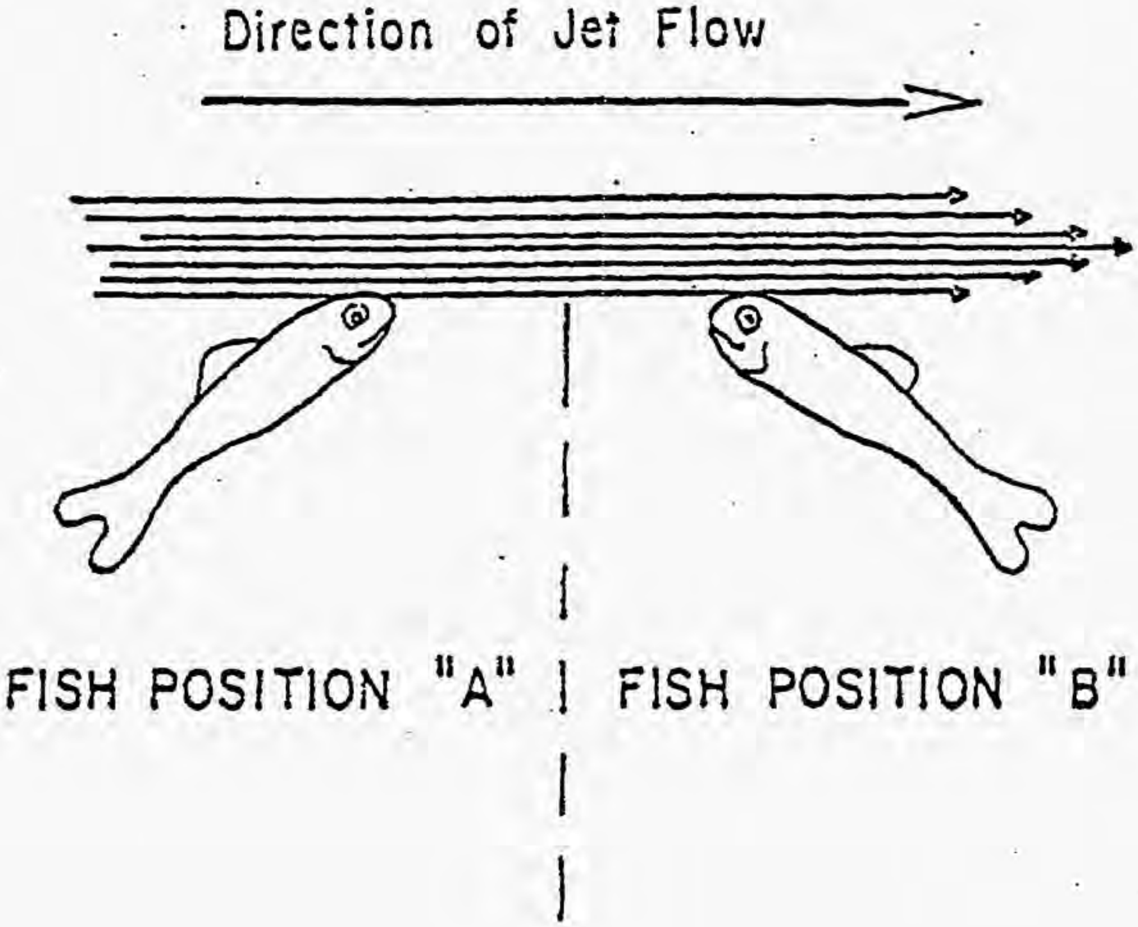


Figure 4.2-6

Effects of shear on the orientation of fish. From Groves (1972).

4.3 FISH SURVIVAL PREDICTION METHODS

4.3.1 MECHANISMS FOR FISH INJURY

4.3.1.1 Background

Overview of Prediction Methods
Fish survival prediction methods developed here are primarily oriented toward direct effects that may be verified by field studies of fish passage through turbines. Other effects that could cause indirect mortality such as stress induced effects leading to mortality after a longer period of time or abrasion related infection, disorientation, visual or other sensory impairment resulting in predation and so forth have not been explicitly considered. Predation, an indirect effect, resulting from turbine powerhouse design and its impact on the backroll in the tailrace is discussed in Section 4.3.3.5. The types of analyses performed here may be extended to other effects such as disorientation (leading to an increased probability of predation), if biological data become available showing the effects of quantified disorienting phenomena.

Several mechanisms inducing fish injury may be amenable to numerical prediction. These include mechanical, fluid and pressure mechanisms. Mechanical mechanisms inducing damage to the fish body may be classified as being related to:

- **leading edge strike** The effect of a fish impact on a turbine blade leading edge, possibly a gap between a blade and adjacent structure, a stay vane leading edge, a wicket gate leading edge, or the leading edge of a support pier in an intake or draft tube.
- **gap grinding** The effect of a fish caught in a narrow region formed between a blade and an adjacent component
- **abrasion** The effect of a fish sliding against a turbine structure
- **wall strike** The effect of a fish impact on a relatively flat turbine structure
- **mechanical chop** The effect of a fish cut by a rotating blade against a stationary wicket gate trailing edge

Fluid mechanisms inducing damage to the fish body may be classified as being related to:

- **excess energy dissipation (Theoretically Avoidable Loss)** The effect of a fish passing near or through a region where the flow is experiencing turbulence, velocity gradients, vortices, or related phenomena that dissipate more energy (cause more losses) than the most benign flow field that could exist in an idealized turbine flow field. These phenomena, when sufficiently intense, can create a force on a fish body of high enough level to cause damage or mortality. A variety of mechanisms will induce such damage, such as:
  - non-optimum incidence on blades, vanes, gates, etc.
  - flow through and downstream of gaps (blade or wicket gate overhang) and clearances which produce vortices and shear zones
  - vortices that form at a blade leading edge, or as a result of complex flow phenomena
  - non-optimum flow entering the draft tube or developing in the draft tube as a result of flow separation or flow interaction with pier structures.

- **cavitation** The effect of water vapor bubble collapse. The bubble which forms in a region of low pressure, moves to a region of higher pressure and collapses due to the increase in pressure above the vapor pressure. Depending on the shape of the vaporized region and the pressure and velocity gradients involved, the collapse may create intense local pressure waves, jets of high velocity fluid and regions of strong flow turbulence.

Pressure related mechanisms inducing damage to the fish body may be related to

- **pressure reduction** Injury resulting from the inability of fish to adjust from the regions of high pressure immediately upstream of the turbine to regions of low pressure downstream of the turbine. This includes both bladder rupture and gas embolism effects. The pressure change in the turbine environment itself may not be of sufficient magnitude and duration to be significant;

Two types of predictions have been considered: 1) global one-dimensional methods that are based on overall turbine characteristics, and 2) detailed three-dimensional methods that are based on precise details of the flow field that are calculated for all internal components for all operating conditions. Sections 4.3.2 through 4.3.4 discuss these methods in the application to survival prediction. For additional insight, Section 5.3 presents results of three-dimensional Computational Fluid Dynamics (CFD) flow field analysis methods applied to typical turbine geometry to quantify characteristics of turbine flow fields.

Several types of predictive approaches have been considered. In the study of mechanically induced damage, particularly leading edge strike, one dimensional methods with some guidance from 3-D methods have shown some success and are examined further. For fluid-induced damage mechanisms, another approach was developed. Here, the mortality caused by the dissipation of energy in the turbine was evaluated in a novel way. This evaluation does not explicitly calculate the individual effects of turbulence and strong velocity gradients, nor the effects of grinding, scraping or wall strike. Instead, the mortality is implicitly addressed based on empirical data through knowledge of the flow field. Based on the analysis of measured survival data, new insights into turbine operation to maximize fish survival were obtained. For shear, using the results of turbulence modeled two-dimensional calculations and correlation with previous experiments utilizing fish, a basis for prediction of a critical shear zone was developed

To assist in the quantification of pressure and time scales which are characteristic of the turbine environment so that the data can be used for further biological evaluation, pressure versus time determination was done for flow through typical turbine components and the results are presented in Section 5.3. The design of the power plant, and in particular, the turbine intake structure and its location in the dam, its length, the time of passage and so forth are important as the plant civil design may allow fish to become acclimated to a high pressure so that when fish are discharged into a low pressure in the tailrace, decompression trauma may occur. For this effect, no formal predictions have been made as they lay outside the scope of this study.

With better biological basis it may be possible to derive mortality predictions associated with fish body forces determined from the computation of some of these fluid and mechanically based effects arising from fish swimming through actual turbine environments. Sections 5.3 and 5.4 demonstrate some of the flow field features that can be calculated with modern three-dimensional turbulence modeled calculation methods.

Operation Limits and Hill Curves

The performance of a turbine is typically displayed on a hill curve as shown in Figure 4.3.1-1. On the abscissa(x axis) the head (H) or head coefficient (E_{ed}) is presented. On the ordinate (y axis) the discharge rate (Q) or discharge coefficient (Q_{ed}) is presented. The head coefficient is based on the net head, the acceleration of gravity, as well as the rotational speed and the turbine diameter. The discharge is the volume of water discharged through the turbine per unit time. The discharge coefficient is based on the discharge as well as the rotational speed and the turbine diameter. The definitions of these coefficients are given below.

The use of coefficients rather than dimensional data for a particular size and speed turbine is a common practice. In this way the performance data of a turbine is known for any choice of turbine size, head, rpm, etc. If a turbine is constructed to be geometrically similar to another turbine (such as to a scale size laboratory model), then the turbine performance characteristics shown in coefficient form are also similar.

The turbine efficiency is presented in the form of iso-efficiency contours. Each of the contour curves represents a constant efficiency level. The center point of these contour curves is the best efficiency point of the machine. There are several reasons that the operation of a turbine is not always made at this best efficiency point. As conditions change the location of the operating point can move to different regions on the hill curve. As head varies, the point of operation moves to the left or right of the peak. As the demand for power varies, more or less water is required to be discharged and the operation moves above or below this optimum point. There are also operational limitations associated with power (generator limitations), cavitation, gate opening capability, and/or pressure pulsations. Operation of a Kaplan turbine anywhere on this performance curve requires operation of the machine at the "on-cam" combination of wicket gate opening and blade tilt. On-cam operation means that the wicket gate opening and the blade tilt vary with head and output in order to maximize the efficiency of the machine at the given head and discharge of the point of operation. If not operated on cam, efficiency values will be less and undesirable fluid and vibratory conditions can exist. For a more in depth presentation of operating characteristics of turbines, the reader is referred to ASME Hydro Power Technical Committee (1996).

4.3.1.2 **Nomenclature**

The reader is referred to Figures 4.3.2-1 through 4.3.2-3 to help understand the variables defined here.

variable	description	(units)
B	Runner height at inlet	(length)
d	Distance having shear greater than the critical value	(length)
D	Diameter of runner	(length)
D*	Non-dimensional shear distance = d / blade spacing	(-)
D_1	Diameter of runner at inlet	(length)
D_2	Diameter of runner at discharge	(length)
D_{2M}	Mean diameter of runner at discharge $= 0.707 D_2$	(length)
g	Acceleration of gravity	(length^2 / time)
H	Turbine net head	(length)
K	Proportionality factor relating losses to Q^2	(none used)
K_{tm}	Theoretical minimum value of K	(none used)
K_{TAL}	Theoretical avoidable value of K	(none used)
L	Fish length	(length)
N	Number of blades or buckets	(-)
P	Probability of strike	(-)

Q	Turbine discharge	(length^3 / time)
Q_{opt}	Turbine discharge at best efficiency	(length^3 / time)
r	Radius	(length)
R	Maximum radius	(length)
RPM	Revolutions per minute	
t	Time for passage of fish or runner blade	(time)
V_a	Axial velocity	(length / time)
V_r	Radial velocity	(length / time)
Vref	Reference velocity	(length / time)
$V_{\theta 1}$	Absolute tangential velocity at runner inlet	(length / time)
$V_{\theta 2}$	Absolute tangential velocity at runner exit	(length / time)
α_t	angle to tangential of absolute flow upstream of runner (for Francis turbines)	(-)
α_a	angle to axial of absolute flow upstream of runner (for Axial flow turbines)	(-)
β	Relative flow angle at runner discharge	(-)
ξ	Ratio between Q with no exit swirl and Q opt	(-)
λ	Strike mortality correlation factor (lambda)	(-)
η	Turbine efficiency	(-)
ω	Rotational speed $= RPM \cdot \frac{2\pi}{60}$	(1/ time)
$E_{\omega d}$	Energy coefficient $= \frac{gH}{(\omega D)^2}$	(-)
$Q_{\omega d}$	Discharge coefficient $= \frac{Q}{\omega D^3}$	(-)

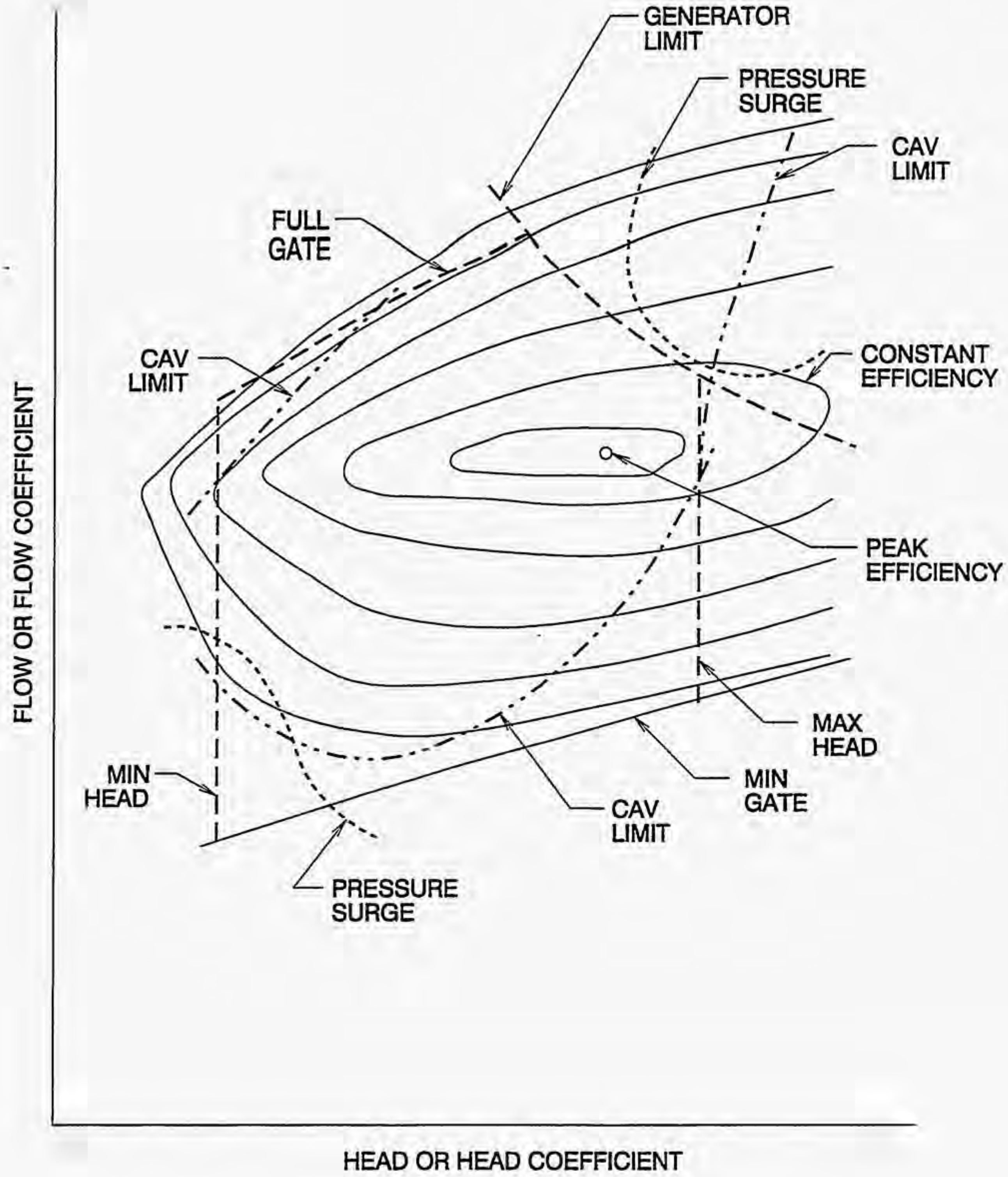


Figure 4.3.1-1 Turbine Operation is Quantified on a Hill Curve

4.3.2 MECHANICAL MECHANISMS LEADING TO FISH INJURY

4.3.2.1 Development of New Leading Edge Strike Equation

Summary

The existing strike equations are reviewed, an improved set of equations are developed and better methods are developed to apply the equations. The improvement to the equations is based on adding consideration for the tangential projection of the fish length. Better methods to apply the equations are based upon calculating the flow angles based on overall operating head and discharge parameters. New insights are obtained by analysis of a non-dimensional form of the equations. While their development is similar in concept, the equations for Kaplan, Propeller and Francis turbines are presented separately.

Discussion

Review of Existing Method

An existing method for strike prediction was examined (Von Raben 1957, cited by Bell 1991). This method considers the fish length, turbine runner size and number of buckets, turbine rpm, and assumes the fish is aligned with the local flow. The basis of the derivation is that the strike probability is the ratio of time for the meridional length of the fish to pass the leading edge of a runner blade divided by the time between passage of successive runner blades. The analysis assumes that the fish remains in a two-dimensional plane of revolution of a point at a given radius on the blade. Essentially, the fish is modeled as a meridional line segment and the blade is modeled as a point. It was also initially assumed that any impact by the blade along any portion of the fish length would be fatal. A factor to correlate experimental data was included. The phenomena that small fish (relative to blade size) may be transported around the blade leading edge was not considered.

Note that the terms mortality and survival are related. This section discusses the probability of strike, and with a correlation factor, strike could be considered to be related to mortality. Most of the presentation of experimental turbine passage data is presented in terms of survival. Survival can be determined from mortality, for example, using percentages, as:

Survival = 100% - Mortality

The physics of strike are the same for Francis turbines and for axial flow turbines (both Kaplan, Propeller and Bulb turbines), but the equations have slightly different form because of the different geometries of these turbines. Figure 4.3.2-1 and 4.3.2-2 illustrates the terminology used for the flow field variables and Figure 4.3.2-3 shows the geometry variables for Francis turbines. For example, for a Francis turbine, the times of passage are:

$$t_{runner} = \frac{\frac{\pi D_1}{N}}{\frac{\omega D_1}{2}}$$

$$t_{fish} = \frac{L \cdot \sin \alpha_r}{V_r}$$

The resulting Von Raben strike probability equations are:

Francis

$$P_{VonRuhen} = \frac{L \cdot \sin \alpha_t}{V_r} \cdot \frac{N \cdot \frac{\omega D_1}{2}}{\pi D_1}$$

Axial Flow

$$P_{VonRaben} = \frac{L \cdot \cos \alpha_a}{V_a} \cdot \frac{N \cdot \frac{\omega D}{2}}{\pi D}$$

Derivation of New Equation
Since the existing method has demonstrated some promise in correlating fish survival, the basic physics used to derive these equations was reexamined. It was observed that the meridional component of the fish length was considered, but the tangential component of length was not considered. The Von Raben equation gave results that could be physically unrealistic in some circumstances. As an example, in a situation where the tangential projection of the fish length is greater than the blade to blade spacing, it is not possible for a fish to pass through the entrance edge region of a runner without touching a runner blade, Figure 4.3.2-4. In this case, the actual strike probability is 100%. The strike probability calculated by the above equations gives a false value less than 100%.

By considering the tangential projection of fish length, a more accurate strike prediction is obtained. This contribution to strike probability is the ratio of the tangential projection of fish length to distance between successive runner blades, Figure 4.3.2-5. The additional contribution to strike probability is:

Francis

$$P_{\tan gential} = \frac{L \cdot \cos \alpha_t}{\frac{\pi D}{N}}$$

Axial Flow

$$P_{\tan gential} = \frac{L \cdot \sin \alpha_a}{\frac{2\pi \cdot r}{N}}$$

With some rearrangement, both contributions can be expressed as:

Francis

$$P = \frac{N \cdot L}{D} \cdot \left[\frac{\omega D_1 \cdot \sin \alpha_t}{2\pi \cdot V_r} + \frac{\cos \alpha_t}{\pi} \right]$$

Kaplan and Propeller

$$P = \frac{N \cdot L}{D} \cdot \left[\frac{\cos\alpha_a \cdot \omega}{2\pi \cdot V_a} + \frac{\sin\alpha_a}{\pi \frac{r}{R}} \right]$$

Although the terms N L / D have been grouped to clarify their importance, this form of the strike equations has several drawbacks. Key relationships between variables are not clear and their application to a specific turbine operating condition, i.e. calculation of the flow angle, requires a number of assumptions. Note that if the orientation of the fish were not equal to the local flow angle, and if this angle were available from other means, then the orientation could be used directly. The following section presents an alternative form of the new leading edge strike equations.

An Improved Form of the New Leading Edge Strike Equation

Summary

The use of the strike equations can be improved by replacing some of the ad hoc estimates of flow angle that had been used by previous investigators. The use of non-dimensional head and discharge parameters clarifies important relationships between variables. Additional generalizations were added to provide a correlation factor that may be based on experimental fish survival data.

Discussion

By using more accurate assumptions regarding the internal flow field in a turbine and expressing the result in a non-dimensional form, the usefulness and accuracy of the strike equations is greatly enhanced. Two main concepts have been used. The first is the use of Euler's equation to evaluate the flow angle based on known values of key operating parameters, such as head and discharge. The second concept is the use of non-dimensional parameters. Euler's equation states the reaction torque on the runner is equal to the change in angular momentum of the flow through the runner. As was done by other investigators, the discharge and cross sectional area are used to calculate an average through-flow velocity.

An additional correlation function, lambda (λ), is added to the equations to account for several factors. One is that the fish may not lie entirely in a plane of revolution. This could be caused by the forces that act on a three-dimensional body in a flow or fish free will. Another is, as has been suggested by other investigators, a length related fraction could be applied because an impact on a sensitive portion of the fish body, particularly the head region may be more damaging than an impact to a different region, such as near the tail. Yet another factor is the phenomena that the local details of the flow at the leading edge of a blade will transport a fish in a manner that can carry it around the leading edge. While readily observed in physical tests, this factor has not been quantified numerically at this time.

In addition to strike phenomena, the use of lambda extends the applicability of these equations to all injury mechanisms that are related to the variable N L / D. Such mechanisms could include mechanical mechanisms of leading edge strike and gap grinding as well as fluid induced mechanisms related to flow through gaps or other flow phenomena associated with blades. Subsequent discussion will conclude that these "strike" equations may be generalized and could be termed "Blade Zone Encounter" (BZE) equations. The lambda factor is further discussed in Sections 4.4.3 and 4.4.6.

Axial Flow Turbines

For axial turbines, the average axial velocity is assumed to be

$$V_a = \frac{Q}{\frac{\pi}{4} D^2}$$

and a useful form of Euler's equation is

$$gH\eta = \omega \cdot r\left(V_{\theta 1} - V_{\theta 2}\right) \text{ with}$$

$$\tan\alpha_a = \frac{V_{\theta 1}}{V_a}$$

A Kaplan turbine is double regulated with both wicket gates and runner blades being adjustable in position. When operated "on - cam", the blade and gate positions are coordinated to achieve the highest possible efficiency. As a result, the flow discharging the blades, entering the draft tube, has a small and reasonably constant amount of swirl (tangential velocity) at all operating conditions. A good assumption is to use $V_{\theta 2}$ equal to zero. Figure 4.3.2-6 illustrates the difference between the exit swirl characteristics of an adjustable blade machine, i.e., a Kaplan, versus a fixed blade machine such as a propeller turbine or Francis turbine. The complete strike equation including the lambda function can be expressed as:

Kaplan Turbine Leading Edge Strike Equation

$$P = \lambda \frac{N \cdot L}{D} \cdot \left[\frac{\cos\alpha_a}{8Q_{\omega d}} + \frac{\sin\alpha_a}{\pi \frac{r}{R}} \right]$$ with the value of α_a most conveniently obtained from

$$\tan\alpha_a = \frac{\pi E_{\omega d} \cdot \eta}{2Q_{\omega d} \frac{r}{R}}$$

Several important relationships can be deduced from this form of the equation. A primary variable is the non-dimensional grouping of N L / D. This is the ratio of fish length to blade spacing. The individual values of L, N, or D are not important, only their ratio is important. Fish strike probability is linearly proportional to this ratio. The use of non-dimensional head and flow variables shows that factors such as head, discharge, rpm, or tip speed do not appear individually. Therefore, they do not affect the strike probability. The variables head, discharge and rpm occur only in the terms $E_{\omega d}$ (the head coefficient) and $Q_{\omega d}$ (the discharge coefficient). For a given design of Kaplan turbine, the variables $E_{\omega d}$ and $Q_{\omega d}$ uniquely specify the operating point. A given design of Kaplan turbine with the same number of blades N can be built in different sizes, rpm, and for different heads to suit specific site requirements. Therefore, machines of varying size, head, etc. (such as a model in a laboratory or an extremely large unit) will have identical fish strike characteristics, and will be a function only of the non-dimensional grouping of L / D.

Figure 4.3.2-7 shows a strike probability calculation at the blade mid span non-dimensional radius r/R = 0.75 for all operating conditions for a particular Kaplan design. Since the values of the correlation factors are not yet determined, the strike probability is normalized by the value that occurs at the best efficiency point. The primary trend which can be observed is that strike probability decreases significantly as the discharge increases, and that the strike probability increases somewhat when operating at heads less than the optimum head. This leads to the proposition that higher discharges are beneficial for fish survival influenced by blade leading edge strike. This proposition is further evaluated in Section 4.4. Also, operation at heads lower than the optimum head will have higher strike probability and presumably higher mortality than operation at the optimum head.

The probability of strike varies with the radius along the blade entrance edge. Figures 4.3.2-8 and 4.3.2-9 show the calculated effect for two discharges as a function of radius. The currently developed formula as well as the previous formula are used to demonstrate that the previously accepted belief that strike probability is lower at the hub is valid for low discharges, but not at high discharges. Figure 4.3.2-10 presents the result of the calculated ratio of strike probability at the hub to strike probability at the periphery as a function of discharge.

The contribution of the additional term in the strike equation due to the tangential length of a fish was examined by calculating the strike probability for a number of Kaplan units. This calculation included the full range of specific speed for axial flow turbines, from three bladed Bulb units to seven bladed Kaplan's. Figure 4.3.2-11 shows that the additional term, although generally small compared to the overall strike probability, is growing in magnitude as specific speed decreases. For reference, a discussion of specific speed in contained in Appendix 10.2.

Propeller Turbine

A propeller machine is similar to a Kaplan, but has runner blades that are fixed to the hub. They can not be adjusted. The discharge swirl of such a machine varies continuously with the machine discharge. Under the assumption that at the propeller turbine's best efficiency point, its operation is similar to a Kaplan turbine, the exit swirl velocity, $V_{\theta 2}$ is assumed to be zero at the best efficiency point. Using this known relationship, $V_{\theta 2}$ is calculated for any other operating condition. The strike equation is the same as for a Kaplan, but the relationship for the flow angle can be expressed as:

Propeller Turbine Leading Edge Strike Equation

$$\tan\alpha_a = \frac{\frac{\pi}{2} E_{ed} \cdot \eta}{Q_{ed} \frac{r}{R}} + \frac{\frac{\pi}{8} \frac{r}{R}}{Q_{ed}} - \tan\beta \quad \text{where}$$

$$\tan\beta = \frac{\frac{\pi}{8} \frac{r}{R}}{Q_{ed} opt}$$

Francis Turbines

A similar procedure may be followed for Francis turbines. The average radial speed is calculated from the discharge and the area at the runner inlet.

$$V_r = \frac{Q}{\pi D_1 B}$$

This assumption is most applicable to turbines of low specific speed. For turbines of high specific speed, the meridional curvature of the water passage causes a variation in the flow velocity. The velocity near the band is higher than the average value and the velocity near the crown is lower than the average, Figure 4.3.2-12. The strike probability will vary along the inlet edge, from crown to band. The equations presented below are based on a constant inflow velocity from band to crown, therefore, it is likely that these equations will suffer reduced accuracy for higher specific speed Francis turbines. An attempt was made to approximate the velocity profile based on a simple radius of curvature method, however, the low accuracy of this approximate method does not seem to justify its additional complexity. Higher accuracy velocity profiles may be obtained through application of CFD analysis tools.

A useful form of Euler's equation is:

$$gH\eta = \omega\left(\frac{D_1 V_{\theta 1}}{2} - \frac{D_{2M} V_{\theta 2}}{2}\right) \text{ with}$$

$$\tan\alpha_t = \frac{V_r}{V_{\theta 1}}$$

In a similar manner to a propeller turbine, the discharge swirl from the runner changes continuously with discharge. It is assumed that a zero swirl discharge flow occurs at a user specified multiple of the discharge at the best efficiency point $\left(\xi \cdot Qopt\right)$. At this value of flow, the discharge conditions are known, in particular, the relative flow angle β. Using this known flow angle, $V_{\theta 2}$ is calculated for any other operating condition. Note that a mean radius is used at the runner discharge. The complete equation can be expressed as:

Francis Turbine Leading Edge Strike Equation

$$P = \lambda \frac{N \cdot L}{D} \cdot \left[\frac{\sin\alpha_t \cdot \frac{B}{D_1}}{2Q_{\omega d}} + \frac{\cos\alpha_t}{\pi}\right]$$ with the value of α_t most conveniently obtained from

$$\tan\left(90 - \alpha_t\right) = \frac{2\pi E_{\omega d} \cdot \eta}{Q_{\omega d}} \cdot \frac{B}{D_1} + \frac{\pi \cdot 0.707^2}{2Q_{\omega d}} \frac{B}{D_1}\left(\frac{D_2}{D_1}\right)^2 - 4 \cdot 0.707 \cdot \tan\beta \frac{B}{D_1}\frac{D_1}{D_2} \text{ and}$$

$$\tan\beta = \frac{0.707 \cdot \frac{\pi}{8}}{\xi \cdot Q_{ed} opt \left(\frac{D_1}{D_2}\right)^3}$$

A value of $\xi = 1.1$ is suggested as a typical value for a zero swirl discharge. The resulting leading edge strike probability is shown in Figure 4.3.2-13. The strike probability is normalized by the value at the best efficiency point. Since the flow mechanisms are different than a Kaplan turbine, the resulting strike probability is different. Generally, the strike probability increase at lower heads (similar to the Kaplan) but the strike probability can increase with increasing discharge. This strike probability increase is caused by the change in the flow angle compared to the change in discharge and occurs differently than in a Kaplan turbine. The implications for fish passage are that increased discharge may not contribute to lower mortality in the same manner as is expected for Kaplan turbines.

The accuracy of the improved method for calculating values of flow angle, α_t and α_a is evaluated in Appendix 10.4

4.3.2.2 **Gap Grinding**

Summary

The presence of a gap as defined below creates a source of both mechanical and fluid mechanisms which may injure fish. The prediction of mortality due to gap induced velocity fields and mechanical gap grinding requires an advancement of technology from today's basis. Estimations of the gap related effect on fish injury are developed based on two experimental observations discussed in Section 4.4.5.

Discussion

Kaplan blades rotate with a hub around the shaft axis of rotation and also in the hub about a blade axis of rotation and change position as the turbine operates. To accommodate the rotation on the unit in the stationary discharge ring and the blades within the hub assembly between high blade tilts and low blade tilts, traditional Kaplan blades have clearances and gaps. Before discussing gaps, it is important to differentiate between clearances and gaps. Clearance describes the minimal distance required between two surfaces for relative movement of those two surfaces. In order for the individual blades to change tilt within the runner, there must be a minimal clearance between the inner radius of the blades and the hub (approximately 0.00035 times the runner diameter). In order for the runner assembly to rotate within the stationary discharge ring there must be a minimal clearance between the outer radius of the blades and the discharge ring (approximately 0.0007 times the runner diameter). These minimal clearances are unavoidable; they must exist for the unit to function. Typically, these clearances are much smaller than a typical fish, and are presumed to have minimal impact on fish mortality. The terminology "gap" as used here describes a distance that is significantly greater than a clearance. The causes of gaps are described below.

In a conventional Kaplan design the hub and discharge ring surfaces are combinations of spherical, cylindrical and conical sections in the areas where they are in close proximity to the blades. The hub and discharge ring surface shapes are designed to accommodate mechanical needs such as disassembly, or to provide space for conventional blade servomotor and linkage mechanisms to operate. Figure 6.2.1-2 shows a Kaplan blade at minimum and maximum tilts and the gaps typically created by cylindrical and conical sections on the hub and periphery. With the blade set at its maximum tilt, the inner radius surface of the blade is machined to fit any adjacent non-spherical surface of the hub. At maximum tilt there is no gap between the blade and hub. At minimum tilt a significant gap is created between the blade and hub. Some turbines have had special considerations whereby gaps between the blades and the hub have been minimized in certain regions between the blade and the hub. Because of the ease of assembly and disassembly, most Kaplan turbines are provided with discharge rings having a cylindrical surface upstream of the blade centerline and with a partially spherical surface downstream of the blade centerline. With the blade set at its minimum tilt, the outer radius surface of the blade is machined to fit any adjacent non-spherical surface of the discharge ring. At minimum tilt there is no gap between the blade and discharge ring. At maximum tilt there is a significant gap between the blade and discharge ring, particularly upstream and occasionally at the lower region of the discharge ring as the blade extends beyond the often provided partially spherical segment. Gaps can exist in other locations in the turbine. For example, some wicket gate and discharge ring designs are such that the wicket gate overhangs the top of the discharge ring at high gate openings creating a gap. Thus, gap geometries vary significantly depending on the turbine design and the point of operation of the turbine.

Gaps between the blade and hub may cause significant mortality (from approximately 2% to more than 5%) for the fish entering the turbine near the roof of the intake and passing through the runner at it's inner radius (Section 4.4.5). Similar gap related mortality may exist for the fish entering the turbine blades near the outer blade diameter. Fish in this outer diameter region could come from those near the floor elevation of the intake which pass through the runner at it's outer radius (Section 4.4.6) or come from fish which entered the turbine intake at other elevations but were then transported to the region of the blade tip gaps because of cross flow currents caused by fish diversion devices, or by large separation regions or secondary flows in the intake, stay vanes, wicket gates or the runner inlet. Gap related mortality includes not only the effect of mechanical gap grinding (a fish caught in the narrow gap), but mortality arising from fluid related effects such as excess energy dissipation effects (Section 4.3.3).

The prediction of gap grinding is not amenable to a one-dimensional analysis other than to account for the presence or absence of a gap. A preliminary CFD analysis is presented in Section 5.3.4 that verifies some of the expectations of the flow through a gap. A prediction of the loads applied to a fish being caught in this region would require a CFD calculation including the presence of a "virtual fish" . Gap effects are related to the number of blades.

Observations of carefully planned and executed experiments do give some indications of mortality that has occurred in these regions. Section 4.4.5 reviews the results of a fish passage survival determination at the Rocky Reach Dam and concludes that a gap at the trailing edge of a blade caused approximately 4.4% additional mortality for a particular operating condition. Section 4.4.6 reviews the results of a fish passage survival determination at Wanapum Dam and concludes that fish injected at a location expected to transport the fish near the runner inner radius experienced approximately 3% additional mortality in comparison to fish passing through the middle section of the blades. This mortality is attributed to fluid and mechanical effects related to the blade leading edge gap and other geometries in the vicinity of the runner hub.

4.3.2.3 Abrasion

Summary
Use of 3-D advanced CFD methods is probably necessary for abrasion prediction. However, a new insight into the role of cross sectional area is presented here. The probability of occurrence of scrape due to the space between blades is unknown.

Discussion

While abrasion is a mechanical effect arising from a fish scraping a mechanical surface, it can also be related to the degree of excess fluid energy dissipation in a design, and therefore the point of operation of a turbine and to the overall quality of the hydraulic design. As an example, fish in a swirling draft tube flow may have a higher chance for wall scrape. A fish having a large size in relationship to the minimum space between turbine blades (the vent dimension) could lead to scrape. The scrape effect related to vent depends on turbine type. The blades of Francis and Propeller turbines are fixed in position and have a fixed value of vent. Kaplan turbine blades change their position as a function of operating point. At low blade tilts (openings), the space between blades is smaller, while at high blade tilts, which correspond to high discharge, the space is larger. Also the smaller number of blades in Kaplan and Propeller turbines will give larger values of vent.

The prediction of scrape generally requires the use of advanced CFD analysis tools for any quantitative assessment. It is expected that scrape could occur due to both primary and secondary flows, centrifugal and buoyancy effects, or fish volitional movement. The flow field effects leading to abrasion will be discussed in Section 4.3.3, Fluid Mechanisms Leading to Fish Injury.

One aspect of scrape, however, that can be clarified is the role of cross sectional area. Several references have presumed that a turbine design goal should be to maximize the cross sectional area through the turbine. Figure 4.3.2-14 illustrates the concept of "vent" of a turbine. The vent of a turbine is the minimum distance between adjacent blades. It occurs near the blade trailing edge. In general, a fish passing through the turbine runner would experience a continuous decrease in cross sectional area. If the body of the fish were not flow aligned, and if the fish length were larger than the vent dimension, then it seems possible that the decreasing cross sectional area could cause the fish to scrape against a runner blade.

The value of vent for a particular turbine design is determined by the number of blades and the local geometry of the blade outlet edge, particularly the blade angle. A smaller number of blades will have a larger vent. For a given design condition, (number of blades, head, discharge, rpm, etc.) the blade angle is essentially fixed, therefore, the vent is fixed and can not be arbitrarily adjusted.

The frequency of scrape type injuries and factors that influence scrape are uncertain. The fish survival studies that have attributed sources of injury to possible causes are limited.

4.3.2.4 **Wall Strike**

Summary

Use of 3-D advanced CFD methods is probably necessary for wall strike prediction

Discussion

While wall strike could be thought of as similar to abrasion, we differentiate it here as a mechanical injury caused by a fish body impact with a wall with enough energy to cause mechanical injury. For this to occur a relatively high rate of change of velocity over a distance related to fish length is necessary. While it is basically a mechanically induced injury, wall strike is assumed to be induced by vortices and high turbulence levels related to the fish length. It is related to the point of operation and the quality of the hydraulic design. Section 5 addresses fluid conditions that could lead to wall strike.

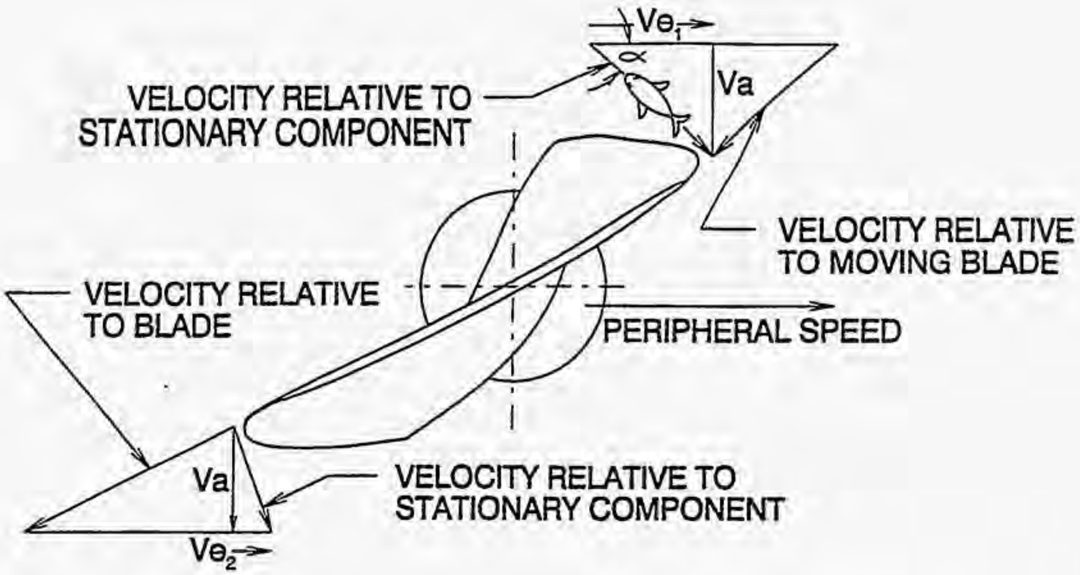


Figure 4.3.2-2 Illustration of Axial Flow Turbine Flow Inlet and Outlet Velocities and Angles

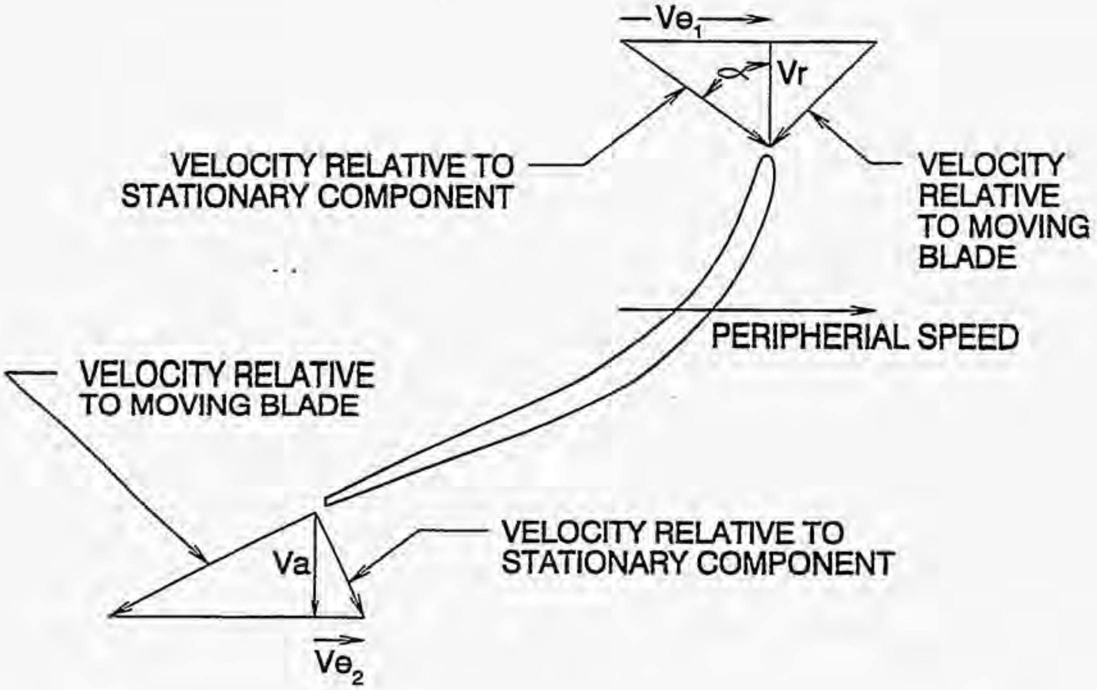


Figure 4.3.2-1 Illustration of Francis Turbine Flow Inlet and Outlet Velocities and Angles

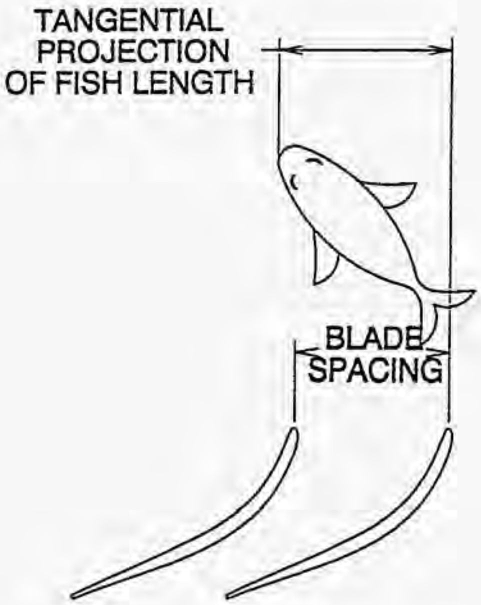


Figure 4.3.2-4 Illustration of Tangential Fish Length that Exceeds Blade Spacing

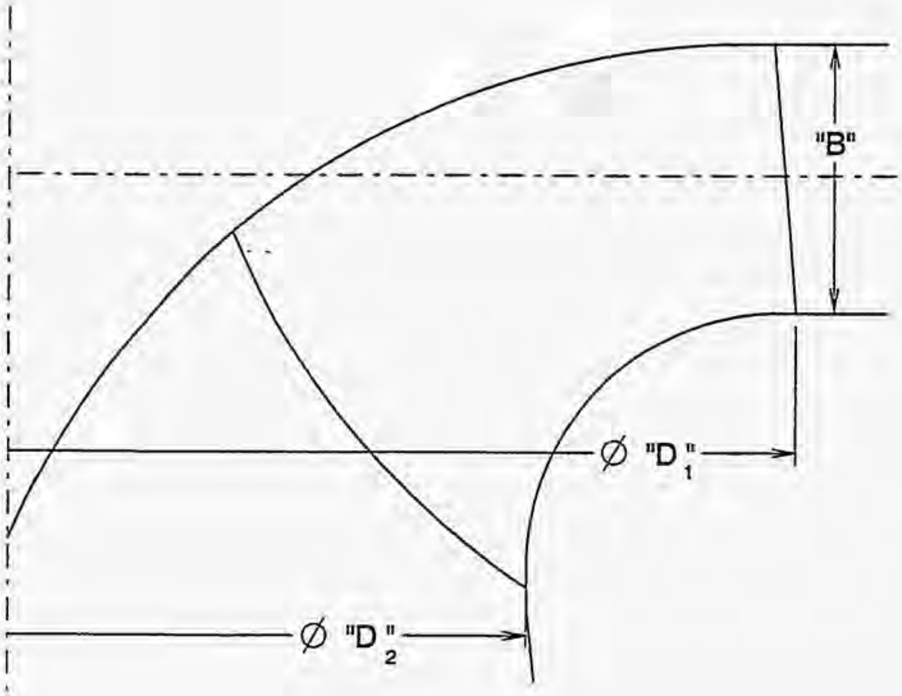


Figure 4.3.2-3 Geometry Notation for Francis Turbines

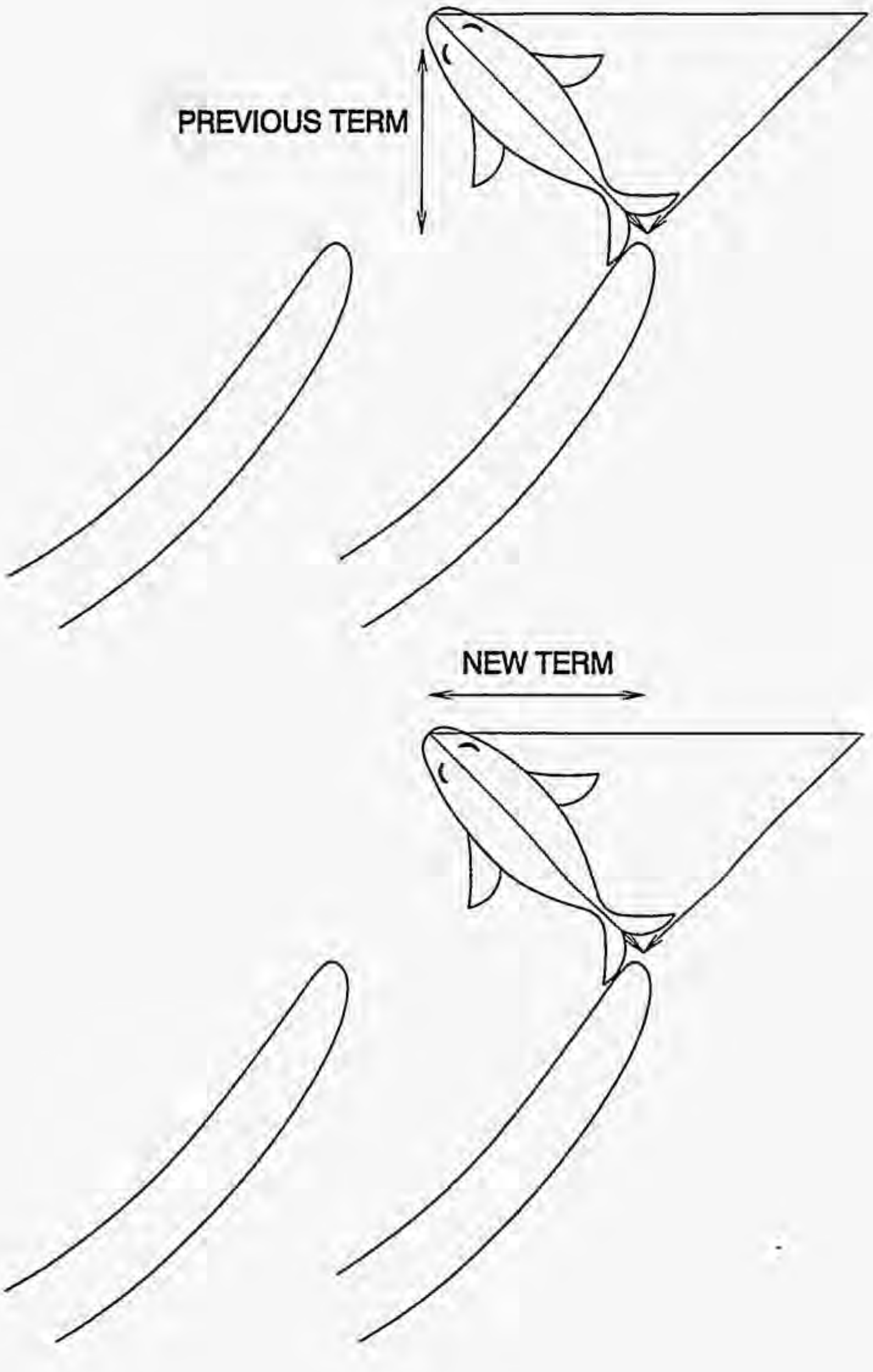


Figure 4.3.2-5 Illustration of Additional Term in Leading Edge Strike Equation

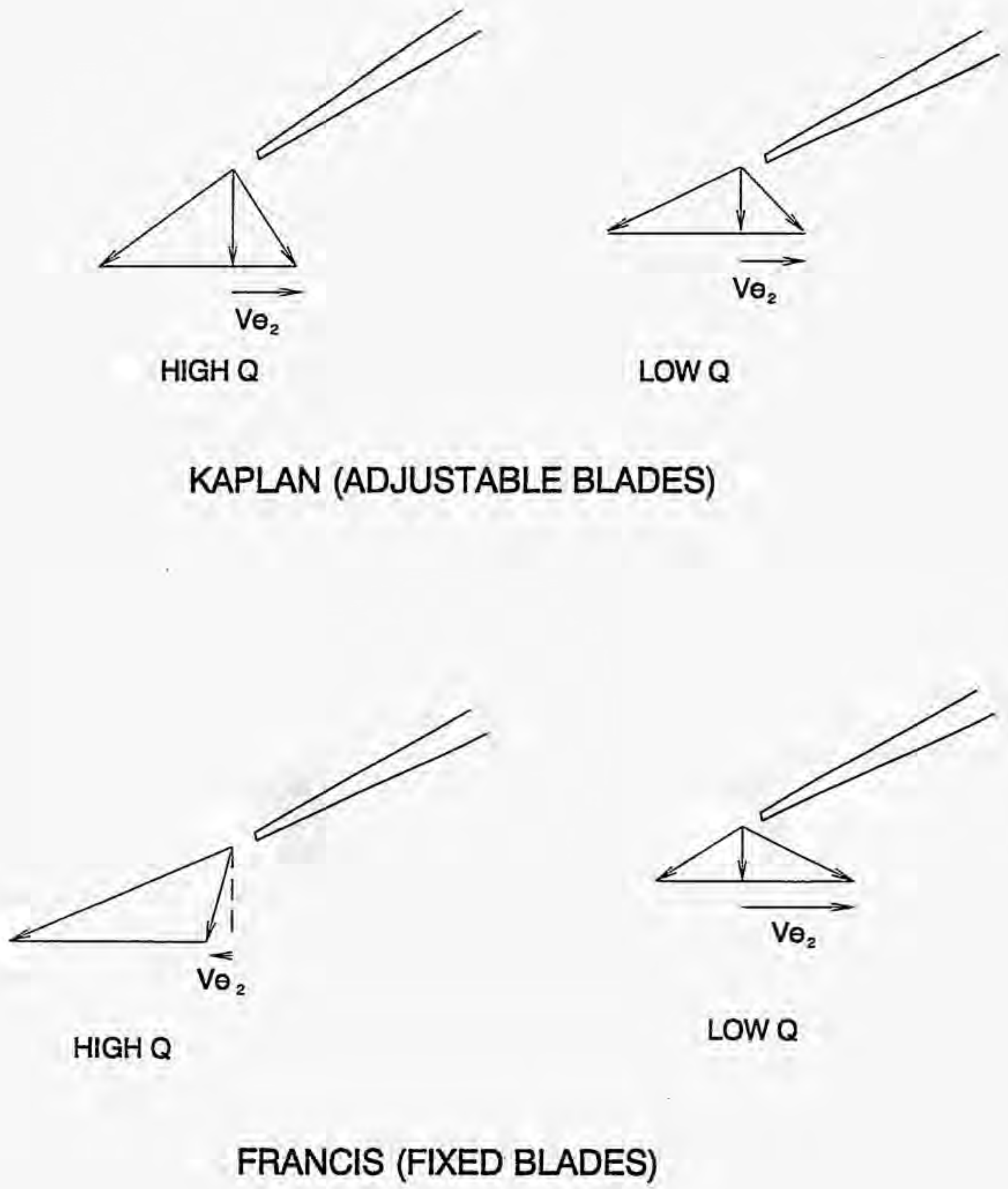


Figure 4.3.2-6 Exit Swirl Characteristics of Francis and Adjustable Blade Turbines

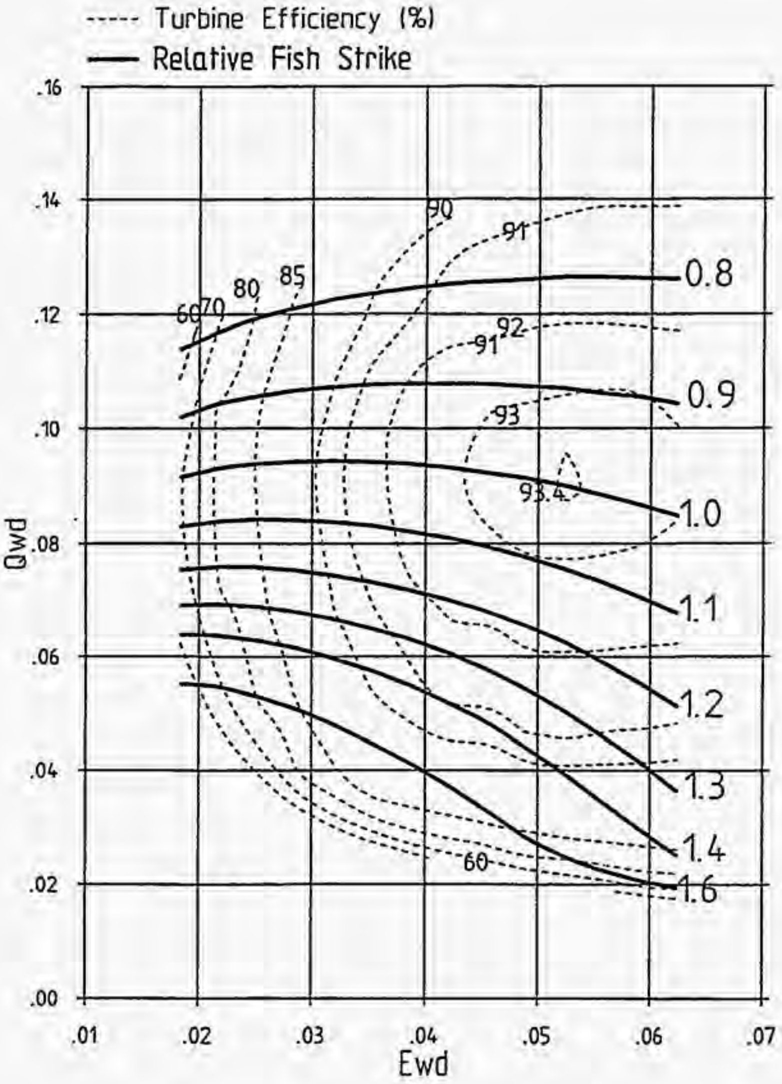


Figure 4.3.2-7 Relative Strike Probability for Kaplan Turbines (at blade tip)

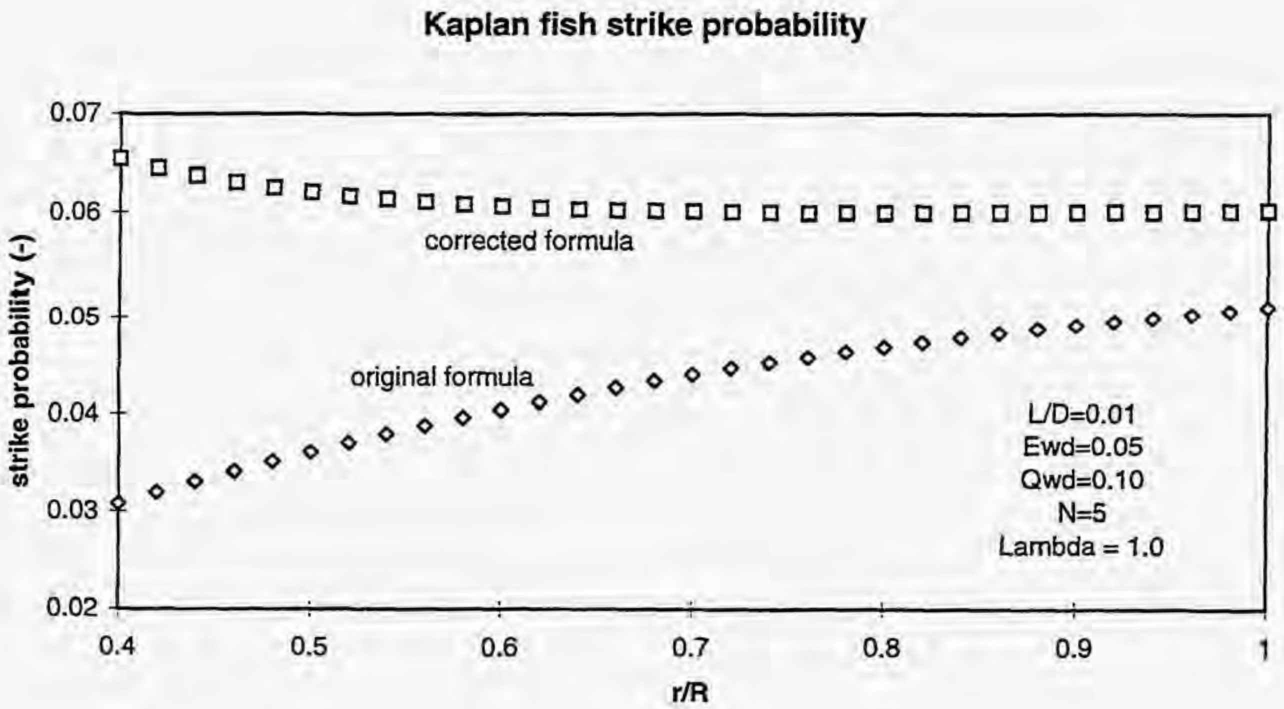


Figure 4.3.2-9 Comparison of Original and New Leading Edge Strike Equation at Higher Flow

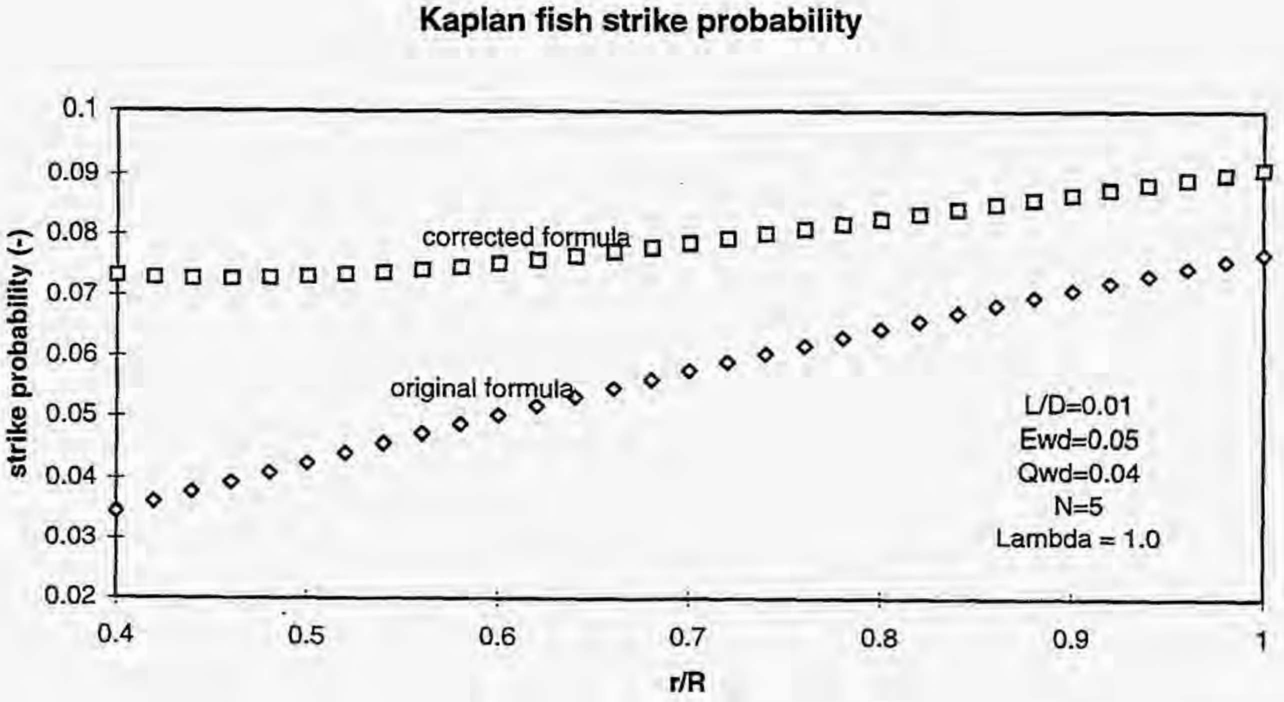


Figure 4.3.2-8 Comparison of Original and New Leading Edge Strike Equation at Low Flow

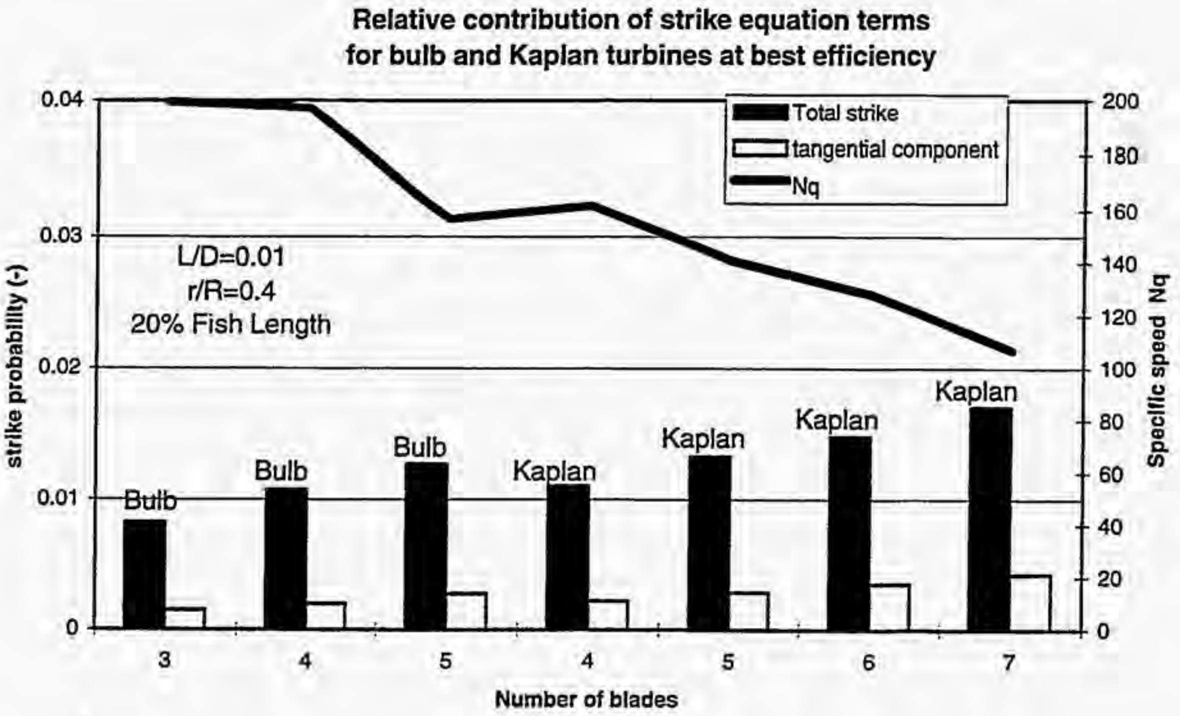


Figure 4.3.2-11 Illustration of Contribution of the Tangential Fish Length to Total Strike Probability

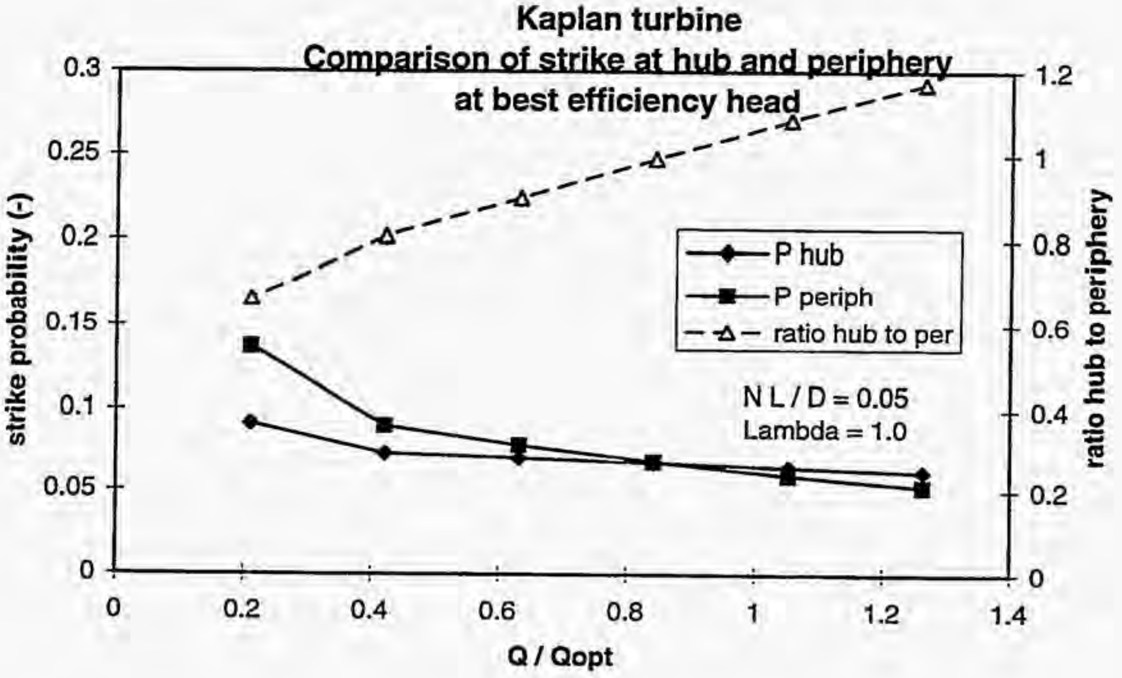
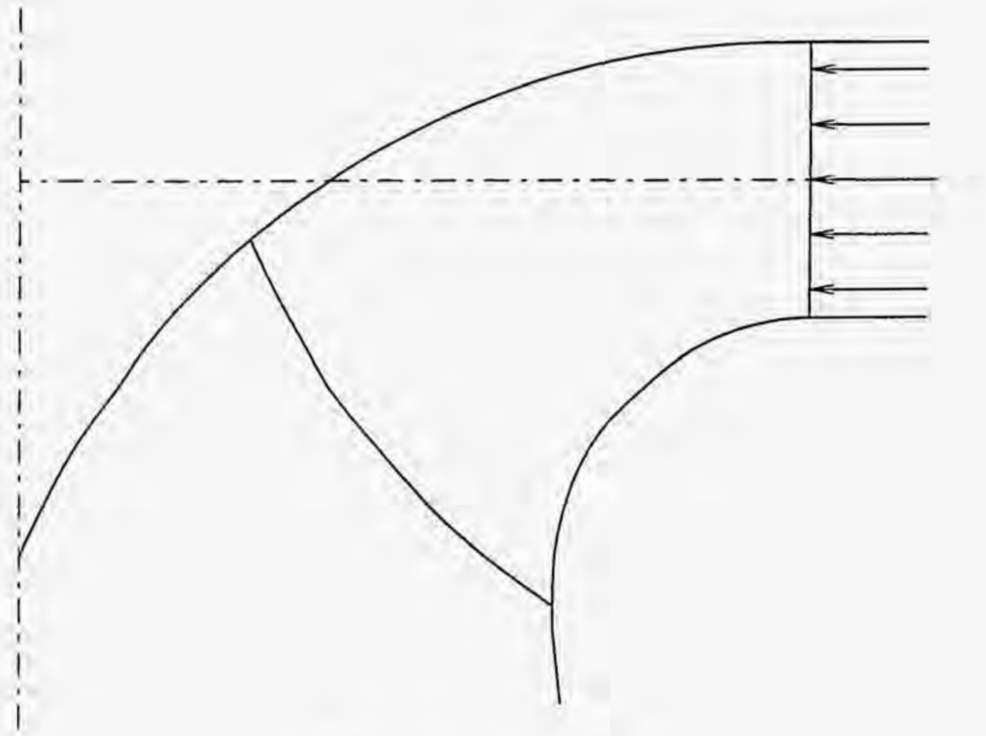
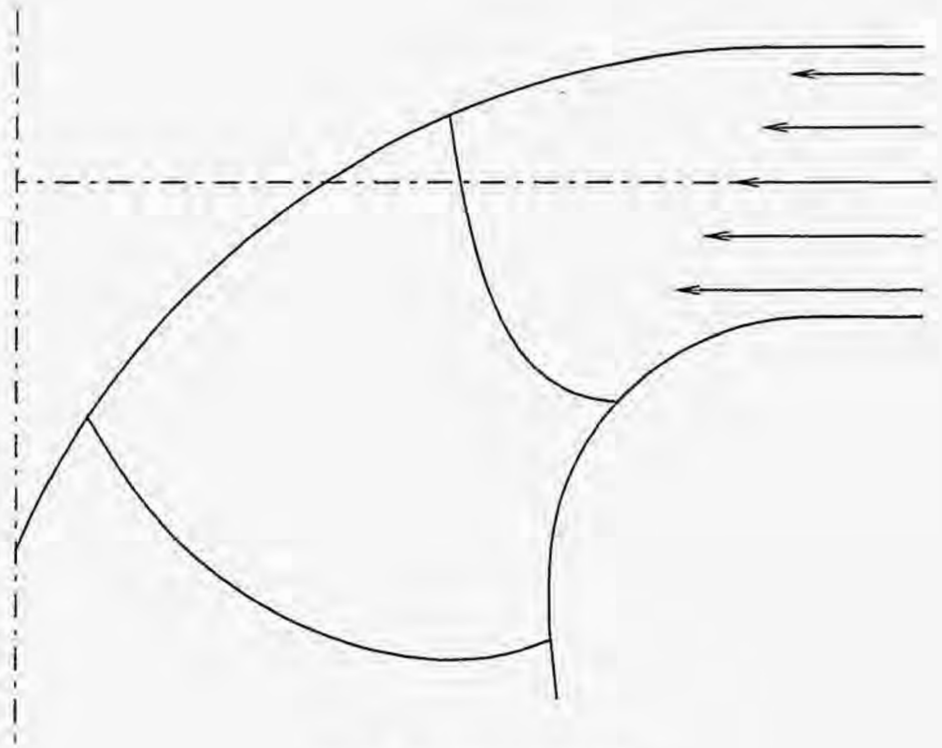


Figure 4.3.2-10 Comparison of Strike Probability at Hub and Periphery vs Flow Rate



LOW SPECIFIC SPEED



HIGH SPECIFIC SPEED

Figure 4.3.2-12 Francis Turbine Runner Inlet Velocity Profile Varies with Specific Speed

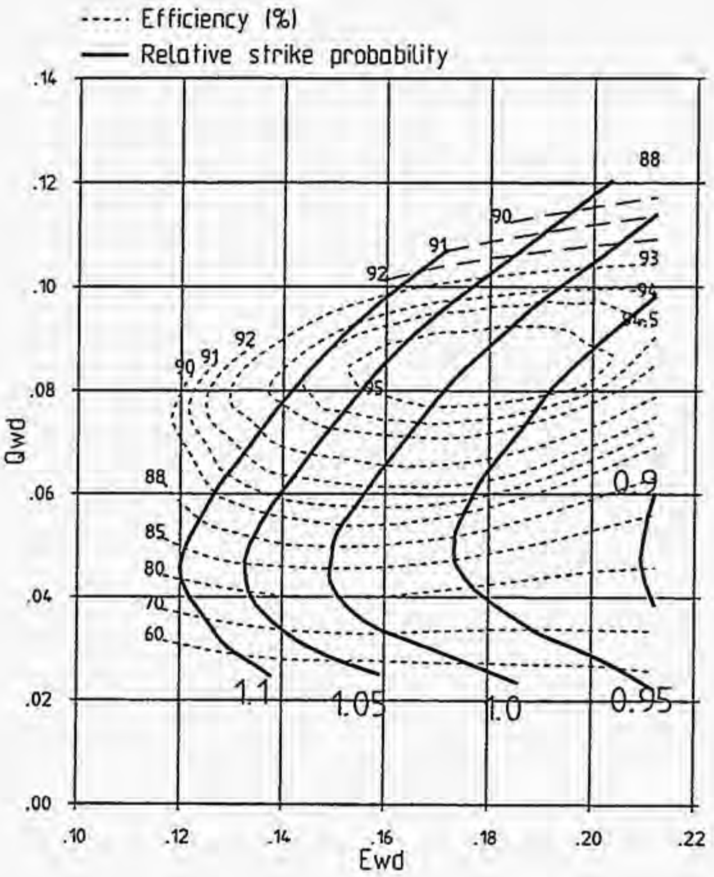


Figure 4.3.2-13 Relative Strike Probability for Francis Turbines

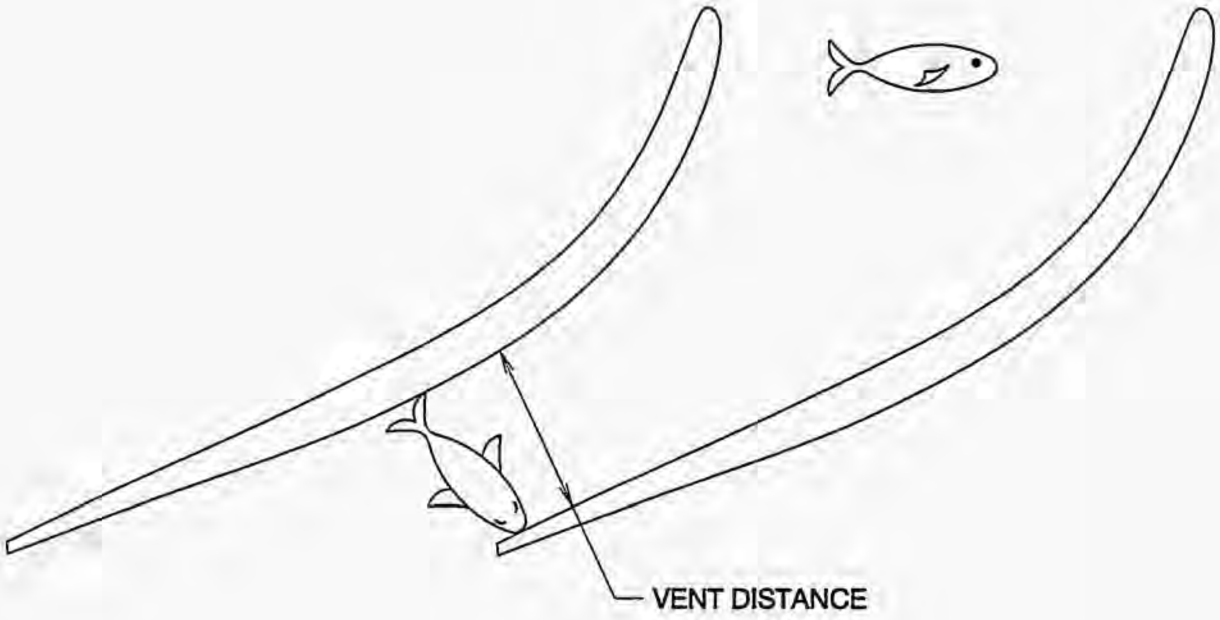


Figure 4.3.2-14 Runner Vent Could Lead to Scrape Injuries

4.3.3 FLUID MECHANISMS LEADING TO FISH INJURY

4.3.3.1 **Overview**

The concept of excess energy dissipation is an attempt to quantify the injurious effects that a non-ideal flow field can cause. Excess energy dissipation can be equated to Theoretical Avoidable Loss (TAL), a concept which has been observed to correlate with measured fish survival. The concept of critical shear is developed and shows some promise of being a credible prediction method. A one-dimensional prediction method based on critical shear is not adequate to predict observed fish survival.

The general assumption regarding fish survival and turbine operating condition is that inefficient turbine operation causes adverse flow field conditions leading to fish mortality. This assumption agrees with the fact that inefficiency is a measure of losses caused by the dissipation of energy (head loss) in the flow. Some of this energy is presumably available to harm fish. It is also known that energy losses are a direct result of flow that is not smoothly aligned with hydraulic surfaces. Two paths have been taken in this study to quantify this phenomenon.

The first method, the overall energy dissipated in the flow, is analyzed without consideration of the precise flow field details. It considers that some energy dissipation is inevitable and causes no harm to fish, while other energy dissipation exceeds a theoretical minimum and this excess energy dissipation is available to cause mortality. The second method, the analysis of shear, is an attempt to directly quantify the details of the flow field. A quantity defined as shear is postulated as a quality of the flow that can be linked to fish mortality. A critical value of shear was estimated from experiments in the literature. Based on a simplified flow field analysis, a general method for estimating shear for any turbine operating condition was developed. The foundation was established for subsequent, more detailed, analyses by CFD methods.

It should be pointed out that both TAL and fluid shear are related to the geometry of the turbine. The number of turbine blades, stay vanes, and wicket gates as well as the presence or absence of blade gaps and the overall quality of the hydraulic design influence the fluid mechanisms which can damage fish. Mortality correlations with the number of turbine blades can relate to the physical strike by a blade structure as well as to the fluid mechanisms associated with the number of blades.

4.3.3.2 **Evaluation of Avoidable Loss**

Summary

Some loss of energy in a turbine can not be avoided. Other sources of energy loss are dependent on the turbine operating condition. The losses that are in excess of a theoretical minimum possible were correlated to measured fish survival. When losses are greater than the minimum amount, fish survival is adversely affected. The point of minimum theoretically avoidable loss usually occurs at a discharge greater than the best efficiency point discharge.

Discussion

Two explicit methods have been applied to predict and evaluate mortality:

- **Theoretically Avoidable Loss (TAL)** An explicit calculation of the portion of the overall energy that has been dissipated in the turbine and that is available to damage fish.

- **Shear** The effect of a fish passing near or through a region where the flow is experiencing a velocity gradient that is sufficiently intense to cause damage. This effect is listed separately from excess energy dissipation because of a different prediction methodology that is used.

The concept of a theoretically avoidable loss has been found to be a useful parameter to correlate a turbine operating condition to internal flow mechanisms capable of causing damage to fish. The flow of water over any surface results in friction that causes a reduction in the useful energy of the flow. This frictional energy loss (actually a transfer of energy into heat) attains a minimum value when the flow is smooth and the velocity is a minimum. For example, the flow of water through a pipe causes losses, but unless the fish comes in near proximity to the pipe wall, in fact so close that it would touch the wall, this energy exchange as a result of the discharge (therefore deemed unavoidable) does not cause damage to the fish. In the turbine, energy losses occur due to several mechanisms. Flow through the spiral case can have losses associated with vortices and secondary flows, but mostly contains losses related to friction on the spiral case walls. Flow over stay vanes, wicket gates, and runner blades may encounter the structure with an angle of attack (depending on the particular operating condition) and the resulting velocity patterns and associated losses are higher than could be achieved under optimum conditions. The flow in a draft tube is typically swirling, with a flow pattern that can cause flow separation from the wetted surfaces. This generates increased turbulence and flow mixing, resulting in energy losses. Flow through various clearance spaces and over thick trailing edges also induce losses. Some of these losses are related to the requirement of moving flow through the machine, and cannot be avoided. Other losses are related to non-optimum flow patterns and could theoretically be eliminated (at a particular operating condition) by redesign of the non-optimum components. Therefore, a method has been developed to separate the theoretically avoidable losses from the losses that cannot be eliminated.

In a turbine the sum of all losses is quantified by the turbine efficiency. Efficiency can be related to losses with the relationship:

efficiency = 1- (losses / Head)

where the losses are taken as the sum of all losses in the turbine. Since some of the losses are proportional to the velocity squared, this offers a means to examine the distribution of losses as a function of operating condition. A procedure was developed to assess the magnitude of frictional losses that are proportional to the velocity squared, and are thus unavoidable, and the magnitude of losses caused by other mechanisms that are considered to be theoretically avoidable. The discharge is used as an indicator of the local velocity. Defining a proportionality between losses and discharge gives:

$$\text{losses} = K \bullet Q^2$$

and to be able to separate avoidable and unavoidable losses, K is further separated

$$K = K_{TAL} + K_{tm}$$

K_{tm}, where tm means theoretical minimum, accounts for the unavoidable frictional losses. K_{TAL} accounts for the theoretically avoidable losses. The value of K is dimensional, having units of (length^5 / time^2). The use of this concept is to compare the value of the losses at various operating points. Therefore only relative values of K at various operating points are most useful. The value of K_{tm} corresponds to the most favorable flow pattern possible for a particular turbine. After a value of K_{tm} is selected, K_{TAL} may be calculated. The value of K_{TAL} is used to calculate the avoidable losses and can be deemed to be Theoretically Avoidable Losses (TAL). They are called "theoretically" avoidable because the turbine design to avoid them would need to have a geometry which can be adjusted to be "theoretically ideal" at every operating condition. TAL then, are an overall measure of the energy available to drive high velocity gradients, swirling flows, vortices and turbulence. These fluid effects, when of high enough energy can cause forces on fish bodies which can result in injury and direct mortality. They can also drive fish into mechanical structures with sufficient energy to create mechanical injury.

In Section 4.4.6, experimental fish survival data from the Wanapum project are analyzed with this approach to provide a quantitative measure of the impact of TAL on the overall measured fish mortality.

4.3.3.3 Evaluation of Fluid Shear

Summary

Shear may be a quantifiable indicator of a flow field condition that causes fish injury. Based on the evaluation of a single study, an injury threshold value of shear (450 / s) has been tentatively selected. A simplified design study has verified the adverse effects of off-design turbine operation on shear, i.e. operation with the turbine blades for a range of values of angle of attack. However, the simplified design study seems inadequate to explain observable mortality effects. Probability equations are developed to predict mortality based on the size of a critical shear zone. The shear concept is applied to confirm the existence of a non-lethal velocity and, for the first time, to explain the minor role that tip speed is observed to have in turbine fish passage survival studies.

Discussion

Definition of Shear

A flow field that causes forces on a fish body that are high enough to do damage are a cause of mortality. Velocity fields within a turbine that change significantly over a distance characteristic of a fish dimension could cause these types of forces. The concept of using this definition of shear to predict fish mortality has been evaluated.

Shear is defined as a change in velocity divided by a change in length. This definition of shear is an indicator of force on a finite three-dimensional body such as a fish. The velocity change causing the shear may be induced by viscous forces as in a boundary layer, in a separated flow region or in the wake downstream of a blade or vane trailing edge. Velocity changes can also be caused in a non-viscous manner and do not necessarily imply energy losses, viscosity, vorticity (rotational flow), or an unfavorable flow field. Note that this definition yields a dimension for shear of (1 / s) as the length dimension in velocity cancels the length in the denominator. An evaluation of shear in Alden, 1997 used a dimension of (ft / s / inch), since these authors chose not to cancel the length units.

The definition of shear as a stress, related to laminar or turbulent viscous forces has been used in the literature. This definition was not chosen here because it excludes certain types of velocity gradients. A flow field having velocity gradients will exert forces on a finite size body, such as a fish. Figure 4.3.3-1 shows velocities and hence velocity gradients, in a runner that are the result of a non-viscous calculation. A definition of shear that was based on viscous forces would incorrectly imply that no fluid induced forces would exist for this flow field.

Shear can be considered as an influence of a mechanical structure that exists some distance away from the structure. Intuitively, this effect is related to the energy contained in the speed between the fluid and the structure. The energy of a moving body is it's kinetic energy, which is proportional to velocity squared. Therefore, as an initial evaluation of the energy in a given flow field that may be available to cause injury to fish, a calculation of the kinetic energy of the flow relative to the blades was made.

As background information, the variation of blade impact energy for different operating conditions is provided. In contrast to initial expectations, this information did not lead to new insights. The fish approaches the blade at a relative velocity. This velocity difference defines the kinetic energy of a possible collision. Figure 4.3.3-2 shows the square of the ratio of relative flow velocity to tip velocity. This calculation was made for the same 5 bladed Kaplan turbine as was used for the sample strike calculation. These results may be used as a general indication of some trends for all turbines. A different view of the strike energy is accomplished by the following shear analysis. The magnitude of the effects caused by strike energy is evaluated and testable predictions are made.

Estimation of the Critical Value of Shear

An important challenge in using the shear concept to evaluate fish survival is to determine if a threshold value exists. Shear above such a value would be described as a critical shear and would be sufficient to cause injury or mortality to a fish, while lower values would cause no injury or mortality. Of course this critical value may be species dependent and also dependent on the size of the region exhibiting this characteristic relative to the fish. If additional biological data become available and a value of shear that can cause disorientation or minor injury is known, then additional types of prediction could be made.

A series of experiments having a nozzle discharging into a quiet pond was used as a benchmark to look for a threshold value (Johnson, 1970a, 1970b, and 1972). A first test with a 150 mm (6 in) nozzle operated at 17.5 m/s (57.5 feet / s). The species used were 120 to 180 mm (5 to 7 in) chinook salmon, and 180 to 200 mm (7 to 8 in) coho salmon. The results were recorded with a high speed movie camera, and it was noted that the fish came out in random positions (head first, tail first or broadside) and location in the jet (left, right, above, or in the center). Some fish broke out of the jet within one foot of the nozzle, while others remained in the jet throughout the picture. Many fish were subjected to violent distortions, both in and breaking out of the jet, with no apparent physical harm. No fish were killed.

A second test with a 100 mm (4 in) nozzle was operated at 20.4 m/s (67 feet / s). The results were as follows:

length of fish inches	coho	chinook	steelhead	Number killed	percent killed
8 to 9		350		19	5.4
8 to 9	150			3	2.0
2.5 to 3.5	150			0	0.0
3 to 4			150	1 **	0.7

** In addition to the one mortality, three fish had popped eyeballs.

Apparently the critical factor governing mortality was their precise location when they were ejected from the jet. To further investigate the shear phenomenon, 100 each of the 8 to 9 inch chinook and coho were anesthetized and were allowed to be entrained into the jet. Approximately 30 percent were seen to enter the jet within one or two feet of the nozzle; none were killed.

A detailed CFD analysis of this experiment was performed as an axisymmetric calculation. Figures 4.3.3-3 and 4.3.3-4 show the grid and velocity vectors. A four inch nozzle protruded into a large tank. The inflow boundary was the small pipe, the discharge boundary was far downstream. The large tank basically provides a large, low velocity area for a gentle recirculation region. Intense mixing occurs at the boundary of the discharge jet. Grids were concentrated in this region.

The shear was evaluated at all locations in the flow, Figure 4.3.3-5. Shear values change rapidly in the near vicinity of the nozzle exit. Considering the 57.5 feet / s experiment, (where fish were ejected from the jet within one foot downstream from the nozzle exit), shear values in the one half to one foot distance ranged from about 400 to 600 (1/s), Figure 4.3.3-6. Considering the 67 feet / s experiment, (where anesthetized fish entered from the jet one or two feet downstream from the nozzle exit, shear values ranged from about 300 to 450 (1/s) in this region Figure 4.3.3-7. Based on these two ranges of shear that caused no mortality, a value of 450 was selected as a tentative critical value. It is also noted that in a different experiment, a lower value of shear (360 (1/s) was found to cause no harm to alewives, a very fragile species that is highly susceptible to injury, (Alden, 1997).

This critical value of shear was evaluated at a point in the flow field. Although the size of a shear zone with respect to fish size, and the direction of the velocity with respect to a fish are important, (Groves, 1972) these effects were not considered in this study. No information is available regarding the orientation of fish during turbine passage. This lack of information regarding fish orientation is a limitation in the application of mortality prediction.

Evaluation of Shear by a One-dimensional Method

The prediction of mortality based on the shear concept, mathematically developed in the next section, is based on the spatial extent of the region having shear values greater than the previously defined threshold value. During CFD evaluations of shear for certain turbine geometries, discussed in Section 5, it was observed that the highest values of shear occurred near the entrance edge of blades, vanes, and gates. Two types of analyses may be useful in the determination of the size of a shear-affected region. A detailed three-dimensional CFD analysis of the exact turbine geometry and operating condition would provide the most accurate information. A second possibility is to approximate a precise analysis with a simplified blade shape and approximate the details of the three-dimensional flow by a two-dimensional calculation with an a priori estimate for the local inflow condition. For the simplified type of analysis, a series of calculations might be performed one time that could then be applied for a wide range of turbine geometries and operating conditions to provide guidance on the shear values that might exist. Based on these approximate calculations, it was hoped that the results could be distilled into a method that could be based only on readily known turbine operating parameters. In order to determine whether an approximate approach could give reasonable results, a series of two-dimensional analyses were performed.

An evaluation of flow over airfoils was performed for a range of conditions. An uncambered NACA airfoil (Abbott and Von Doenhoff 1959) in a two-dimensional linear cascade was used. A variation of angle of attack, blade thickness, and blade spacing was performed. Calculations were done with a 1m long blade in a 10 m/s inlet velocity flow field. An important part of this procedure was to non-dimensionalize the geometry and flow field to be able to relate these results to any turbine operating condition. Distances were non-dimensionalized by dividing by the blade spacing. Velocity and velocity related quantities were non-dimensionalized by dividing by a reference velocity (Vref). In this two-dimensional study the inlet velocity was used as Vref. For subsequent use, the non-dimensionalized values can be applied to particular situation by multiplying the non-dimensional distance by appropriate blade spacing and non-dimensional velocity by a reference velocity.

A typical grid is shown in Figure 4.3.3-8. This grid also shows Cartesian coordinate axes, the blade is in the x-y plane, with the z axis normal to the paper. The label of the z axis is more clearly visible in subsequent figures that show more detail of the blade leading edge. In these figures the z is rather large, and since it is shown rotated, it has the appearance of a large capital N. For small angles of attack, the velocity field is smooth with high shear values occurring near the leading edge stagnation point, in the boundary layer, and in the trailing edge wake, Figures 4.3.3-9 and 4.3.3-10. For higher angles of attack, the velocity field shows flow separating near the leading edge. This leads to high shear values that occur some distance away from the blade, Figures 4.3.3-11 and 4.3.3-12. Plots of the shear were made and a visual judgment was made as to the location having the greatest distance from the blade to the critical shear location.

The distance, d, from the airfoil to the critical value of shear (450 /s) was determined and was non-dimensionalized by the blade spacing, i.e.

$D^* = d / \text{spacing}$

and the shear was non-dimensionalized by the blade spacing and Vref, i.e.

$S^* = \text{shear} / (\text{Vref} / \text{spacing})$

Figure 4.3.3-13 summarizes the results of the parametric study. The non-dimensional distance , D*, is presented as a function of angle of attack, since the angle of attack was found to have the strongest effect. Generally, the thin blades (having a maximum thickness of 2% of the blade length) have lower shear when operated at small angles of attack, while thick blades (4% thick) have lower shear when operated at larger angles of attack. This result confirms general expectations.

In order to use these data to predict mortality in a turbine, several steps are required. The value of the critical shear, i.e., the value that causes mortality, must be known. One must evaluate the angle of attack on the turbine entrance edge, and using the runner inlet velocity and the blade spacing, the distance where the critical shear value occurs can be calculated. This distance could be used to calculate the probability that a fish would pass through the critical shear zone. The development of this calculation is addressed in the next section.

Development of Shear Mortality Equations

Equations describing the probability that a fish will encounter a critical shear zone were developed in a manner similar to the leading edge strike equations. The dimension of a zone in the flow field that contains a critical value of shear can be subtracted from the blade spacing. The magnitude of the mortality due to shear calculated using general estimates seems too low to account for observed fish survival effects. It may be possible to use CFD results, possibly with a virtual fish, to find accurate critical shear distances and correlation functions for use in these equations. The details of the derivation are included in Appendix 10.3.

Application of Shear Concept to Non Damaging Velocity and Explanation of the Minor Role of Tip Speed

Summary

Through a consistent application of the shear principles, a velocity of 5 m/s adjacent to a structure will not generate a shear zone of sufficient strength to cause fish mortality. Velocities of this magnitude can exist in several locations in a typical turbine. Also, the mechanism causing blade tip speed to play such a minor role in fish mortality is explained.

Discussion

A further use of this shear analysis is the evaluation of the existence of a velocity that is low enough to avoid shear damage to fish. A typical calculation was examined to represent a flow over a blade, vane, or any other type of obstruction that is not ideally hydraulically streamlined. A 2% thick vane at 20 degree incidence was chosen because, due to the vane's sharpness at this incidence, a region of flow separation exists. This separation causes high shear values that can be studied. The non-dimensional shear values from this calculation were used to calculate the shear values that would exist for a range of velocities. A 1m long vane was chosen with a 1m vane spacing. The shear was calculated from the non-dimensional shear, S* by multiplying different values of (Vref / spacing). Figure 4.3.3-14 shows these shear values as a function of the distance from the vane. In general, lower values of velocity cause lower shear values, as expected. When the velocity decreases sufficiently, the highest value of shear attained does not reach the presumed critical value of 450 (1/s). This occurs at approximately 5 m/s. A strict interpretation of the 5 m/s data would conclude that the shear does not reach the threshold value, however, recognizing some numerical inaccuracy easily leads to a judgment that the shear is quite close to the threshold and occurs at a distance of about 3.3% of the spacing. An exact value for a non-lethal velocity cannot be determined precisely, but at or slightly below 5 m/s it appears that no mortality is expected based on this shear analysis. The results may be summarized in Table 4.3.3-1.

Vref (m/s)	D* distance from vane / spacing
20	.044
10	.039
5	.033 or 0
2	0

Table 4.3.3 -1 Distance from vane to location of critical shear.

Although the injury mechanism of impact with a solid object seems likely to be different than a shear induced injury, results from two impact experiments are mentioned. First, an impact velocity of 5.2 m /s was found to cause little damage and no mortality (Turnpenny, 1992), based on an experimental study using airfoil shapes mechanically propelled in a test tank. A second test showed that a velocity of 4.9 m/s (16.1 ft / s) was required to cause mortality based on impact of fish by solid objects, (Bell, 1991). It is noted that the results of this experiment were presented without information regarding test protocols or test details. The magnitude of velocity required to cause mortality in a direct impact is quite similar to the velocity required to cause a critical shear zone adjacent to a solid object.

A second aspect of this study illuminates the manner in which strike energy, represented by higher head turbines with higher tip speeds, has not been observed to play an observable role in fish mortality. The distance from the vane to the location of the critical shear value varies with velocity, as is noted in Table 4.3.3-1. At 20 m/s, the distance from the vane to the location of the critical shear has increased by a mere 1% compared to the distance when the velocity is 5 m/s. The shear equation predicts that a 1% change in the non-dimensional shear distance leads to a 1% change in the probability that a fish will enter the critical shear region. Therefore, once the velocity is high enough to cause injury, a significant increase in velocity above this value will give a small increase in the extent of the critical shear zone and hence a small increase in the shear mortality probability.

Outlook for Shear Based Mortality Prediction

The previous sections have defined shear, estimated a threshold value of injuring shear, established general characteristics of probable shear distances, developed a one-dimensional shear mortality prediction equation, and used the shear concept to find a velocity that is low enough to avoid fluid induced damage. While these concepts are a necessary prerequisite to a useful mortality prediction method, several steps remain as a subject for further evaluation. These include

- evaluate threshold shear values and direction with respect to finite sized fish
- evaluate estimated shear zones compared with more accurate CFD results
- determine local angles of attack for any operating condition, both for axial flow and for Francis turbines and find a more accurate method for predicting high specific speed Francis turbine inlet velocities

4.3.3.4 Cavitation

Summary

It is perceived that fish passing through cavitating flow fields can be damaged by fluid effects arising because of cavitation and subsequent vapor bubble collapse. The turbine operating condition that coincides with the onset of cavitation can be determined by CFD analysis. Designs for existing turbines can be developed to eliminate cavitation while increasing power production. Operational guidelines to minimize operation in cavitation regimes will reduce maintenance costs and reduce fish mortality associated with fluid induced loading on fish bodies related to cavitation.

Discussion

The phenomena of a water vapor bubble formation is referred to as cavitation. When the value of static pressure in a fluid reaches vapor pressure, cavitation is presumed to begin. CFD methods can reasonably determine when this condition occurs. However, damaging effects of the phenomena are associated with vapor bubble collapse. A bubble that forms in a region of low pressure is transported to a region of higher pressure and the increase in pressure above the vapor pressure causes the bubble to collapse. Depending on the shape of the vaporized region and the pressure and velocity gradients involved, the collapse may create intense local pressure waves, jets of high velocity fluid and regions of strong flow turbulence. These fluid effects can cause injury to fish and may be associated with mortality.

Numerical modeling of the growth and collapse of cavitation bubbles and the quantification of the velocities and forces created by the collapse are beyond the scope of current CFD methods. However, benefits are still obtained from the use of CFD. Utilizing today's technology, replacement runner designs can be developed to improve cavitating characteristics of existing turbines and in many cases eliminate cavitation while improving power output. Figure 4.3.3-15 shows a comparison of the cavitation performance improvement (expressed as a coefficient signifying the beginning of detrimental cavitation) that was obtained for the Bay D'Espoir Francis turbines. For this project, a turbine design which was normally operated in the cavitation region was replaced with a higher efficiency runner design which eliminated the cavitation while enabling operation at higher outputs than previously achievable.

The work conducted for this contract cannot directly add to the current state of knowledge regarding cavitation induced fish mortality. However, carefully designed fish survival experiments can be used to place fish into turbine cavitating flow fields. From these experiments, correlation between observed mortality and the state of the cavitating flow field estimated from scale model testing of turbine components may be able to be made. Section 4.4.6 discusses fish passage survival testing at Wanapum Dam. For one of the operating conditions fish were injected into the turbine intake leading some of the fish to a region where cavitation was observed during model testing. An estimate of the effect of cavitation was derived from these turbine passage tests. More data are required to be able to develop a quantified prediction method.

4.3.3.5 Draft Tube Backroll

Summary

The flow leaving the draft tube and entering the tailrace can cause a complex flow field that may trap and possibly disorient fish. CFD methods may be applied, but would challenge current hardware and software.

Discussion

The flow leaving the draft tube and entering the tailrace is quite complex. A backroll is defined as a region of the flow that recirculates. This occurs due to the interaction of the velocity field inside the draft tube and the tailrace. The draft tube exit flow is commonly believed to behave like a fluid jet entering a quiescent pool. Such a jet has separated or recirculating flow on all sides. The draft tube exit jet typically has a solid boundary on the bottom and an atmospheric boundary at the tailrace pool surface. Adjacent turbines may or may not cause an interaction. Both civil structure and internal flow fields may interact to have significant effects. Figure 4.3.3-16 shows alternate arrangements that were used for the draft tube designs at the old and new sites at Bonneville Dam. The draft tube with greater upsweep, had a more visible disturbance to the tailrace surface, compared to the more horizontally oriented discharge. The internal draft tube flow field is believed to have an appreciable influence. Figure 4.3.3-17 shows a tailrace surface that is more disturbed subsequent to a turbine runner replacement. Despite good turbine performance, it is postulated that the draft tube discharge has become more nonuniform.

Overall, the backroll phenomena seems to be relatively unexplored. CFD methods may be applied, but the analysis of a single operating point would require calculation of both the draft tube (with a suitable link to a runner calculation) and the tailrace. The tailrace would require that the location of the surface be determined as a result of the calculation. Calculations of this complexity may be possible or are becoming possible as computing hardware and software advances, but are not normally performed by turbine manufacturers.

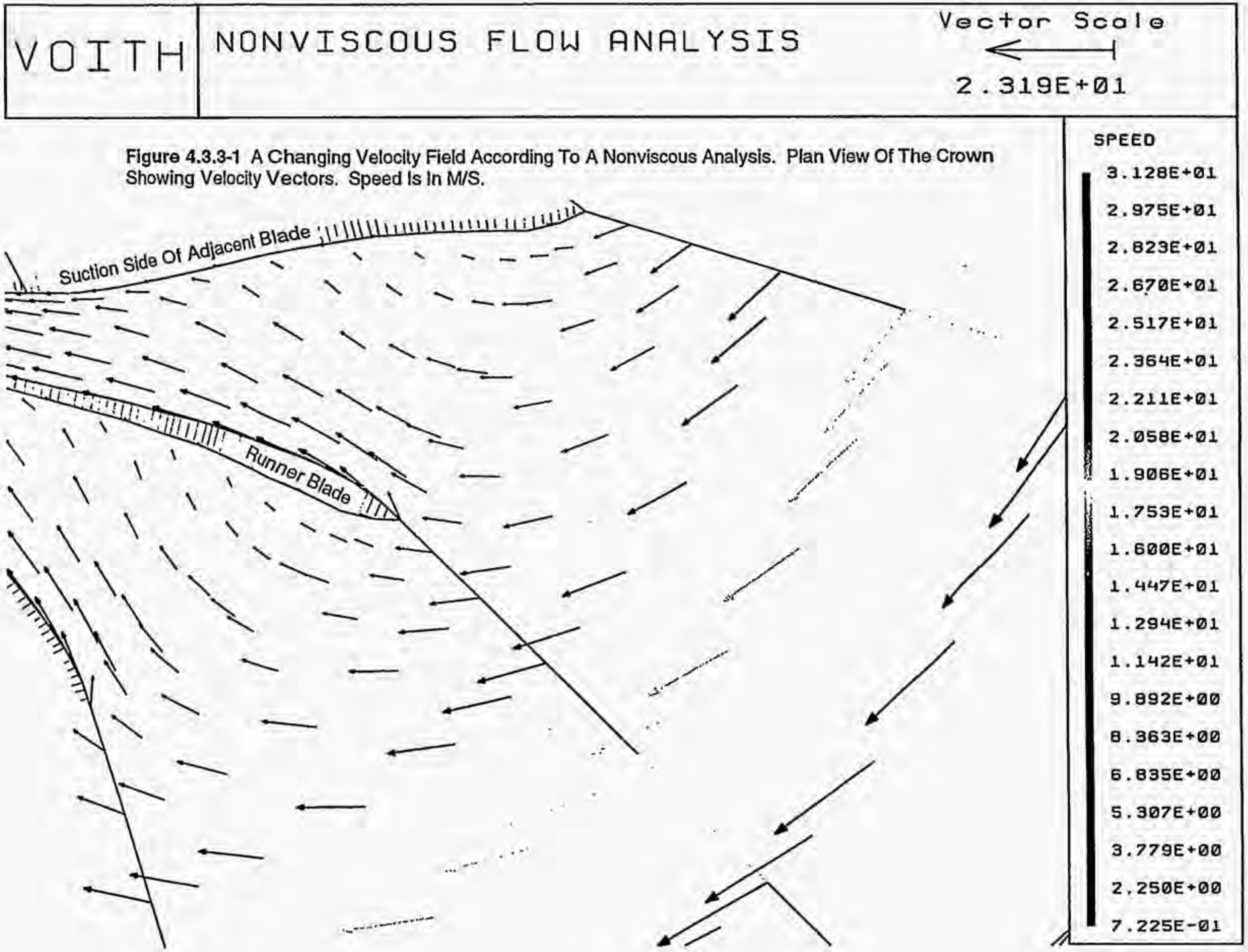


Figure 4.3.3-1 A Changing Velocity Field According To A Nonviscous Analysis. Plan View Of The Crown Showing Velocity Vectors. Speed Is In M/S.

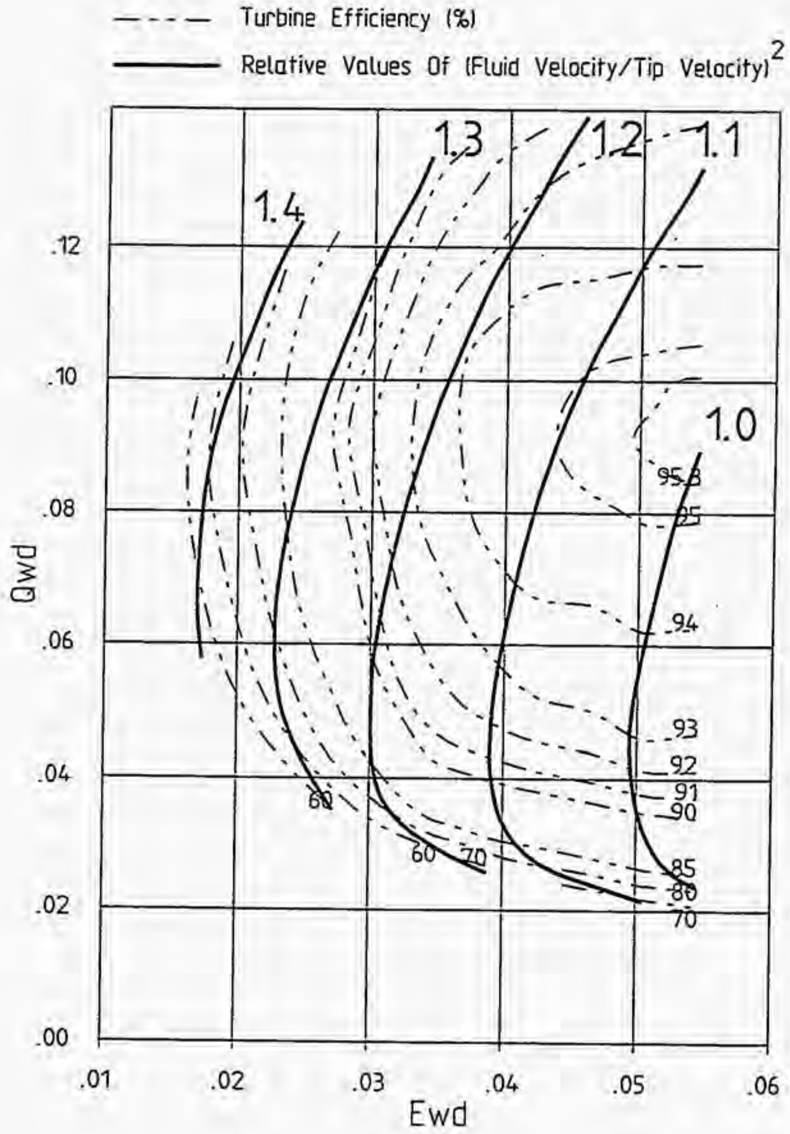


Figure 4.3.3-2 Square of the Ratio of Relative Flow Velocity to Tip Velocity for 5-Bladed Kaplan.

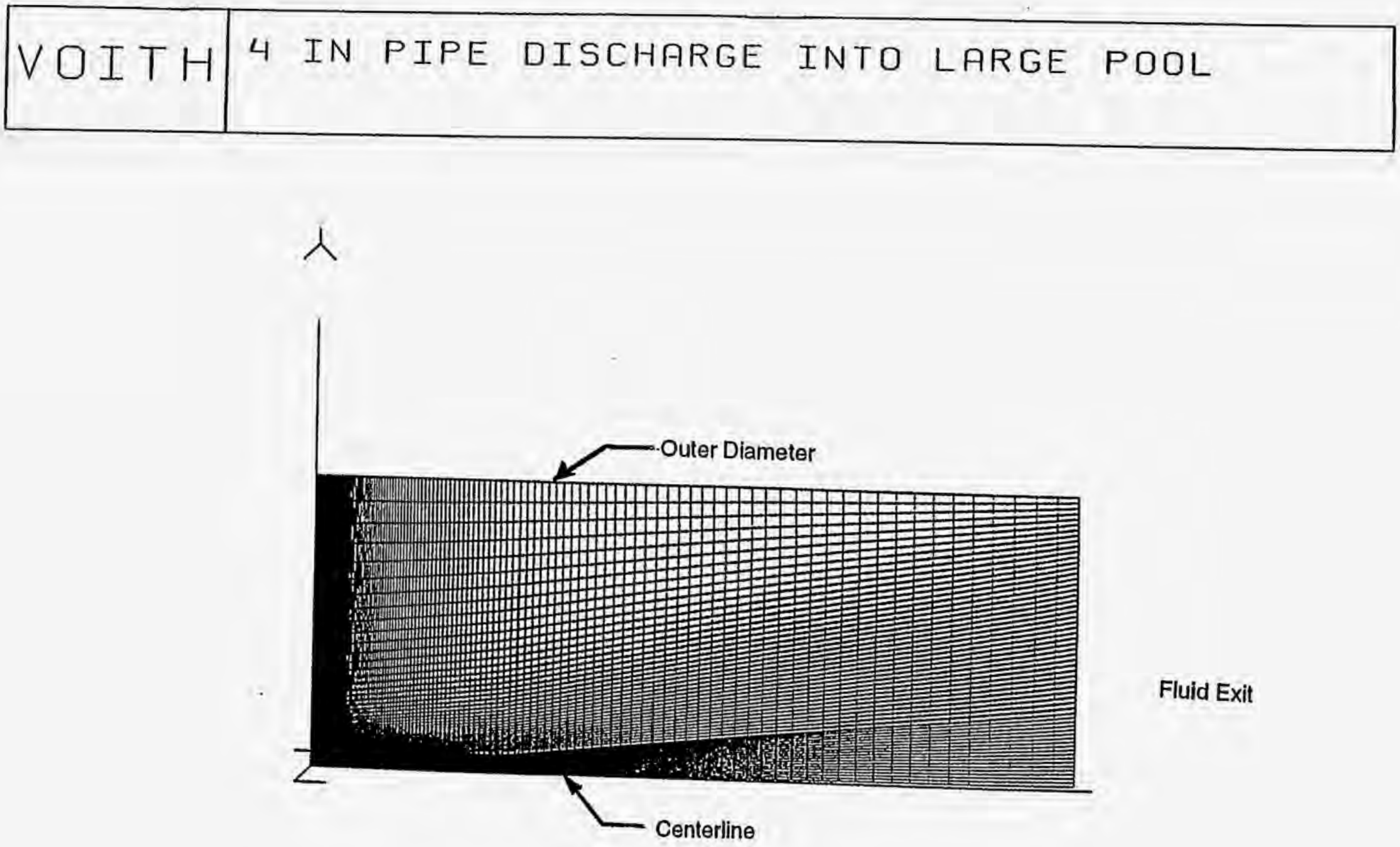


Figure 4.3.3-3 The Grid Used to Calculate to Pipe Discharge into a Large Pool

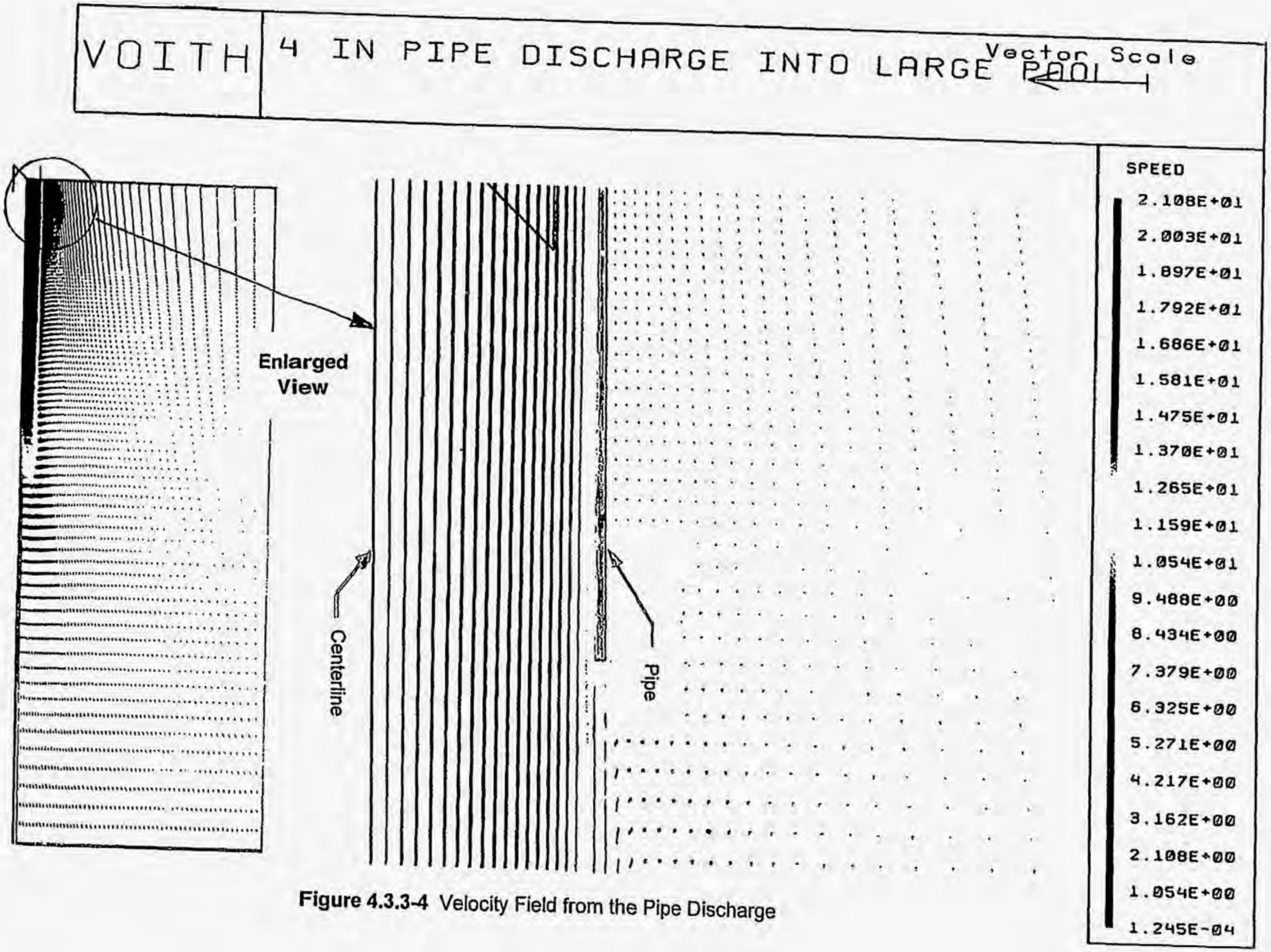


Figure 4.3.3-4 Velocity Field from the Pipe Discharge

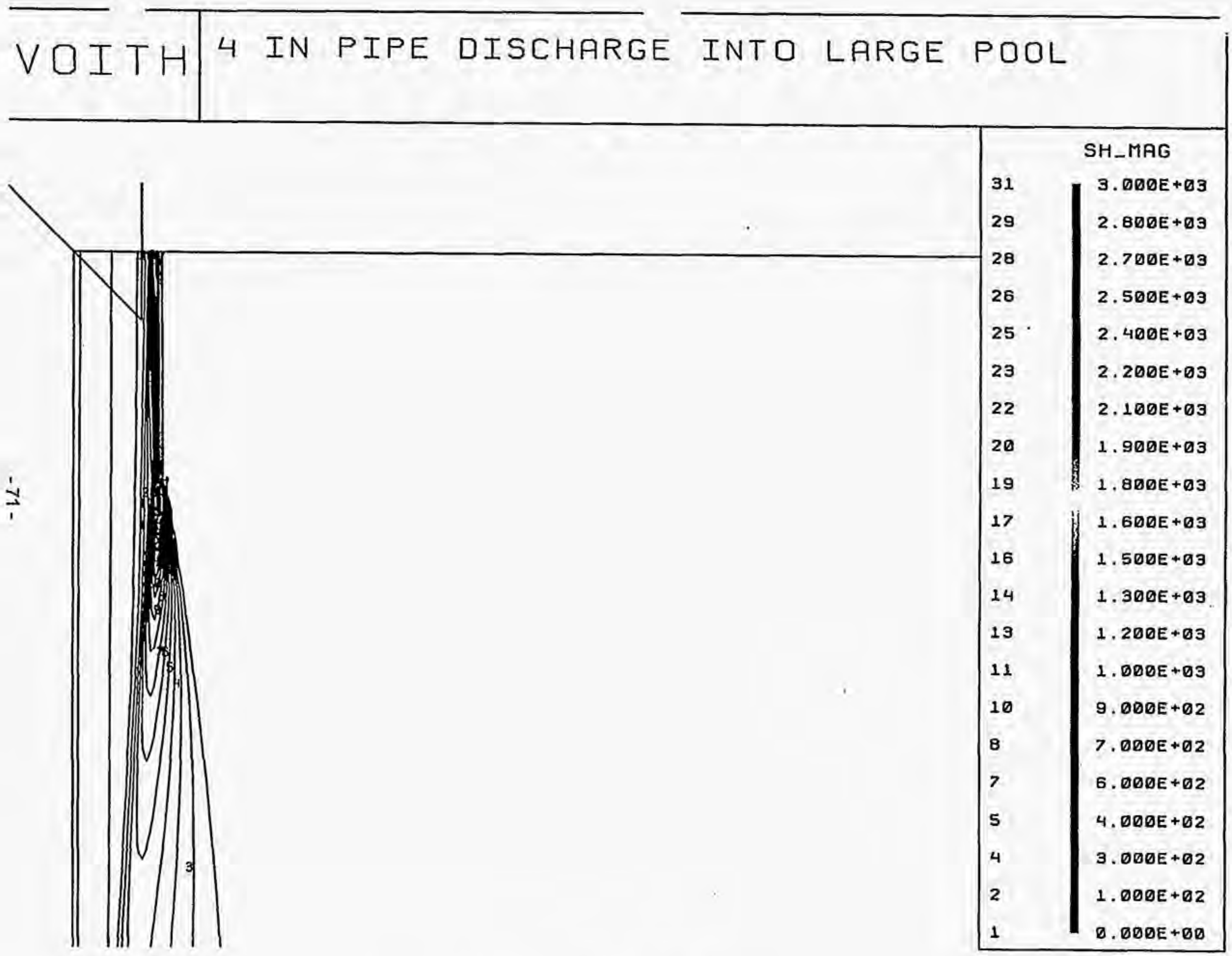


Figure 4.3.3-5 Contours of Shear for Pipe Discharge

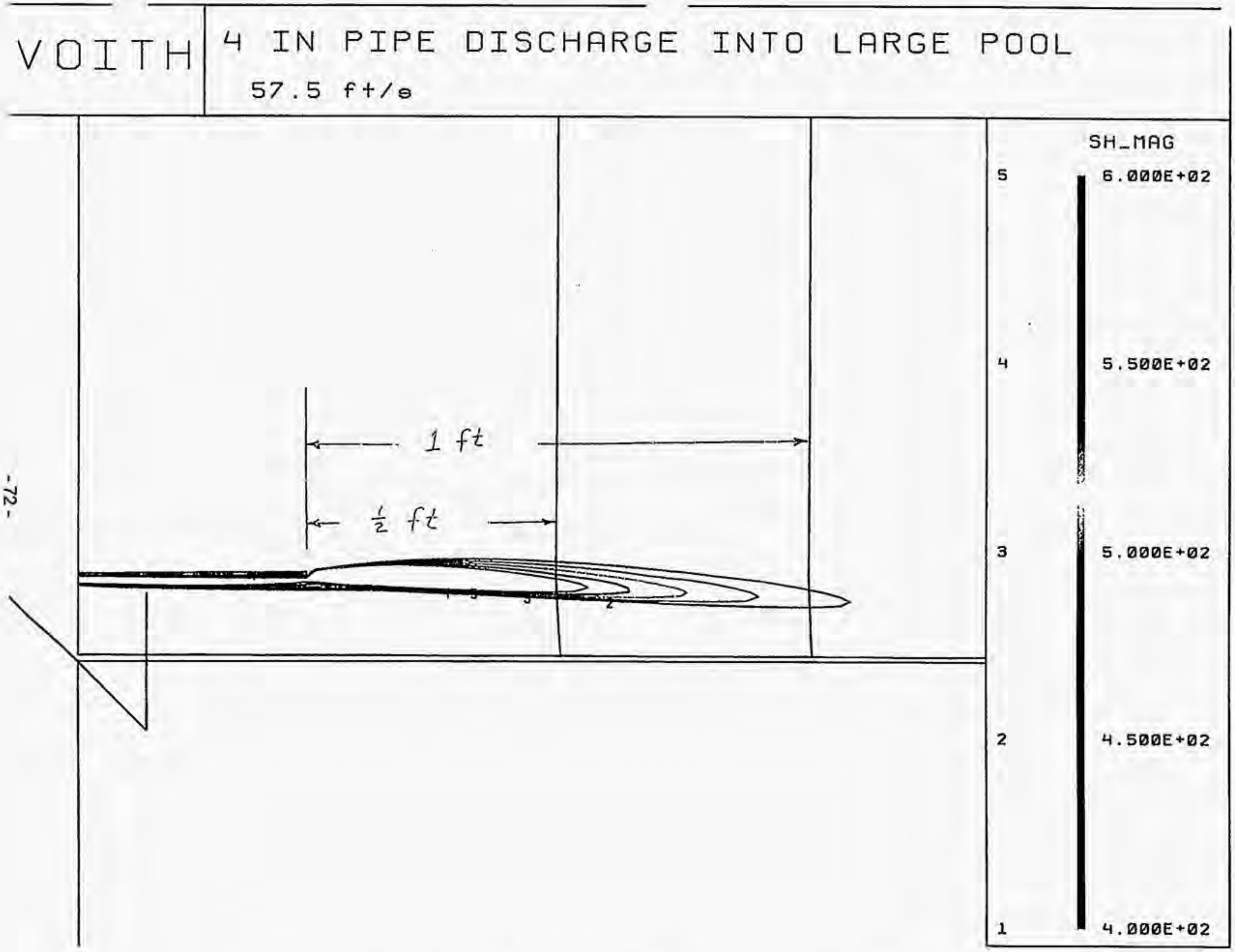


Figure 4.3.3-6 Shear Contours at One-Half to One Foot from Nozzle, 57.5 ft/s

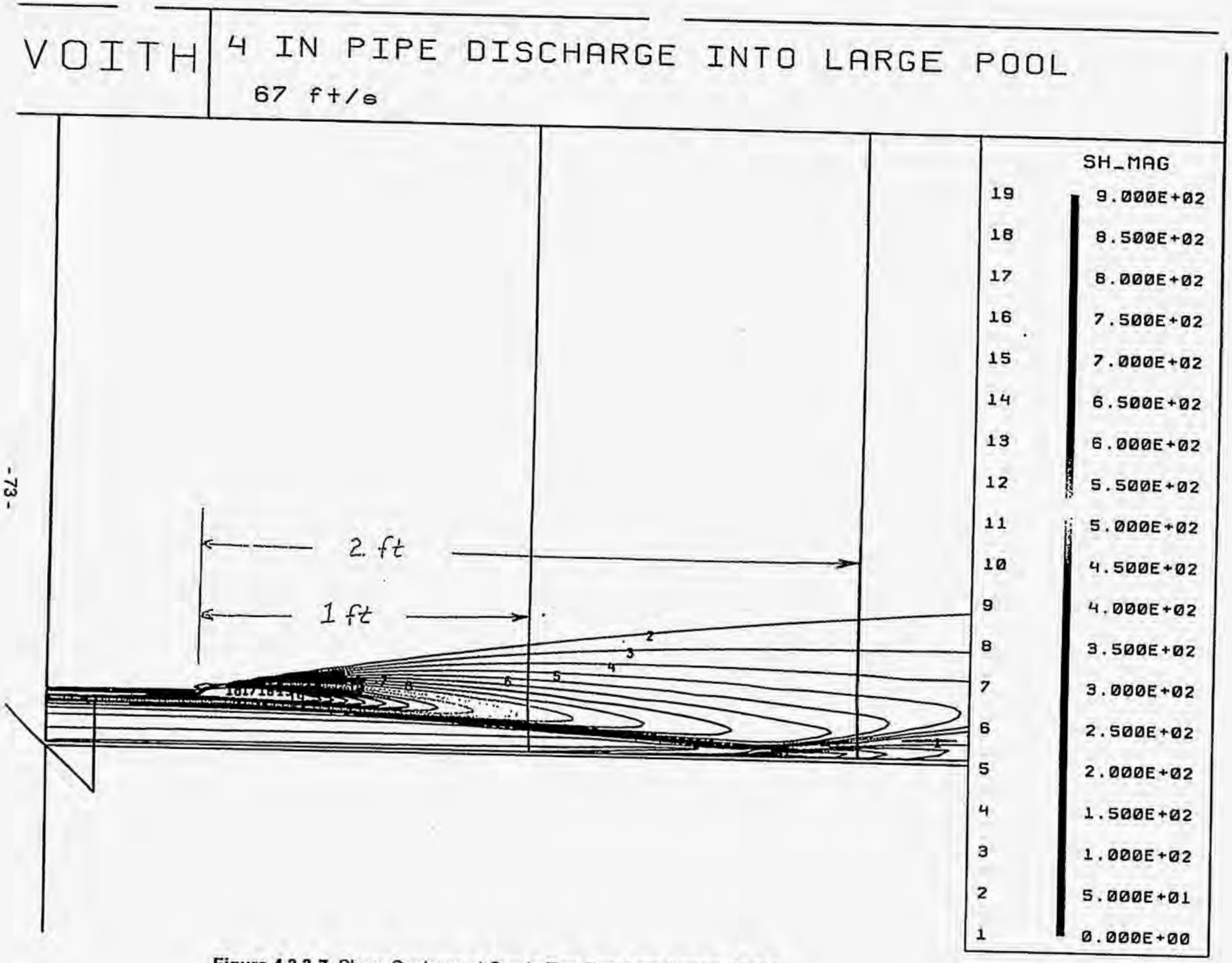
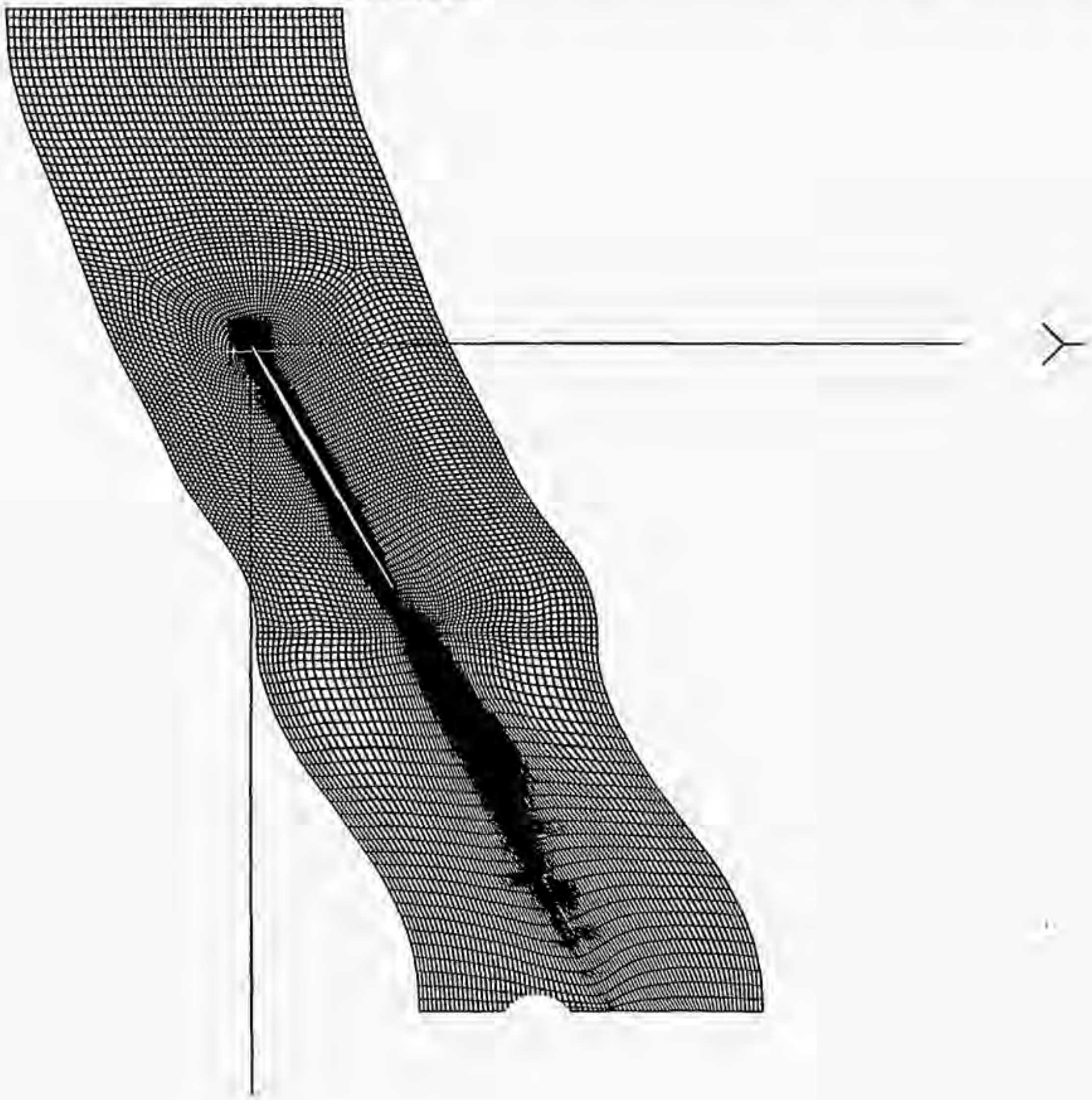


Figure 4.3.3-7 Shear Contours at One to Two Feet from Nozzle, 67 ft/s

Figure 4.3.3-8 Grid For Two-Dimensional Airfoil Study. Blade Is 2% Thick, Spacing To Adjacent Blade Is One Blade Length.



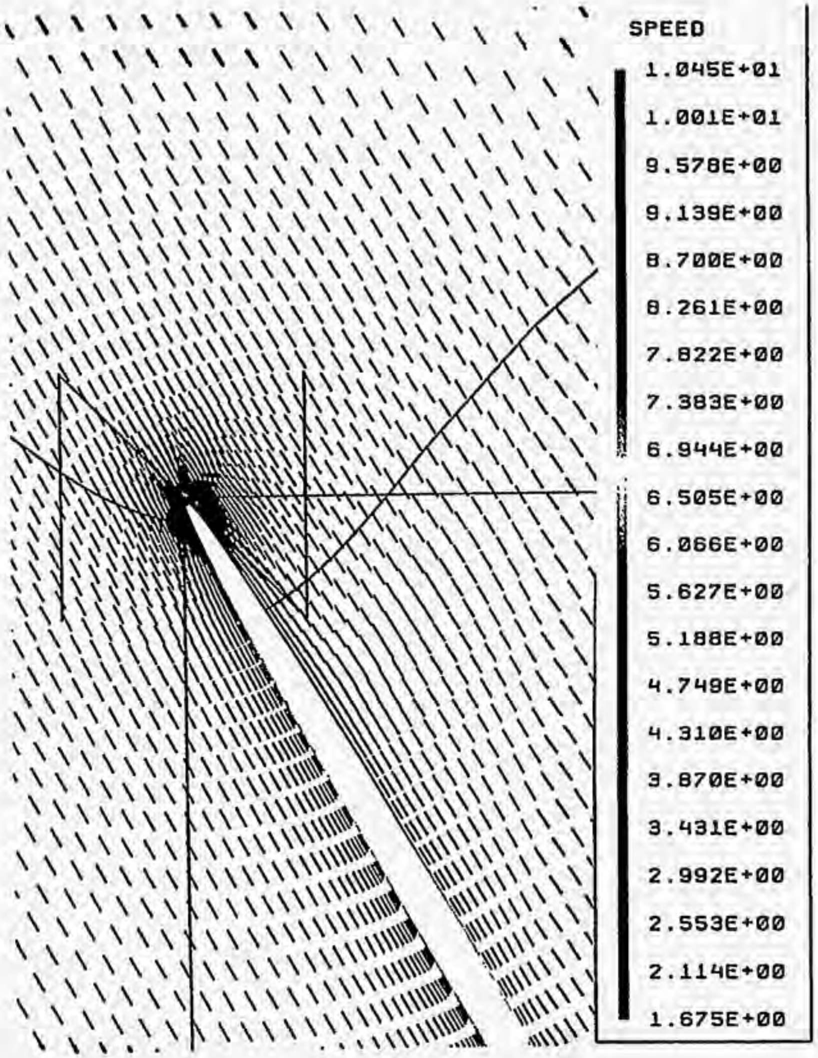


Figure 4.3.3-9 Velocity Field Near A Leading Edge At Zero Incidence. Speed Is In m/s.

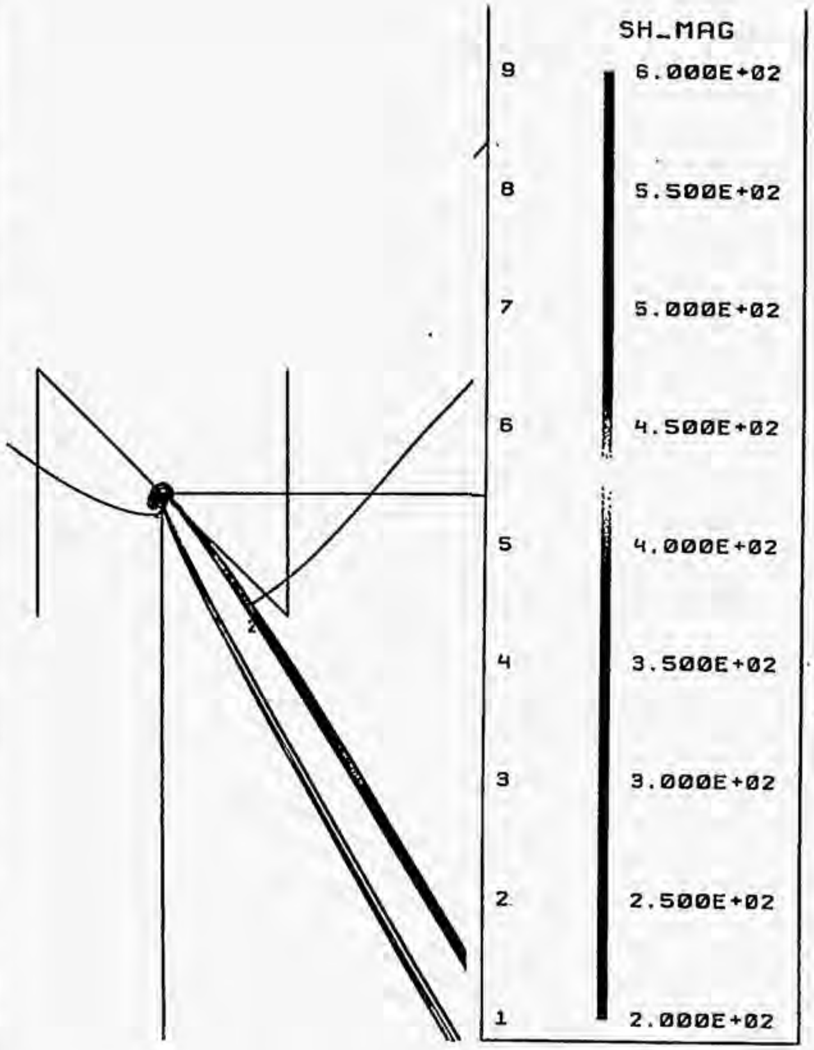


Figure 4.3.3-10 Shear Values Near A Leading Edge At Zero Incidence. The Spacing Of The Shear Contours Indicates A Small Critical Shear Zone.

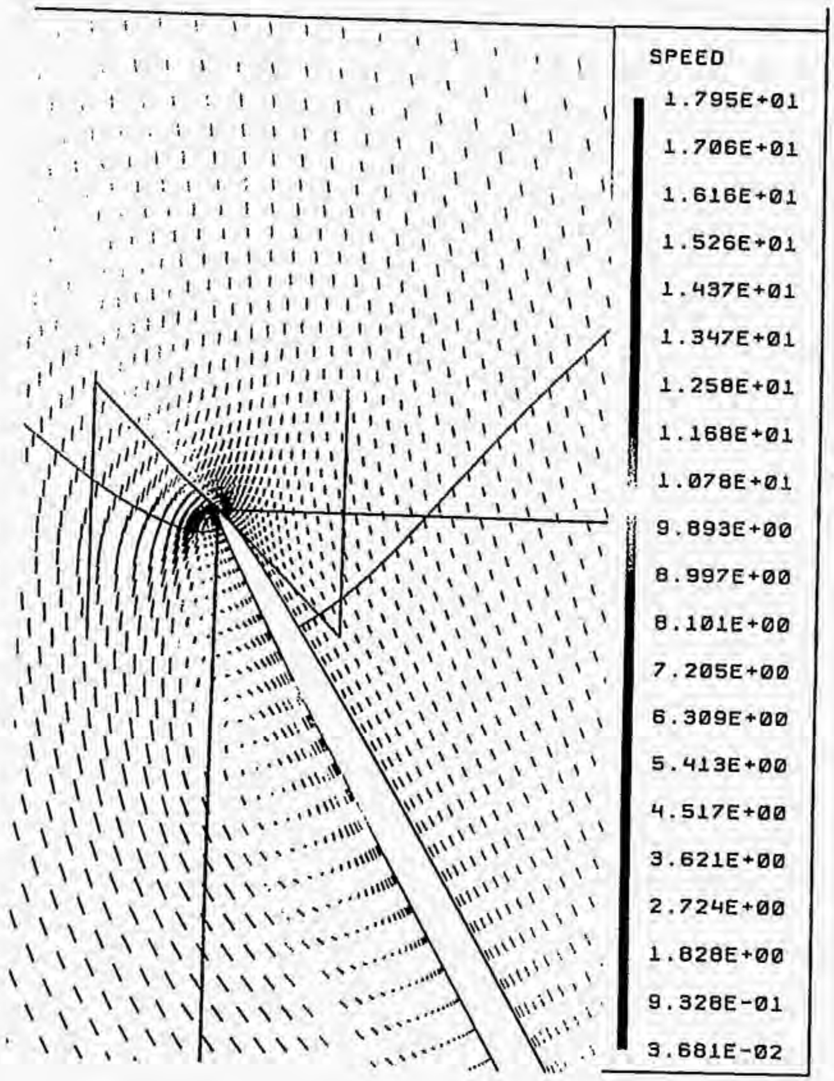


Figure 4.3.3-11 Velocity Field Near A Leading Edge At 20 Degree Incidence. Speed Is In m/s.

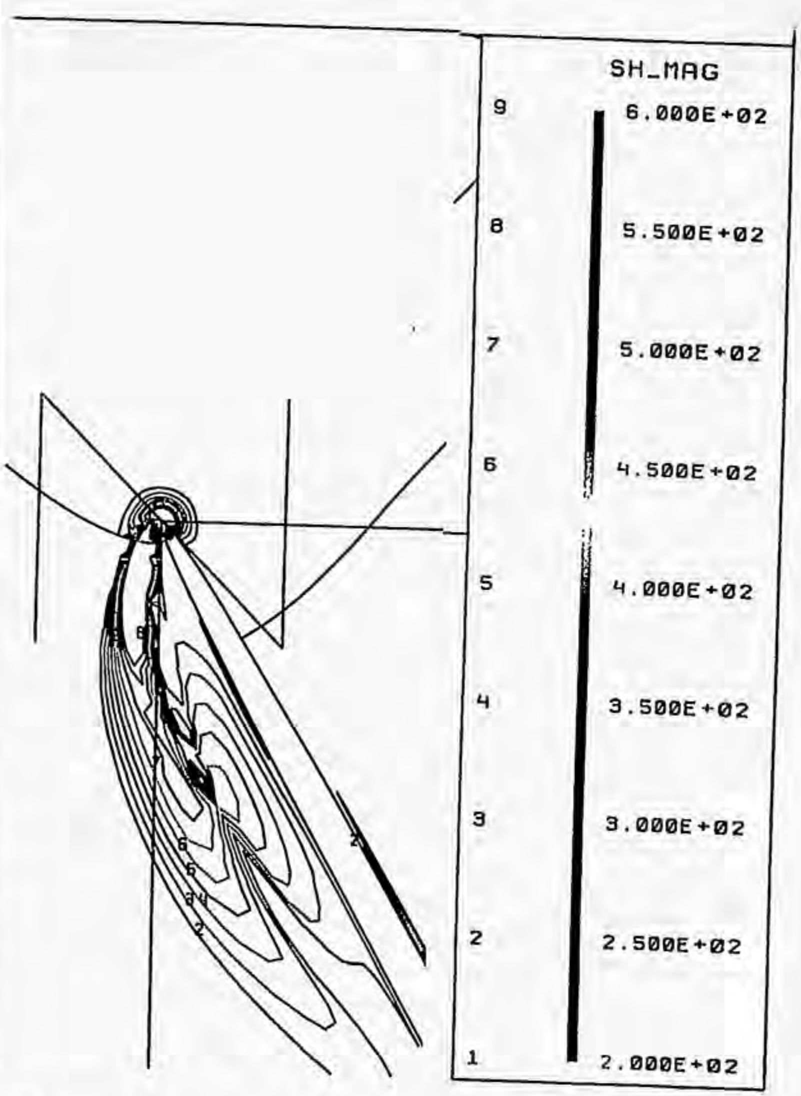


Figure 4.3.3-12 Shear Values Near A Leading Edge At 20 Degree Incidence. The Spacing Of The Shear Contours Indicates A Large Critical Shear Zone.

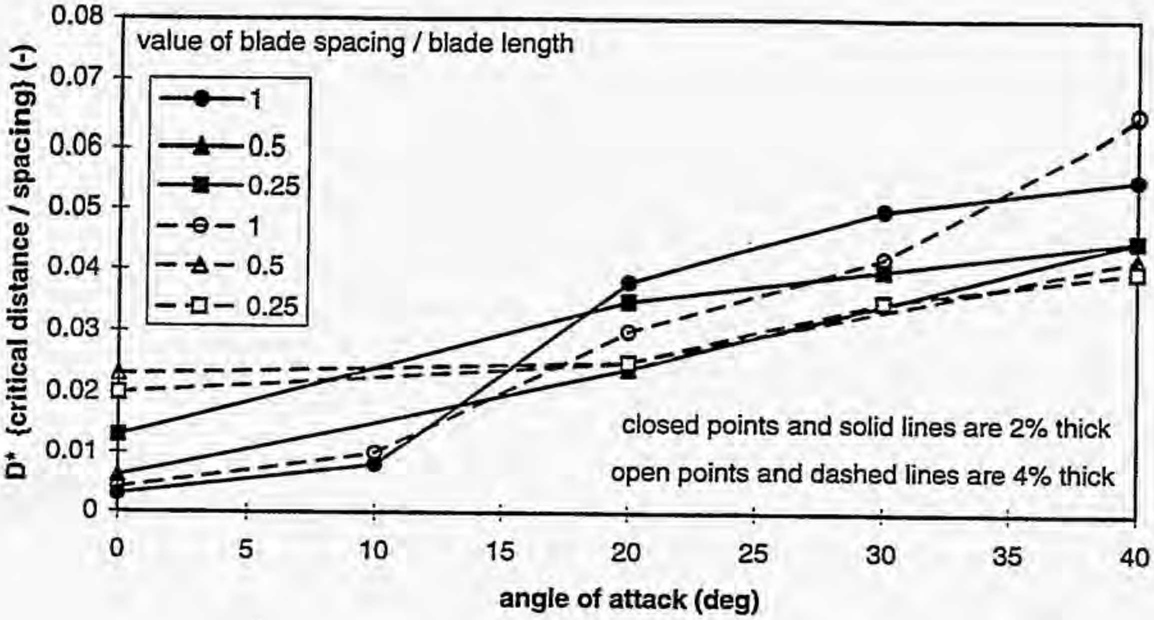


Figure 4.3.3-13 Summary of Critical Shear Distance

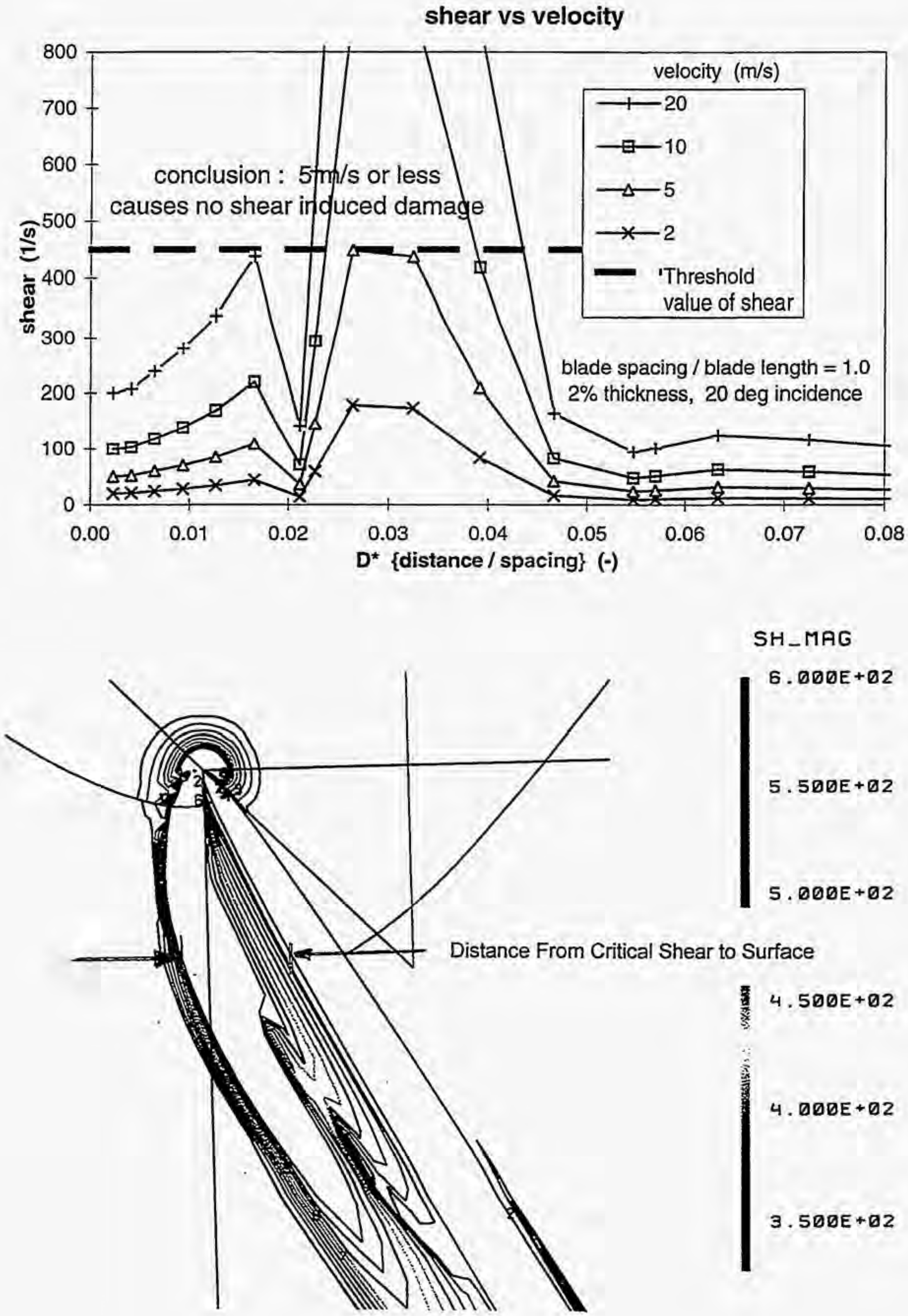


Figure 4.3.3-14 Illustration of Shear Magnitude and Critical Distance for Different Velocities

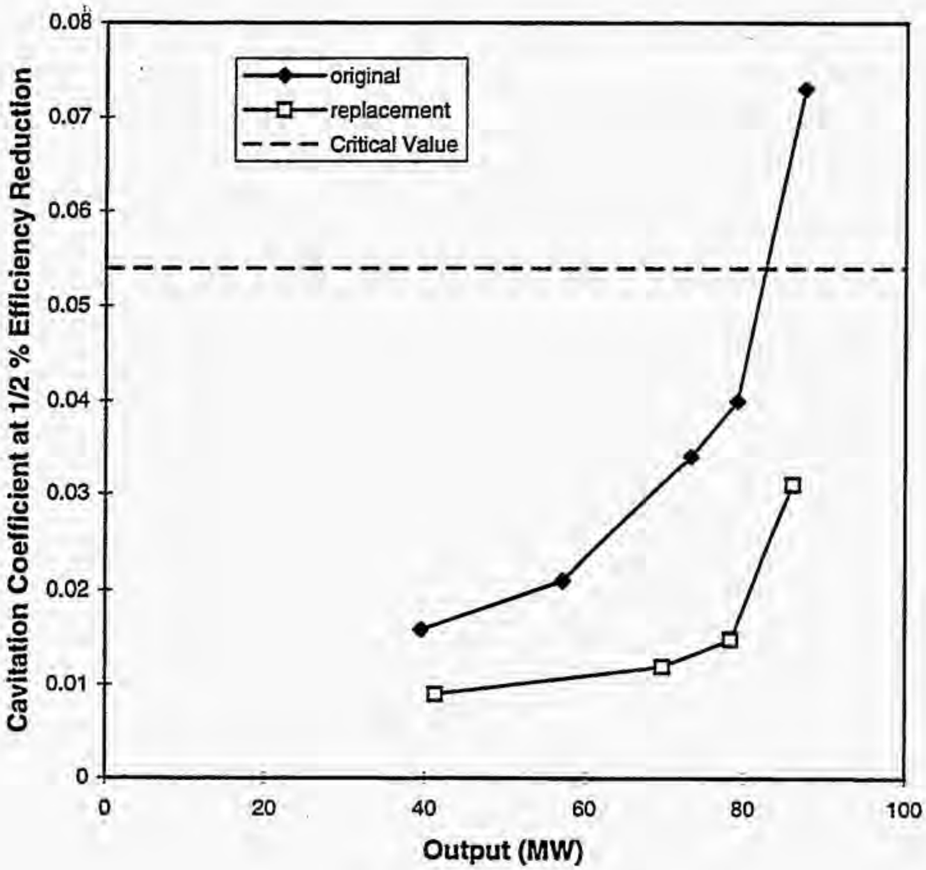


Figure 4.3.3-15 Rehabilitation Can Improve Cavitation Performance

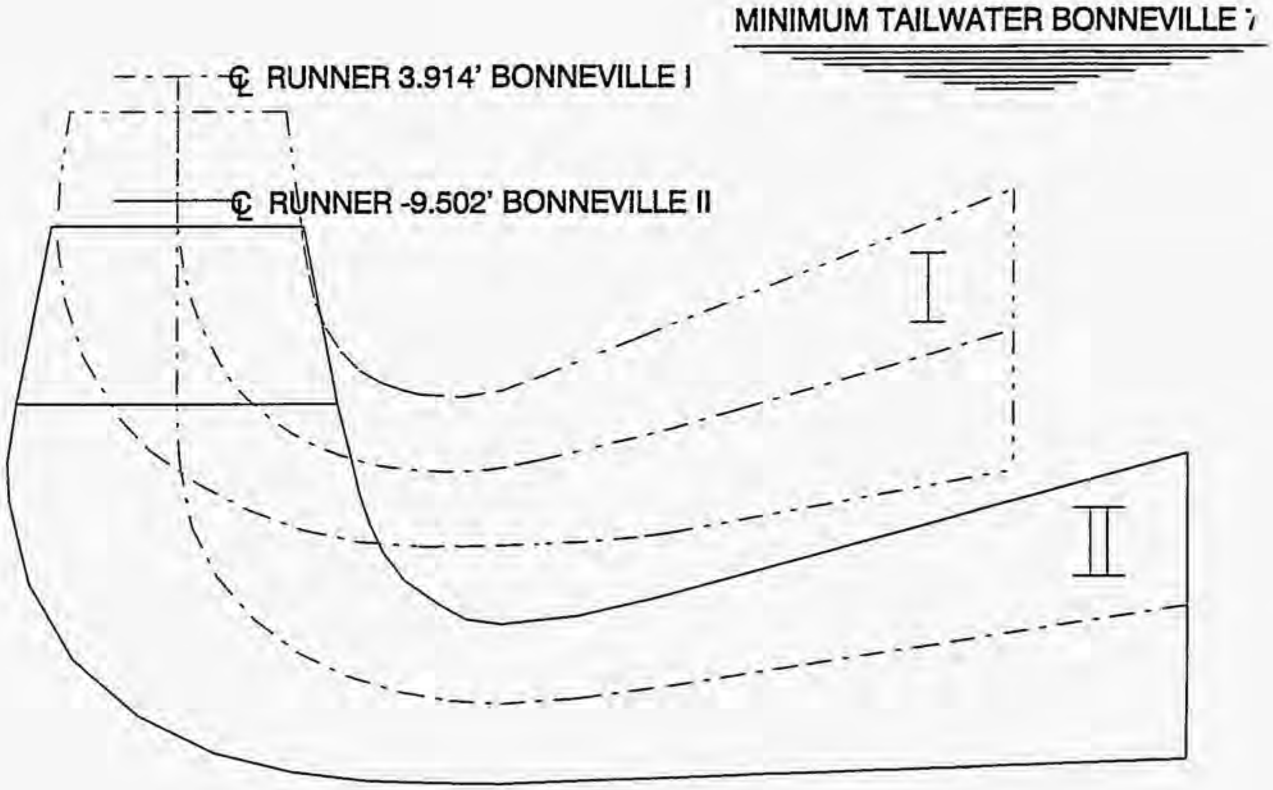


Figure 4.3.3-16 Bonneville Draft Tubes with Different Discharge Orientation

Figure 4.3.3-17 A Disturbed Tailrace Surface

4.3.4 PRESSURE MECHANISMS LEADING TO FISH INJURY

Summary

Inside of turbines, the pressure distribution and the rate of change of pressure with time can be determined accurately by CFD analysis. Transit times from the high pressure region at the turbine spiral case inlet to the exit of the draft tube are relatively short, even for big turbines. Transit times through low pressure regions in the runner blade region are quite short. The rapid transit through this low pressure region in the blades is felt to cause no significant mortality. More important is the change in pressure from that to which the fish has become acclimated to a lower value at the draft tube exit. Controlled experiments on fish at low head projects (<30 m or 100 ft) indicate that pressure related injuries/mortality is low. However, pressure related mortality can be significant at higher head dams, whether equipped with hydroelectric turbines or not, if sufficient time is given to fish to acclimate to greater depths (high pressure) and subsequent rapid exposure to low pressures. Such environments can be associated with fish passage as fish move from the bottom of an upstream deep reservoir or long penstocks to a downstream reservoir. To minimize the adverse pressure effects, the turbine plant design may be more critical to fish passage survival than turbine machine design. For new advanced turbines, plant designs should be developed to minimize long times associated with penstock passage, and therefore the acclimation to deeper depths associated with them

Discussion

The absolute levels of pressure and their rate of change over time (between intake and the draft tube exit) are the result of complex interactions of factors associated with the environment and turbine design. For a particular turbine design, the head and discharge cause a unique flow pattern that can be calculated by advanced CFD methods. Subtle changes to the turbine runner geometry or the turbine operating condition can cause significant changes to the resulting pressure. However, in the absence of cavitation, pressure changes that fish experience for exceedingly short time cause no significant mortality (RMC 1994a,b,c,d; RMC and Skalski 1994a,b; RMC et al. 1994; Normandeau Associates et al. 1995, 1996a). Time to acclimation is probably too short to cause significant pressure related fish damage. However, as mentioned earlier acclimation history and the time to reach full acclimation are not precisely known.

On the other hand, time to acclimation may be sufficient to cause significant damage in plants where the civil design draws fish from lower regions of the reservoir, or allows fish to transit the penstock in a manner wherein the fish can become acclimated to the higher pressures by the time they reach the turbine spiral intake. Figure 4.3.4-1 shows schematics of two alternative turbine plant configurations, and the pressure history for fish passage. For all sites, the transit times for fish passing the turbine is relatively small in comparison to the transit times for fish passing from the dam to the turbine. For typical Kaplan projects the heads range from 6 to 26 m (20 to 120 ft) and the pressure changes associated with the difference in pressure from the headwater to the tailwater are not sufficient to create serious decompression trauma. For Francis turbines, however, heads are higher (15 to over 350 m or 40 to over 1,200 ft), the time of passage is longer (particularly at sites with long penstocks or tunnels leading to the turbine) and the possibility for decompression trauma is real.

Testing has been done to help understand the effects. The information derived from laboratory tests is quite variable and is based mostly on subjecting fish to pressure regimes that may not mimic those encountered by fish in passage at dams with greater than 14 m (45 ft) of head. Therefore, the development of criteria for maintenance of safe pressure differential at the turbine runners may be risky if based on currently available laboratory data. This is especially true if the fish tolerance to pressure differential (expressed as a ratio or proportion of fish exposure pressure to acclimation pressure) observed in a laboratory is applied to designing an advanced turbine. A numerical example may help clarify the importance of considering the site specific depth characteristics. A pressure reduction of 60%, that is a ratio of final pressure to acclimation pressure of 0.4, can be observed in two different situations with significantly different consequences. For example, fish acclimated at 30 m or 100 ft depth (about 4 atm) and suddenly exposed to 6 m (20 ft) deep tailwater (about 1.6 atm) and fish acclimated to 18 m (60 ft) depth (about 2.9 atm) and rapidly exposed to 2 m (5 ft) deep tailwater (about 1.16 atm), each experience a 60% reduction in pressure. The latter pressure condition is frequently encountered by surface oriented fish intercepted by extended length screens at large hydroelectric dams (e.g., Pacific Northwest) and collected in gatewells or surface bypass structures. Recent studies at hydro dams with less than 15 m (50 ft) of head have not shown pressure related injuries/mortality (see Section 4.2.3). Section 4.2.4.2 provides other specific examples of the effects of intake configuration, location, and depth on fish injuries/mortalities caused by changes in pressure.

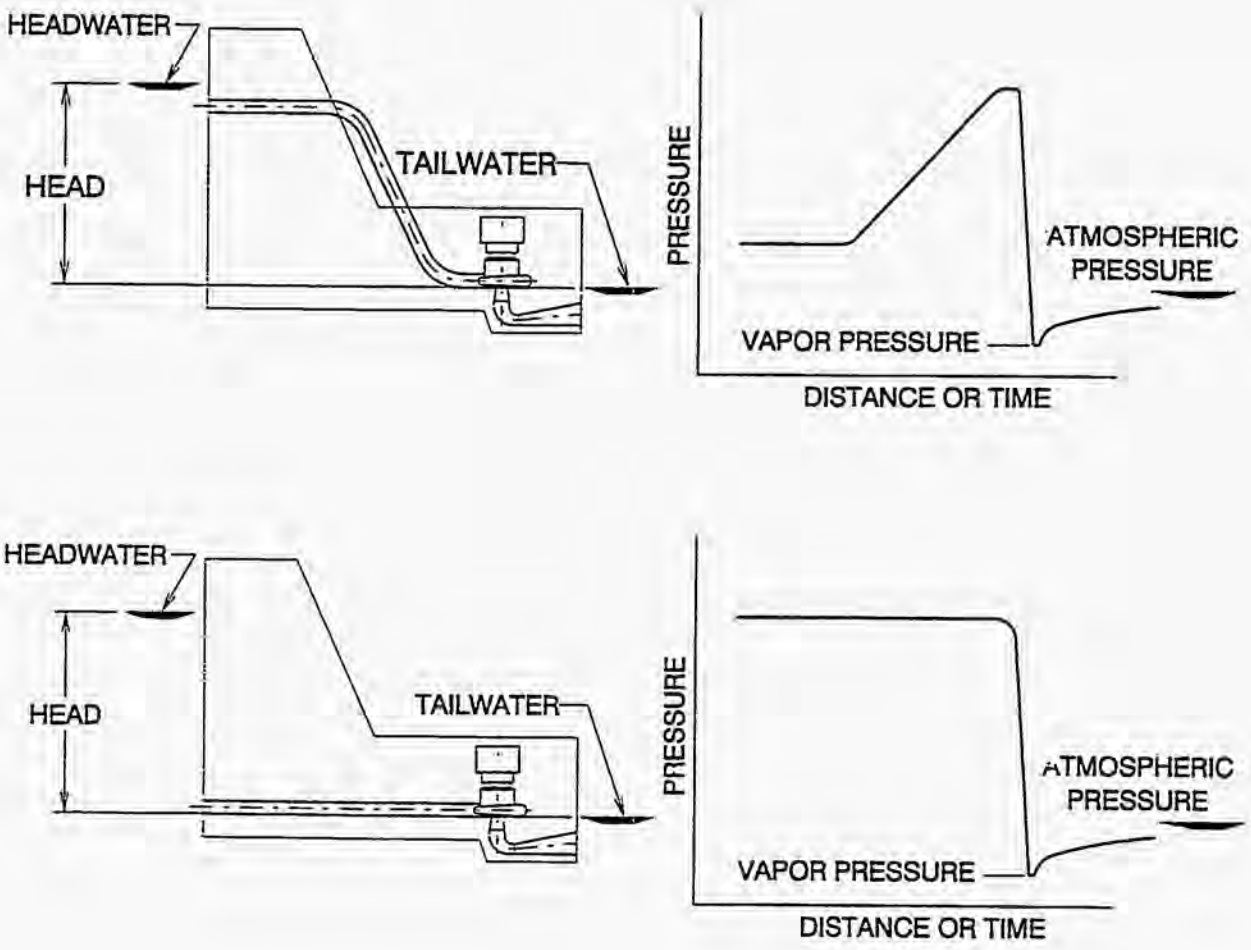


Figure 4.3.4-1 Alternative Plant Configurations Having Different Pressure Acclimation

4.3.5 IMPLICATIONS OF SURVIVAL PREDICTION FOR TURBINE DESIGN AND OPERATION

4.3.5.1 Evaluation of Francis Turbine Number of Blades

Summary

Fewer blades appear to be a feasible design concept. For large turbines passing small fish, high survival can be achieved. For small Francis turbines with large fish, the best solution for fish survival may be to keep the fish out of the turbines. This section presents the results of a design study where a series of different turbine runner designs were developed and verified to operate cavitation free at the desired design point. Geometrical differences in the designs are shown along with estimates of fish passage strike survival.

Discussion

A detailed hydraulic design was made for a series of Francis turbine runners. Each design had a different number of blades. Precise contours were developed for the shape of the crown, band, and runner blades. Each runner was designed to have identical performance characteristics, i.e. had identical specific speed, discharge, power output, and cavitation characteristics. To achieve identical cavitation performance, designs with fewer blades required that each blade be longer. The determination of the complete characteristics of each design (efficiency, pressure pulsations, runaway speed, etc.) is beyond the scope of this study. A preliminary conclusion from the study is that a reduction in number of blades seems to be a modest extension to current design methods and seems likely to produce good turbine characteristics. Some performance and cost tradeoffs may need to be accommodated for designs having fewer than the traditional number of blades.

The designs were developed for a specific design condition: a diameter of 5.41m, a head of 91.4 m, a discharge of 215 m^3/s, and a power of 180MW. Survival prediction was based on the strike equation. The fish length used was 150 mm and the lambda correlation value was arbitrarily chosen as 0.2. The value of lambda that would be most appropriate for Francis turbines is unknown, but values in the range of 0.1 to 0.2 were determined from Kaplan survival tests. Runner designs having 25, 18, 15, 13, and 11 buckets were developed. Figure 4.3.5-1 compares the water passage shapes of the designs. Table 4.3.5-1 shows the impact of bucket number on calculated survival, as well as the survival for a smaller turbine with the same operating point (head coefficient and discharge coefficient).

Design	Number of Blades	**using D=1.0m** Survival Probability (%)	**using D=5.41m** Survival Probability (%)
New	25	89.7	98.1
Original	18	92.6	98.6
New	15	93.8	98.9
New	13	94.6	99.0
New	11	95.5	99.2

Table 4.3.5-1 Predicted Survival for Various Number of Blades

4.3.5.2 **Evaluation of Francis Turbine Specific Speed**

Summary
Through the use of a simplified sizing exercise for a turbine at a new hydro site, the role of specific speed is demonstrated. The principal effect of choosing a high specific speed for the head at the site is to reduce the turbine size, thus increasing the strike probability. The new term in the strike formula was found to have a significant contribution to the calculated strike probability.

Discussion
The strike equation was used to conduct an evaluation of design tradeoffs that might occur for the analysis of a new hydro site. The characteristics of turbines vary as a function of specific speed. As the selected specific speed increases, the turbine becomes smaller, operates at higher speed, and has increased susceptibility to cavitation for a given centerline elevation. The following scenario was used to evaluate calculated fish strike for different turbine designs operating at their best operating point.

Number of units	1	
Discharge	28.3 cms	(1000 cfs)
Head	25.91 m	(85 feet)
Number of blades	13	
fish length	152 mm	(6 inch)

A number of considerations were ignored, such as unit submergence required to avoid cavitation, operational flexibility, energy production during a yearly flow duration cycle, etc. The result shown in Figure 4.3.5-2 is that fish survival due to strike is enhanced by selection of lower specific speed units for the head at the site. Further study for a specific plant would be required to evaluate other operating conditions to permit an overall judgment.

Since the strike equations used here are new, the contribution of the additional term in the strike equation due to the tangential length of a fish was examined. The strike probability was calculated separately for each term in the strike equations for these same Francis units. Figure 4.3.5-3 shows that the additional term is significant compared to the overall strike probability, and becomes relatively larger as specific speed decreases.

4.3.5.3 **Evaluation of Adjustable Speed Turbines**

Summary
Adjustable speed offers the possibility of improved fish survival. Adjusting the speed to compensate for head changes will allow the head coefficient characterizing the turbine performance location on the turbine performance hill curve to be kept at a more favorable operating point than a design having constant speed.

Discussion
When a turbine is operating at head and discharge coefficients that are not optimum for fish survival, alteration of the rotational speed offers the opportunity to move the operating point to a more favorable location on the hill curve. Consideration of the Kaplan turbine leading edge strike probability, Figure 4.3.2-7 indicates that higher discharge coefficients are always better, and higher head coefficients are typically better. Consideration of the Francis Turbine leading edge strike probability, Figure 4.3.2-13 indicates a different variation with discharge coefficient but, higher head coefficients are always better. The excess energy dissipation mechanisms relating to fish mortality, based primarily on the results of the Wanapum

fish survival tests, indicates that the optimum fish survival occurs at the minimum TAL, which occurs at a head coefficient near best efficiency heads and a discharge coefficient greater than the best efficiency discharge, but less than the maximum possible discharge. Combining both of these considerations leads to the presumption that the overall minimum fish mortality location on the hill curve occurs near the best efficiency head coefficient and at a discharge coefficient that is greater than the best efficiency discharge, but less than the maximum possible discharge.

Operational considerations will have an effect on mortality. In the case of operation at a low head, a lower rpm will change the head coefficient to a higher value. If it is desired to discharge the same discharge at the lower rpm, the discharge coefficient must increase. This would be accomplished by opening the wicket gates. If this desired operating point is greater than the maximum gate opening, then this condition can not be realized. Also, it may be that this operating point is in a region of higher TAL, or possibly cavitation. A precise analysis of some of these effects could be evaluated for a particular design but is beyond the scope of this work.

4.3.5.4 **Critical Velocity Implications for Specific Turbine Components**

Summary

A survey type of analysis was performed to evaluate average values of velocity in several regions of a turbine to permit a rough assessment of potential mortality. For structural piers, in turbine intakes, typical velocities are significantly less than 5 m/s. Therefore, strike on these bodies seems to be of no concern.

For stay vane entrance edges and for runner blade tips, typical velocities are shown on Figure 4.3.5-4, as well as mortality results for fish impacting solid objects and entering water adapted from Bell (1991). It is noted that that Bell provided this data without information on test design, uncertainty estimates or protocols used. Although Turnpenny (1992) found that an impact with an airfoil shape at 5.2 m/s caused little damage and no mortality, little supporting data is available to evaluate these mortality results.

The velocity on stay vane entrance edges is a function of the head. Velocities of 5 m/s at the stay vanes of Kaplan turbines are expected when the head exceeds approximately 17 m (56 feet). Therefore, stay vanes should not be ignored except for very low head turbines. The tip speed of runners essentially always exceeds 5 m/s and all blade strikes may be lethal. Also, the lowest speed portion of a Kaplan blade, (near the hub), will experience relative velocities between the blade and the water exceeding 5 m/s when the head exceeds 3 m.

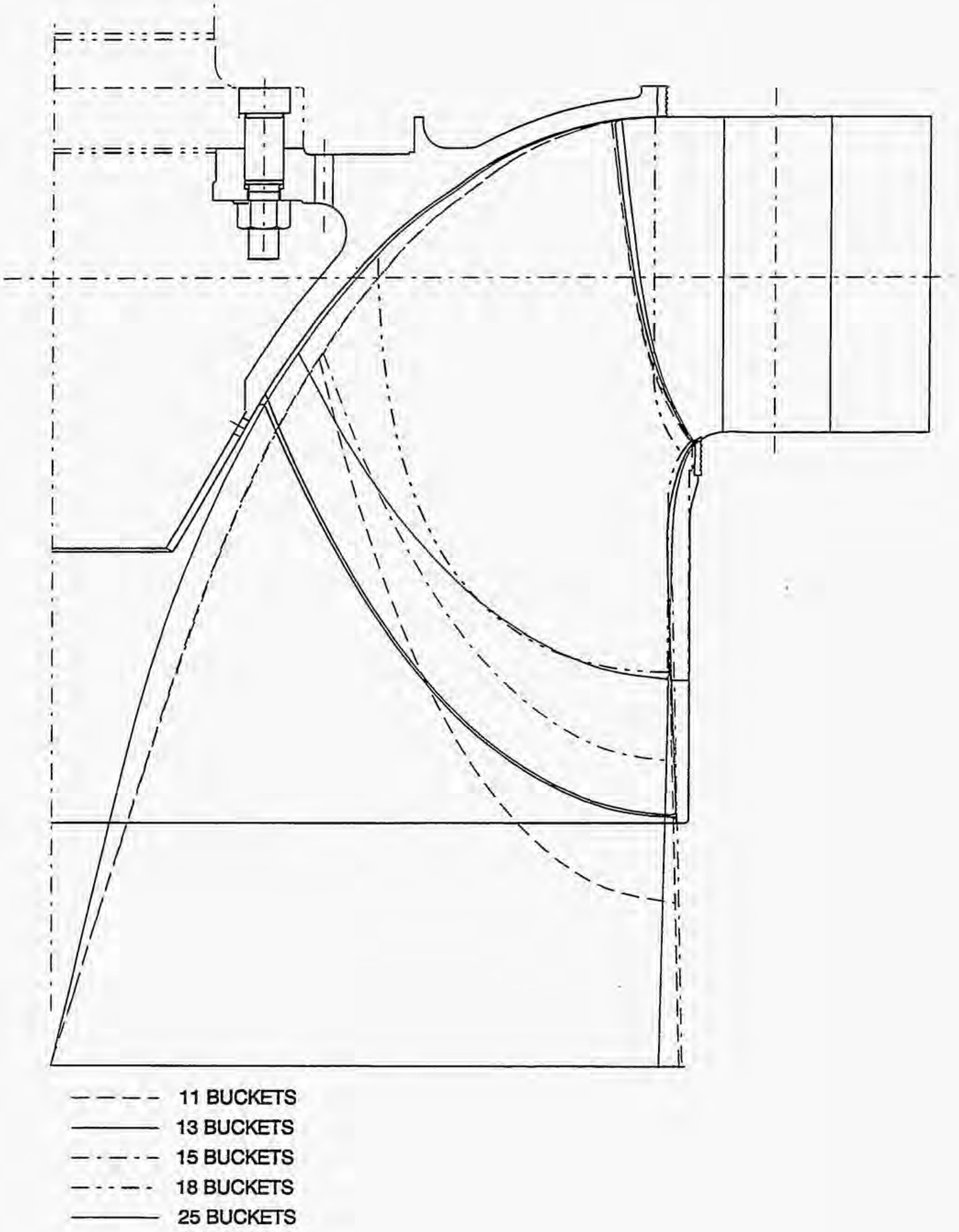


Figure 4.3.5-1 Francis Turbine Design Study: Comparison of Water Passage Shapes as a Function of Number of Blades

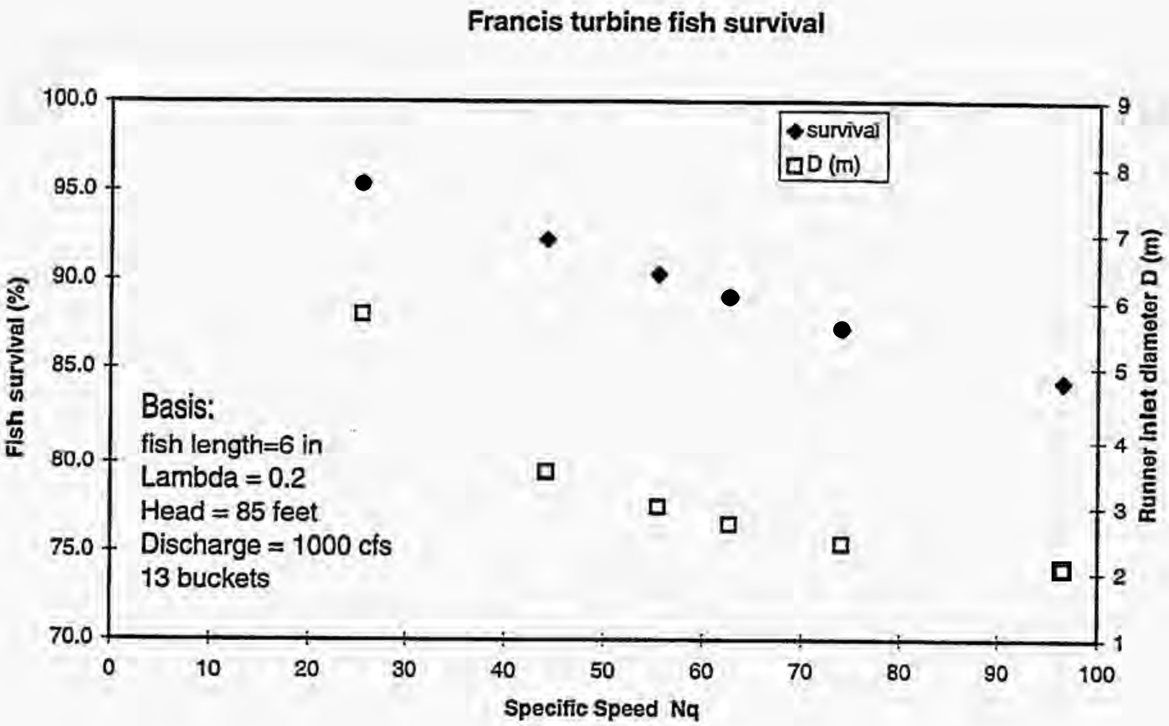


Figure 4.3.5-2 Francis Turbine Design Study: Comparison of Leading Edge Strike Probability and Turbine Size for Several Values of Specific Speed

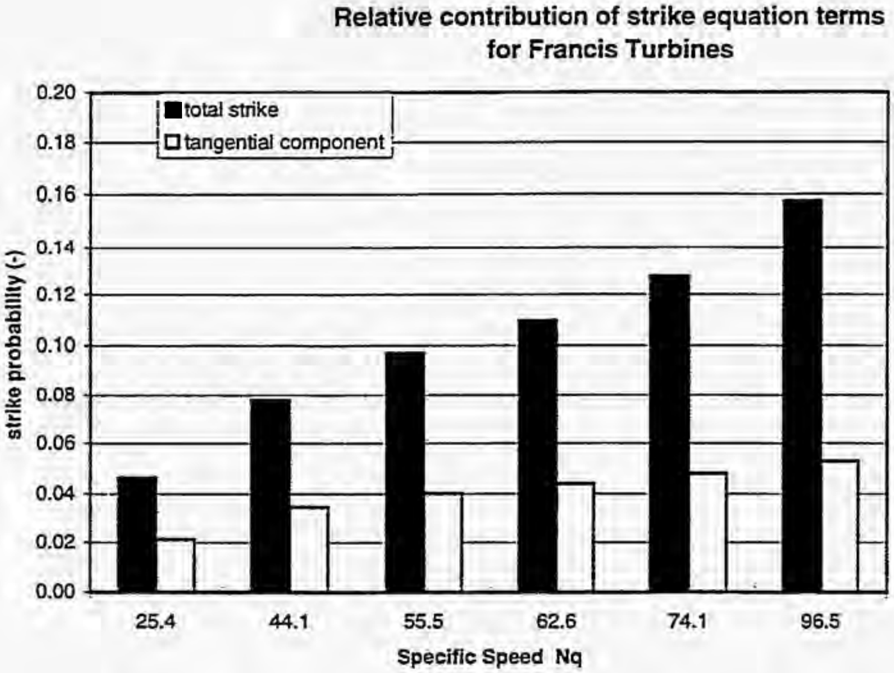


Figure 4.3.5-3 Illustration of Contribution of the Tangential Fish Length to Total Strike Probability

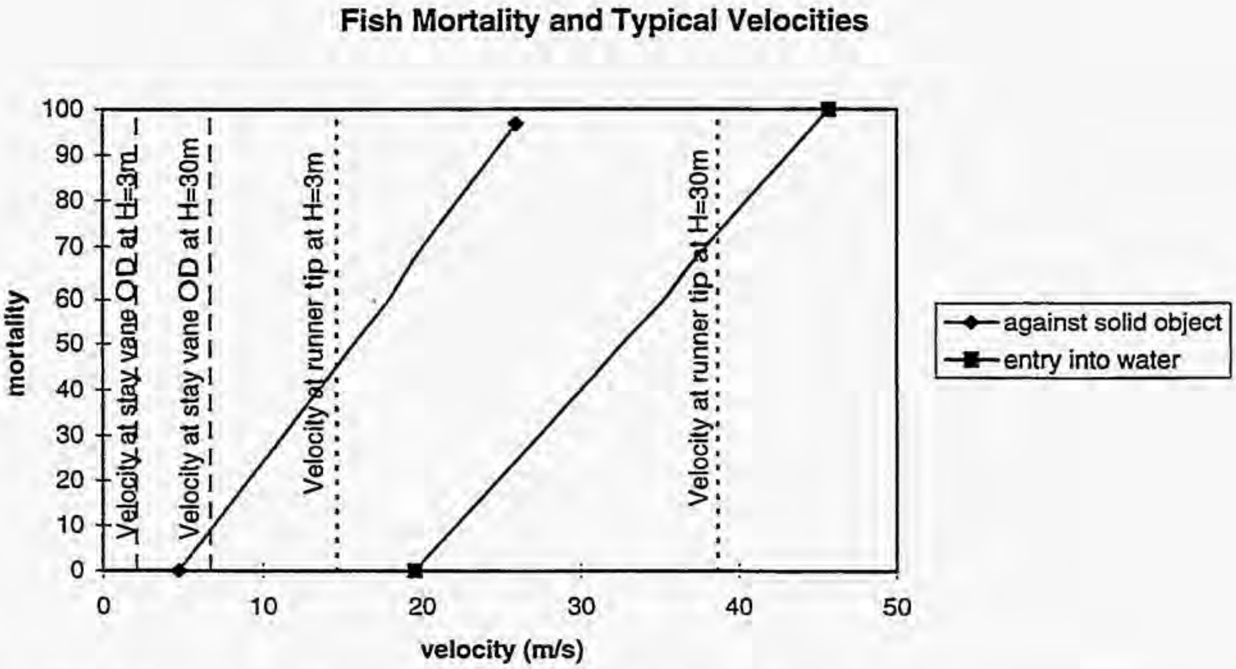


Figure 4.3.5-4 Comparison of Typical Turbine Velocities with Mortality Results. Mortalities Adapted from Bell. 1991

4.3.6 ZONAL DEPENDENCE OF INJURY/MORTALITY MECHANISMS

Summary

Injury mechanisms are associated with specific zones of turbine geometry. This section introduces the concept.

Discussion

A turbine should not be treated as a black box into which fish are carried and out of which some injured fish and some uninjured fish emerge. The survival of the fish is highly dependent on the zones of the turbine traversed by the fish path.

For the purpose of evaluating the fish injury potential of a turbine, the turbine should be considered to be comprised of a number of separate zones and sub zones. Each zone's geometry and fluid characteristics will have a unique effect on the fish. Some zones are fish friendly, while others cause damage. As an example, the turbine can be divided into zones associated with the principal structures. Zones associated with the intake upstream of the turbine (penstock or other civil structure with trash racks), the near inlet (spiral case or semi spiral case), the stay vanes and wicket gates, the runner and the draft tube could be considered. In the vicinity of the runner, the zones could be further subdivided (Figure 4.3.6-1). Here, the annular regions of the water passage could be designated as the hub zone, the mid zone and the tip zone for Kaplan turbines, and other zones for Francis or propeller turbines. The annular regions could be further subdivided into near blade zones and between blade zones. Within the hub zone, there would be a separate zone per blade, each of which may contain gaps with sharp edges and associated fluid injury mechanism sources such as cavitating vortices, regions of high shear and so forth. Experimentally measured mortality correlated with the number of turbine blades, for example, would contain the effects of both mechanical strike and fluid induced mortality. Experiments to determine the effect of certain mechanisms on fish mortality will need to recognize the zonal nature of damage mechanisms and be designed to carry fish into appropriate zones.

In the discussions that follow, the concepts of the zonal nature of the turbine and the zonal effects on injury/mortality mechanisms will be used.

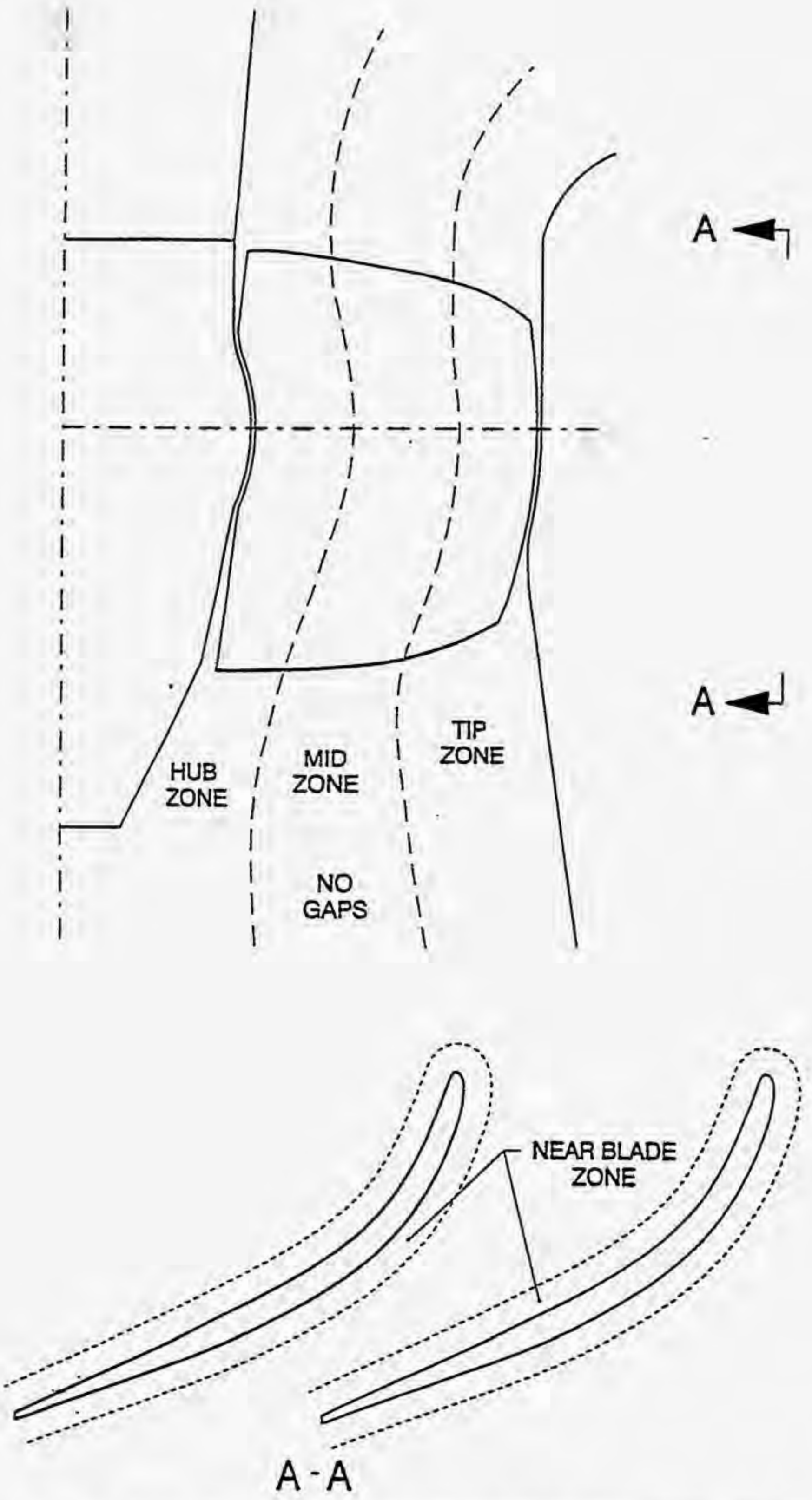


Figure 4.3.6-1 Fish Survival is Strongly Affected by Transit Through Different Zones of the Runner

4.4 INSIGHTS TO INJURY MECHANISMS AND SURVIVAL PREDICTION METHODS OBTAINED FROM EVALUATION OF FISH SURVIVAL TEST RESULTS

4.4.1 INTRODUCTION

Data selected in Section 4.2 are evaluated herein to look for trends supporting concepts presented in Section 4.3. Kaplan, Propeller and Francis data are used, but treated separately in the evaluation. Note that statistics have not been relied upon for our evaluations. Instead, each evaluation outcome was examined for it's merits in integration of biological significance with turbine flow physics. This is to avoid confusion between statistical significance and practical significance

Fish injury during turbine passage and therefore survival is dependent on a number of factors. These have been outlined in Section 4.3. No single mechanism can be correlated with observed injury or mortality in fish passage survival testing. Instead, a combination of effects come into play which are dependent on the type of turbine, the turbine geometry, how the turbine and the turbine plant is operated, as well as the species and size of fish, the fish's location in the water column and the fish's behavior in approaching and passing through the turbine. The mechanisms of injury can be characterized into zones within the turbine. Fish passing through different zones can encounter different mechanisms and therefore experience different survivability.

Because of some unique characteristics of the hydroelectric dams on the Columbia River Basin, and the intensity of effort in improving fish passage survival there, significant fish passage testing has been conducted on Kaplan turbines providing a relatively good availability of comparable data, under relatively controlled conditions. The testing investigated the effect of intake modifications, turbine operating conditions, and turbine geometry with the specific objectives of improving turbine passage conditions. Insights gained from these tests are discussed separately below. The tests include those at Rocky Reach Dam and Wanapum Dam on the Columbia River and Lower Granite Dam on the Snake River. At Rocky Reach Dam the experiments were conducted to identify potential sources of fish injuries so that turbine design and structural modifications could be incorporated into a new replacement turbine. At Wanapum Dam, the investigation was done to determine the effects of turbine operating performance on the survival of fish entrained at two depths. Studies at Lower Granite Dam were conducted to provide baseline survival and sources of fish injury for comparison with turbine operation during the proposed reservoir drawdown and to evaluate the potential effects of extended length screens on unguided fish in passage through turbines, (Figure 4.4.1-1). For all tests, data were specifically obtained to evaluate the effects of turbine operating efficiency, fish entrainment depth, differences between turbine intake bays, and presence or absence of intake screens on fish survival and injury rates. With some minor exceptions, these types of data are generally not available from other regions of the country. However, even in this database a wide range of operating conditions, configurations, and fish behavioral reactions have not been tested to draw a predictive statistical relationship. The above data and others from previous testing are discussed below.

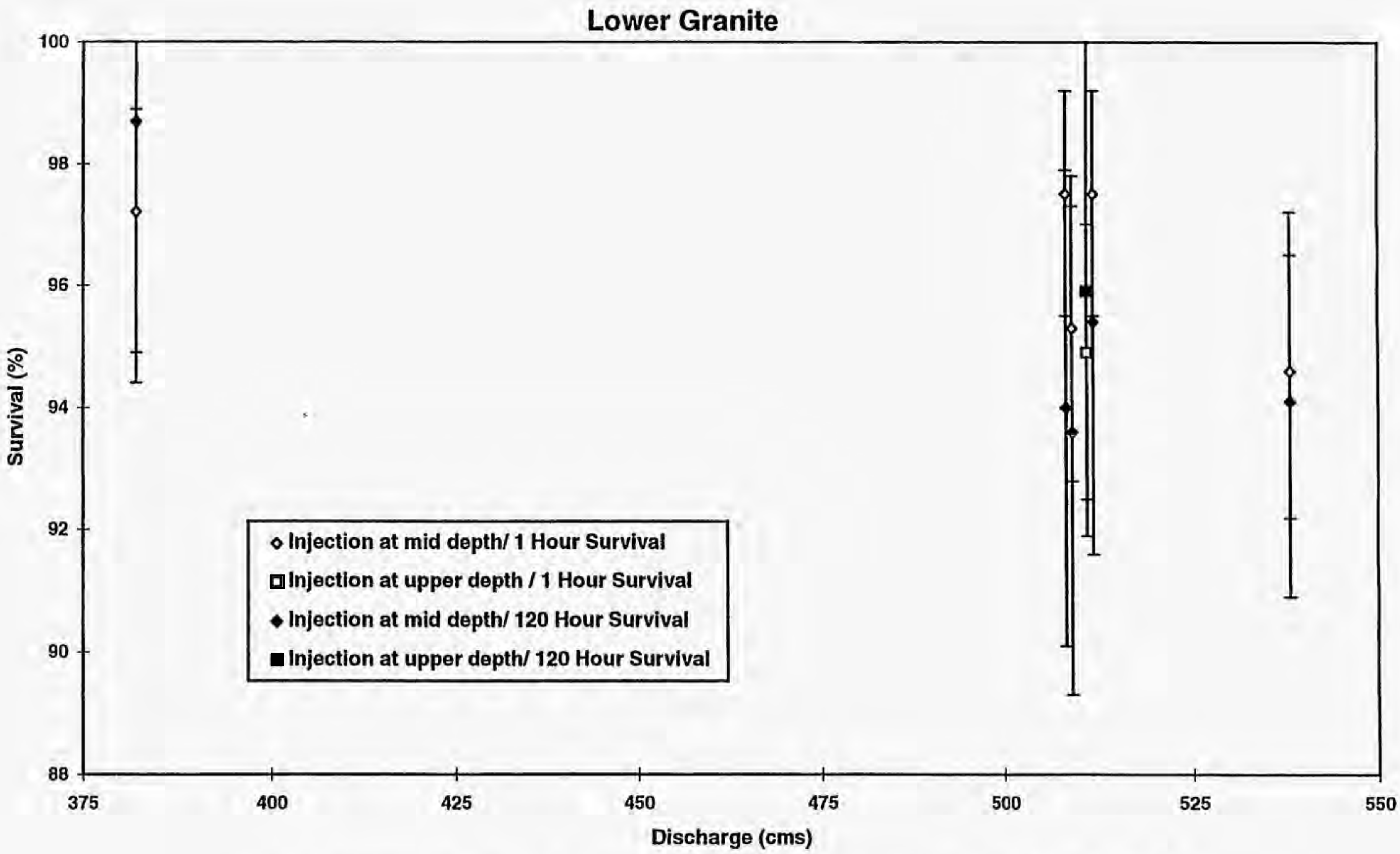


Figure 4.4.1-1 Survival Results At Lower Granite, Effect Of Injection Location And Operating Point

4.4.2 EFFECT OF LOCATION OF FISH WITHIN THE WATER COLUMN

Summary

The geometry of the turbine intake influences the distribution of fish as they pass through the turbine internal geometries. For Kaplan turbines with large well designed intakes and low inflow distortions (such distortions may be induced by upstream flow nonuniformities, inserted structures, trash on the trashrack or the state of operation of the neighboring units), the location of fish relative to the height in the intakes reasonably predicts the relative height of their path through the stay vanes , wicket gates and within the runner. For Francis turbines this is often not the case. If the fish position is known as it passes through the turbine, observation of fish injuries incurred can be related to the local geometry and to the resulting injury mechanism.

Discussion

Based on the background developed in Section 4.3, the mechanisms associated with fish injury are related to the turbine geometry and its operation. In general, the mechanisms are localized and do not occur at the same time and in the same manner throughout the turbine water passageway. Thus, the probability of fish injury and the type of injury are related to the location of the fish relative to the turbine geometry or to the zone through which the fish pass. The location of the fish within the turbine geometry zones are related to their starting position in the intake, the flow characteristics of the turbine and their own free will response to fluid stimuli.

The geometry of a Kaplan intake frequently takes the form of a semispiral intake where the initial cross sectional geometry is a box like structure that has a significant distance from the upper portion of the water column to the bottom portion. Section 5.3 presents the results of CFD analysis of such a Kaplan turbine intake showing flow paths of neutrally buoyant particles injected into the flow. Fish without the effect of fish volitional movement (FVM) would pass like neutrally buoyant particles. For Kaplan turbines, in the absence of FVM and significant disturbances within the turbine flow field, the flow will transport fish through the turbine in such a manner that fish in the upper water column pass by the top of the wicket gates and stay vanes and enter the runner near the hub. Fish near the lower portion of the water column will be transported past the stay vanes and wicket gates near the lower portions of these structures and into the blades near the runner tips. Fish entering the intake in the mid elevation will be transported to the zone in the center of the blades. Some Kaplan turbines have full spiral intakes connected to the upper reservoir by a penstock. In this case, the characteristics of the intake flow distribution will be as described below.

The geometry of a Francis intake is significantly different from that of a semispiral Kaplan. Typically, Francis turbines draw water into either the bottom or upper portion of the dam or intake structure and lead the water into a penstock (pipe) which carries the water to the full spiral intake. Section 5.3 presents the results of CFD analysis of a Francis full spiral turbine intake showing predicted flow paths of neutrally buoyant particles injected into the flow. In contrast with Kaplan turbines, neutrally buoyant particles seeded into the flow at the full spiral intake for Francis turbines enter the stay vanes and runner at different locations dependent on their starting radius. Particles near the top on the spiral inlet can pass the stay vanes at the top, mid region or bottom of the vanes and then in similar positions of the wicket gates and into the runner entrance edge. The effect of the geometry of the penstock, which leads water into the spiral case, on particle distribution is also significant. Bends in the penstock can set up swirling flow patterns, redistributing the neutrally buoyant particles. Some Francis turbines have semispiral intakes much like the classic Kaplan and in that case have similar intake flow distributions as a Kaplan described above.

Based on the above, it can be seen that the location of the fish in the intake of the turbine can be related to flow characteristics and geometry of each zone through which the fish pass. This has been observed in prior studies. For Kaplan turbines, it has been hypothesized that survival may be lower for fish entrained at greater depth because fish are likely to enter turbines nearer the blade tip (Eicher Associates 1987; Ferguson 1993; Turner *et al.* 1993). However, this hypothesis did not take other factors into consideration such as the detailed geometry of the turbine, the quality of the hydraulic design, the change in geometry as the point of operation changes, the impact of flow obstructions within the water passages such as fish screens and so forth. At Lower Granite Dam on the Snake River, the estimated survival (93.6 to 95.4%) of chinook salmon smolts entrained at mid-depth elevations was similar to that at the upper elevation (94.9%) with the turbine operation held constant at (Normandeau Associates *et al.* 1995). At Rocky Reach Dam Unit 3 on the Columbia River the point estimate (93%) of survival of chinook salmon smolts released at upper elevation was slightly lower (though not significantly) than for the mid-elevation releases, 94.7% (Mathur *et al.* 1996a). The turbine operated over a wide range of normal power outputs and discharges during the test. However, as discussed earlier, survival at Unit 5 varied with power output and was higher at 9 m (30 ft) depth than at 3 m (10 ft) depth. At Wanapum Dam, coho salmon smolts were introduced at two depths within a turbine that presumably swept them either near the hub or near the middle of the blade at four different operating efficiencies (Normandeau Associates *et al.* 1996a). The two release depths were selected on the basis of CFD model predictions which assumed that fish was incapable of changing its path while entrained in these flow streams. Fish released 3 m (10 ft) below the turbine intake ceiling were expected to pass near the hub and those released at 9 m (30 ft) below the intake ceiling were expected to end up at the mid blade region. The survival was consistently lower for coho salmon introduced at 3 m depth than at 9 m depth; depth-related differences in survival increased with increased discharge; the largest difference (9.3%) occurred at 184 cms (17,000 cfs). At other discharges survival at 3m depth was 4.4% to 5.2% lower that at 9 m depth.

The interrelationships between the mechanisms for injury and the flow characteristics that occur in different zones and how they vary with discharge will be explored in more detail below.

4.4.3 EVALUATION OF (N L / D)

Summary

While fish passage survival is influenced by N L / D, the effect of other parameters such as turbine geometry, turbine performance (efficiency, cavitation), the point of operation, and the quality of the experiment lead to enough data scatter that it is difficult to define the effect of a fish passage experiment without a carefully designed and executed test. For Kaplan turbine data (from 3 to 6 blades) the newly developed leading edge strike (or blade zone encounter) equations verify that N L / D correlates measured data well. An estimate for the strike mortality correlation factor, lambda, derived in Section 4.3 is developed for Kaplan turbines.

Background Information for a New Data Analysis Technique

A new data analysis technique is used. Previous analysis techniques have presented survival as a function of individual variables, such as peripheral speed. These previous presentations mask several effects that are known to occur simultaneously. For example, at a particular value of peripheral speed, various data points could exist for different fish length, or even different turbines having different number of blades. A useful data analysis technique would account for known correlations of certain variables and therefore, clarify whether other variables play a role. This is accomplished by combining measured survival with predicted survival as a ratio. This single quantity then contains information regarding the correlation of survival by the prediction method. In this case, the prediction uses the variable N L / D, as well as the correlation factor, lambda. Note that the predictive method used here does not include the variables head, peripheral speed, acclimation pressure, etc. Future use of this technique could employ more sophisticated prediction methods, when they become available. The success of this method requires that the prediction have reasonable success. A plot of this ratio versus a variable of interest, permits conclusions to be drawn regarding the isolated effects of that variable. If this variable had observable influence, with other effects occurring in a random manner, the ratio values would form a definite trend. If this variable had no observable influence, the ratio values would vary about unity with data points scattered above and below unity in a random manner.

Note that although the "strike" equation is used as the prediction method, this does not imply that strike is the only phenomena occurring. As an example, fluid mechanisms such as those associated with Kaplan blade gaps and entrance edge shear are related to the flow in the zone near the blades and are therefore, related to the number of blades. In the analysis of the data, several mechanisms of injury may contribute to the correlation, as long as they are strongly related to the geometrical variable N L / D, and are contained within the lambda value. Because of the zonal nature of the mechanical and fluid mortality mechanisms associated with the blade zone, the term "blade zone encounter" (BZE) might be a better choice of words replacing "leading edge strike" in the sections that follow.

Discussion

Fish Length: Fish length has long been known to directly influence fish survival. Evaluation of limited data selected in Section 4.2 where significant changes in length were evaluated confirms the effect. To evaluate whether the strike (BZE) equation correctly accounts for the fish length, the ratio of measured fish survival to fish survival predicted by the strike equation is used. Overall, this ratio will approximately equal unity due to the choice of lambda equal 0.2. If the strike (BZE) equation captures the essential influence of fish length on mortality, these ratio data points will occur at values both higher and lower than unity without any significant deviations at large or small values of fish length. For example, if less severe mortality effects would occur for large fish, their higher survival would cause the ratio value to increase. This would cause a cluster of data points at higher ratio values for large fish.

For a Kaplan turbine at Hadley Falls, where significantly different length fish were used, the ratio of survival data shown in Figure 4.4.3-1 shows no clustering. The data point for the largest fish length, for example exceeds unity by a similar amount that the data points at the smallest fish length differ from unity. A similar analysis for all data from Table 4.2-1 (Figure 4.4.3-2a and b) shows greater scatter (presumably due to various uncontrolled aspects of the experiments), but shows no obvious data clustering. Since the fish trajectories are unknown, calculations are presented for two values of the radius where fish are assumed to enter the runner, r/R = 0.4, and r/R = 0.75.

For reference, Figure 4.4.3-3 presents survival data directly. For fish of approximately 300 mm, the measured survival varies from 70 to 97%. This large scatter is due to the effect caused by the simultaneous variation of other important variables. With this type of data analysis, even sophisticated statistics have not been successful in identifying fundamental physical relationships.

Number of Blades: Another physical effect considered by the one-dimensional leading edge blade strike (BZE) model is based on the physical size of the turbine, the fish and the number of blades. For large Kaplan turbines, this aspect of fish survival was examined for turbines having the most common number of blades (5 and 6) where fish of nearly the same size were tested. Figure 4.4.3-4 shows that within the scatter associated with operating conditions, fish location, turbine geometry and so forth, the effect of the number of blades was not significant. However, when data from 3 and 4 bladed turbines were considered, the N L / D parameter of the strike equation did correlate the data, Figure 4.4.3-5. The measured mortality (calculated as 100% - survival) includes both effects related to strike (BZE) and to other effects. These other effects include stay vane or gate strike, wicket gate overhang gaps, TAL (theoretical avoidable loss) related to the inlet, stay vanes, gates and draft tube flows, as well as cavitation etc.

Plotting the ratio of measured mortality divided by predicted mortality, Figure 4.4.3-6, yields a ratio of approximately 0.2 which may be used to place an upper bound on lambda derived in Section 4.3.2 , at least for Kaplans. It is also noted that the overall correlation achieved with the strike (BZE) equation indicates the substantial role of the variable N L / D.

To more precisely estimate the lambda factor, more accurately designed survival tests which lead fish into zones dominated by leading edge strike (BZE) need to be conducted. Such a test is discussed in Section 4.4.6. Based on those test results, lambda was estimated as having a value of 0.1.

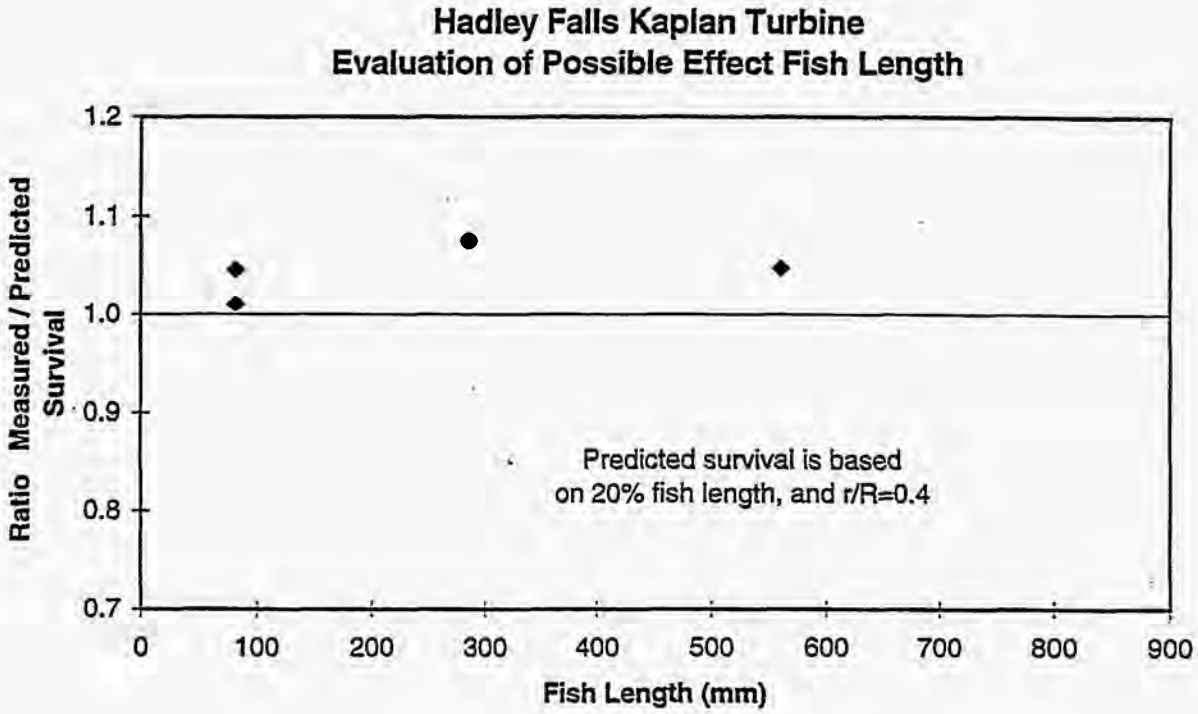


Figure 4.4.3-1 The Leading Edge Strike Equation Correctly Accounts For Fish Length

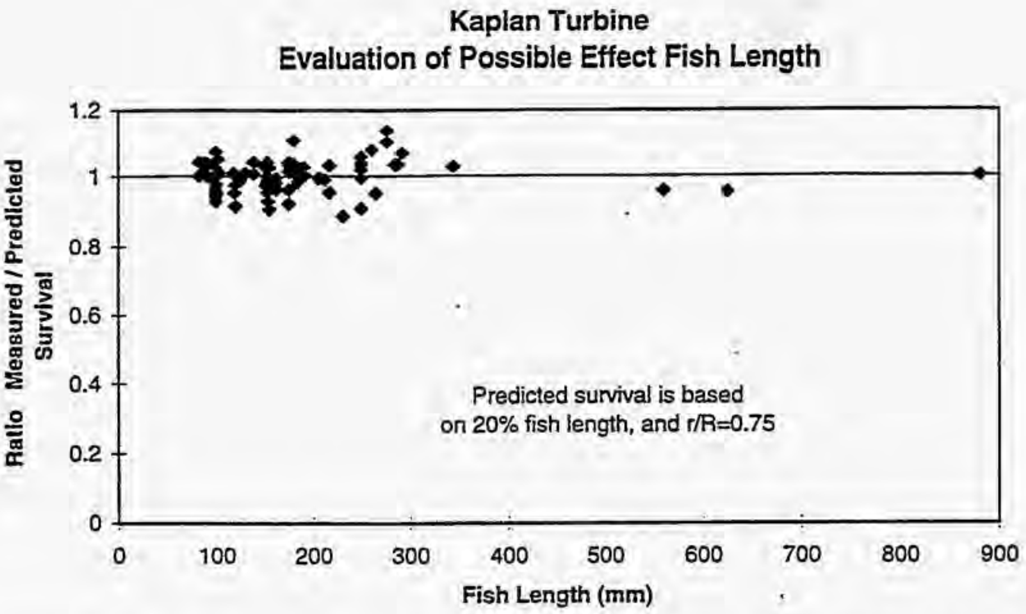


Figure 4.4.3-2a Evaluation of Fish Length Shows Good Correlation with Measurements if Fish Enter the Runner at 75% of the Blade Tip Radius

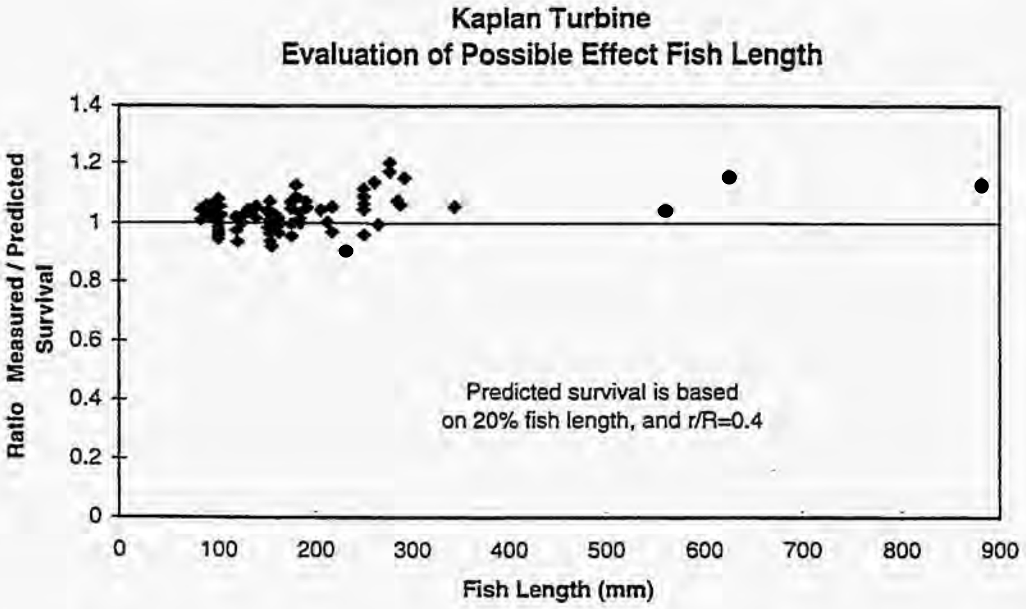


Figure 4.4.3-2b Evaluation of Fish Length Shows Reduced Correlation with Measurements if Fish Enter the Runner at 40% of the Blade Tip Radius

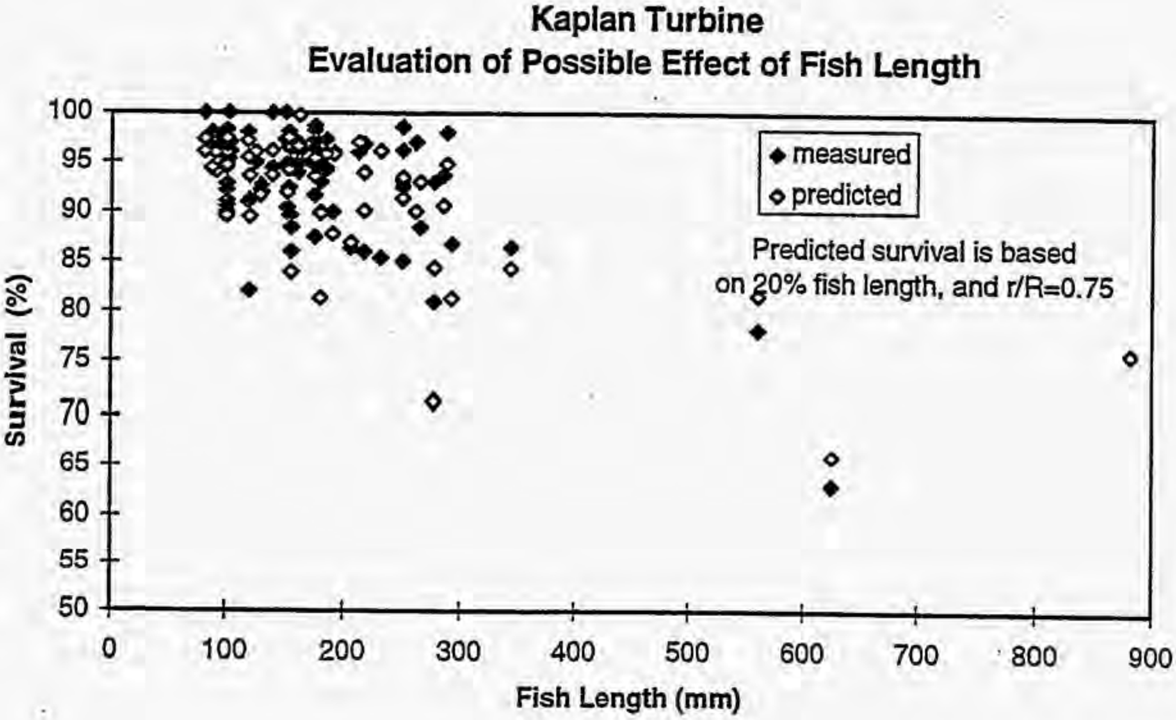


Figure 4.4.3-3 Survival Is Related To Fish Length But Simultaneous Variation Of Other Effects Prevents Clear Insights

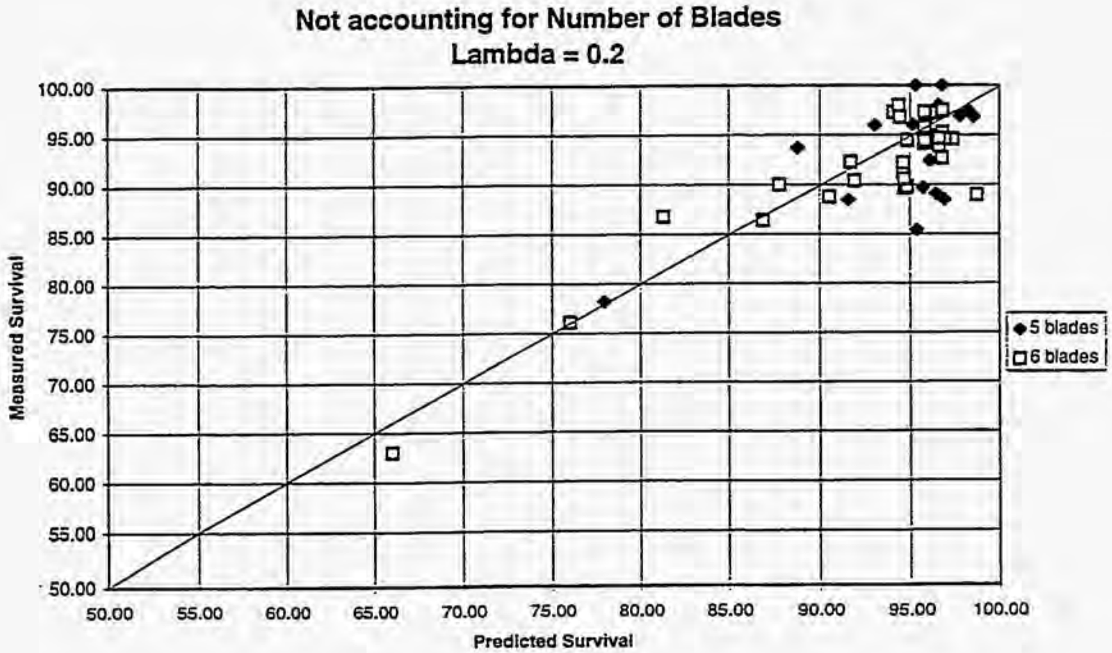


Figure 4.4.3-4a Predicted Survival is Insensitive to Five or Six Bladed Kaplans

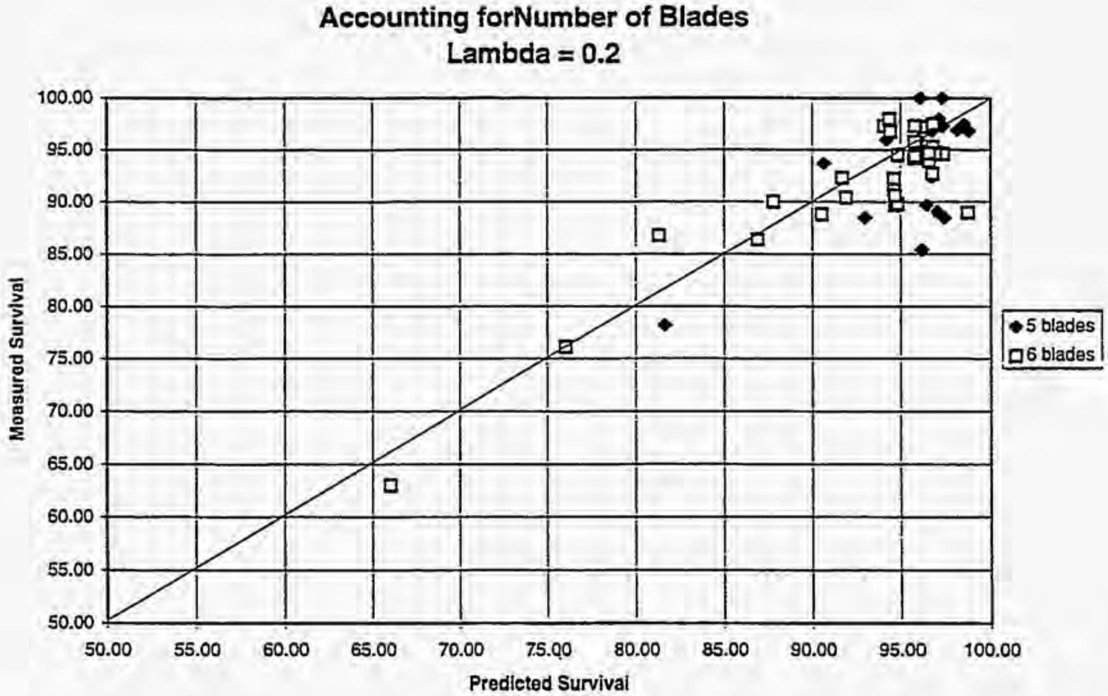


Figure 4.4.3-4b Predicted Survival is Not Materially Changed by Accounting for Five or Six Blades

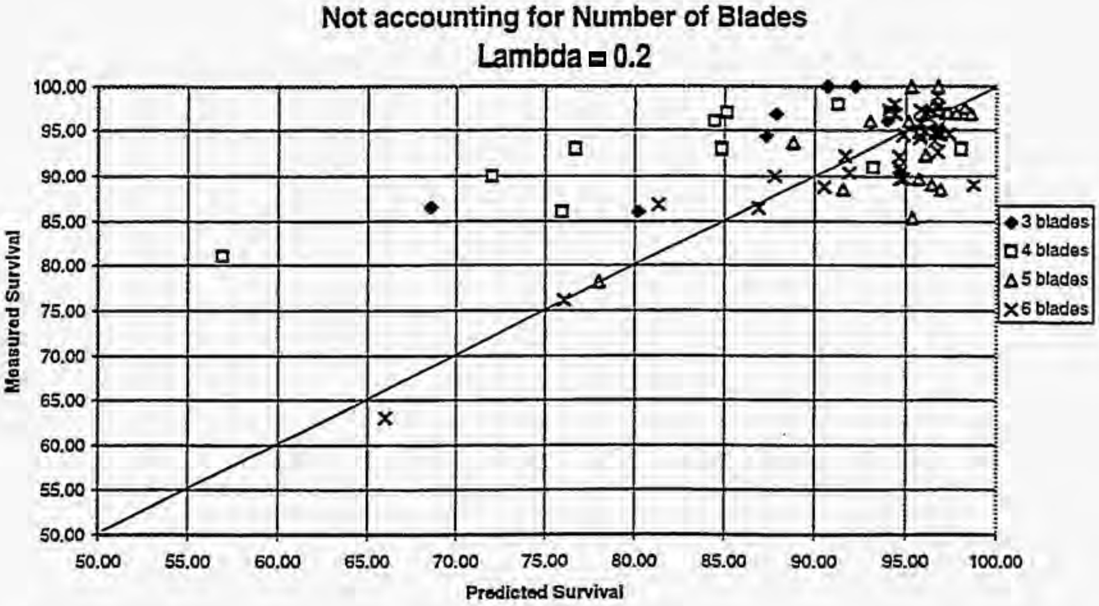


Figure 4.4.3-5a Predicted Survival is Degraded by Not Accounting for Number of Blades

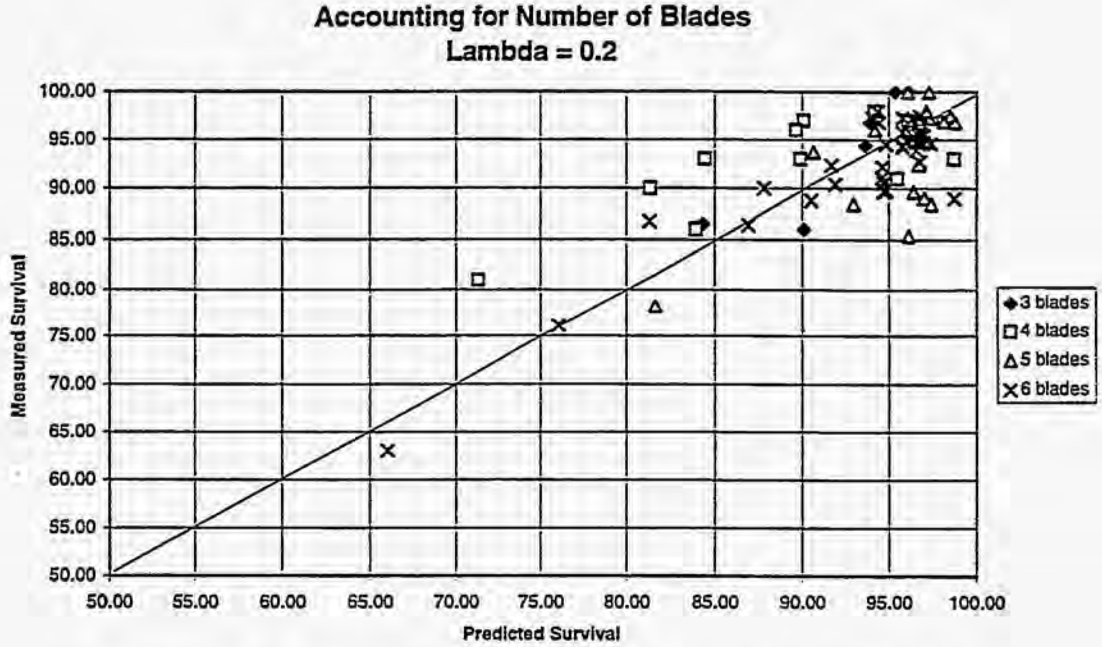


Figure 4.4.3-5b Prediction is Improved by Accounting for Number of Blades

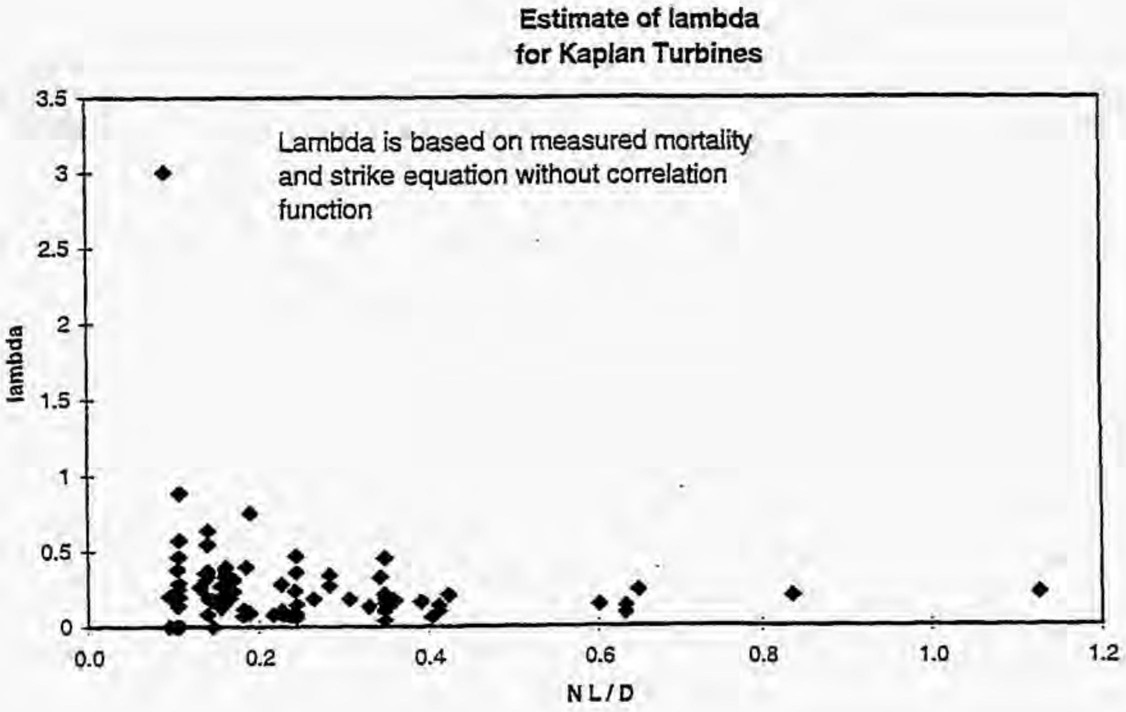


Figure 4.4.3-6 Calculated Lambda Factor For All Test Data

4.4.4 INVESTIGATION OF PERIPHERAL SPEED, TURBINE HEAD, INDIRECT EFFECTS, AND SPECIES

Summary

Using the data sets selected in Section 4.2, little correlation between fish survival and peripheral speed , turbine head, indirect effects, or species was observed.

Discussion of Peripheral Speed

One of the mechanisms associated with fish mortality is strike. While it had been suggested in the literature that peripheral speed was related to fish passage mortality, the following analysis of existing test data shows little correlation between fish survival and runner peripheral speed. All data from Tables 4.2-1 for Kaplan turbines and Table 4.2-2 for Propeller turbines was used. The use of all data introduces the variability of different test protocols, species used, fish length, fish injection location, internal turbine geometry, and other effects. These effects may introduce scatter to the data set, or, since the effects are not well understood, may introduce bias.

In an axial flow turbine, the peripheral speed varies with radius. Since the actual radius that would be appropriate for each passage test point is unknown, the peripheral speed at the blade tip is used. Figure 4.4.4-1a shows the ratio of measured to predicted survival for the Kaplan turbine data. If any clustering of the data exists, it's magnitude is modest compared to the scatter of the data. Figure 4.4.4-1b shows the result of Propeller turbine data analysis. Since no significant clustering of the data exists, no significant role of peripheral speed in fish survival can be demonstrated.

Several factors support this empirical finding. Peripheral speed is related to both energy of impact and the size and intensity of the shear zone near blade leading edges. The discussion of Section 4.3.5.4 indicates that for most turbine runners, independent of size and rpm, the peripheral speed of the entrance edge is above the critical value for injury and mortality due to mechanical impact. The theoretical analysis of Section 4.3.3.3 showed that although the blade entrance edge fluid shear zone intensity and size increases with peripheral speed, the size of the region having a lethal shear value is relatively small compared to the blade. Also, the total lethal zone (including mechanical and fluid effects) grows slowly with increasing runner peripheral speed. Virtually all runner blades have sufficient impact energy for a direct impact to cause mortality. Thus, that the mortality is insensitive to peripheral speed is not surprising.

Discussion of Turbine Head

The available data from Table 4.2.1 was analyzed for evidence that the head influences survival. This could occur due pressure reduction effects, or possible TAL or other energy dissipation mechanisms. Within the scatter of the data, no mortality differences were found (Figure 4.4.4-2).

Discussion of Direct and Indirect Effects

Several data sets are available that quantify the combined effects of direct and indirect mortality. Figure 4.4.4-3a and b and Figure 4.4.4-4 contrast the comparison of predicted and measured mortality for these effects. Since the effect of the location of fish during runner passage is unknown, calculations were performed for $r/R = 0.4$ and $r/R = 0.75$ for the direct data. For the rather narrow range of N L / D of the combined direct and indirect data, that only partially overlaps the direct effect data, no trend is evident.

Discussion of Species

The available data from Table 4.2.1 was analyzed for evidence that species have different susceptibility to injury. Within the scatter of the data, no mortality differences were found (Figure 4.4.4-5).

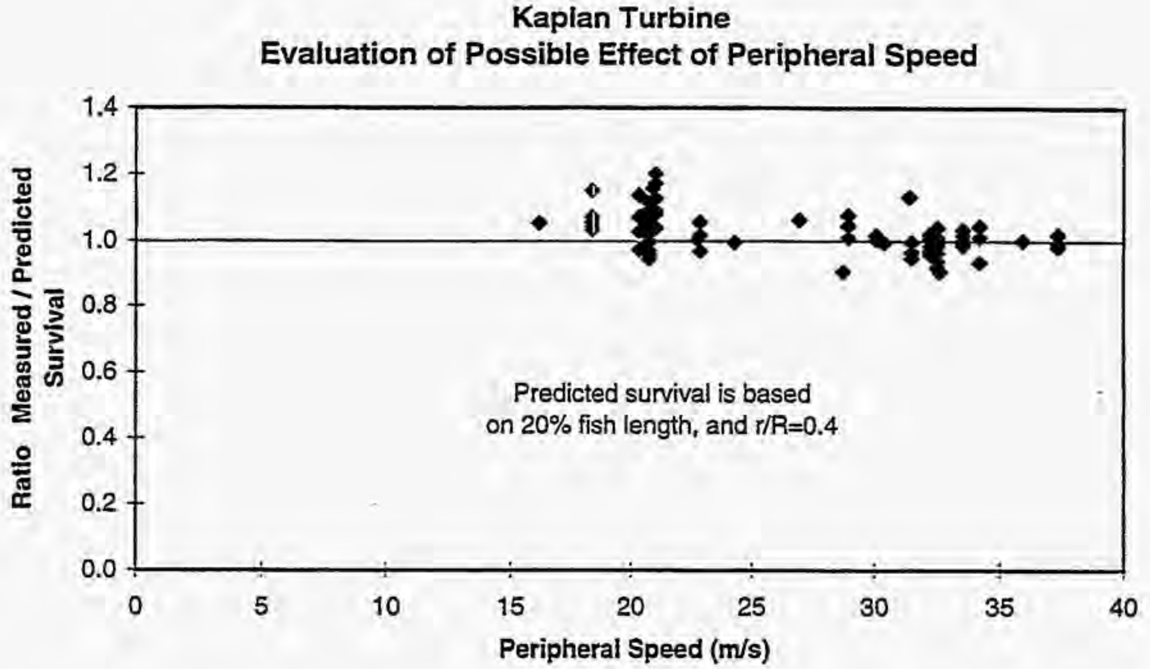


Figure 4.4.4-1a Peripheral Speed Is Shown To Have No Significant Effect For Kaplan Turbines

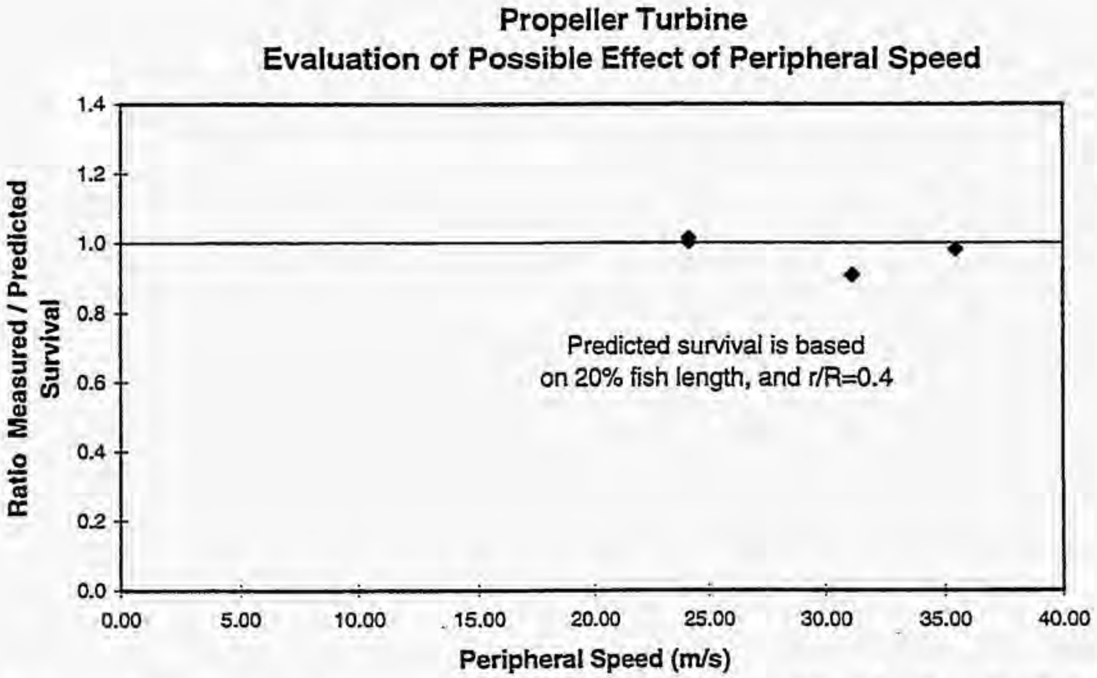


Figure 4.4.4-1b Peripheral Speed Is Shown To Have No Significant Effect For Propeller Turbines

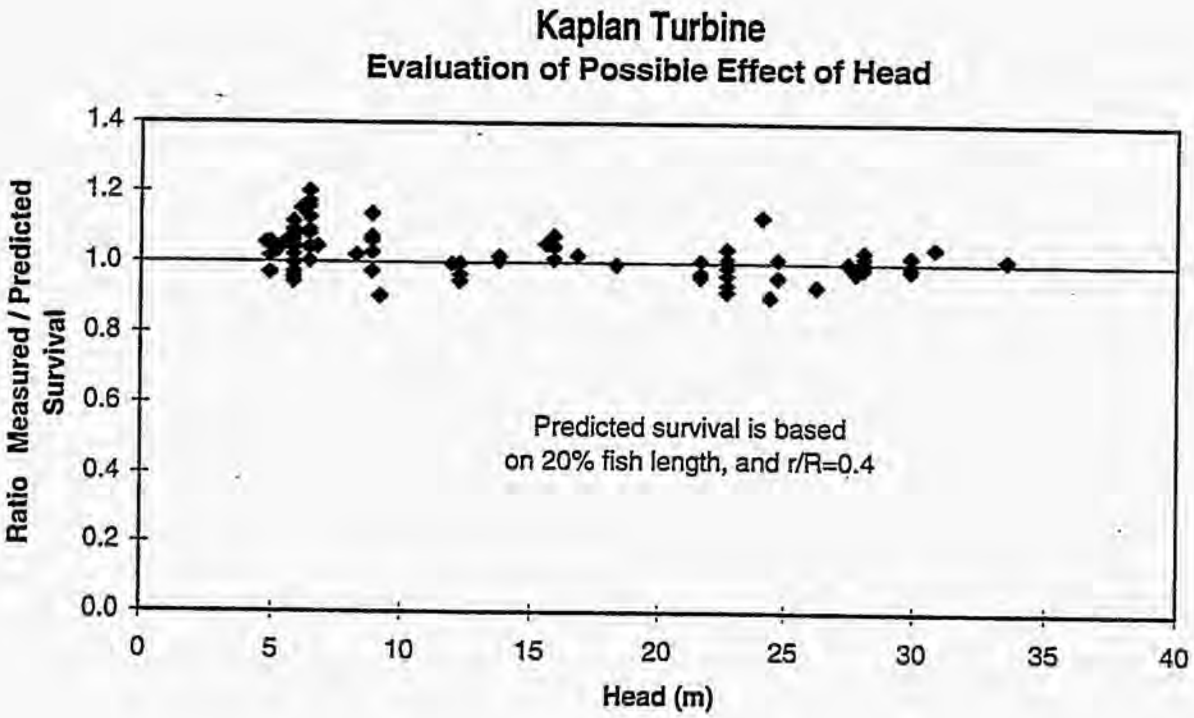


Figure 4.4.4-2 Head Is Shown To Have No Significant Effect For Kaplan Turbines

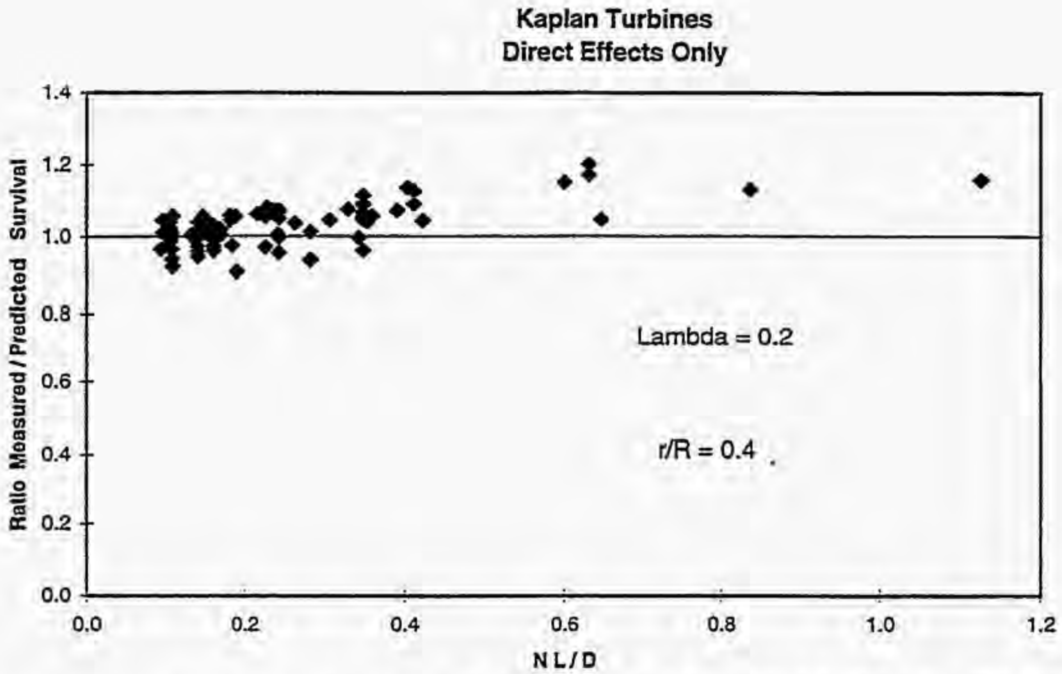


Figure 4.4.4-3a Direct Effects For Kaplan Turbines Are Well Correlated By N L / D – The Position Where The Fish Enter The Runner Has An Effect

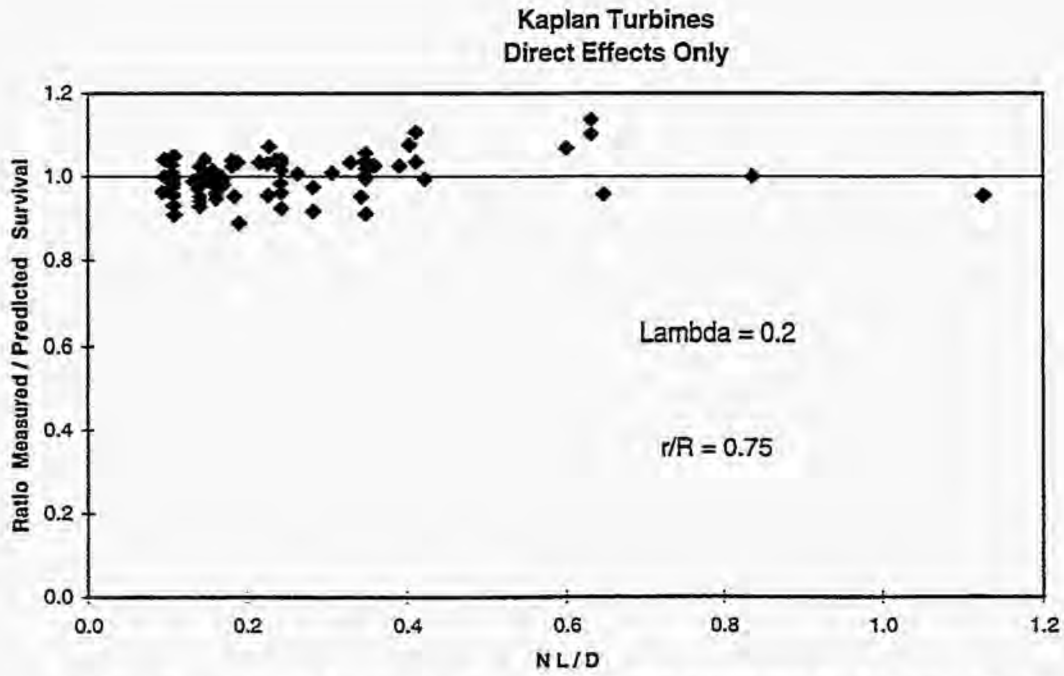


Figure 4.4.4-3b Direct Effects For Kaplan Turbines Are Well Correlated By N L / D – The Position Where The Fish Enter The Runner Has An Effect

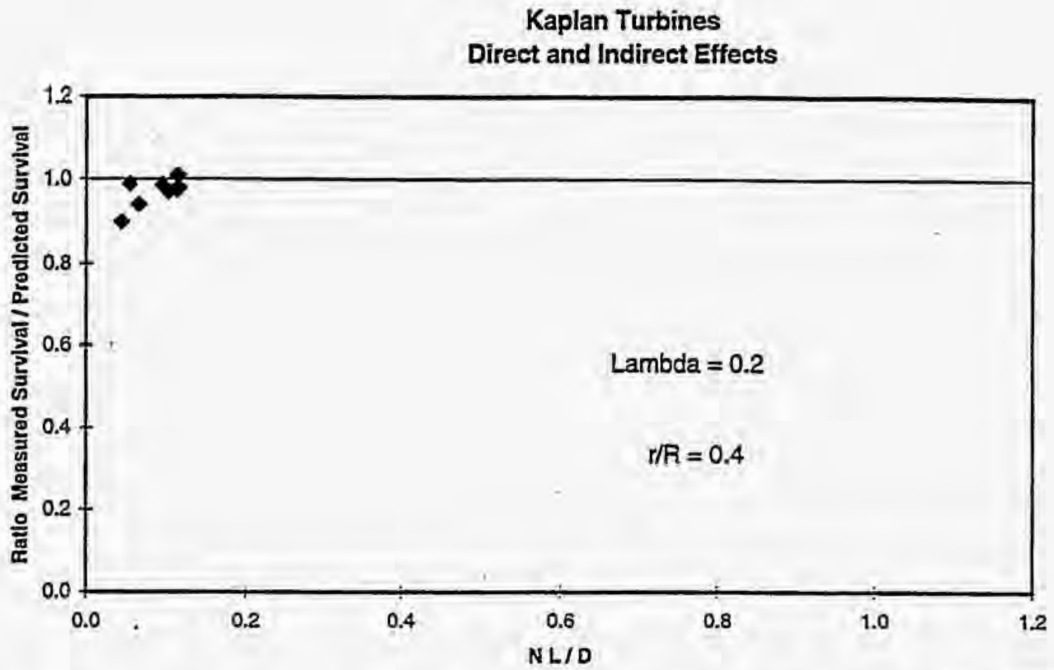


Figure 4.4.4-4 No Conclusion Is Drawn Regarding Possible Differences Between Direct And Combined Direct And Indirect Effects

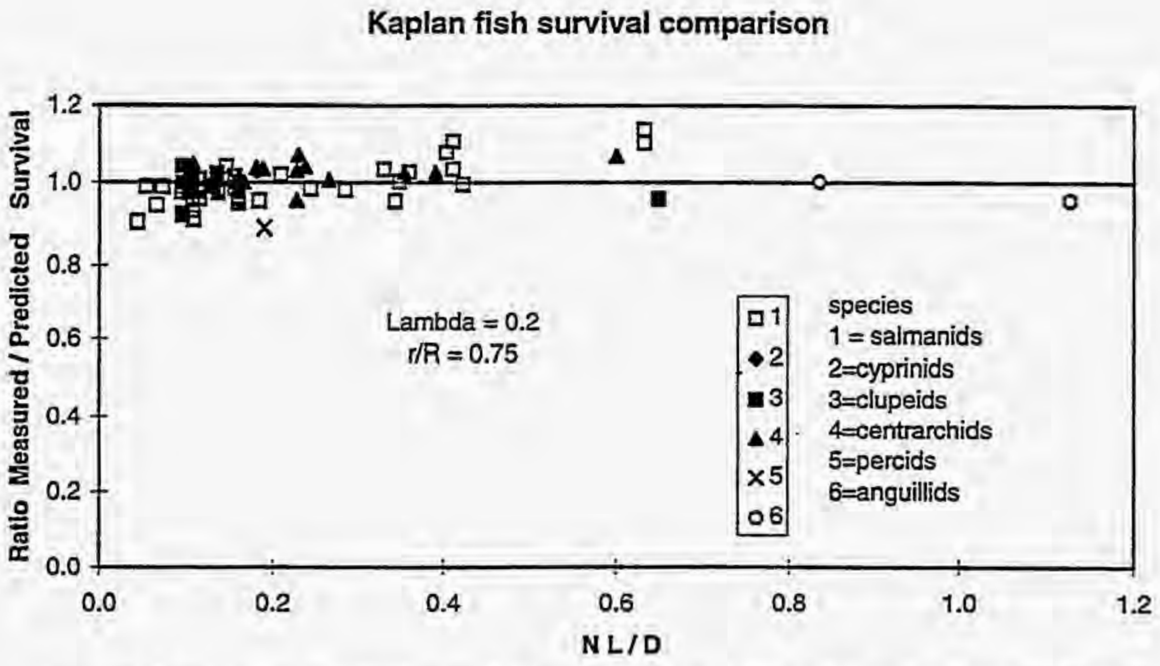


Figure 4.4.4-5 No Effect Of Species Is Observed For Fish Survival Through Kaplan Turbines

4.4.5 CLUES TO GAP RELATED INJURY

Summary

Fish passage tests at Rocky Reach provided data used to estimate fluid and mechanical mechanisms associated with gap related injury. At smaller blade tilts a gap exists between the blade trailing edge and the hub. For one blade tilt, a filler piece was installed to close this gap. When fish were injected to enter the runner blade zone near the hub, a minimum improvement in fish survival of approximately 4.4% with the trailing edge gap filled could be inferred from the reported data.

Discussion

Passage tests of a new Kaplan turbine runner at Rocky Reach Dam using chinook salmon smolts were conducted at a range of blade tilts providing power outputs between 60 MW and 100 MW. The new Kaplan runner design closed the leading edge hub gaps upstream of the blade rotational axis over the full range of blade angles, but conventional Kaplan blade gaps were present downstream of the spherical portion of the hub. Fish were injected into the turbine intake at locations 3 m (10 ft) and 9 m (30 ft) below the intake ceiling. The lower (88.8%) than expected survival rate of fish entrained at 3 m (10 ft) depth at 60 MW prompted exploration of potential causes and subsequent solutions. It was suspected that the fish released at 3 m (10 ft) depth may have been transported through the gaps between the blade inner edge and the hub downstream of the spherical portion of the hub at the turbine operating point tested. Therefore, an additional test using the same sized chinook salmon smolts was performed at a blade tilt corresponding to 60 MW, where a filler piece was constructed to temporarily fill in the gap between the hub and the blade, downstream of the spherical portion of the hub near the outlet edge for testing purposes. The primary purpose of these fish releases (for testing purposes) was to evaluate whether the injury rates were the same. Because no concurrent controls were released to adjust for the potential effects of tagging, handling, release, and recapture, survival probability was not reported (Normandeau Associates and Skalski 1996). However, the 48 h control survival rate of the same stock and size of chinook smolts used in the primary releases was slightly greater than 99%. Therefore, it may be reasonable to assume that the effects of tagging, handling, induction, and recapture may not change over a short time period. Similar observations have been made in other investigations lasting two to three weeks (RMC et al. 1994; Normandeau Associates et al. 1995, 1996a,b,c,d). The reported data suggest that if the rate of severe injury decreased then survival should have correspondingly increased. The injury rate declined from 5% with gaps to 2.8% with closed gaps. A minimal improvement in survival of about 4.4% compared to the unmodified turbine could be inferred from the reported data. The closure of the gap, and the minimization of its associated fluid and mechanical injury mechanisms, could be related to an increase of survival of approximately 4.4% in this zone.

4.4.6 INSIGHT FROM THE TESTS AT WANAPUM DAM

Summary

Tests at Wanapum Dam were designed, conducted and evaluated to shed light on injury mechanisms and how they changed with zone and the point of operation. The overall survival was correlated with the new leading edge strike equation. A value of 0.1 was estimated for the lambda coefficient of the strike equation. The effect of the point of operation on fish passage survival provided insight to the TAL relationships and the cavitation effects. Peak survival did not coincide with peak efficiency, but occurred at a discharge where the blade strike probabilities were low while the TAL were at a minimum and before cavitation began to be significant. Additional mortality was attributed to flow phenomena in the zone of geometry near the vicinity of the hub. There, a dramatic decrease in survival occurred for fish injected at 3 m (10 ft) below the intake ceiling compared to those injected at 9 m (30 ft) below the intake ceiling, and was attributed to gaps between the blade and the hub.

Discussion

For Unit 9 at the Wanapum Dam on the Columbia river, fish passage survival tests were conducted at the originally designed Kaplan turbines. The tests were sponsored by the Public Utility District No. 2 of Grant County as part of its on going efforts to improve fish passage survival at the project. The dam houses 10 vertical Kaplan turbines, each rated at 89.5 MW (120,000 HP) at 24.4 m (80 ft) net head. Rehabilitation of the existing turbines is underway to improve turbine performance and fish survival (Hron et al, 1997). Model testing of the original turbine design was conducted as part of the testing to develop an upgraded design. The tested Unit 9 has 5 adjustable runner blades, a speed of 85.7 rpm, and runner diameter of 7.2 m (285 inches). A systematic variation of turbine discharge (and therefore, efficiency, blade and wicket gate gap geometry and cavitating conditions) and fish injection location was performed. Four blade positions were selected for evaluation covering a range of discharge from below the peak efficiency to nearly maximum output; 250, 310, 425, and 480 cms (9,000, 11,000, 15,000, and 17,000 cfs), Figure 4.4.6-1. Two fish injection locations were used, 3 m (10 ft) and 10 m (30 ft) from the intake ceiling. Based on CFD analyses, Figures 4.4.6-2 through 4.4.6-5, these heights in the intake are believed to transport fish to the runner entrance edge at 52% and 75% of the blade maximum radius. No information regarding possible fish volitional movement or dynamic effects on three-dimensional fish shaped bodies was simulated in the CFD analyses, so these locations must be considered as tentative. The 10 ft location fish are assumed to pass through an annular zone relatively near the hub and therefore in the vicinity of the hub gaps, while the 30 ft location fish are assumed to pass through the blades in an annular zone near the middle of the blade, Figure 4.4.6-6.

In addition to the fish passage survival testing, scale model test measurements and visual observations of a 1:20 scale model operated at the equivalent conditions to the 9,000, 11,000, 15,000, and 17,000 cfs discharges were made. These observations were used to document turbine efficiency and define cavitation patterns associated with gap leakage and blade surface cavitation at the highest discharge. To complement the model and field testing, CFD analyses of selected turbine geometries were made.

The geometry of the 5 bladed turbines is such that the blades have a leading edge hub to blade gap upstream of the spherical portion of the hub, Figure 4.4.6-7. A special three-dimensional localized contour on the hub exists downstream of the spherical portion of the hub to minimize the gap between the blade inner edge and the hub, Figures 4.4.6-8a and b. Along the hub surface at low blade tilts (Figure 4.4.6-8a) and especially at high blade tilts (Figure 4.4.6-8b), the special hub contour is oriented in a way that it creates a flow distortion as water passes along the hub surface. At the blade tip, a gap exists between the runner blade and the discharge ring upstream of the runner centerline because the discharge ring is machined cylindrically upstream of the blade centerline to allow removal of the runner. The wicket gate geometry is such that the gates overhang the discharge ring at higher gate openings. Characteristics of the Unit 9 turbine gaps as a function of discharge are shown in Figure 4.4.6-9.

A sequence of reasonable assumptions and data comparisons were made to draw conclusions about sources of mortality that refine the prediction methods.

- The mortality of fish at the 30 ft location (assumed to pass the blade in the mid blade annular zone centered around r/R=0.75) is assumed to be due to leading edge strike and TAL. This assumption was made because the runner gaps and the special three-dimensional localized contour on the hub are relatively far from the middle of the blade, and no cavitation was present in the middle of the blade according to model test observations.
- The gaps between the blade and the hub are minimum at the maximum blade tilt tested (17,000 cfs). Therefore, fish mortality for the 10 ft location at the maximum flow rate includes no contribution from gaps.
- Some cavitation was observed in the model on the blade near the hub at the maximum tilt tested (17,000 cfs). Figure 4.4.6-10
- There is an entrance edge gap. There is essentially no blade to hub gap downstream of the spherical portion of the hub due to a special three-dimensional localized contour on the hub.
- At flows above 15000 cfs there is a wicket gate overhang gap.
- At all flows there is a blade tip to discharge ring gap at the blade leading edge.
- The mortality attributed to the flow conditions in the annular zone along the hub, including the entrance edge gap, was 3% higher than the mortality in the mid blade zone.

30 ft location (mid blade zone) Figure 4.4.6-11 presents the measured survival data with uncertainty bands and a prediction of survival based on the strike equation with a lambda value of 0.2. Using the leading edge strike equation with a 0.2 factor for lambda would indicate that virtually all of the mortality for each test point at the 30 ft location would be attributed to strike and at the 15000 cfs condition, survival is higher than the strike equation would indicate is possible. Also, no mortality would be due to any other source. As this seems unrealistic, a value of lambda equal to 0.1 was arbitrarily chosen for the strike equation and leading edge strike mortality was recalculated. This choice of a value of lambda equal to 0.1 allows other damage mechanisms to exist and gives a survival prediction for the 15000 cfs condition that is within the experimental uncertainty band. It is recognized that future tests can be expected to refine this estimate. In order to assign damage mechanisms to the remaining mortality, several damage mechanisms were evaluated specifically for this location:

⇒ Gap grinding effects are not believed to exist because the fish are assumed to enter the runner near the mid blade zone.
⇒ Cavitation effects are not believed to exist because no cavitation was observed at the three lower discharges and for the higher discharge, no cavitation occurred on the mid blade region.
⇒ Pressure reduction effects are not believed to be significant. At the 15,000 cfs test condition no mortality was attributed to probable pressure related changes (Normandeau Associates et al. 1996c). The pressure reduction effect would be the same for all test points, if it is based on the acclimation pressure and the final pressure.

The remainder of the mortality for the mid blade position were assumed to be related to TAL. TAL related losses are in a sense a one number characterization of the fluid energy losses which may cause damage to fish. They are not distributed spatially throughout the machine as a function of location (stay vanes, runner, draft tube) or blade radius even though the fluid mechanisms related to the losses (shear, turbulence) are localized mechanisms. It is beyond the scope of the current work to be able to predict which percentage of the losses are of a characteristic which may cause mortality to fish and where those occur. Even though the magnitude of the losses associated to TAL are somewhat arbitrary, it can be seen from Figure 4.4.6-12 that the shape of the TAL assumed mortality characteristics as a function of discharge correspond to the calculated total TAL based on the observed model efficiency distribution. For a different choice of lambda, the magnitude of the TAL induced mortality would change, but the shape of the curves would be similar. The distribution of the estimated mortality sources is shown on Figure 4.4.6-13.

The above assessment related to pressure reduction related mortality contrasts with the field observations made during the fish passage testing. For the Wanapum study, the field attribution of pressure related mortality varied with operating condition. While some operating conditions had no mortality attributed to probable pressure related causes, overall, 23% of the injuries were attributed to pressure related causes. However, a lack of reliable quantifiable criteria for such mortality assignments appears to be a significant obstacle to obtaining an accurate mechanism causal assessment of the mortality.

10 ft location (near hub zone) Figure 4.4.6-14 presents the measured survival data with uncertainty bands. The leading edge strike equation with a 0.1 factor for lambda was used with the radius ratio ($r/R = 0.52$) for the 10 ft location to calculate the portion of mortality due to strike. The same TAL induced mortality from the 30 ft location was assigned to the 10 ft data. The remaining mortality is assigned to specific sources based on experimental guidance, estimation, or the process of elimination. Hub gap mortality was assigned as 0% at the maximum tilt (17,000 cfs) and 2.5% at minimum blade tilt (9,000 cfs). The value of 2.5% was chosen based on consideration of the Rocky Reach study of blade trailing edge gap mortality in Section 4.4.5. The hub gap related mortality versus discharge is distributed in a manner that assumes that some gap creates a leakage velocity jet and shear values high enough to cause mortality and that gap grinding also occurs. The sources of mortality that were assigned so far were based on results of the 30 ft location analysis, or on other specific tests. The assignment of the remaining mortality requires more arbitrary judgments. At the minimum tilt, the remaining unassigned mortality was assumed to be caused by a higher concentration of flow induced mortality arising from unfavorable geometry in the hub annular zone including the abrupt sharp edge disturbance between blades and under the runner hub. A flow calculation that was developed for a different purpose illustrates the potential for injury in the hub region. Figure 4.4.6-15 shows a vortex located under a runner hub. At larger tilts, and discharges, this fluid induced mortality was assigned an increasing value. At the maximum tilt, 3.3% mortality remained unassigned and was attributed to cavitation related effects both by the process of elimination and by guidance from model observations. Figure 4.4.6-10 shows the cavitation pattern observed during the model testing. Note that based on the CFD streamline tracing cavitation occurred in this flow zone. The resulting distribution of mortality is shown on Figure 4.4.6-16.

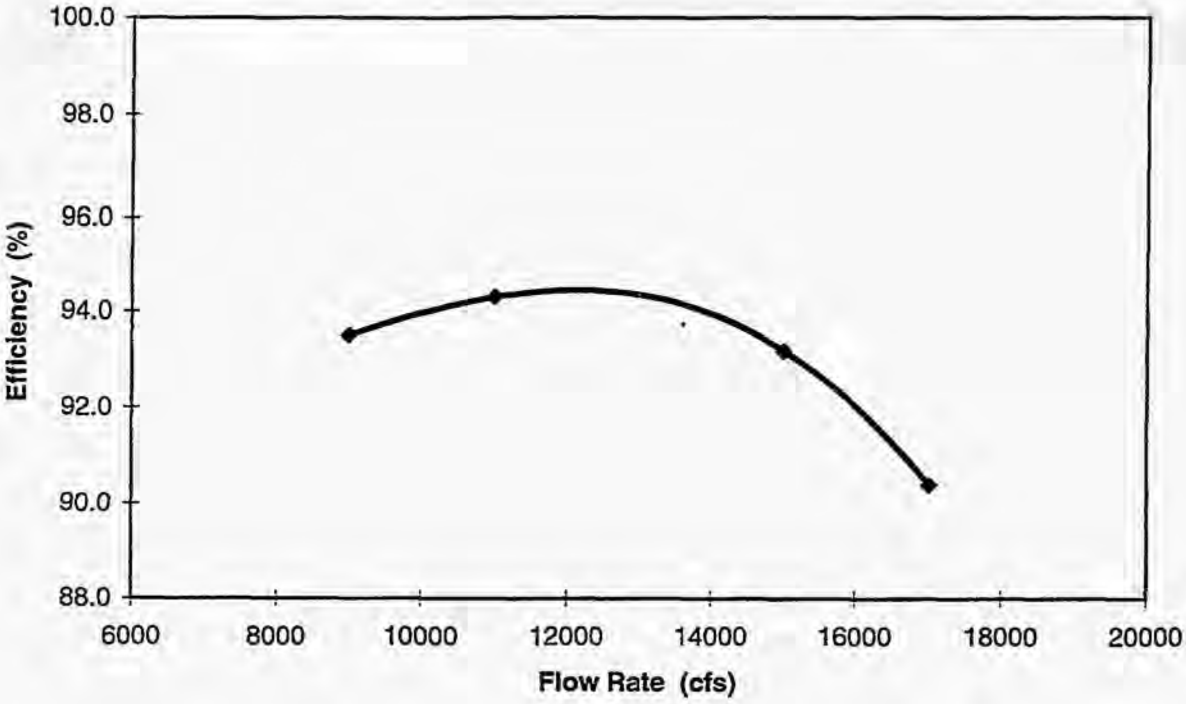


Figure 4.4.6-1 Fish Survival Test Points for Wanapum

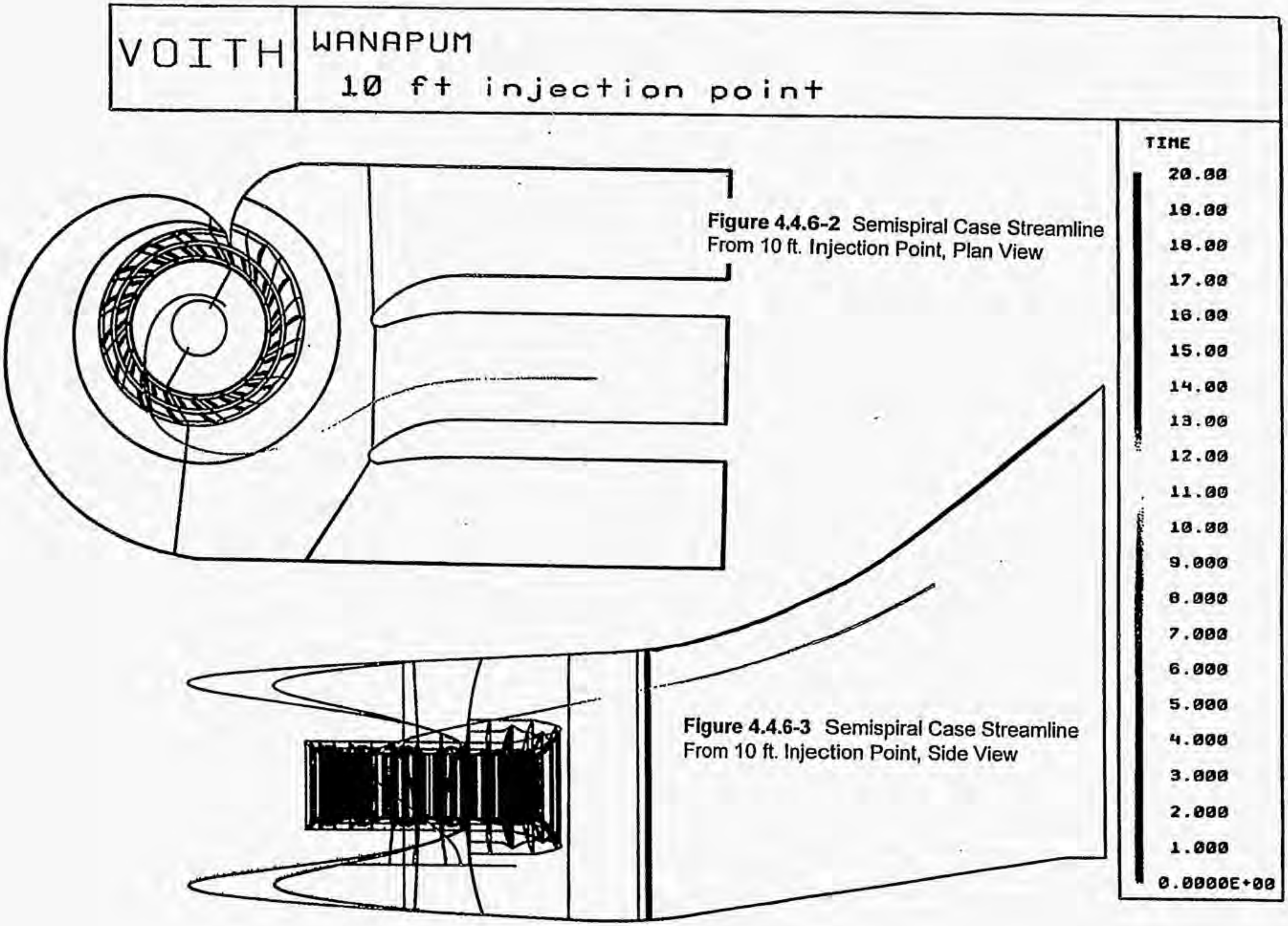


Figure 4.4.6-2 Semispiral Case Streamline From 10 ft. Injection Point, Plan View

Figure 4.4.6-3 Semispiral Case Streamline From 10 ft. Injection Point, Side View

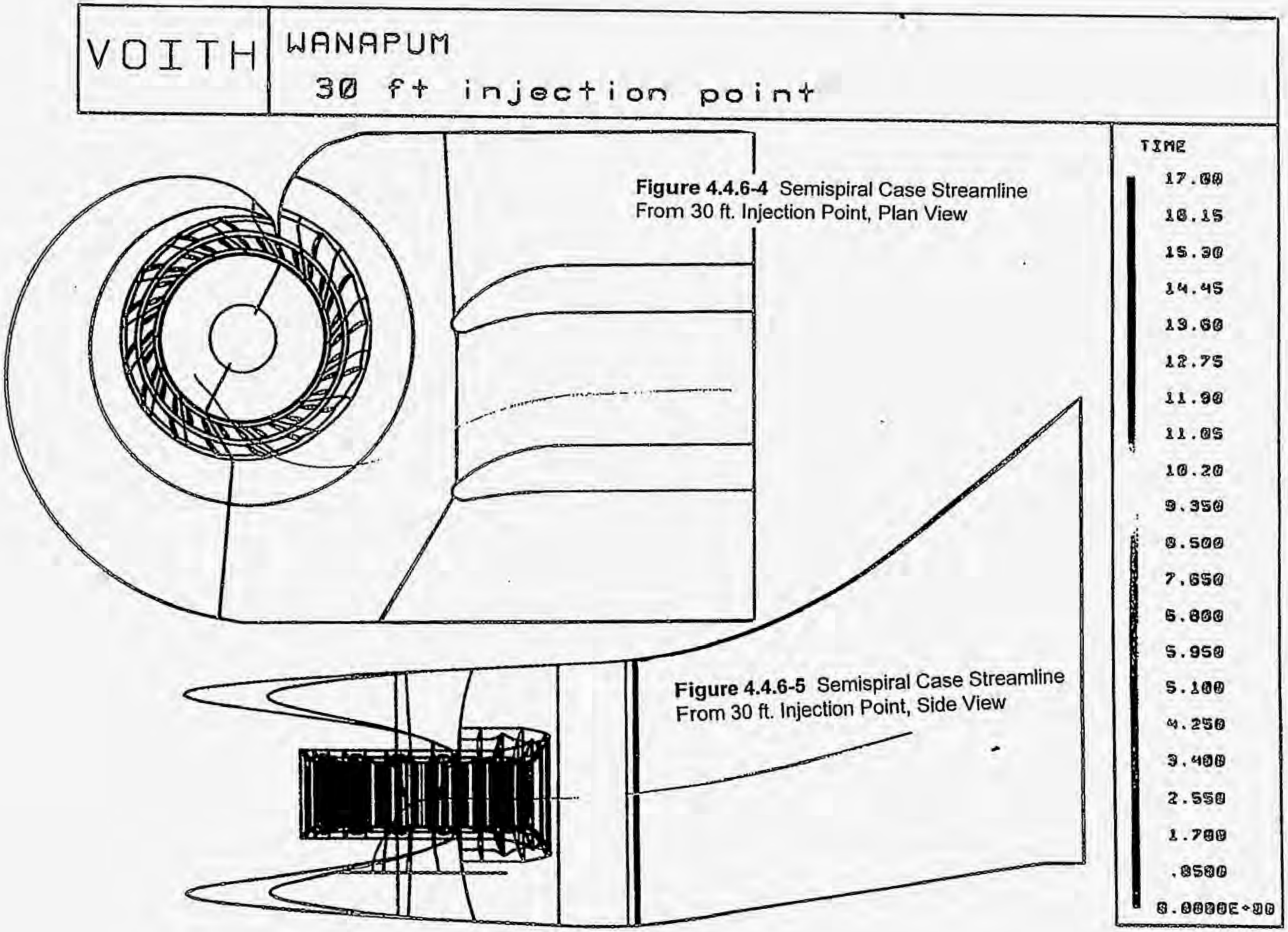


Figure 4.4.6-4 Semispiral Case Streamline From 30 ft. Injection Point, Plan View

Figure 4.4.6-5 Semispiral Case Streamline From 30 ft. Injection Point, Side View

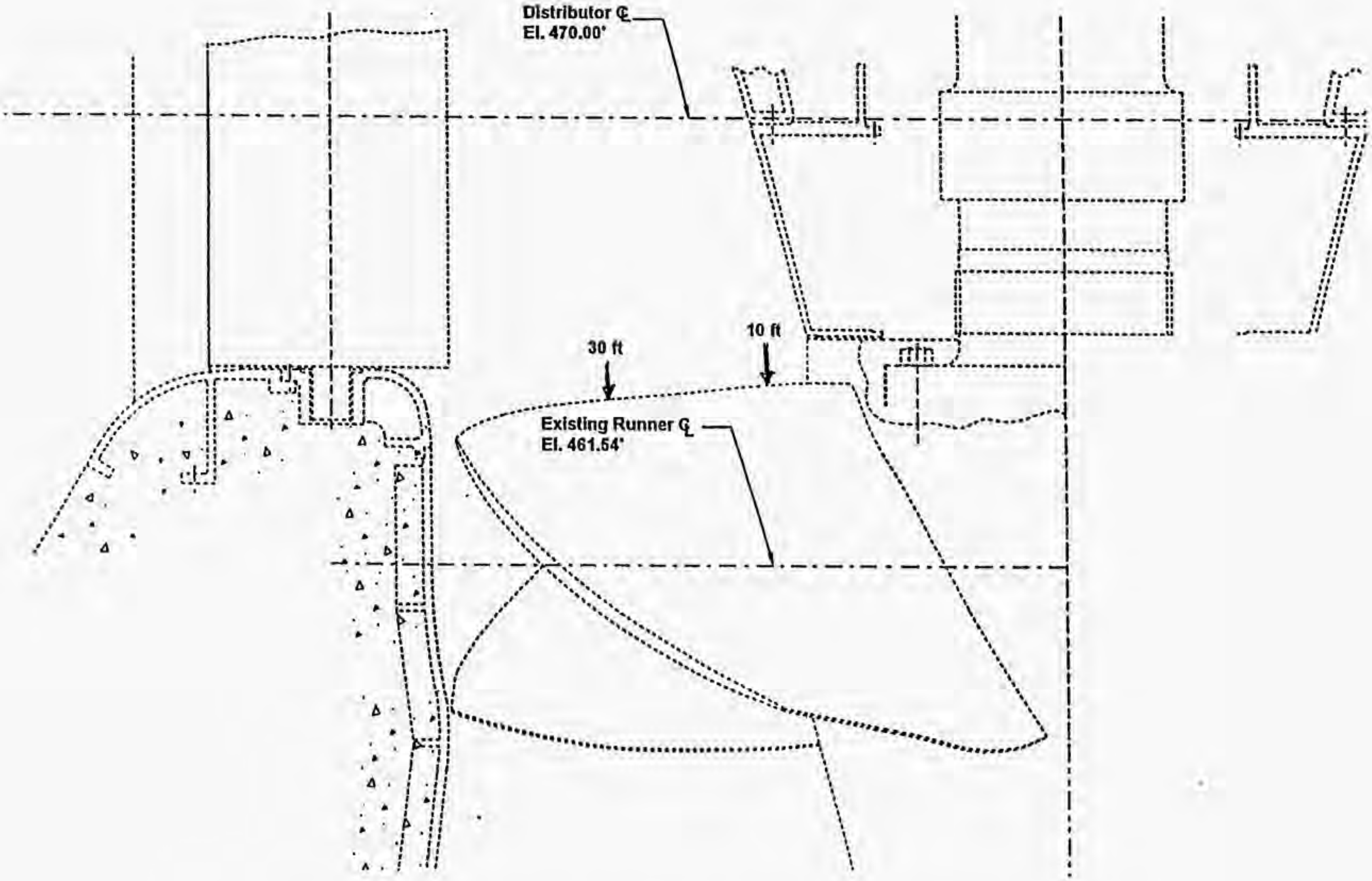


Figure 4.4.6-6 Expected Fish Location for 10 ft and 30 ft Injection Points at Wanapum

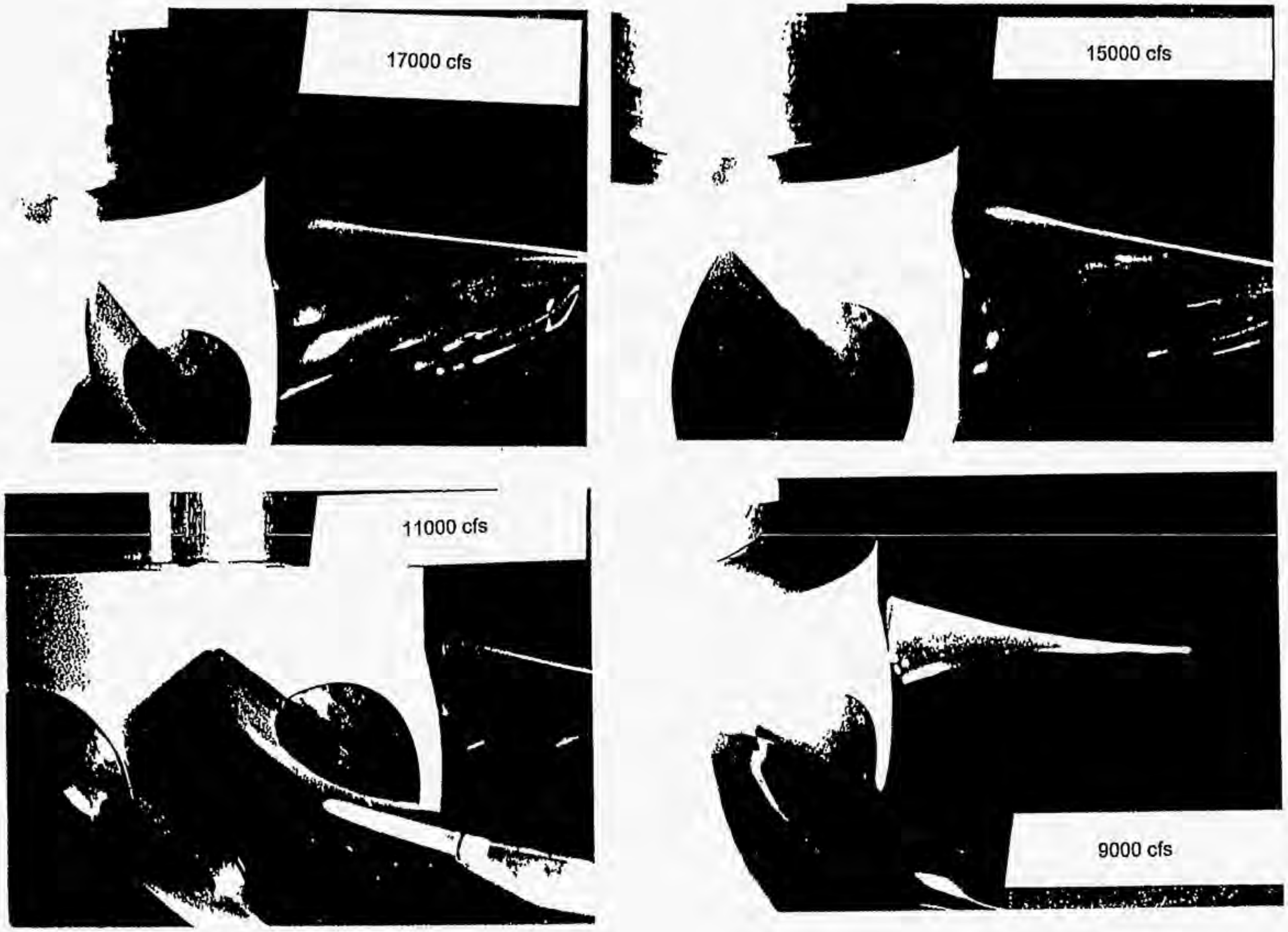


Figure 4.4.6-7 Wanapum Blade Entrance Edge Hub Gaps

Figure 4.4.6-8a Wanapum Special Hub Contour at Higher Blade Tilt

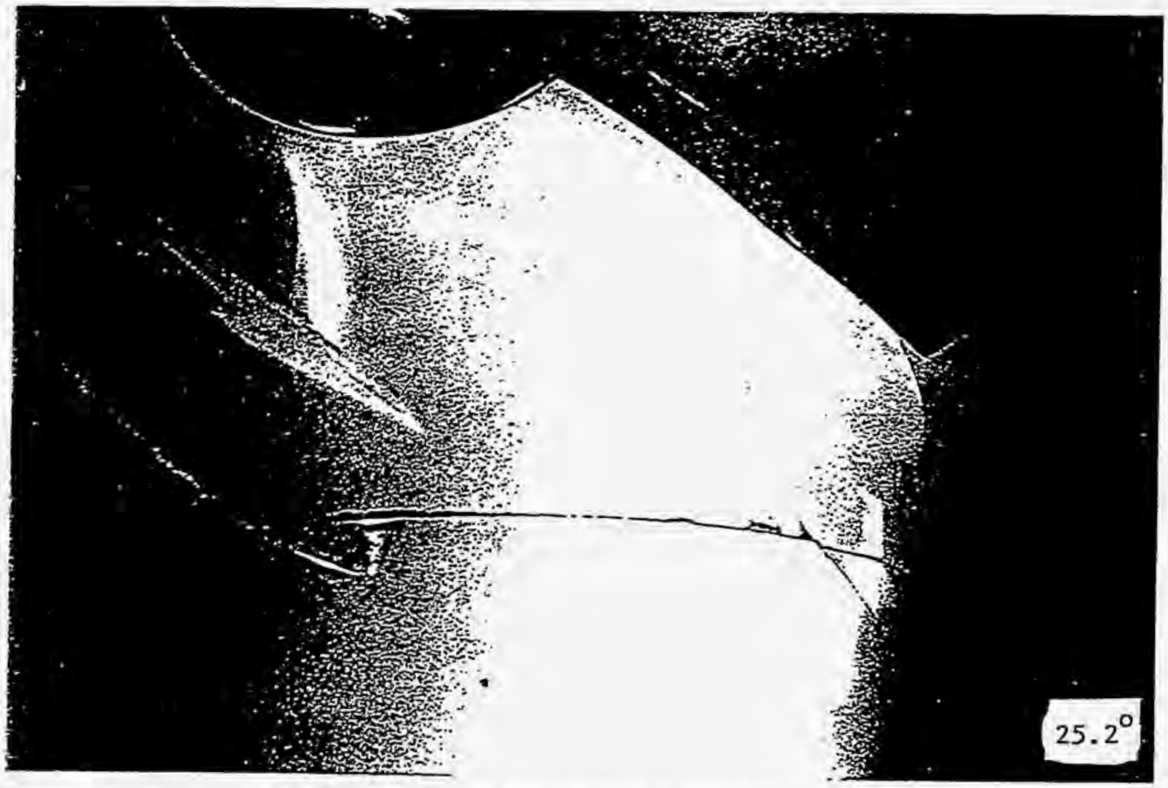


Figure 4.4.6-8b Wanapum Special Hub Contour at Lower Blade Tilt

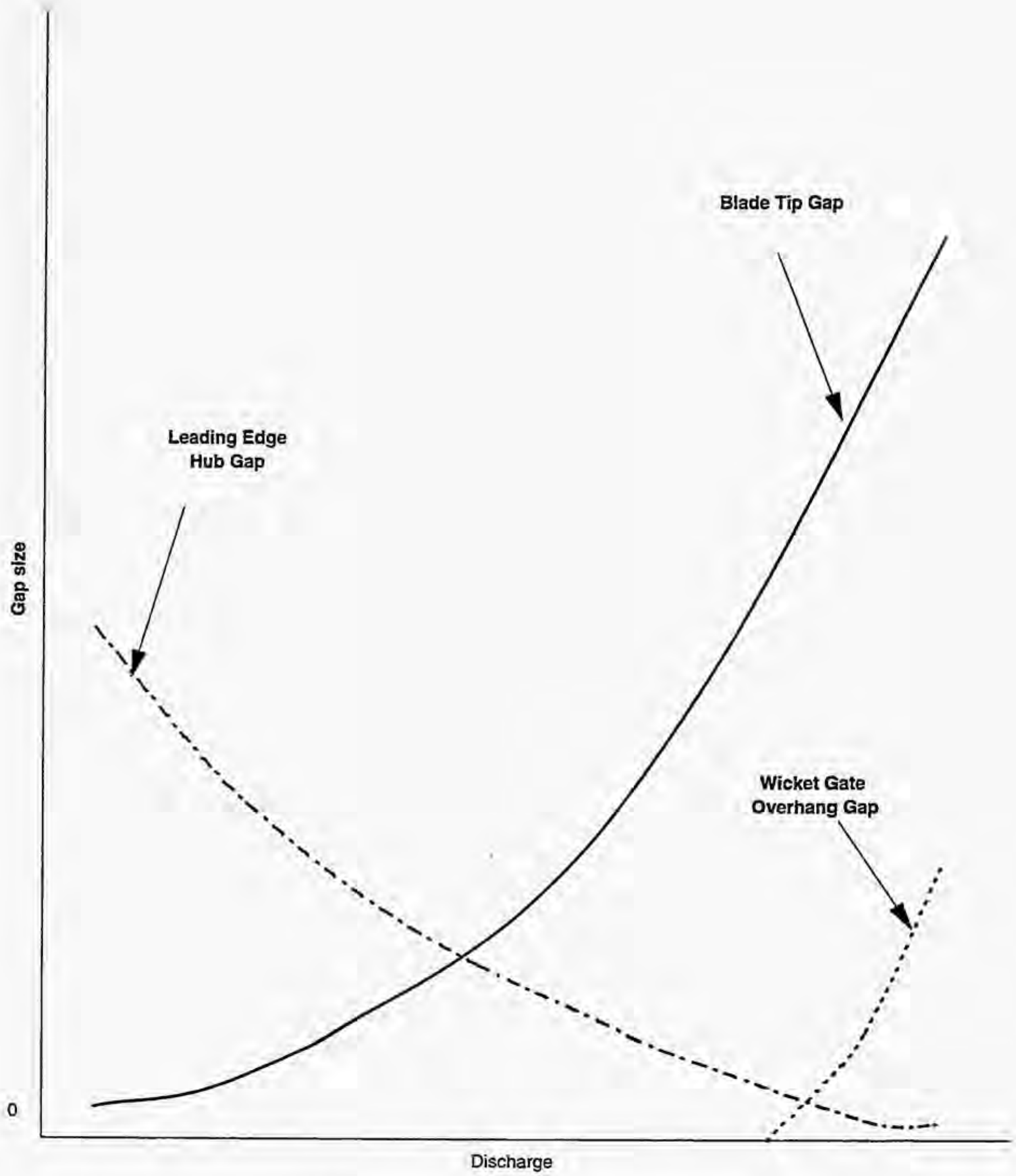
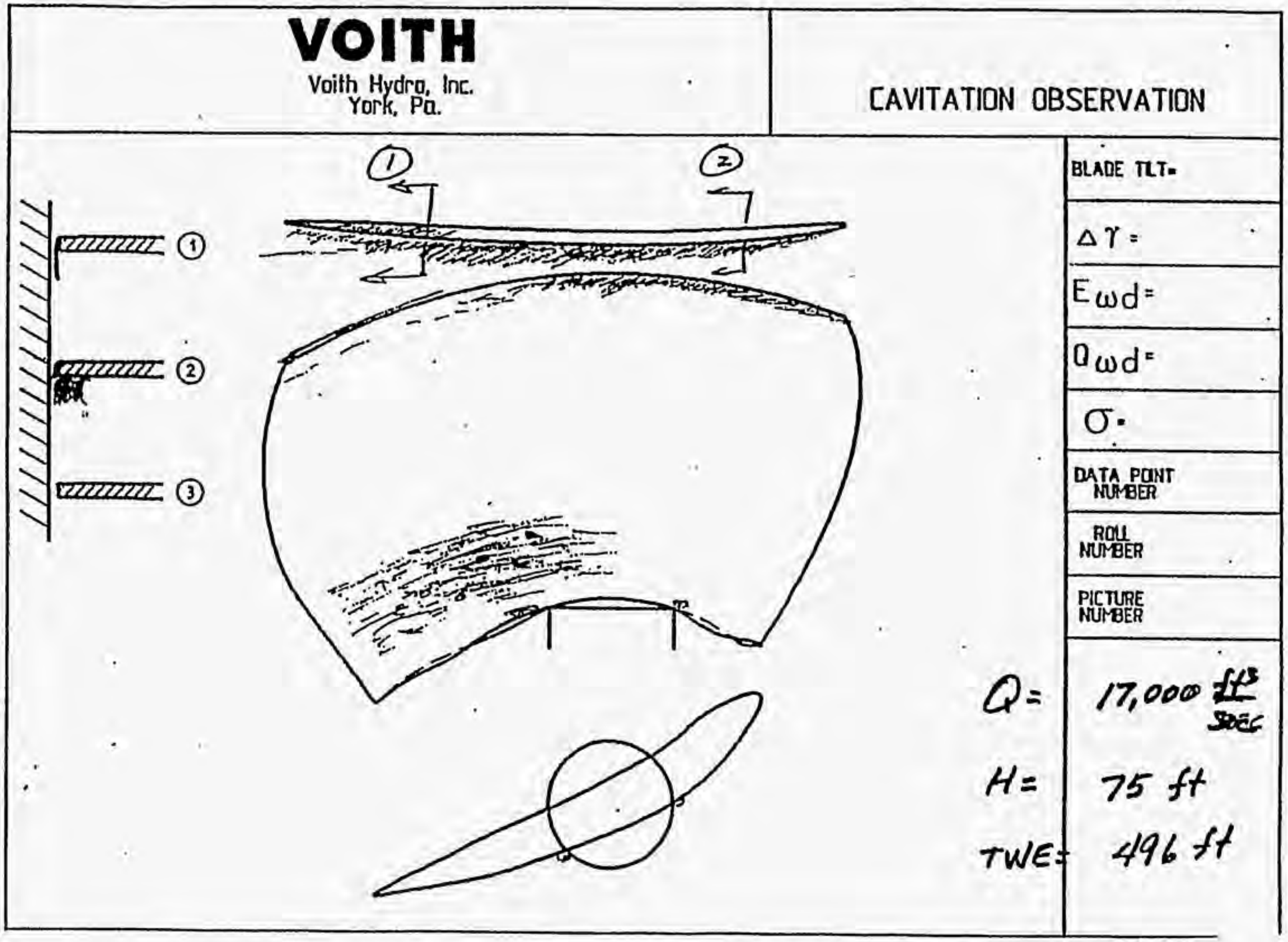


Figure 4.4.6-9 Tendencies for Blade Gaps and Wicket Gate Overhang



VOITH
Voith Hydro, Inc.
York, Pa.

CAVITATION OBSERVATION

BLADE TLT=

Δγ =

Eωd=

Qωd=

σ=

DATA POINT NUMBER

ROLL NUMBER

PICTURE NUMBER

Q= 17,000 ft³/sec

H= 75 ft

TWE= 496 ft

Figure 4.4.6-10 Wanapum Cavitation Sketch at 17,000 CFS

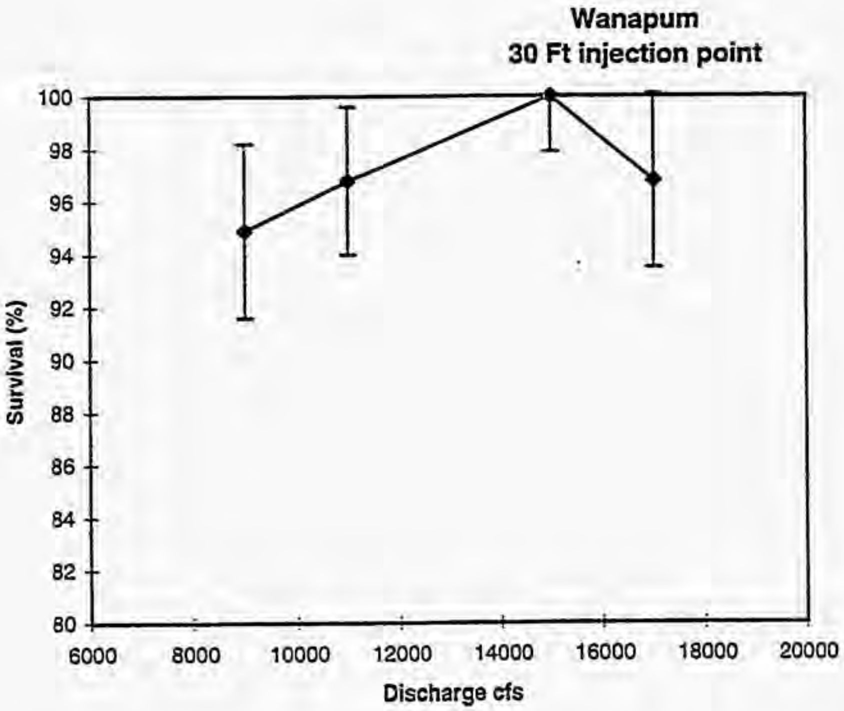


Figure 4.4.6-11 Wanapum Measured Survival For 30 Ft. Injection Location

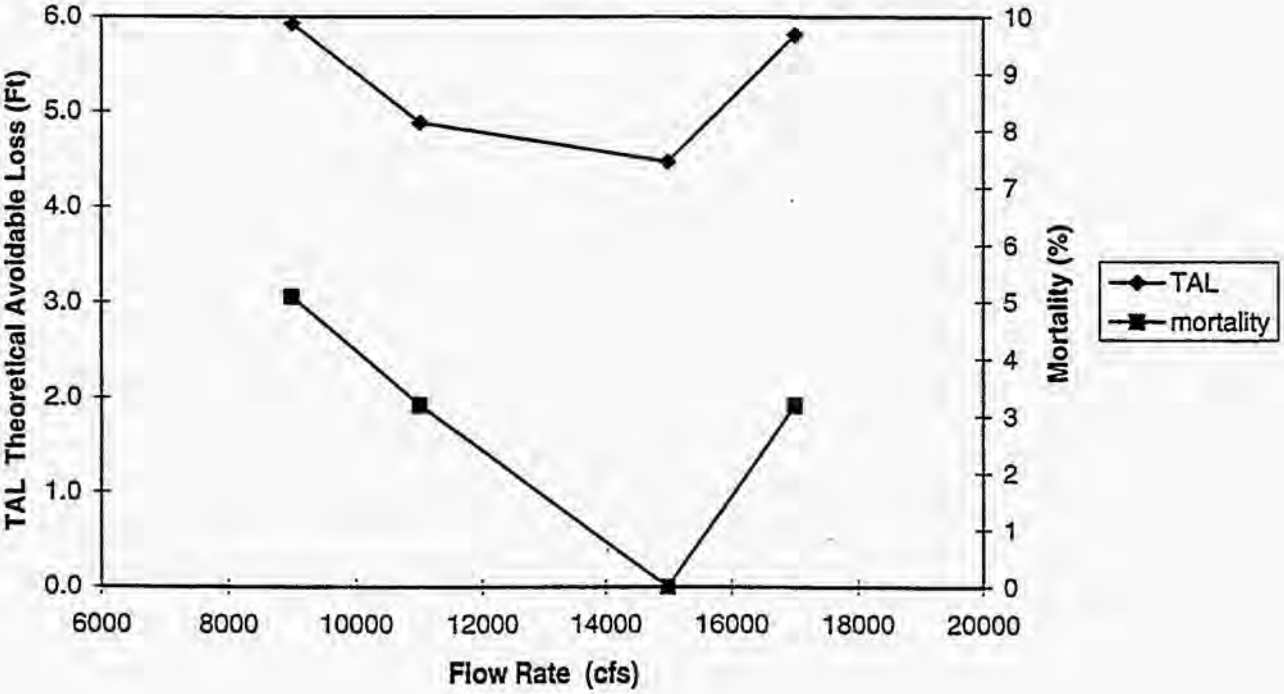


Figure 4.4.6-12 Mortality due to TAL has similar shape to TAL

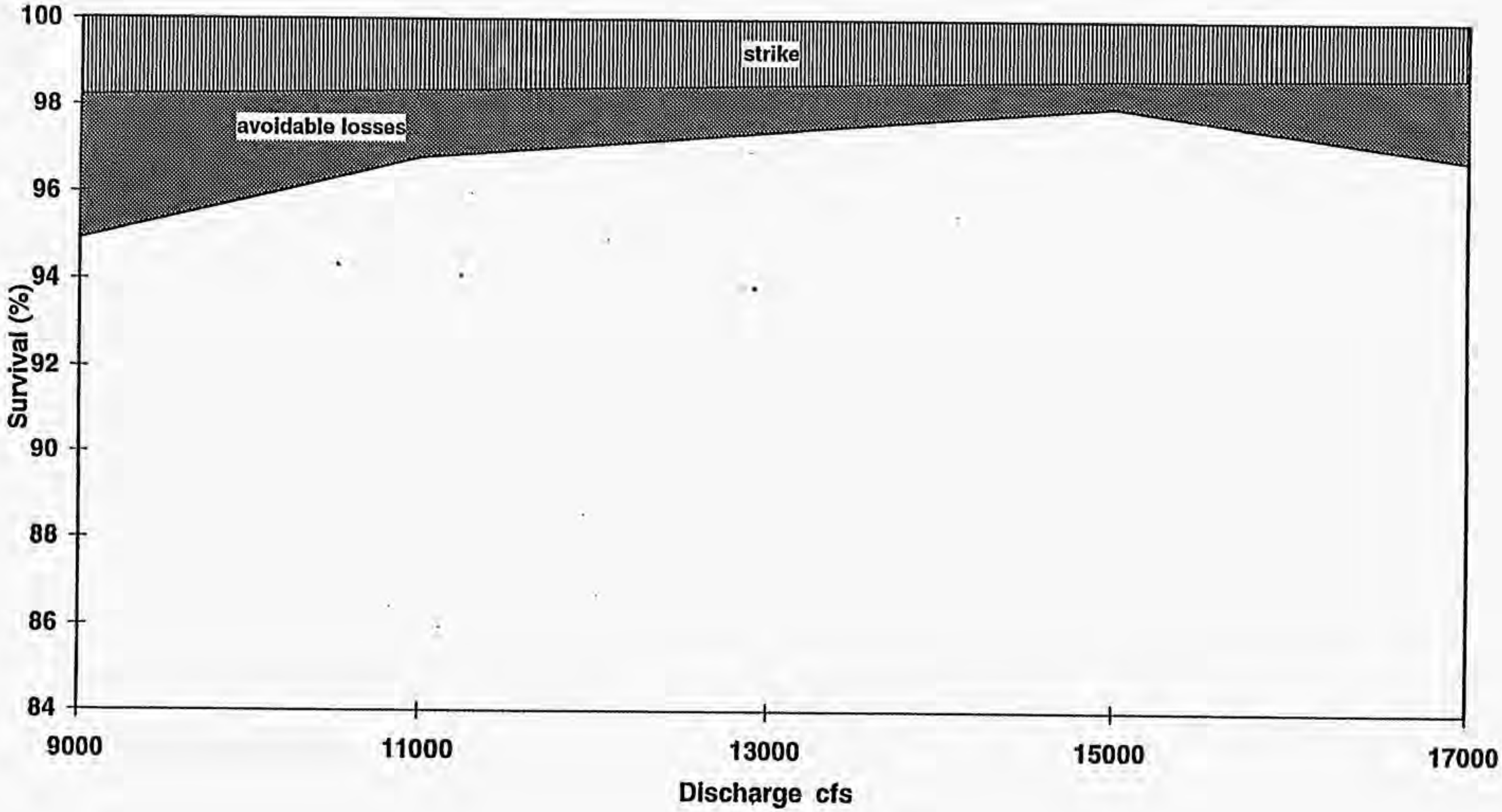


Figure 4.4.6-13 Distribution of Mortality Sources for 30 ft. Injection Location

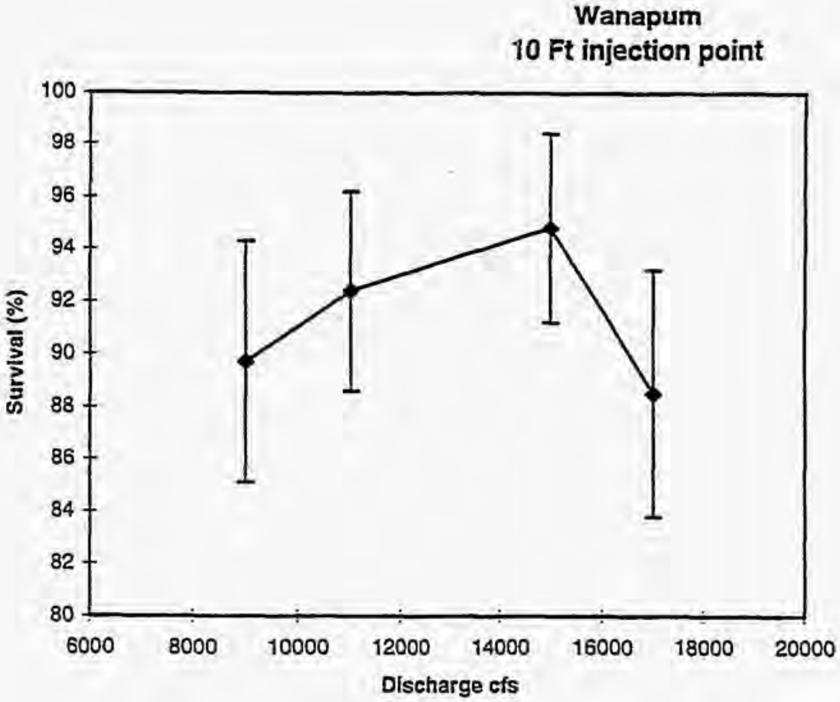


Figure 4.4.6-14 Wanapum Measured Survival For 10 Ft. Injection Location

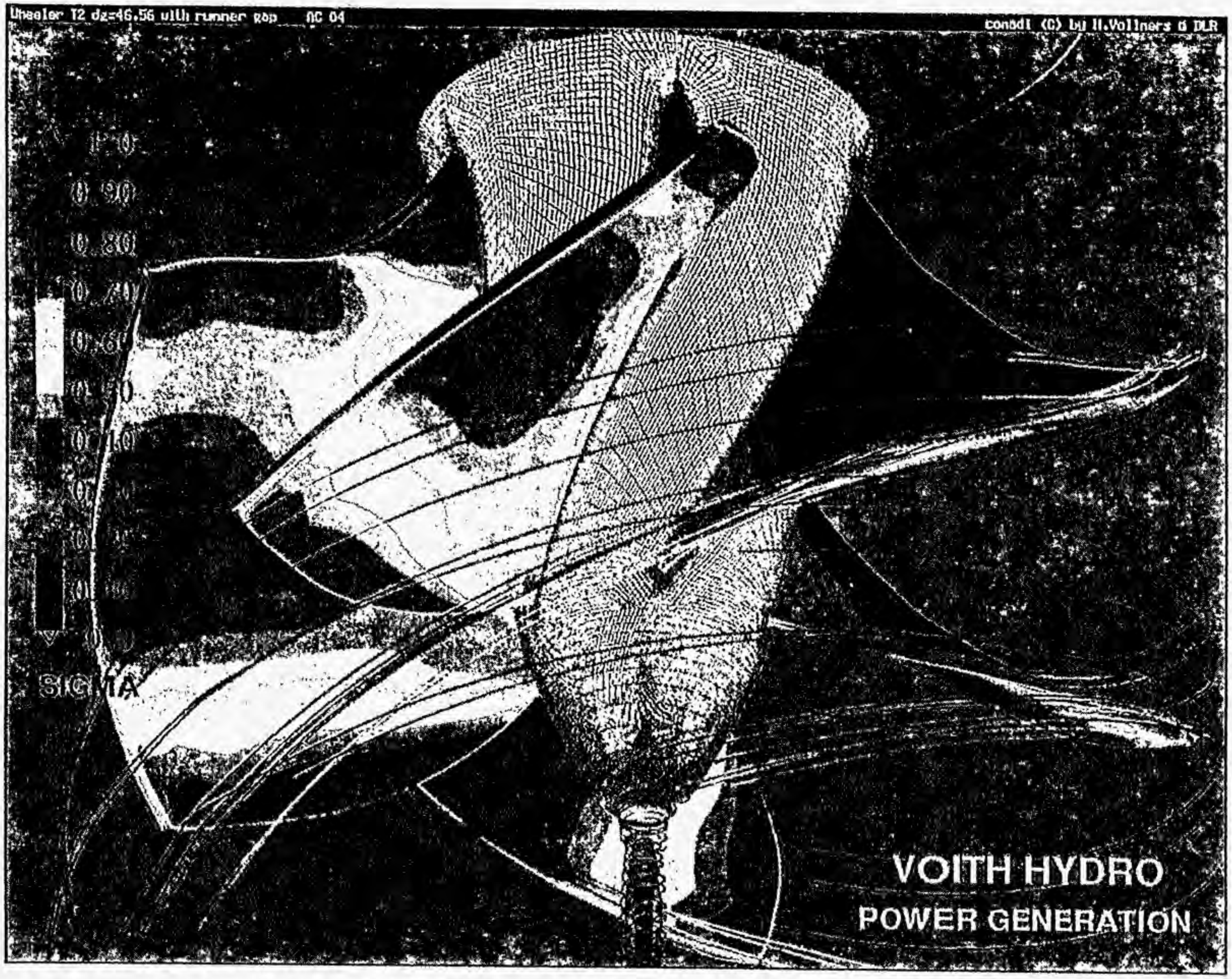


Figure 4.4.6-15 A Vortex Typically Exists Beneath A Runner Hub

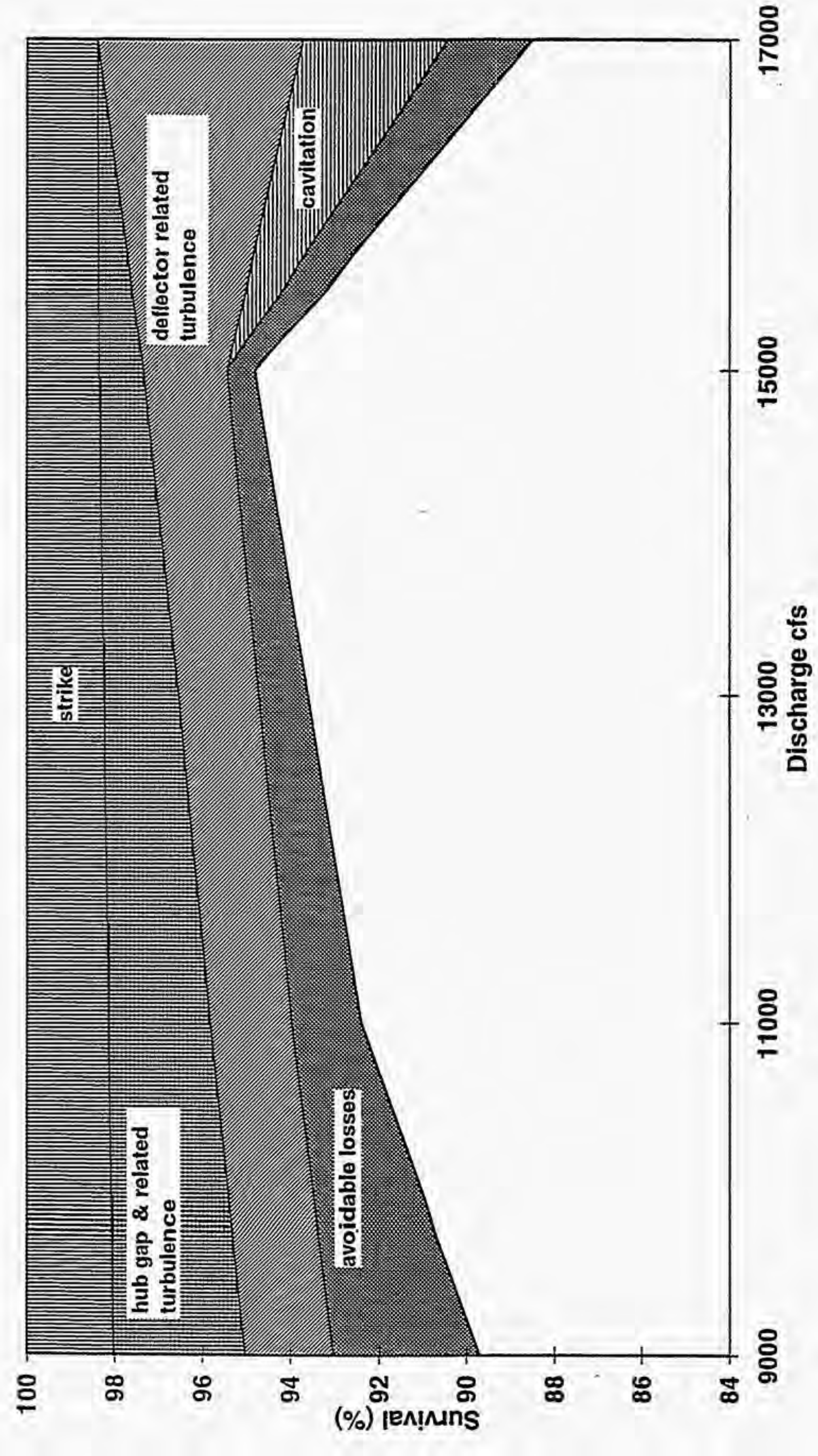


Figure 4.4.6-16 Distribution of Mortality Sources for 10 ft. Injection Location

4.4.7 KAPLAN TURBINE OPERATION FOR MAXIMUM FISH SURVIVAL

Summary

The proposition that operation at discharges within 1% of peak efficiency will maximize fish survival was examined. Historical data that had been generally believed to support this belief, from the Big Cliff and Foster Kaplan turbines was reanalyzed. This data does not show that maximum fish survival occurs at discharges within 1% of peak efficiency. Analysis of the complex factors involved reveal tendencies for fish survival that operate in opposing directions. That is, some factors maximize survival at high discharges while other factors maximize survival at low discharges. For the geometry of the Wanapum Dam turbines and for fish located predominately in the upper water column, fish survival was enhanced by operation at higher discharges where efficiency is more than 1% lower than maximum, where TAL is a minimum, the hub gaps are small, and no cavitation is present in the blade region.

Reexamination of the 1% Criteria

Three separate studies containing survival measurements over a range of discharges were evaluated.

The data from Oligher and Donaldson (1966) were reexamined. A series of survival tests were conducted at three heads at Big Cliff Dam on the North Santiam River, Oregon. Although uncertainty estimates were not provided, the description of the experimental protocols used compare favorably with those used today. Plots of all their data are reproduced in Figure 4.4.7-1. The general trend that maximum fish survival occurred in the area of highest operating efficiency, concluded by the authors, does not hold. In fact, in some cases survival increased at turbine discharges greater than the best efficiency point.

At Foster Dam, a series of tests were conducted to evaluate the effects of turbine operating efficiency on fish survival (Figure 4.4.7-2, adapted from Bell, 1991). Tests were conducted for on cam and off cam operation. Only the on cam data are presented here. Generally, no trend is evident except at the lowest head tested where survival was a maximum at a discharge less than the peak efficiency.

Data from the tests at Wanapum were discussed in detail in Section 4.4.6., Figures 4.4.6-11 and 4.4.6-14 show the measured survival, and Figure 4.4.6-1, shows the turbine efficiency. Maximum fish survival occurs at a discharge greater than those discharges having efficiencies less than 1% below the maximum efficiency.

Current Understanding of Operation To Maximize Survival

Since factors affecting fish survival do not uniformly increase or decrease with changes in turbine operation (Figure 4.4.7-3) each factor is briefly reviewed, and the consequence for fish survival is summarized.

The strike probability equation for a Kaplan turbine as presented in Figure 4.3.2-7 shows that the probability of strike and also blade zone encounter is lowest at highest discharges and also varies somewhat as a function of head. Generally, lower strike and BZE probabilities occur at higher heads.

The injury effects of gaps between the blade and the hub, both fluid induced and mechanically induced, are related to the size of the gap. This gap size varies with operating point and the turbine design and is minimized at higher discharges (higher blade tilts), and is at maximum at lowest discharges. Therefore, survival due to hub gaps is increased at higher discharges. The actual variation of survival with the size of the gap has not been studied experimentally, thus requiring an assumption of survival as a function of gap size.

The injury effects of gaps between the blade and the periphery are presumed to cause fluid and mechanically induced damage to fish similar to hub gaps. However, fish have not been introduced in a controlled way in the vicinity of these gaps to verify survival mortality effects. The size of these gaps also varies with operating point, but in contrast to the hub gaps, is minimal at low discharges. Therefore, for fish passing through the runner near the periphery, survival is greater at lower discharges.

The location of fish in the water column has been shown to be important. For fish that do not traverse the runner in zones including the hub or periphery gaps, the gap related mortality tendencies do not apply. These fish may be more affected by strictly fluid induced effects such as TAL. In this case, survival may be enhanced by operation at the minimum TAL condition. TAL losses are a one dimensional representation of the energy available to injure fish. While the losses are zonal, the distribution of TAL to different zones is not possible with the algebraic calculation method employed in this study. Therefore, quantitative predictions are not made.

Cavitation bubbles typically do not appear for operating points near the best efficiency. As discharge increases, cavitation bubbles may appear, and with further increases in discharge, may grow to form larger regions. At heads larger or smaller than the head at the best efficiency point, cavitation may begin at lower discharges compared to the discharge at the best efficiency point. Operating conditions having cavitation are presumed to reduce survival, and should be avoided. A detailed analysis of cavitation patterns is required to make an assessment for a specific turbine and operating point.

Some turbines may also have somewhat uncommon features, such as the three-dimensional surfaces on the hub of the Wanapum turbines (Figures 4.4.6-8a and b). These surfaces function to essentially eliminate the gap between the hub and the inner edge of the blade downstream of the spherical portion of the hub, but also create rather sudden changes in the local contour of the water passage. Possible mortality effects of these surfaces is difficult to predict without the use of CFD and a "virtual fish".

All of these effects operate simultaneously. The aggregate of all effects may be examined through analysis of carefully planned fish survival tests designed to evaluate zonal survival, such as were conducted at Wanapum. There, it was found that fish survival was a maximum at discharges greater than the best efficiency discharge. The maximum fish survival occurred at an operating discharge that had an efficiency that was more than 1% lower than the best efficiency..

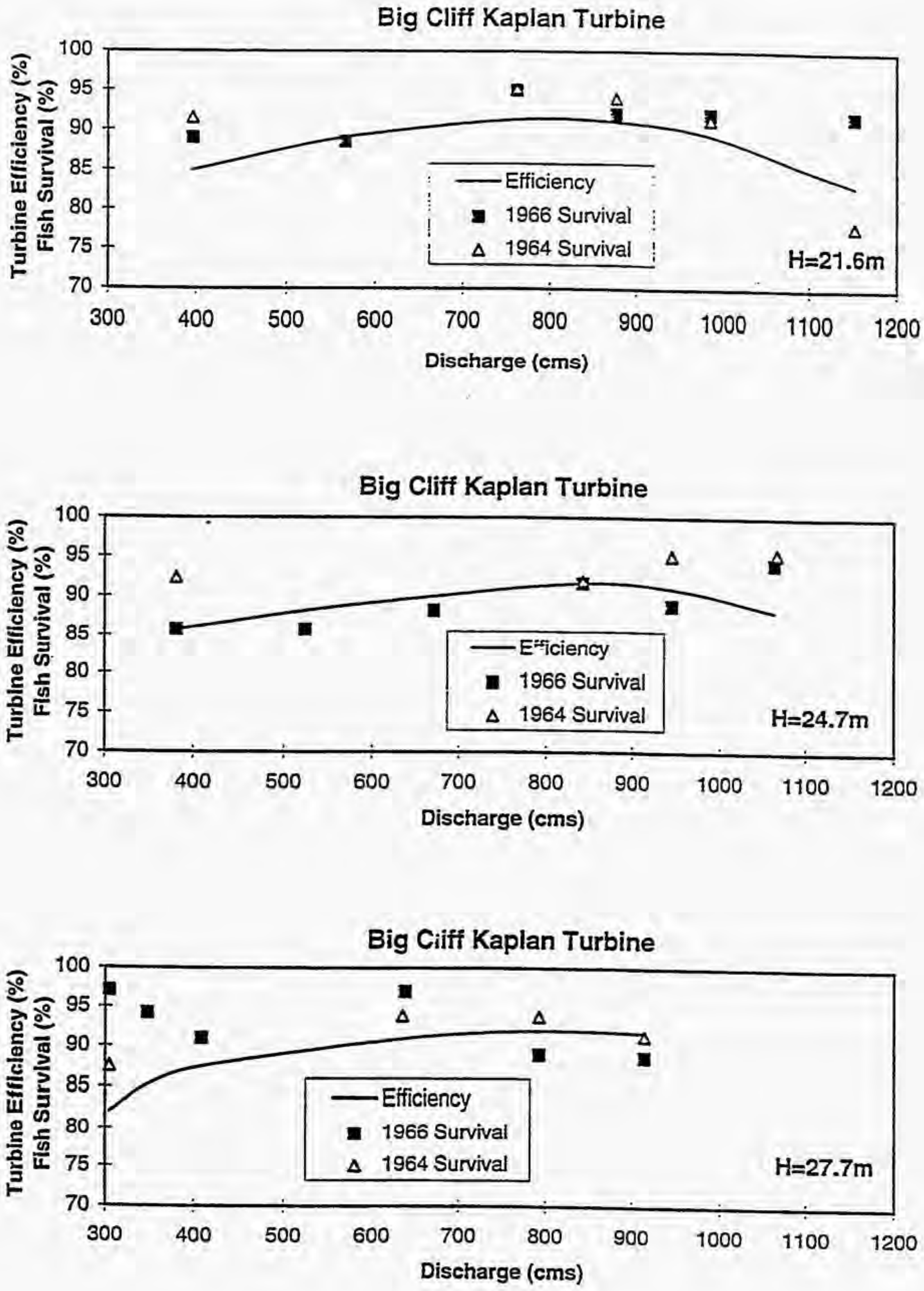


Figure 4.4.7-1 Survival and Efficiency at Big Cliff

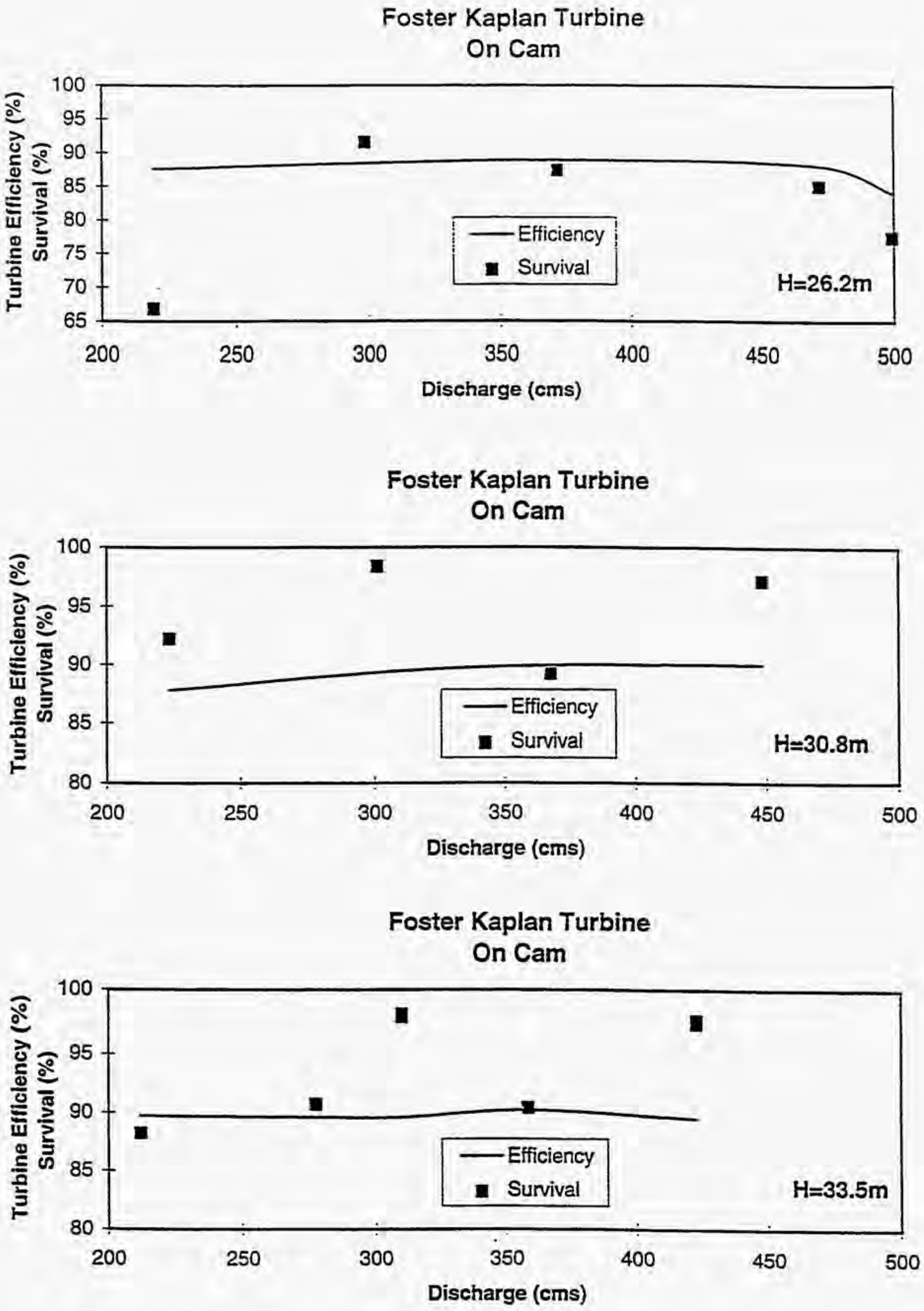


Figure 4.4.7-2 Survival and Efficiency at Foster, On Cam Data Only, From Bell, 1991

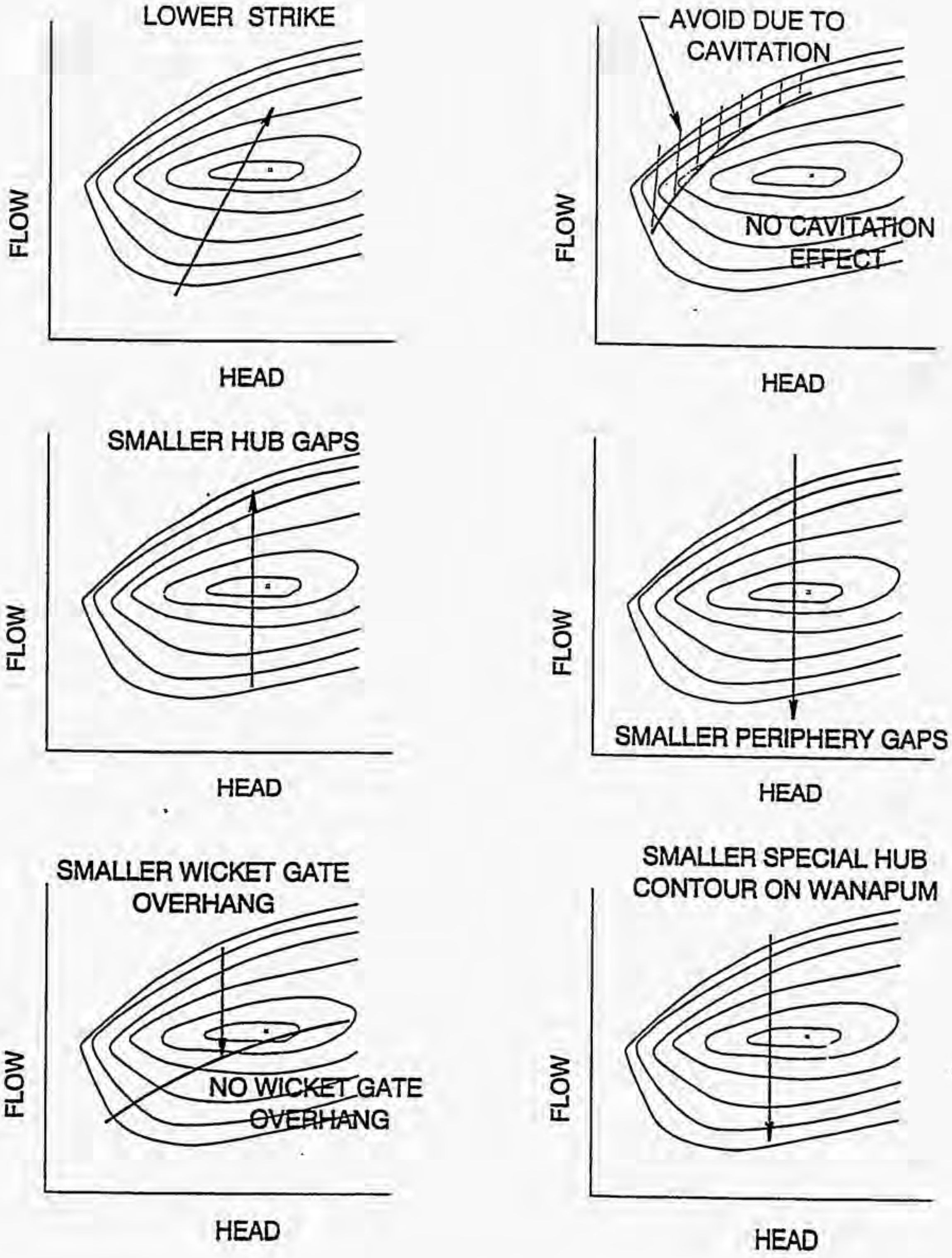


Figure 4.4.7-3 Mortality Tendencies for Several Factors in Kaplan Turbines

4.4.8 EFFECT OF FISH SCREENS AND FLOW DISTURBANCES ON FISH PATHS

Summary

Physical model testing and CFD analyses have investigated the impact of the presence of fish screens on velocity distributions in turbine intakes. Physical model tests indicate significant redistribution of flow takes place. Fish screen effectiveness testing has shown that the exclusion of fish entering the turbine is not 100%. Therefore, some portion of the unguided fish go under the fish screen either by being carried there by the water or by their own free will. Fish passing the intake under the fish screens may then find themselves in the lower portion of the water column where they are expected to pass through the lower zone on the wicket gates and near the outer radius zone of the blades. These fish will experience a different set of turbine geometry and associated fluid effects compared to fish passing through the upper portion of the wicket gates and the mid to inner regions of the blades. Compared to units without fish screens, the presence of fish screens may cause different fish survival characteristics.

Results from Lower Granite Dam show the importance of site-specific characteristics (i.e., whether intakes are equipped with screens or not), entrainment depth, point of operation of turbine, experimental protocols, and perils of extrapolating data from one site to another for the purposes of turbine rehabilitation. For tests at Lower Granite turbines, when equipped with extended length fish guidance screens, survival was highest for fish introduced at lower elevation when turbine operated towards its lower end of operating efficiency range. No significant differences in survival of fish introduced in the three intake bays at the same depth and turbine operation occurred. Survival in cavitation mode was similar to that at turbine operating modes. Injuries due to gaps between the runner and the hub were more common for fish introduced at upper elevation than at lower elevation.

Discussion

Despite efforts to exclude fish entry into turbines by installation of intake guidance screens or a surface bypass collector system some proportion of fish population remains unguided and is transported through the turbines. The entry of these fish into turbines most likely occurs differently than through turbines not equipped with protection devices. Thus, the development of "fish friendly" passage through these turbines needs to consider the altered hydraulic conditions, depth and trajectory of fish entrainment, fish distribution, etc. Limited field experiments have been conducted to offer some insights into potential sources of fish injury/mortality at turbines equipped with protective devices. Two separate tests were conducted at Lower Granite Dam turbine Unit 4, one in 1994 and the other in 1995.

The 1994 test was conducted to establish benchmarks of turbine passage fish survival prior to the proposed reservoir drawdown so that a pre- and post-drawdown comparison could be made. This test involved introducing fish at about 9 m (30 ft) below the turbine Unit 4 intake ceiling (Bay B) equipped with standard length screens; the turbine operated near the upper discharge limit of the within 1% of normal peak operating efficiency discharge range (about 513 cms or 18,000 cfs discharge). The 1995 test was conducted to evaluate the potential effects of (1) extended length screens installed at turbine Unit 4, (2) passage through different intake bays, (3) turbine operation, and (4) entrainment depth in one intake bay (A).

Although in Appendix Section 10.1 only immediate survival rates (1 h) from Lower Granite are presented for consistency with other studies the experimental protocols utilized at Lower Granite Dam were identical in 1994 and 1995 and 120 h survival estimates were made (RMC et al. 1994; Normandeau Associates et al. 1995). These data are discussed herein for drawing general conclusions for this site. The potential effect of extended length screen on fish survival was almost non-existent; in 1994 the 120 h survival was estimated at 93.4% and 94.0% in 1995. In both years, the turbine operated within the discharge limits set by a 1% drop in efficiency from the peak efficiency at the head of operation. However, the probable sources of fish injury differed between the two years. In 1994, 67% of the injuries were attributed to probable mechanical causes, 21% to shear and pressure, and remainder to multiple sources. In 1995, mechanically caused injuries accounted for 40%, shear and pressure 40%, and the remainder to multiple sources. Again, we emphasize that these were field determinations of probable causes of injury mechanisms and some uncertainty exists in exact classifications.

Tests to determine the impact of turbine operation were conducted at 385 cms or 13,500 cfs (lower discharge limit of the 1% down from peak efficiency at the head, at 513 cms (18,000 cfs), the upper discharge limit, and at 541 cms (19,000 cfs),.beyond the upper discharge limit, off cam and above the discharge for the beginning of cavitation. The effects of the point of turbine operation on fish survival were evident. The highest survival (97.2%) was estimated at lowest turbine operating efficiency (within ±1% towards the lower end of efficiency, discharge of 385 cms or 13,500 cfs) for fish introduced in intake bay A at 3 m (10 ft) below the extended length screens. The estimated survival at a discharge of 19,000 cfs was 94.1% and comparable to that estimated (93.6%) at the discharge of 513 cms (18,000 cfs).

The question remains; where did the fish go and into which zones of the turbine were they transported. What mechanical and fluid mechanisms for fish injury were associated with the zones transited by the fish, and what survival models could be associated with the transit. To shed light on this issue, a closer look at the Lower Granite Dam Unit 4 data are discussed below.

The presence of fish screens alters the velocity field in the intake (Figure 4.2-2). The velocity field alteration results in fish passing the screen being transported to deeper portions of the intake. CFD studies of a similar Kaplan intake with a screen are presented in Section 5.3.2. From these studies it can be seen that water from the lower region of the intake under the screens is transported to the lower portion of the stay vanes and wicket gates and then into the runner blades at the mid to outer radius of the blades. In this region, fish near the lowest region of the wicket gates are in a fluid environment which is affected by the wicket gate overhang gaps. In the runner blades, fish transiting the zone near the outer radius experience fluid and mechanical injury mechanisms which are strongly influenced by the blade tip gaps. Lower Granite turbines have a design where the upper portion of the discharge ring is cylindrical. At small blade tilts, the gaps are at a minimum. At larger tilts associated with higher discharges, the gaps are large and the gap flow characteristics provide strong vortices, high turbulence, shear and leakage cavitation. Mechanically, the gap shapes are sharp edged and tapered providing for a high potential for gap grinding. At high discharges, the wicket gates overhang the discharge ring and give rise to a leakage vortex.

To examine the effects of the above fluid and mechanical mechanisms, an analysis of the observed fish mortality was done. Fish injection was conducted at a location approximately 3 m (10 ft) below the bottom of the extended length screen but downstream of the screen. Velocity patterns are unknown here but would affect the fish transport path. Whether or not fish injected into the flow would be transported near the lower portion of the stay vanes and gates is unknown. The experimental results are shown in Figure 4.4.1-1. Test results including those from injection in three bays and at two elevations in bay A as well as 1 hour survival and 120 hour survival are presented. Maximum survival occurred at the minimum discharge. The survival trends for higher discharges is less clear. Consideration of the overlapping nature of the estimated uncertainty bands leads to the conclusion that survival trends with discharge cannot be established. No support for any survival hypothesis is inferred from these data.

It can be concluded that experiments designed to specifically address a single issue are best able to advance the understanding of fish survival.

4.4.9 PROPELLER TURBINES

Summary

Although measured data are few, good correlation was achieved between the new leading edge strike equation with lambda = 0.1 and the measured survival data. This improved correlation compared to Kaplan turbines may be due to the absence of blade gaps and limitations of the dataset.

Discussion

Few data for propeller turbines exist. Table 4.2-2 summarizes the available data from three installations. Two turbines are conventional propeller designs, the units at Safe Harbor (which are sometimes referred to as a mixed flow design) were included in the propeller turbine analysis. This was based on a judgment that the runner experiences an inflow that is substantially axial rather than the radial inflow that was assumed to exist for the development of the Francis turbine strike equations. All of these survival tests were conducted at the best efficiency point. Therefore, the propeller performance characteristic is virtually the same as that of a Kaplan at this point, and the strike prediction equation is also the same as a Kaplan. An appropriate value of lambda may be different than for a typical Kaplan turbine. Considering that a propeller turbine has no hub or periphery gaps, a value of lambda lower than 0.2 may be expected. A value of lambda equal to 0.1 was inferred for the strike only portion of Kaplan mortality. Because of sources of mortality in addition to strike lambda would be expected to be larger than a strike only value. Although little cavitation is likely to exist at the best efficiency point, some TAL is expected, and some deflector vortex rope phenomena may occur, a value of lambda may be greater than 0.1. Without additional data, a value for lambda could not be further refined, so a value of lambda equal to 0.1 was used and all fish were presumed to enter the runner at r/R = 0.4. The strike prediction equation and the measured mortality are compared in Figure 4.4.9-1. Good correlation between predicted and measured mortality is observed when correlated with the variable N L / D.

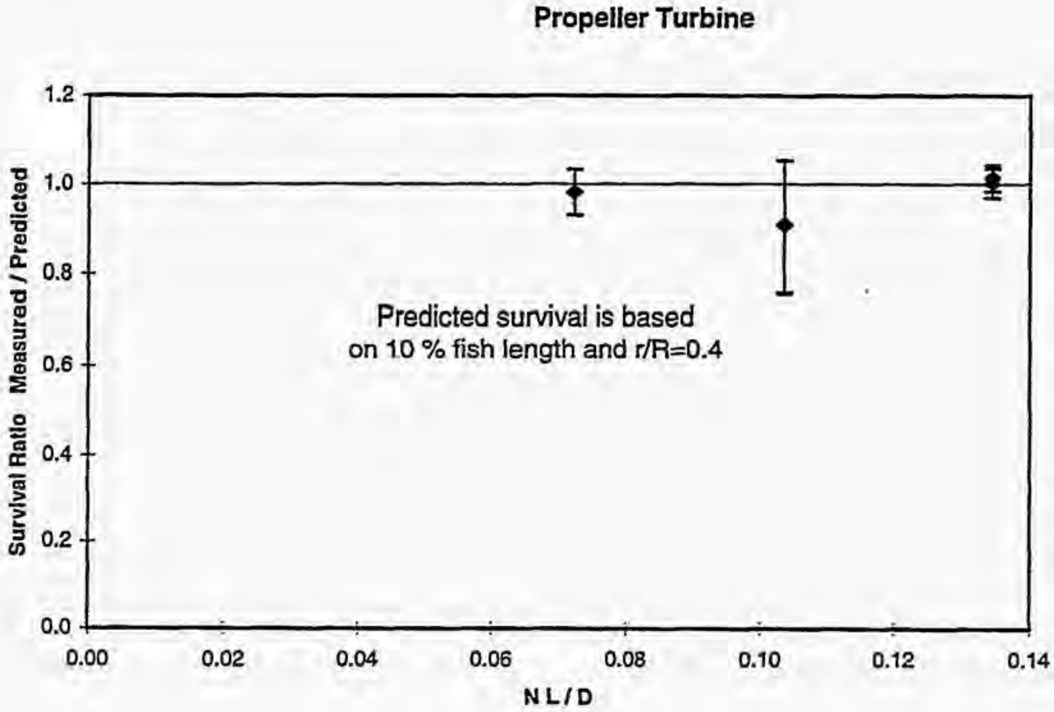


Figure 4.4.9-1 Survival Through Propeller Turbines is Well Correlated by N L / D

4.4.10 FRANCIS TURBINE EVALUATION OF PREDICTION METHODS

Summary

Poor correlation was achieved between measured survival and the strike equation prediction. Data inaccuracies, reduced applicability of the strike equations for high specific speed turbines, and pressure reduction inducted mortality are possible causes.

Discussion

The Francis turbine data is more extensive than the Kaplan data. It encompasses a greater range of turbine sizes, thus leading to a greater range of the variable N L / D. The specific speed range is also large. The accuracy of the Francis turbine strike (BZE) equations is believed to decrease as specific speed increases. It also appears that the credibility of the turbine operating data is not as high as the Kaplan data. In spite of this, all available data (Table 4.2.4) was used to evaluate the strike prediction formula. The correlation coefficient, lambda, was reevaluated for Francis turbines. Figure 4.4.10-1 shows considerable scatter for the calculation of lambda for each survival data point. A value of lambda equal to 0.2 was chosen, based on Kaplan turbine results, and on the absence of a reliable estimation method. Figure 4.4.10-2 shows poor correlation between measured and calculated survival. The use of data obtained from balloon tag experiments, which may have higher accuracy, does have less scatter.

Injury types were examined for two studies at Shasta and Cushman No. 2. At Shasta, injury allocation was as follows: 74.4 to 76.8% due to mechanical causes; 7.6% to 13.4% due to pressure, 6.2% to 7.1% shear related; 2% to cavitation (in one test); and the remainder to unknown causes. At Cushman, probable causes of injury were 57.1% to mechanical causes and 42.9% to pressure related causes.

The literature contains occasional references to correlation to runner tip speed. Section 4.3.3.3 presented a mechanism by which higher heads will increase the strike energy available. The experimental data were also used to examine a possible correlation. Figure 4.4.10-3 shows large scatter in the data. No such correlation is evident. High head Francis turbines, with typically longer penstocks than low head turbines, could have increased mortality due to pressure effects, if longer penstocks provide time for fish to acclimate to higher pressures. No correlation is evident for the effect of turbine head (Figure 4.4.10-4).

Different velocity profiles at the runner entrance were noted to occur for different specific speed designs in the strike equation development (Section 4.3.2.1) and it was speculated that the accuracy of the strike equation may decrease with higher specific speeds. No correlation is evident for the effect of specific speed (Figure 4.4.10-5).

The available experimental data does contain some interesting information. Figure 4.4.10-6 compares the value of the non-dimensional length parameter N L / D for Francis and for Kaplan turbines. Due to the generally smaller size and larger number of blades of Francis turbines, the non-dimensional length parameter is an order of magnitude larger than for Kaplan turbines. It is presumed therefore, that fish survival would be significantly lower

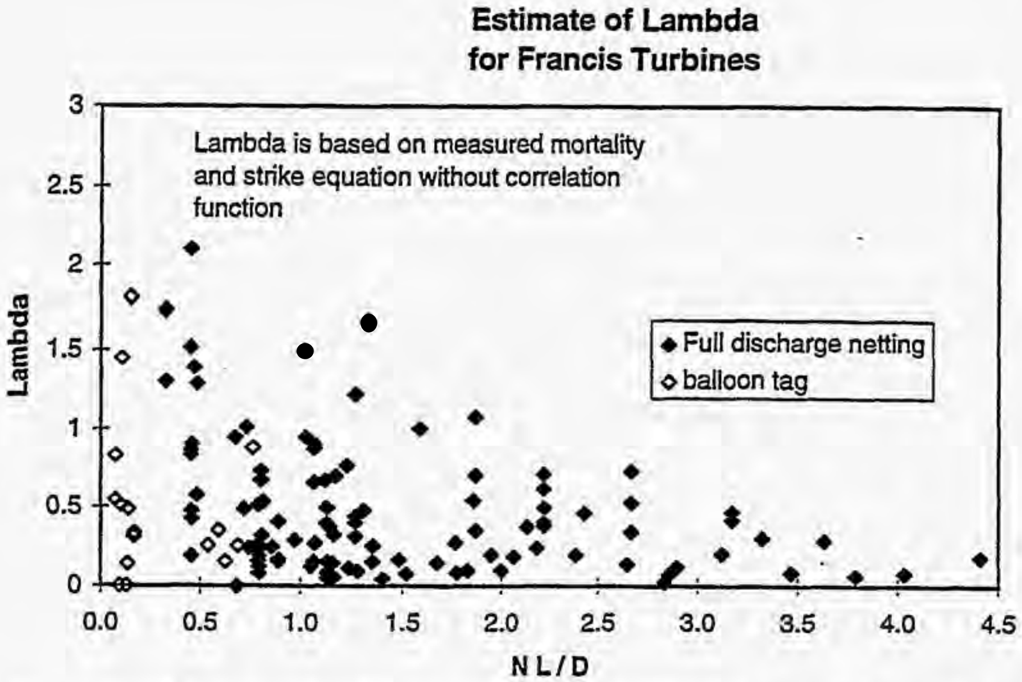


Figure 4.4.10-1 Calculation Of Correlation Coefficient For The Francis Turbine Data Set

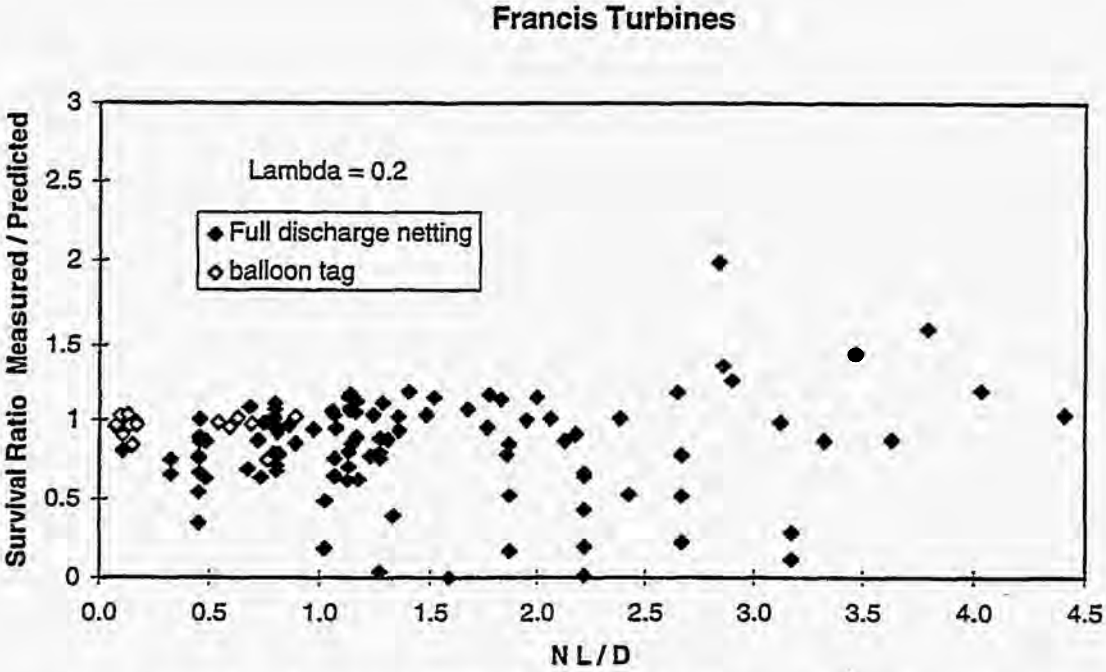


Figure 4.4.10-2 A Correlation Of The Francis Turbine Data Set Shows Large Scatter

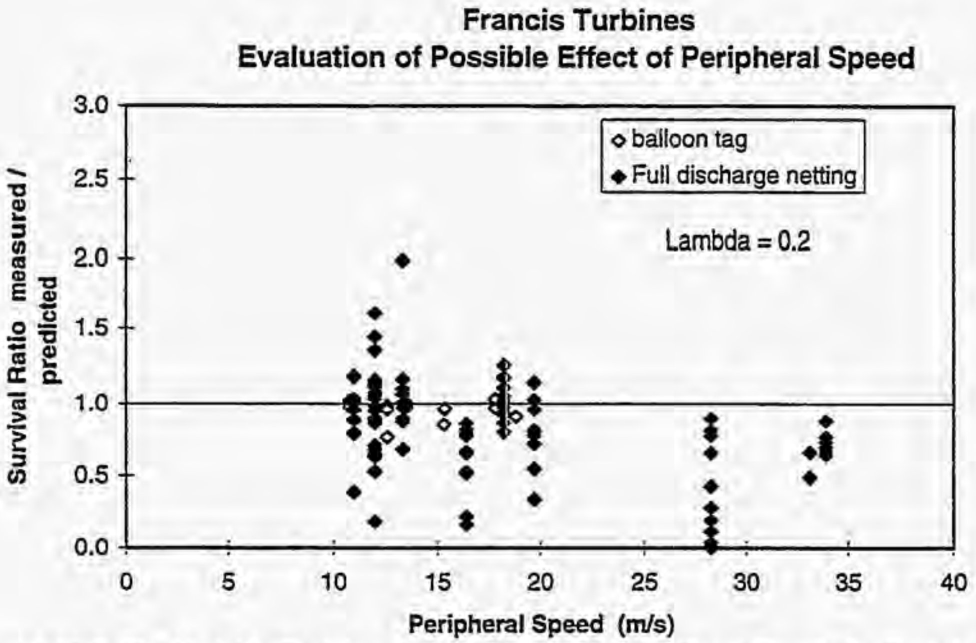


Figure 4.4.10-3 The Variable Peripheral Speed Shows No Significant Trends for Francis Turbines

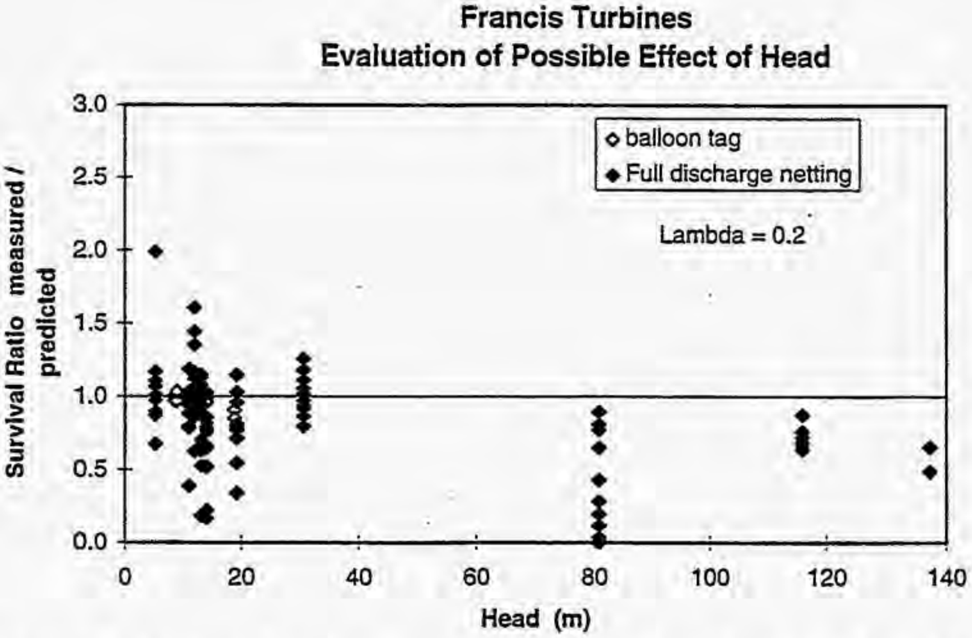


Figure 4.4.10-4 The Variable Peripheral Head Shows No Significant Trends for Francis Turbines

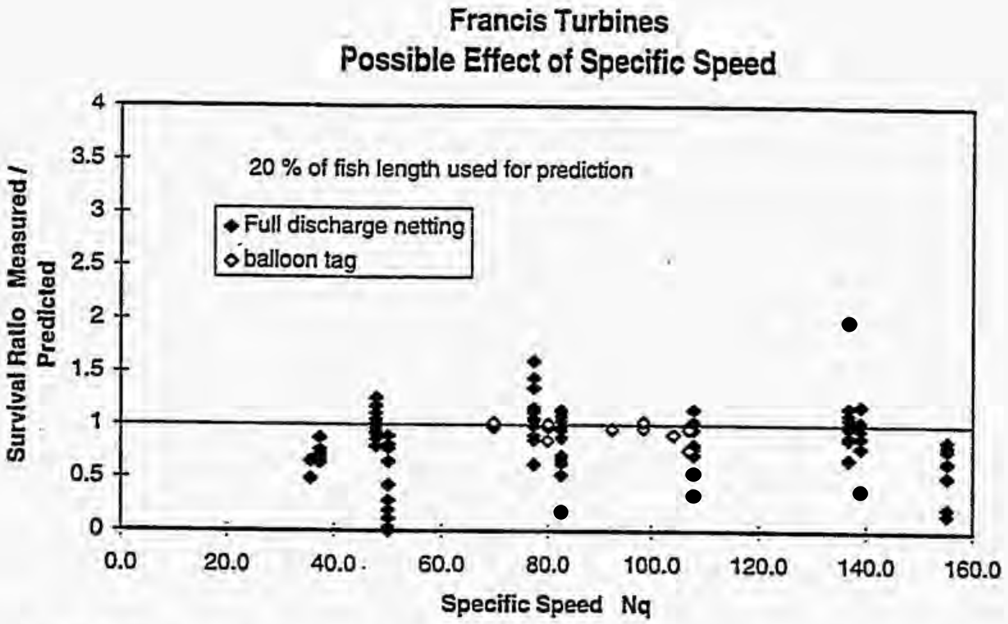


Figure 4.4.10-5 The Variable Peripheral specific Speed Shows No Significant Trends for Francis Turbines

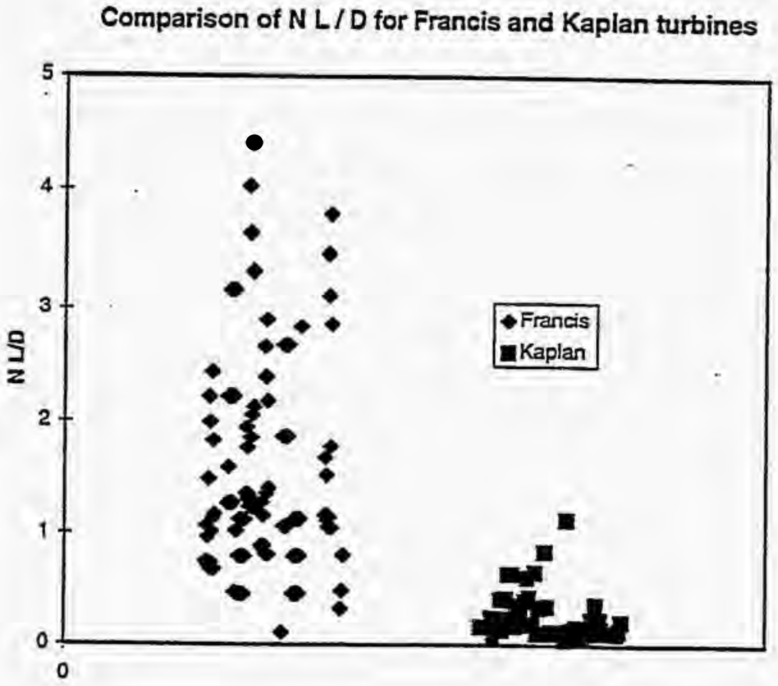


Figure 4.4.10-6 Higher Values Of N L / D For Francis Turbines Indicate Greater Strike Potential

4.4.11 EFFICIENCY AND FISH SURVIVAL AT FRANCIS SITES

Summary

Tests at the relatively high head turbines at Shasta and Cushman are the only data available to evaluate the effect of turbine operating point on fish survival. Survival does not appear to reach a maximum at flows less than peak efficiency, but there is no conclusive evidence that survival is highest at peak efficiency, The data do not preclude the possibility that the complex factors involved in survival cause maximum survival to occur at discharges greater than the peak efficiency discharge.

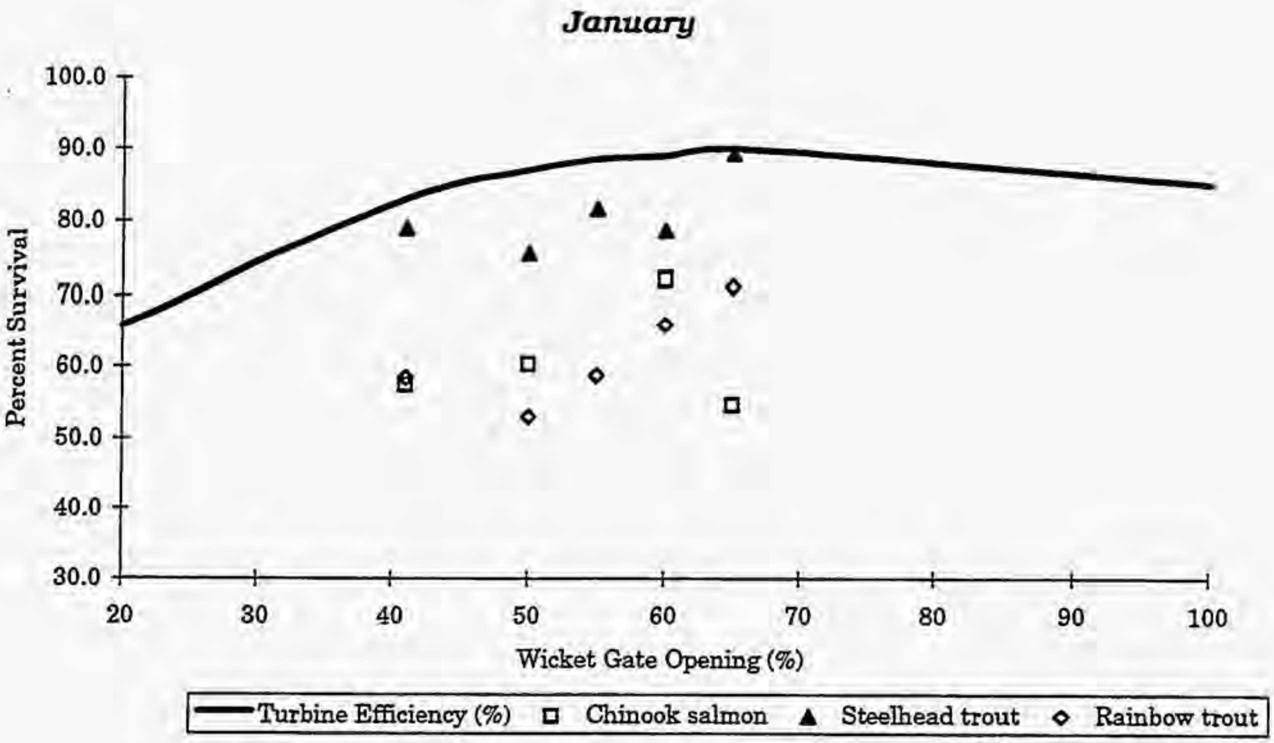
Discussion

In contrast to the fish survival data on Kaplan and propeller turbines, data on Francis turbines are characterized by studies at high head (>30 m or 100 ft) dams (primarily prior to 1970's) and those conducted in the 1990's, primarily at low head dams. Only two studies, prior to 1970, were conducted specifically to evaluate fish survival at several turbine operating efficiencies (Cramer and Oligher 1964). Both these studies were conducted at relatively high head dams (Cushman No. 2, head of about 136 m (450 ft), and Shasta, head of about 124 m or 410 ft). At Cushman No. 2, although two years of testing occurred, data presentation was different for each year. For the tests conducted in 1960 a composite survival rate of mixed salmonid species was given for each wicket gate opening; for the 1961 tests survival rates for individual species were provided (Cramer and Oligher 1964). Because of the different sizes of species tested (chinook salmon averaged 102 mm, steelhead 152mm, and rainbow trout 254 mm) pooling of data may mask the effects of interest. Results of these studies are discussed below.

Studies by other investigators have generally involved tests either at the prevailing turbine operating condition (with no accompanying data) or at the "worst case" scenarios (narrowing of the wicket gates to the lowest operable level). Many of these studies utilized small sample size, no controls, poor experimental protocols, use of mixed species, unknown fish lengths, unknown depth of fish introduction, etc. and thus are of little value for the present discussion. These deficiencies are apparent from data listings given in Appendix 10.1.

For the tests at Shasta, discharges were not available, so the relationship between wicket gate opening, efficiency, and fish survival was used, and is shown in Figure 4.4.11-1. Significant variability between the January and November tests is evident. Fish length varied with species, but different species had the best survival for different test points, with no pattern being evident. For the same species at similar wicket gate openings, survival differed up to 30%; the largest difference (30%) occurred between the January and November tests for chinook salmon at the highest wicket gate opening tested. Differences in survival between the two test periods for any species were generally less than 21%. No survival data could be obtained at discharges equal to or greater than the best efficiency discharge due to limitations on generator output. The highest survival did not consistently coincide with the discharge nearest to the best efficiency discharge. These data do not support a conclusion that operation at best efficiency enhances fish survival.

Two test series at Cushman No. 2 (Cramer and Oligher 1964), also provide survival information at different operating conditions. In the 1960 test data, Figure 4.4.11-2, changes in tailwater elevation had a large and perhaps dominant effect. Survival was extremely similar at the best efficiency discharge and at 100% gate opening , except for the low tailwater data. This point would have the most severe cavitation effects which may have caused disproportionate mortality. The 1964 data (Figure 4.4.11-3) show considerable variation as a function of gate opening. Maximum survival for Steelhead trout occurred at 50% gate opening (51.9%)and nearly the same survival was obtained at 76% gate opening (50.0%). Maximum survival for silver salmon occurred near the best efficiency point (72.0%), but nearly the same survival was obtained at 90% gate opening (71.7%). Overall these data do generally support the conclusion that survival is maximized near the best efficiency point, but due to the large variability in survival rates, the reliability is not high.



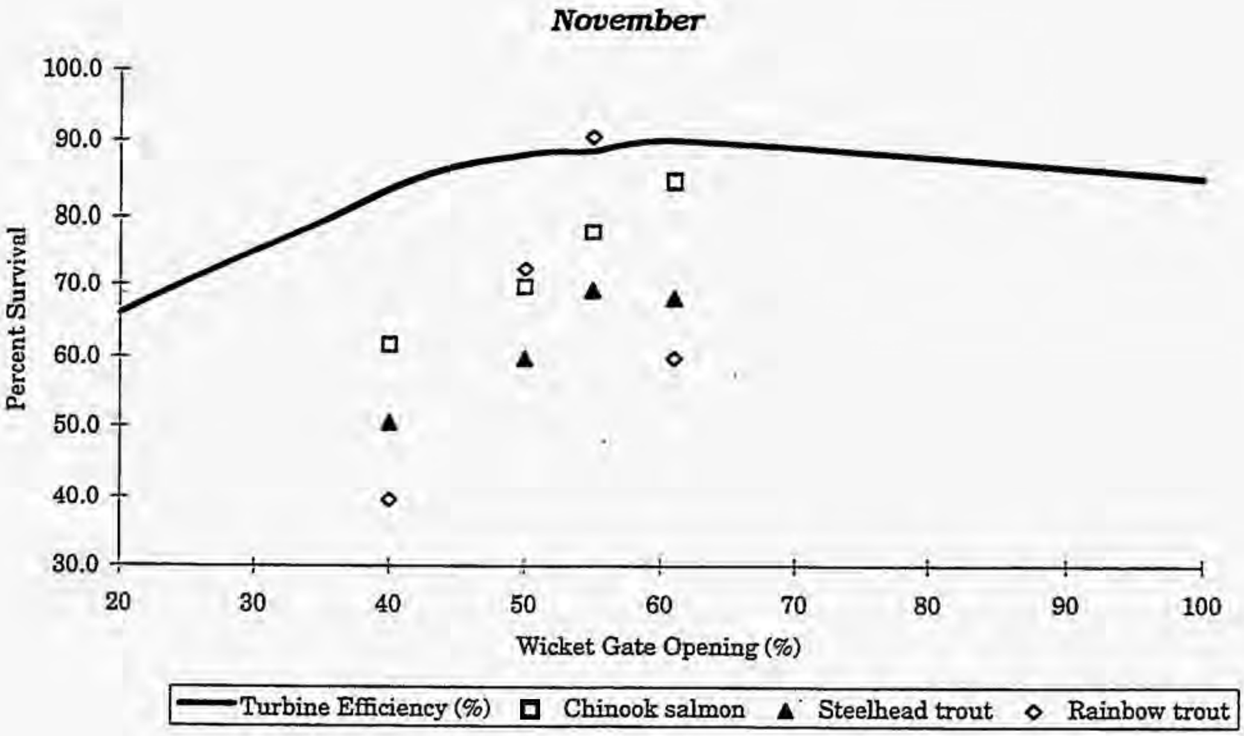


Figure 4.4.11-1 Survival Data at Shasta

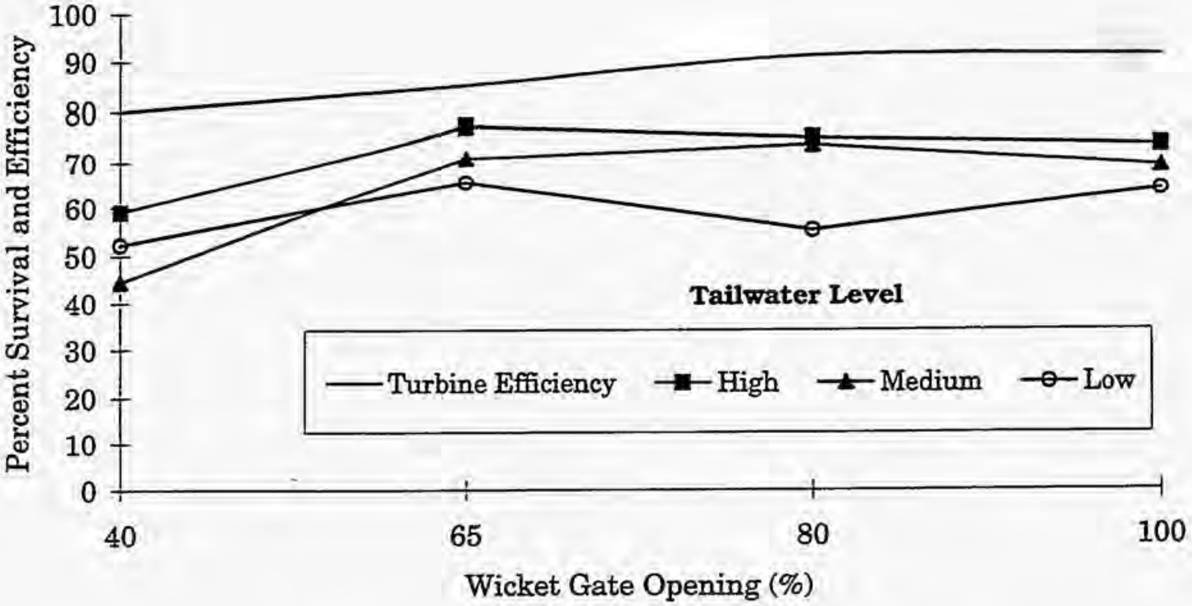


Figure 4.4.11-2 Survival Data at Cushman No. 2, 1960 Data

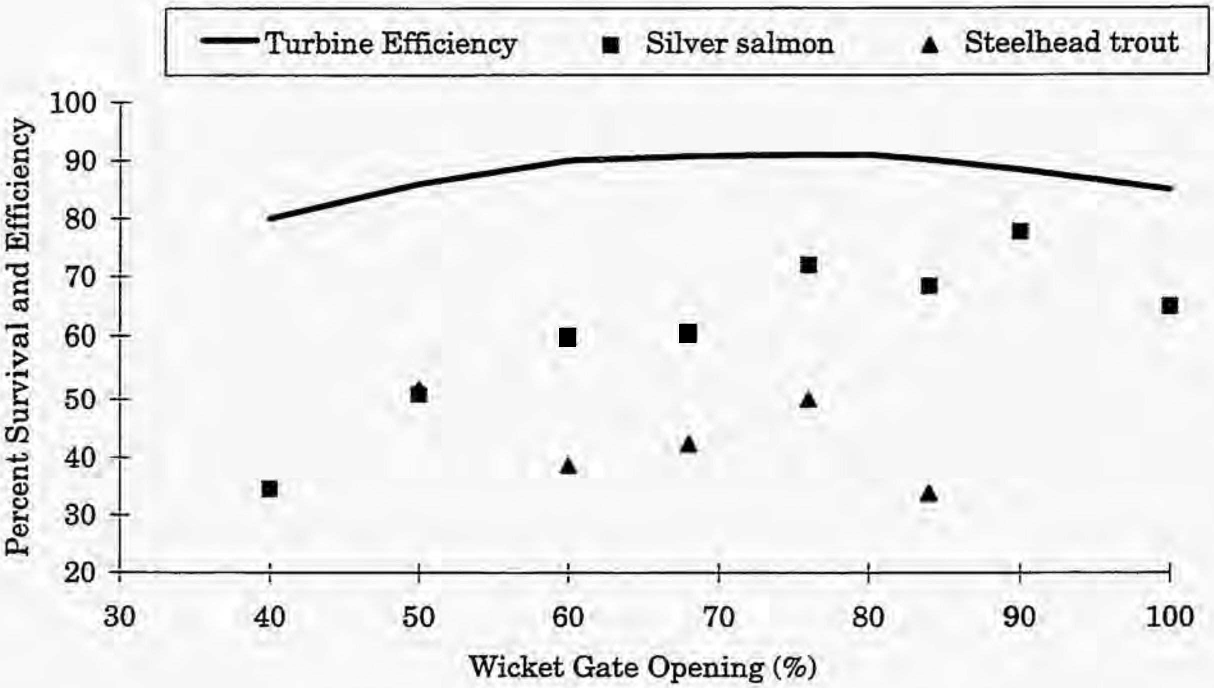


Figure 4.4.11-3 Survival Data at Cushman No. 2, 1964 Data

4.5 SURVIVAL THROUGH SLUICES/SPILLWAYS

Summary

Fish survival in passage through exit routes without moving parts (sluices, spillways, etc.) is not 100% at all sites; most likely because these passage routes were constructed primarily to transport excess river flow and debris, and not fish. Survival rates vary between sites and may be reflective of differences in unique physical and hydraulic features. However, fish transported through these conduits must contend with the potential effects of the same hydraulic forces (e.g., impact velocity, pressure change, shear, cavitation, etc.) as those passing the turbines. Thus, fluid related information obtained at these passage routes may be applied to the Advanced Hydropower Turbine System Program.

Discussion

Historically, spillways and sluiceways (Figures 4.5-1 and 4.5-2) at hydroelectric dams were deemed strictly as conduits for transporting excess river flow or debris with little focus on their potential for fish passage routes. In recent times, however, these conveyances are increasingly viewed as viable fish passage routes. Consequently, at many hydro dams, particularly on the Columbia River Basin, spill is used as an alternative to turbine passage because of reported higher survival rates for juvenile salmonid emigrants (Bell 1981; Eicher Associates 1987). However, spill is expensive in terms of lost power generation and with some spillway configurations can result in potentially lethal levels of total dissolved gas in the river. Additionally, many spillways are equipped with bottom opening tainter gates and the surface oriented fish, such as salmonids, may not be effectively guided by these spillways. These fish would be required to sound approximately 6 to 18 m (20 to 60 ft) to exit at such a conventional spillway.

To alleviate the dissolved gas saturation problem and to take advantage of the surface oriented behavioral patterns of fish, two major structural modifications of spillways have occurred at some hydro dams: installation of flow deflectors for total dissolved gas abatement (Figure 4.5-3), installation of top flow structures such as overflow weirs (Figure 4.5-4) or vertical slots to improve spill effectiveness for attracting surface oriented salmonid emigrants. Although there are no moving parts that fish encounter in passage through sluices or spillways, they are subjected to varying hydraulic forces (e.g., turbulence, pressure changes, variable terminal velocity), potential impact collisions with rock outcrops, abrasive surfaces, obstructions in flow path, etc. (Ruggles and Murray 1983). Therefore, the survival data obtained at these passage routes can provide some insight into factors (exclusive of mobile mechanical parts) that affect fish survivability and perhaps a better perspective on survival through turbines. The importance of impact velocity, pressure change, cavitation, and collision with structural objects has been noted mostly in laboratory experiments (Muir 1959; Harvey 1963; Groves 1972; Bell *et al.* 1972; Turnpenny *et al.* 1992). However, some recent survival studies at several hydro projects where fish were immediately recaptured upon passage through sluices and spillways provide empirical estimates of survival, injury types, and probable sources of injury/mortality (RMC 1992f; Normandeau Associates *et al.* 1996b,c,d).

4.5.1 SPECIFIC DATA

Survival rates of fish passage through sluices or over spillways have been estimated almost exclusively for juvenile migratory fish (e.g., salmon, American shad, and herrings) and only at limited sites. However, comparable survival rates in passage through turbines are not available from all sites. Table 4.5-1 shows some recent fish survival data along with the discharge, estimated impact velocity, species size, and head. The velocity data in Table 4.5-1 were estimated from the equation $v=\sqrt{2gh}$, where g=gravitational acceleration, 10 m/s (32 ft/s), and h=height (Bell 1972). As in the case of turbine passage survival database little species-related differences are evident. Additionally, the effects of head, discharge, or fish size on survival are difficult to discern from these limited data.

The effect of estimated impact velocity entering stilling basin on fish survival can differ with site, exit route, flow volume, and spillbay configuration Table 4.5-1. At spillways, with estimated impact velocities 18 m/s (60 ft/s) entering stilling basin survival exceeded 95%; while velocities greater than 18 m/s (60 ft/s) seemed to produce variable results depending upon the type of structural modification and flow volume. At Wanapum Dam, the survival was 92.0% at 57 cms (2,000 cfs) and 96.9% at 114 cms (4,000 cfs) in passage through a spillbay equipped with an overflow weir (estimated velocity was 19 m/s or 62 ft/s). At sluiceways, survival ranged from 93% to 99%; the impact velocities ranged from 13 to 22 m/s (41.6 to 71.1 ft/s). The three lowest survivals (92.0%, 93.0%, and 93.3%) observed included for Atlantic salmon in passage through sluiceways at Vernon and Wilder dams, and chinook salmon in passage through an overflow weir at Wanapum Dam, respectively. Results of some laboratory experiments reported in Bell (1984) and reproduced here in Figure 4.5-5 show that velocities of up to 18 m/s (60 ft/s) entering water had little effect on fish survival, however, velocities of 15 m/s (50 ft/s) striking against a solid object caused about 50% mortality. Velocities of <6 m/s (20 ft/s) striking solid objects caused little mortality. Turnpenny *et al.* (1992) reported similar findings for several species.

The potential effect of differential flow volume was observed only at two sites, Wilder and Wanapum dams (Table 4.5-1). The sluice passage survival of Atlantic salmon at Wilder Dam was lowest at intermediate spill volume tested, 9 cms (300 cfs), and highest at the lowest volume (6 cms or 200 cfs) tested. Water cascades down a sloping concrete channel with two concrete pillars adjacent to the discharge from the sluice. It is possible that passage conditions at 6 cms (200 cfs) were such that the probability of collision with the pillars was lower than at other discharges. At Wanapum Dam overflow weir an opposite effect of increasing flow on survival was observed (Table 4.5-1). At 57 cms (2,000 cfs) the estimated survival was only 92% but increased to 96.9% at 114 cms (4,000 cfs). Whether the hydraulic conditions improved at the higher spill volume is unknown. Field observations indicated great turbulence in the area between the overflow weir and tainter gate, but the difference in turbulence levels between 57 cms (2,000 cfs) and 114 cms (4,000 cfs) could not be quantified.

The principal causal mechanisms for injury/mortality to fish transported via spillways have been attributed to shear forces, turbulence, rapid deceleration after high terminal velocity, impact against the base of the spillbay, scraping against the concrete face of the spillbay, and rapid pressure change (Ruggles and Murray 1983). Although experiments have not been conducted to identify the relative importance of these factors in affecting fish condition/mortality at most spillways, reported injury types sustained included eye damage, embolism, hemorrhaging, and abrasions (Ruggles and Murray 1983). At Wanapum Dam, the scrape, cut, and bruise wounds were suspected to be caused by the fish physically contacting structural components at the spillbay including the frame of the weir, tainter gate, and flow deflector (Normandeau Associates *et al.* 1996d). The internal injuries (ruptured or hemorrhaged organs) could have resulted from pressure changes; bulging or hemorrhaged eyes have been attributed to pressure effects, as have corroborating symptoms such as expanded or burst air bladders and entrapped gas bubbles (Cramer and

Oligher 1964). A relatively high proportion of suspected pressure-related injuries was observed on overflow weir and sluiceway fish (Figures 4.5-6 and 4.5-7). The passageways for these fish are different; the sluiceway fish do not have to sound and presumably are not subject to significant pressure changes. However, fish attracted by surface currents to the overflow weir must sound to exit; thus, these fish may be exposed to pressure changes. Wanapum spillbay 2 is equipped with a flow deflector and the overflow weir was characterized by severe turbulence. Sluiceway fish experience a substantial vertical drop and the plunge pool is quite turbulent (Figure 4.5-1).

Injury rates and types may vary with site-specific exit route characteristics (Normandeau Associates *et al.* 1996b,c). At The Dalles Dam, chinook salmon smolts exhibited primarily hemorrhaging eyes and body injuries (Figures 4.5-8 and 4.5-9) in passage through an overflow weir; the estimated injury rate was 2.5%. At another spillbay modified with an "I" slot configured overflow weir, the injury was characterized by hemorrhaging and bulging eyes; the injury rate was estimated at 1.5%. The primary causative mechanisms for fish injury was attributed to collisions with structural components of the spillbay (tainter gate, baffles, or downstream end sill) and large boulders in the stilling basin. No observed injury was attributed to pressure, cavitation, or shear-related causes. At the unmodified spillbay the injury rate was 0.5% (Normandeau Associates *et al.* 1996c).

At Bonneville Dam, the injury rates of chinook salmon smolts in passage through a spillbay equipped with a flow deflector and a standard spillbay were similar (1.8% and 2.2%, respectively) and were attributed to contact with flow deflectors, tainter gates, or downstream dentates. Which of the structural components contribute most to injuries was not apparent (Normandeau Associates *et al.* 1996b).

Table 4.5-1

Short-term (one hour) fish survival over spillways or ice-log sluices at hydro projects. All studies utilized balloon tag-recapture technique.

Station/Location	Passage Route	Species	Height (m)	Estimated Velocity (m/s)*	Discharge (cms)	Survival (%)
Bellows Falls, VT	Sluice	Atlantic salmon	18.0	18.7	8.5	96.0
Wilder, VT	Sluice	Atlantic salmon	15.8	17.6	5.7	99.0
Wilder, VT	Sluice	Atlantic salmon	15.8	17.6	8.5	93.0
Wilder, VT	Sluice	Atlantic salmon	15.8	17.6	14.2	98.0
Vernon, VT/NH	Sluice	Atlantic salmon	8.2	12.7	1.1	93.3
Cabot, MA	Sluice	American shad	21.0	20.3	6.4	98.0
					0.0	
Crescent, NY	Spillway**	Blueback herring	4.0	8.8	1.1	100.0
Bonneville, WA	Spillway**	Chinook salmon	7.6	12.2	339.8	100.0
Bonneville, WA	Spillway**	Chinook salmon	7.6	12.2	339.8	100.0
The Dalles, WA	Spillway**	Chinook salmon	15.2	17.3	297.4	95.4
The Dalles, WA	Spillway**	Chinook salmon	15.2	17.3	297.4	99.2
The Dalles, WA	Spillway**	Chinook salmon	15.2	17.3	127.4	98.9
Wanapum, WA	Spillway**	Chinook salmon	6.4	11.2	121.8	99.6
Wanapum, WA	Spillway**	Chinook salmon	6.4	11.2	121.8	95.7
Wanapum, WA	Spillway**	Chinook salmon	18.3	18.9	56.6	92.0
Wanapum, WA	Spillway**	Chinook salmon	18.3	18.9	113.3	96.9
Wanapum, WA	Sluice	Chinook salmon	24.1	21.7	56.6	97.4

• Estimated velocity (V)= 2gh; g=gravitational acceleration, h=head or height (Bell *et al.* 1972).

** Modified spillbay with flow deflectors or overflow weir.

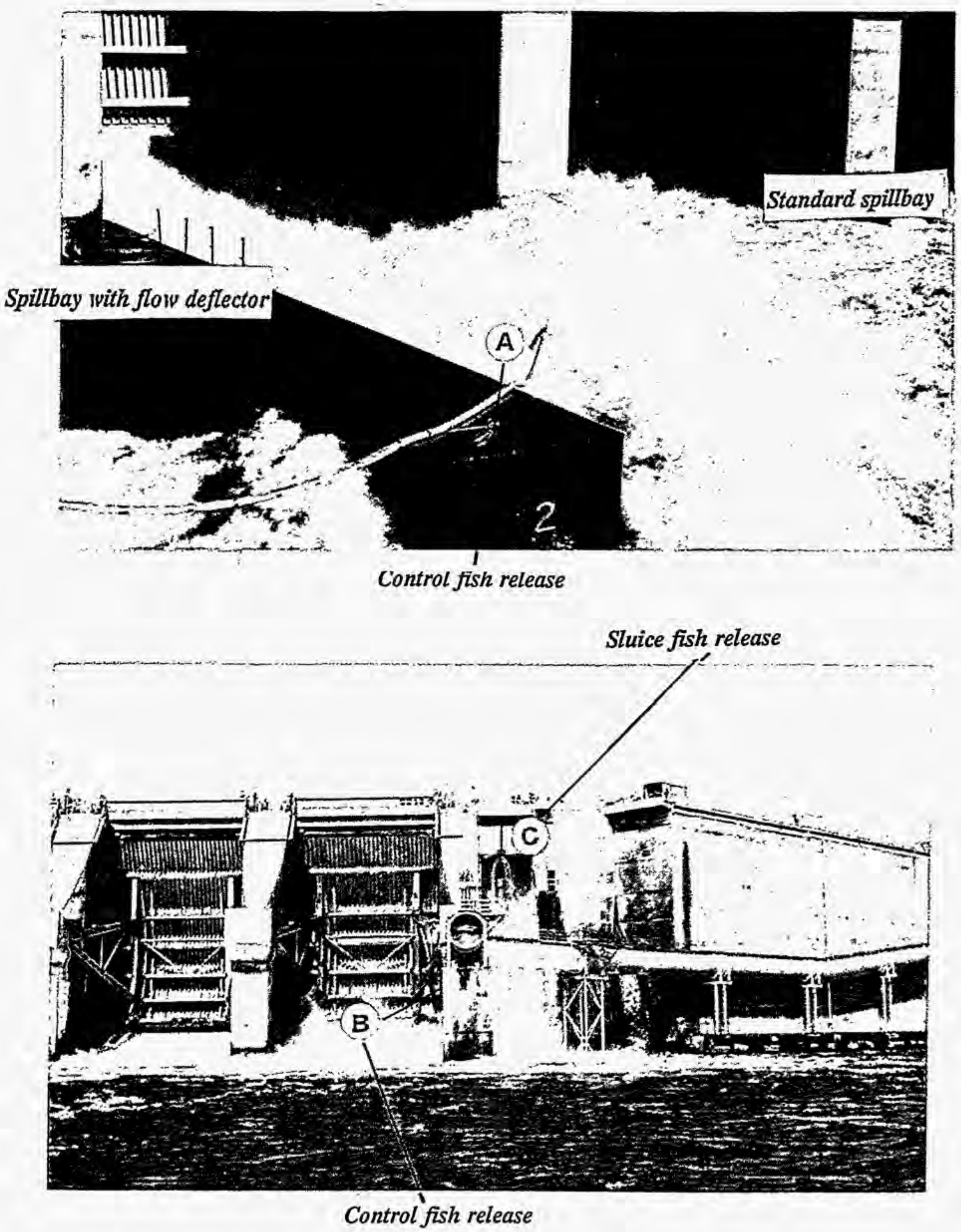
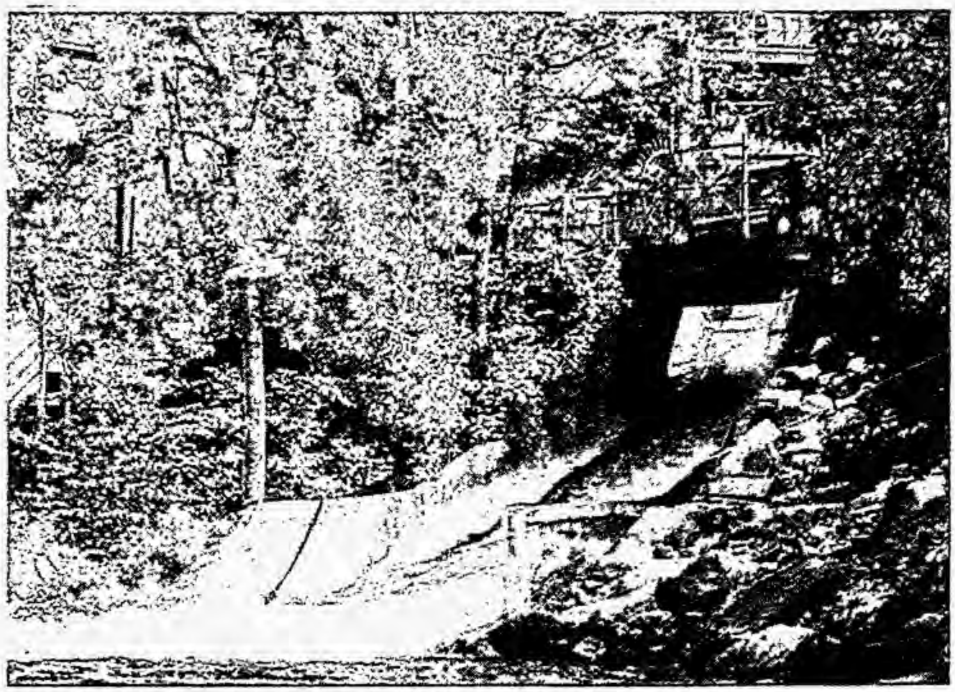


Figure 4.5-1

Spillbay and Sluice Discharge Conditions. From Normandeau *et al.* (1996d)

10 ft spill at top of sluice

Sluice exit



Deflection spill at bottom of sluice

Figure 4.5-2 Log sluice bypass; note deflection spill in C. From RMC (1994c).

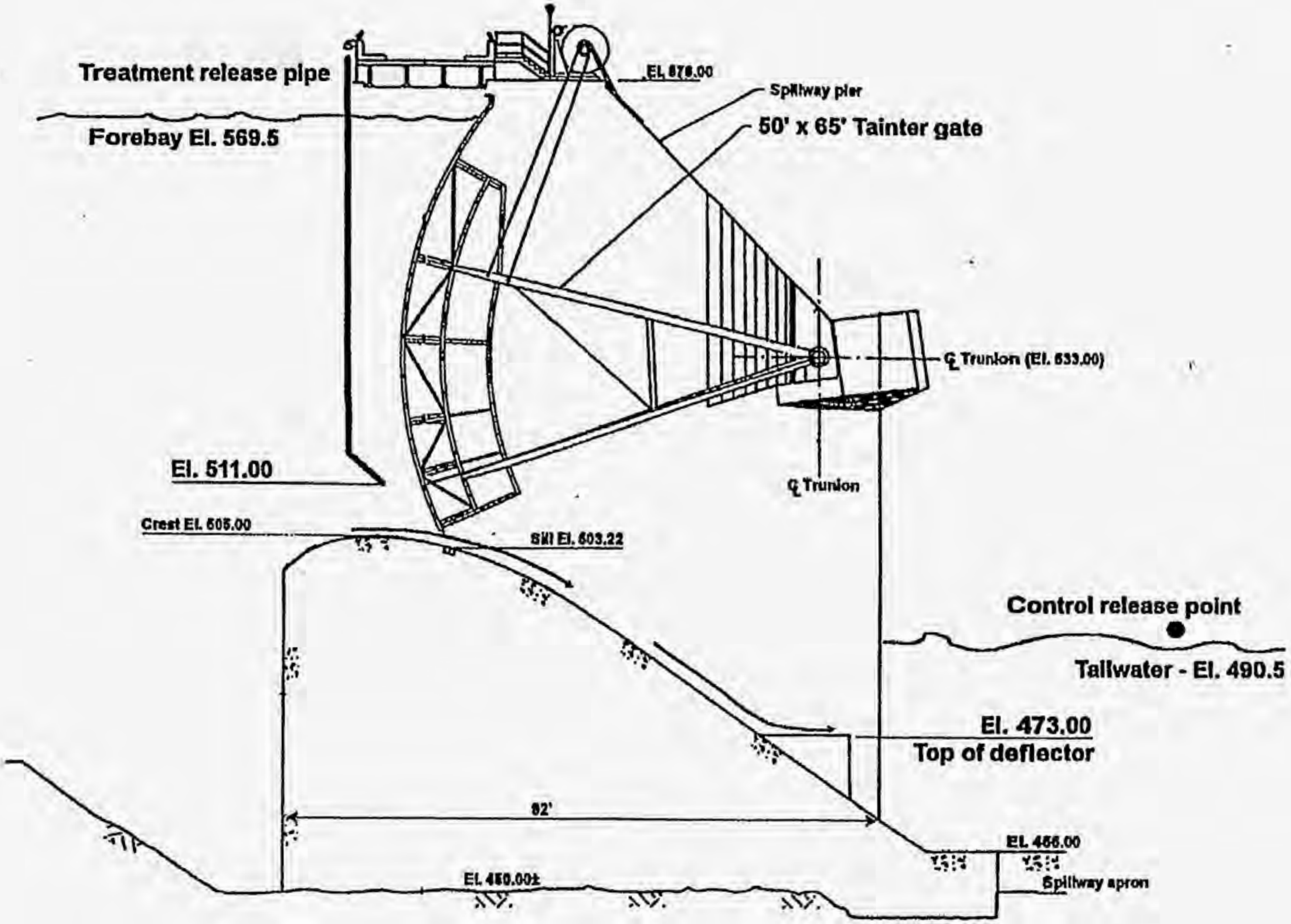


Figure 4.5-3

Schematic of a spillbay equipped with flow deflector.

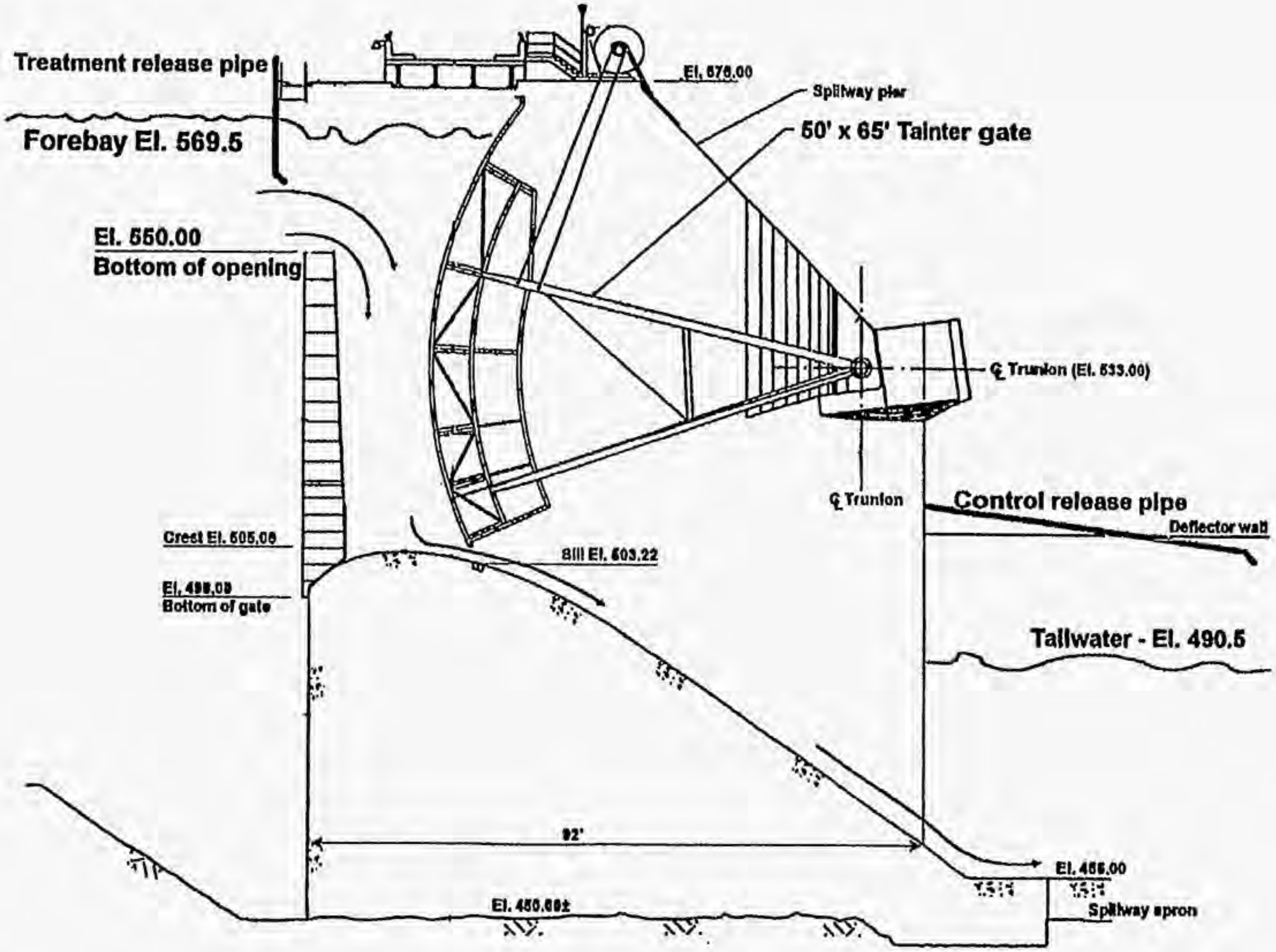


Figure 4.5-4

Schematic of a spillbay equipped with overflow weir.

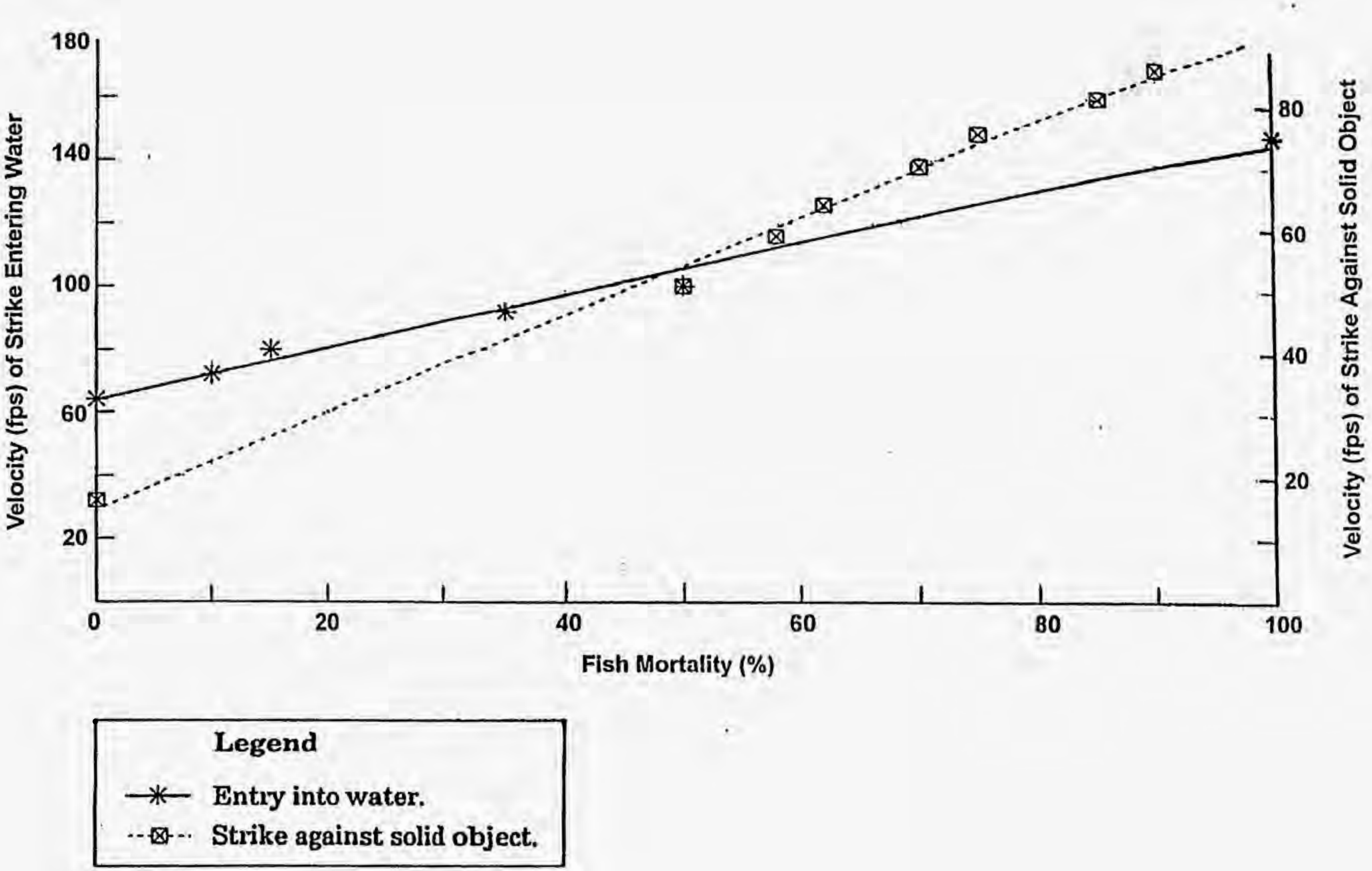


Figure 4.5-5

Relationship of velocity (fps) and fish mortality. (From Bell 1991).

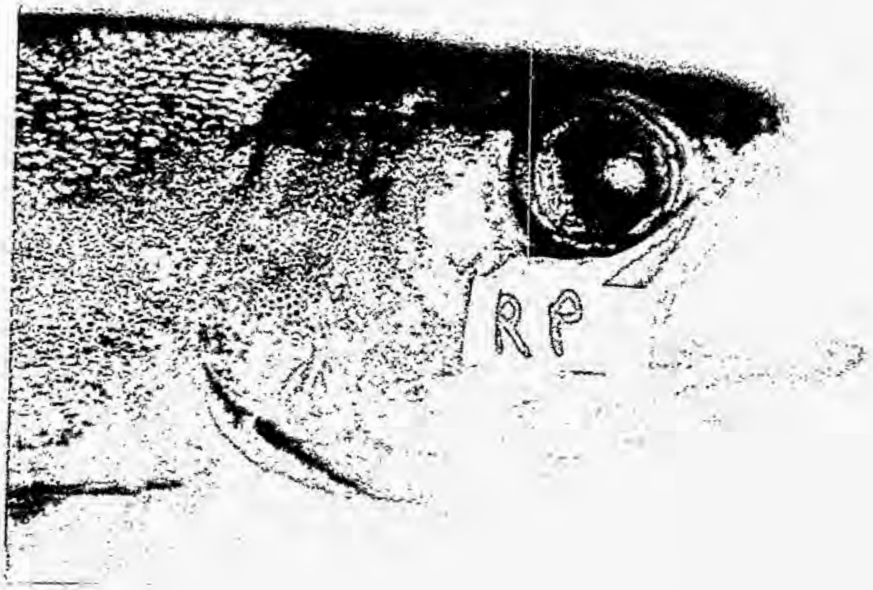


Figure 4.5-6

Example of pressure related injury (hemorrhaged eyes) at a spillway. From Normandeau Associates *et al.* (1996d).

Figure 4.5-7

Example of probable shear/mechanical injury in passage at a spillway. Note torn/bent left operculum. From Normandeau Associates *et al.* (1996d).

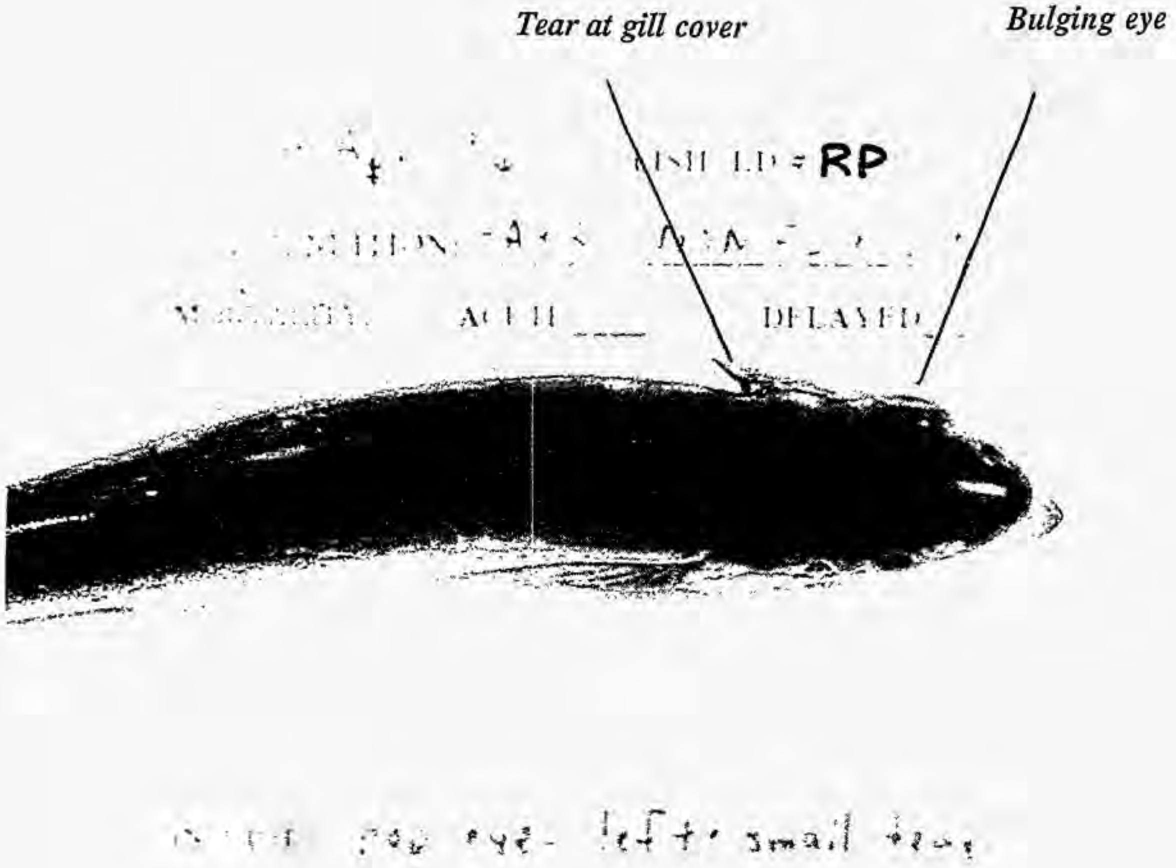
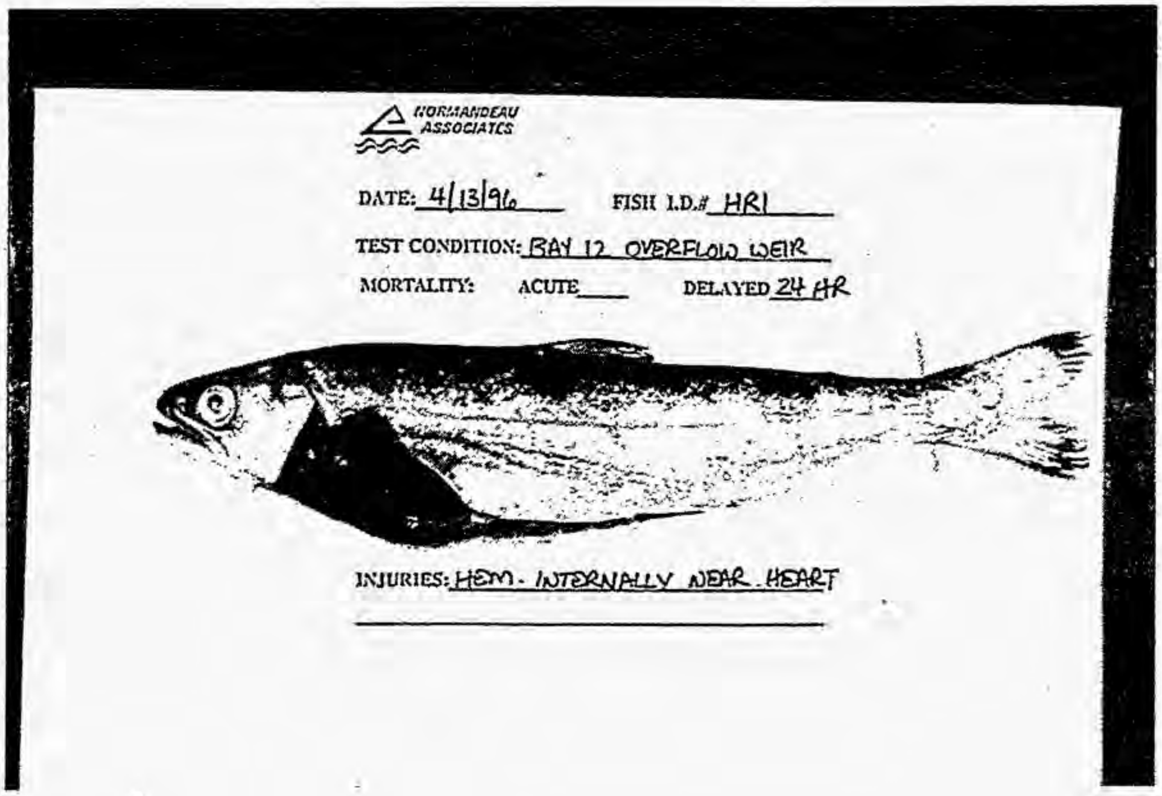


Figure 4.5-8

Example of pressure/shear injuries in passage at a spillway with flow deflector. Note bulging left eye, tear at top of gill cover. From Normandeau Associates *et al.* (1996d).



NORMANDEAU ASSOCIATES

DATE: 4/13/96 FISH I.D.# HR1

TEST CONDITION: BAY 12 OVERFLOW WEIR

MORTALITY: ACUTE ___ DELAYED 24 HR

INJURIES: HEM. INTERNALLY NEAR HEART

Figure 4.5-9

Example of injuries sustained by fish in passage through overflow weir at Wanapum Dam. From Normandeau Associates *et al.*(1996d).

4.6 PERSPECTIVE ON IMPROVEMENTS OF FISH PASSAGE SURVIVAL

Summary

The fish passage survival of any turbine design should be evaluated against a standard of comparison (benchmark) survival rate based on the "best of class" alternatives for passage including bypass systems or spillways, other turbines of similar type with similar L/D ratios, or other turbines of any type and characteristic. Fish passage survival of any modified turbine design should be evaluated against a benchmark where the passage survival has been evaluated in a before and after manner using comparative controlled experiments. The Rocky Reach experiments with the new turbine show that the relative effectiveness of engineering solutions (e.g., closure of gaps at the trailing edge of the runner blade) can be quantified by comparative controlled experiments performed before and after implementation of engineering solution to enhance fish survival.

Discussion

A quantitative evaluation of a turbine design for its enhanced fish passage survival capability should be an integral component of the AHT program. It should include considerations for the establishment of guidelines for "best of class" survival and for "comparative" survival and use these benchmarks to determine a magnitude of improvement that is achievable or possible or that was obtained, and could contain the following steps:

1. establishment of "best of class" benchmarks for survival rates; these may be established using the existing databases (passage through turbines or alternative exit routes such as bypasses, spillbays, ice-log sluiceways, etc.), or be based on new experiments utilizing well designed comparative controlled experiments. These "best of class" benchmarks would establish the maximum possible limit of improvement in fish survival. For example, if the benchmark is established at 95% at a hydroelectric project scheduled for rehabilitation, then the possible improvement would not be measured based on the ideal of 100% survival, but rather on the difference between the actual survival and the benchmark. Whether this benchmark survival rate is achievable for the particular installation is unknown but it can help in comparisons with other acceptable passage alternatives.
2. actual tests to determine the fish friendliness of the modified design. The same experimental protocols used in establishment of the "comparative" benchmark survival rates (in this case, the unmodified design) should again be used to obtain the new data.

The passage survival study at Rocky Reach Dam on chinook salmon smolts (Normandeau Associates and Skalski 1996), having well designed comparative controlled experiments, used the "comparative" benchmark to evaluate whether changes in fish injury/survival due to engineering design modifications of turbines can be detected. Results from the secondary fish releases showed that the zonal injury rate was reduced from 5% to 2.8% and a minimal improvement in survival rate of 4.4% for chinook salmon smolts introduced at 3 m (10 ft) depth below the intake ceiling after a temporary engineering solution was implemented (a steel wedge was placed to close the gap between the runner trailing edge and the hub) could be inferred from the reported data. Although controls were not released concurrently with the secondary releases to evaluate the effects of tagging, handling, and induction results from primary releases of chinook showed control survival rate exceeded 99%. The probable sources of zonal injuries changed as well; prior to the interim engineering solution most injuries appeared to be pressure- and shear-related. After the modification most of the injuries were attributed to probable mechanical causes. Obviously, the hydraulic conditions were altered due to "re-engineering" the turbine's local geometry by closing the gap and changing the zonal injury mechanisms.

4.7 SUMMARY AND CONCLUSIONS

Section 4 combined insights of biologists and turbine designers to develop an understanding of the mechanisms of injury/mortality which may occur within hydropower projects. The combined work of the team in discussing the problems and probable causal relationships as well as the reevaluation of existing data and evaluation of new data sets to achieve an insight toward potential mechanisms led to some new understandings, some perceptions which need further testing through future experiments, and to some conclusions for design concepts. Based on the material presented, the criteria for improvements to features of existing turbines and for new turbine designs are derived based on causal mechanisms.

Zonal Observations

The mechanisms of injury/mortality are zonal. Some zones of geometry within the turbine are relatively free of injury/mortality mechanisms. Others exist having high mechanism concentrations. The zones a fish traverses in its passage through a turbine has a significant effect on its injury/mortality. At Rocky Reach, a change to the geometry of the blade-hub zone led to the inference of a 4.4% increase in survival. At Wanapum, fish that were believed to pass through the runner near the hub zone experienced a 5% increase on mortality compared to fish passing through the middle zone of the runner.

The development of the blade encounter zone (BEZ) concept and the development of an improved encounter equation with the significant parameter N L / D led to a better analysis technique whereby new correlations and insight were achieved. The BEZ contains a number of injury/mortality mechanisms. Some of these are related to blade strike. Others are related to blade end gaps and local fluid effects. Quantification of exact sources of injury/mortality of fish transported through turbines is difficult due to a lack of controlled experiments and to the fact that the observed symptoms could be manifested by two different sources. Correlation of predictive methods with zonally planned and executed fish passage survival testing is the key to understanding injury mechanisms and developing improved survival prediction methods. Historical studies primarily focused on juvenile salmonids of limited size range. Most studies did not provide turbine operating data or location of fish injection.

In spite of the limitations of the existing data, they are the basis for correlations with the BEZ prediction method. Some of the new insights obtained through this correlation effort are:

- The combined variable N L / D plays a major role. Individual values of N, L, or D play no significant role.
- Because impact velocities related to peripheral speed are always above the critical value where direct impact is assumed to be fatal, peripheral speed has no observable effect on survival in data analyzed for Kaplan turbines, and for Francis turbines plays no obvious role. Other analyses of the shear damage mechanism have provided the physical basis to explain this observation.
- Data did not show a species effect.

A new concept of theoretical avoidable loss (TAL) is introduced, and in tests at Wanapum, has provided a tool to further understand the relationship between fish injury/mortality and energy dissipation within a turbine. For these tests, low values of TAL correlated with high survival. However the TAL concept treats the entire turbine as a single zone and does not lead to specific injury/mortality insight.

A number of zones in a turbine have regions of fluid shear. Effects of shear induced forces have been studied primarily under laboratory conditions. However, these studies indicate that shear effects may be species and size specific and are related to the orientation of fish in the shear zone. Larger sized fish and those facing the water jet appear to suffer less injuries. CFD studies have been used to estimate a threshold value of shear (450/s). Shear values greater than this threshold or critical value can cause mortality. Through a consistent application of the shear principles, a velocity of greater than 5 m/s adjacent to a structure will generate a shear zone above the critical value to cause mortality to fish. Velocities of this magnitude can exist in several zones in a typical turbine.

In all zones fish experience pressure changes. Pressure-related injuries appear to be more a function of acclimation history of fish upstream of turbine than passage through turbines *per se*. At dams (>30 m or 100 ft head), without hydro turbines, fish transported through bottom sluices or openings suffer decompression trauma (as evident by rupture of air bladder and other internal organs) when rapidly exposed to shallow tailrace conditions. Inside of turbines, the pressure distribution and the rate of change of pressure with time can be determined accurately by CFD analysis. Transit times from the high pressure region at the turbine spiral case inlet to the exit of the draft tube are relatively short, even for big turbines. Transit times through low pressure regions in the runner blade region are quite short. The rapid transit through this low pressure region in the blades is felt to cause no significant mortality. More important is the change in pressure from that to which the fish has become acclimated to a lower value at the draft tube exit.

It is perceived that fish passing through zones with cavitating flow fields can be damaged by fluid effects arising because of cavitation and subsequent vapor bubble collapse. The turbine operating condition that coincides with the onset of cavitation can be determined by CFD analysis. Designs for existing turbines can be developed to eliminate cavitation while increasing power production. Operational guidelines to minimize operation in cavitation regimes will reduce maintenance costs and reduce fish mortality associated with fluid induced loading on fish bodies related to cavitation.

Physical model testing and CFD analyses have investigated the impact of the presence of fish screens on velocity distributions in turbine intakes. Physical model tests indicate significant redistribution of flow takes place. Fish screen effectiveness testing has shown that the exclusion of fish entering the turbine is not 100%. Therefore, some portion of the unguided fish go under the fish screen. Fish passing the intake under the fish screens may then find themselves in the lower portion of the water column where they are expected to pass through the lower zone on the wicket gates and near the outer radius zone of the blades. These fish will experience a different set of turbine geometry and associated fluid effects compared to fish passing through the upper portion of the wicket gates and the mid to inner regions of the blades. Compared to units without fish screens, the presence of fish screens may cause different fish survival characteristics.

Operation

The operation point of turbines has a large effect on fish survival. Tests at Wanapum Dam were designed, conducted and evaluated to shed light on injury mechanisms and how they changed with zone and the point of operation. Peak survival did not coincide with peak efficiency, but occurred at a discharge where the blade strike probabilities were low, while the TAL were at a minimum, and before cavitation began to be significant. The proposition that operation at discharges within 1% of peak efficiency will maximize fish survival was examined. Historical data that had been generally believed to support this belief, from the Big Cliff and Foster Kaplan turbines, was reanalyzed. This data does not show that maximum fish survival occurs at discharges within 1% of peak efficiency. Analysis of the complex factors involved reveal tendencies for fish survival that operate in opposing directions. That is, some factors maximize survival at high discharges while other factors maximize survival at low discharges. Tests at the relatively high head turbines at Shasta and Cushman are the only data available to evaluate the effect of Francis turbine operating point on fish survival. Survival does not appear to reach a maximum at discharges less than peak efficiency, but there is no conclusive evidence that survival is highest at peak efficiency, The data do not preclude the possibility that the complex factors involved in survival cause maximum survival to occur at discharges greater than the peak efficiency discharge.

Fish survival in passage through exit routes without moving parts (sluices, spillways, etc.) is not 100% at all sites; most likely because these passage routes were constructed primarily to transport excess river flow and debris, and not fish. Survival rates vary between sites and may be reflective of differences in unique physical and hydraulic features. However, fish transported through these conduits must contend with the potential effects of the same hydraulic forces (e.g., impact velocity, pressure change, shear, cavitation, etc.) as those passing through turbines.

Design Observations

For Kaplan turbines, blade end gaps are judged to be a significant source of injury/mortality.

Adjustable turbine and generator speed offers the possibility of improved fish survival. Adjusting the speed to compensate for head changes will allow the head coefficient characterizing the turbine performance location on the turbine performance hill curve to be kept at a more favorable operating point than a design having constant speed.

The effects of some of the injury/mortality mechanisms are significantly greater than others and depend on turbine type, size and project variables. For some Kaplan turbine projects, increasing the number of blades to reduce gaps results in a significant fish passage survival improvement. For other projects, decreasing the number of blades may be the correct solution. The effect is project related. Fewer blades for Francis turbines appears to be a feasible design concept. For large turbines passing small fish, high survival can be achieved. For small Francis turbines with large fish, the best solution for fish survival may be to keep the fish out of the turbines.

Other

The fish passage survival of any turbine design should be evaluated against a standard of comparison (benchmark) survival rate based on the "best of class" alternatives for passage including bypass systems or spillways, other turbines of similar type with similar L/D ratios, or other turbines of any type and characteristic. Fish passage survival of any modified turbine design should be evaluated against a benchmark where the passage survival has been evaluated in a before and after manner using comparative controlled experiments.

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5.0 TASK 3 REPORT -- INVESTIGATION OF INDIVIDUAL DESIGN ELEMENTS

5.0 TASK 3 REPORT – INVESTIGATION OF INDIVIDUAL DESIGN ELEMENTS

5.1 INTRODUCTION

Different aspects of individual design elements were investigated by three team members. Voith utilized its family of CFD tools to provide details of the flow fields in many locations in Kaplan and Francis turbines. The goal was to demonstrate the nature of internal turbine flows, quantify velocity and pressure fields, and sharpen knowledge regarding fluid mechanisms leading to possible fish injury. Georgia Institute of Technology studied more basic flow physics to improve our understanding of advanced CFD methods. TVA reviewed the state of the art in mitigation of low dissolved oxygen flows, and reported on the testing, and operation of aerating turbines.

5.2 DISCUSSION

Voith analyzed the entire water passage of Kaplan and Francis turbines. Several computer programs were used in this study including proprietary and commercial codes. Detailed CFD calculations were performed using a commercial three-dimensional Navier-Stokes turbulence modeled code, TASCflow, developed by Advanced Scientific Computing Ltd. (Raw 1995, Thomas et al. 1989). A standard K - epsilon method was applied for the turbulence modeling. The calculations were applied for semi-spiral intakes, spiral case, stay vanes and wicket gates, runners and draft tube. Many of these calculations were performed explicitly for this project. Some results were taken from ongoing efforts. The goals included:

- ascertain probable fish paths
- illustrate the complex flow fields that exist in turbines
- evaluate and sharpen expectations regarding fluid induced injury mechanisms of shear, rate of pressure reduction, cavitation, and gap flows.
- examine draft tube flow conditions for recirculation (entrapment regions), draft tube pier impact, and disorienting effects

Georgia Institute of Technology developed and tested advanced numerical methods and turbulence models for simulating hydroturbine flows. The combined effects of numerical resolution and turbulence modeling on the accuracy of complex flow predictions were investigated in great detail. An advanced turbulence model was proposed which was shown to yield superior results as compared to existing models. A computational framework for predicting unsteady flow phenomena in Francis-type hydroturbines was also developed and applied to simulate formation of rope-like vortices. These advancements provide us with a better understanding of the complex flow environment inside the powerplant which, in turn, will lead to more accurate evaluations of fluid induced fish injuries. The potential of coupling advanced CFD methods with a virtual fish numerical model, capable of estimating fish trajectories and flow induced loads on fish bodies, was also demonstrated.

TVA reviewed the state of the art in mitigation of low dissolved oxygen flows, and studied the testing, and operation of aerating turbines.

5.3 THREE-DIMENSIONAL CFD STUDIES

5.3.1 KAPLAN INTAKE

A Kaplan turbine intake with no fish screen was analyzed. The region from the trashracks through the runner, including all stay vanes and wicket gates was calculated. The purpose of the calculation was to analyze the semi-spiral case with an emphasis on flow through the stay vanes and wicket gates and into the runner. This calculation does not contain features that would permit accurate calculations through or below the runner. The inlet flow boundary to the intake was assumed to have constant total pressure. This simulates inflow from a large quiet reservoir, and permits a velocity profile to develop. The exit plane, below the runner, was specified to have the desired mass flow. The presence of the runner was simulated by applying a tangential body force to remove swirl that was induced by the stay vanes and wicket gates. The grids on the outer surface of the model are shown on Figure 5.3.1-1

The results of the calculation are demonstrated by streamline plots, Figures 5.3.1-2 through 5.3.1-4. The path of a streamline is not affected by forces on a three-dimensional body, centrifugal effects, or buoyancy. In the absence of a "virtual fish" calculation that could account for more realistic fish effects, streamlines are tentatively accepted as an approximation of fish paths. Streamlines at a given height at the trash racks travel to a constant radius in the runner region. Generally, streamlines near the roof travel to the hub and streamlines near the floor travel to the periphery. The results for this geometry support the previously assumed hypothesis that fish from the near surface region would tend to be discharged through the runner near the hub. These streamlines are colored by transit time for a laboratory scale model turbine. Figure 5.3.1-5 shows a transit time of 30 seconds for a prototype size (Wanapum) intake.

VOITH

VOITH SEMI-SPIRAL CASE
SURFACE GRID

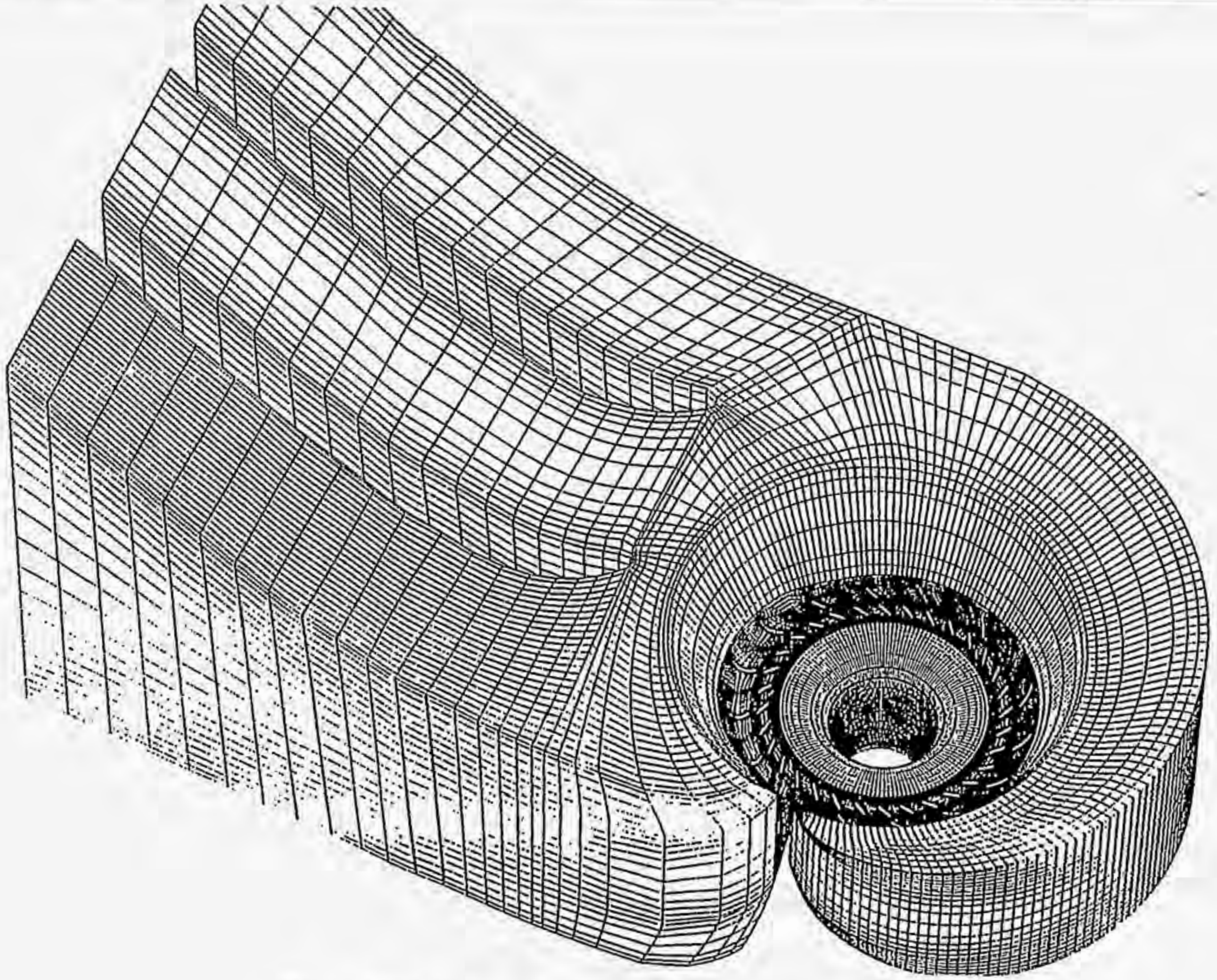

Figure 5.3.1-1 Grids On The Surface Of A Semi-Spiral Case

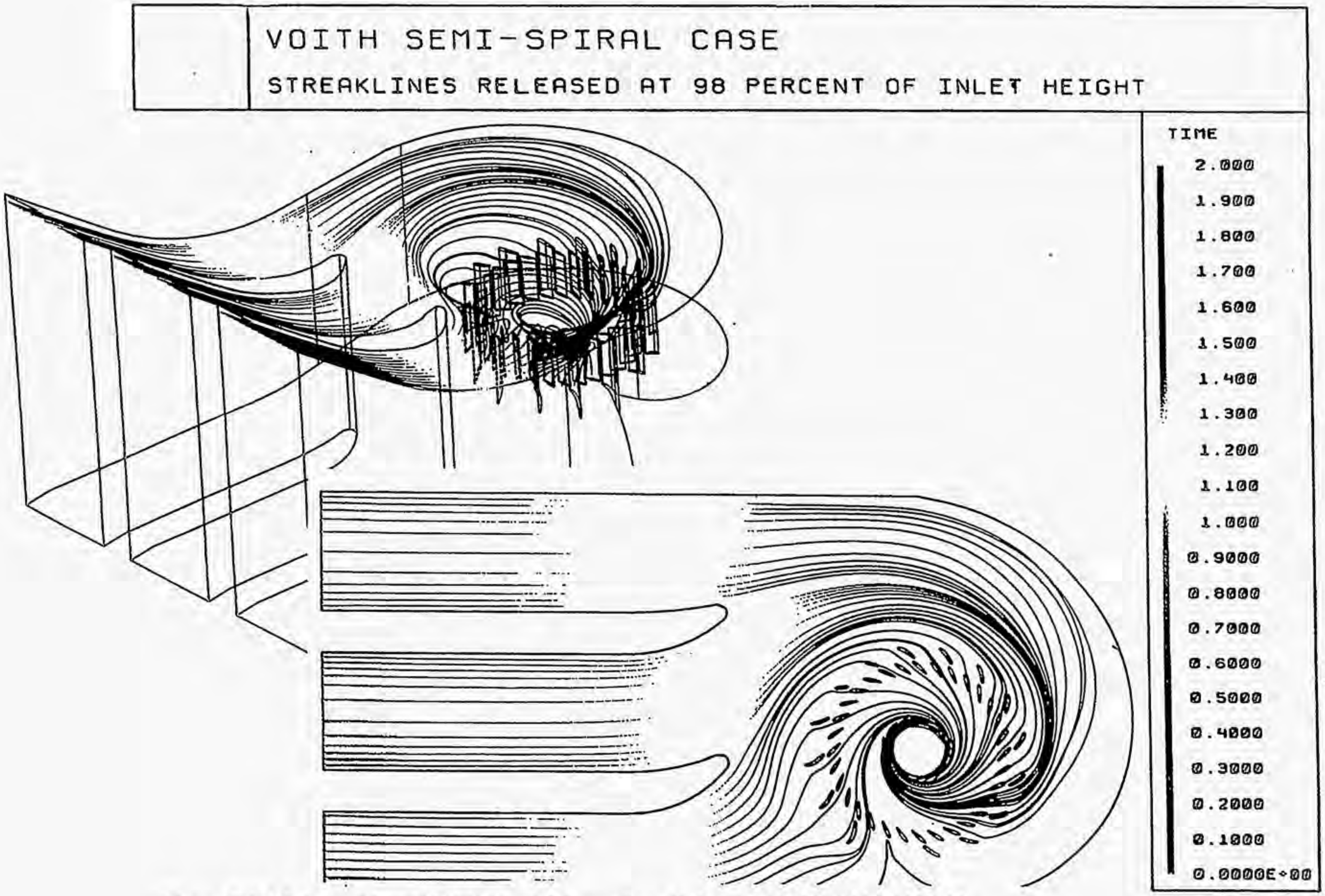

Figure 5.3.1-2 Isometric and Plan View Of Streamlines Released At 98% Of Inlet Height

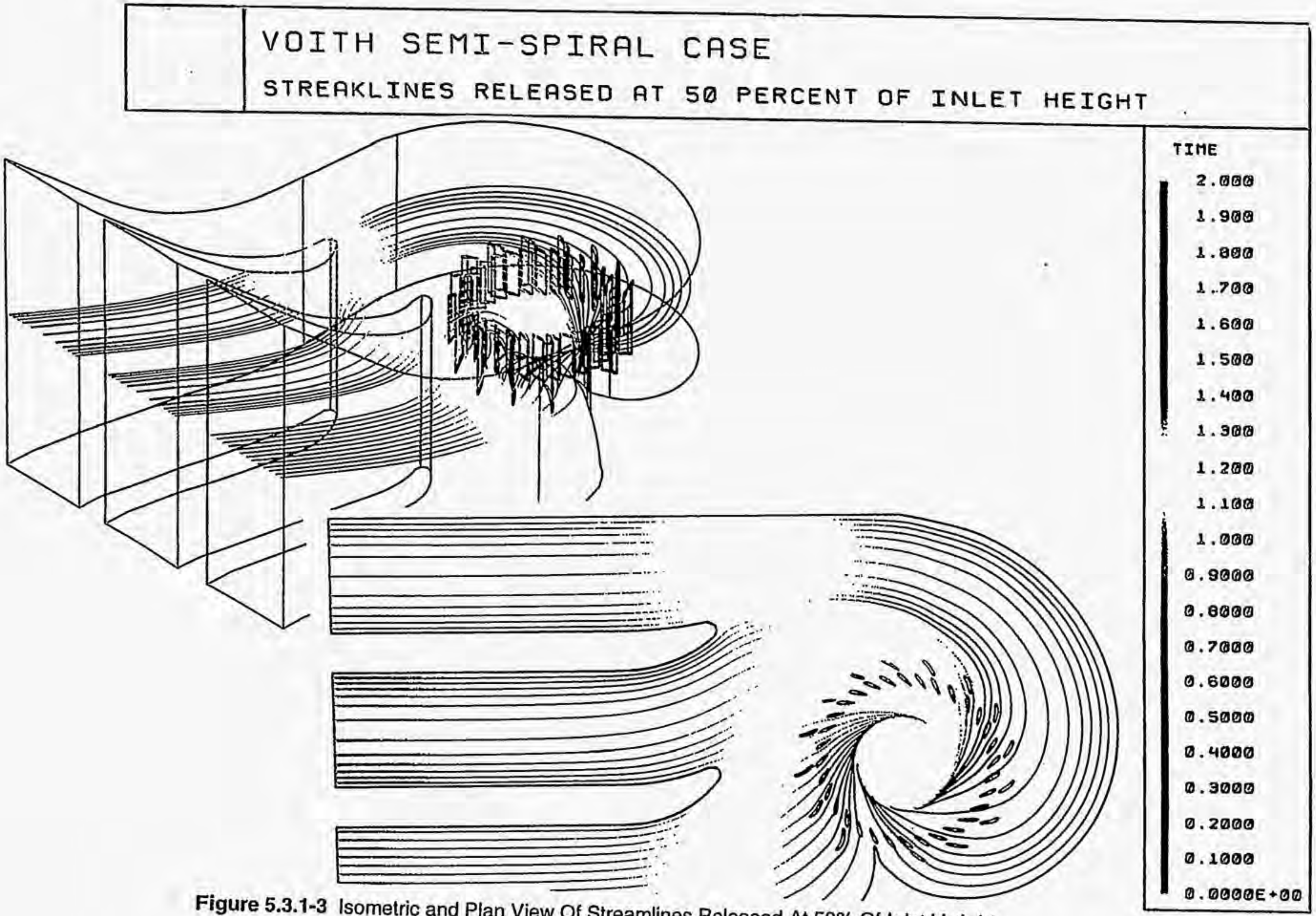

Figure 5.3.1-3 Isometric and Plan View Of Streamlines Released At 50% Of Inlet Height

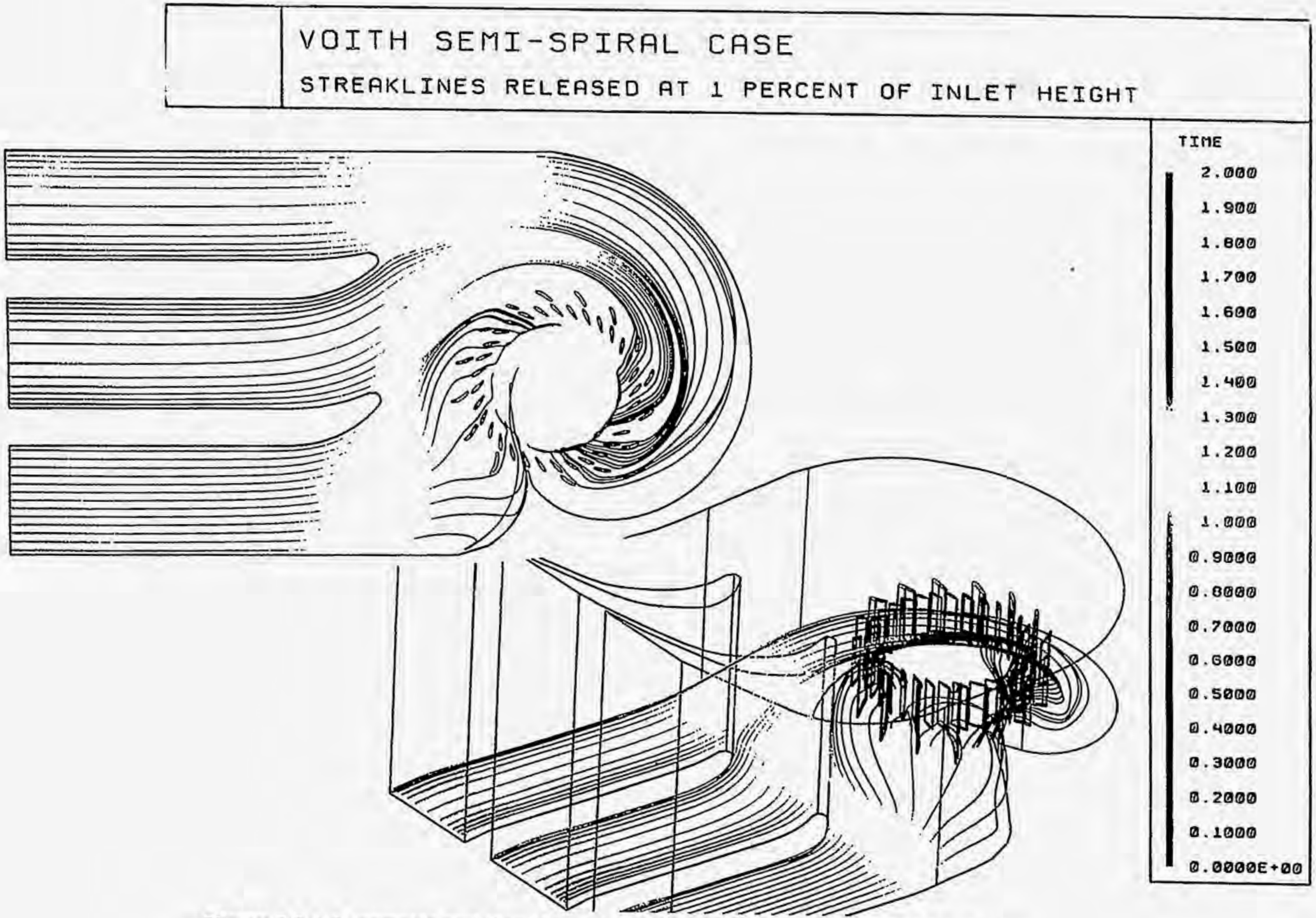

Figure 5.3.1-4 Isometric and Plan View Of Streamlines Released At 1% Of Inlet Height

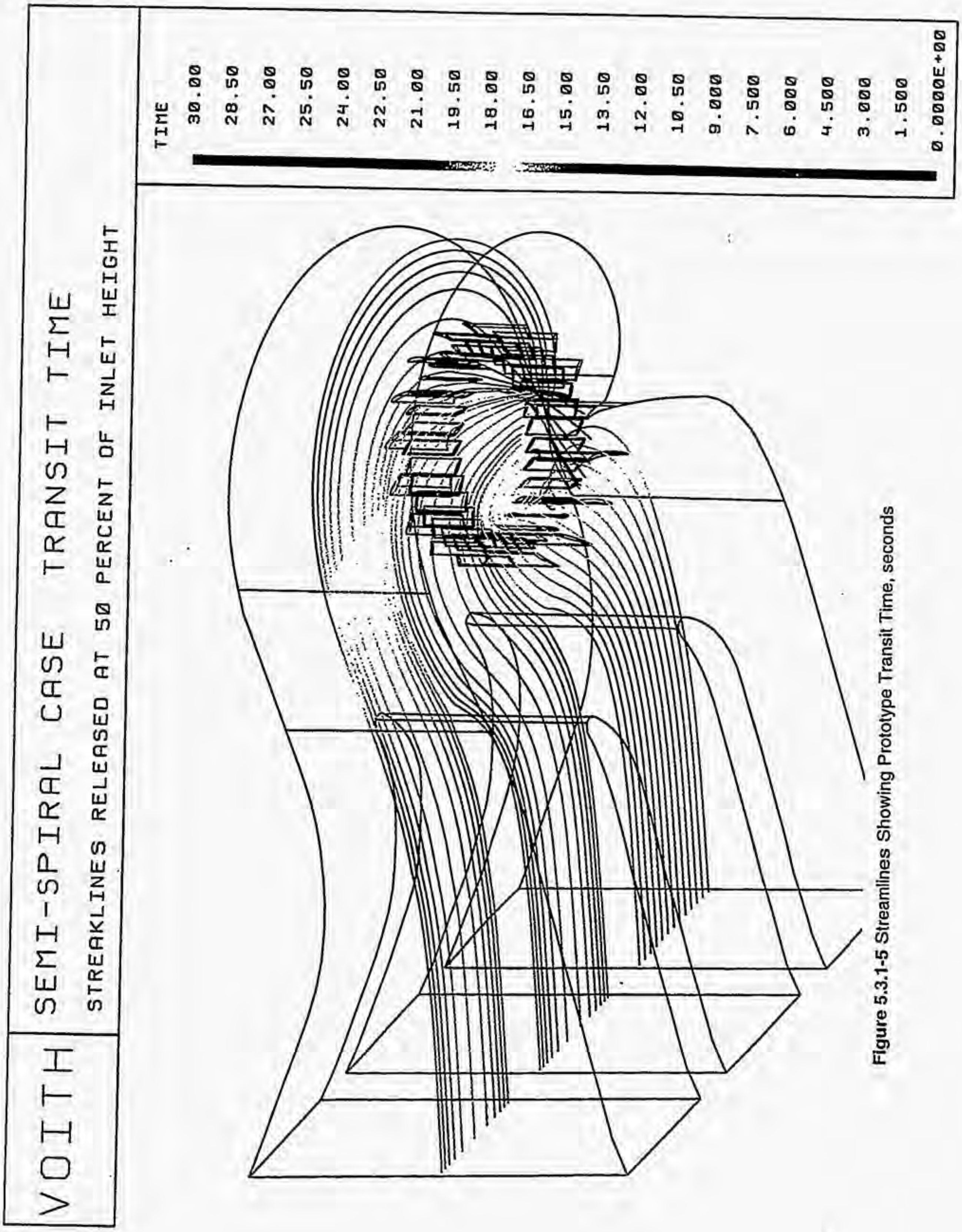

Figure 5.3.1-5 Streamlines Showing Prototype Transit Time, seconds

5.3.2 KAPLAN INTAKES WITH FISH SCREENS

Upon recognizing the significant influence that a fish screen would have on the internal intake flow, the previous grids were altered to include additional grids that could simulate a typical fish screen (from the Wanapum project) and to remove grids in the runner region. The runner grids were removed to prevent the large total pressure drop across the runner from overwhelming the relatively smaller total pressure loss due to the fish screen. The calculation outlet boundary was near the stay vane outer diameter. The fish screen region with more refined grids, shown on Figure 5.3.2-1 (note that this figure shows stay vane and wicket gate grids that were not used) was specified to have properties of a porous media to simulate the fish screen, or was specified to have no special properties to simulate the absence of the fish screen. The properties of the porous media were adjusted to cause a total pressure loss of 0.73 times the local velocity head. Subsequent analysis of model screen testing on the Bonneville project indicated that screen caused a loss of total pressure equal to 2.24 times the local velocity head. The current analysis therefore, has insufficient total pressure loss and would have less flow disruption than an equivalent Bonneville type screen. The diversion of flow through the gate slot was also modeled. 10% of the flow was specified to be discharged from the model at the gate slot location.

The results of the calculation are demonstrated by streamline plots. Plots are shown in a sequence of side and plan views with no fish screen present and with the fish screen active. Figures 5.3.2-2 and 5.3.2-3 show streamlines released at the inlet from 87% of the inlet height (100% is the top at the inlet). Note that the outline of the all grid blocks are shown in black, including the fish screen, even if the porous feature of the fish screen is not active. Streamlines near the roof are deflected significantly upwards due to the flow entering the gate slot. This 87% of the inlet height is approximately the dividing surface between flow entering the gate slot and flow entering the turbine. Streamlines released at the inlet from 30% of the intake height are shown in Figures 5.3.2-4 through 5.3.2-5. This height is approximately the dividing surface between flow passing beneath the fish screen and flow through the screen. These streamlines are affected by both the upward deflection of the flow entering the gate slot and a downward diversion of the flow that would tend to flow around the obstruction of the fish screen. The result is an approximate cancellation of these effects, and the 30% height streamline is approximately the dividing streamline both with a fish screen and in the absence of a fish screen. The presence of the fish screen affects the streamline paths for both release locations. While streamline paths in the absence of the fish screen terminate at a nearly constant height, the presence of the fish screen causes a redistribution of the flow, and subsequent variation in the location of the streamline paths. Presumably, fish trajectories would also be similarly affected. Velocity vectors and total pressure plots that also highlight the flow disturbance caused by the fish screen are shown on Figure 5.3.2-6 while Figure 5.3.2-7 compares streamlines released behind the fish screen with and without the fish screen.

Flow disturbances that the fish screen generate which could alter the details of flow in the stay vane, wicket gate, or runner could not be analyzed with this calculation. Such a calculation would require a grid fine enough to model the turbulent mixing behind the fish screen. The calculation model used 300,000 nodes. A model of this size was nearly impossible several years ago and even today requires an advanced workstation.

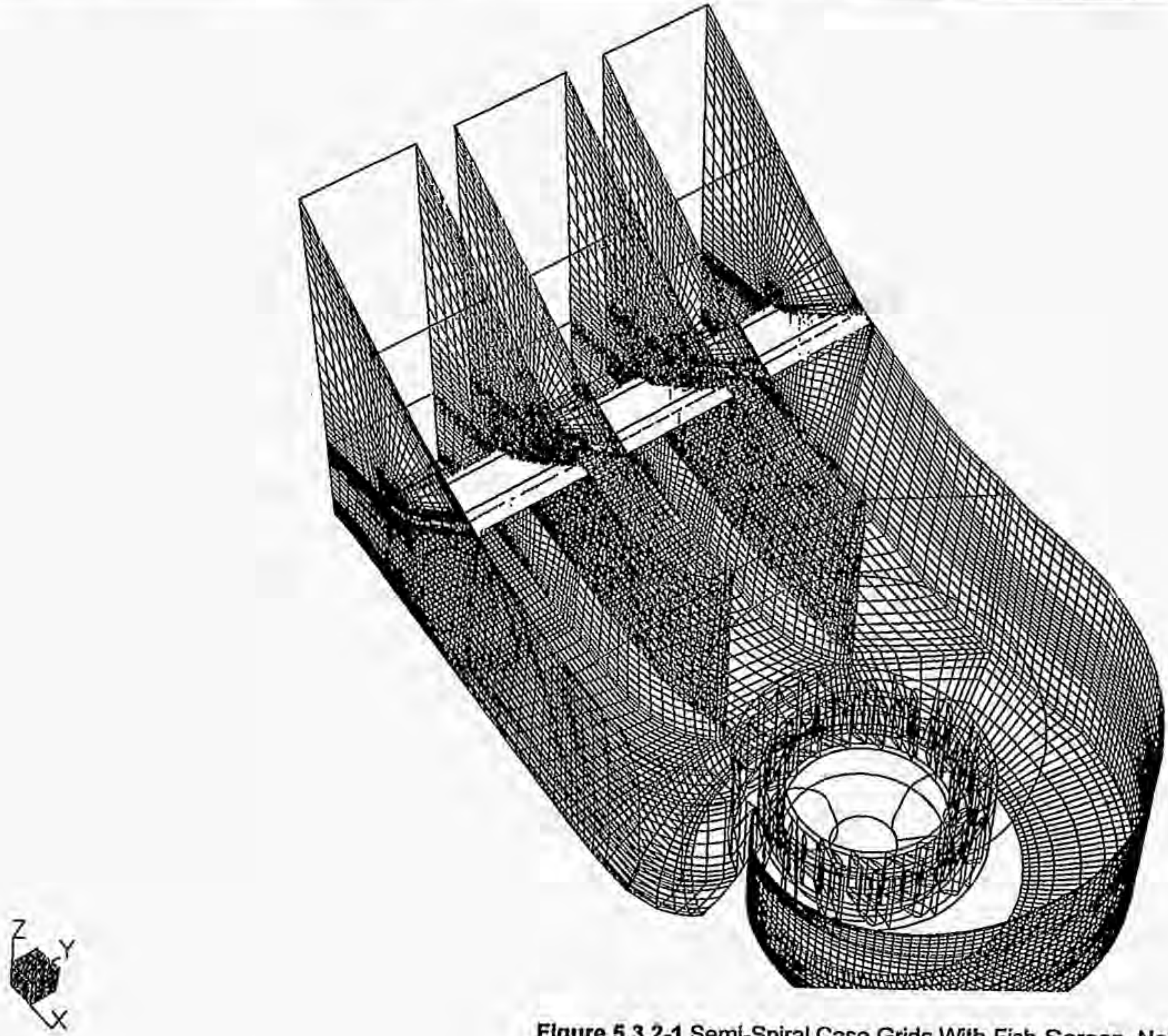

Figure 5.3.2-1 Semi-Spiral Case Grids With Fish Screen. Note That Stay Vane And Wicket Gate Grids Were Not Used.

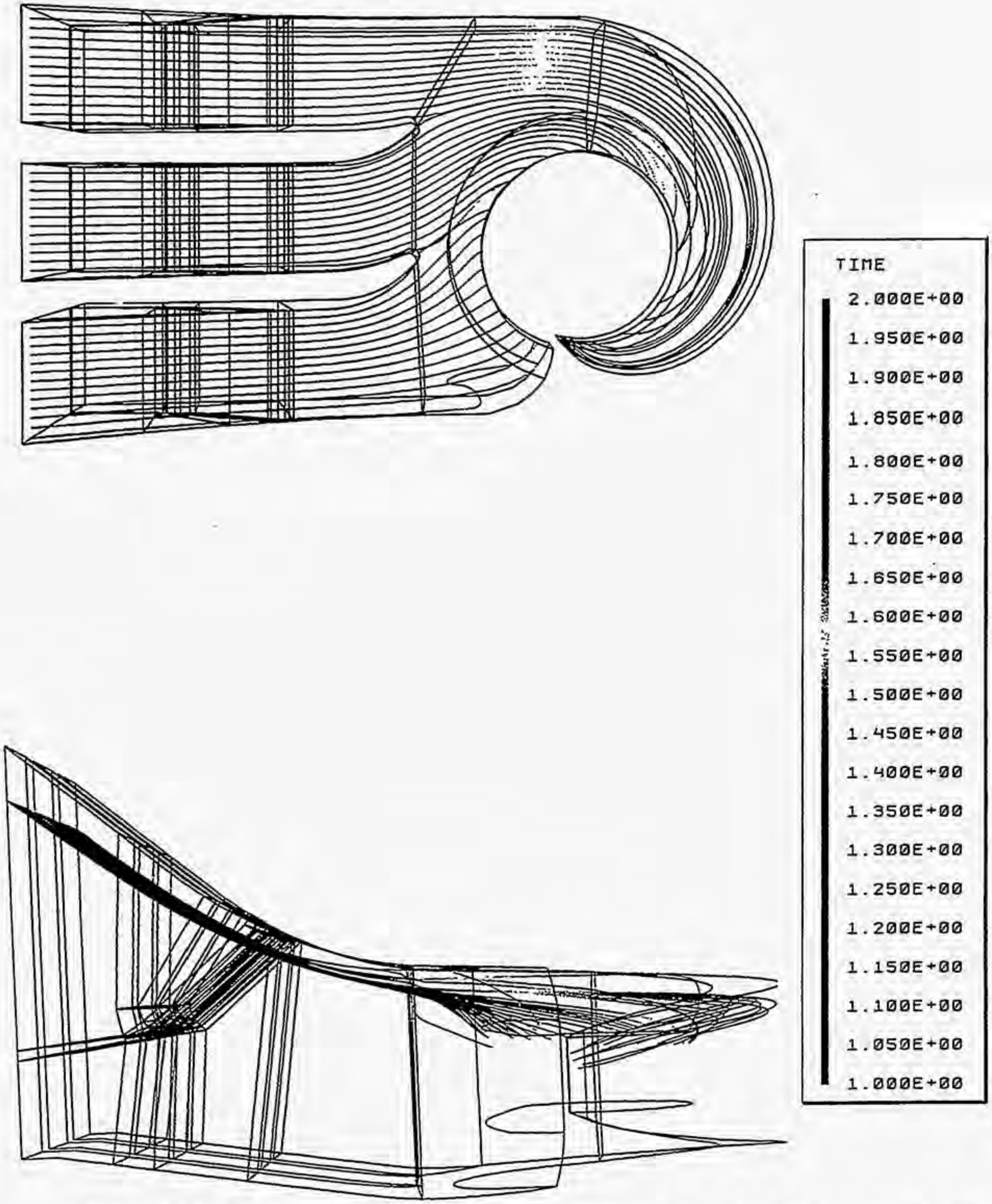

Figure 5.3.2-2 Plan and Side View of Streamlines Released From 87% Of Inlet Height. Fish Screen Not Activated.

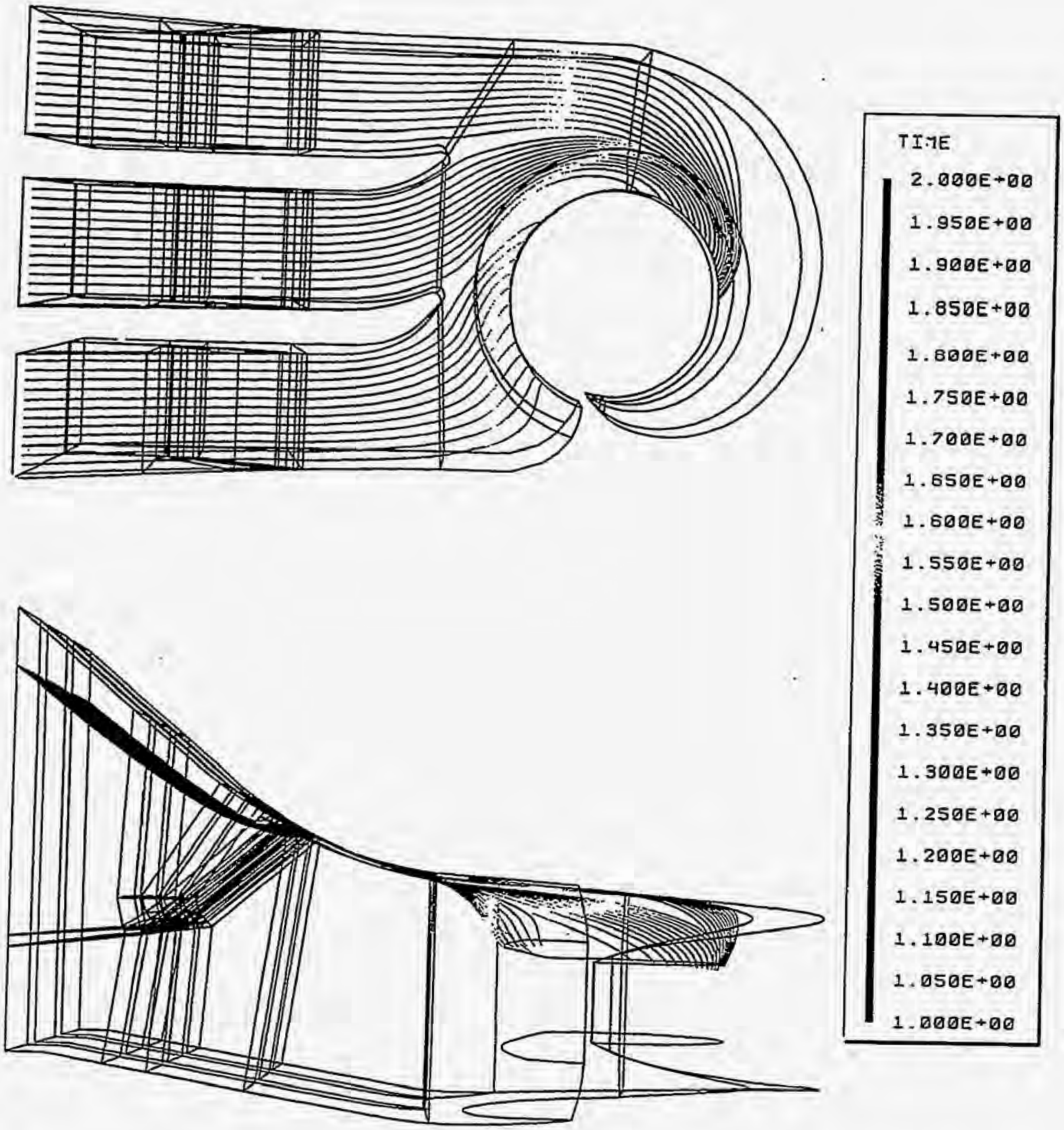

Figure 5.3.2-3 Plan and Side View of Streamlines Released From 87% Of Inlet Height. With Fish Screen Activated.

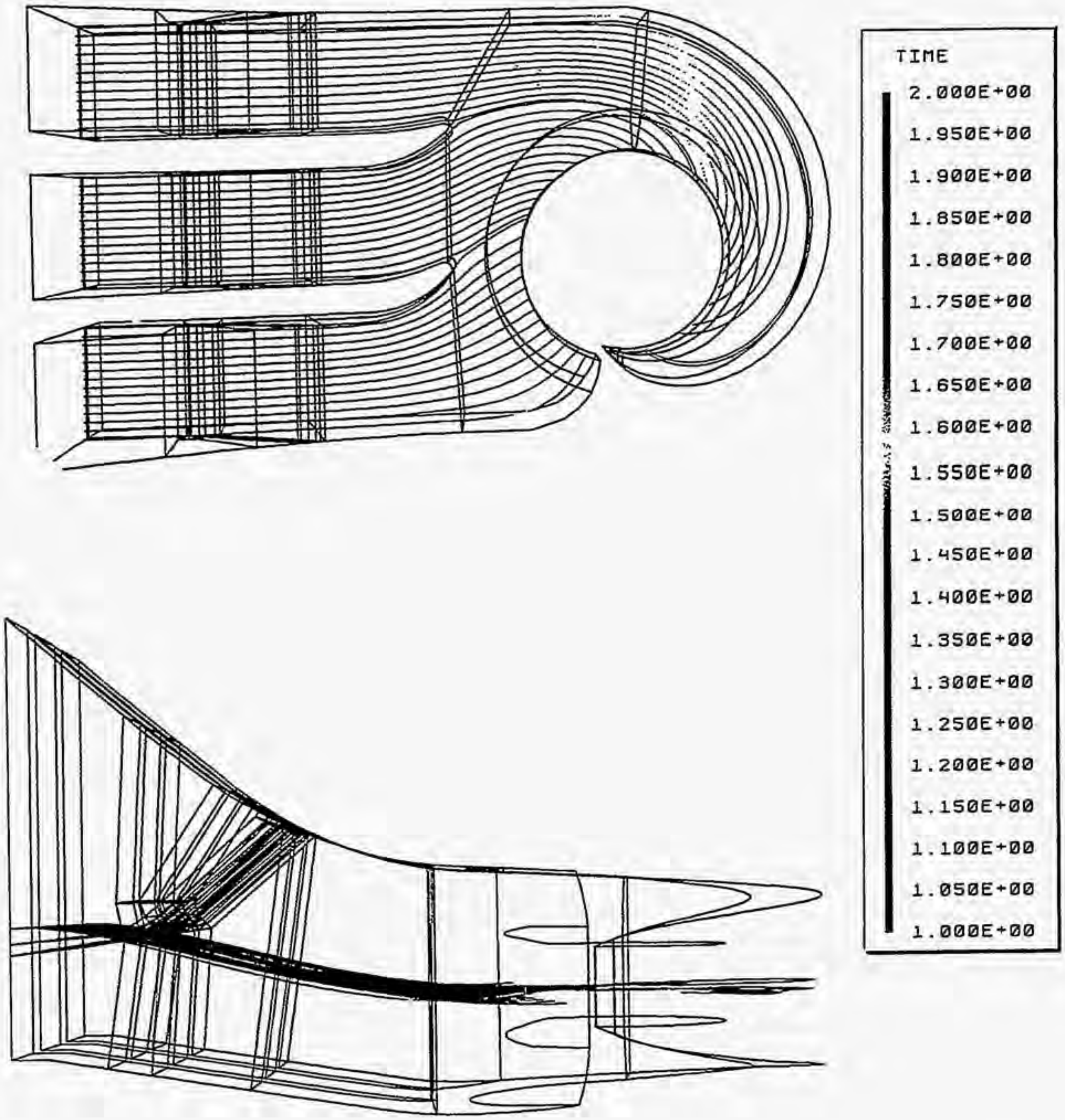

Figure 5.3.2-4 Plan and Side View of Streamlines Released From 30% Of Inlet Height. Fish Screen Not Activated.

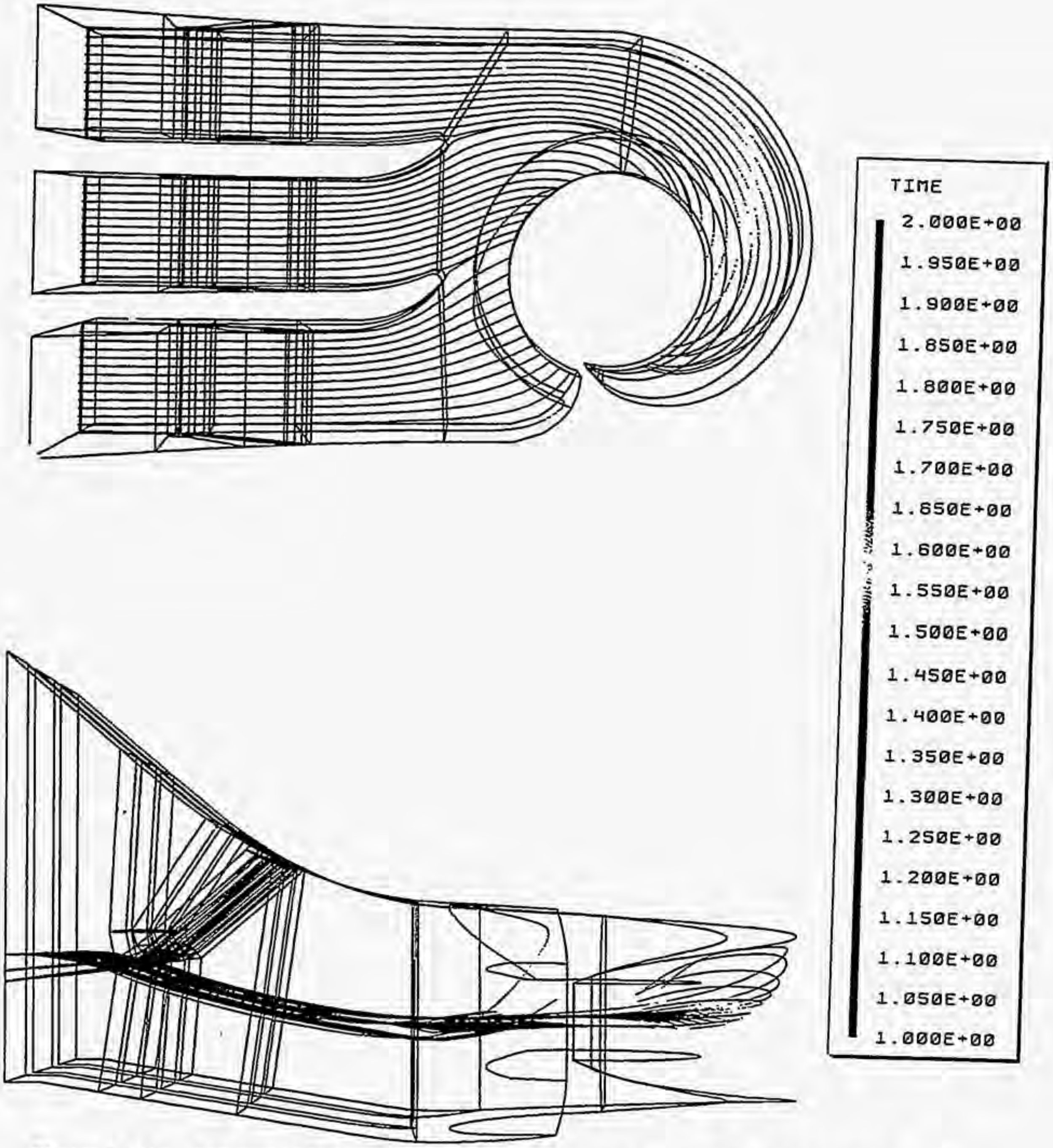

Figure 5.3.2-5 Plan and Side View of Streamlines Released From 30% Of Inlet Height. With Fish Screen Activated.

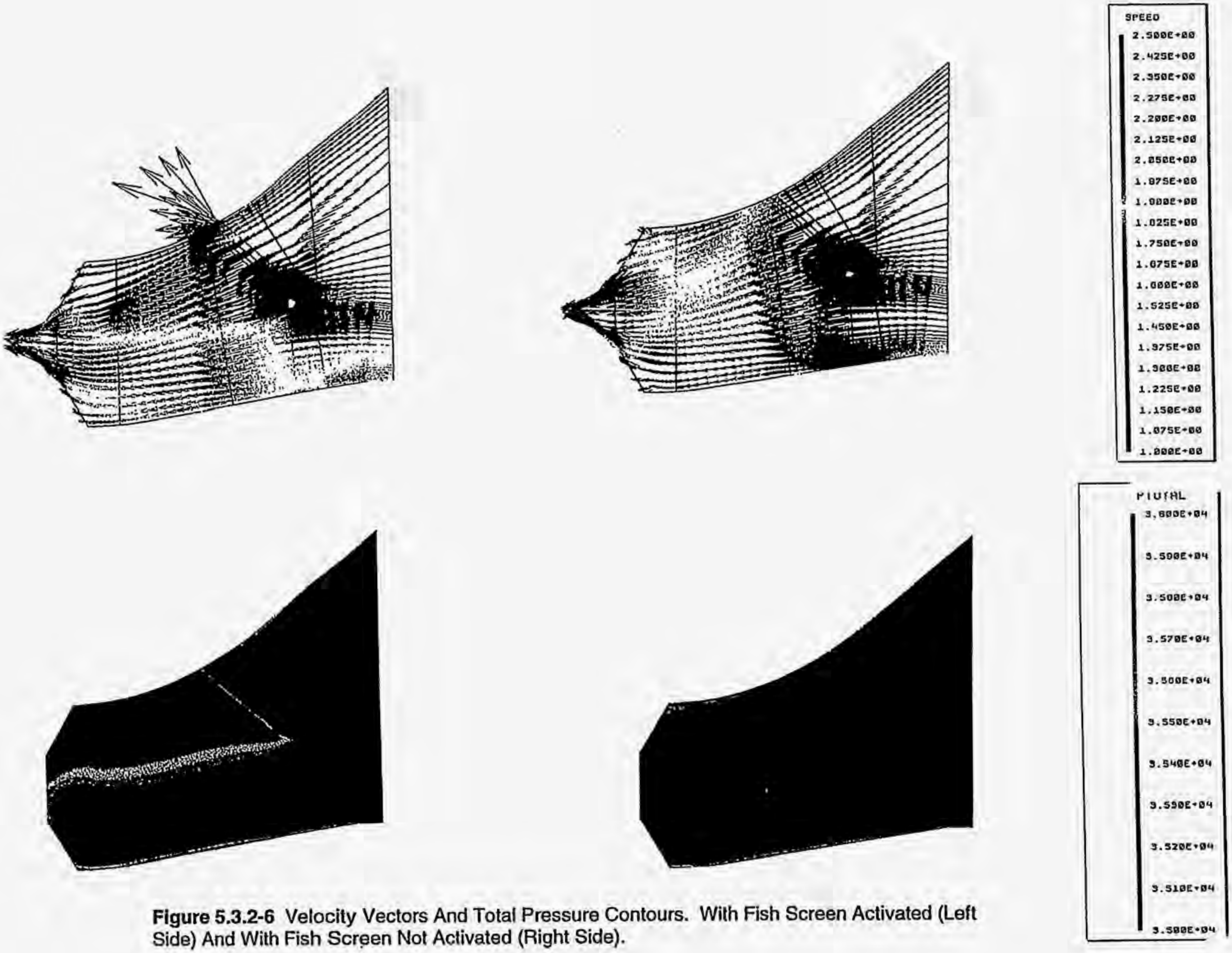

Figure 5.3.2-6 Velocity Vectors And Total Pressure Contours. With Fish Screen Activated (Left Side) And With Fish Screen Not Activated (Right Side).

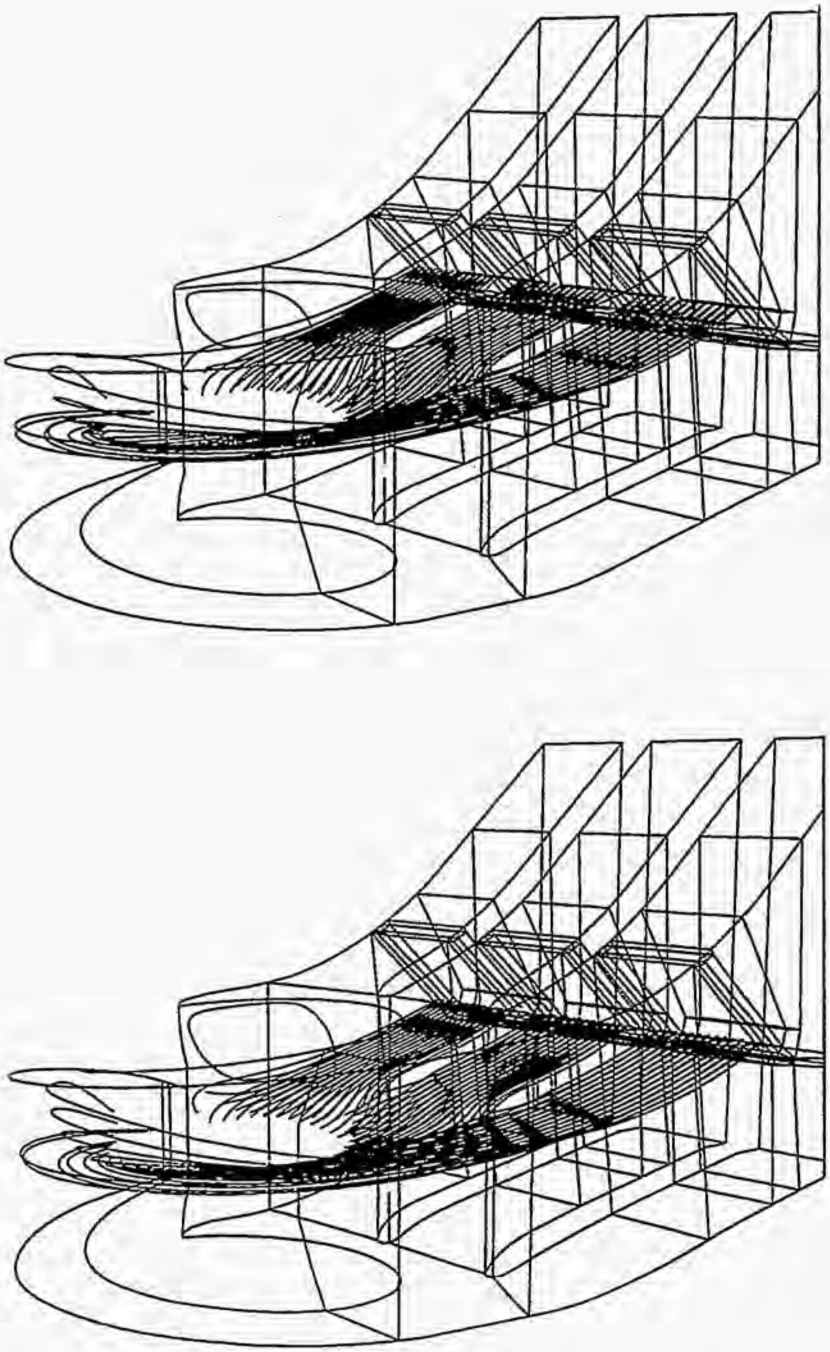

Figure 5.3.2-7 Streamlines Released Behind Fish Screen At 50% Height (Top) And 30% Height (Bottom). Green Indicates Fish Screen Activated, Red Indicates Fish Screen Not Activated.

5.3.3 STAY VANES AND WICKET GATES

Kaplan and Francis Turbines

The flow field in stay vanes and wicket gates and related fish damage mechanisms are generally the same for Kaplan and Francis turbines. One modest difference between typical Kaplan and typical Francis turbine stay vanes and wicket gates is related to the difference between a spiral case and a semispiral case. Streamline tracings in a spiral case, as is more commonly used for Francis turbines, show that initial position in the spiral case inlet does not coincide in a clear manner to a definite position at the runner inlet (Figure 5.3.3-1).

Stay vanes and wicket gates are analyzed from the spiral case centerline to approximately the runner entrance. Periodic boundaries are used assuming that all flow channels are identical and therefore, only one flow channel needs to be analyzed. A typical grid of the water passage wetted surface is shown on Figure 5.3.3-2. Velocity fields, shear magnitude and pressure gradients are analyzed. Two cases were studied.

A gate and vane that are not optimized was analyzed. This gate and vane were existing shapes that were encountered during a turbine rehabilitation project (Wanapum). The velocity and shear field are shown in Figures 5.3.3-3 and 5.3.3-4. In general, the result is unremarkable. At the stay vane leading edge, velocities are low enough to prevent the occurrence of critical shear values, as shown in a magnified view of the stay vane leading edge, Figures 5.3.3-5 and 5.3.3-6. Shear values greater than the critical value were found at the stay vanes trailing edge and at the wicket gate leading edge, Figures 5.3.3-7 and 5.3.3-8. However, the region having shear greater than the critical shear (450 /s) is quite small. Pressure gradients were small, varying from zero to approximately 500 psi / sec. As these local pressure gradients are presumed to have little effect on fish survival, figures are not provided.

To illustrate the effect of design modifications on flow field characteristics, improvements were made to the shape of both the stay vanes and wicket gates. The stay vane shape was modified to reduce the angle of attack and the wicket gate shape was also changed. The flow field analyzed in this case is not precisely the same as the first case. At a different circumferential location in the semi-spiral case, the inflow to the stay vanes is more tangential. The velocity and shear field are shown in Figures 5.3.3-9 and 5.3.3-10. In general, the flow field is smoother and values of shear are reduced. Values of critical shear no longer occur on the gate leading edge. Only a smaller region of critical shear is observed near the stay vane trailing edge, Figures 5.3.3-11 and 5.3.3-12. This optimized gate and vane was found (through laboratory testing) to increase turbine efficiency.

The wicket gate openings analyzed above were chosen to avoid large openings where the gate overhangs the bottom ring. At higher gate openings, a gap is created at the bottom of the gate, allowing a leakage flow to develop. No CFD calculations were made to verify the unfavorable flow field that is presumed to exist at the higher gate opening.

In summary, the risk of fish mortality for optimized stay vanes seems low for Kaplan turbines at typical heads, and a good indicator of fish friendliness is a hydraulic design that gives good efficiency. Higher velocities on the wicket gates may have greater potential for injury. Although not analyzed, other gate openings that occur at different operating conditions will cause different flow fields and will have different mortality. In general, it seems that good turbine designs that optimize turbine efficiency are beneficial for fish survival.

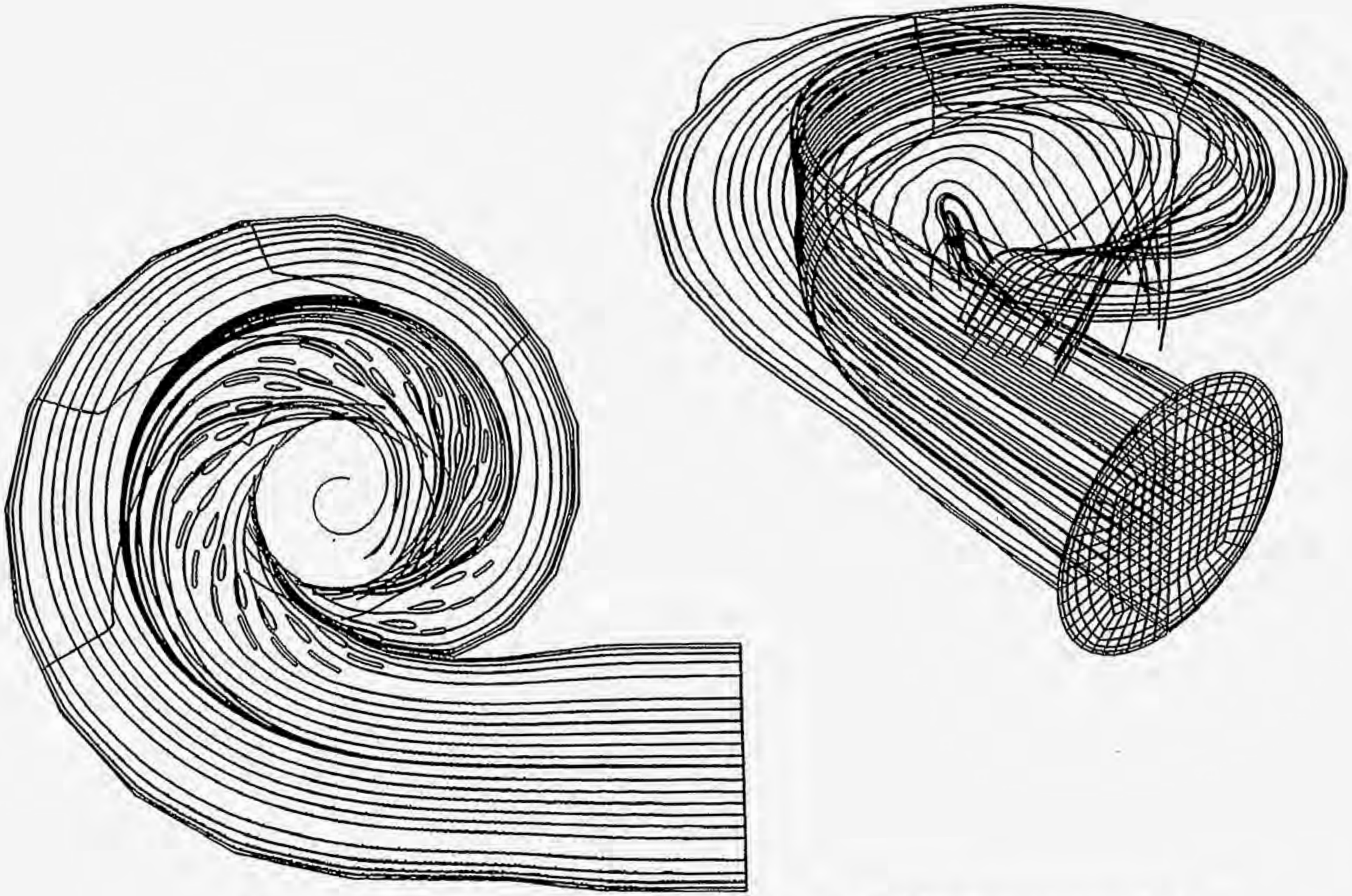

Figure 5.3.3-1 Streamlines in a Spiral Case. Initial Position Does Not Correspond to a Particular Runner Inlet Position

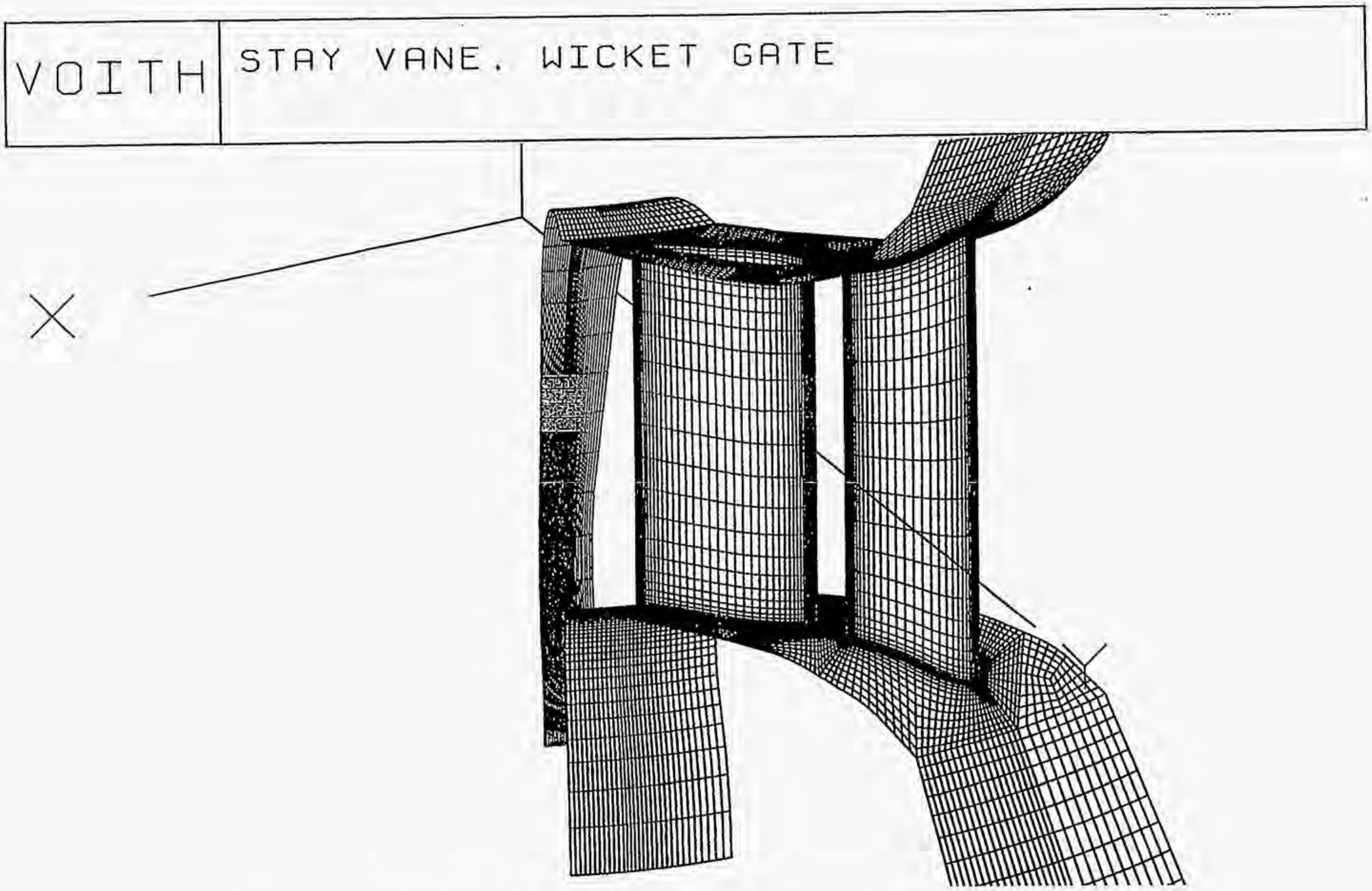

Figure 5.3.3-2 Typical Grid Showing Wetted Surfaces Of A Typical Stay Vane And Wicket Gate

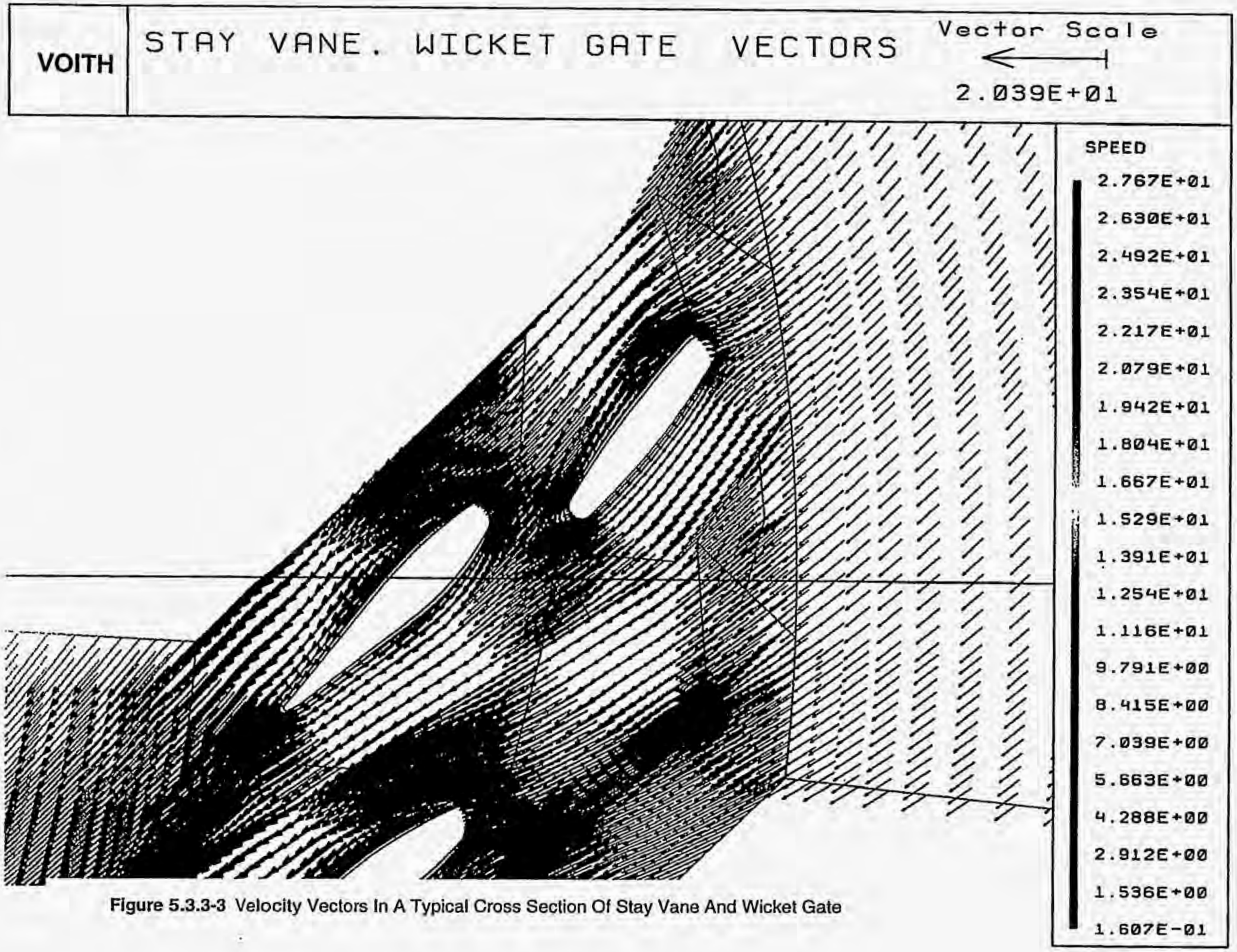

Figure 5.3.3-3 Velocity Vectors In A Typical Cross Section Of Stay Vane And Wicket Gate

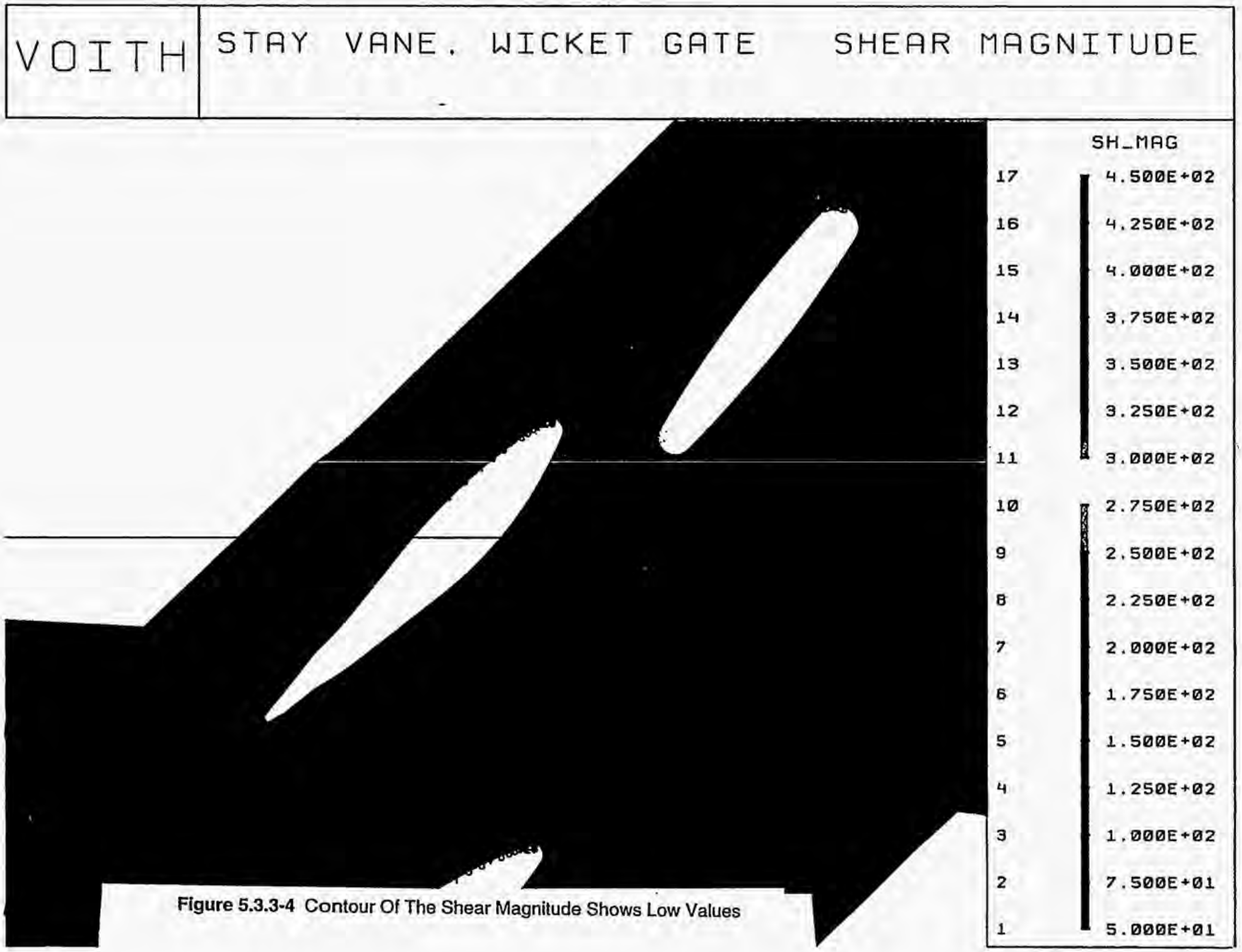

Figure 5.3.3-4 Contour Of The Shear Magnitude Shows Low Values

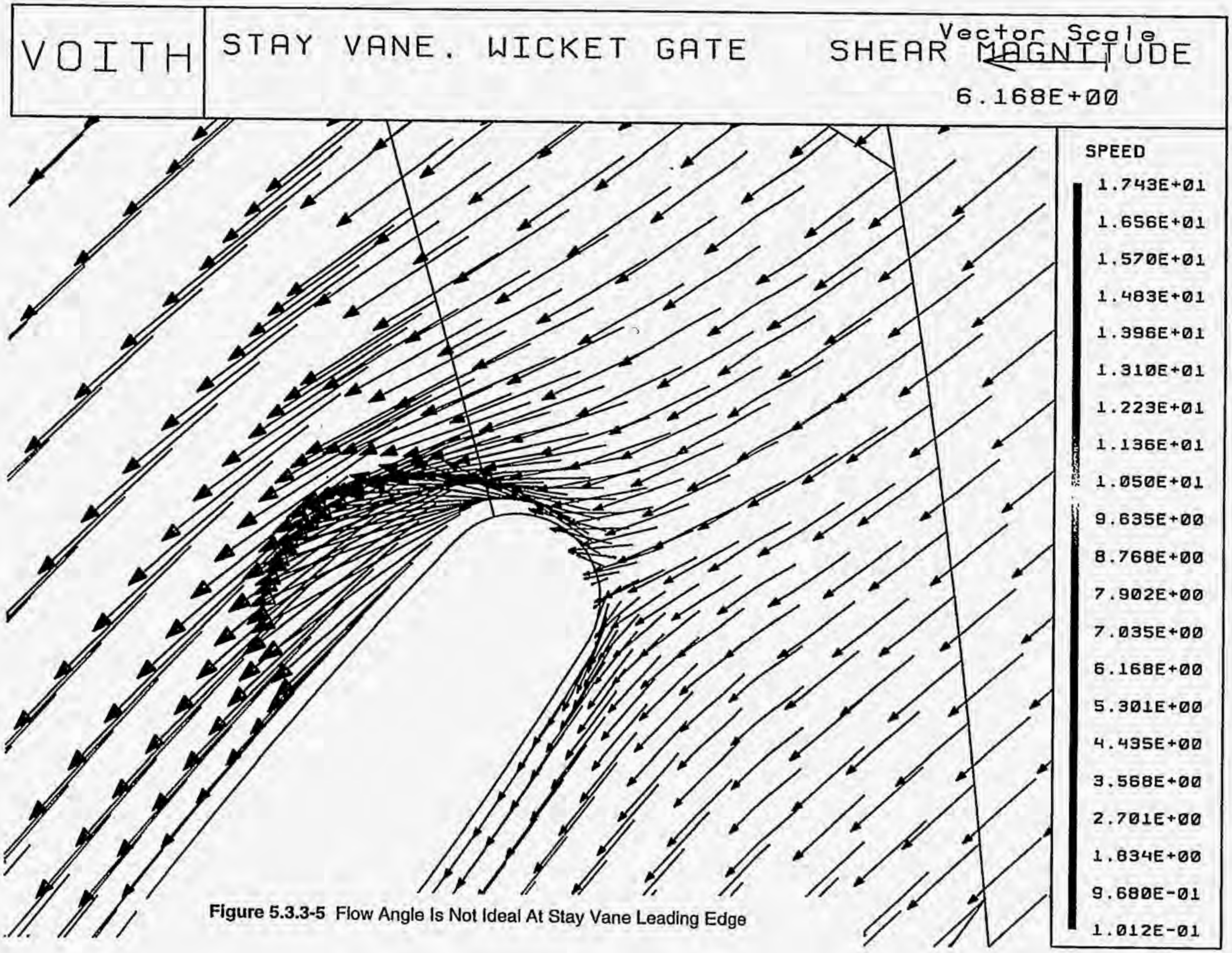

Figure 5.3.3-5 Flow Angle Is Not Ideal At Stay Vane Leading Edge

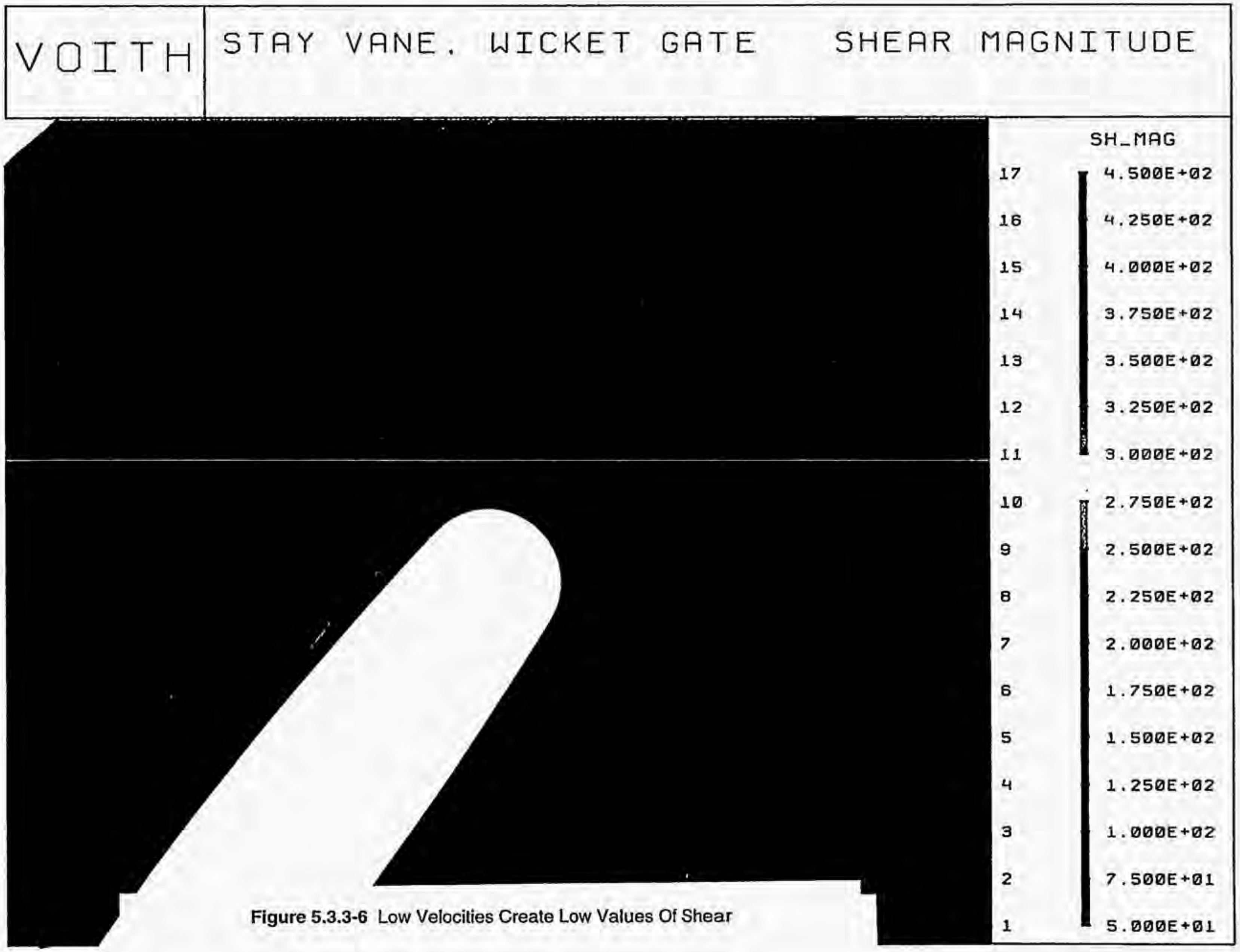

Figure 5.3.3-6 Low Velocities Create Low Values Of Shear

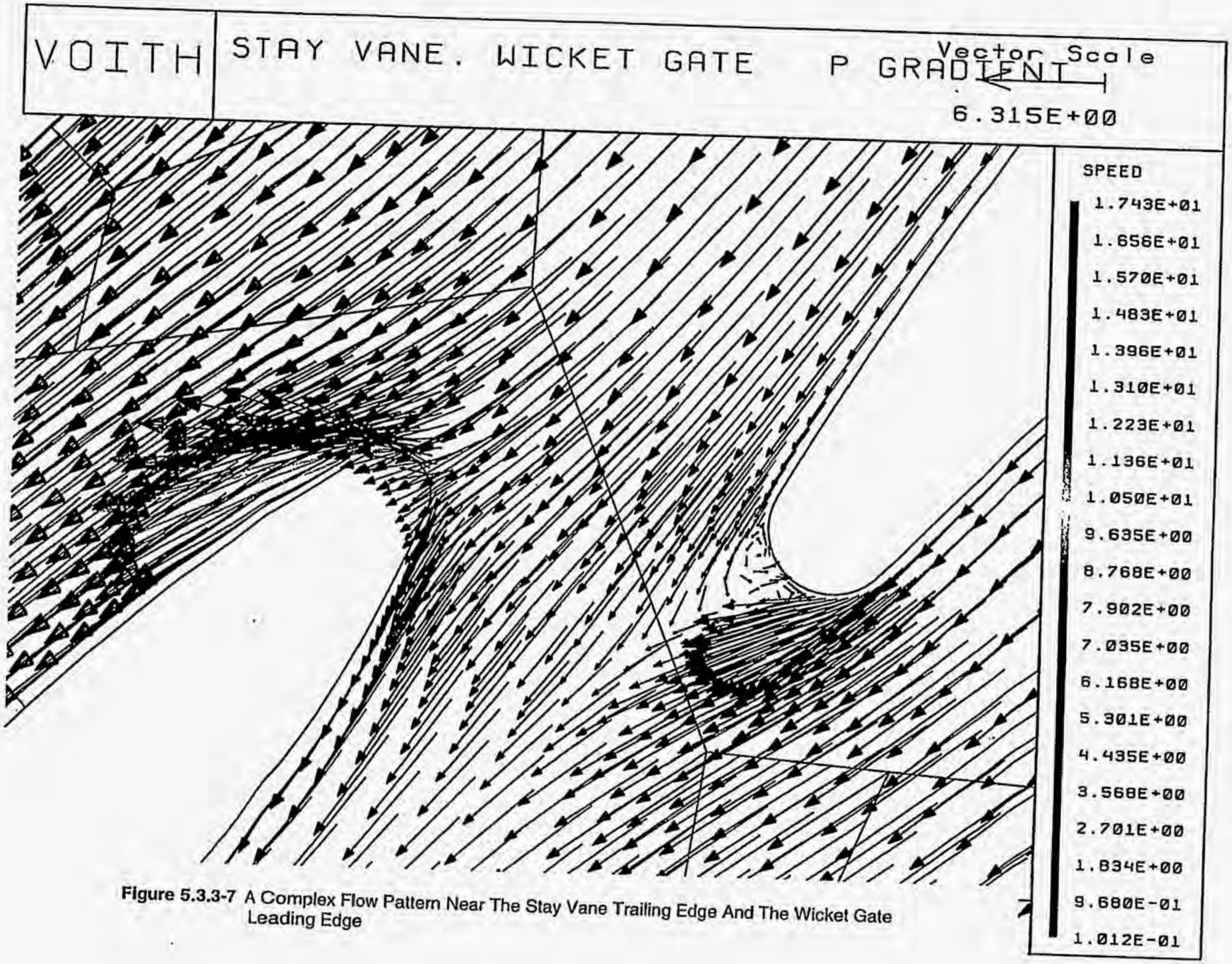

Figure 5.3.3-7 A Complex Flow Pattern Near The Stay Vane Trailing Edge And The Wicket Gate Leading Edge

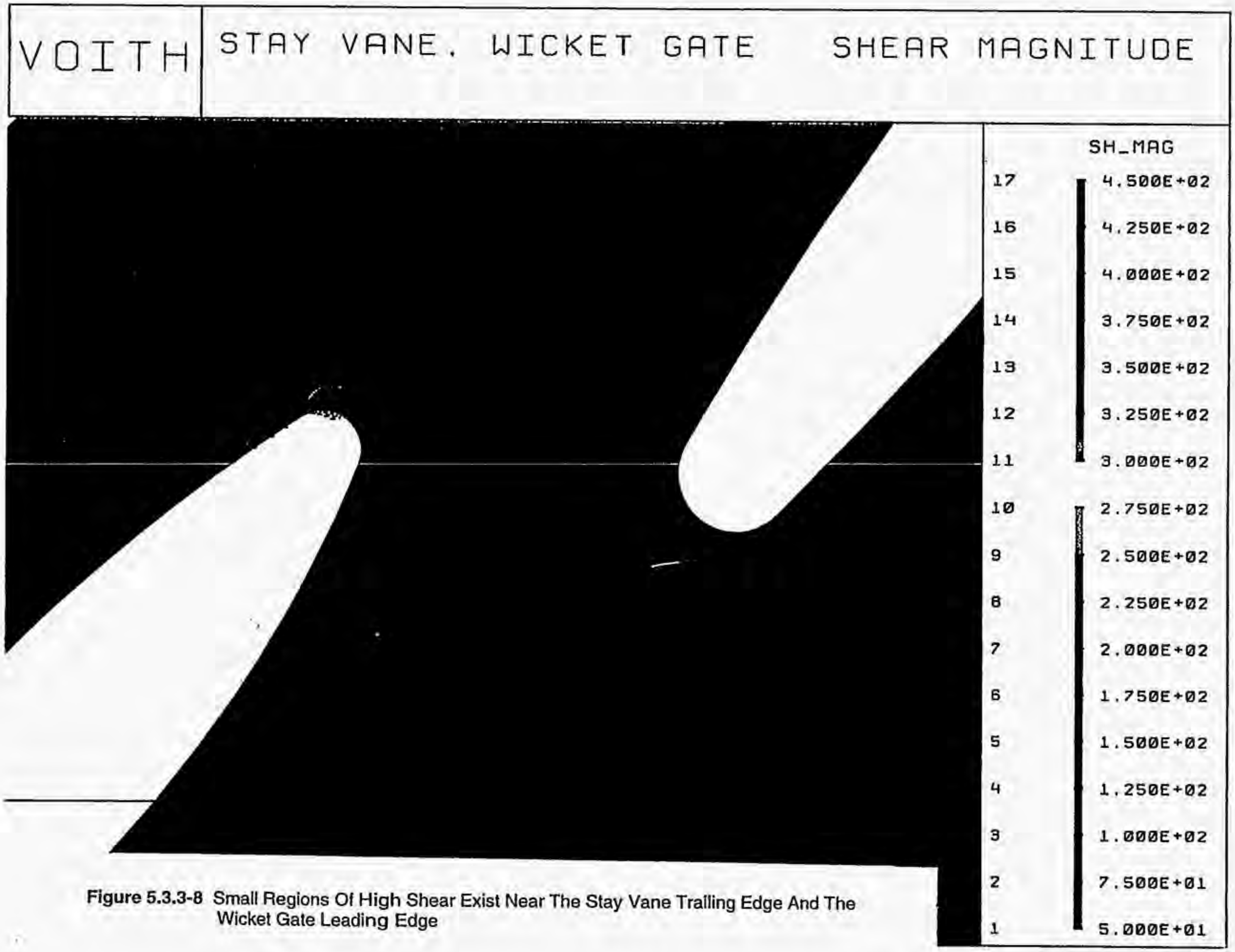

Figure 5.3.3-8 Small Regions Of High Shear Exist Near The Stay Vane Trailing Edge And The Wicket Gate Leading Edge

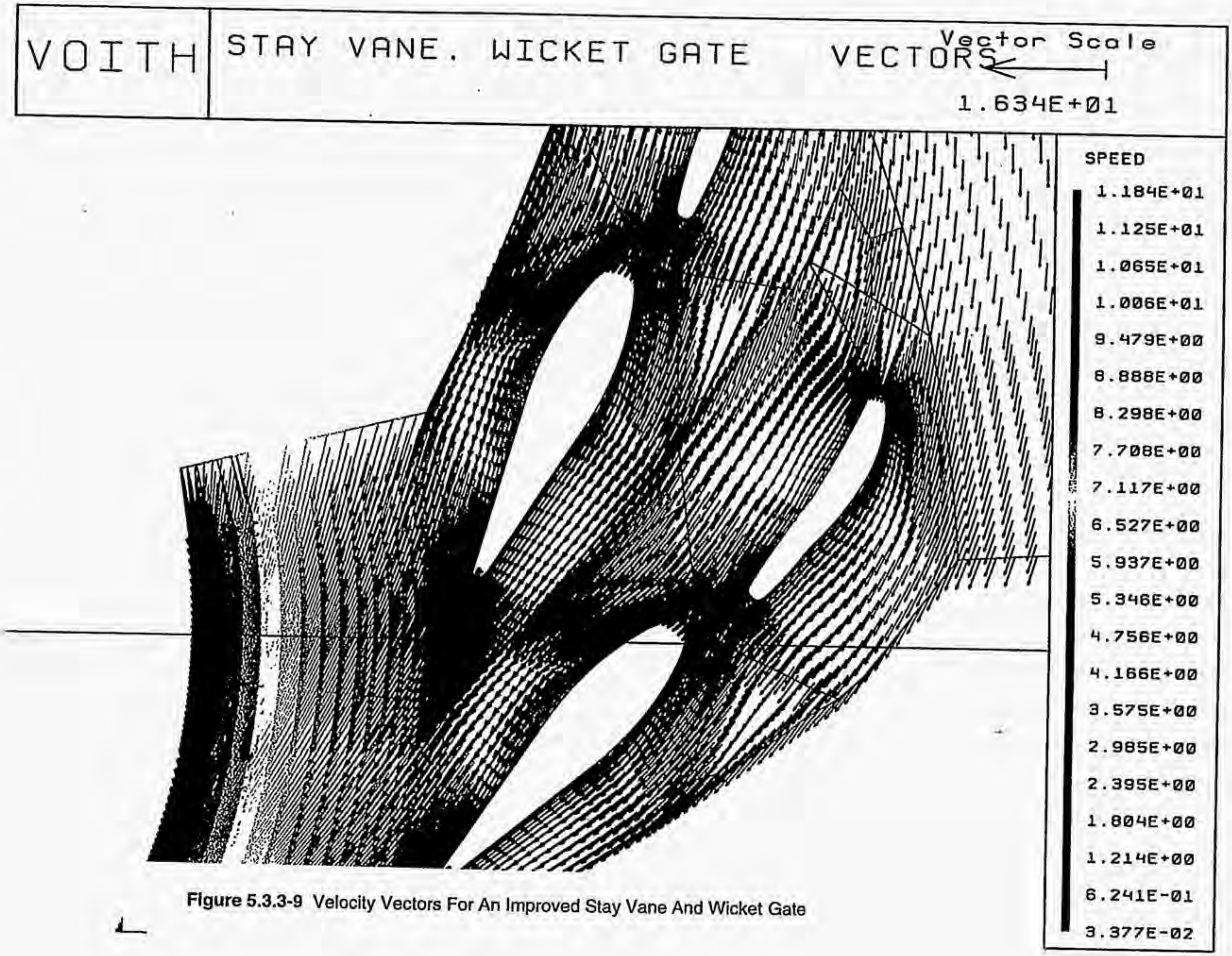

Figure 5.3.3-9 Velocity Vectors For An Improved Stay Vane And Wicket Gate

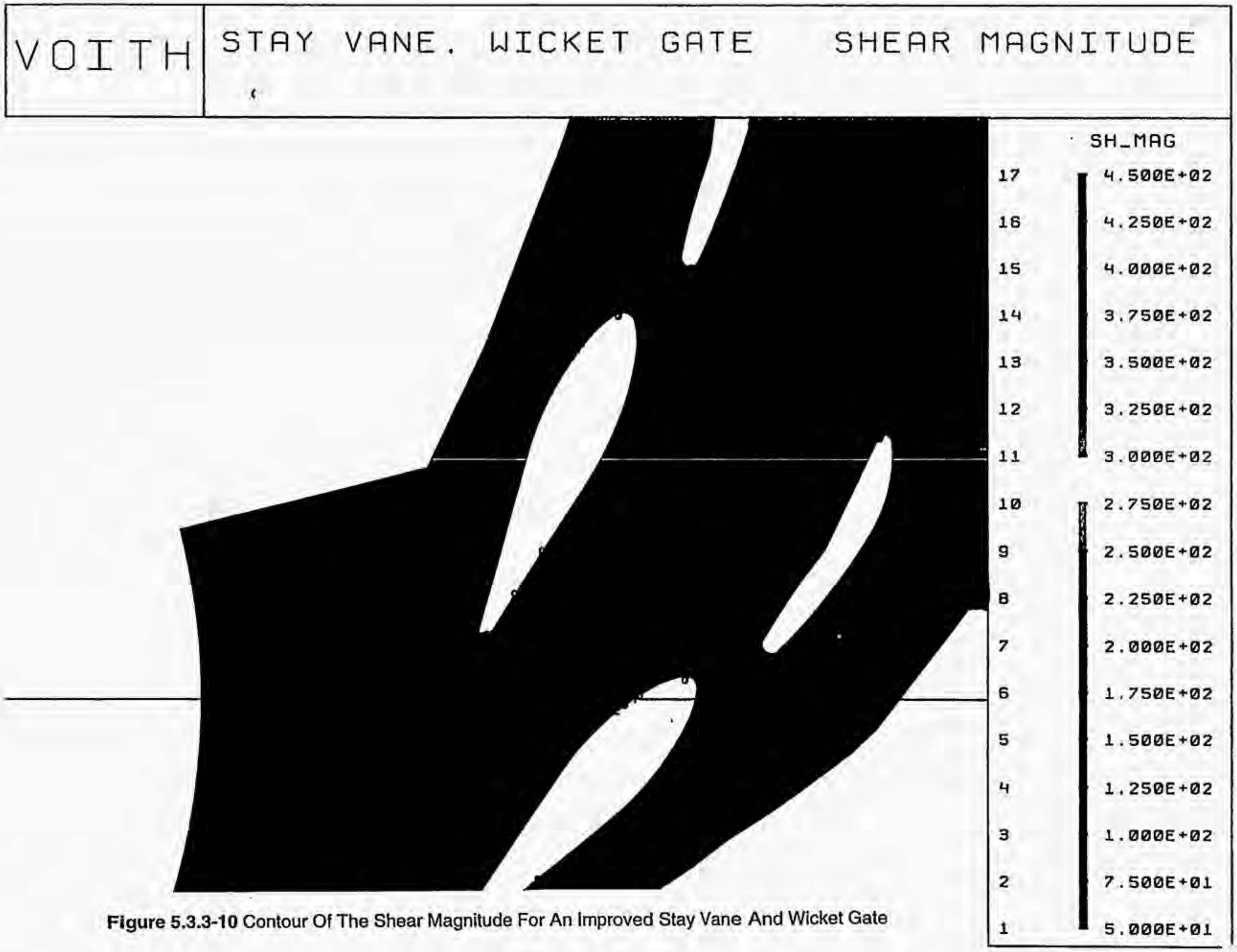

Figure 5.3.3-10 Contour Of The Shear Magnitude For An Improved Stay Vane And Wicket Gate

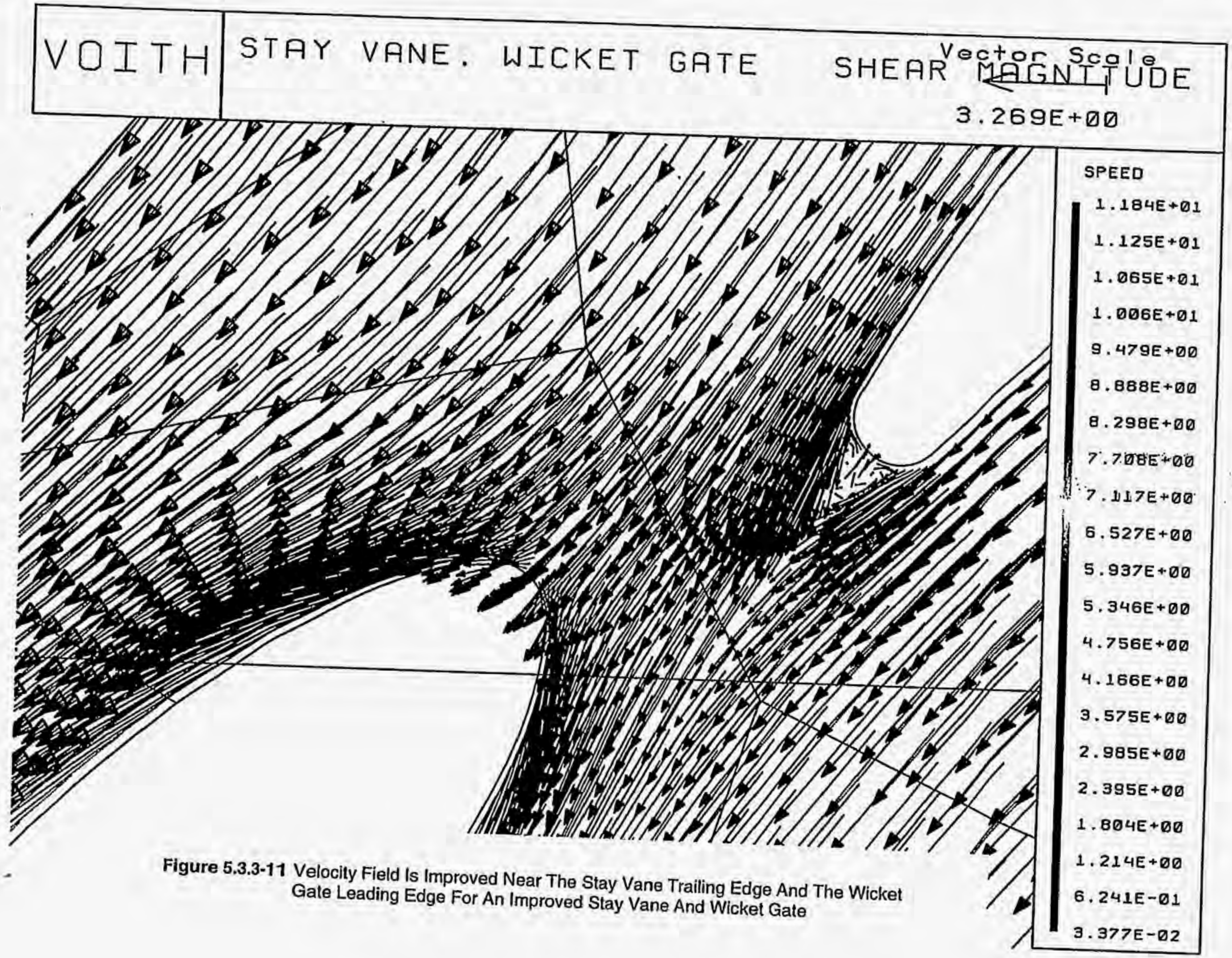

Figure 5.3.3-11 Velocity Field Is Improved Near The Stay Vane Trailing Edge And The Wicket Gate Leading Edge For An Improved Stay Vane And Wicket Gate

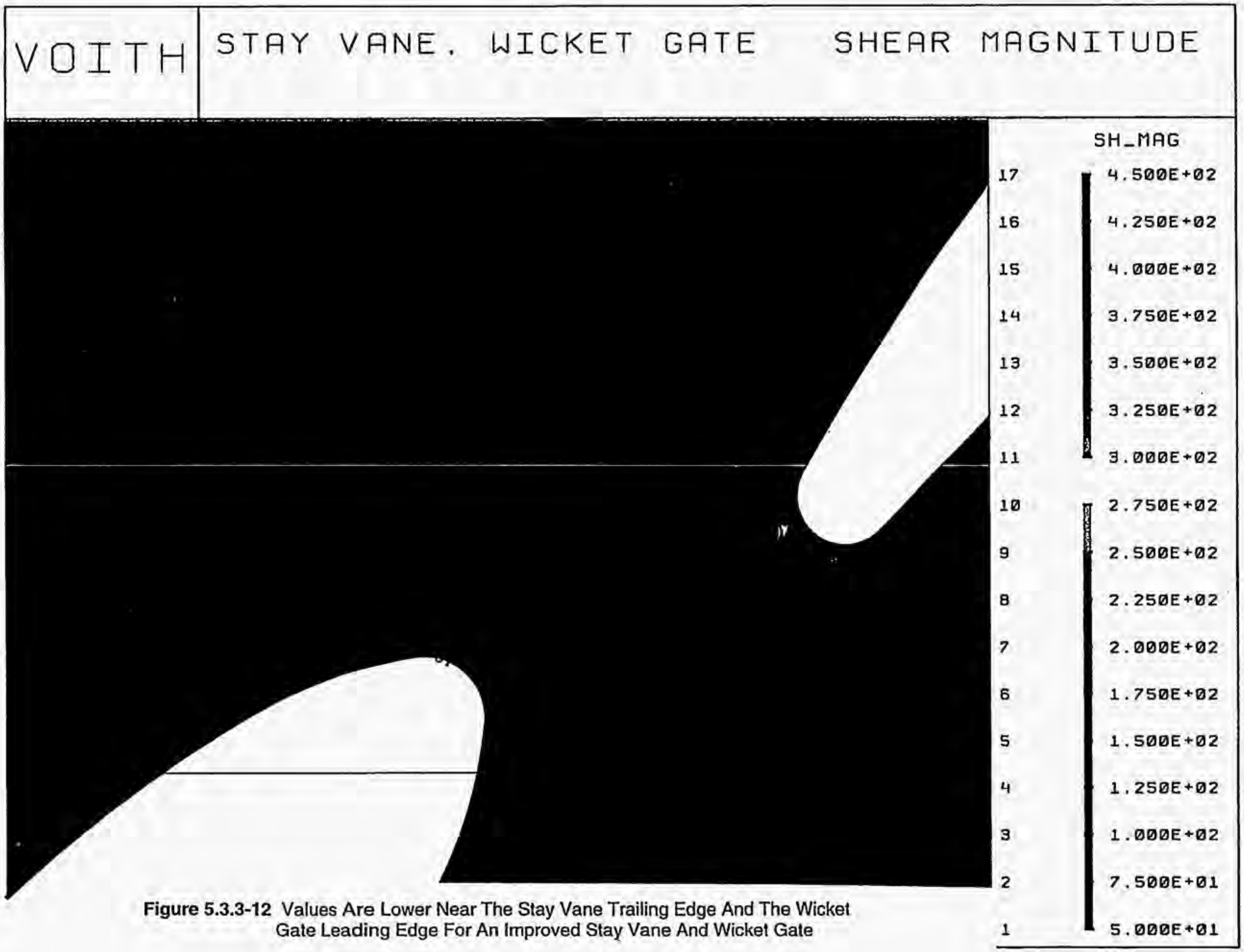

Figure 5.3.3-12 Values Are Lower Near The Stay Vane Trailing Edge And The Wicket Gate Leading Edge For An Improved Stay Vane And Wicket Gate

5.3.4 KAPLAN TURBINE RUNNER

Kaplan turbine runner blades were analyzed at an on cam condition. The inflow condition to the runner is obtained from a stay vane and wicket gate analysis. Periodic boundary conditions are used so that only one flow channel needs to be analyzed. The runner blade extended completely from hub to periphery, without any gaps or clearances.

Figure 5.3.4-1 shows a typical grid as well as streamlines that have been colored to show prototype transit time. Streamlines typically exit the runner in approximately 0.2 to 0.3 sec.

The prediction of cavitation is important for avoiding fish injury as well as for achieving good operating characteristics. Figure 5.3.4-2 shows a prediction of cavitation on the suction side of the blade and 5.3.4-3 is a prediction for the pressure side. The results are presented such that the pressure reaches vapor pressure if the values shown reach the plant sigma parameter. Based on a plant sigma of 0.7, this blade shape is not ideal as cavitation would occur on both pressure side, on a small region near the periphery leading edge, as well as on larger regions of the suction side.

Two locations through the runner blade were analyzed to study fluid-induced damage mechanisms, near the periphery, and near the hub. Figures 5.3.4-4 through 5.3.4-10 present velocity, shear, pressure gradient, and absolute pressure at a location near the periphery. Figures 5.3.4-11 through 5.3.4-15 present similar information for a location near the hub. Overall critical shear values are small when viewing the entire blade, but in the detailed views of the leading and discharge edges, values of critical shear exist at the leading edge near the periphery. The lower flow velocities near the hub do not induce regions of critical shear. The higher speed flow at the periphery also causes a thin layer of critical shear on both the pressure and suction sides of the blade. This identifies another possible mechanism of fish injury. Secondary flows that might cause a fish to move toward the blade surface could cause injury that might be independent of possible fish scrape or contact with the blade. This shear region is quite thin and is therefore, not suspected as a major mechanism of injury. Pressure gradients are greater in the runner than in the stay vanes and wicket gates, achieving values in excess of 750 psi / sec. These values are not believed to induce any fish injury. Generally, high pressure gradients occur near the leading edge and near the trailing edge. These regions are somewhat larger for the periphery than the hub.

In general, a well designed Kaplan blade operating on cam, having no gaps, and operating without cavitation, seems to cause only small local regions of critical shear that might cause fish injury.

A preliminary analysis was made of a conventional Kaplan blade at its mid range tilt position with hub and periphery gaps. This calculation was a first effort and results are viewed as preliminary. Two analyses were made, one analysis assumed the blade extended exactly to the hub and periphery, so that no gaps existed. A second analysis included gaps at the hub entrance and discharge and the periphery entrance and discharge. Figure 5.3.4-16 shows the blade and hub grid to permit visualization of the hub gap. Visualization of the results is presented with streamlines that have been colored to show the value of shear. Streamlines were released in all gap locations. In Figure 5.3.4-17, streamlines were seeded in the gap at the blade periphery and also the blade was colored according the value of local velocity. High shear values exist on the streamlines in the gap and for a short distance downstream of the gap. Subsequently, the flow moves downstream without evidence of vortices or other significant flow disturbance. Figures 5.3.4-18 and 5.3.4-19 show shear in a region near the periphery leading edge for both gap and no gap calculations. The calculation that includes the gap has a larger region of critical shear. Streamlines seeded in all four gap locations (hub entrance edge and discharge edge, and periphery entrance edge and discharge edge) are shown in Figure 5.3.4-20. The periphery gap leakage flow seems to not form a strong vortex at this particular operating condition, but the hub entrance edge leakage forms a large vortex and the streamlines do not travel along the hub, but move radially outward and are at a rather large radius when they leave the blades. The implications for a fish traversing such a vortex can not be quantified at this time, but are presumed to be significant. The flow through the hub discharge edge gap also forms a strong vortex, with similar implications. and indeed with tested adverse consequences for fish survival (see Section 4.4.5).

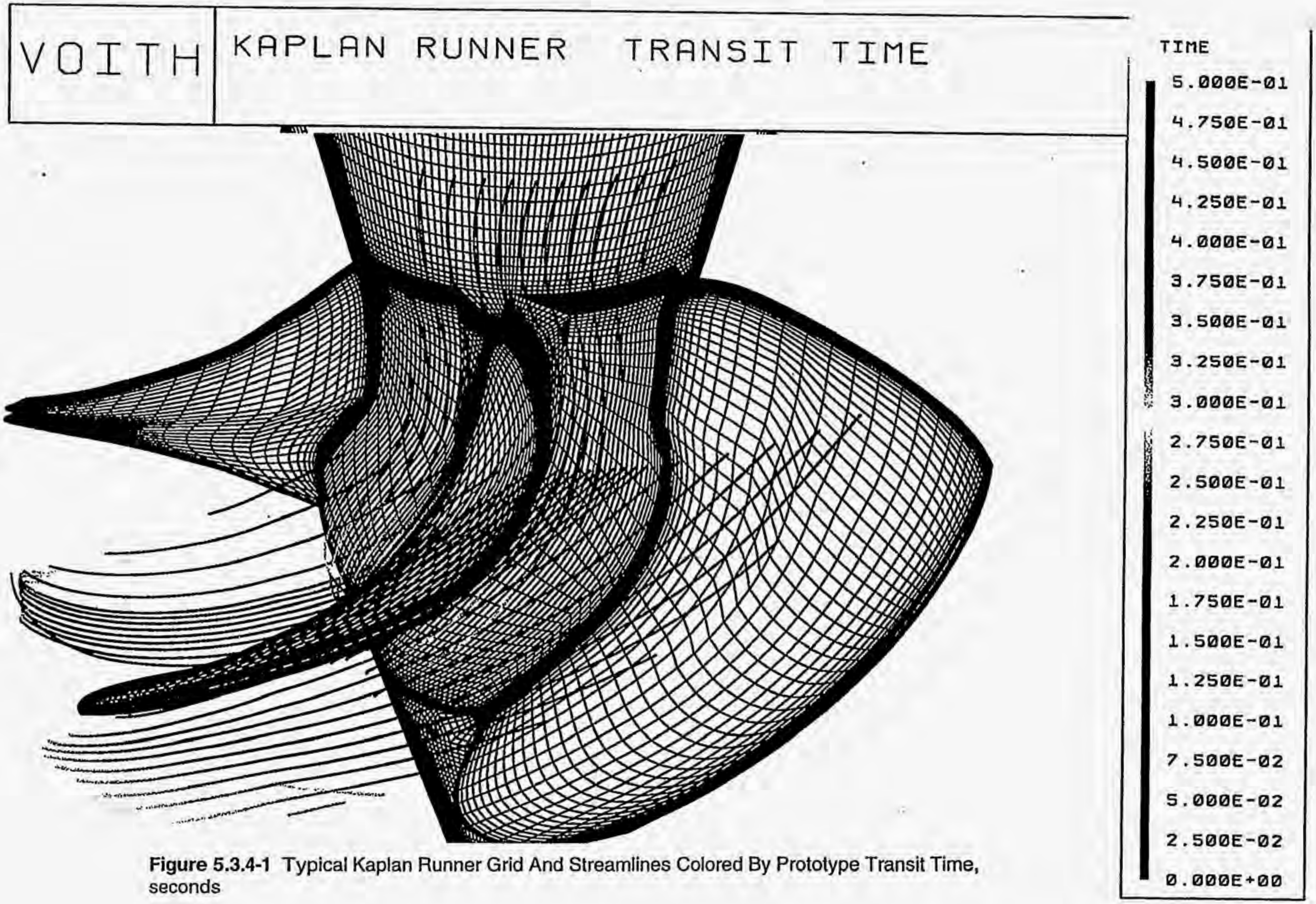

Figure 5.3.4-1 Typical Kaplan Runner Grid And Streamlines Colored By Prototype Transit Time, seconds

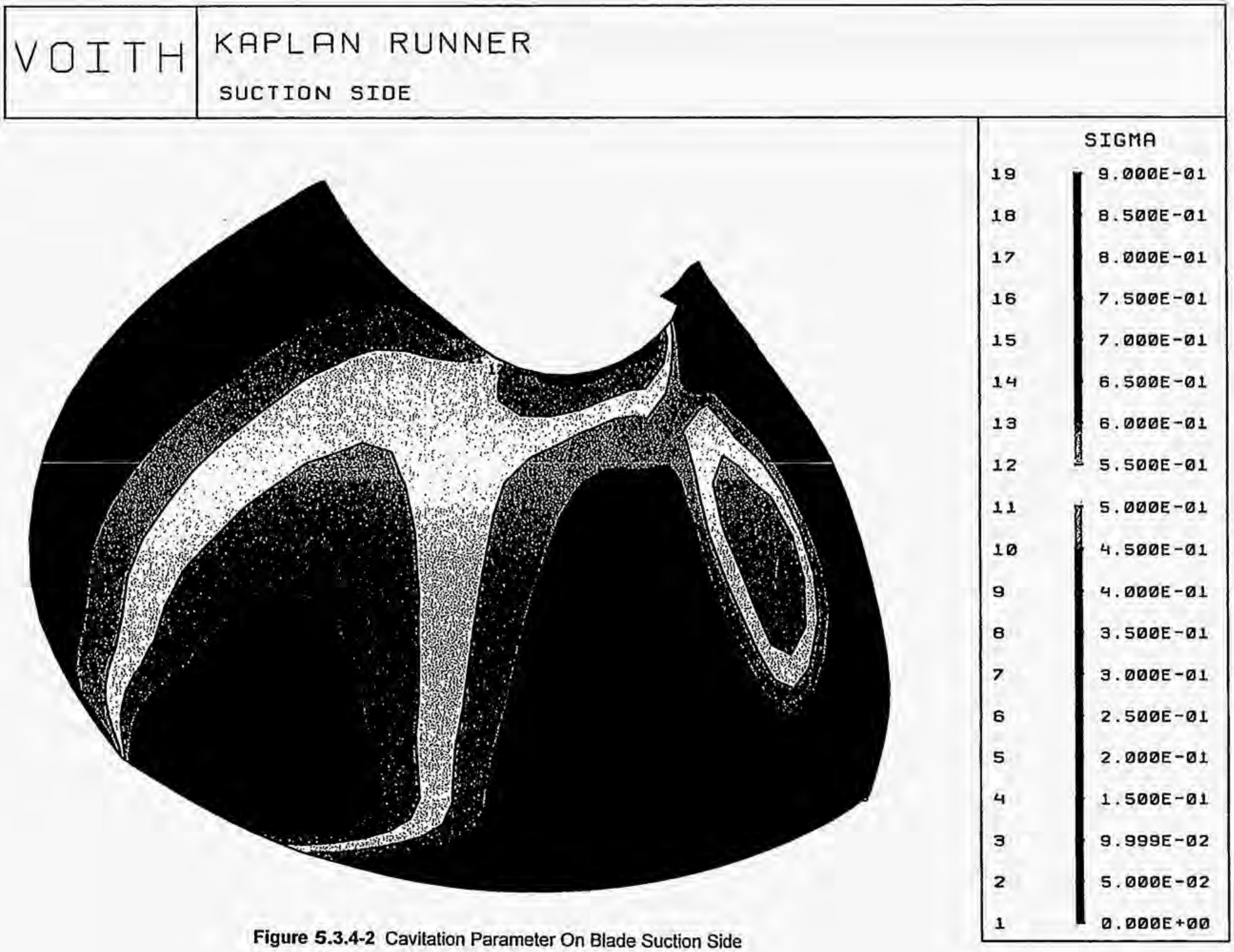

Figure 5.3.4-2 Cavitation Parameter On Blade Suction Side

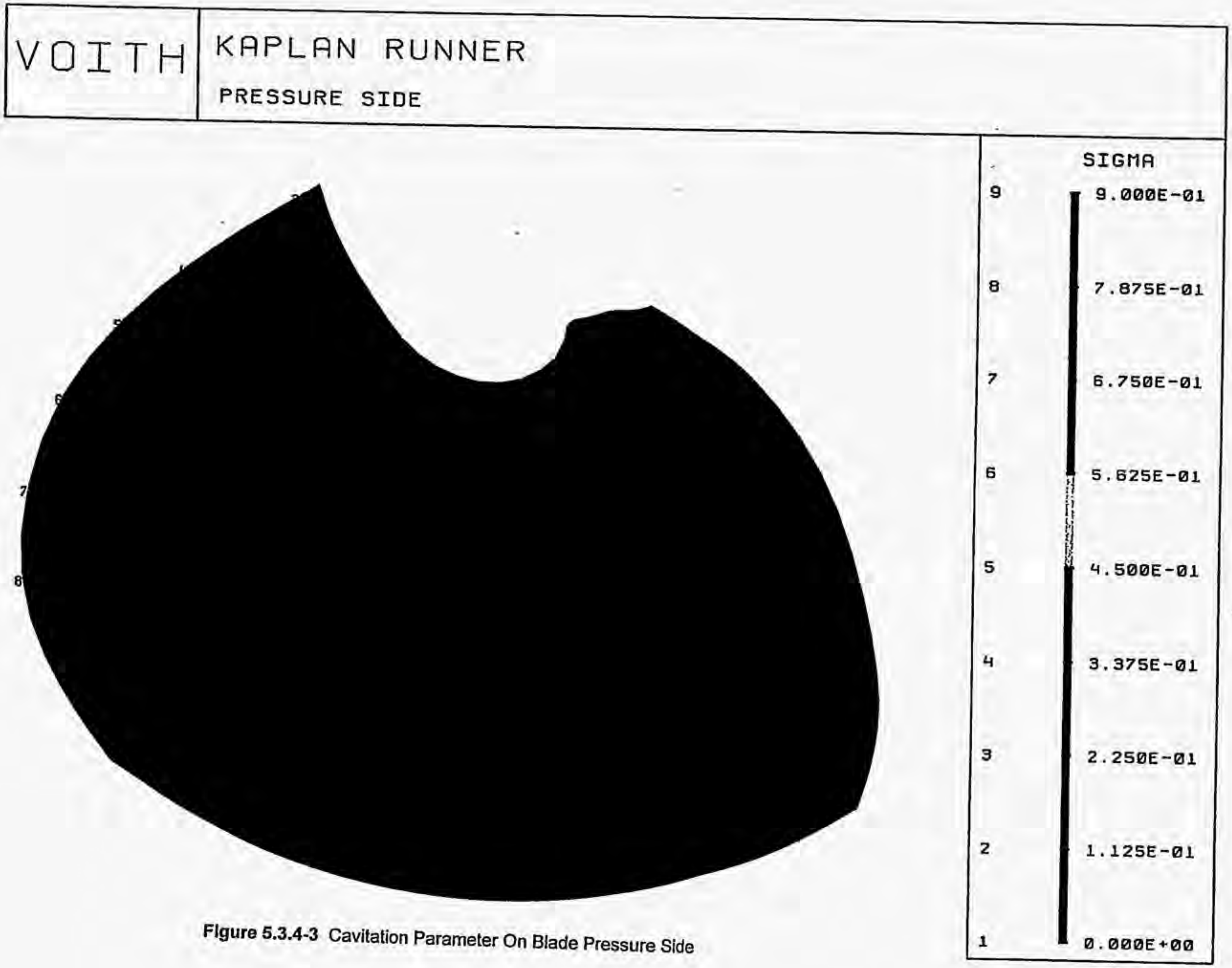

Figure 5.3.4-3 Cavitation Parameter On Blade Pressure Side

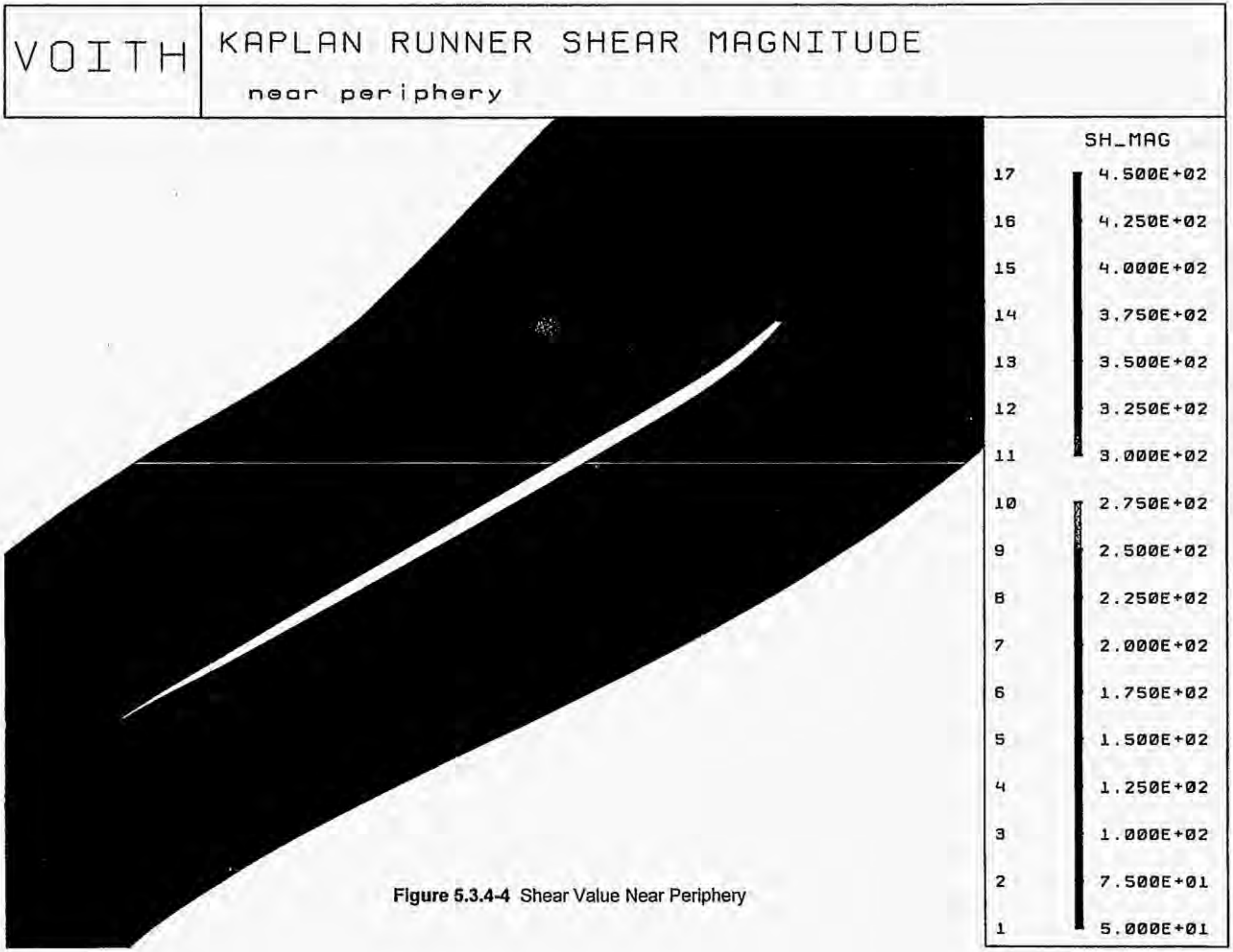

Figure 5.3.4-4 Shear Value Near Periphery

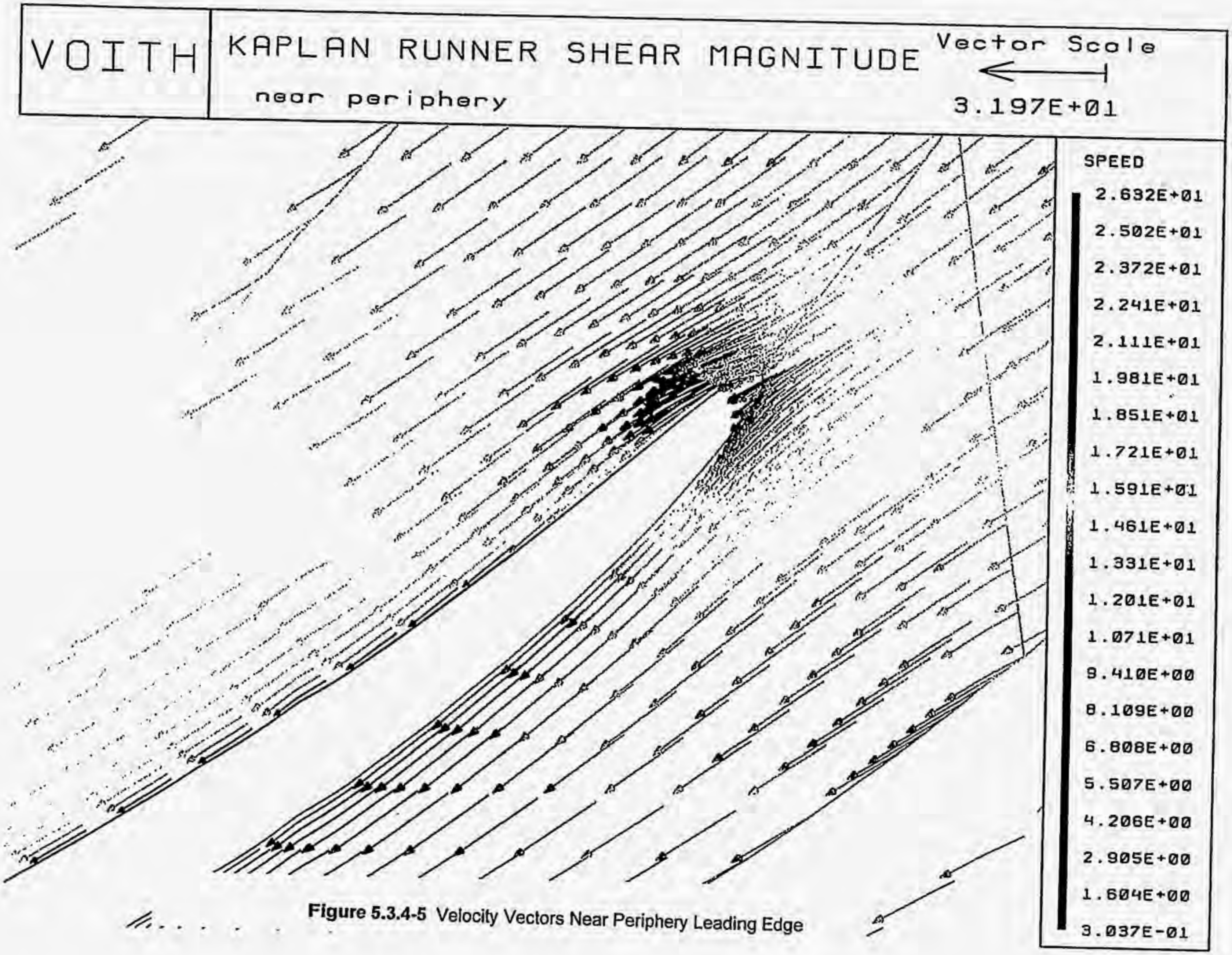

Figure 5.3.4-5 Velocity Vectors Near Periphery Leading Edge

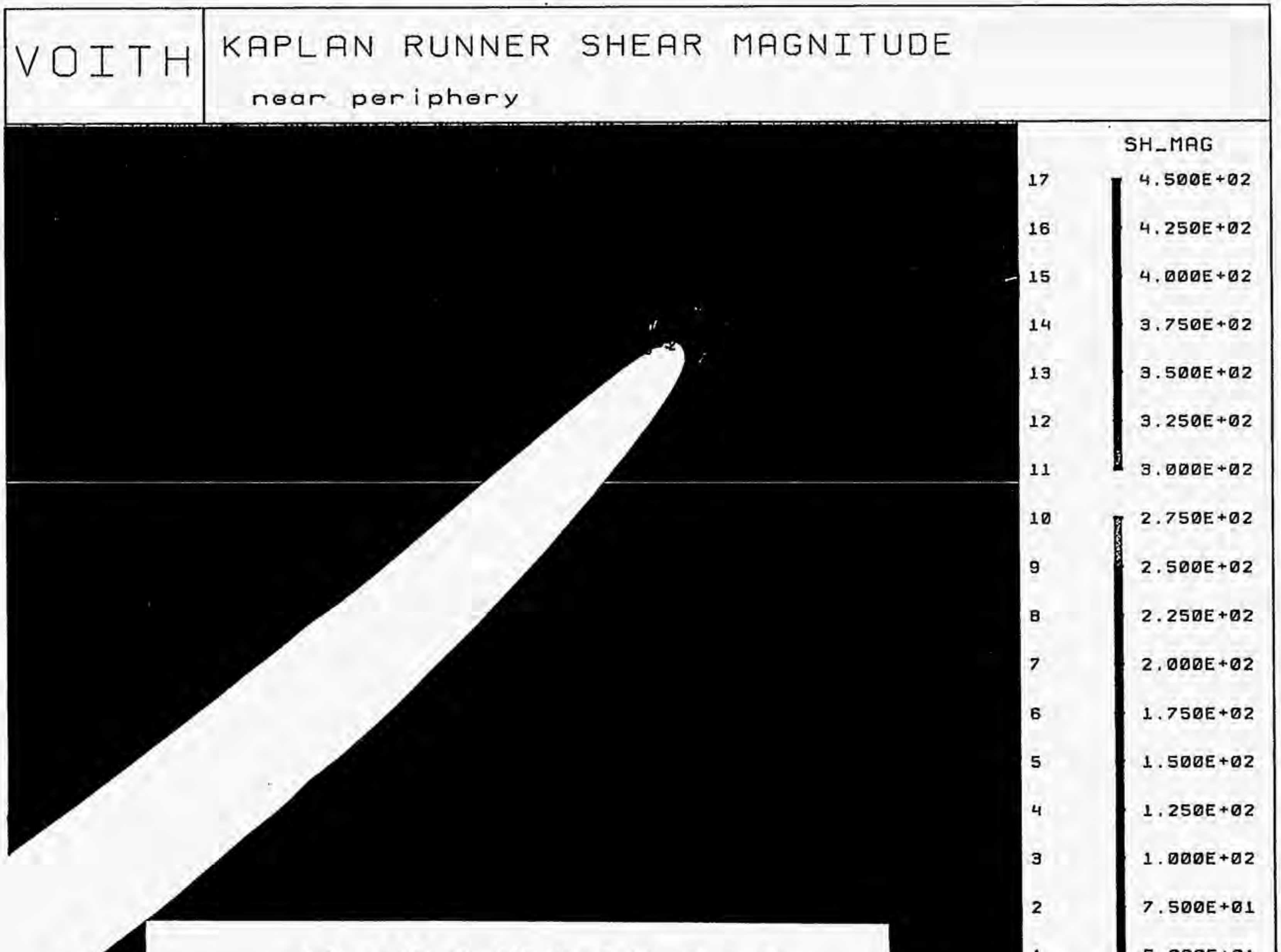

Figure 5.3.4-6 Shear Value Near Periphery Leading Edge

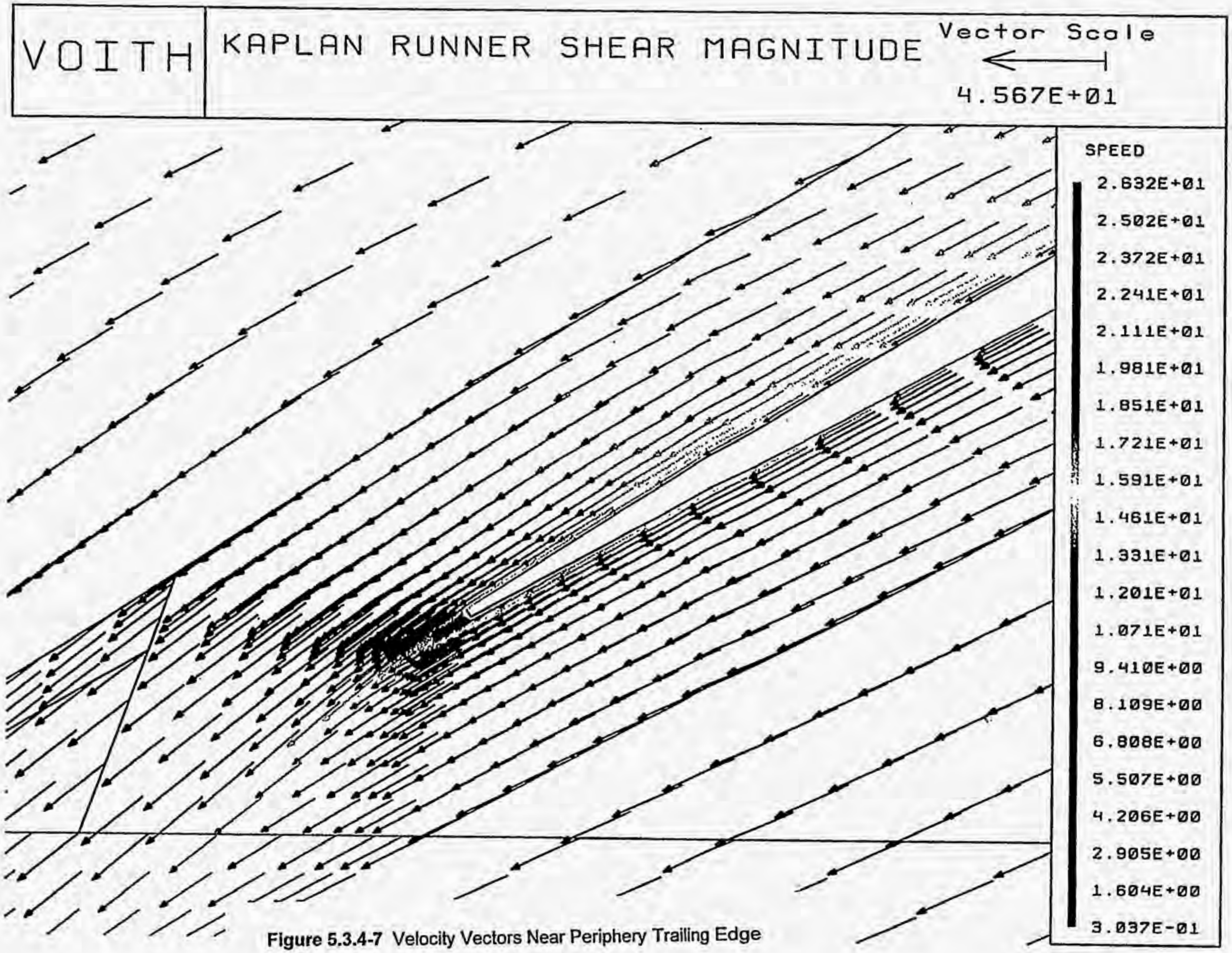

Figure 5.3.4-7 Velocity Vectors Near Periphery Trailing Edge

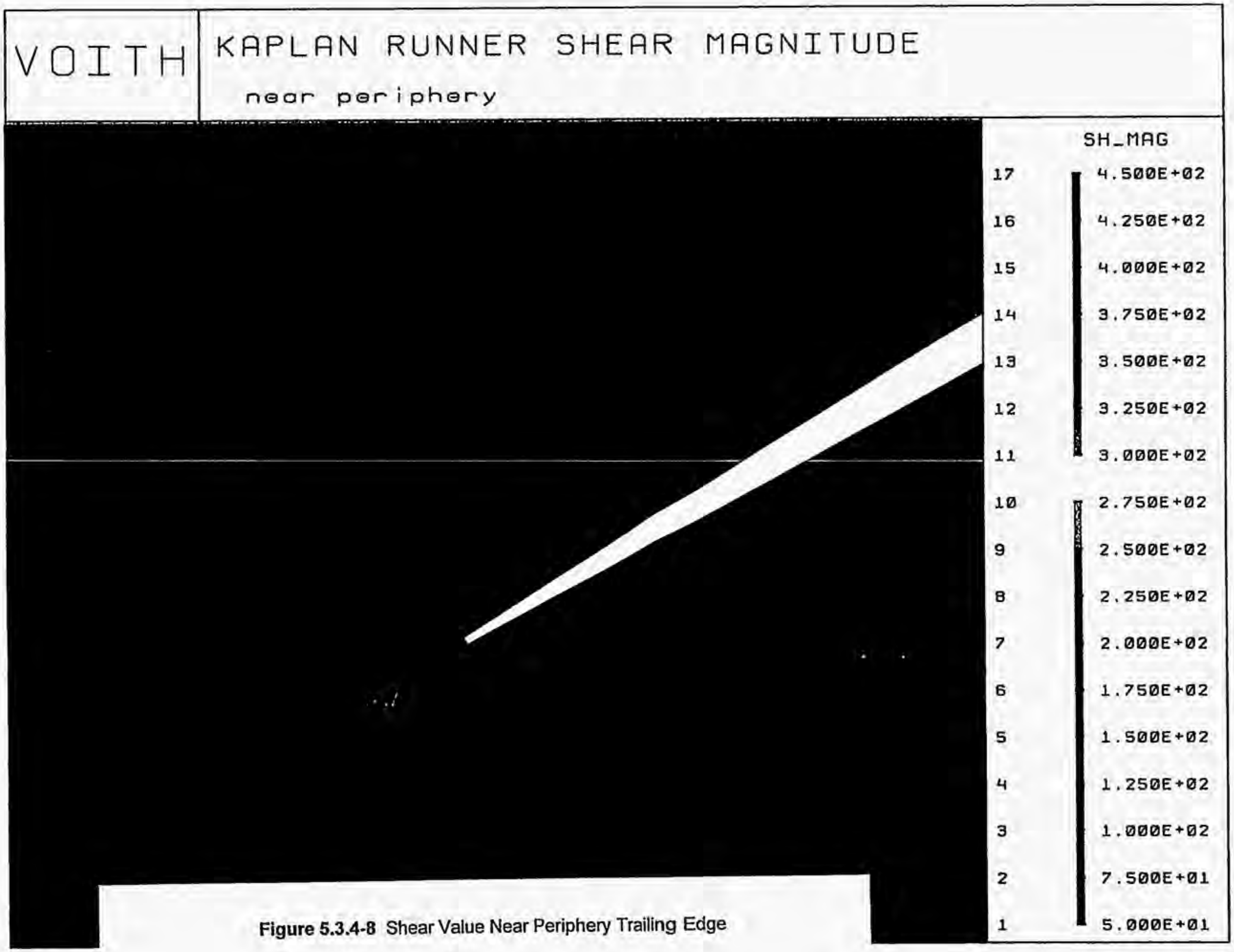

Figure 5.3.4-8 Shear Value Near Periphery Trailing Edge

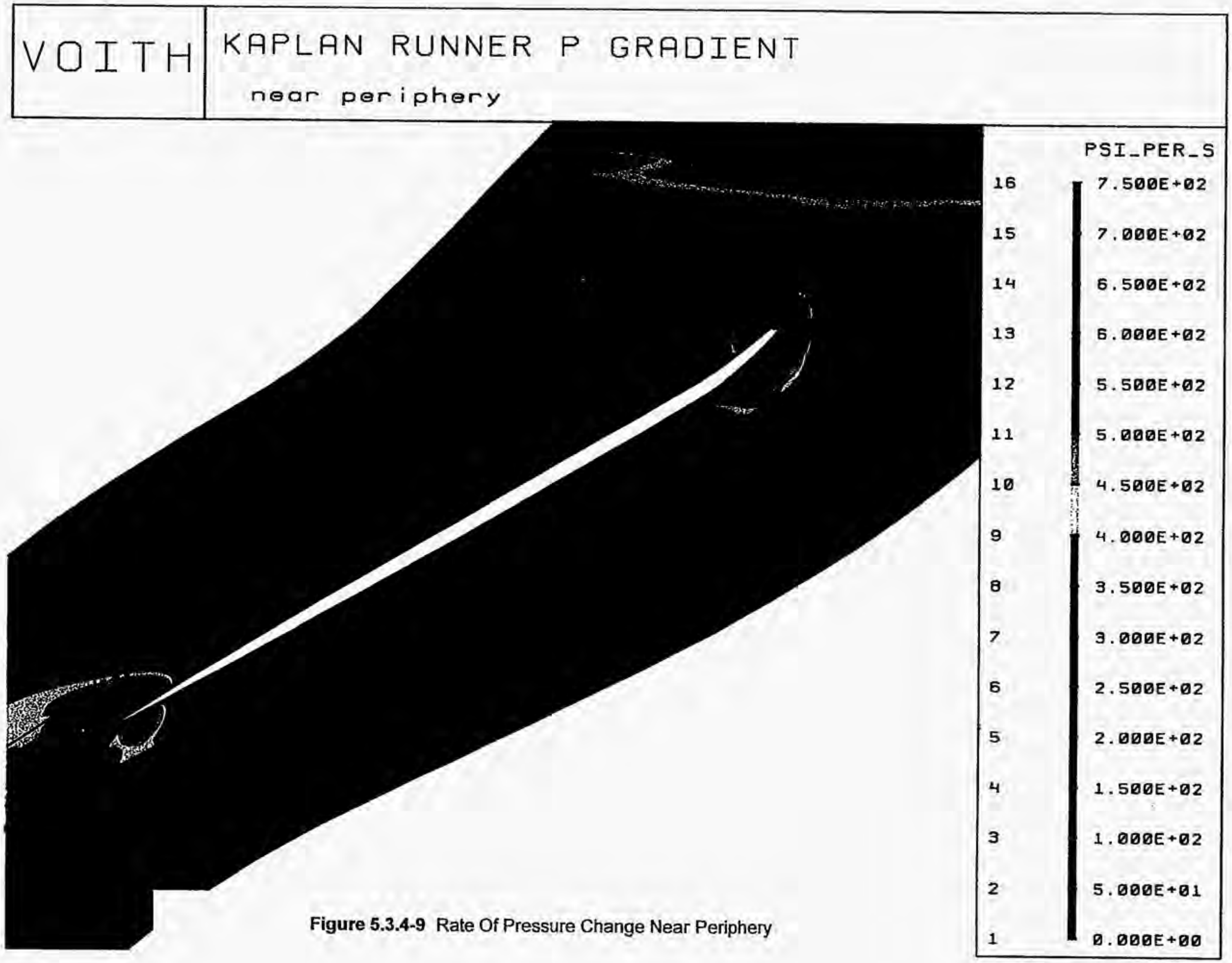

Figure 5.3.4-9 Rate Of Pressure Change Near Periphery

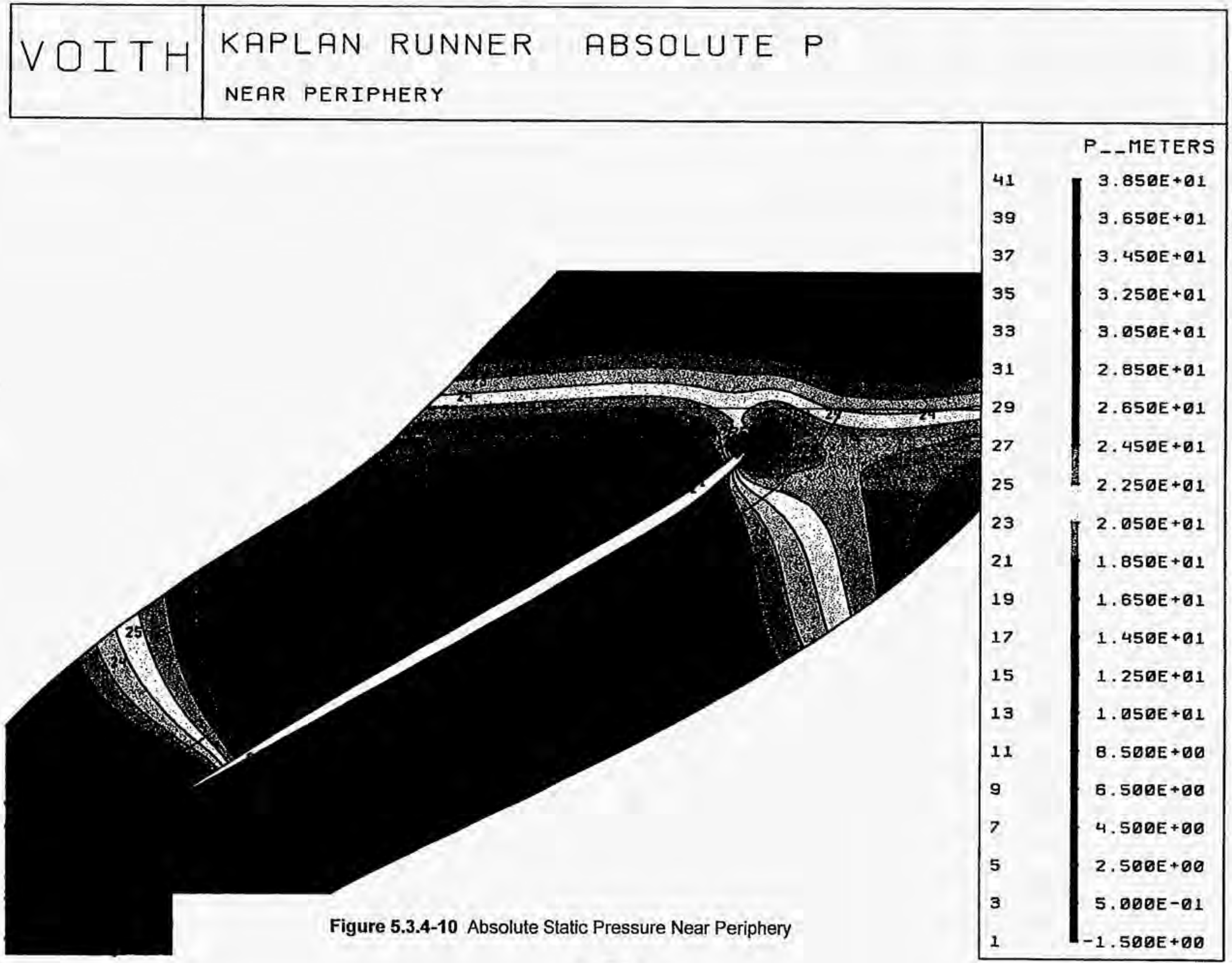

Figure 5.3.4-10 Absolute Static Pressure Near Periphery

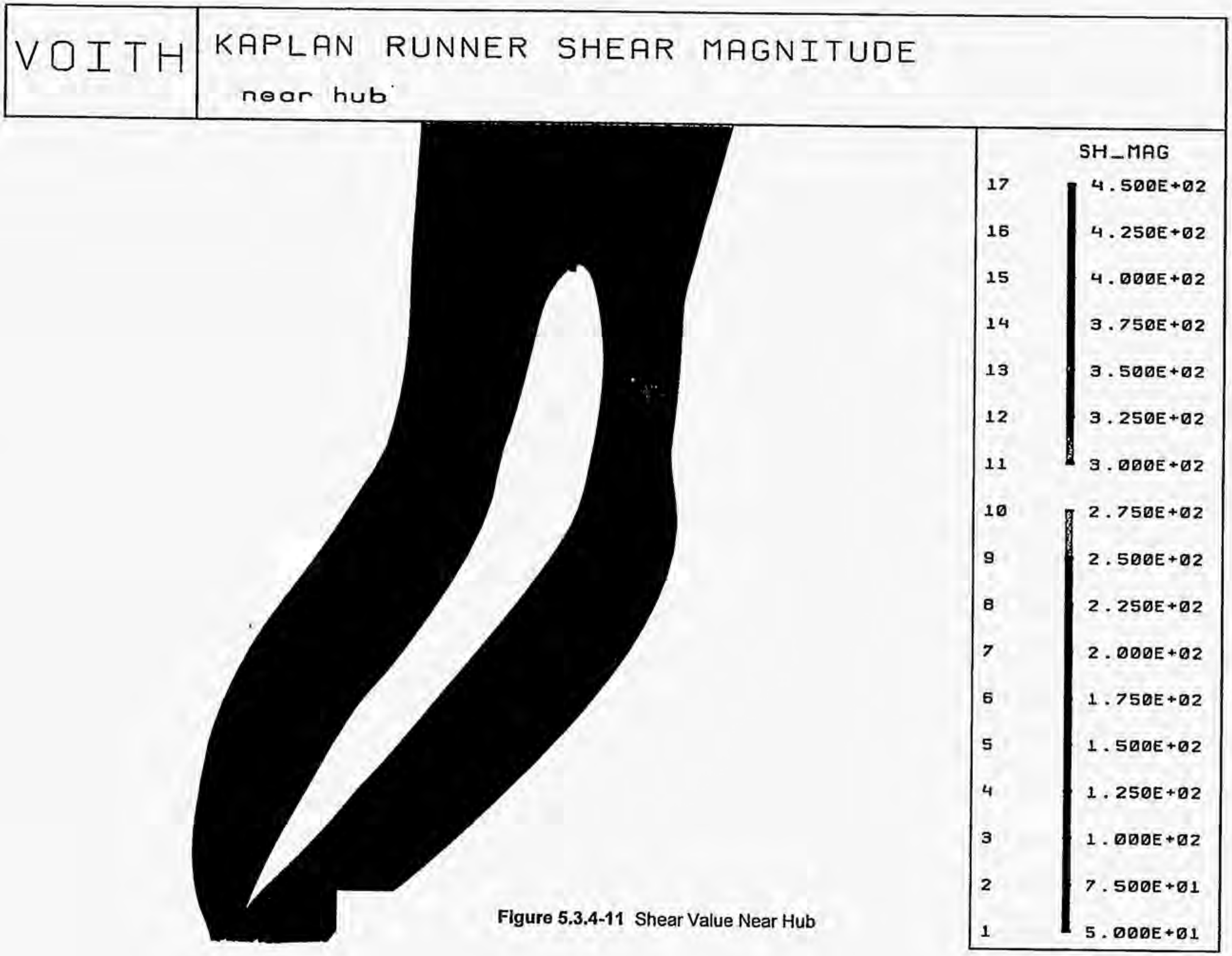

Figure 5.3.4-11 Shear Value Near Hub

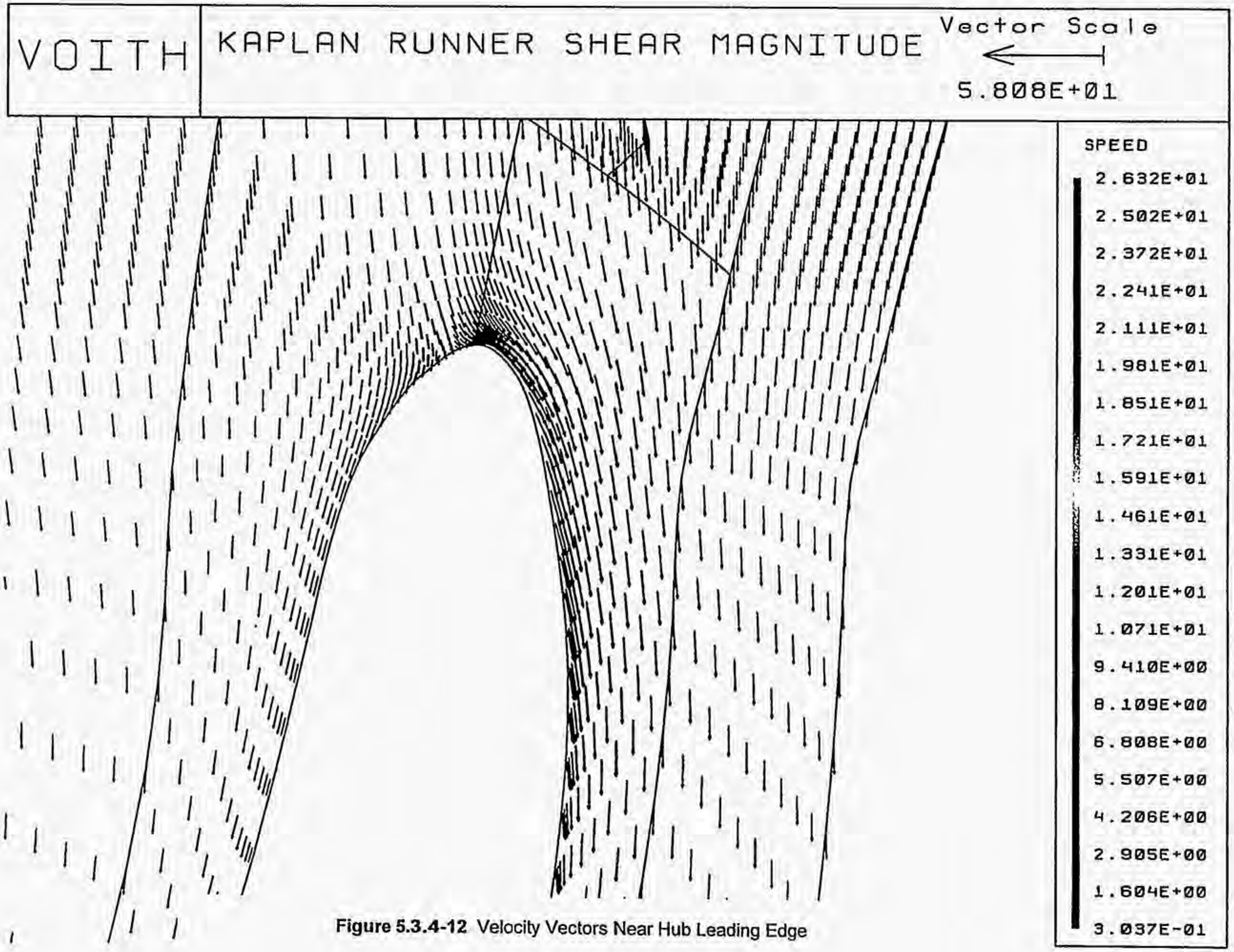

Figure 5.3.4-12 Velocity Vectors Near Hub Leading Edge

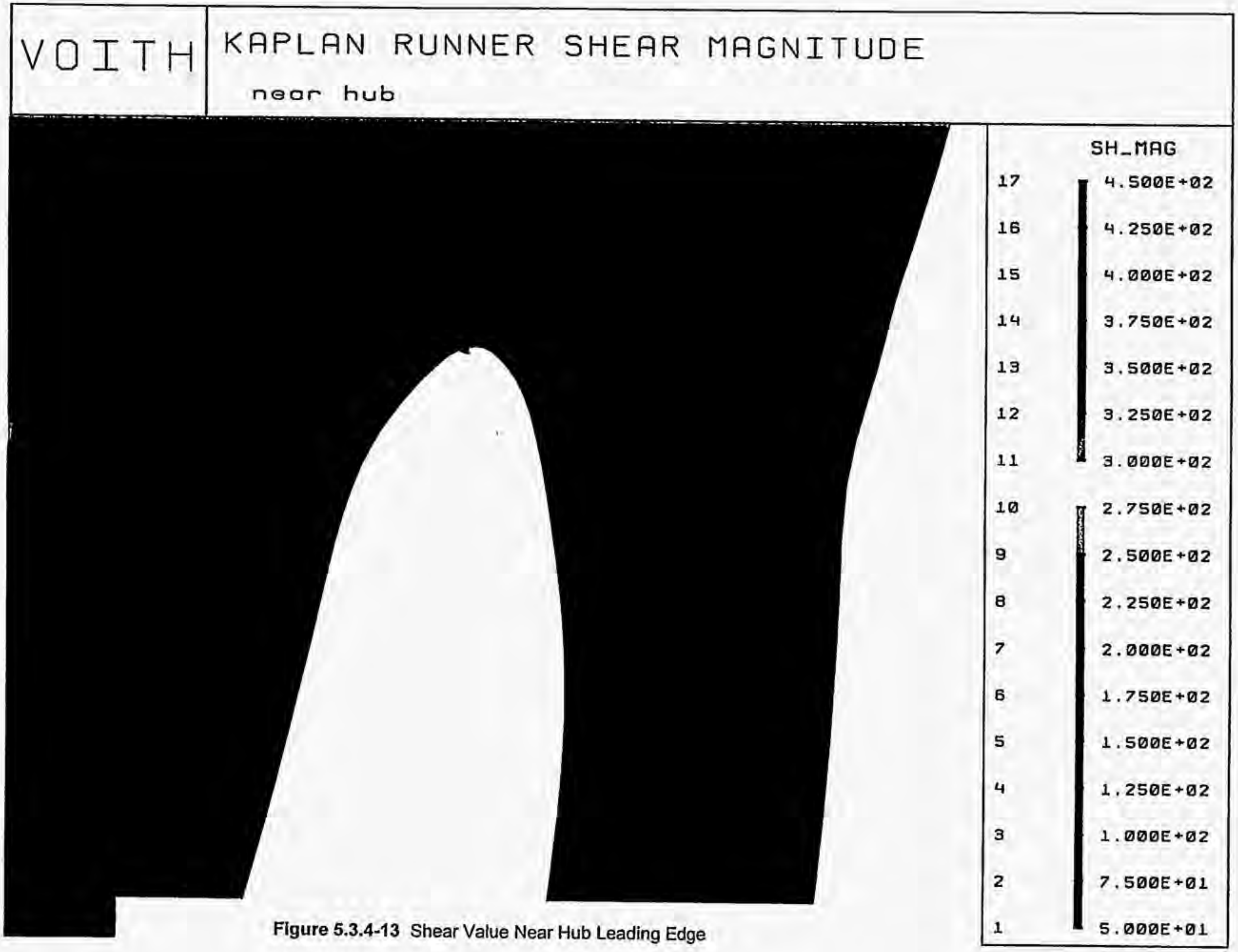

Figure 5.3.4-13 Shear Value Near Hub Leading Edge

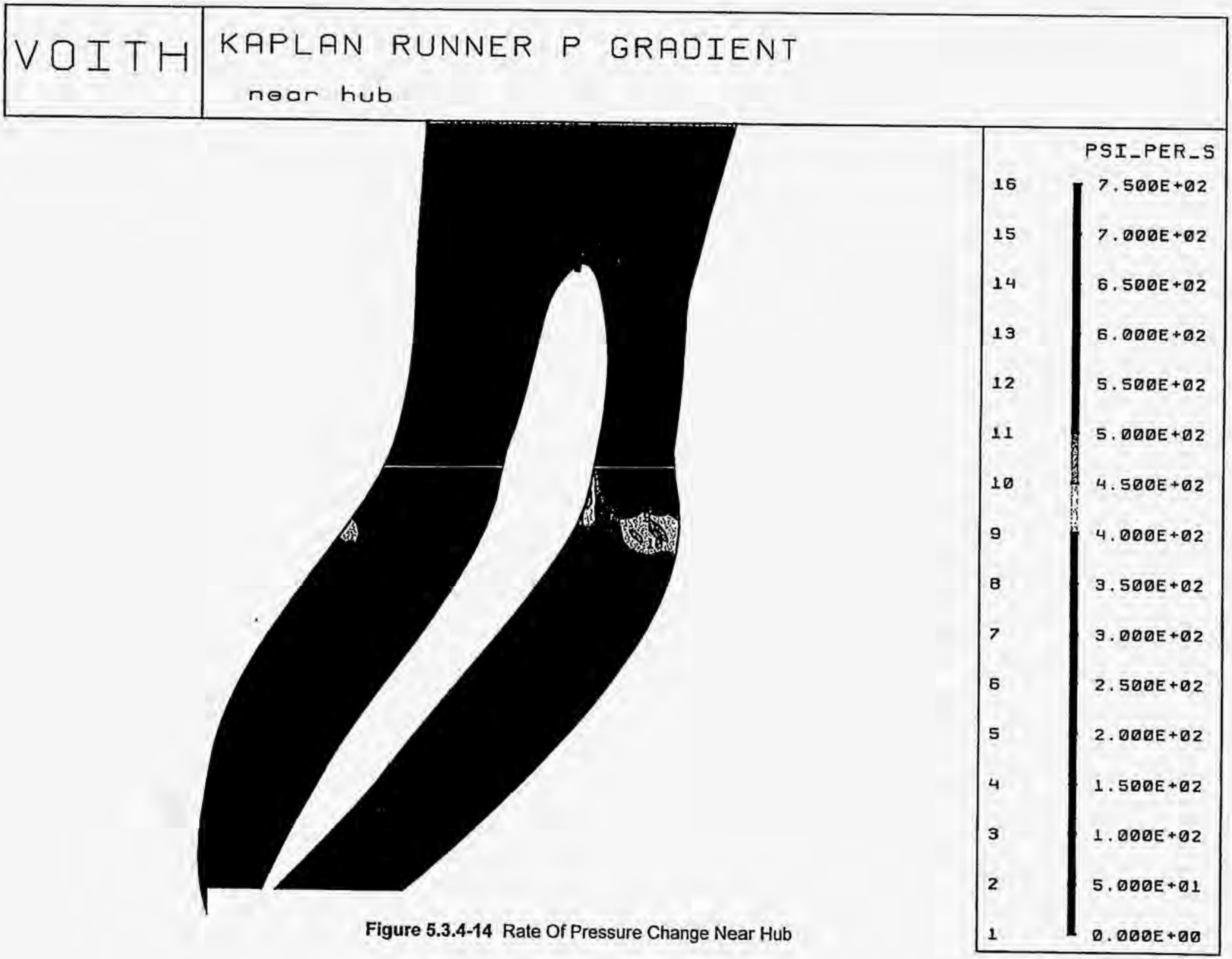

Figure 5.3.4-14 Rate Of Pressure Change Near Hub

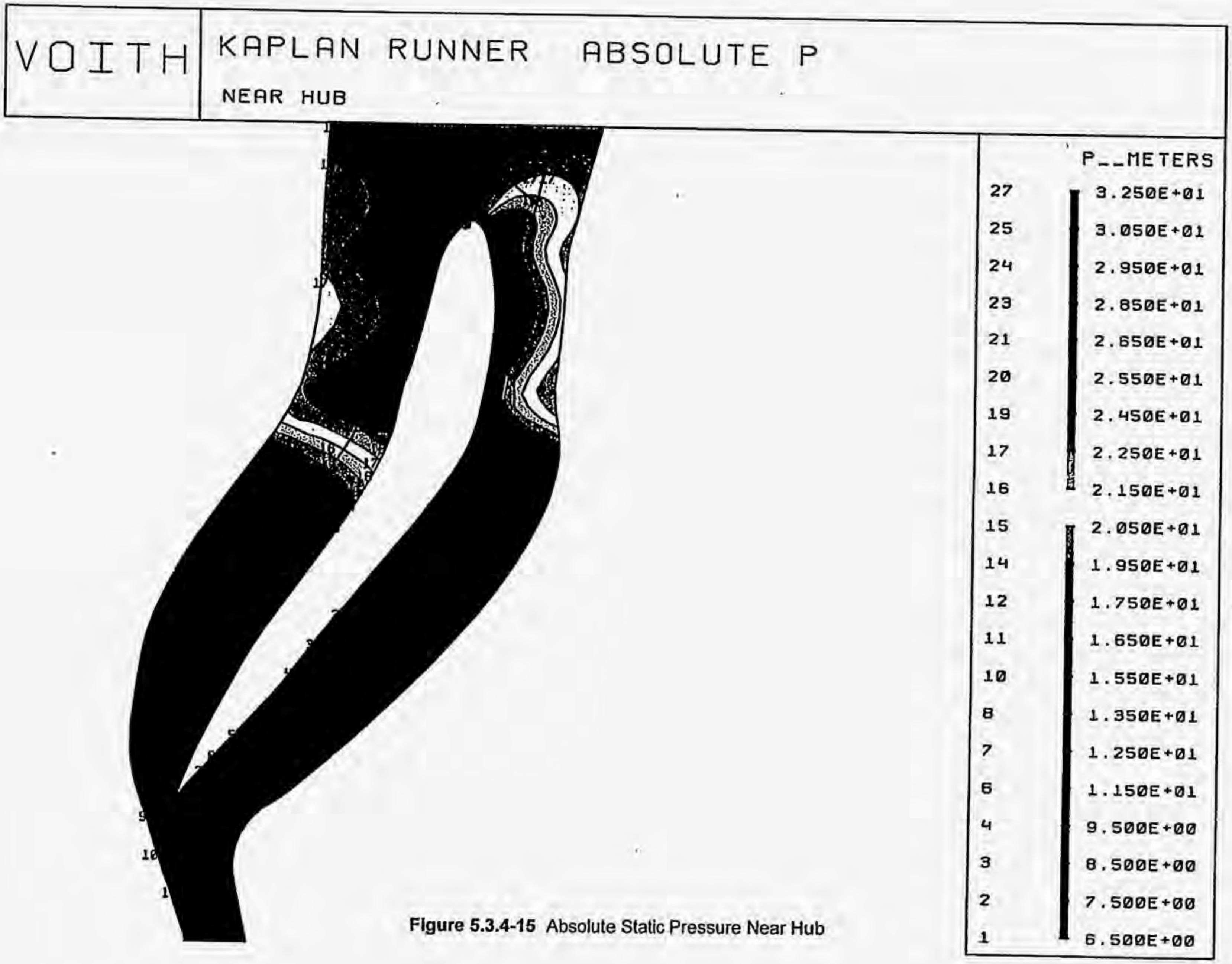

Figure 5.3.4-15 Absolute Static Pressure Near Hub

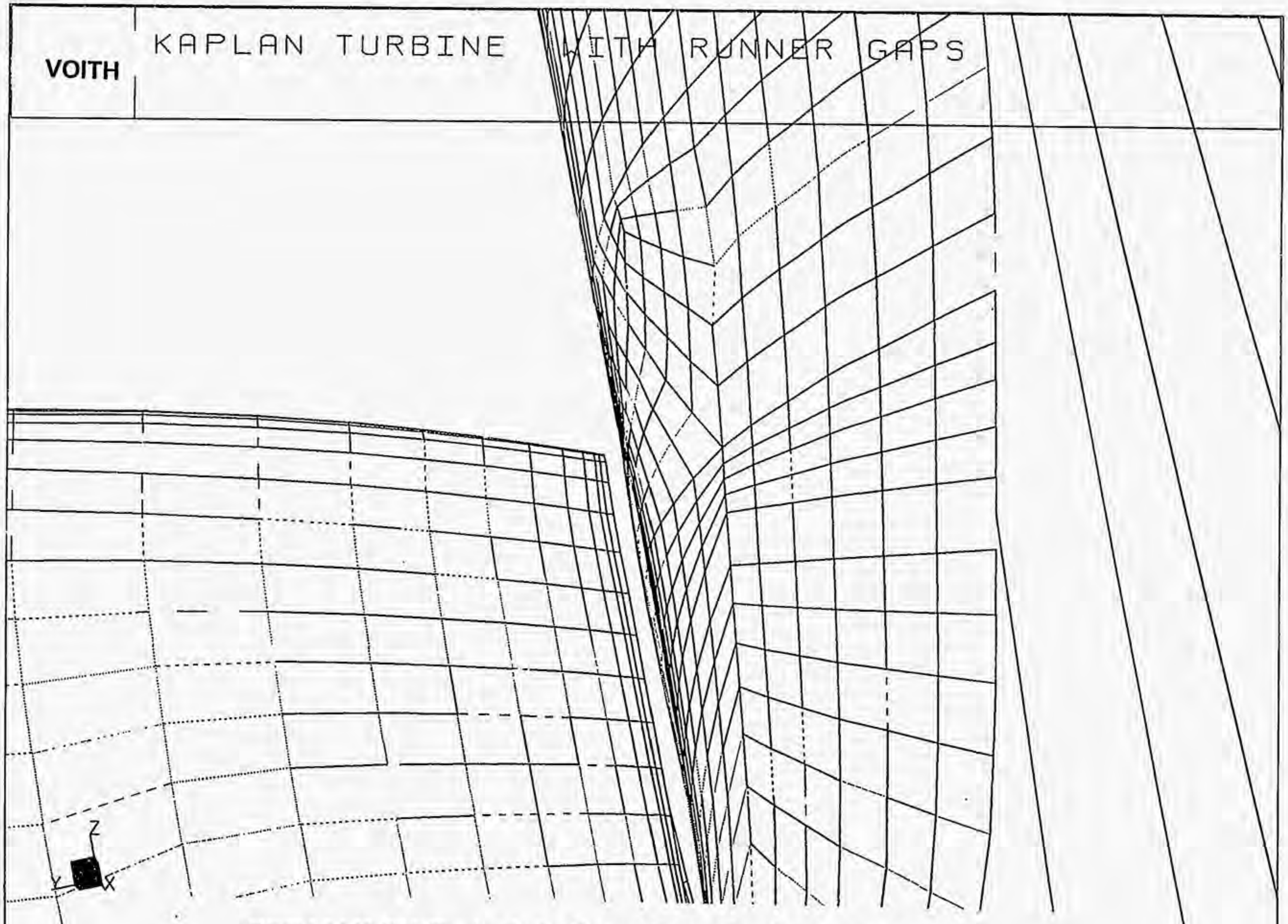

Figure 5.3.4-16 Grids Showing The Gap Between Entrance Edge Of Runner Blade And Hub

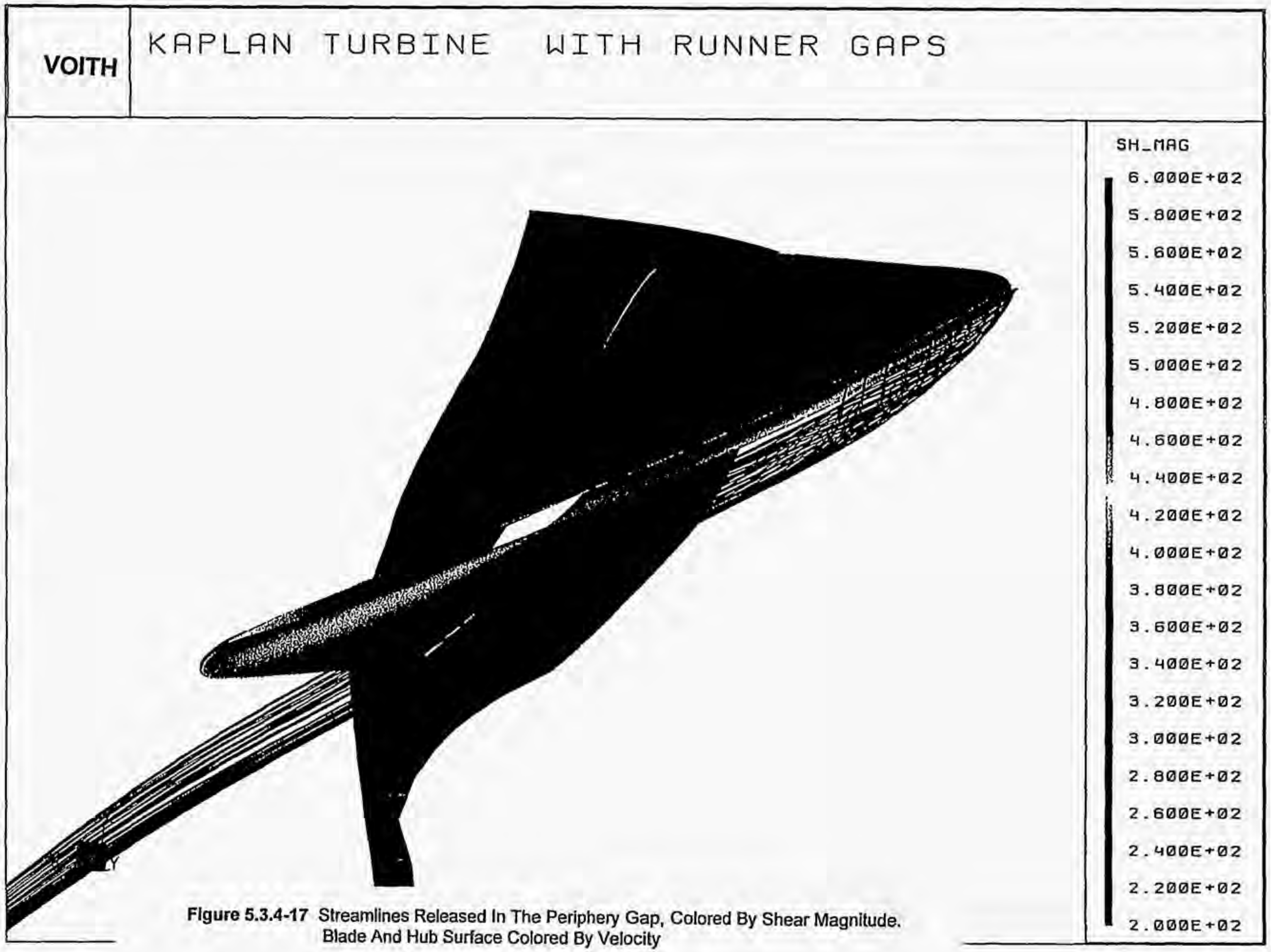

Figure 5.3.4-17 Streamlines Released In The Periphery Gap, Colored By Shear Magnitude. Blade And Hub Surface Colored By Velocity

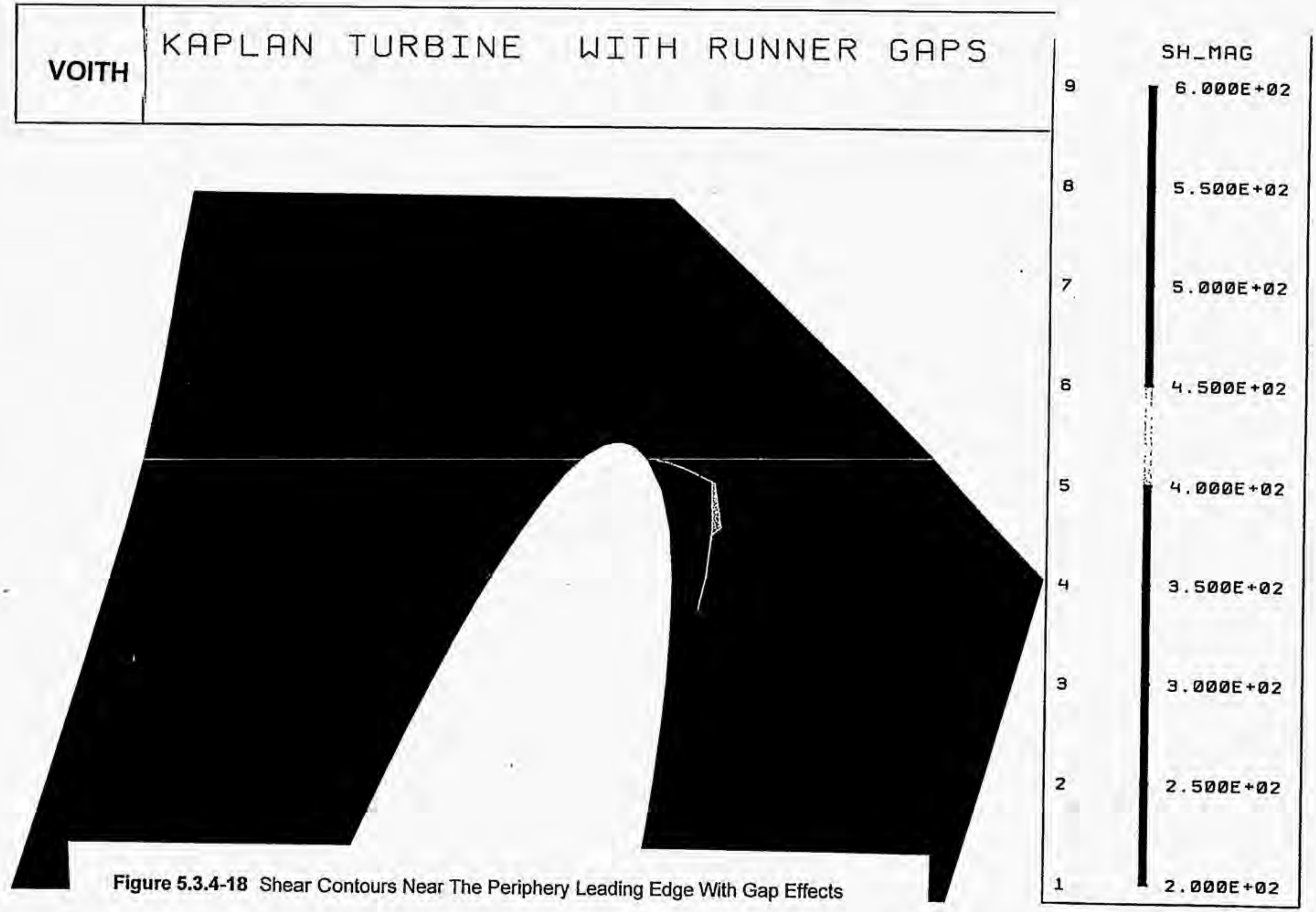

Figure 5.3.4-18 Shear Contours Near The Periphery Leading Edge With Gap Effects

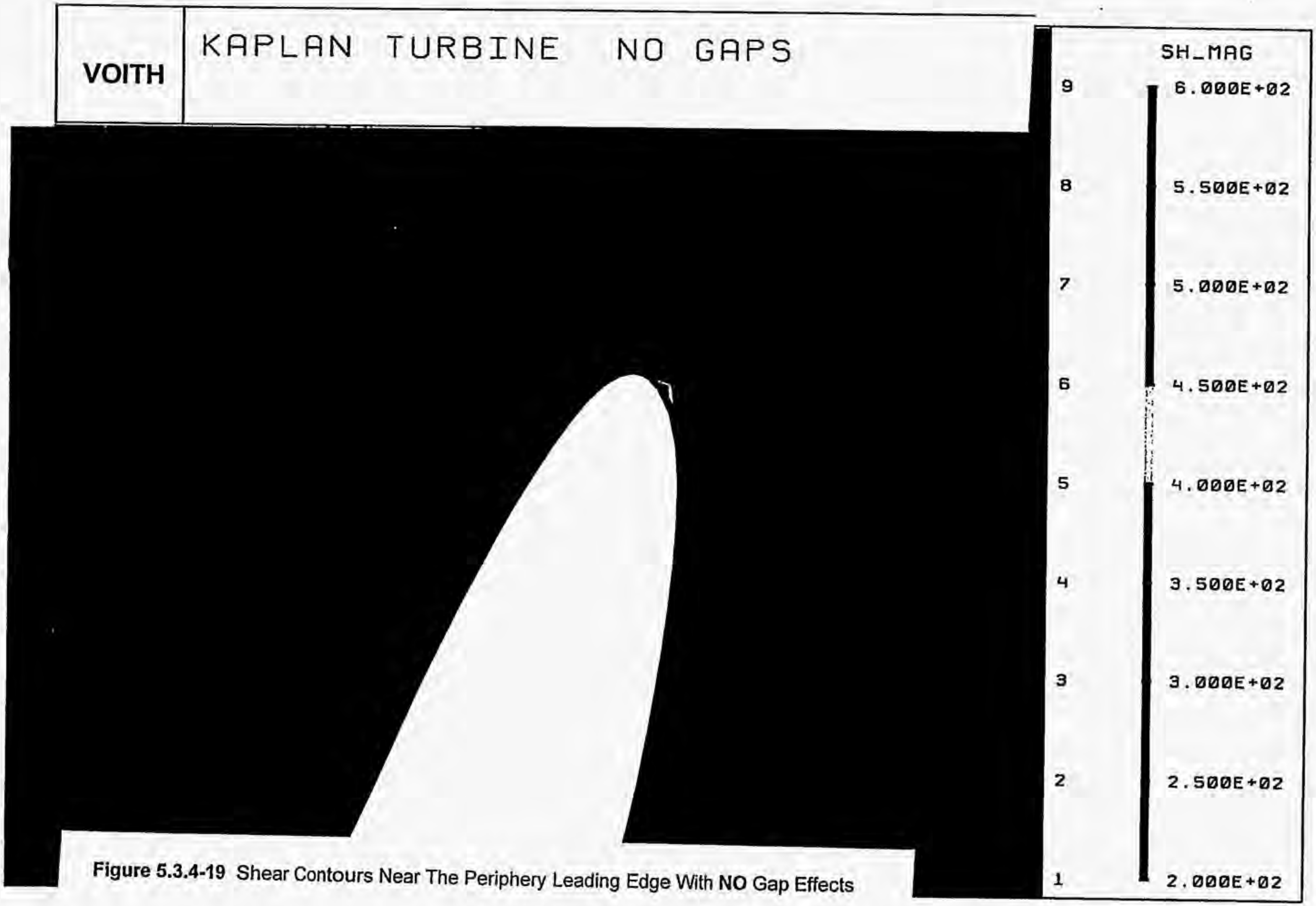

Figure 5.3.4-19 Shear Contours Near The Periphery Leading Edge With NO Gap Effects

Figure 5.3.4-20 Streamlines Released In All Gaps, Colored By Shear Magnitude. Blade And Hub Surface Colored By Velocity

5.3.5 FRANCIS TURBINE RUNNER

Two aspects of Francis turbine operation were evaluated, blade shape and off design operation. An ongoing project was used, with the following characteristics: Inlet diameter of 6.4 m , Rated speed of 107 RPM. The best efficiency point occurs at head of 112 m and a discharge of 296 m3/s. Two blade shapes were analyzed, one blade having a relatively thin leading edge near the crown, and a second blade with a considerably thicker leading edge near the crown. Off design effects were evaluated by analyzing operation at a low head (70% of the best efficiency head). Section 10.2 reviews terminology that may be useful on this section. Grids for a typical calculation are shown of Figure 5.3.5-1. This figure also shows streamlines colored by prototype transit time.. Near the band, transit time is typically 0.2 sec while at the crown, transit time can be up to 0.4 sec. The streamlines near the crown also demonstrate unexpected flow paths that occur due to secondary flows, even at the best efficiency point. Some of these streamlines remain near the crown, while others respond to the complex flow field and travel toward the band. This phenomenon has a variety of fish survival implications, affecting leading edge strike, zones of shear and / or energy dissipation, scrape, etc.

Operating Condition Evaluation with a Thick Entrance Edge Shape

Operation at the best efficiency condition for the thicker blade is shown in Figures 5.3.5-2 through 5.3.5-6. This condition is characterized by smooth flow. The blade shape has been carefully chosen to minimize flow disturbances at this head and discharge. The resulting flow field shows minimal shear regions both near the crown and near the band. The shear magnitude is higher at the band than at the crown due to the higher velocity near the band. Figure 5.3.5-2 shows that critical shear regions are small and occur near the blade and in the wake flow at the discharge edge. Figures 5.3.5-3 and 5.3.5-4 show in greater detail the velocity and shear values near the band entrance edge. The pressure gradient is significantly larger than for a low head Kaplan turbine, is shown on Figure 5.3.5-5. Shear values near the crown are low (Figure 5.3.5-6). Values of absolute pressure are shown on Figure 5.3.5-7.

In contrast to the best efficiency point, at the low head operating condition, the blade is operating at an angle of attack. Flow conditions are considerably more complex at an "off design" point of operation. The same series of figures as for the best efficiency point are shown in Figures 5.3.5-8 through 5.3.5-12. The resulting flow field shows higher shear values, especially at the band. Figures 5.3.5-9 and 5.3.5-10 show the leading edge details near the band where the region having a value of shear greater than the critical value extends noticeably away from the blade. On Figure 5.3.5-12, it is observed that shear levels are higher than for the best efficiency point, but the lower velocities at the crown, in combination with the blade shape do not create large shear regions. Values of absolute pressure are shown on Figure 5.3.5-13.

Blade Thickness Evaluation

Figure 5.3.5-14 shows two blade shapes, referred to as thin and thick. Near the band, both blades have equal maximum thickness. Near the crown, however, the "thin" blade is approximately one fourth as thick as the "thick" blade. The blades do however, have subtle differences in the shape of the blade nose. The thin blade actually has a slightly thicker region very close to the leading edge.

At the best efficiency operating point, the thin blade has good performance. The flow field and in particular the values of shear are basically the same as for the thick blade at both the crown and band, Figures 5.3.5-15 through 5.3.5-18.

At the low head operating condition, the thin blade has significant differences compared to the thick blade. Near the crown, the thin blade is unable to maintain smooth attached flow. Figures 5.3.5-19 and 5.3.5-20 show the resulting separated region has induced a recirculation zone and a significant region having high shear values. The critical value of shear has been exceeded. Also shown is a circular flow pattern that is a cross section of a vortex. Streamlines further showing the significant extent of the vortex are shown on Figure 5.3.5-21. Near the band, Figures 5.3.5-22 and 5.3.5-23 show that the "thin" blade, with it's superior local shape near the nose, has lower shear values compared to the "thick" (but locally slightly sharper) blade.

Additionally, a blade was designed having a thin entrance edge along it's entire entrance edge, from crown to band. The analysis of this blade at the low head operating condition shows increased shear values at the band compared to the previous "thin" blade (that actually was not thin at the band), Figure 5.3.5-24.

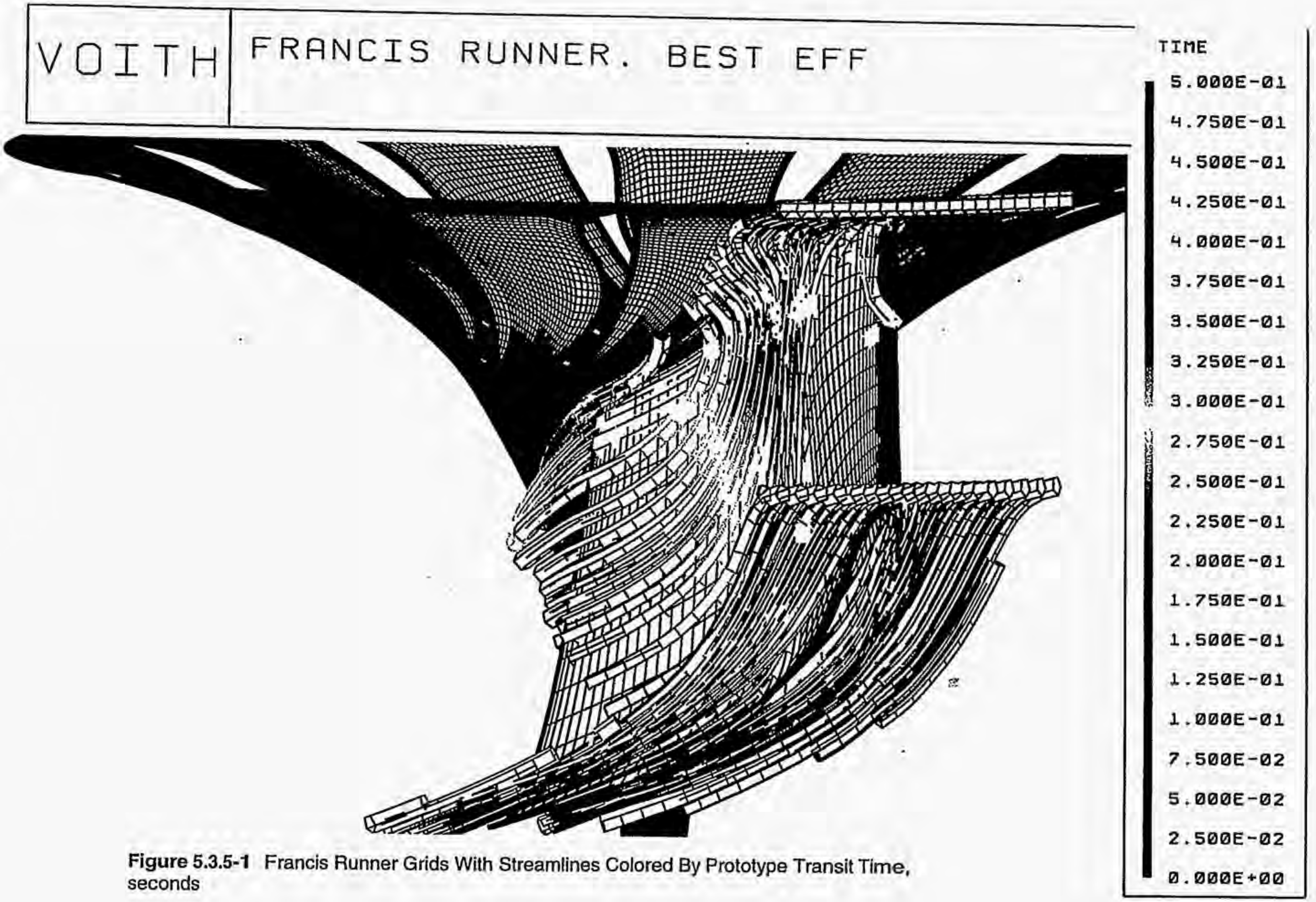

Figure 5.3.5-1 Francis Runner Grids With Streamlines Colored By Prototype Transit Time, seconds

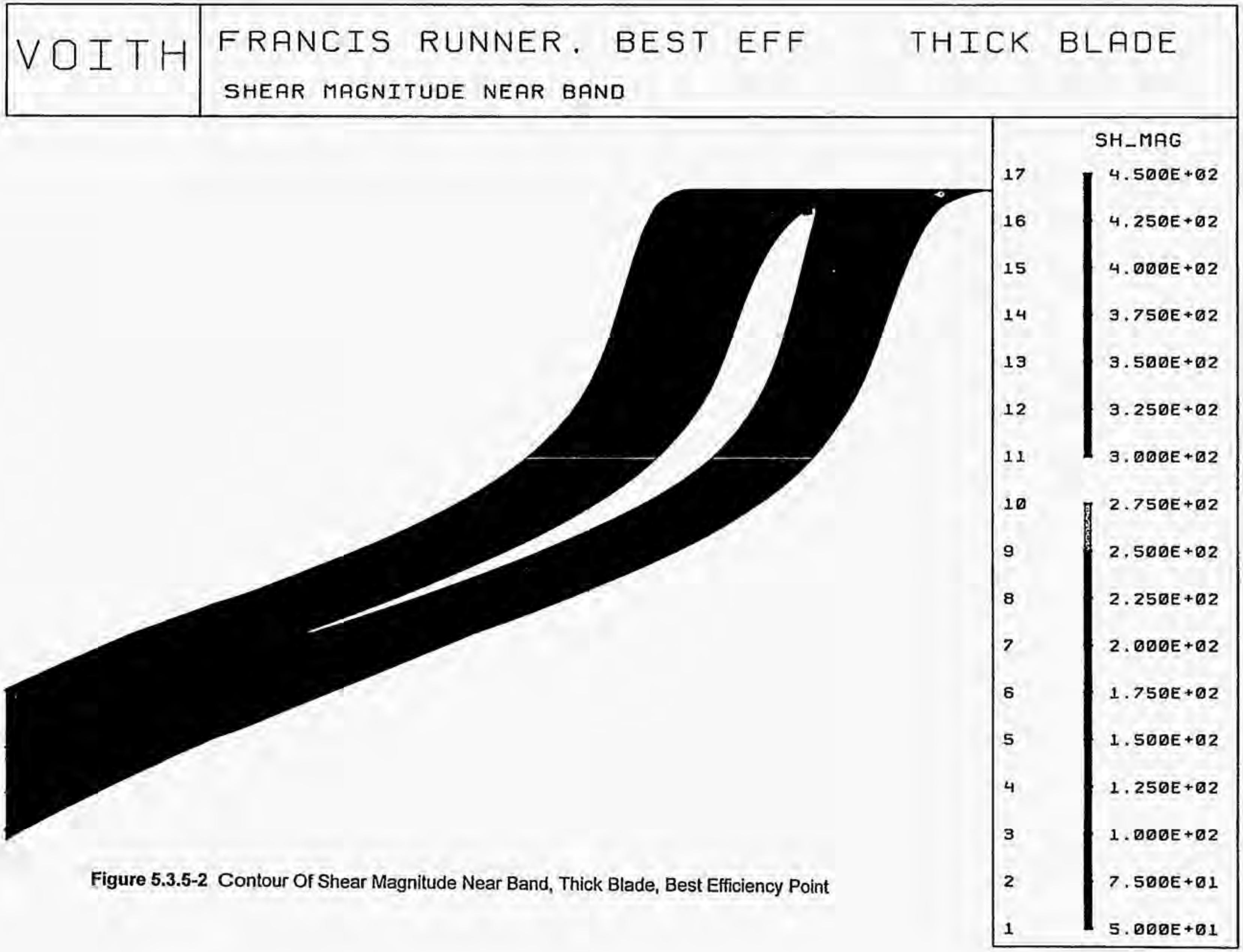

Figure 5.3.5-2 Contour Of Shear Magnitude Near Band, Thick Blade, Best Efficiency Point

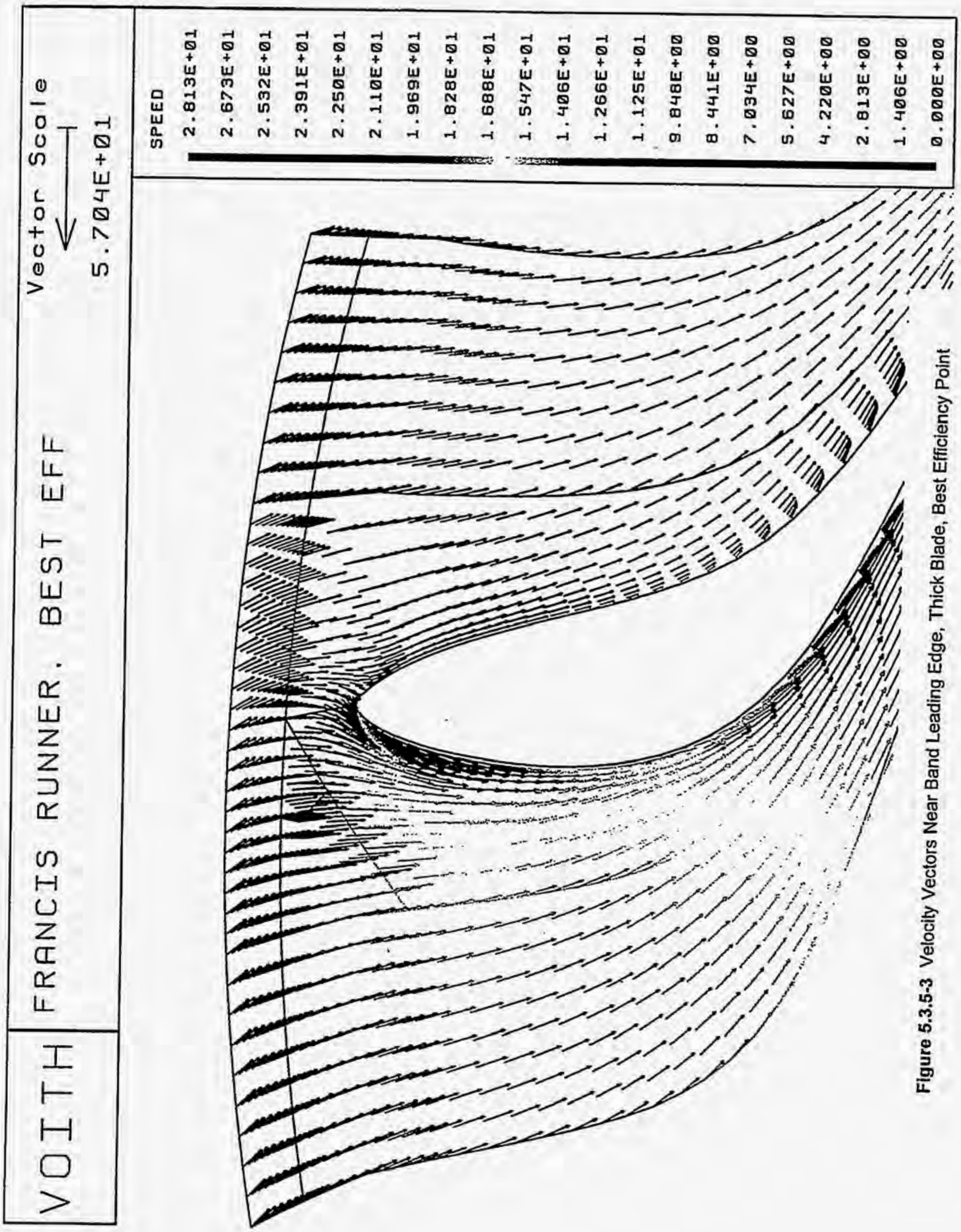

Figure 5.3.5-3 Velocity Vectors Near Band Leading Edge, Thick Blade, Best Efficiency Point

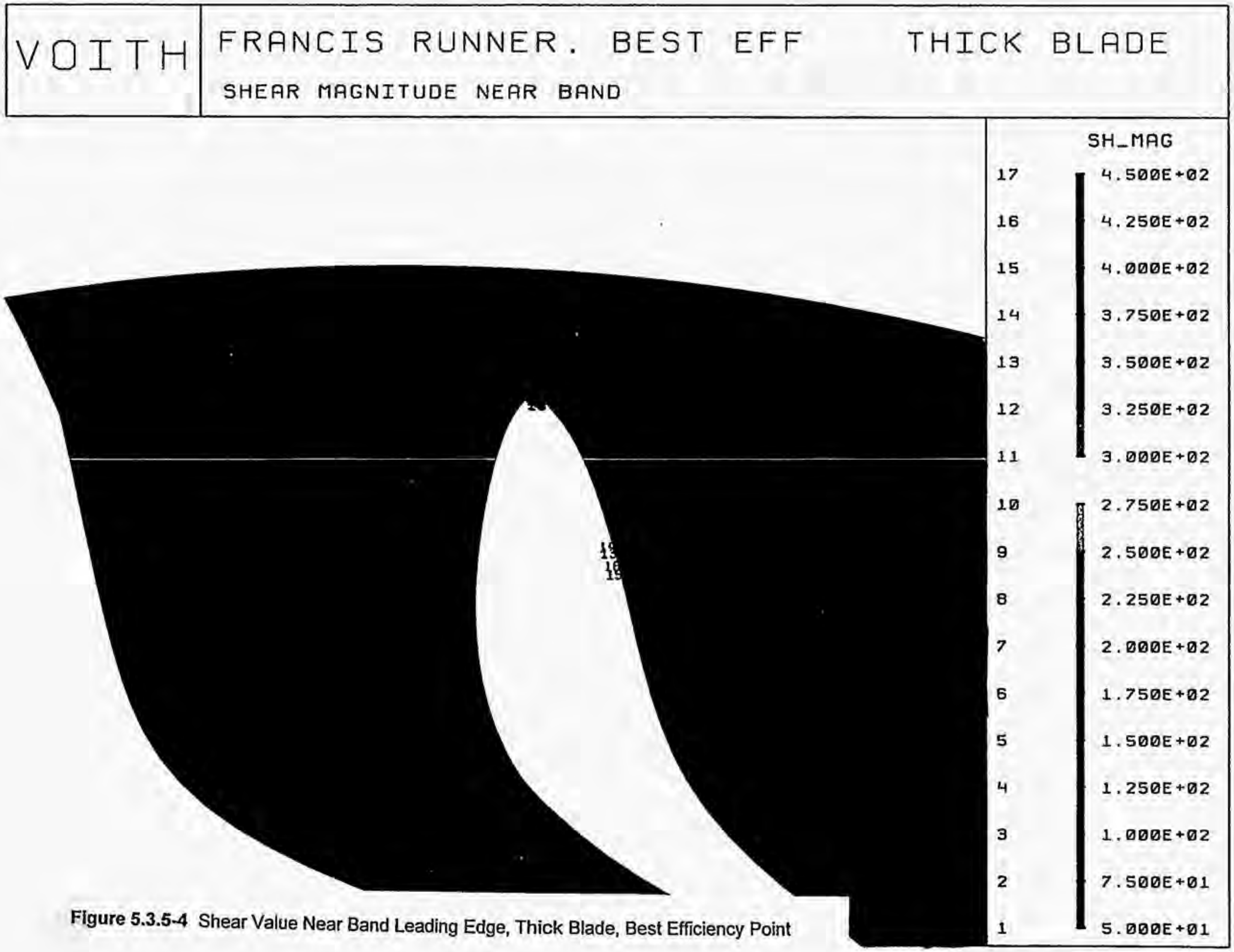

Figure 5.3.5-4 Shear Value Near Band Leading Edge, Thick Blade, Best Efficiency Point

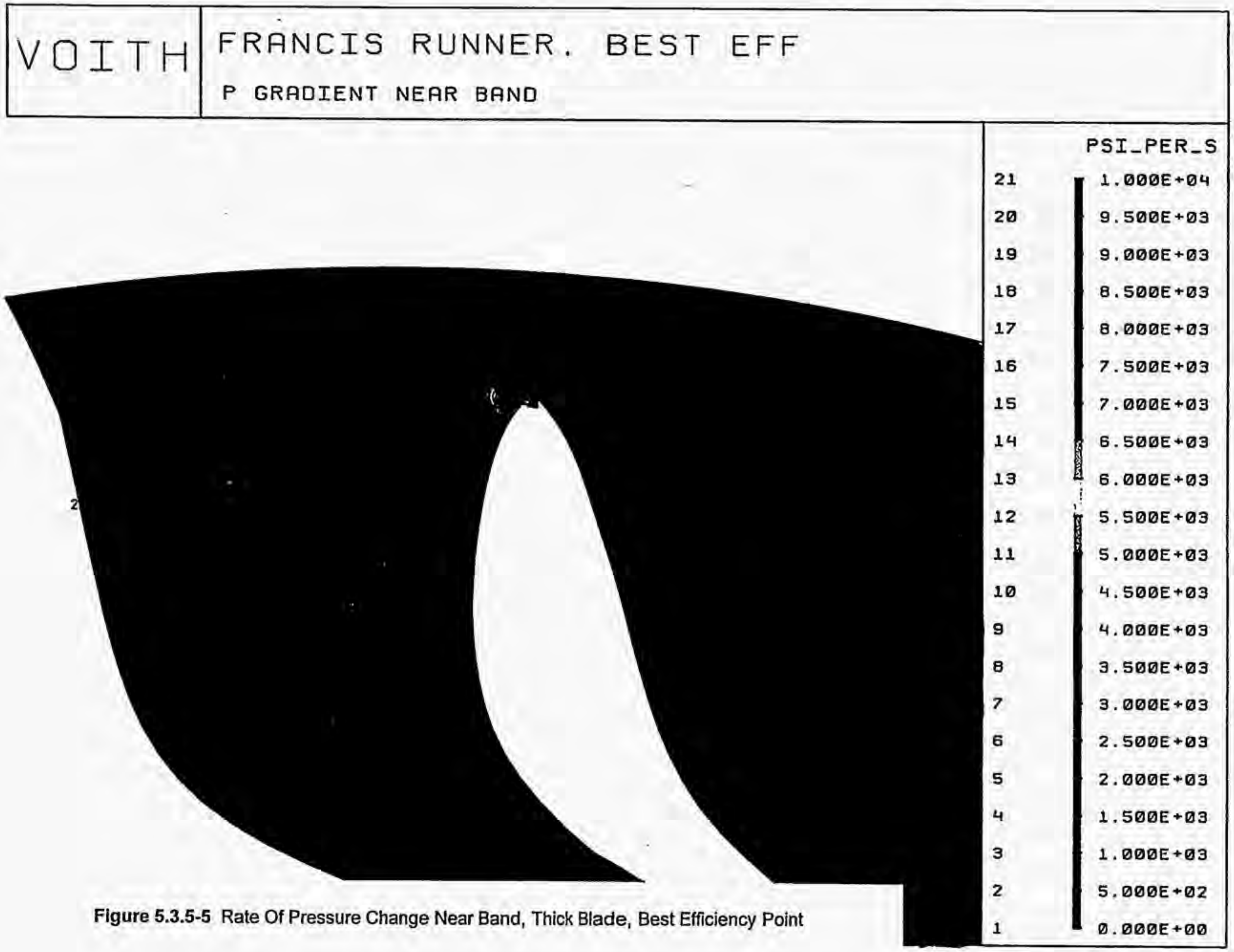

Figure 5.3.5-5 Rate Of Pressure Change Near Band, Thick Blade, Best Efficiency Point

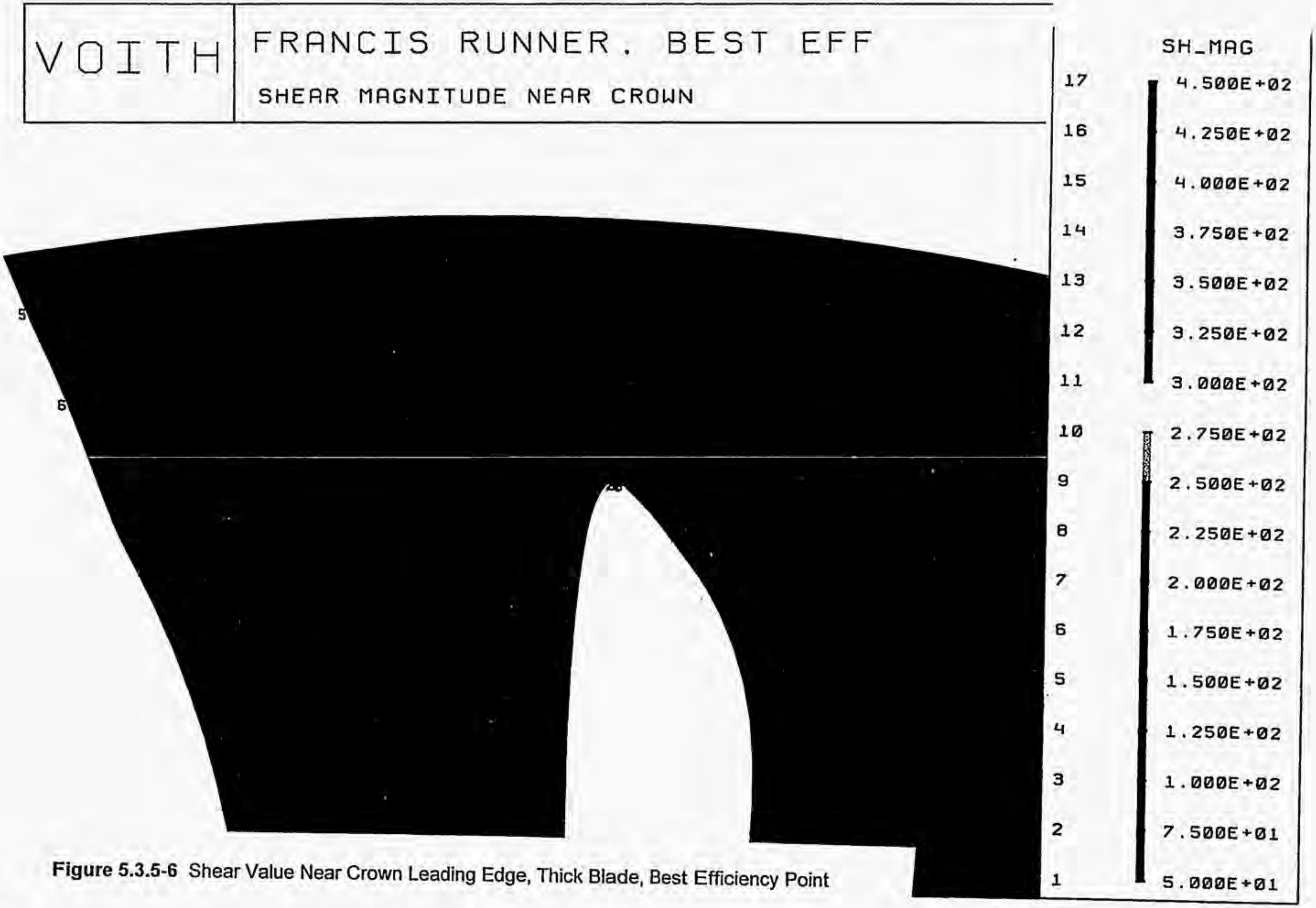

Figure 5.3.5-6 Shear Value Near Crown Leading Edge, Thick Blade, Best Efficiency Point

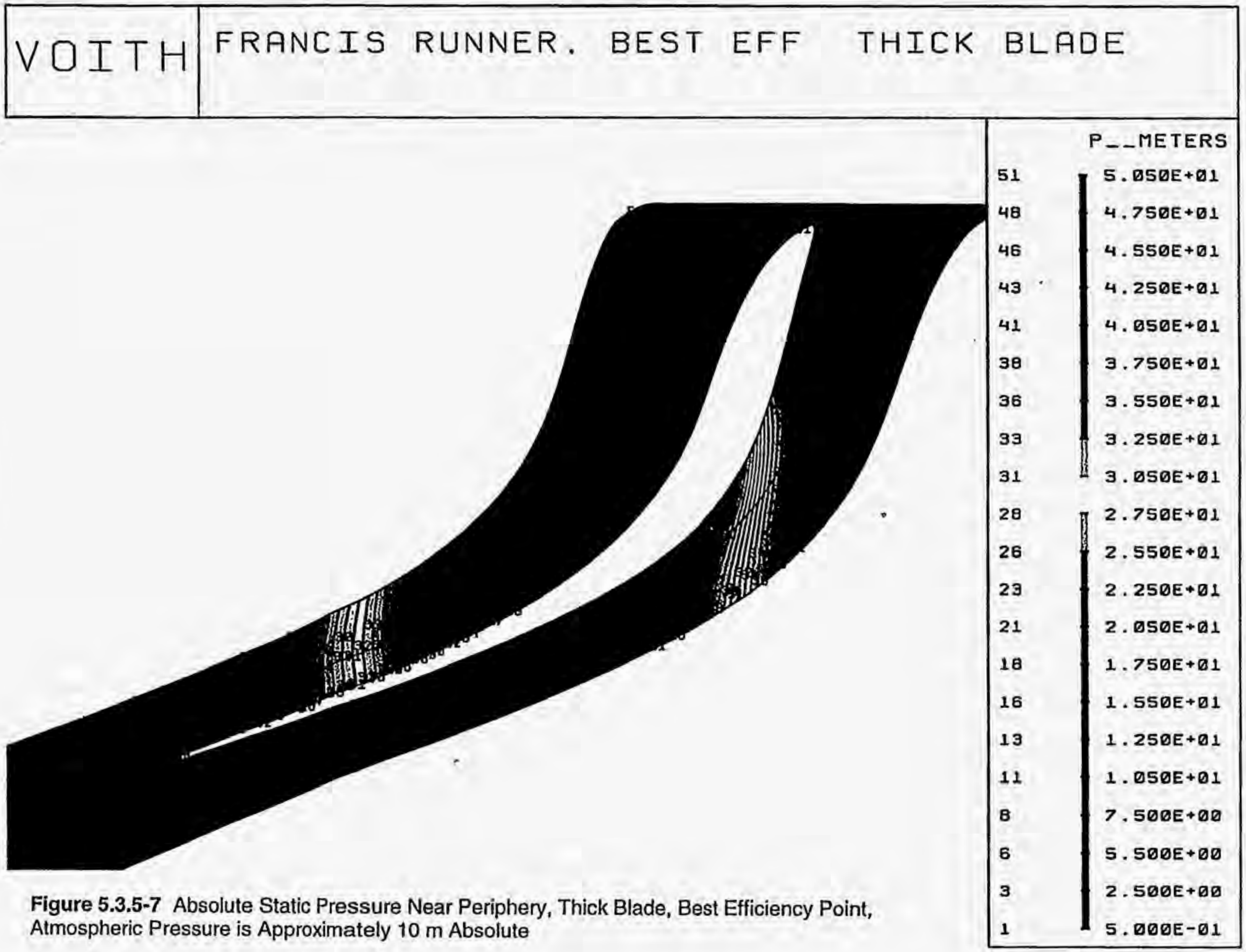

Figure 5.3.5-7 Absolute Static Pressure Near Periphery, Thick Blade, Best Efficiency Point, Atmospheric Pressure is Approximately 10 m Absolute

Figure 5.3.5-8 Contour Of Shear Magnitude Near Band, Thick Blade, Low Head Point

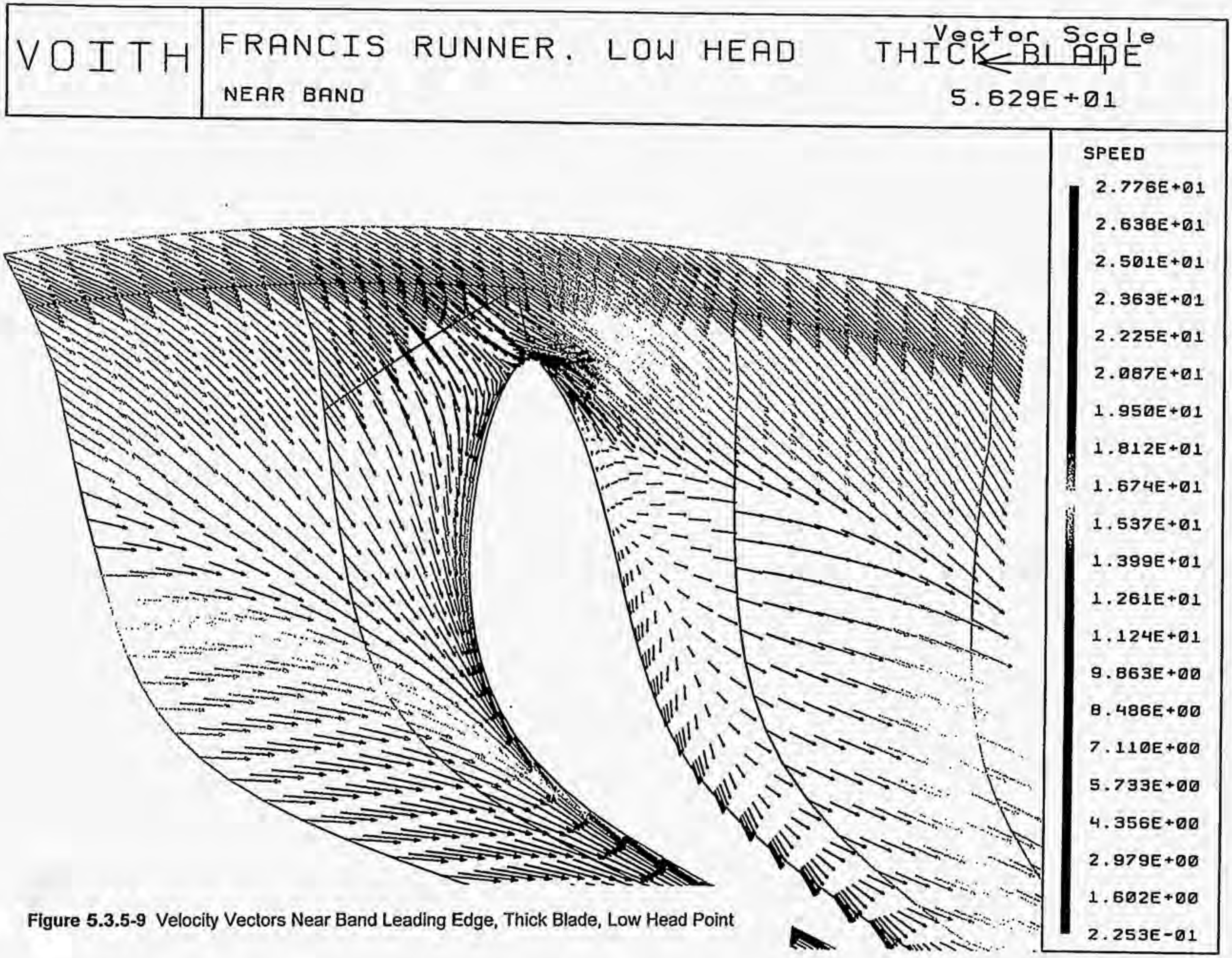

Figure 5.3.5-9 Velocity Vectors Near Band Leading Edge, Thick Blade, Low Head Point

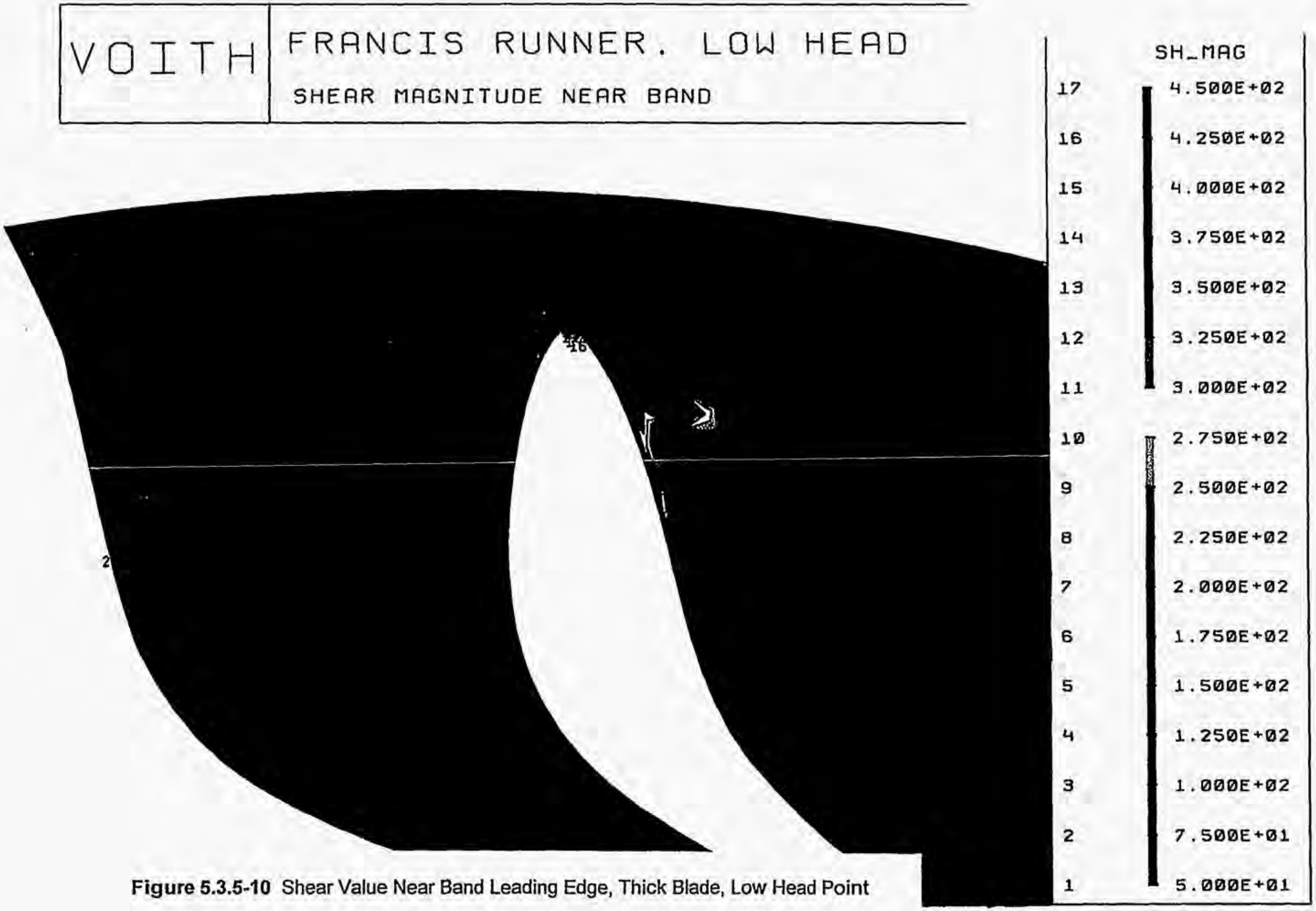

Figure 5.3.5-10 Shear Value Near Band Leading Edge, Thick Blade, Low Head Point

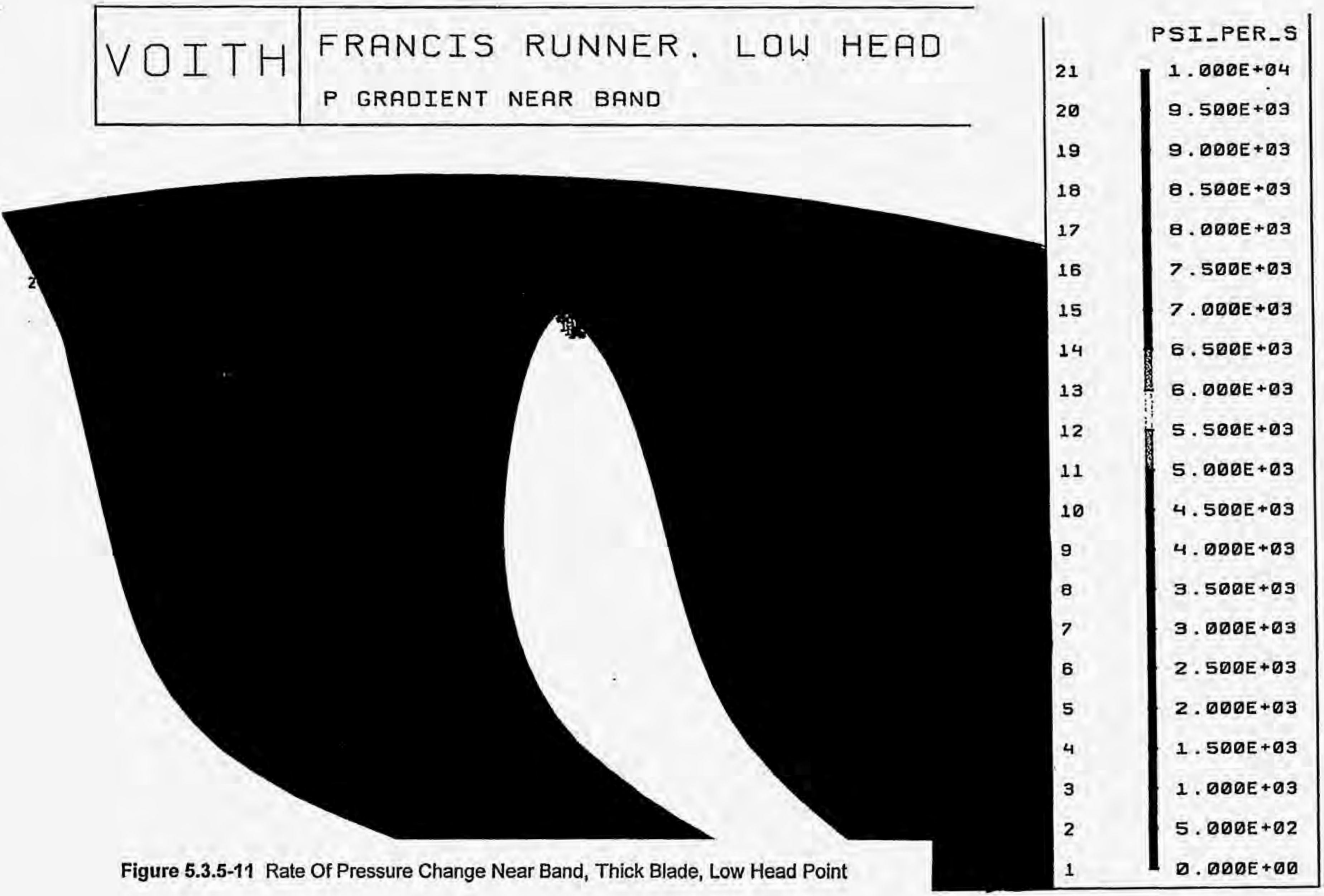

Figure 5.3.5-11 Rate Of Pressure Change Near Band, Thick Blade, Low Head Point

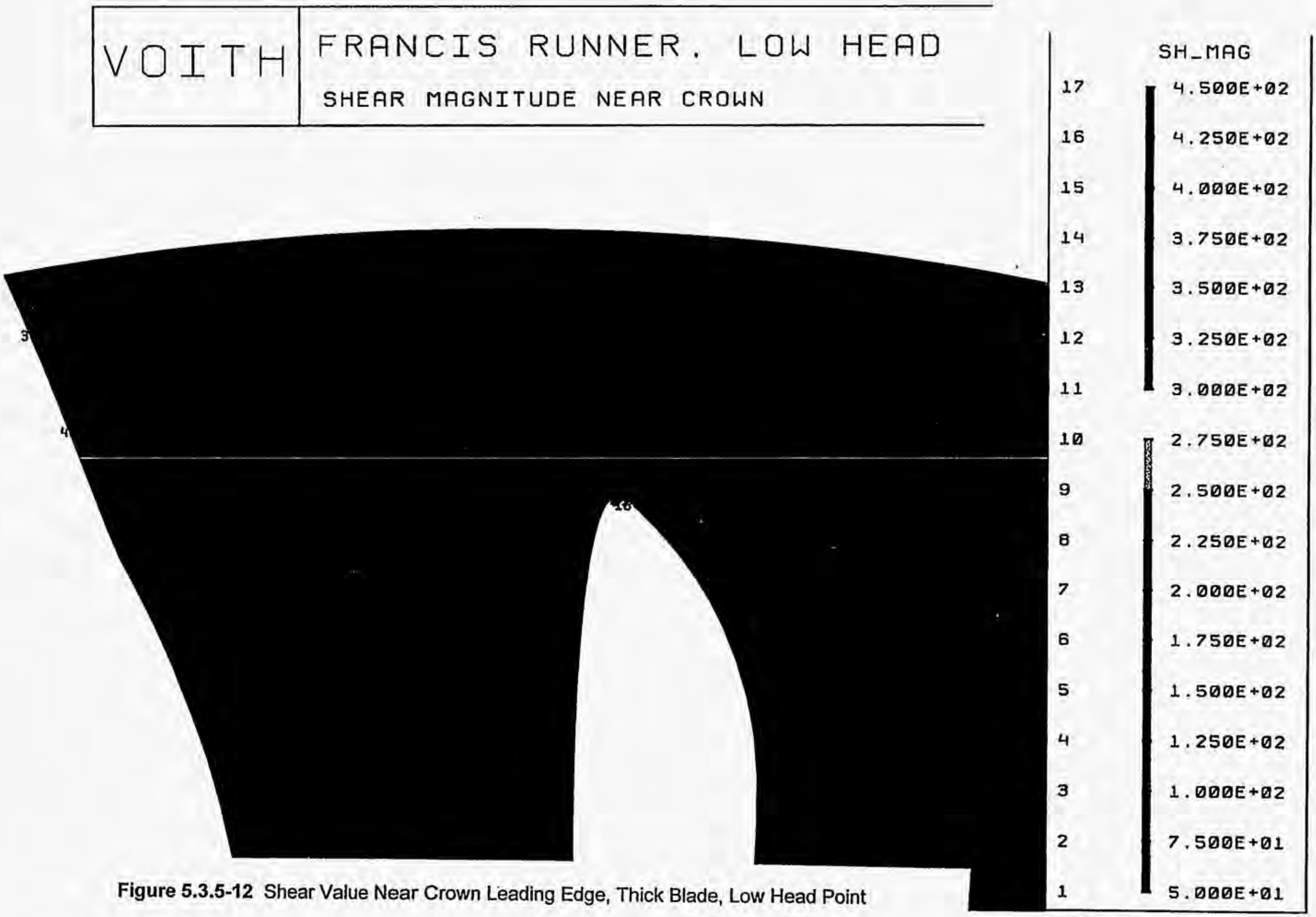

Figure 5.3.5-12 Shear Value Near Crown Leading Edge, Thick Blade, Low Head Point

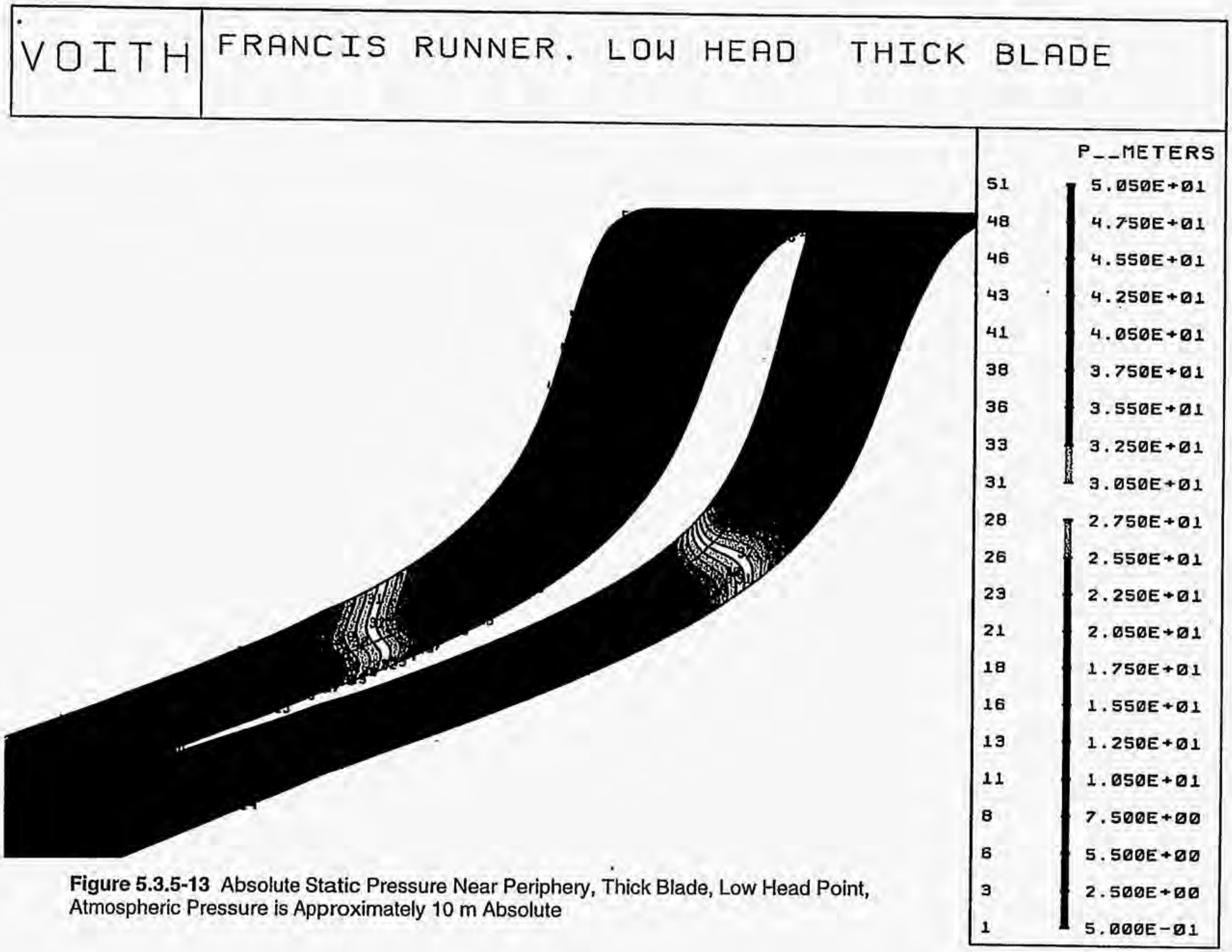

Figure 5.3.5-13 Absolute Static Pressure Near Periphery, Thick Blade, Low Head Point, Atmospheric Pressure is Approximately 10 m Absolute

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FRANCIS RUNNER BLADE COMPARISON

GREEN-THICK BLADE BLACK-THIN BLADE

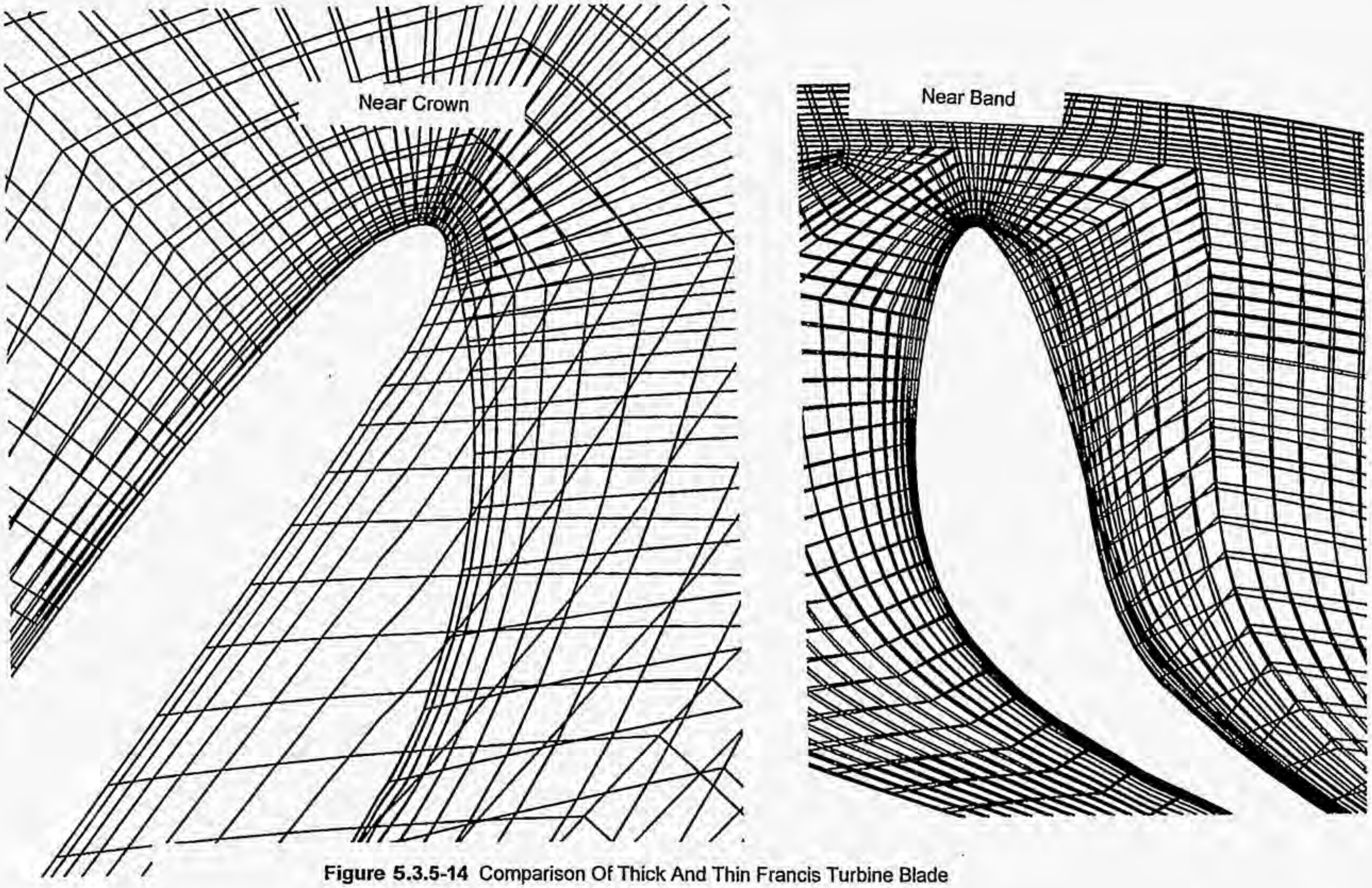

Figure 5.3.5-14 Comparison Of Thick And Thin Francis Turbine Blade

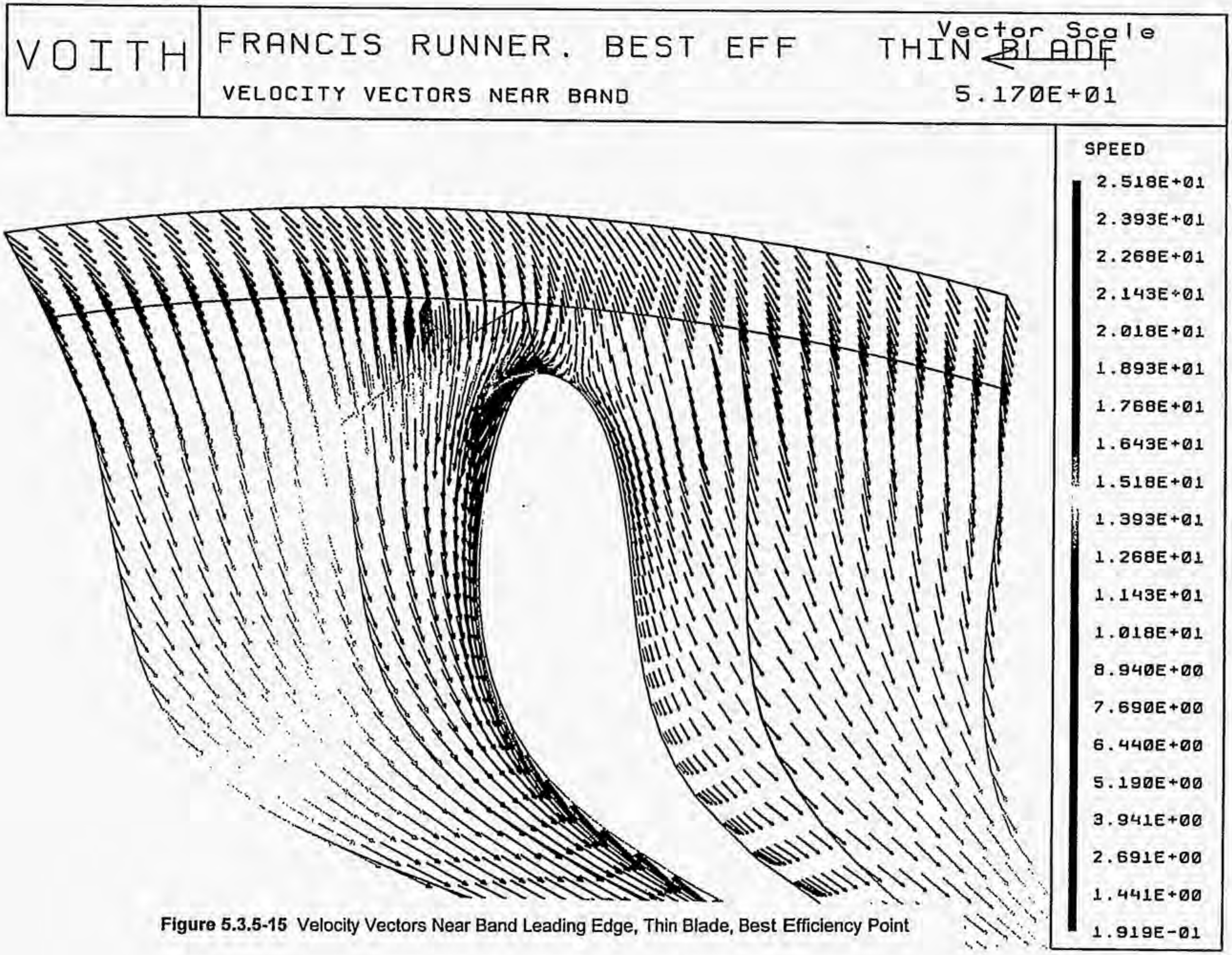

Figure 5.3.5-15 Velocity Vectors Near Band Leading Edge, Thin Blade, Best Efficiency Point

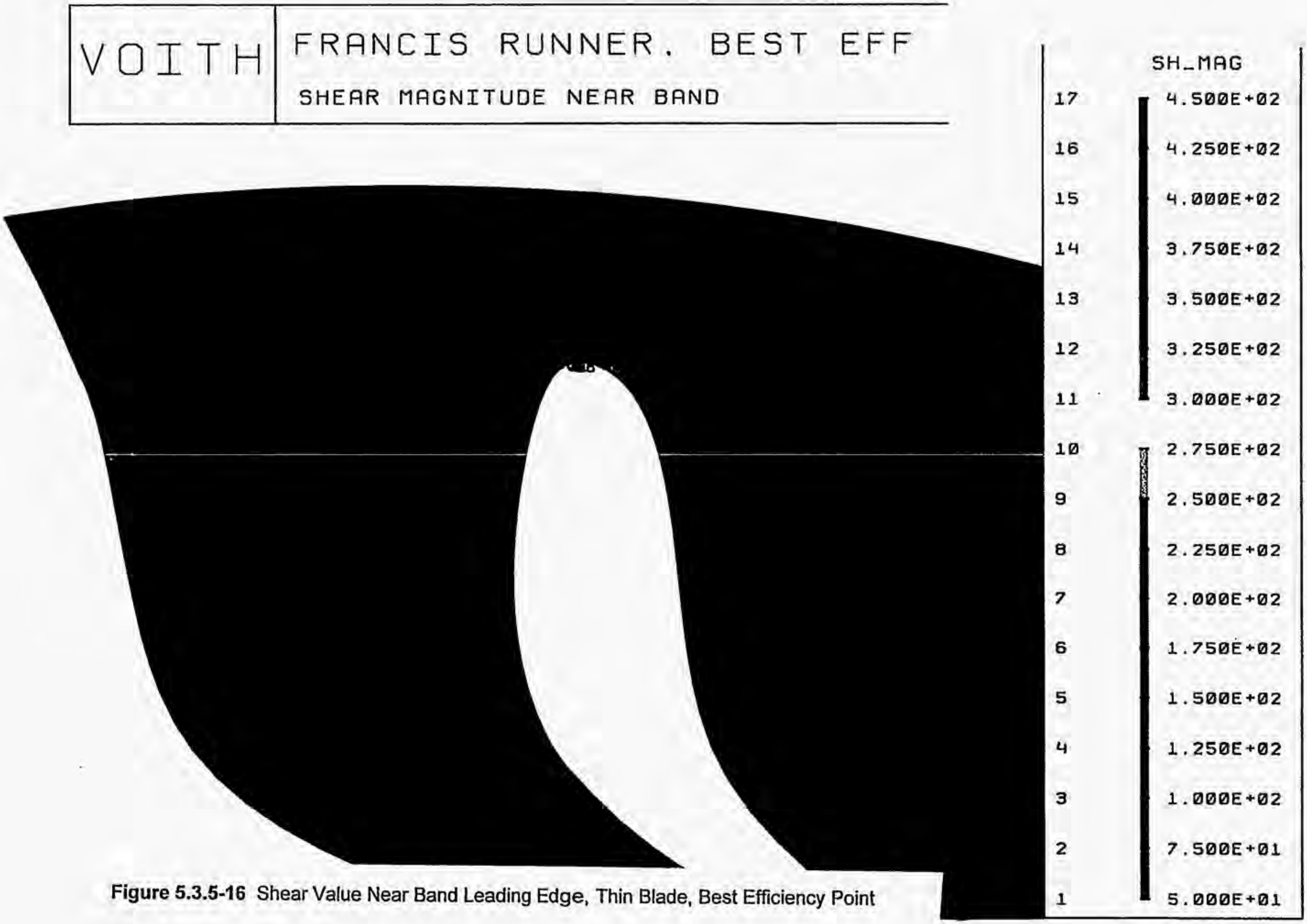

Figure 5.3.5-16 Shear Value Near Band Leading Edge, Thin Blade, Best Efficiency Point

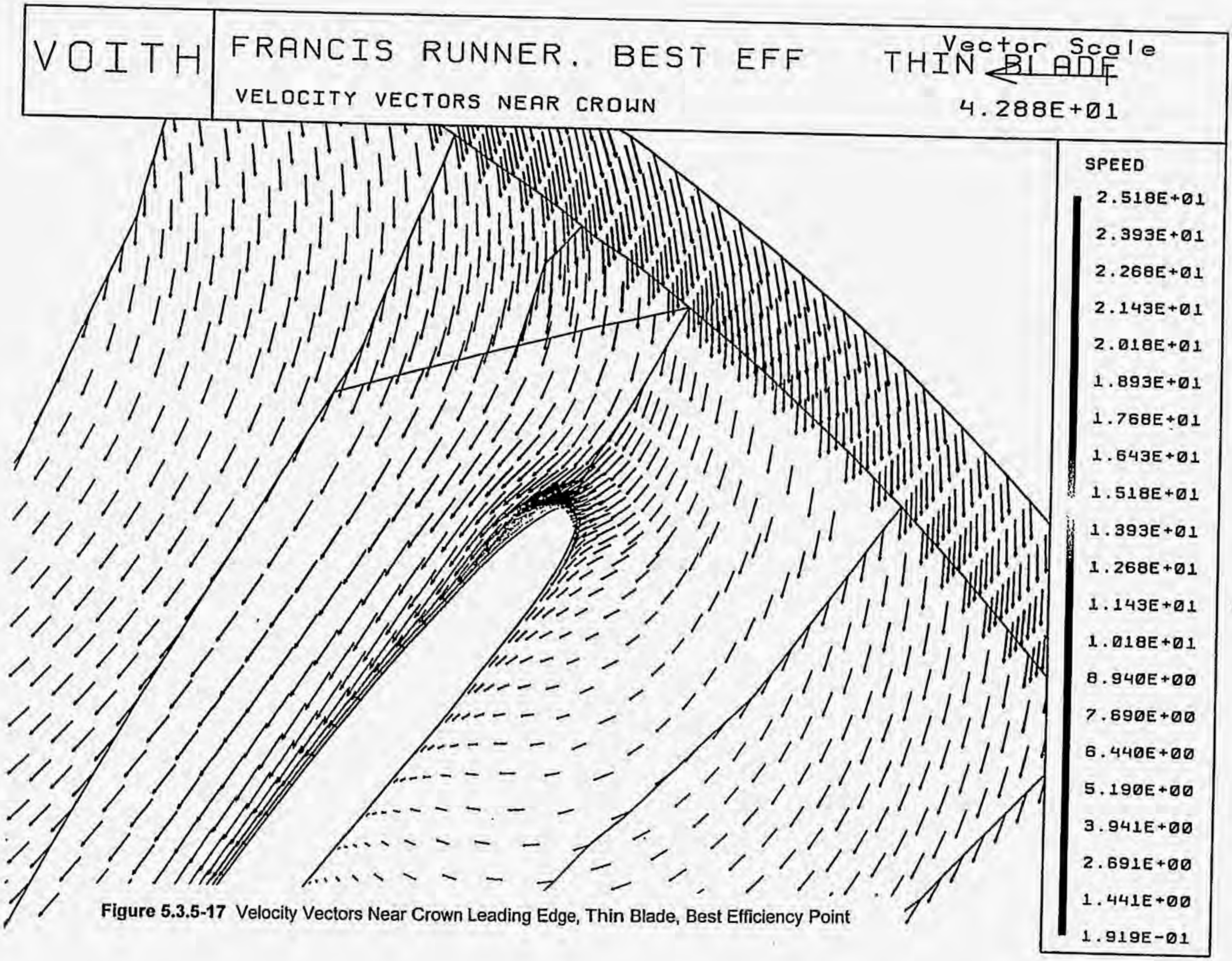

Figure 5.3.5-17 Velocity Vectors Near Crown Leading Edge, Thin Blade, Best Efficiency Point

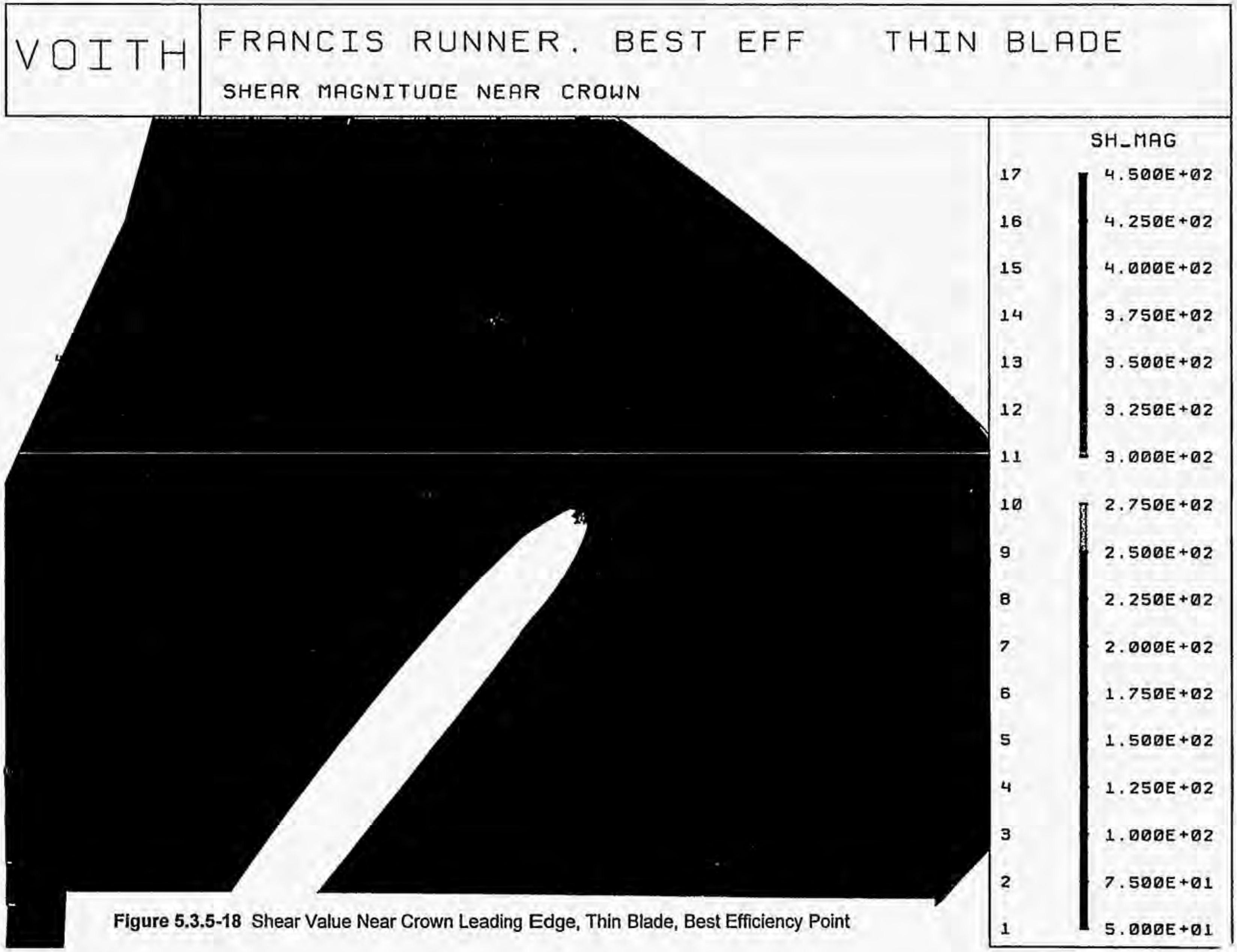

Figure 5.3.5-18 Shear Value Near Crown Leading Edge, Thin Blade, Best Efficiency Point

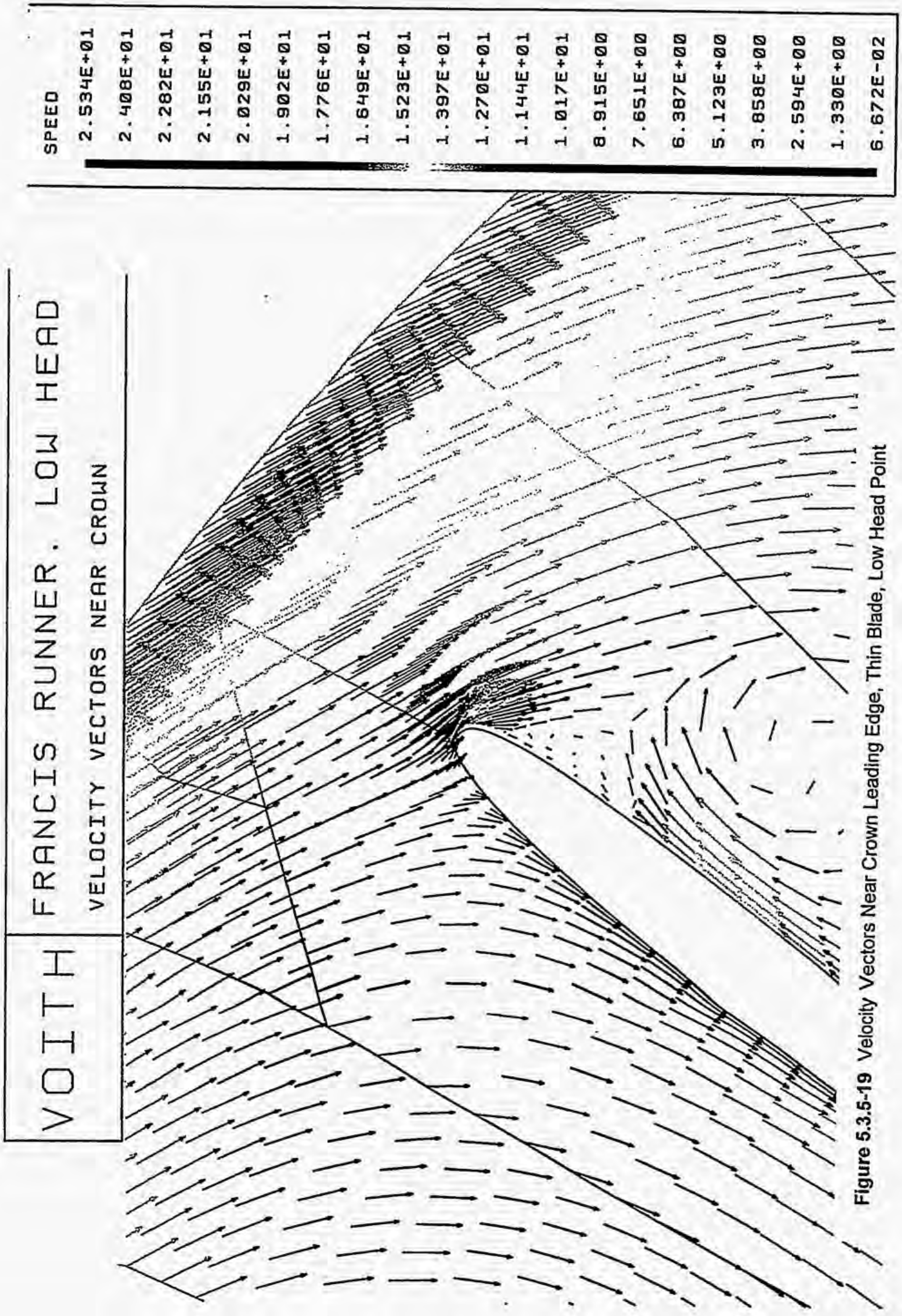

Figure 5.3.5-19 Velocity Vectors Near Crown Leading Edge, Thin Blade, Low Head Point

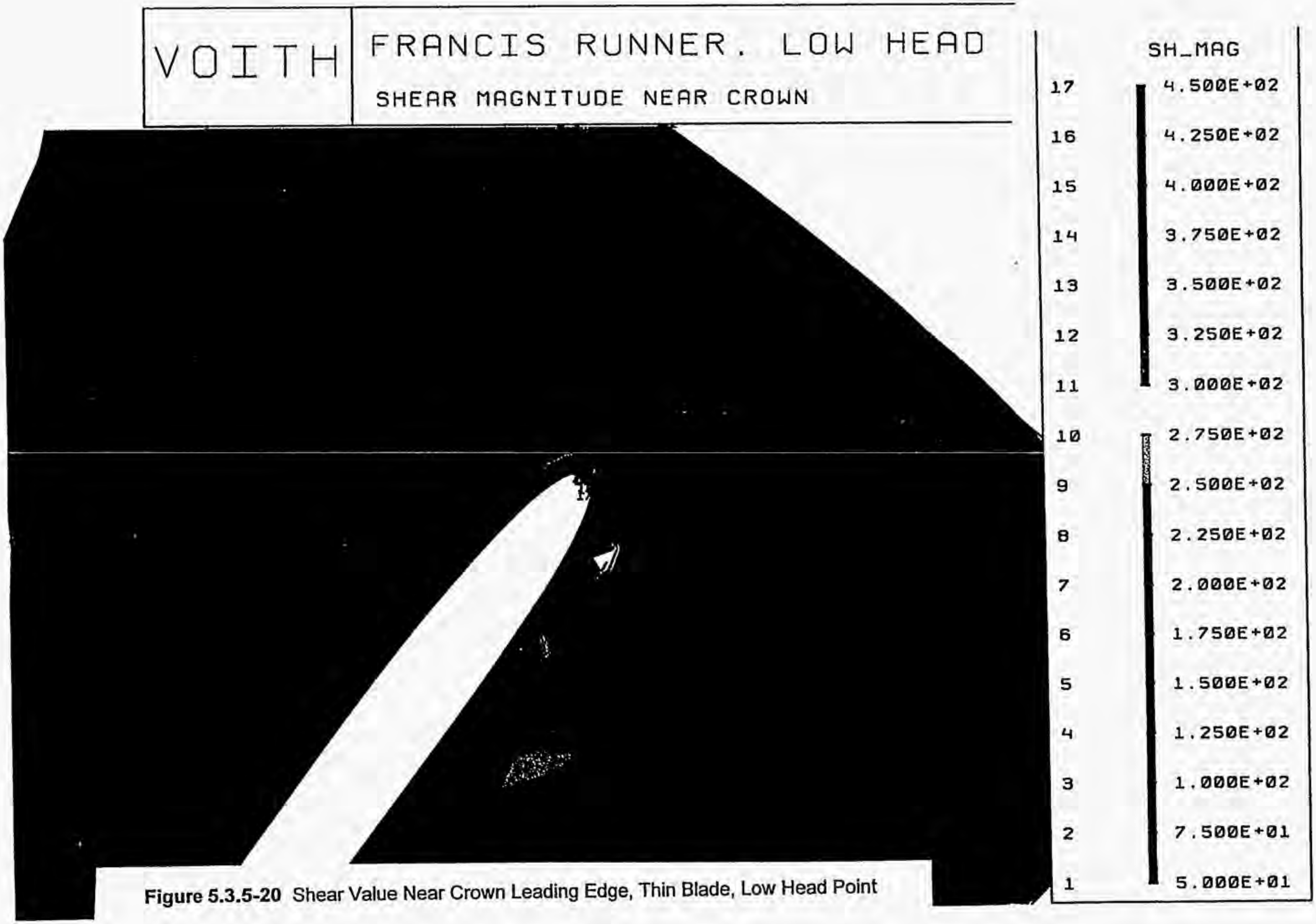

Figure 5.3.5-20 Shear Value Near Crown Leading Edge, Thin Blade, Low Head Point

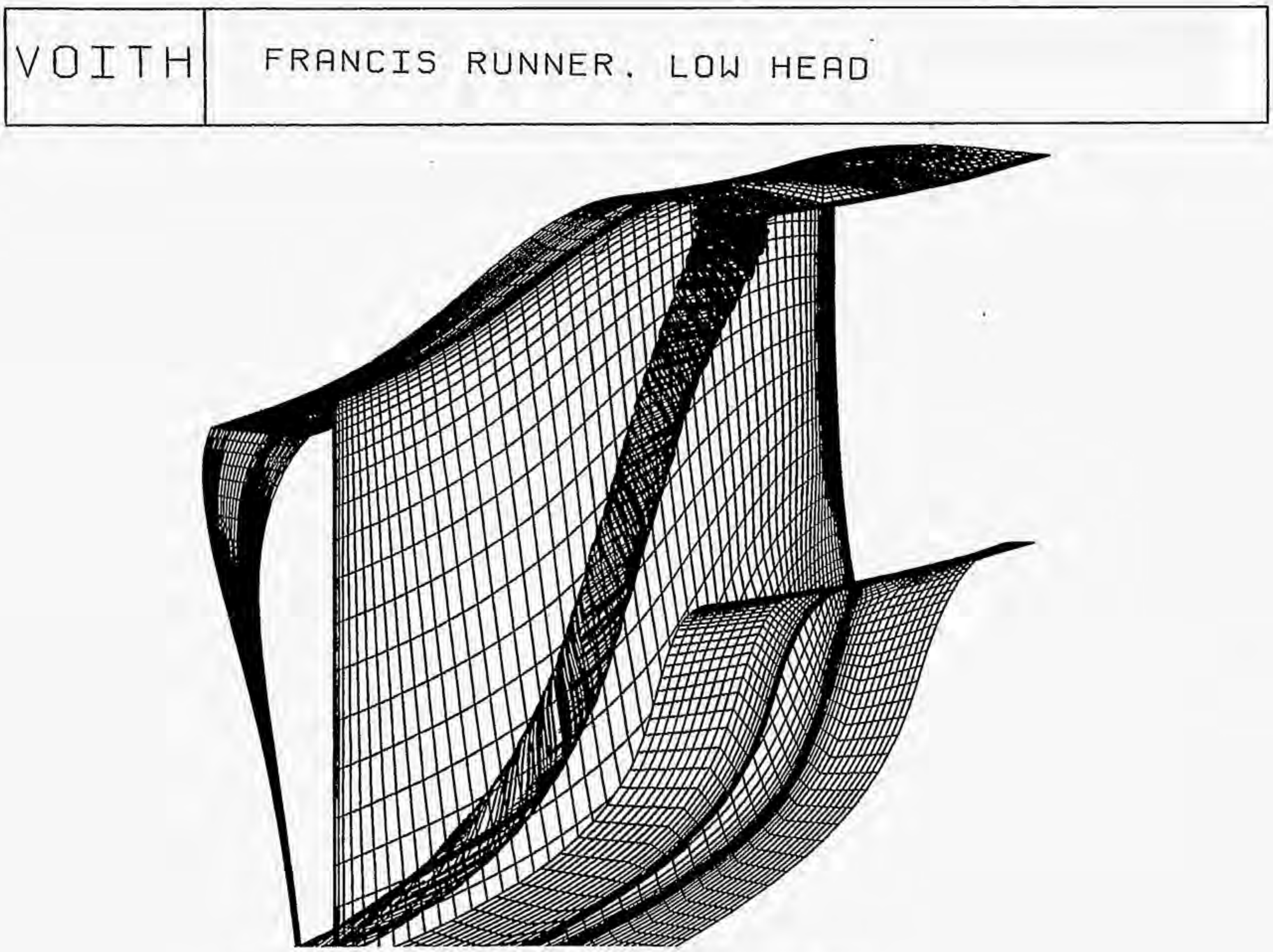

Figure 5.3.5-21 Typical Vortex At Crown Due To Low Head Operating Point

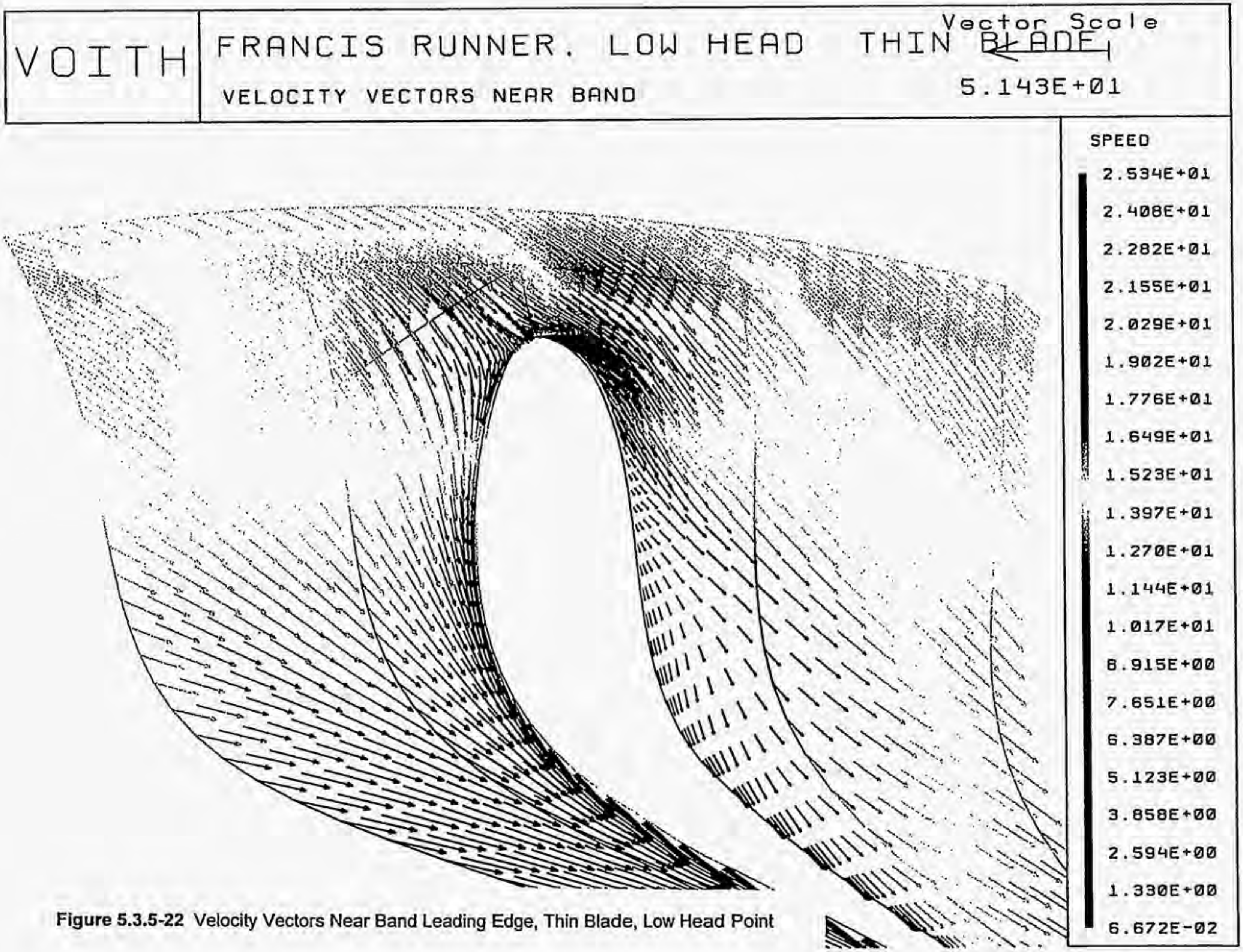

Figure 5.3.5-22 Velocity Vectors Near Band Leading Edge, Thin Blade, Low Head Point

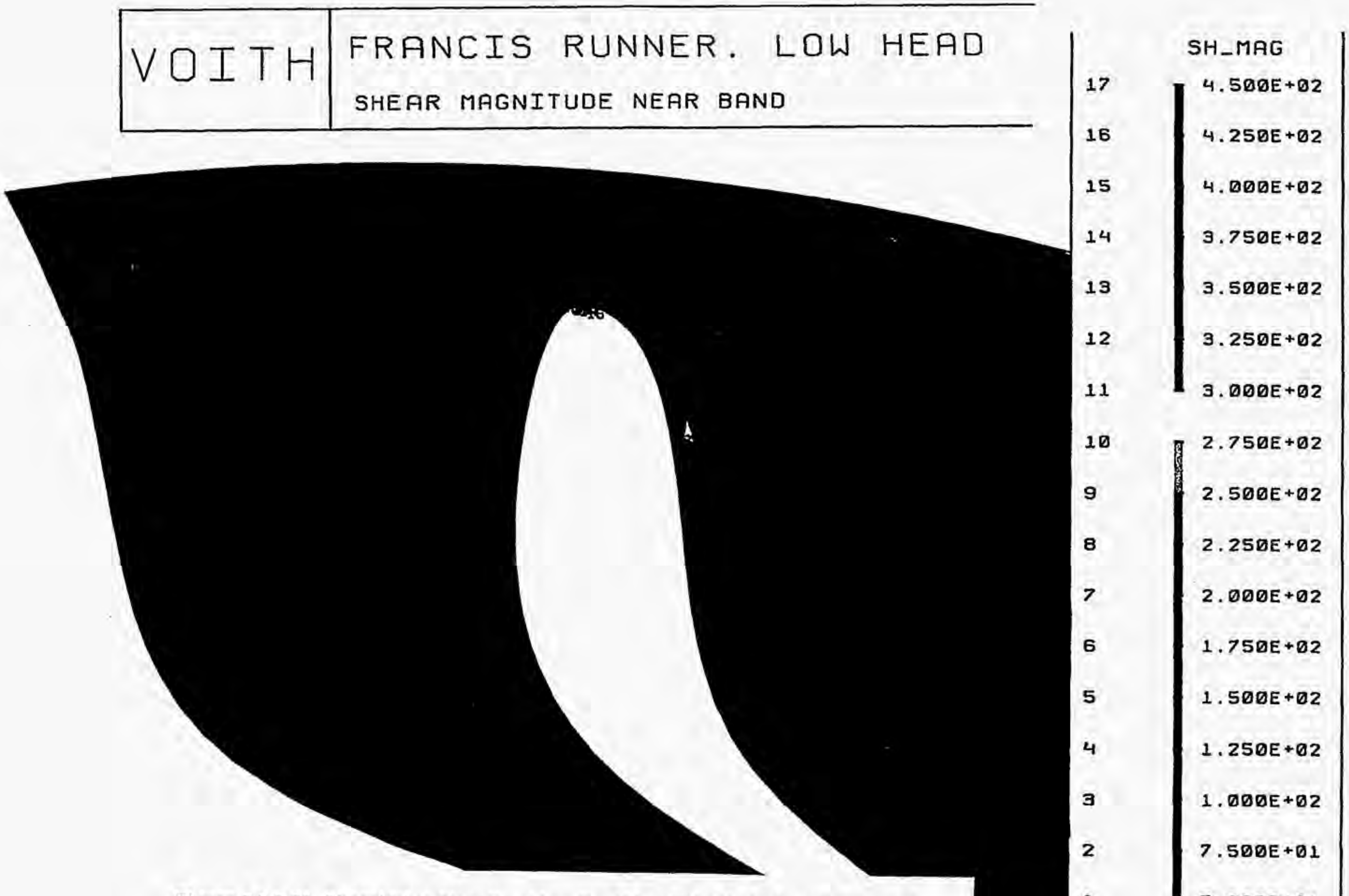

Figure 5.3.5-23 Shear Value Near Band Leading Edge, Thin Blade, Low Head Point

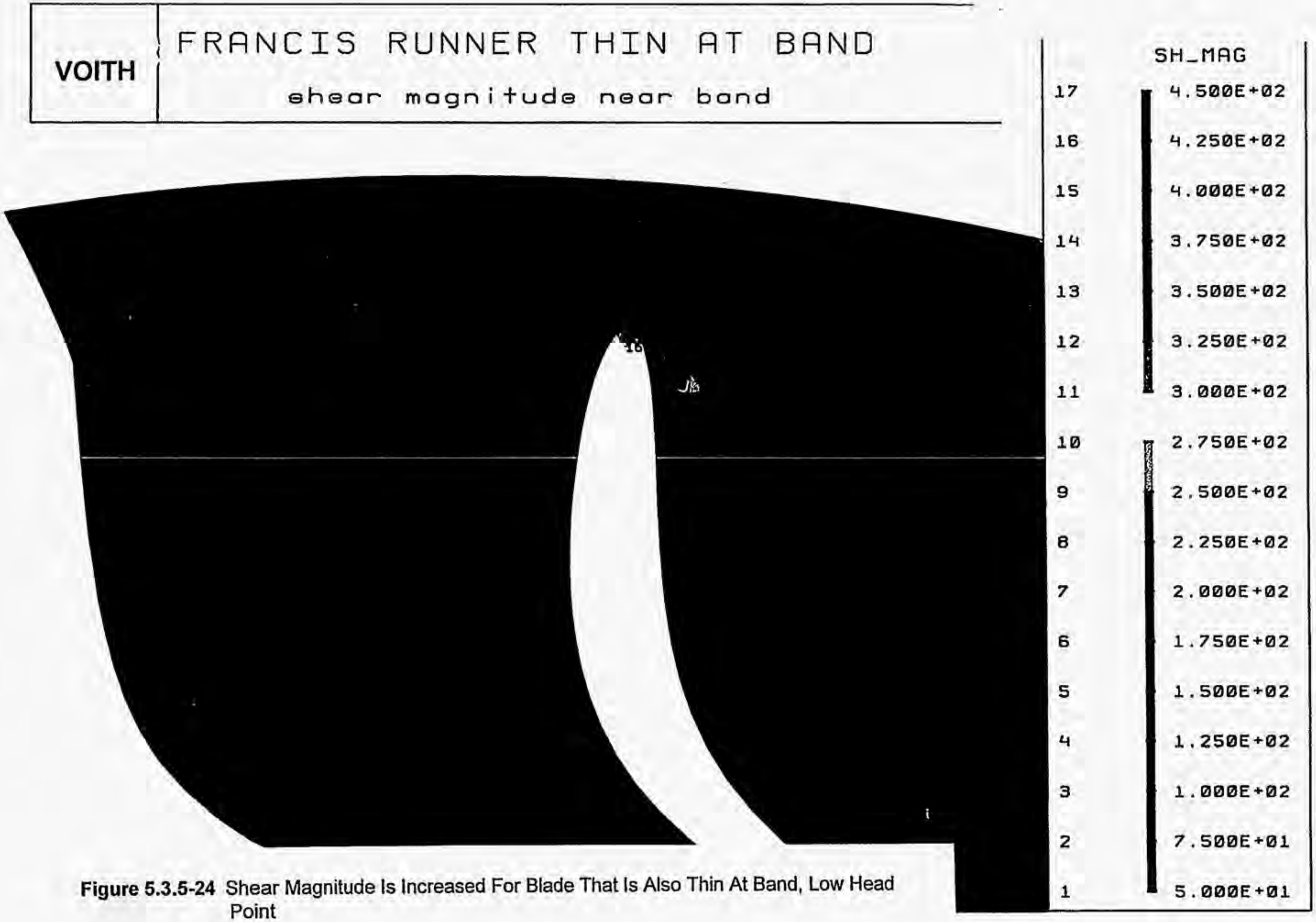

Figure 5.3.5-24 Shear Magnitude Is Increased For Blade That Is Also Thin At Band, Low Head Point

5.3.6 KAPLAN TURBINE DRAFT TUBE

Since Kaplan turbines operate on-cam, the runner discharge conditions do not vary drastically. It is the experience that the draft tube flow field therefore, remains relatively similar at all operating conditions. For this reason an extensive operating range was not examined.

For an on cam condition, a typical Kaplan draft tube (Bonneville) with a single pier was analyzed. The inflow condition was determined from laboratory measurements. The grid on the outer walls is shown on Figure 5.3.6-1. A combination of streamline plots and total pressure contours are used to characterize the results, Figures 5.3.6-2a and b and 5.3.6-3. The streamlines show what is a common theme throughout draft tube analyses performed during this study (also in Sections 5.3.7 and 5.4). The decelerating flow responds to pressure gradients and numerous other factors to create a highly complex pattern. Some streamlines flow in a direct and smooth path to the exit, while adjacent streamlines exhibit features of a vortex or recirculation zone. The existence of such phenomena in a Kaplan turbine draft tube, where one would expect that the on cam or maximum efficiency condition would yield the optimum draft tube conditions, indicates that these complex flow features are unavoidable aspects of decelerating flow in a bended channel that cannot be designed away. Figure 5.3.6-3 shows a section through the centerline of the draft tube, ending on the pier nose. It also shows that the bottom of the hub extends into the draft tube. The flow disturbance in this region (also shown by streamline vortex patterns in Figure 4.4.6-15) is shown her by total pressure contours. Total pressure is a measure of the energy in the flow, a combination of static pressure and kinetic energy. This figure illustrates that the area of the draft tube affected by the hub flow is quite large and is expected to have a significant influence on the entire flow field.

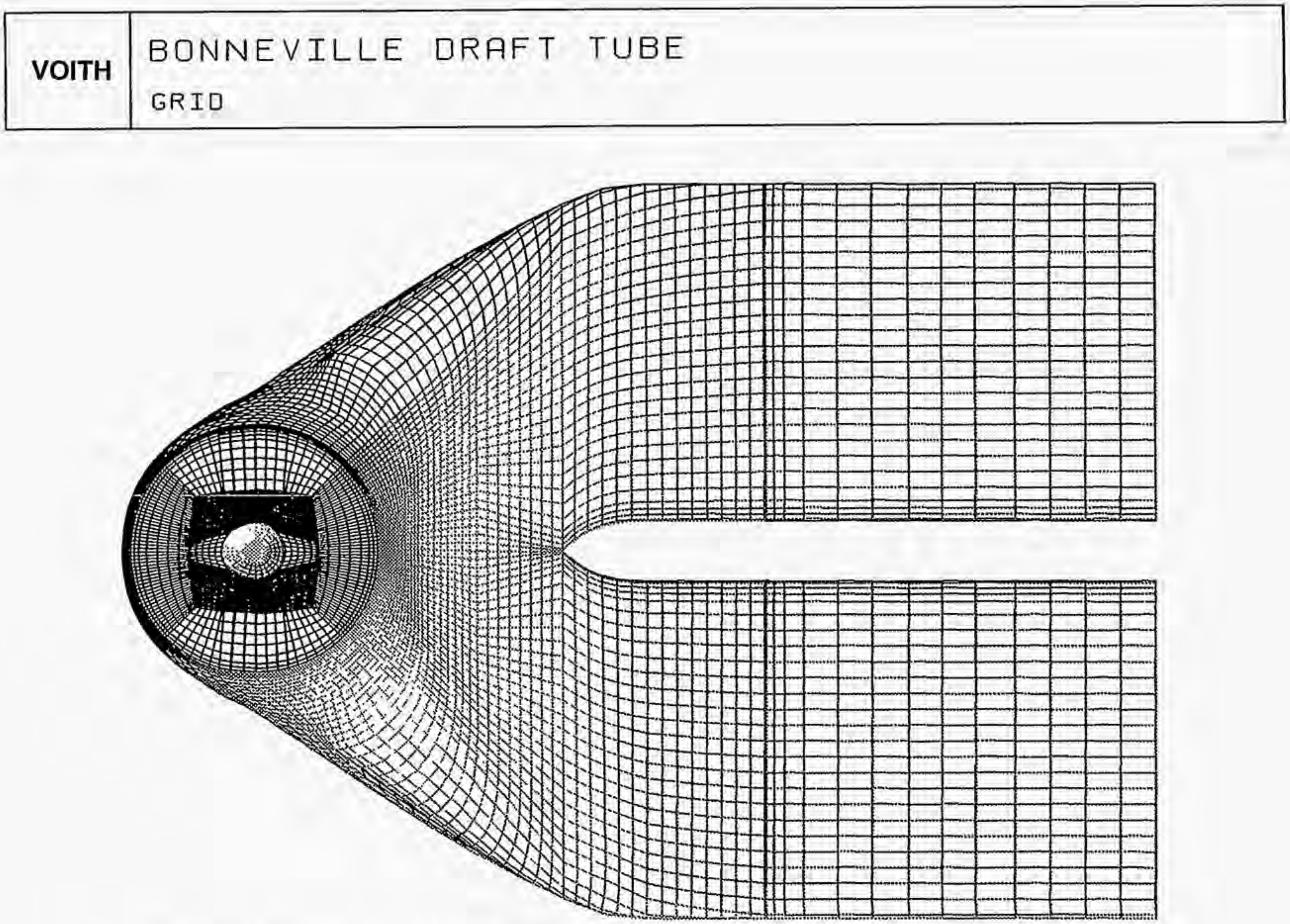

Figure 5.3.6-1 Typical Grid For A Kaplan Draft Tube With A Single Pier

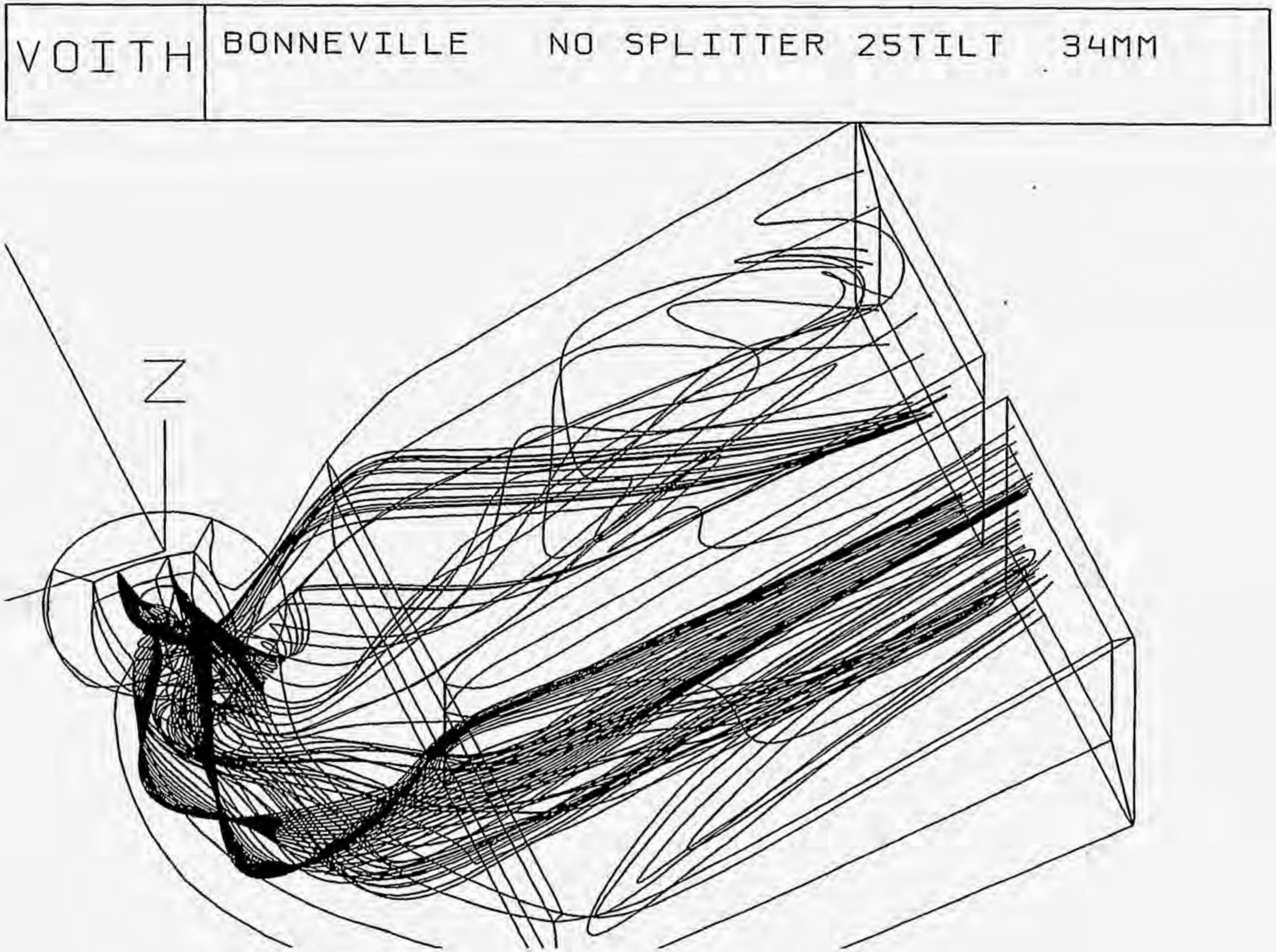

Figure 5.3.6-2a Streamlines Are Complex, Even For An On Cam Operating Point

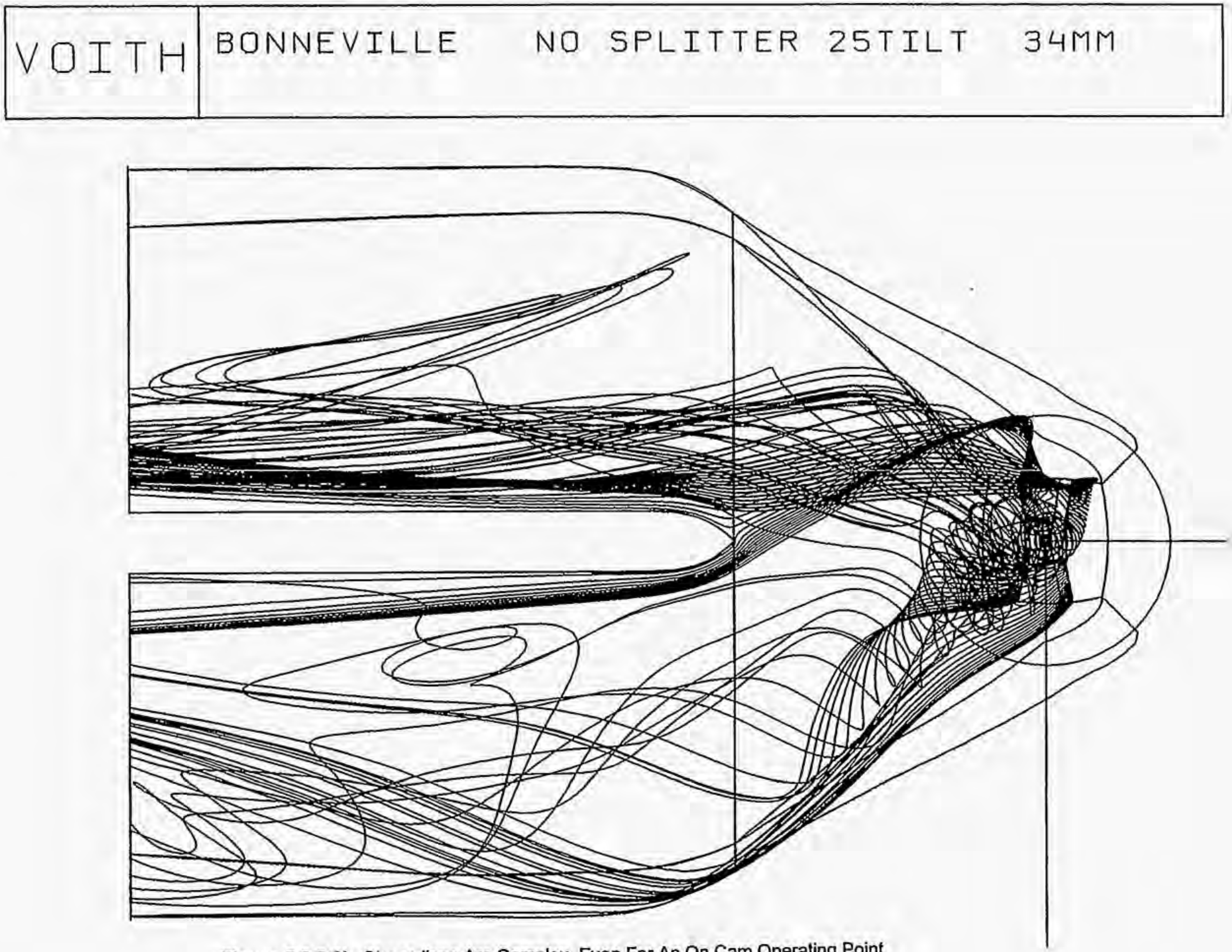

Figure 5.3.6-2b Streamlines Are Complex, Even For An On Cam Operating Point

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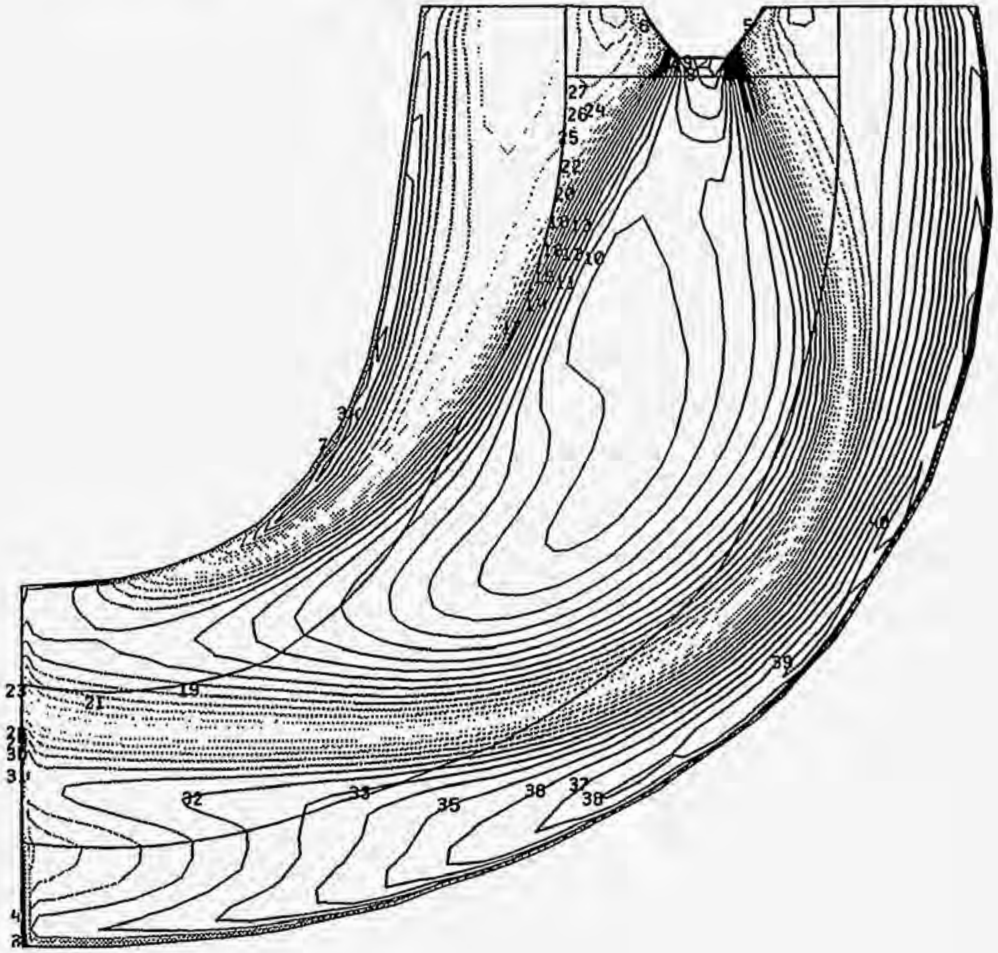
Figure 5.3.6-3 Total Pressure Contour On The Centerline Plane, Up To Nose Of Pier, Showing Hub Contour In Draft Tube And Large Region Of Low Energy



	PTOTAL
41	1.649E+04
39	1.537E+04
37	1.425E+04
35	1.313E+04
33	1.201E+04
31	1.089E+04
29	9.773E+03
27	8.653E+03
25	7.533E+03
23	6.413E+03
21	5.293E+03
19	4.173E+03
17	3.053E+03
15	1.933E+03
13	8.135E+02
11	-3.064E+02
9	-1.426E+03
7	-2.546E+03
5	-3.666E+03
3	-4.786E+03
1	-5.906E+03

5.3.7 FRANCIS TURBINE DRAFT TUBE

A Francis turbine draft tube without piers was chosen for analysis. Three operating points were calculated: maximum output, best efficiency, and low power output. The different discharge characteristics of a Francis turbine runner have different effects on the draft tube flow field. Grids on the outer surface are shown on Figure 5.3.7-1.

Figures 5.3.7-2 through 5.3.7-6 show velocity vectors, and streamlines for the maximum output condition. This condition has a complex inlet condition with a combination of clockwise and counterclockwise swirl. Velocity nonuniformities become accentuated, the secondary flows form a complex pattern, and streamlines show considerable vortex like behavior.

For the best efficiency condition, Figures 5.3.7-7 through 5.3.7-11 show the same flow field information as for the maximum power figures. The inlet flow field is much more uniform, resulting in a much more uniform velocity pattern. Complex secondary flows still exist and vortex patterns are still evident in the streamlines.

At low output, Figures 5.3.7-12 through 5.3.7-15 show a variety of velocity vectors and the total pressure on the symmetry plane. The pattern of the inflow is essentially opposite of the high power condition. A low velocity region in the center region dominates the flow field, resulting in a large recirculation region that extends to the draft tube inlet surface. This flow would have a tendency to reenter the runner. Typically, this operating condition is not stable, and the unsteady flow is manifest in pressure pulsations that are a limitation in plant operation. Unsteady calculations are discussed in Section 5.4.3.

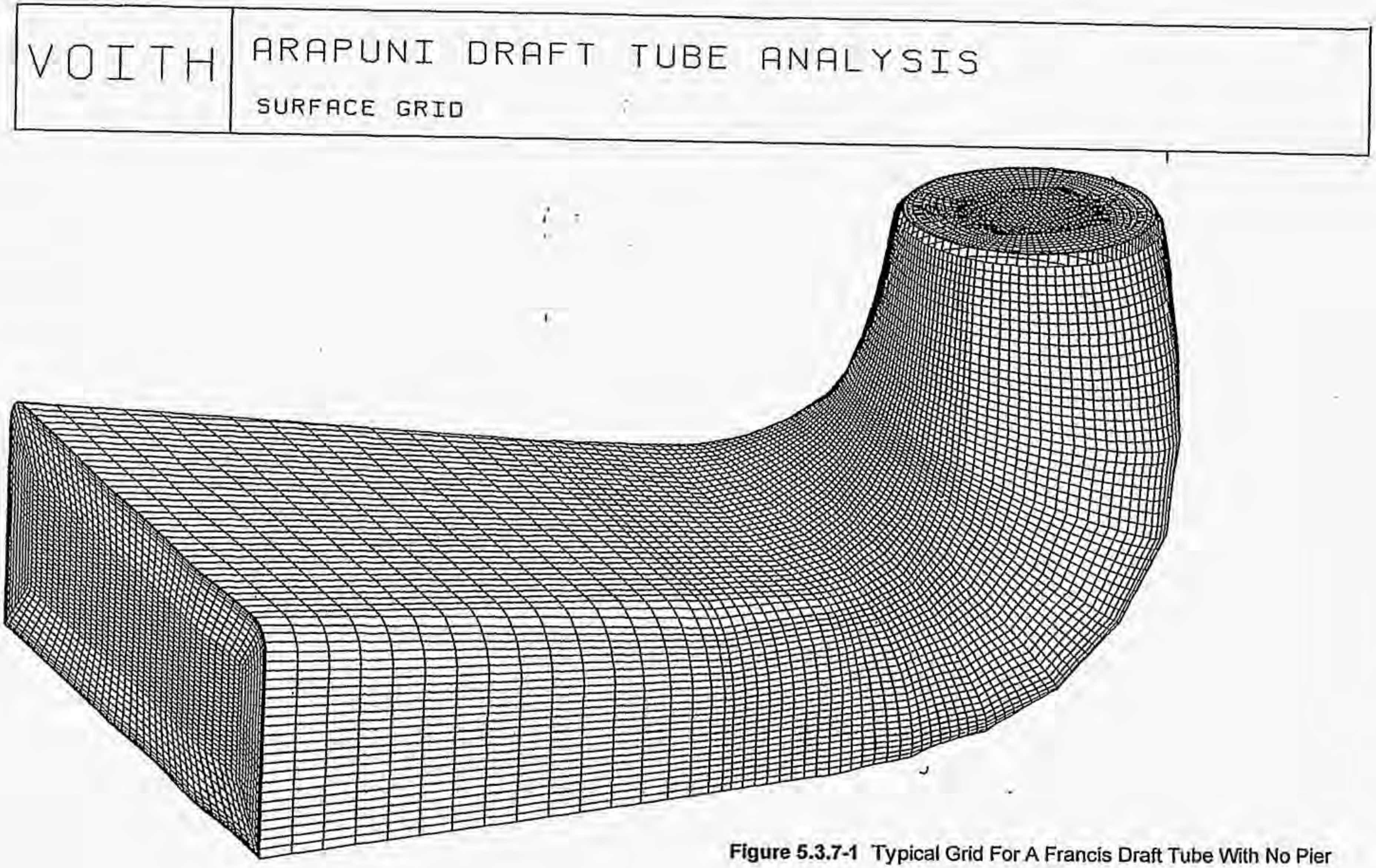


Figure 5.3.7-1 Typical Grid For A Francis Draft Tube With No Pier

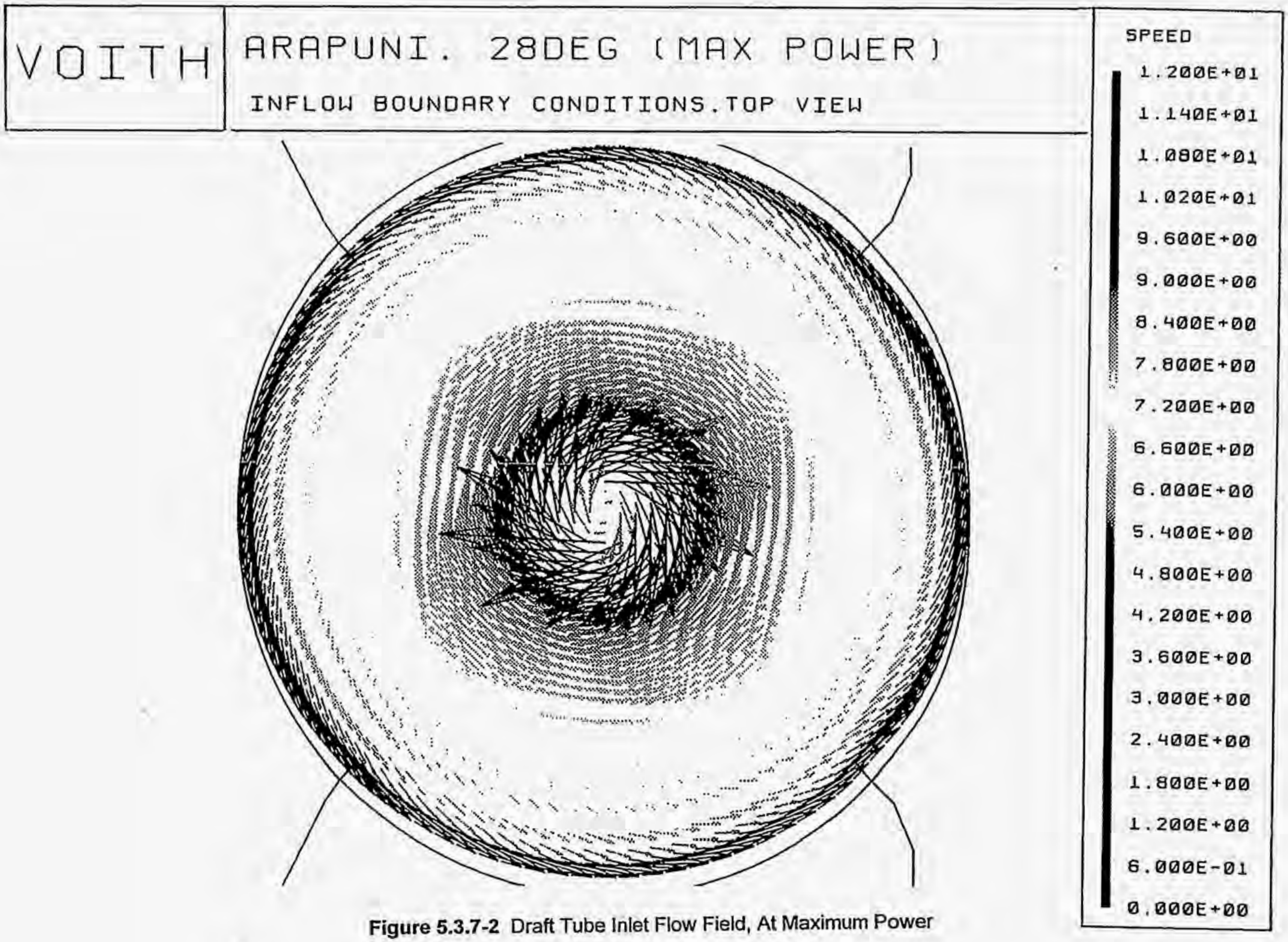


Figure 5.3.7-2 Draft Tube Inlet Flow Field, At Maximum Power

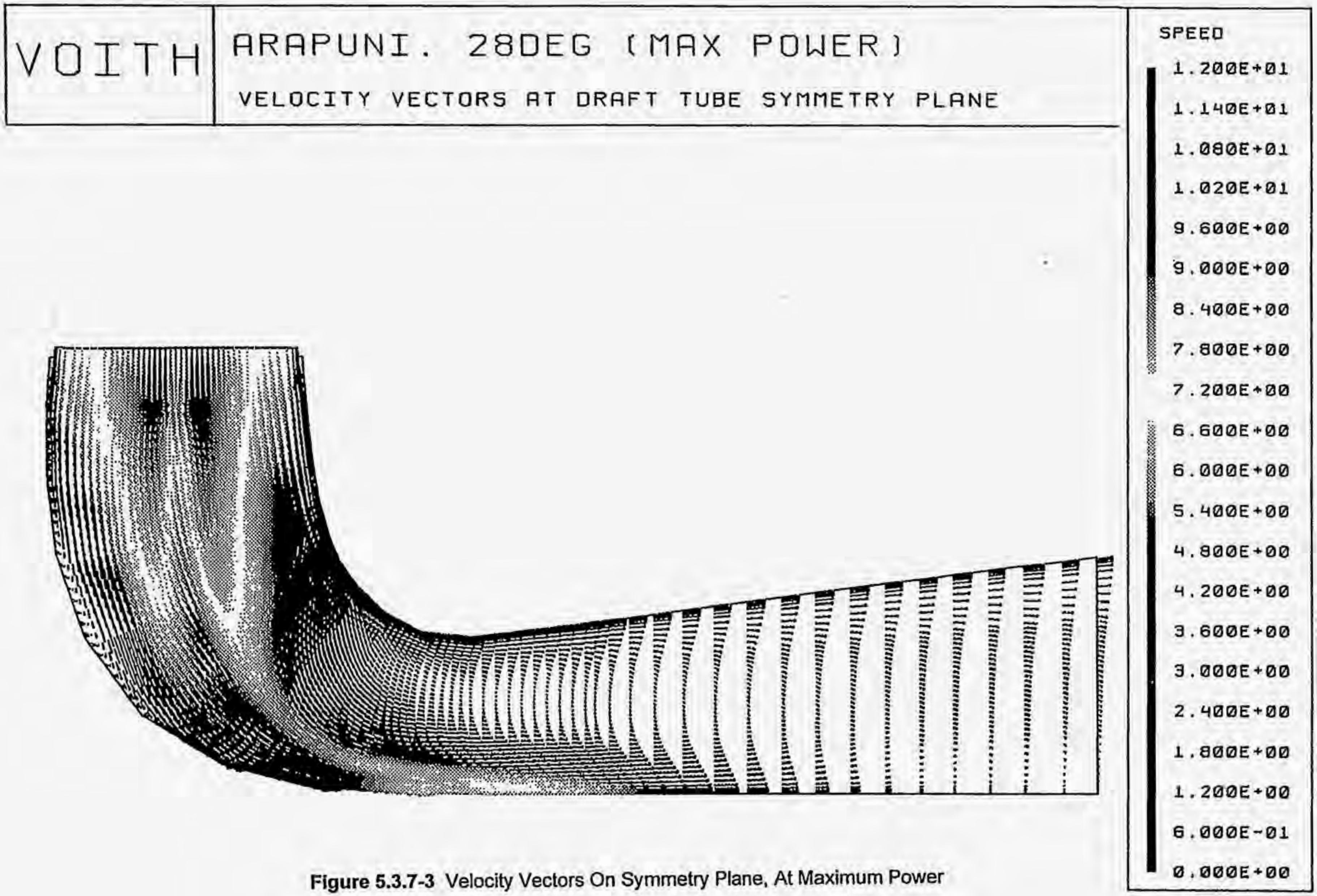


Figure 5.3.7-3 Velocity Vectors On Symmetry Plane, At Maximum Power

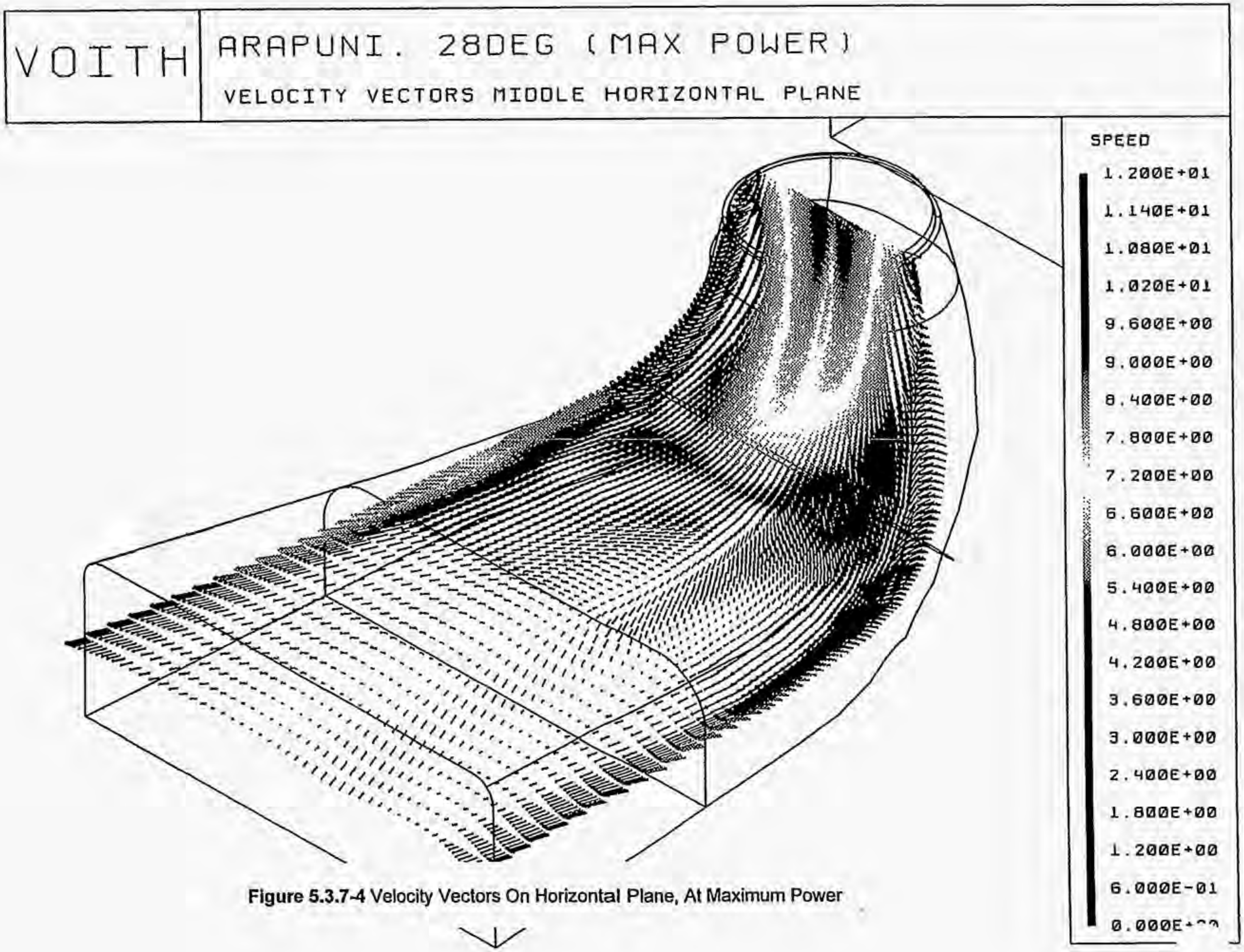


Figure 5.3.7-4 Velocity Vectors On Horizontal Plane, At Maximum Power

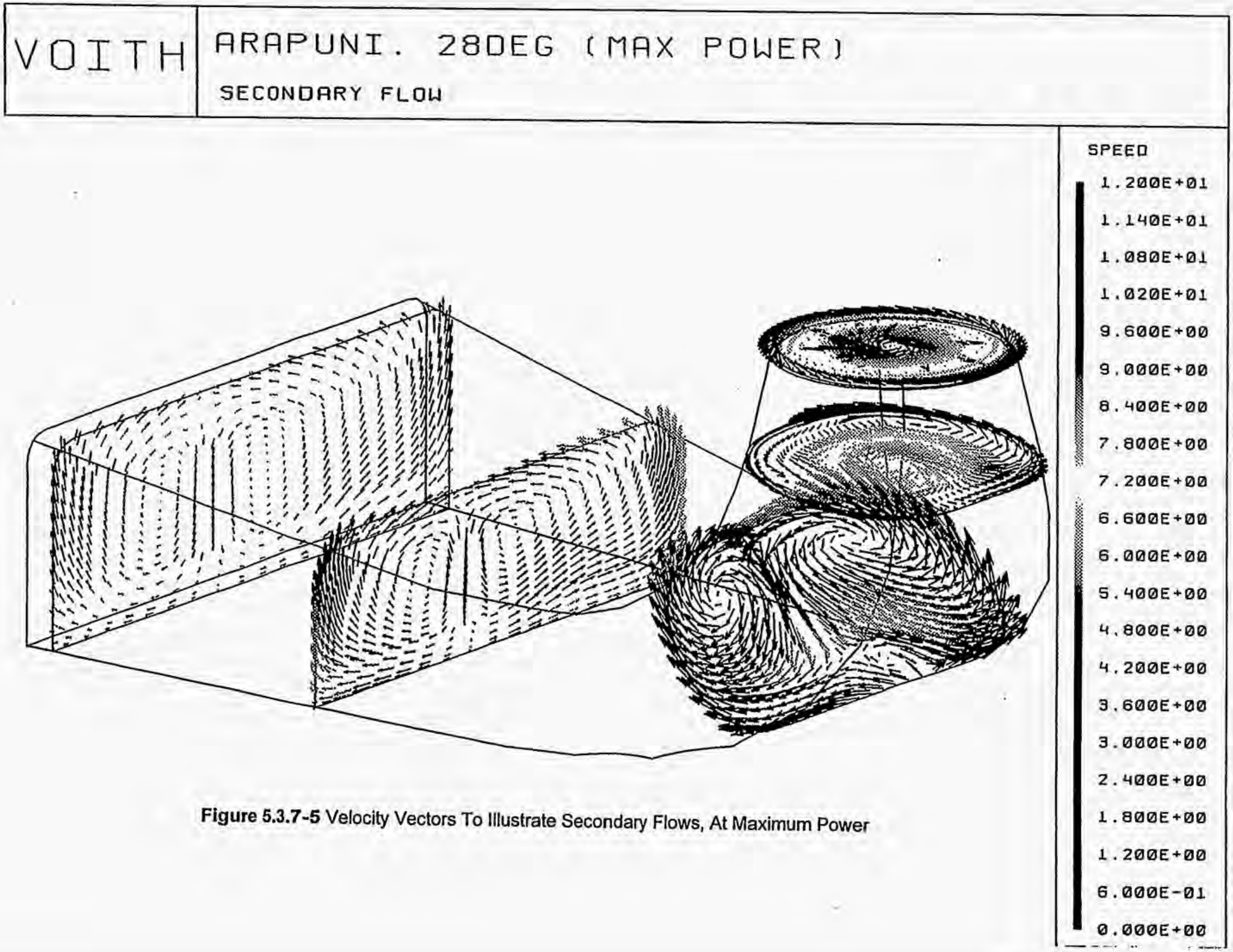


Figure 5.3.7-5 Velocity Vectors To Illustrate Secondary Flows, At Maximum Power

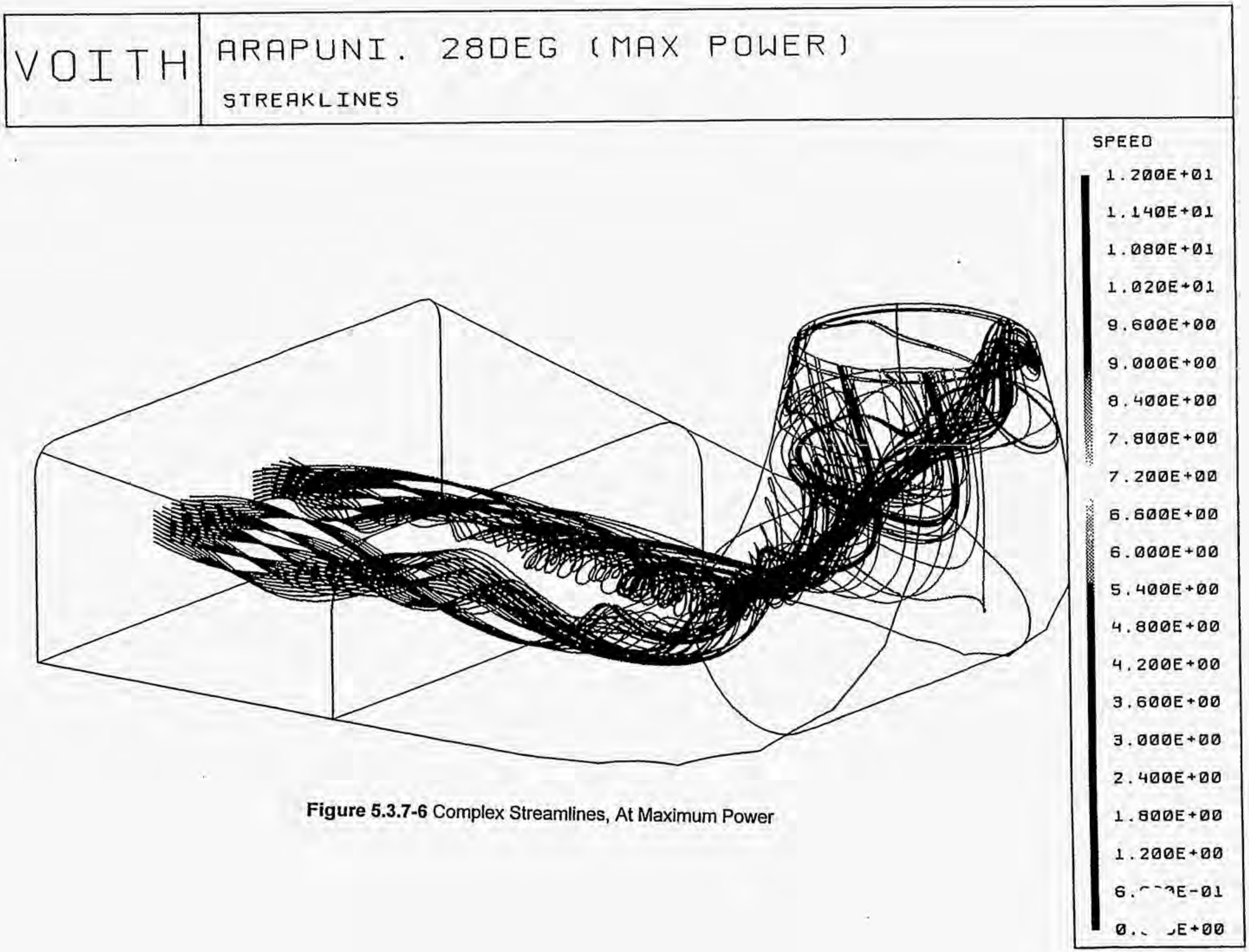


Figure 5.3.7-6 Complex Streamlines, At Maximum Power

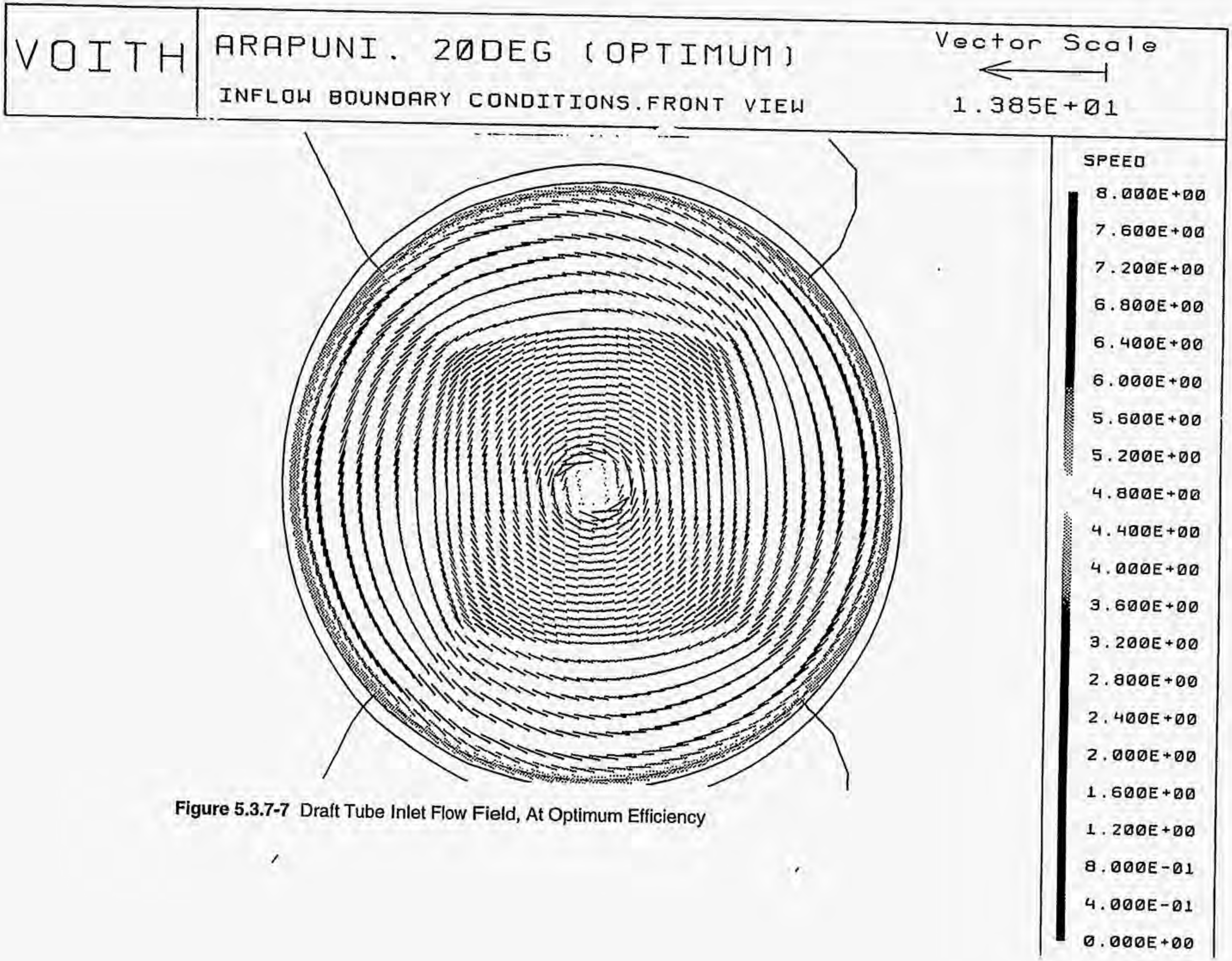


Figure 5.3.7-7 Draft Tube Inlet Flow Field, At Optimum Efficiency

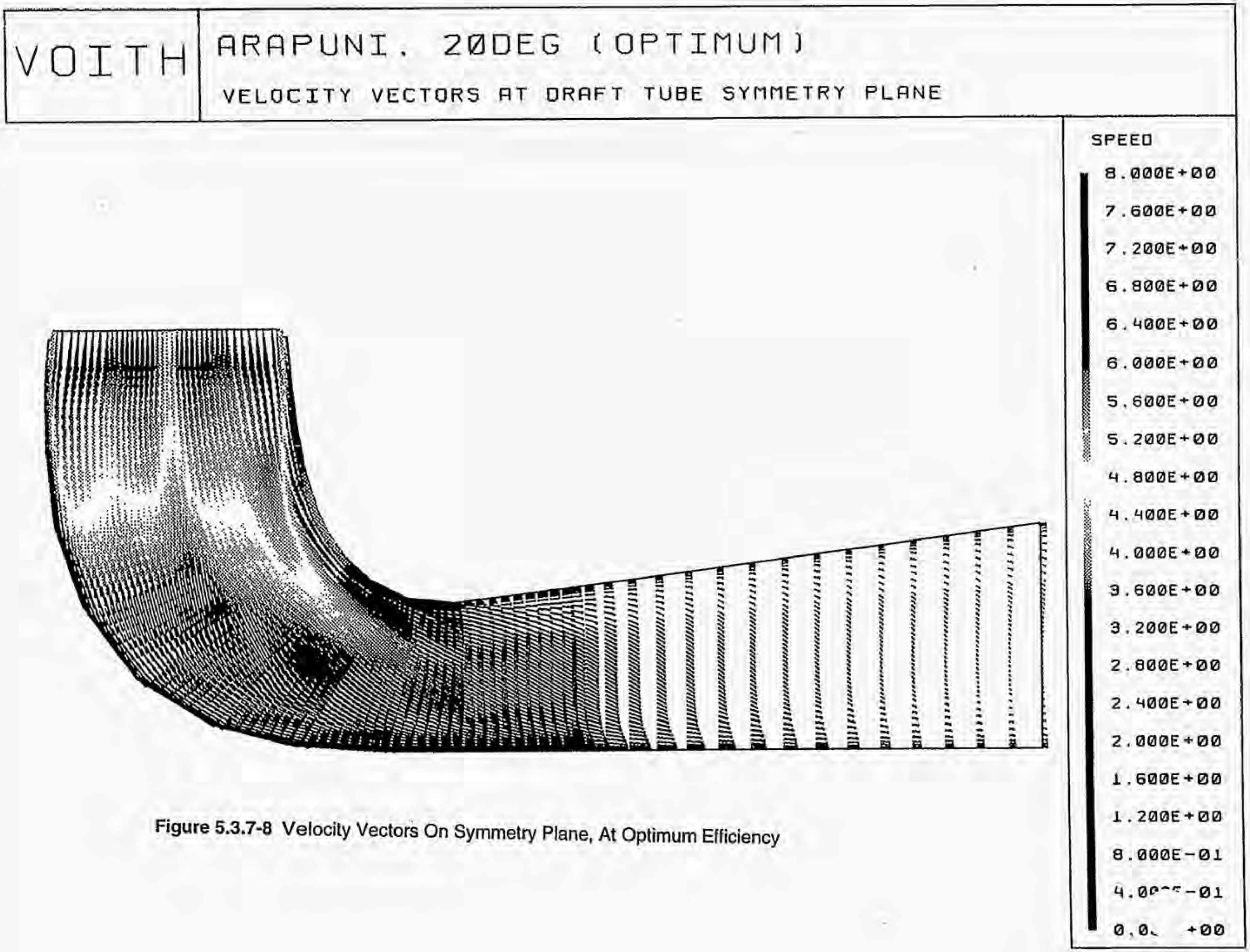


Figure 5.3.7-8 Velocity Vectors On Symmetry Plane, At Optimum Efficiency

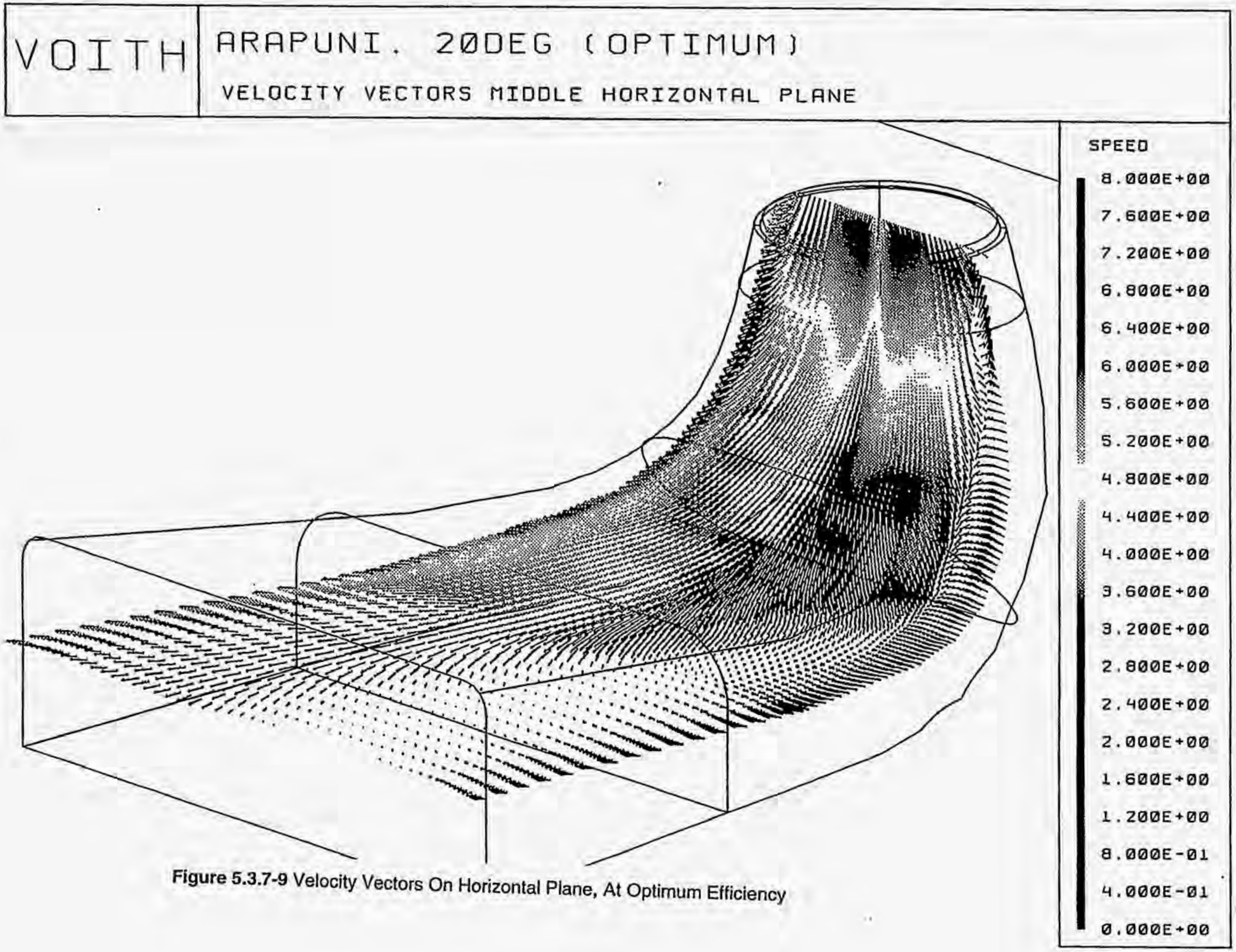


Figure 5.3.7-9 Velocity Vectors On Horizontal Plane, At Optimum Efficiency

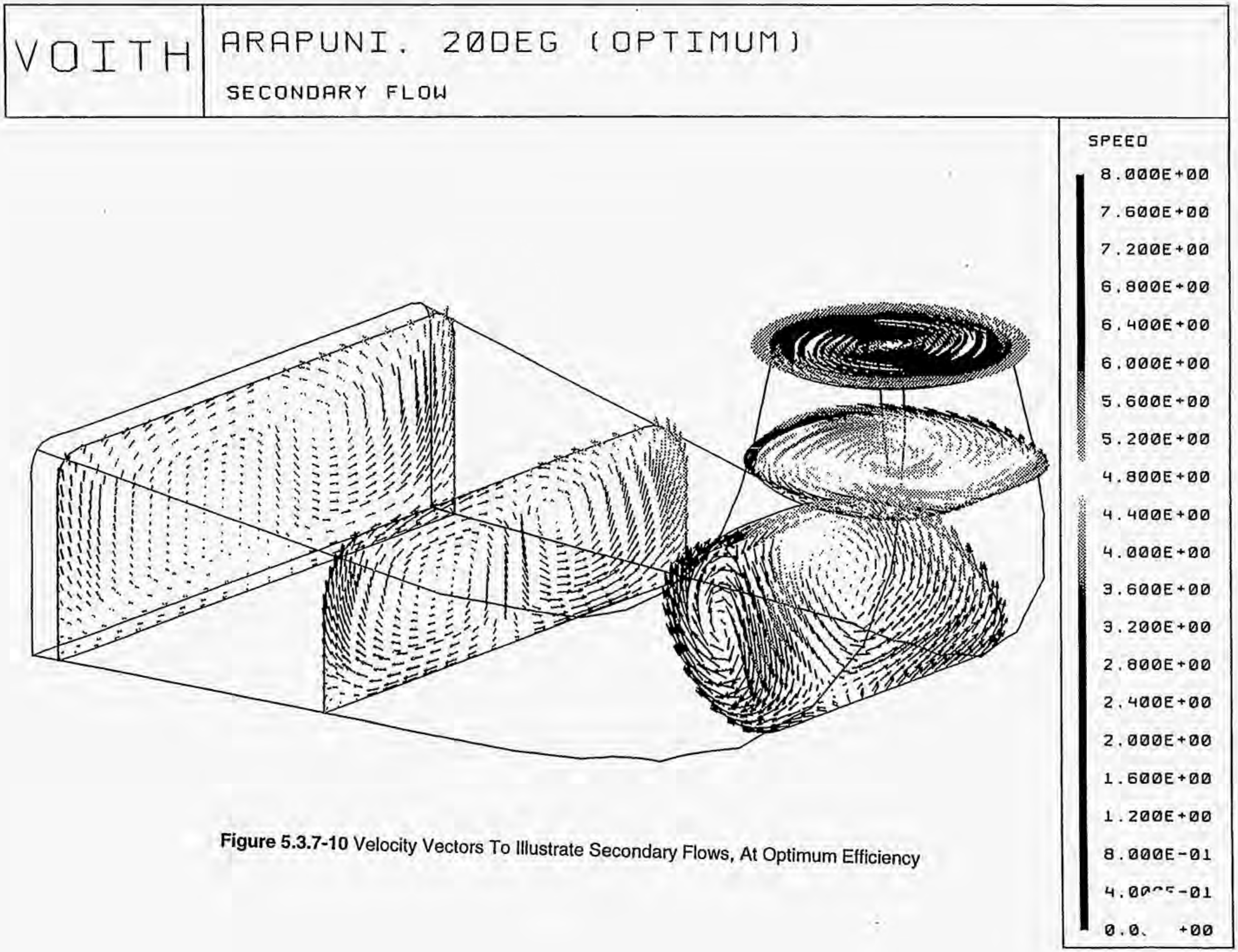


Figure 5.3.7-10 Velocity Vectors To Illustrate Secondary Flows, At Optimum Efficiency

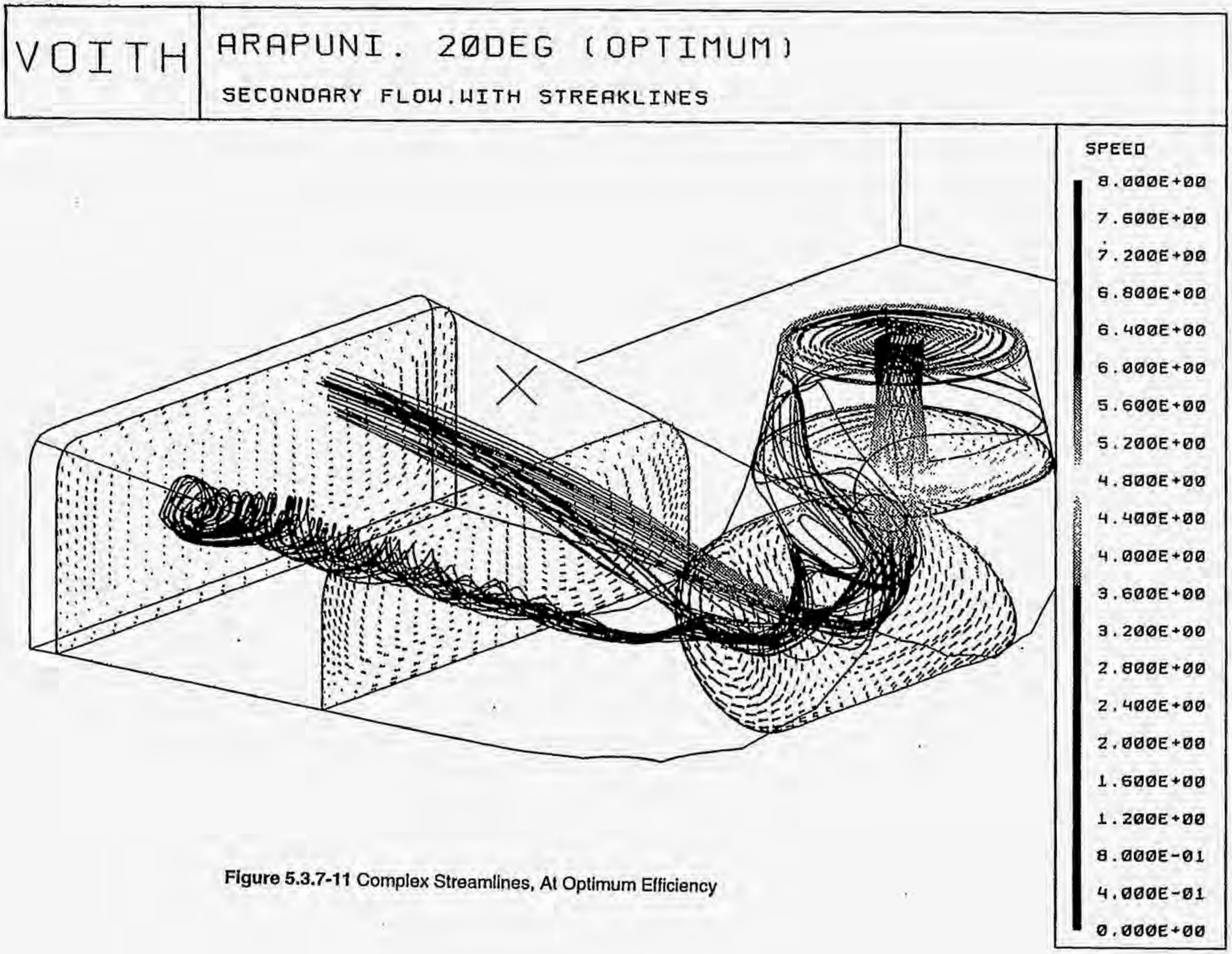


Figure 5.3.7-11 Complex Streamlines, At Optimum Efficiency

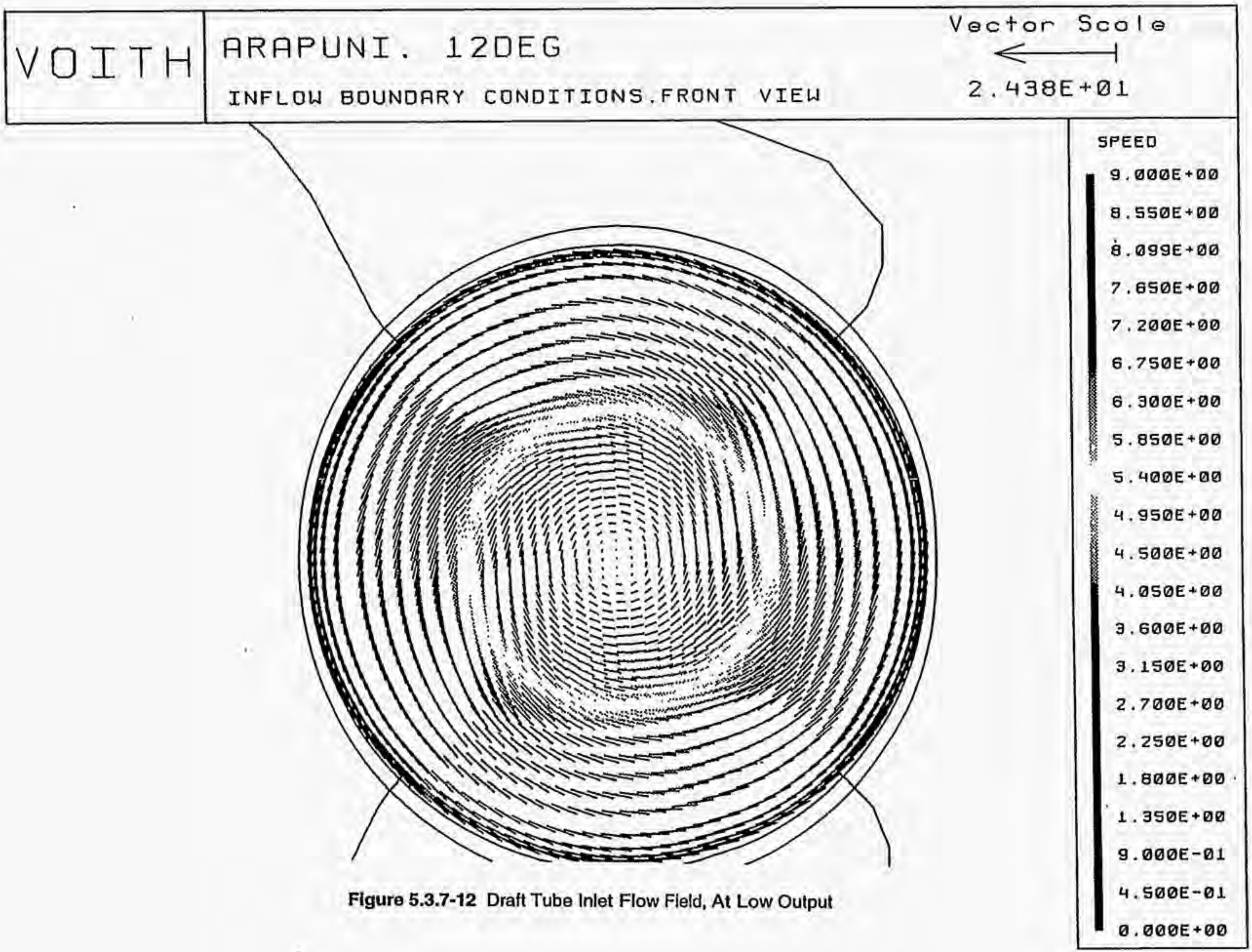


Figure 5.3.7-12 Draft Tube Inlet Flow Field, At Low Output

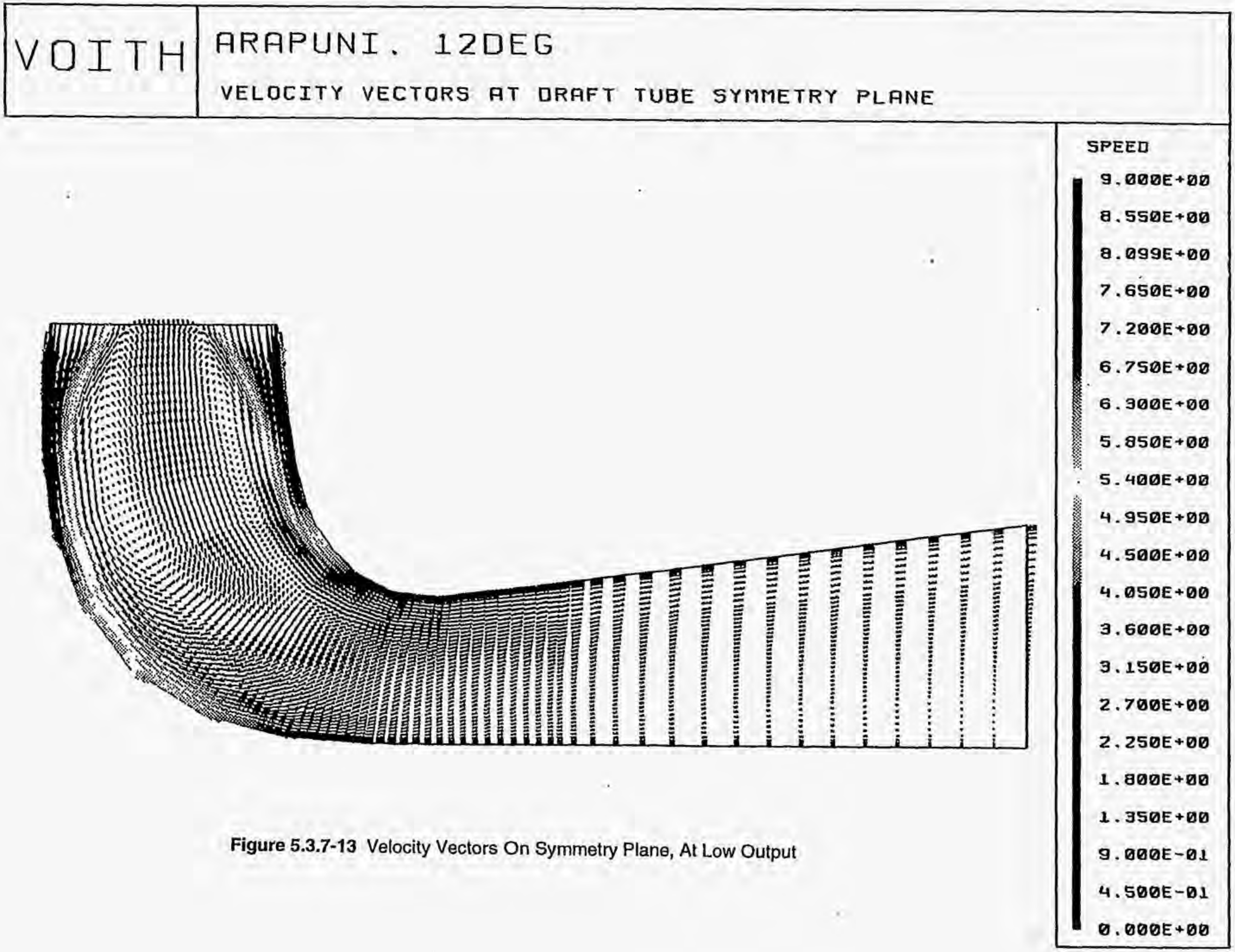


Figure 5.3.7-13 Velocity Vectors On Symmetry Plane, At Low Output

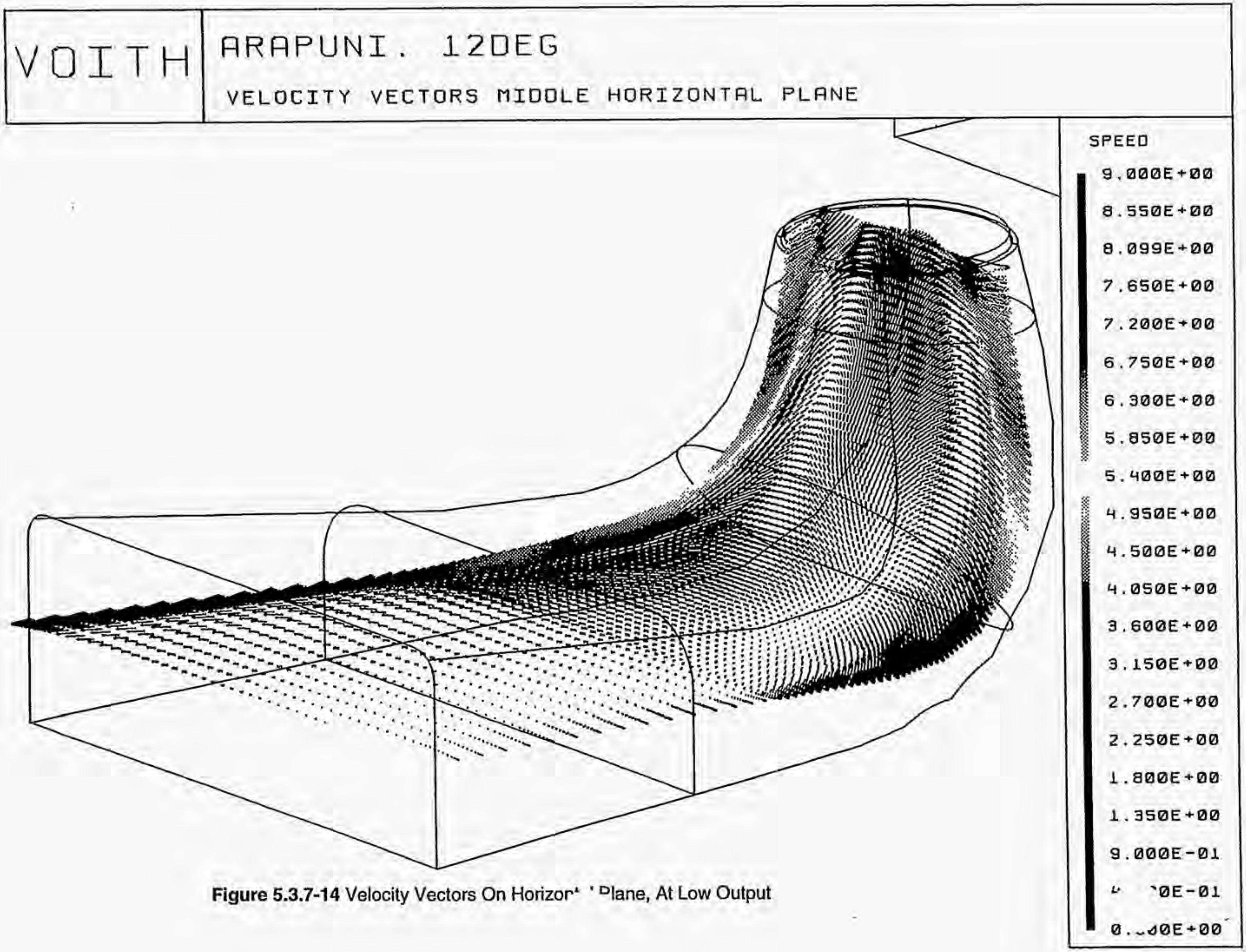


Figure 5.3.7-14 Velocity Vectors On Horizor' ' Plane, At Low Output

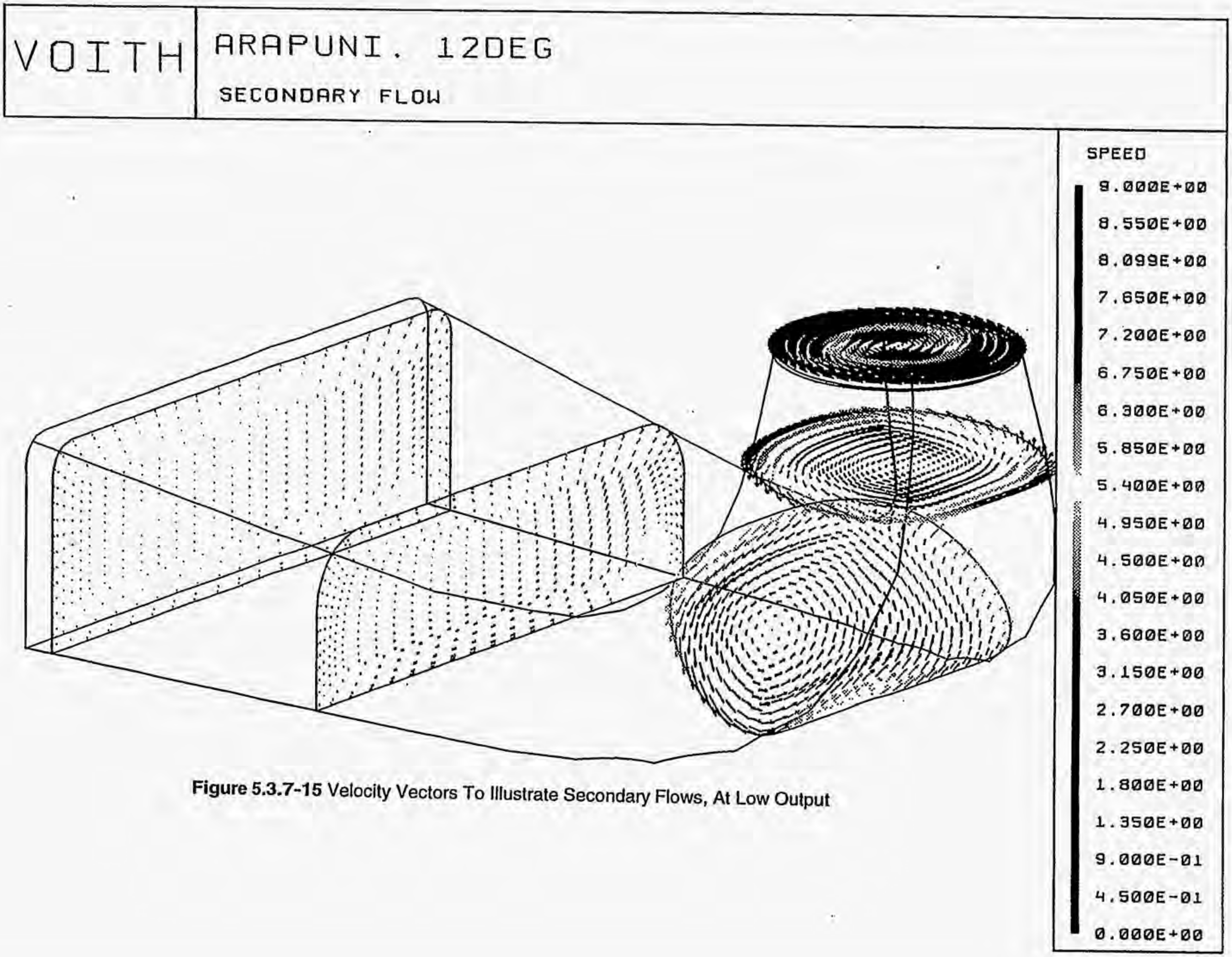


Figure 5.3.7-15 Velocity Vectors To Illustrate Secondary Flows, At Low Output

5.3.8 SUMMARY AND CONCLUSIONS OF THREE-DIMENSIONAL CFD STUDIES

Three-dimensional viscous flow analysis capabilities are effective today for use in quantifying flow field characteristics within turbines and provide valuable insight to designers allowing significant improvements in turbine design sophistication in comparison to designs of the past. The use of CFD tools, coupled with carefully planned experimental investigations, provides a means of quantifying fluid flow characteristics and can lead to a better understanding of the causal mechanisms leading to fish mortality. CFD tools allow the prediction of flow paths, times of passage, local pressures, rates of pressure change, rates of shear, and so forth. To effectively use their capabilities, a skilled user needs an effective computer aided geometry definition system, a grid mesh generator, a numerical calculation system, a post processor, and a means of correlating the results of the calculations with real world flow measurements and observations. The skill of the user in interpreting the results based on a history of correlations is important as the numerical tools improperly applied can lead to erroneous results.

Missing in the tool set of today is a method to integrate fluid forces on fish bodies. Only though the quantification of the loads and the correlation of the calculations with the results of physical experiments can criteria for fish injury/mortality be developed.

Today's tools can benefit from advancements. Grid generation methodologies can be improved to reduce the effort related to this phase of the analysis. Grid densities and grid quality can be improved to improve the accuracy of the results. Turbulence models used in the tools are kept simple to keep calculation times and computer costs to affordable levels. More sophisticated turbulence models using more efficient numerical algorithms will allow better numerical simulation. Particularly in decelerating flow fields, the effect of different turbulence models can be surprising leading to significantly different flow characteristics. Using the CFD tools to map flow particle location (fish paths) versus time will be more meaningful with better turbulence models being used. Postprocessors can be improved to allow the user to more easily detect the regions of flow having characteristics of interest. Solvers and computer hardware can be improved to facilitate finer grids, more advanced turbulence modeling so as to allow computation of the flow field in reasonable time frames. Some of these advancements and their impact will be addressed in the next section.

5.3.9 REFERENCES

Raw, M. A Coupled Algebraic Multigrid Method for the 3D Navier-Stokes Equations Proceedings of the 10th GAMM - Seminar Kiel, January 14 -16, 1994. Notes on Numerical Fluid Mechanics Vol. 49. Vieweg - Verlag, Braunschweig, Weisbaden, Germany, 1995.

Thomas, M.E., Shimp, N.R., Raw, M.J., Galpin, P.F., and Raithby G.D. The Development of an Efficient Turbomachinery CFD Analysis Procedure AIAA/ASME/SAE/ASEE 25th Joint Propulsion Conference, Monterey, CA, July 10 - 12, 1989.

5.4 ADVANCED CFD MODELING FOR FISH-FRIENDLY HYDROTURBINES

5.4.1 INTRODUCTION

The objective of Georgia Tech's contribution in Task 3 was to investigate the role of advanced Computational Fluid Dynamics (CFD) as a tool for enhancing the fish-friendliness of hydro-power installations. The term *advanced CFD* refers herein to computational methodologies which are currently at the leading edge of CFD research. Such methodologies feature state-of-the-art turbulence models, capable of quantitatively accurate predictions of complex three-dimensional flows, in conjunction with advanced numerical techniques for the efficient solution of the turbulence model and mean flow equations. At their current state of development, such methods are not practical enough (in terms of the computational resources they require for obtaining solutions) to be used in the turbine design process but will provide the basis for the next generation of industrial CFD codes.

In the context of the AHT project, the contribution of advanced CFD methods is twofold. First, such methods may be employed to obtain quantitatively accurate predictions of the flow through the various components of a hydropower installation over a broad range of operating conditions. This is a crucial prerequisite for: i) understanding the details of the very complex flowfields encountered in hydraulic turbines; ii) identifying flow phenomena which may be responsible for increased fish mortality rates associated with specific turbine design elements; and iii) suggesting and evaluating rehabilitation remedies for alleviating the adverse effects of such elements. It should be emphasized that achieving the two latter objectives further necessitates the development of computational tools capable of calculating fish trajectories and fish body loads through a given pre-computed flowfield. Such tools must treat fish as three-dimensional bodies with physical and geometrical characteristics closely approximating those of the species under consideration. Demonstrating the need for developing such tools is our second contribution in the AHT project.

In the following sections we outline the progress we made in both areas of accurate flow predictions and numerical modeling of fish passage. In the first area, we demonstrate the ability of advanced CFD methods to yield quantitatively accurate results for complex flows, as well as their potential for predicting hydroturbine discharges at off-design conditions, which are typically associated with complex unsteady flow phenomena. In the area of fish passage, we demonstrate the inadequacy of modeling fish as fluid particles and point to the need for future research aimed at the development of advanced fish passage numerical tools.

5.4.2 ADVANCED TURBULENCE MODELS FOR HYDROTURBINE FLOWS

The original objectives of this task were to apply a number of advanced turbulence models to hydroturbine geometries, correlate their predictions with experimental measurements, and identify models suitable for quantitatively accurate predictions of real-life hydroturbine flows. Unfortunately, however, existing measurements for hydroturbine geometries are not sufficiently detailed to facilitate a meaningful validation of advanced turbulence models. For that reason, we chose to carry out a comprehensive model validation study for a test case for which detailed three-dimensional measurements (in terms of three velocity components, pressure and skin-friction distributions, and turbulence quantities) are available in the literature. The specific case selected is flow through a strongly curved rectangular duct (Kim, 1991). Although geometrically simpler than typical components of hydraulic turbines, the flow in this curved duct exhibits a number of complex phenomena similar to those encountered in various subsystems of hydropower installations--strong streamline curvature, curvature induced axial and transverse pressure gradients, secondary flows, formation, growth and decay of intense streamwise vortices, turbulence anisotropy, etc. It is, therefore, a very challenging test case for exploring the predictive capabilities of advanced turbulence models. Calculations were also carried out for a typical draft-tube geometry (the TVA Norris Dam draft tube) using two different turbulence models. The computed results for this case were compared with the measurements of Hopping et al. (1992) who reported streamwise velocity profiles at few sections within the three bays of the draft tube.

The results of the two studies are summarized in the following sections. More details can be found in Sotiropoulos and Ventikos (1997) and Ventikos et al. (1996) which are included in Appendix 10.5.

5.4.2.1 **Flow through a Strongly Curved Rectangular Duct**

Summary

The non-linear two-equation turbulence model (non-linear k-w model) that was developed and tested in this study was shown to yield significantly improved predictions of complex three-dimensional flows with strong streamwise vortices. It is, therefore, a very promising tool for quantitative accurate simulations of real-life hydroturbine flows.

Discussion

The experiment of Kim (1991) is selected as a test case for this study. Kim (1991) reported detailed mean flow and turbulence measurements for flow through a 90° rectangular duct, of aspect ratio 6, at Reynolds number Re=224,000 (based on the duct width and the mean bulk velocity). An overall view of the wind tunnel and duct geometry, as well as the sections at which measurements were reported, are shown in Figure 5.4.2-1. As seen in the figure, the flow enters the inlet tangent of the curved duct through a short transition duct (a two-dimensional 6:1 contraction). The transverse pressure gradients on the top wall of the contraction induce a pair of vortices inside the top-wall boundary layer resulting in a complex three-dimensional flow at the inlet of the upstream straight tangent (Kim, 1991). In order to ensure that the inlet conditions for the numerical calculations properly represent the experimental situation, the experimental data at station U1 are used to construct appropriate inlet distributions for the mean velocity components and the turbulent quantities (see Kim (1991) for more details on using the measurements to construct inlet conditions for the calculations). The computational domain starts 4.5H upstream from the inlet of the bend (station U1) and extends up to 30H downstream from the exit of the bend. A numerical mesh with 98x69x52 nodes, in the streamwise, radial and normal directions, respectively, is used for all subsequently reported calculations.

A total of five turbulence models were evaluated. These included k-e and k-w models combined with linear (isotropic) and non-linear (non-isotropic) constitutive relations for expressing the turbulent stresses in terms of mean velocity gradients. The details of the various models can be found in Sotiropoulos and Ventikos (1997a). Here we only include a small but representative sample of the computed results. Figure 5.4.2-2 compares measured and computed contours of streamwise vorticity component at the exit of the bend (section D1 in Figure 5.4.2-1) near the inner lower corner of the duct cross-section. The streamwise vorticity component is selected herein because it provides a direct measure of the intensity of the secondary motion that develops inside the bend. As seen in Figure 5.4.2-2, the measurements reveal a very complex secondary motion structure characterized by C-shaped vorticity contours and five distinct vorticity peaks. Out of the five turbulence models tested, we include in Figure 5.4.2-2 only two predictions which clearly gage the progress we made in the course of this work. The first model, denoted in Figure 5.4.2-2 as *linear k-e model*, is the standard two-layer k-e model of Chen and Patel (1988) which is widely used today in computations of complex flows of practical interest. The second model, denoted as *non-linear k-w model* in Figure 5.4.2-2, was developed in the course of this study and is a non-isotropic version of the k-w model of Wilcox (1988). It is seen that the two-layer k-e model of Chen and Patel fails to capture even qualitatively the measured features of the vorticity contours. The proposed non-linear k-w model, on the other hand, reproduces almost every experimental trend with remarkable accuracy. This very promising result underscores the potential of the proposed turbulence model as a practical tool for quantitatively accurate predictions of complex hydroturbine flows.

5.4.2.2 Flow Through a Francis-Turbine Draft Tube

Summary

Several improvements were incorporated in the Georgia Tech CFD code to make it applicable to complex draft-tube geometries with piers. Calculations were carried out for the TVA Norris Dam draft-tube, using a very fine computational grid, and the computed results were shown to be in good quantitative agreement with available measurements. Lack of detailed experimental data, however, is a major obstacle that hinders further advancements in the numerical simulation of real-life draft-tube flows.

Discussion

The draft tube configuration, used for the present computations, is one of the TVA Norris Autoventing Power Plant (Norris, Tennessee) draft tubes designed to operate with 50 MW (66,000 HP) Francis hydroturbines. The area expansion ratio for this draft tube (ratio of the exit to inlet cross-sectional area) is approximately 4.4:1 while the radius of curvature of the elbow is 1.34 diameters of the inlet circular cross-section. Two vertical piers, symmetrically placed about the centerline, support the downstream rectangular diffuser.

The computational grid for every cross-section is generated using an efficient algebraic grid generation method which employs linear and third-order spline interpolation. The grid lines are concentrated near the walls using the hyperbolic tangent stretching function. The cross-sectional grids are then stacked along the centerline of the tube to complete the three-dimensional grid. To accurately resolve the flow in the vicinity of the piers, the streamwise planes are clustered around the pier leading edges also using hyperbolic tangent stretching. Typical cross-sectional views of the computational mesh and the relevant coordinates are shown in Figure 5.4.2-3. All the subsequently reported calculations were carried out on a grid with 85 x 73 x 193 nodes (a total of approximately 1.2×10^{6} nodes), in the streamwise, vertical, and horizontal (x, h, and z) directions, respectively, which is the finest mesh to be used so far for draft-tube calculations.

The numerical method employed herein is based on the work of Lin and Sotiropoulos (1997a,b) who developed an efficient time-marching procedure for solving the three-dimensional Reynolds-averaged Navier-Stokes (RANS) equations, in conjunction with two-equation, near-wall, turbulence closures, in generalized curvilinear coordinates. Several improvements were required in order to make the method of Lin and Sotiropoulos (1997) applicable to complex hydroturbine geometries. These include, among others, the ability to handle multiple connected domains, use of variable residual smoothing coefficients, and implementation of Total Variation Diminishing (TVD) schemes (Yee and Harten, 1987). This latter development was found critical for accurate high Reynolds number turbulent flow simulations for geometries involving stagnation points (such as the piers of a draft tube). None of the high-resolution non-monotone schemes tested by Lin and Sotiropoulos (1997a) were robust enough to handle draft-tube flows. In fact stable simulations could be carried out only when first-order flux-difference splitting upwind was implemented for discretizing the convective terms. The resulting solutions, however, are contaminated due to excessive numerical viscosity. The dramatic effect of spatial accuracy in predictions of draft-tube flows is demonstrated in Figure 5.4.2-4 which compares solutions obtained using the first-order upwind scheme with those obtained using the up-to-second order accurate symmetric TVD scheme of Yee and Harten (1987). Both calculations were carried out on the same mesh using the two-layer k-e model. The particle traces in both figures have been released from exactly the same points located just upstream of the right pier.

Figure 5.4.2-5 shows comparisons of measured (Hopping, 1992) and calculated streamwise mean velocity profiles at two streamwise locations, downstream the start of the piers, in all three bays--these results were obtained using the linear k-w model of Wilcox (1988). The present simulations correspond to experimental run No. 1 (see Hopping, 1992) which was performed with runner speed 898 rpm, and net head 24.8m. These conditions correspond to a Reynolds number $Re=1.1\times10^6$, based on the diameter D, and bulk velocity U_b at the inlet of the draft tube. The velocity profiles, are plotted at two y = constant planes (see Figure 5.4.2-5a for axis definition) along the horizontal (Figure 5.4.2-5a), and vertical (Figure 5.4.2-5b) centerlines of each cross-section. Figures 5.4.2-5a and b also include the measured streamwise and swirl velocity components at the inlet section, which were used to provide inlet conditions for the calculations (all velocities in these figures have been scaled by the bulk velocity at the inlet of the draft tube). It should be noted that the inlet measurements, which were obtained along two mutually perpendicular radii, suggest that the flow is not circumferentially symmetric. Due to lack of more detailed data, however, the calculations were carried out by arbitrarily choosing one of the two profiles and assuming that the inlet flow is axisymmetric. The measurements in Figure 5.4.2-5 suggest that most of the flow passes through the left (with respect to an observer standing at the draft-tube inlet looking downstream) bay. This is evident by the overall larger velocities through that bay and is obviously associated with the clockwise direction of the inflow swirl. The calculations reproduce this flow feature and appear to capture reasonably well most experimental trends. Some discrepancies are observed at the downstream location in the right bay (Figure 5.4.2-5a), where the calculated streamwise velocity profile indicates the presence of a small reversed flow region near the inner wall. Contour plots of the calculated streamwise velocity component, not shown here due to space considerations, reveal a recirculating flow region starting upstream of that section and ending immediately downstream. The measurements, on the other hand, suggest a fuller and almost uniform velocity profile there which appears to have recovered very rapidly from its upstream distorted shape. Similar discrepancies, albeit not as pronounced, are observed at the downstream section in the left bay as well. It should be noted, however, that the experimental measurements are not detailed enough to allow a comprehensive assessment of the accuracy of the numerical solutions. Given the continuous area expansion downstream of the elbow, it is very likely that reversed flow does exist in the experiment, although may be not at the same locations indicated by the calculations, but could not be resolved by the few available velocity measurements. Yet another source of uncertainty is the lack of detailed velocity measurements at the inlet. As discussed in the previous section, the inlet flow was assumed axisymmetric, although the limited available measurements do not support such an assumption (see inlet swirl profile in Figure 5.4.2-

5a). Given the complexity of the draft-tube geometry, even small differences in inlet conditions could account for the observed discrepancies. Obviously, the present calculations can not offer positive answers to all these questions. They do, however, underscore the need for carefully designed, very detailed laboratory experiments.

Figures 5.4.2-6, 5.4.2-7 and 5.4.2-8 depict particle traces released at strategically selected locations to clarify various three-dimensional flow features. A global view of the flowfield is given in Figure 5.4.2-6, which shows the paths of particles originating along two mutually perpendicular diameters at the inlet plane. It is seen that most of the flow passes through the left bay and the left half of the center bay, which is consistent with the trends exhibited by the velocity profiles discussed above. Particles released near the center of the inlet section are seen to form a coherent, rope-like, vortical structure which appears to pass through the left half of center bay. Significant secondary motion is also present in the right bay as indicated by the twisting particle trajectories there. Figures 5.4.2-7 and 5.4.2-8 reveal some very complex three-dimensional flow patterns along the flat wall of the right pier. Figure 5.4.2-7 indicates the existence of a recirculation region which is located near the top (diverging) wall of the draft tube--although not shown herein due to space limitations, the particles that are trapped in this area originate from the near-wall region at the left side of the inlet section. Underneath this recirculating flow region there is a very intense longitudinal vortical structure, shown in Figure 5.4.2-8, which appears to be similar to horse-shoe like vortices known to form at wing-body junctions. These flow patterns serve to demonstrate the enormous complexities of such flows, underscore the challenges for advanced CFD methods, and point, once again, to the need for very detailed laboratory experiments to provide data for numerical validation.

It is important to emphasize that the complex, three-dimensional flow features discussed above may have significant implications from the fish-passage standpoint. One may speculate, for instance, that the strong secondary motion and intense longitudinal vortices forming at various locations inside the draft-tube would tend to disorient passing fish and increase the probability for scrape and de-scaling related injuries. Furthermore, such vortices re-distribute the axial momentum within the draft-tube cross-section, thus, inducing areas of intense velocity gradients, which may result in increased shearing forces acting on the fish body. Therefore, in order to assess the fish-friendliness of a given draft-tube design it is of crucial importance to be able to predict numerically the various flow details identified above. Due to the geometrical complexity of real-life draft tubes, however, this necessitates the use of very fine computational meshes and advanced numerical and turbulence models, such as those employed herein. This further underscores the need for continuing our research efforts to refine and validate advanced CFD techniques for hydraulic turbine flows.

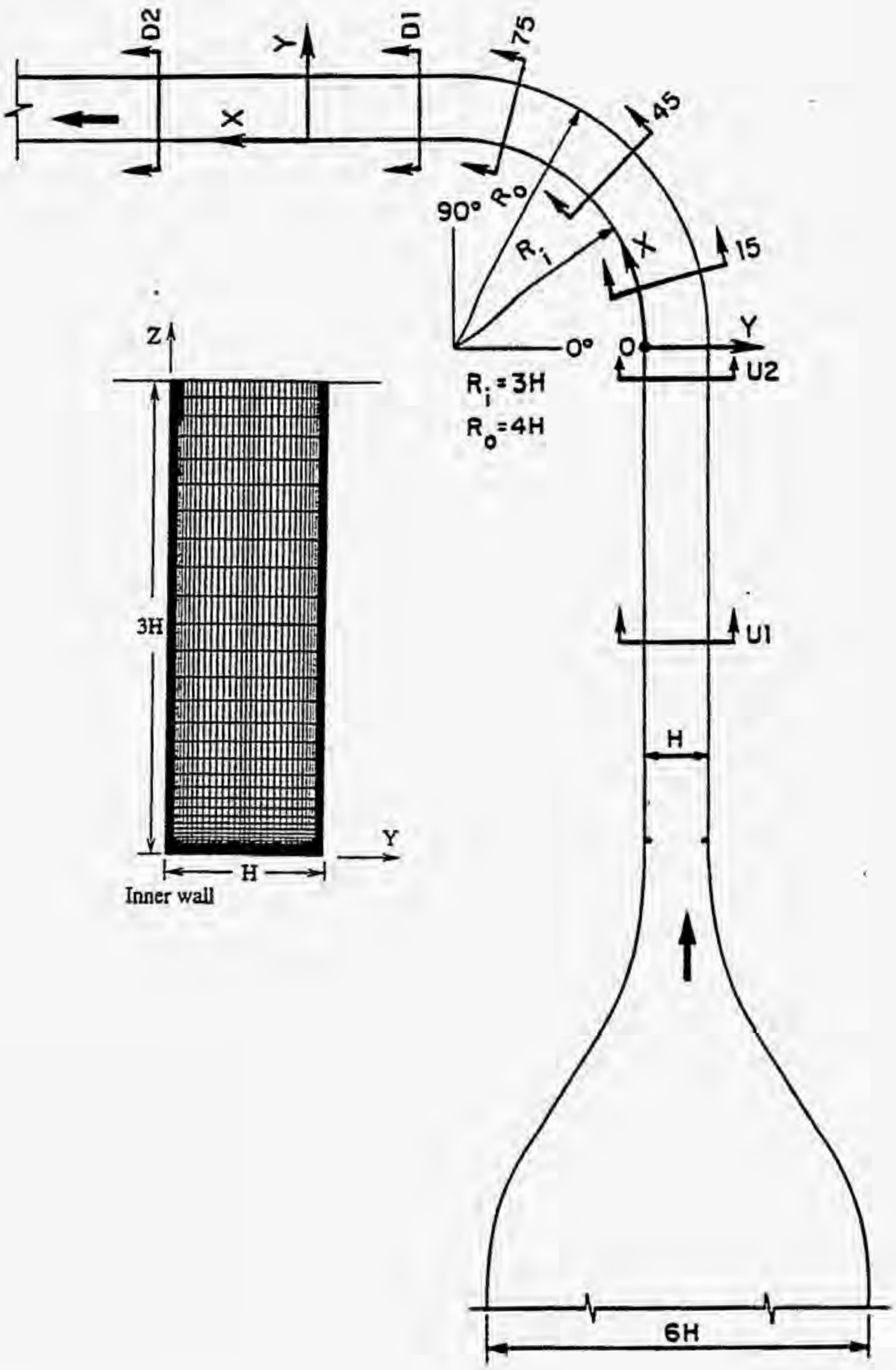
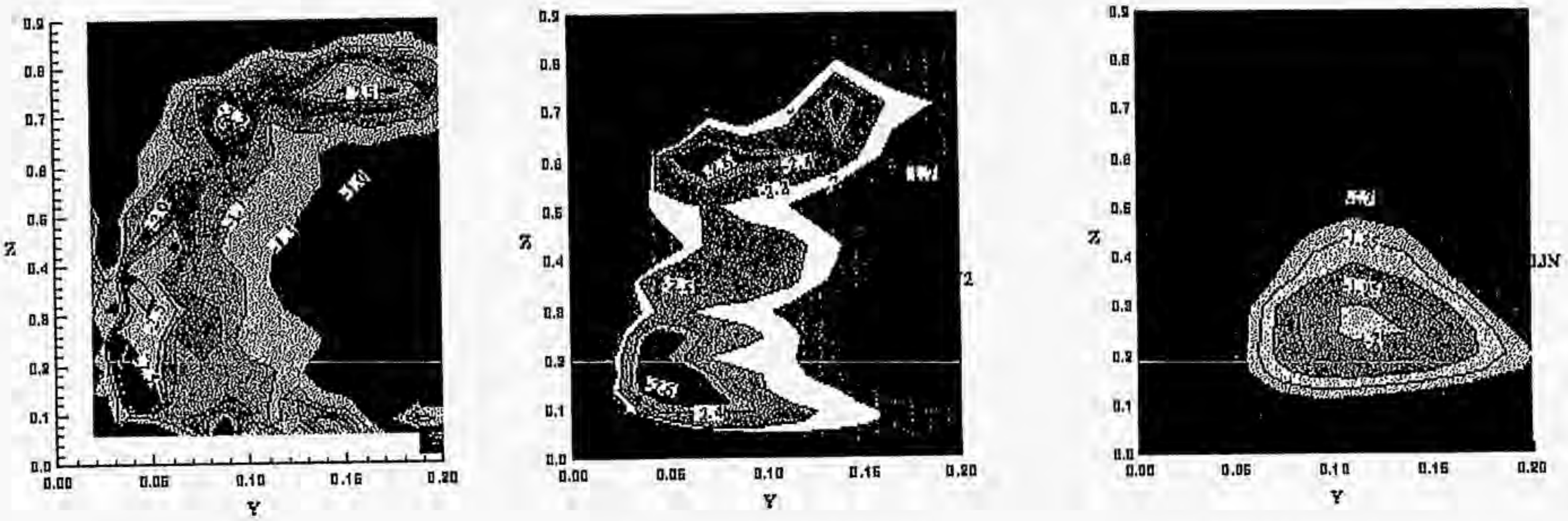


Figure 5.4.2-1 Coordinates, measurement locations, and cross-sectional mesh for the curved duct of Kim (1991).



a) Measurements

b) Present non-linear k-ω model

c) Two-layer k-ε model of Chen and Patel

Figure 5.4.2-2 Measured (Kim, 1991) and computed mean streamwise vorticity contours near the corner of the inner wall at station D1 (axis aspect ratio has been distorted for clarity).

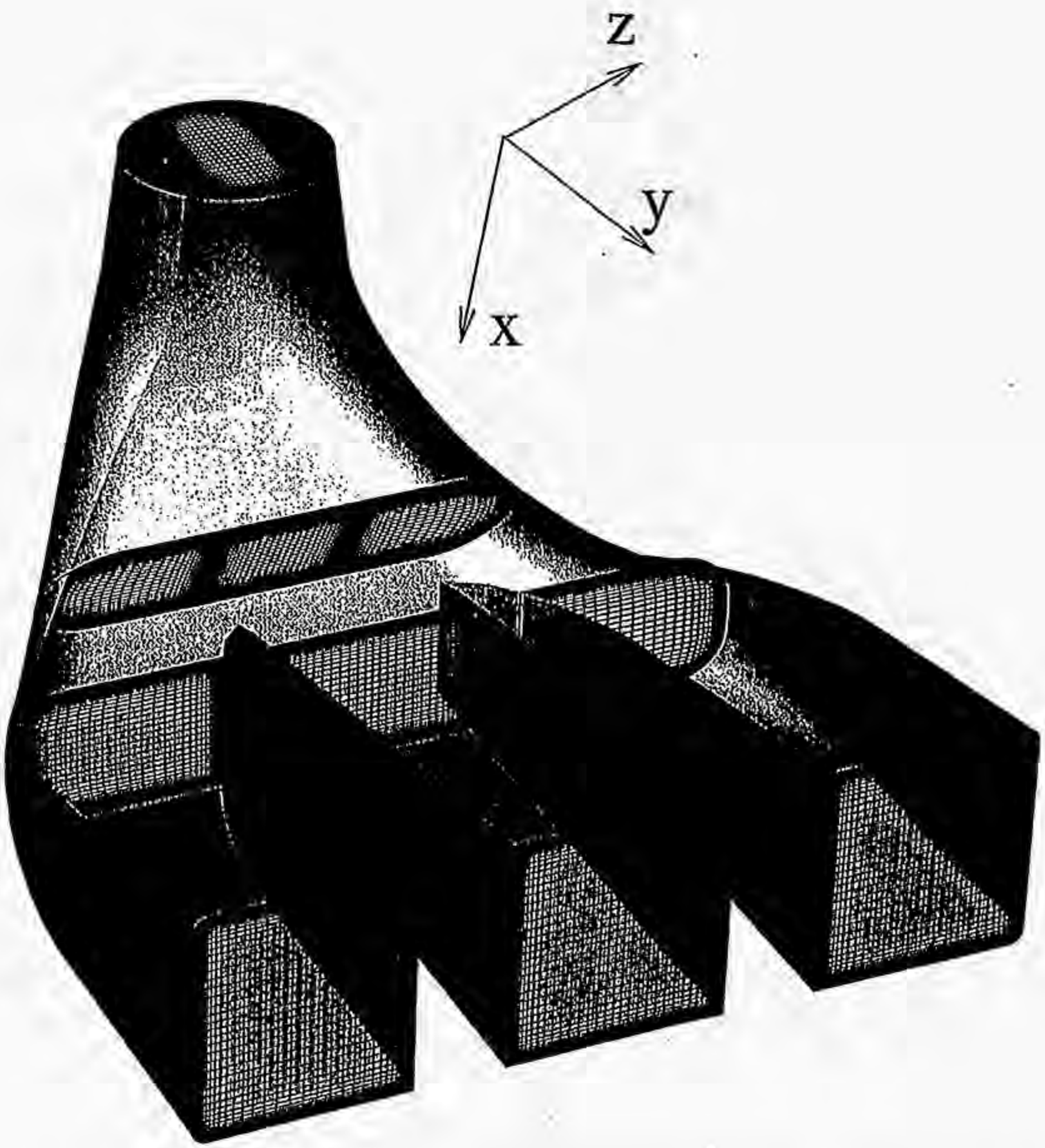
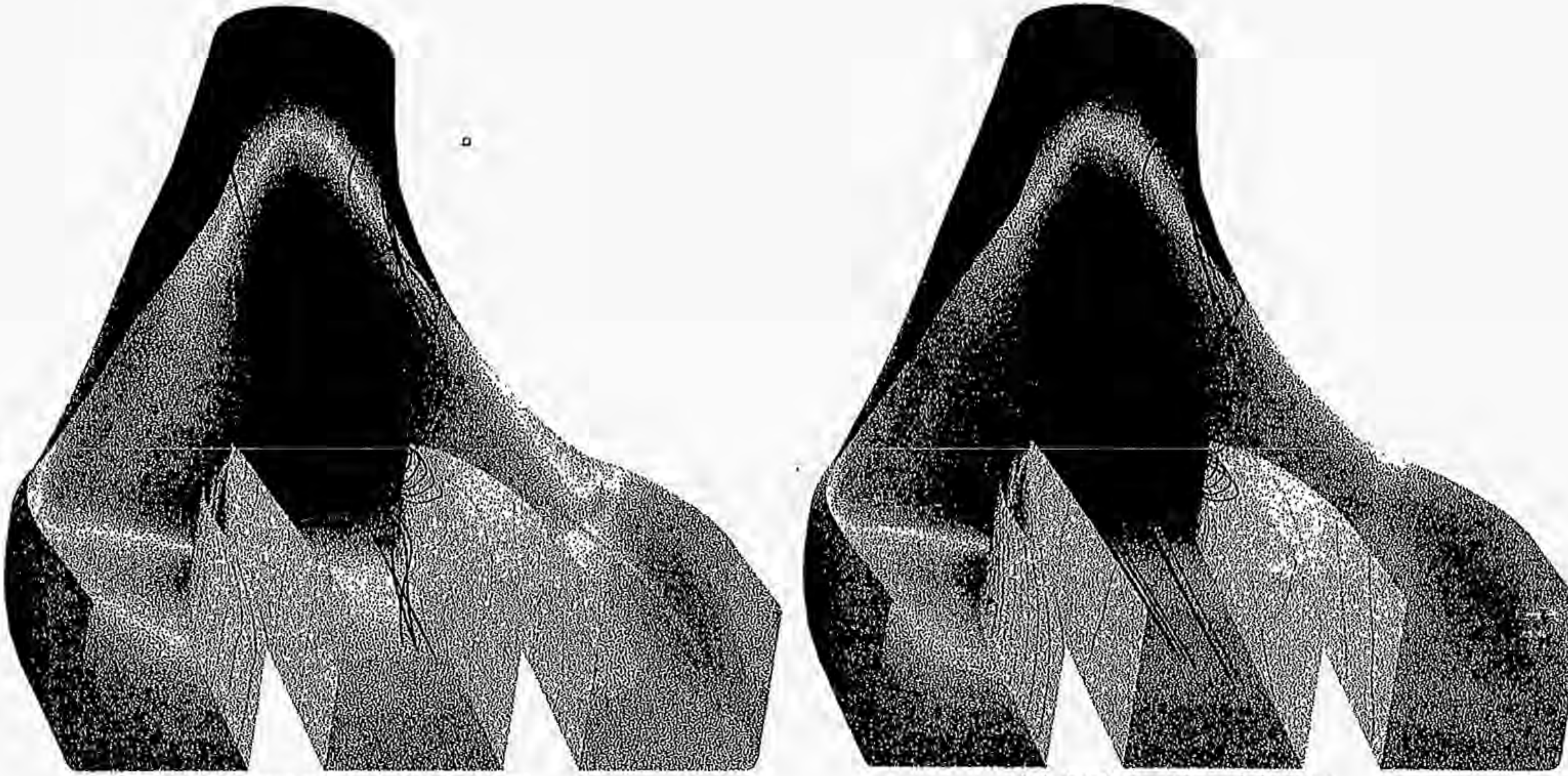


Figure 5.4.2-3 Cross-sectional views of the computational grid for the TVA Norris draft tube.



a) Up-to-second-order accurate TVD scheme

b) First-order accurate upwind scheme

Figure 5.4.2-4 Effect of the accuracy of the numerical scheme on the calculated particle paths.

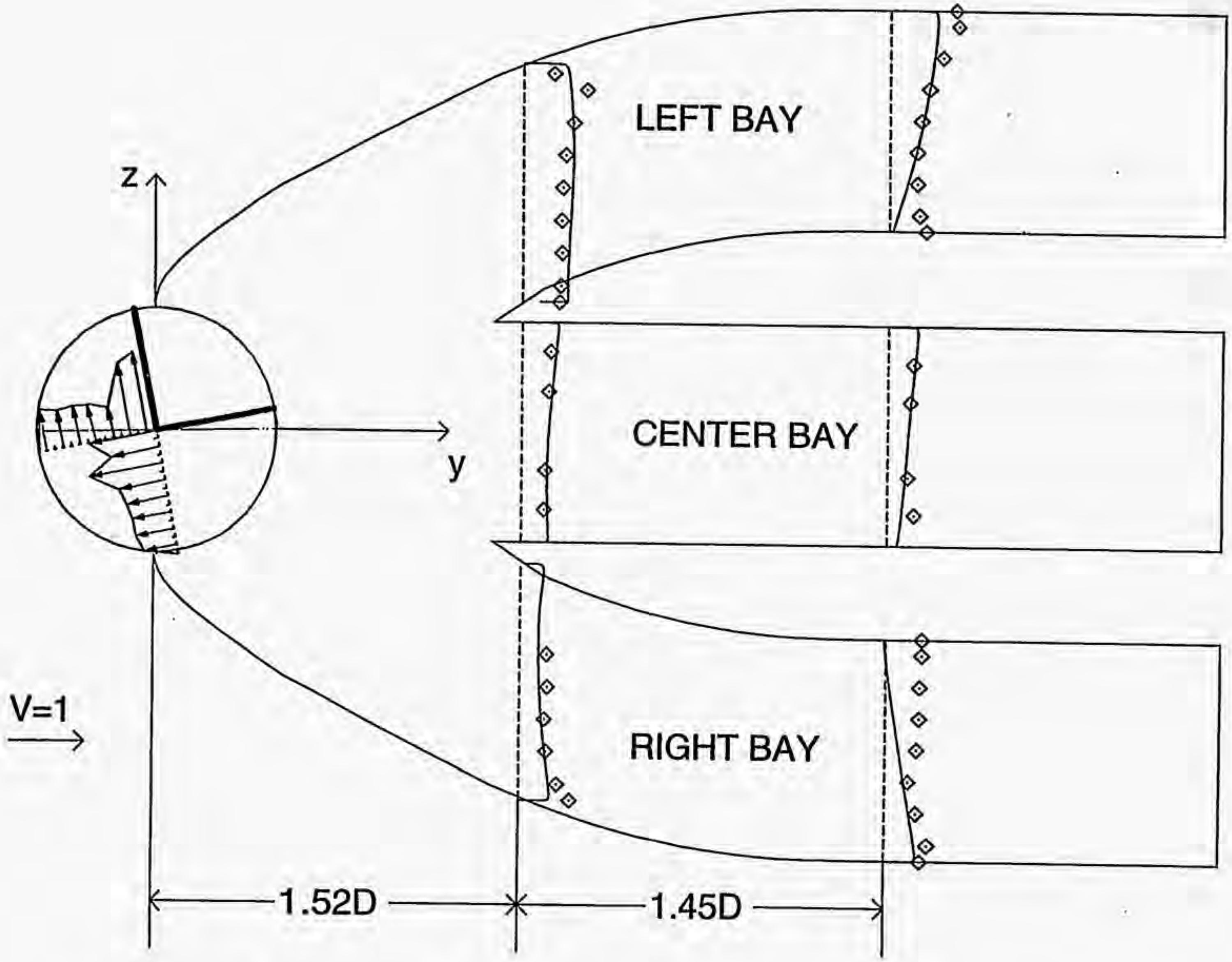


Figure 5.4.2-5a Measured (Hopping, 1992) and computed streamwise velocity profiles.

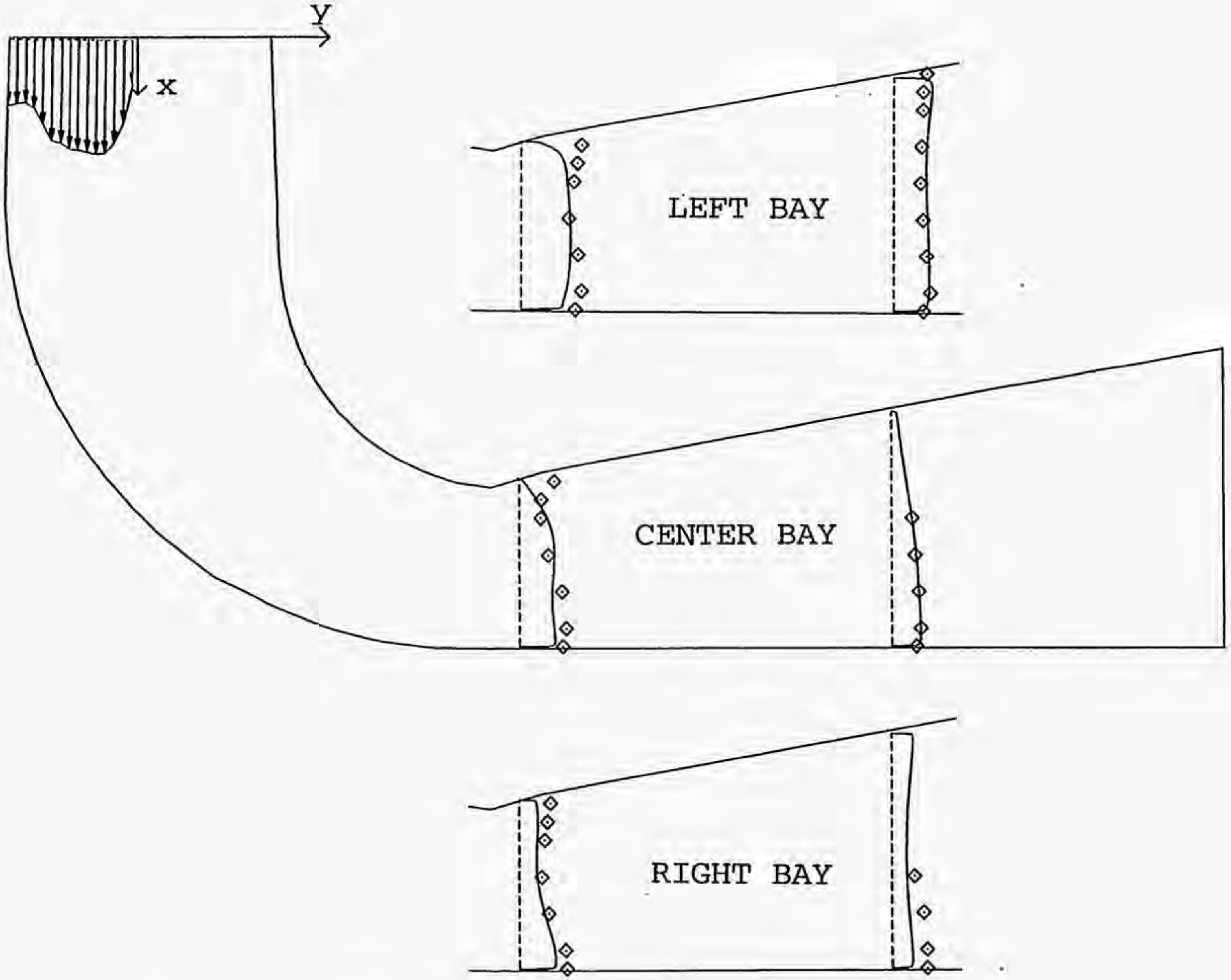


Figure 5.4.2-5b Measured (Hopping, 1992) omputed streamwise velocity profiles.

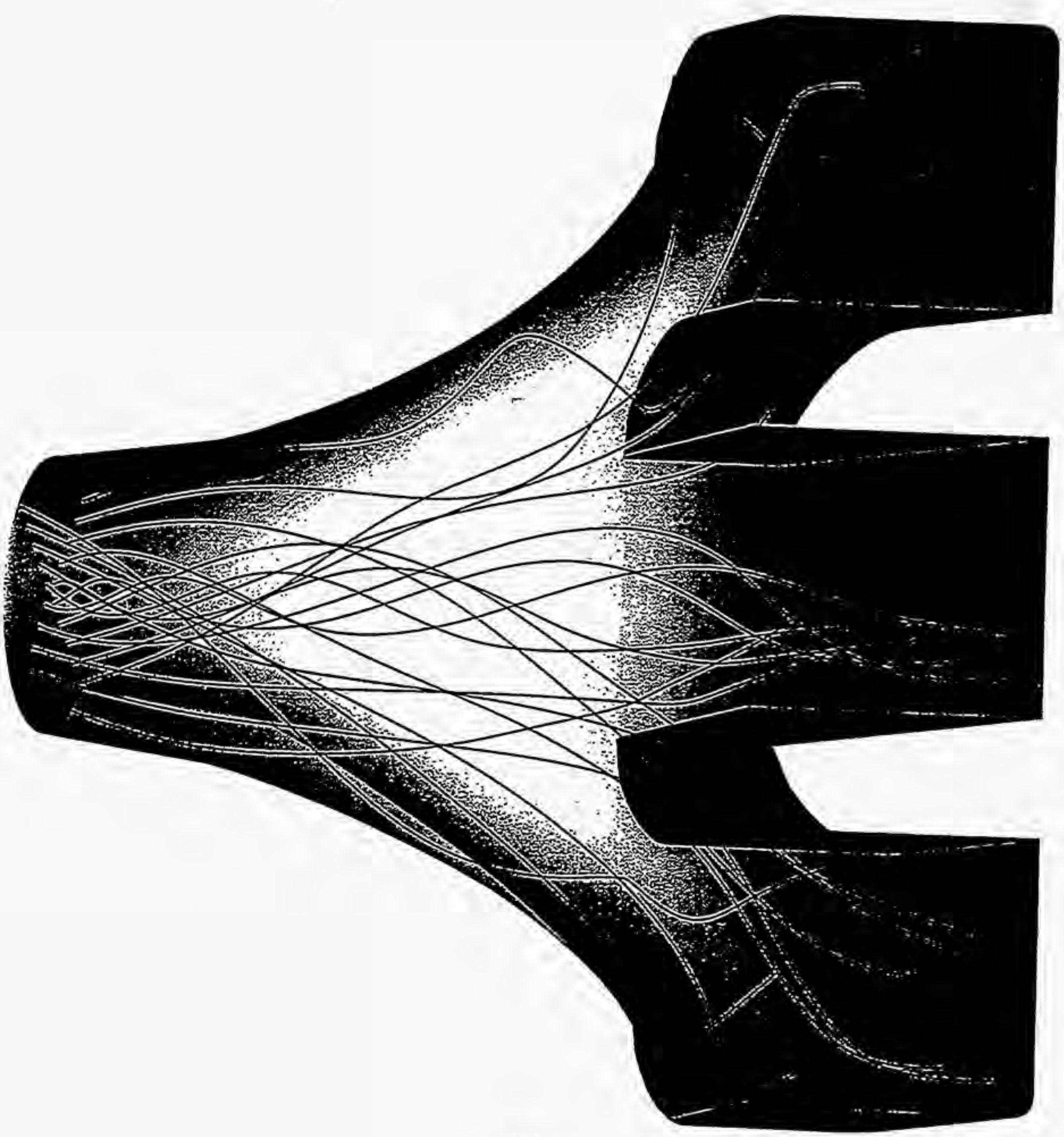


Figure 5.4.2-6 Three-dimensional particle traces: General view

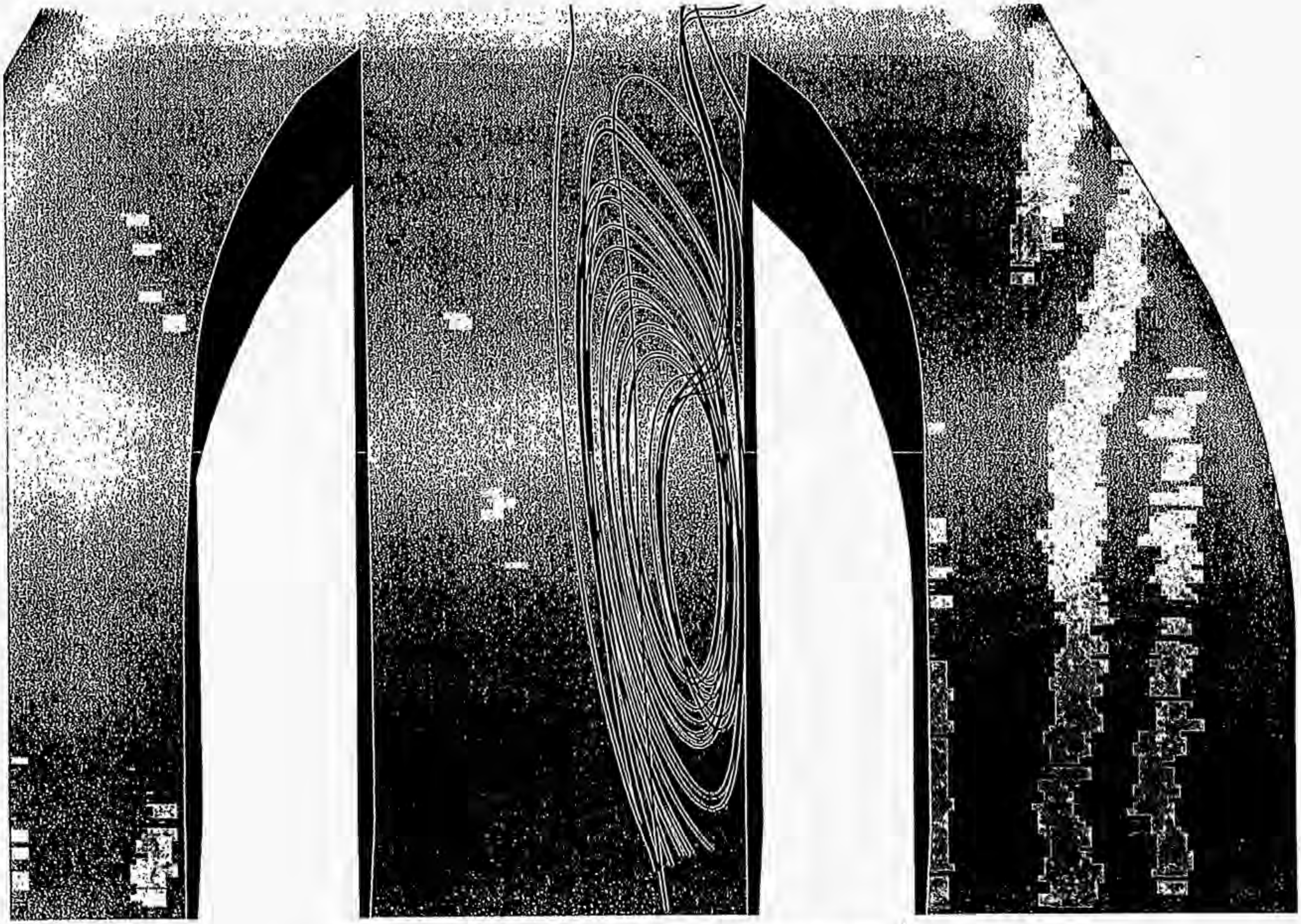


Figure 5.4.2-7 Three-dimensional particle traces: Reversed flow region

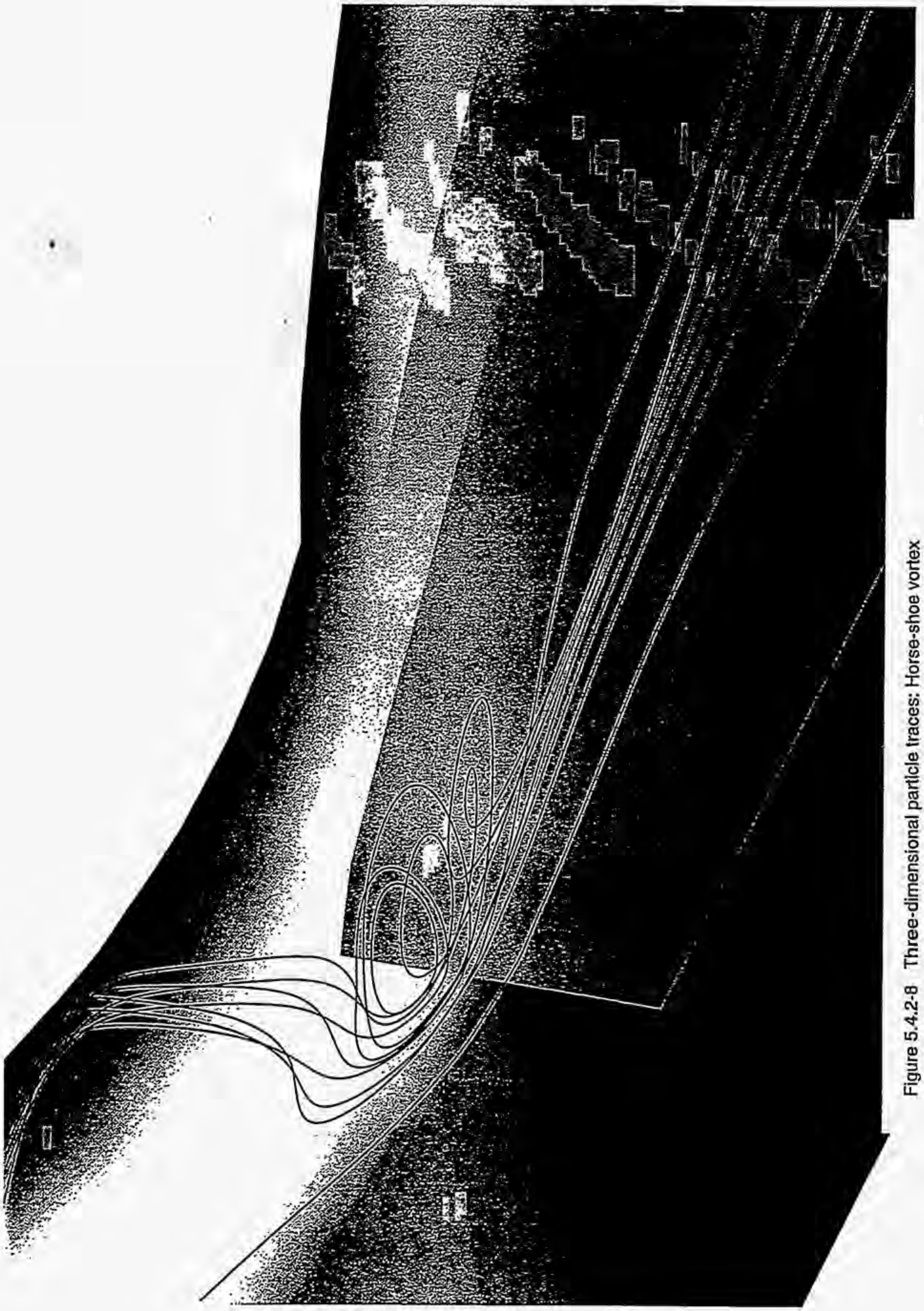


Figure 5.4.2-8 Three-dimensional particle traces: Horse-shoe vortex

5.4.3 UNSTEADY VORTEX PHENOMENA IN FRANCIS TURBINES DRAFT TUBES

Summary

A new numerical method was developed for simulating unsteady, three-dimensional flows and applied to calculate the evolution of swirling flow through a straight circular diffuser. The computed results reveal the formation of a precessing spiral vortex core whose general structure resembles that of a rope vortex known to be associated with draft-tube surge phenomena. This study, which is the first to calculate numerically the formation of such a vortex, underscores the potential of the proposed method for predicting hydroturbine flows at off-design operating conditions.

Discussion

Powerplants with Francis runners are known to experience undesirable pressure pulsations when operating at off-best-efficiency output, a phenomenon known as draft tube surge. When the system operates near best efficiency, the swirling flow exiting the runner forms a vortex core whose axis is stationary and more or less aligned with the geometrical axis of the draft tube (Figure 5.4.3-1.a). At off-design operating conditions, the residual swirl at the exit of the runner could exceed a threshold level causing the vortex core to become unstable and undergo dramatic transformations (Figure 5.4.3-1.b). The specific flow scenario that would emerge depends on the operating conditions and could include, among others, the formation of an unsteady bubble-like structure, with reversed flow entering the runner, as well as spiral or 'rope'-like vortex core which precesses about the draft tube axis. The rotating vortex core causes unsteady variations of the net head across the unit and could result in efficiency losses, noise, power swings, vibrations, and even catastrophic structural failure. Furthermore, these unsteady variations may induce significant pressure and shear loads on passing fish, thus, increasing the potential for serious injuries and possibly mortality. Understanding, therefore, the physics of such unsteady flow phenomena is a crucial prerequisite for not only improving the mechanical performance and extending the stable operational envelope of Francis powerplants but also for enhancing fish-friendliness. Given the complexity and time-dependent nature of these phenomena, experiments alone, such as the flow visualizations shown in Figure 5.4.3-1, can offer only limited insight into the relevant flow physics and should be supplemented by comprehensive time-accurate computations. Our objective herein is to demonstrate the ability of advanced CFD methods to simulate the formation of unsteady rope vortices in hydroturbine draft-tube.

A numerical method was developed for solving the unsteady, Navier-Stokes equations in generalized curvilinear coordinates. The method was applied to simulate unsteady swirling flow through a simplified draft-tube geometry, the straight circular diffuser shown in Figure 5.4.3-2. Detailed description of the computational procedure and analysis of the results can be found in Sotiropoulos and Ventikos (1997) (see Appendix 10.5). A sample of the computed results is shown in Figure 5.4.3-2 which depicts the evolution in time of a surface of constant circumferential vorticity component. As seen, the swirling flow entering the diffuser section breaks down into a spiral vortex (or rope vortex) whose structure resembles that observed in the flow visualization shown in Figure 5.4.3-1. The rope-vortex precesses about the diffuser axis along the same direction as the swirling flow at the inlet. The present results, which are the first to reproduce computationally the formation and precession of a rope-vortex, clearly demonstrate that numerical computation of unsteady flow phenomena in hydraulic turbines is now well within the reach of state-of-the-art CFD methodologies.

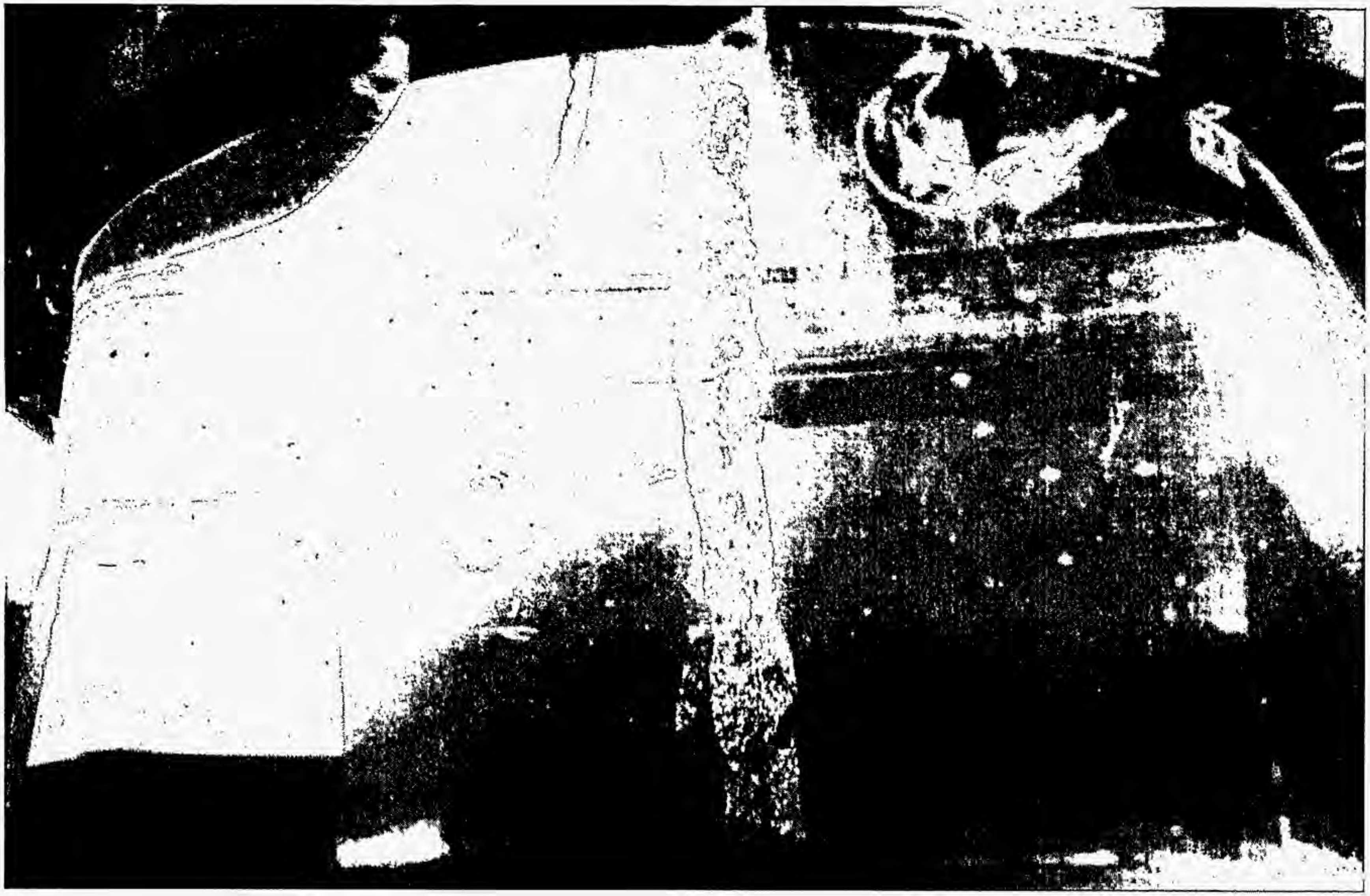


Figure 5.4.3-1a Visualization of cavitating vortex core at draft-tube entrance at design point

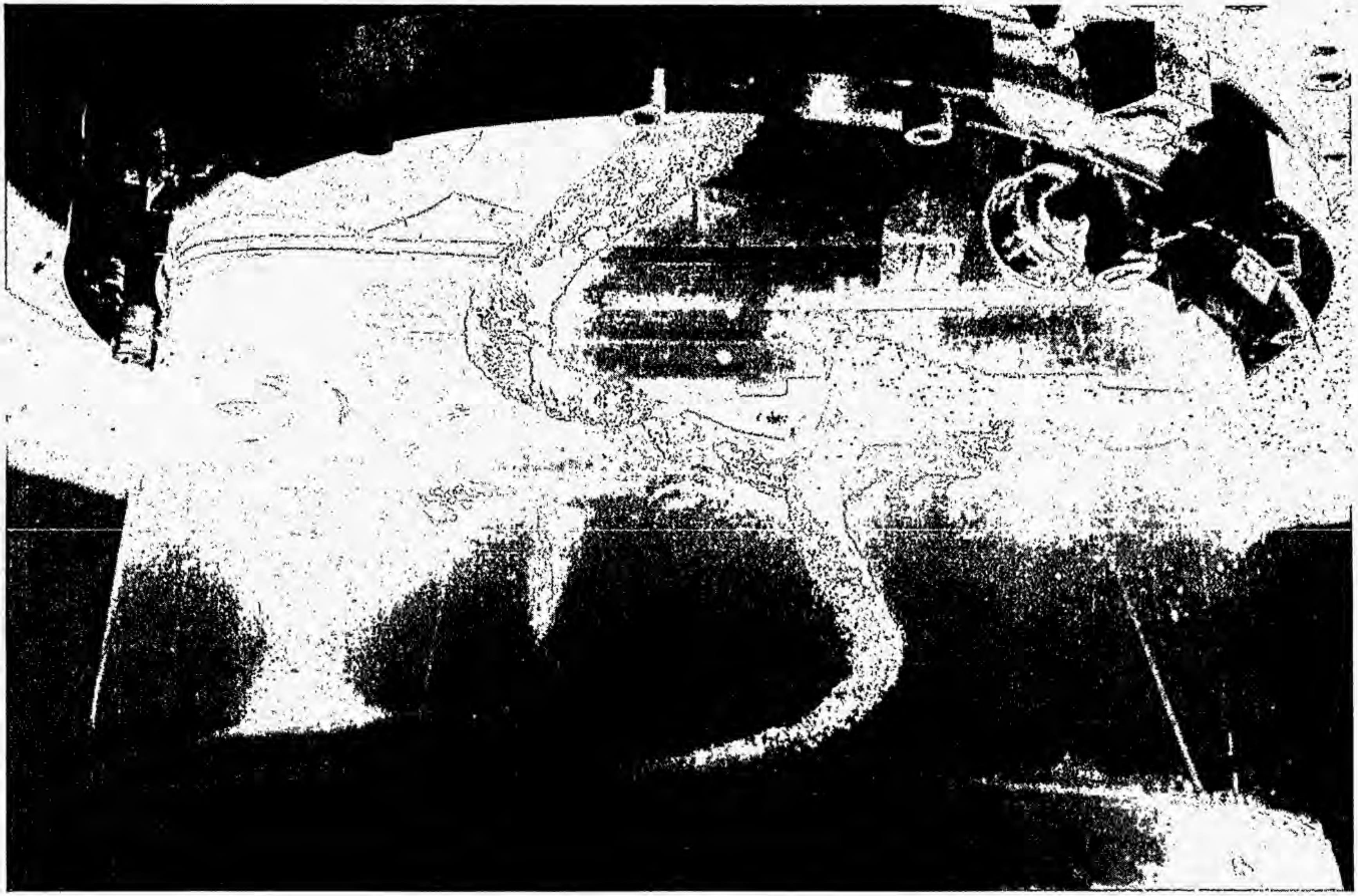


Figure 5.4.3-1b Visualization of cavitating vortex core at draft-tube entrance at off design point

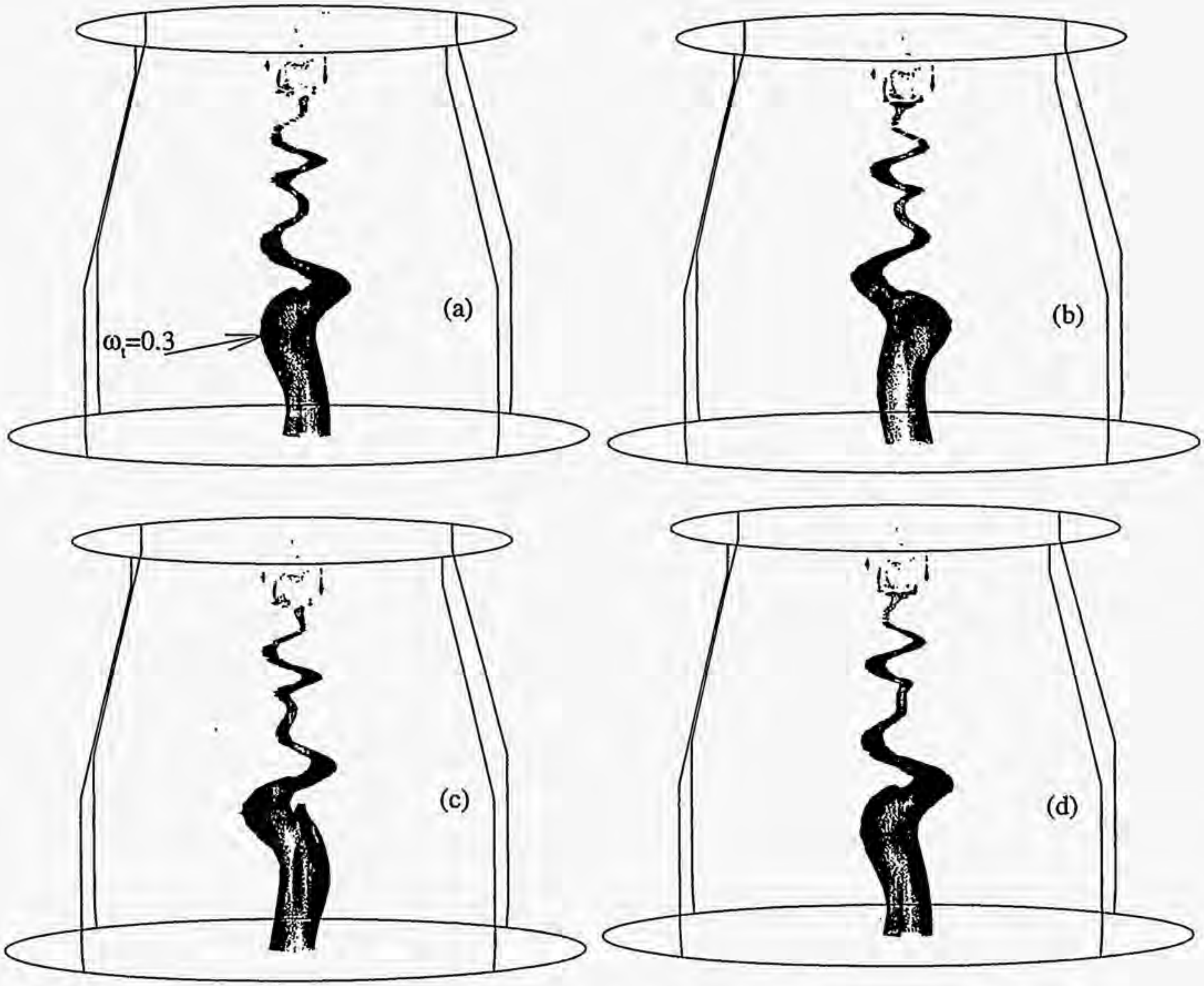


Figure 5.4.3-2 Instantaneous iso-surface of transverse vorticity magnitude for a straight diffuser, depicting the precession of a rope vortex. a) t=30 sec b) t=31.5 sec c) t=33 sec and d) t=34.5 sec. Axis aspect ratios have been distorted for clarity.

5.4.4 NUMERICAL SIMULATION OF FISH PASSAGE: CAN FISH BE MODELED AS FLUID PARTICLES?

Summary

A preliminary fish-passage numerical model was applied to calculate fish trajectories through a typical draft-tube geometry. The model treats fish as three-dimensional bodies with realistic physical and geometrical characteristics. The results demonstrate the inadequacy of modeling fish as fluid particles, underscore the potential of the proposed numerical model, and point to the need for future research in order to transform the present approach into a useful design tool.

Discussion

The progress we have made so far in the numerical simulation of hydroturbine flows clearly shows that advanced CFD methods can greatly enhance our understanding of the complex flow physics inside the powerplant. Understanding and being able to predict the flow details, however, is not, by itself, sufficient for evaluating the fish-friendliness of a given hydroturbine design. Achieving such an objective, further requires the development of a tool that can estimate the trajectories of fish inside a given complex flow environment. A first obvious attempt to develop such a tool could be based on treating fish as fluid particles which would imply that fish behave like material points moving with the local flow velocity. If this assumption were adequate, the output of a three-dimensional CFD calculation could be readily post-processed, by calculating particle paths, to estimate fish trajectories through the entire powerplant. Unfortunately, however, such an approach is not suitable for a number of reasons. First and foremost, a model which treats fish as fluid particles is not capable of predicting mechanical strike, scrape and abrasion related injuries, since a particle will always move with the flow around the various solid structures (vanes, runner blades, piers, etc.). Furthermore, in the vicinity of the runner even very small fish are likely to experience centrifugal forces, due to the runner rotation, which are known to have a profound effect on their trajectories (Cada et al., 1997). Obviously such effects can not be accounted for correctly if fish is modeled as a material point. Therefore, there is a need for developing a computational tool that models fish as three-dimensional bodies with geometrical (body length and thickness distribution) and physical (weight, locations of center of gravity and buoyancy, etc.) characteristics closely resembling those of the species under consideration.

In this section we present some results from the application of a preliminary three-dimensional fish-passage model to a typical draft-tube geometry. The model is based on the assumption that a fish swimming through a complex flow environment, obtained via a separate CFD calculation, can be approximated as a simplified, yet fish-like, geometrical body whose motion does not affect the pre-computed flowfield. The motion of this fish-like body is governed by six ordinary differential equations, for the components of the linear and angular acceleration vectors, whose source terms include hydrodynamic (lift, drag, pressure forces, inertial forces, etc.) and biological (fish "free-will") forces. A preliminary version of this model--which accounts only for inertial forces acting on the fish body--was applied to calculate fish trajectories through TVA's Norris Dam draft tube. It should be emphasized that all subsequently shown results are preliminary and serve only to demonstrate the need for further research on fish modeling. Figures 5.4.4-1 and 5.4.4-2 compare the trajectories of two different fish "species", respectively: a material particle (i.e. fish trajectories coincide with particle paths), and a three-dimensional fish-like body of 0.05 kg weight. One important attribute can be readily observed: the fish-like body enters the draft-tube with much more inertia and tends to hit the walls of the tube. As discussed above, this trend underscores the effect of centrifugal forces exerted on even small fish and points to the inadequacy of treating fish as fluid particles.

Further research is necessary to transform the preliminary fish-passage model into a useful design tool. Specific topics that need to be addressed include: i) accurate implementation of the various hydrodynamic forces acting on the fish body; ii) development of a fish free-will model; iii) execution of carefully designed field and laboratory experiments with real fish to collect data necessary for the calibration and validation of the model; and iv) formulation of the model so that it is applicable to the entire powerplant. It is important to point out that in most of these areas close collaboration will be necessary between the CFD modelers, fish biologists, turbine manufacturers, and powerplant operators.

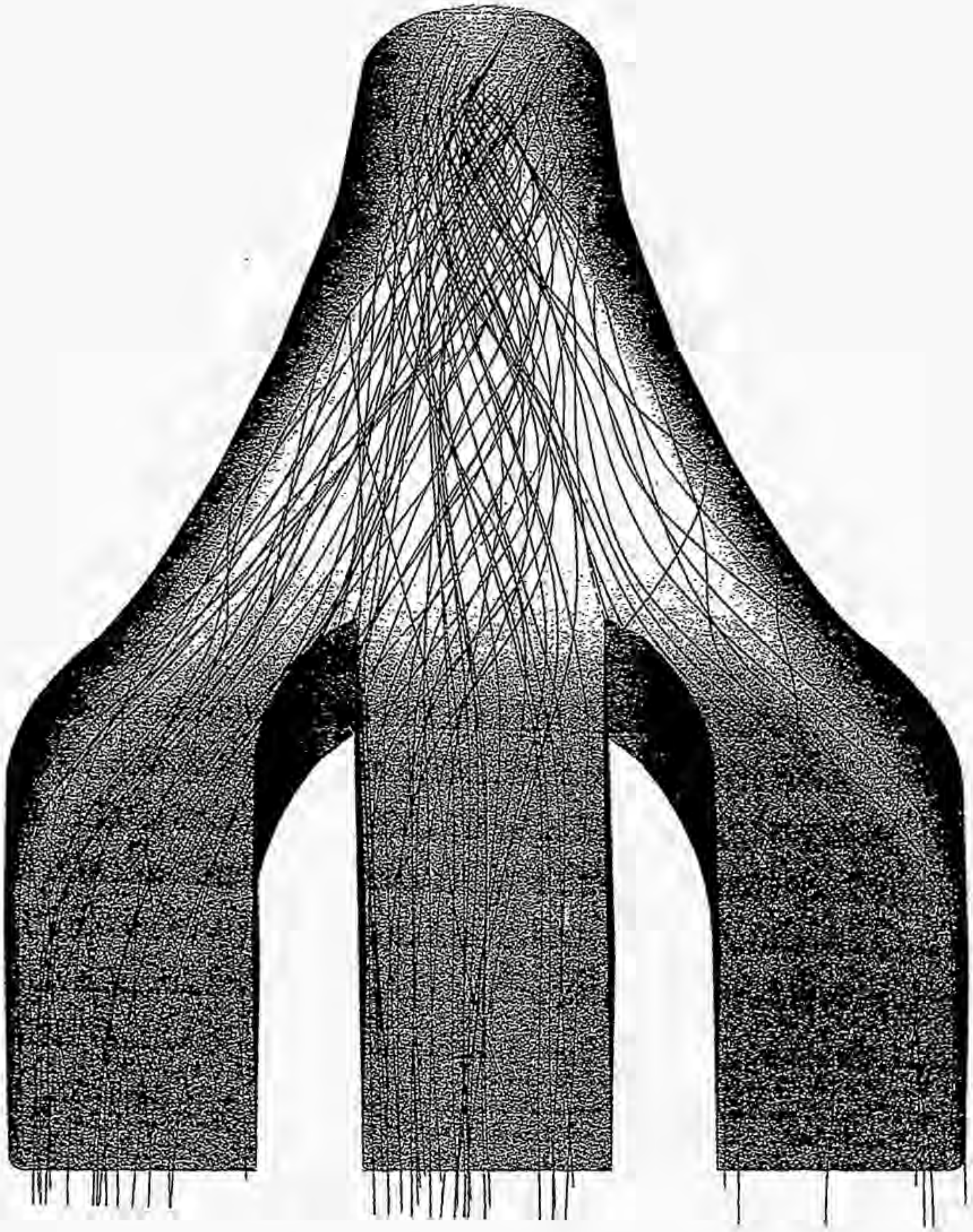


Figure 5.4.4-1 Calculated trajectories for fish modeled as flow particles.

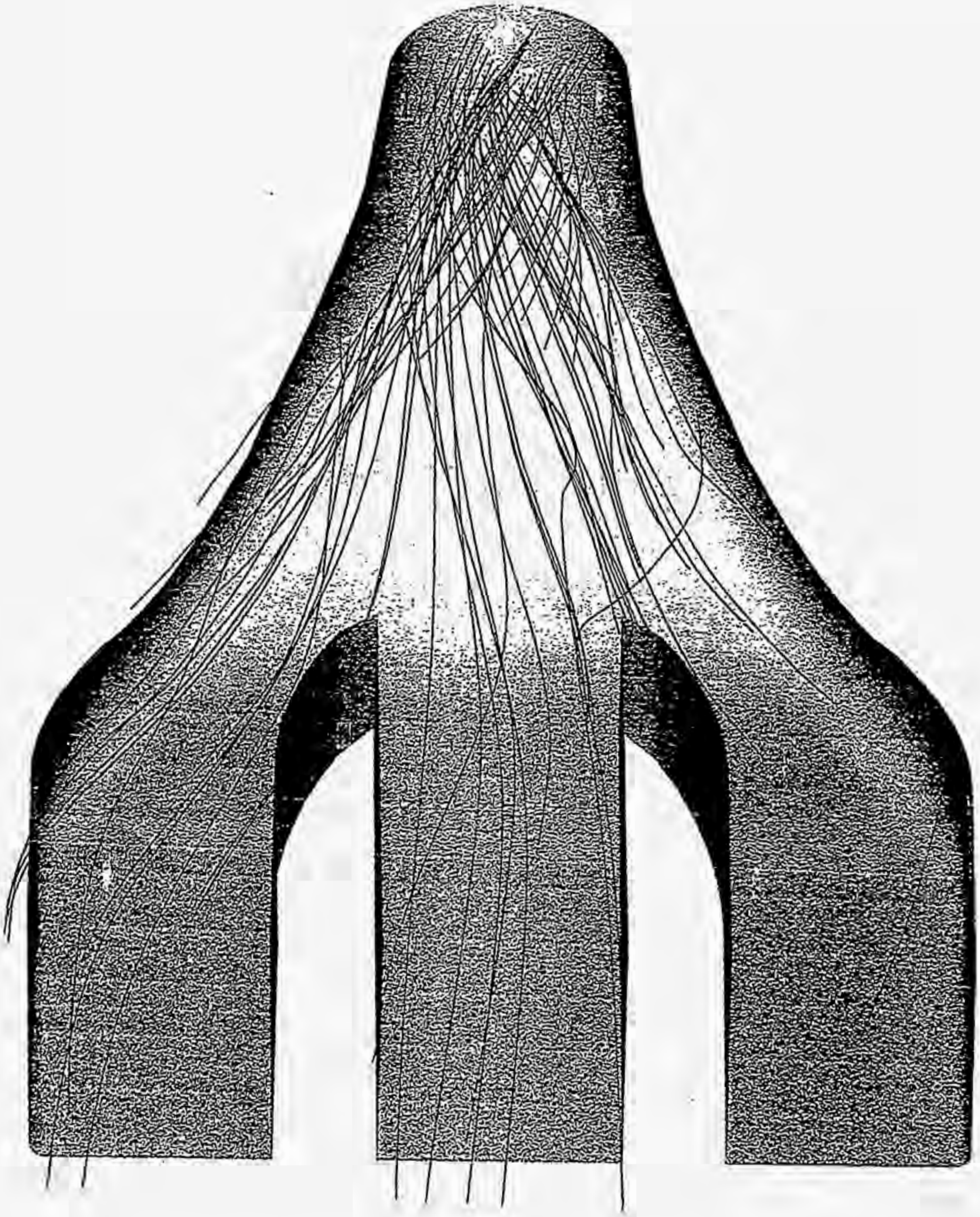


Figure 5.4.4-2 Calculated trajectories for fish-like bodies of 0.05Kg weight.

5.4.5 SUMMARY AND CONCLUSIONS

The most important findings of this study are summarized as follows:

The non-linear k-w turbulence model developed herein was shown to reproduce the measured very complex vortex structure inside a strongly curved duct with remarkable accuracy. Calculations of the same test case with a number of other advanced turbulence models failed to capture, even qualitatively, essential physics of this flow. Therefore, the proposed non-linear k-w model is a very promising turbulence closure for quantitatively accurate predictions of complex flows of engineering interest. It should be emphasized that this model does not require the distance from the wall which facilitates its application to geometries involving multiple intersecting walls, such as those encountered in components of a hydropower installation. Furthermore, this model is computationally very efficient, as it requires the solution of only two additional transport equations and, thus, provides us with a practical tool for engineering calculations on fine computational meshes. Future work will focus on evaluating the performance of this model in real-life hydroturbine flows.

Calculations of flow through a typical draft-tube geometry, with two downstream piers, were carried out using linear k-e and k-w models. The computed results were compared with few mean streamwise velocity measurements taken at selected locations within the three bays downstream of the elbow. The calculations reproduce most experimental trends with reasonable accuracy. Three-dimensional particle traces revealed the presence of very complex three-dimensional flow patterns around the piers. These include longitudinal and horse-shoe vortex formation, and regions of reversed flow. Lack of detailed three-dimensional measurements and uncertainties in the specification of inlet boundary conditions, however, prohibited a comprehensive evaluation of various turbulence models. The present study clearly underscores the need for detailed three-dimensional experiments for draft-tube geometries in order to make further progress in developing and evaluating advanced turbulence models for such flows.

A three-dimensional numerical method was developed for simulating unsteady hydroturbine flows. The method was applied to calculate the time evolution of swirling flow through a simplified draft-tube geometry. Analysis of the computed results revealed the formation of a rope-like vortex precessing in a periodic fashion about the tube axis. The general structure and temporal evolution of this vortex are in qualitative agreement with laboratory and prototype observations of similar unsteady vortex phenomena which occur when Francis units operate at off-design conditions. The present results demonstrate that the numerical computation of complex unsteady hydroturbine flows–which is a crucial prerequisite for understanding such flows over the entire range of possible operating conditions–is now well within reach. Future work will focus on applying the method developed herein to real-life draft-tube geometries as well as extending the present methodology to simulate unsteady coupling of runner and draft-tube configurations.

A preliminary three-dimensional numerical model was applied to calculate trajectories of fish-like bodies through a typical draft-tube geometry. The main conclusion of this study is that treating fish as particles which passively follow the flow is inadequate for realistic simulations. A fully three-dimensional model is, therefore, needed which takes into account the major geometrical and physical characteristics of a given species as well as the various hydrodynamic forces exerted by the flow on the fish body. Such a fish-passage model may be utilized in conjunction with flow solutions obtained via CFD methods to estimate fish trajectories through the various components of a hydroturbine installation. With input from fish biologists and carefully designed laboratory experiments and field measurements with real fish, such a model could be used to estimate the fish friendliness of a given turbine design. It can, thus, provide the hydropower industry with a powerful tool for developing and evaluating innovative rehabilitation strategies for improving the environmental compatibility of its facilities.

5.4.6 REFERENCES

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5.5 DISSOLVED OXYGEN ENHANCEMENT USING TURBINE AERATION

A variety of methods are available to provide dissolved oxygen (DO) in hydropower releases. For the most part, these methods can be grouped into one of four categories–reservoir techniques, powerhouse techniques, tailwater techniques, and operational techniques (see Bohac and Ruane, 1990, or EPRI, 1990). Examples of each are given in Table 5.5-1. Where applicable, turbine aeration is usually the method of choice. Consequently, any work to develop advanced hydroturbine technologies must include objectives for aeration, at least for projects containing low DO.

In general, turbine aeration can be provided by natural aspiration or forced injection. With natural aspiration, air is supplied to openings in the turbine where the pressure is subatmospheric. In some cases, subatmospheric conditions are created by adding small deflectors or baffles on flow boundaries in the turbine. Outside, the openings are vented to the atmosphere, thus providing the pressure difference to draw air into the water. A naturally aspirating turbine is also called an auto-venting turbine (AVT). With forced injection, the pressure in the turbine is above atmospheric, requiring compressors or blowers to push air into the water. In both the AVT and forced injection arrangements, the DO is increased by the transfer of oxygen from the entrained air to the water. Due to the minimal requirements for extra mechanical equipment, as well as reduced expenses for operation and maintenance, the AVT usually is the least-cost option for aeration in hydroturbines.

Reservoir Techniques
• Forebay Destratification
• Hypolimnetic Air or Oxygen Diffusers
• Epilimnetic Pumps
Powerhouse Techniques
• Penstock Air or Oxygen Diffusers
• Turbine Aeration
Tailwater Techniques
• Surface Aerators
• Side-Stream Aeration
• Aerating Weirs
Operational Techniques
• Sluice or Spillway Aeration
• Selective Withdrawal
• Special Turbine Operations

Table 5.5-1 Methods for Improving DO in Hydropower Releases

In examining advanced technologies for turbine aeration, attention must be given both to existing units and to new units. For the latter, a wide range of design factors and, consequently, potential aeration alternatives exists to improve DO in the turbine discharge. This is attributable to the fact that within the overall constrains of most projects, there is usually some flexibility in selecting the shape and position of the turbine components and surrounding equipment. This especially is true in new construction. Although extra physical limitations can be imposed by existing structures, and this also can be true in upgrade situations involving new, modernized units. At most currently operating projects, however, new modernized turbines cannot be economically provided for the sole purpose of improving DO. Hence, any effort to develop advanced technologies for turbine aeration needs also to consider retrofit arrangements for existing units.

In this section, discussions focus on methods to provide aeration in existing and new turbines. Because TVA currently is the industry leader in using this technology, most of the cited experience includes examples of one or more of the agency's hydro projects. Summaries of state-of-the-art practice and advanced technology for turbine aeration are given. Methods of analysis for the environmental and hydraulic performance (i.e., DO uptake and aeration-induced efficiency loss) of aerating turbines are assessed. Recommendations for testing aerating turbines are presented, as well as requirements for efficient operation of these units.

5.5.1 STATE-OF-THE-ART PRACTICE AND ADVANCED TECHNOLOGY

Research to mix air with the water flowing through a turbine for the purpose of improving DO in hydropower releases has been in progress for nearly half a century. Almost all these studies have focused on developing retrofit arrangements to provide aeration in existing turbines. This, of course, is due to the fact that almost all existing turbines are designed for the sole purpose of producing power, not for providing aeration. Since about 1990, however, the need for turbine upgrades combined with regulatory requirements for preserving and enhancing aquatic wildlife in tailrace channels has encouraged research for methods to include aeration as an integral part of the design of new turbines. The following discussions are given to summarize state-of-the-art practice and objectives for advanced technology in providing aeration in existing and new units.

5.5.1.1 **Existing Turbines**

Background
Studies by Wagner (1958) to improve water quality on the Neckar River in Germany are among the earliest examples of aeration for existing turbines. About the same time, turbine venting was introduced in the United States to mitigate the adverse impact on water quality of discharges from pulp and paper factories and municipal sewage systems in Wisconsin (Lueders, 1956). By 1961, turbine aeration was in use at eighteen hydroplants on the Flambeau, Lower Fox, and Wisconsin Rivers (Wiley et al., 1962; Wisniewski, 1965). This and other studies prior to 1982, including studies by both U.S. and European hydropower owners, are summarized by Bohac et al. (1983). A more recent summary of experience with TVA projects is given by Carter (1995).

In existing units, the evolution of turbine aeration has led to a variety of retrofit designs. These typically provide air at one of two locations, the vacuum breaker outlet or the draft tube. The best arrangements include turbines that contain subatmospheric pressures and airflow passageways large enough to insufflate the required amount of air without any changes to the unit. In these cases, natural aspiration is achieved merely by blocking open the control valve for the air supply passageway, typically the vacuum breaker valve. Often, however, physical modifications are required to enhance airflow and obtain the desired DO uptake. As mentioned earlier, baffles placed over the aeration outlets will locally decrease pressures. Ventilation pipes added to the turbine headcover can be used to reduce pressure losses due to friction, bends, valves, and other devices in the airflow passageways. Because these pipes usually "short-circuit" the vacuum breaker valve, they often are called bypass conduits. A typical installation of baffles and bypass conduits is shown in Figure 5.5-1.

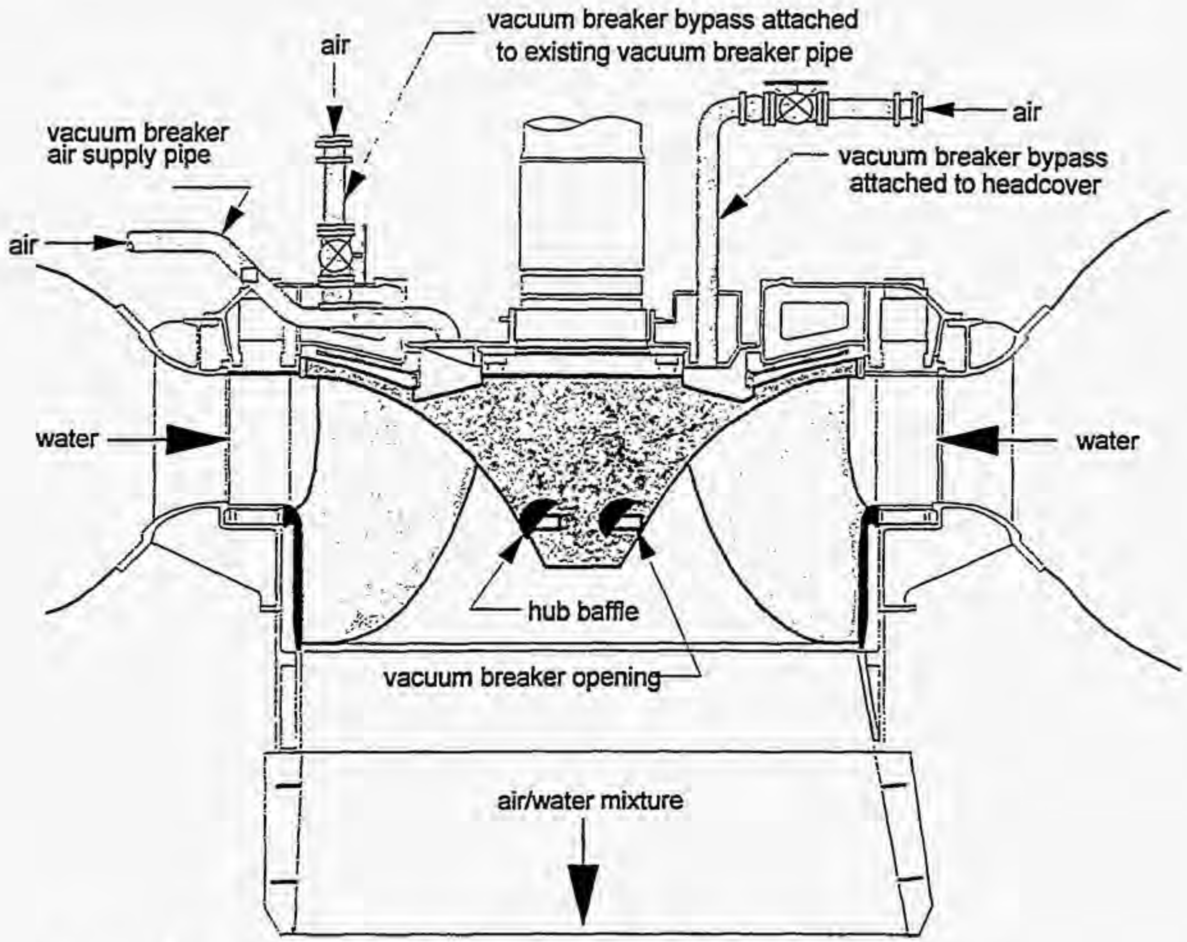
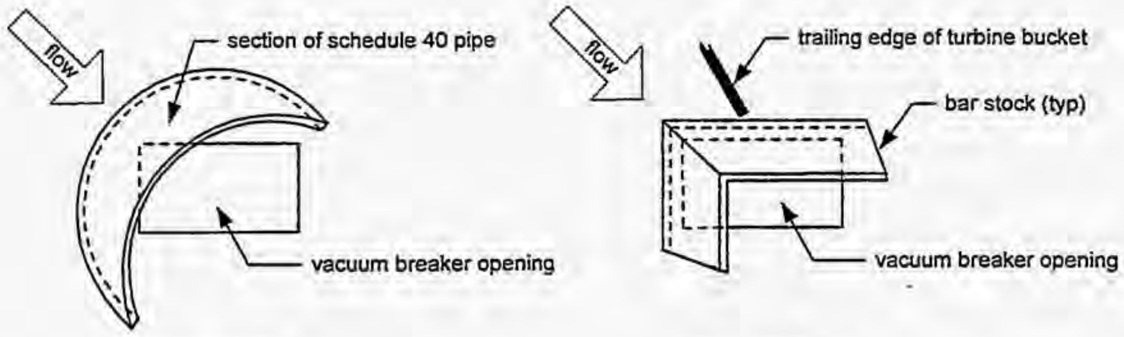


Figure 5.5-1 Francis Turbine Retrofitted with Hub Baffles to Enhance Aeration

If baffle and bypass modifications fail to provide the required airflow, two options remain–operational changes and forced injection. Subatmospheric pressures usually can be induced by operating the turbine at low settings below peak efficiency gate. Due to the resulting efficiency and capacity losses, as well as adverse surging and vibration, such operations are undesirable. However, in situations where an immediate solution is needed (e.g., to avoid a fish kill), operational changes may be the only option available to provide aeration. In forced injection, air is supplied to outlets in the turbine by blowers or compressors. The outlets usually include the vacuum breaker openings or circumferential manifolds near the entrance of the draft tube. The capital and operation and maintenance costs for forced air injection is almost always higher than costs for naturally aspirating turbines.

Hub Baffles

Baffles for retrofit Francis installations usually are positioned on the hub over the vacuum breaker openings (Figure 5.5-1). Two designs are common–streamlined and flat plate baffles. The basic geometry of each is given in Figure 5.5-2. The streamlined baffle is suitable for turbines containing vacuum breaker openings roughly 10 inches or more below the trailing edge crown fillet of the turbine buckets. Although exceptions occur, the clearance between the baffles and buckets is usually too small



Streamlined Baffle

Flat Plate Baffle

Figure 5.5-2 Typical Hub Baffle Arrangements

for shorter distances. The streamlined baffle is typically fabricated by cutting a wedge from standard 10-inch diameter schedule 40 pipe, creating an "eyelid-shaped" piece to shroud the vacuum breaker outlet (see Harshbarger, 1984). The baffle is positioned perpendicular to the direction of flow as determined by the typical erosion wear patterns on the turbine hub. In most cases, this position places the baffle at about 45° from vertical. Along the centerline of the baffle, the deflection angle for flow along the hub is about 60°. An example installation of streamline hub baffles is shown for the original turbines at TVA's Norris Dam in Figure 5.5-3.

The flat plate baffle is typically fabricated from bar stock and is used for vacuum breaker outlets located on the turbine crown near the trailing edge of the buckets. The plates are situated 90° horizontal-to-vertical to create an L-shaped wedge over the openings (see Figure 5.5-2). Due to the narrow clearance between the hub and turbine blades, the overall height of the flat plate baffle is lower than that of the streamlined baffle. The deflection angle at the hub typically is near 45°. In designs used thus far, the baffle is situated so that the downstream edge of the horizontal plate is aligned with the center of the vacuum breaker opening. To permit access for welding, the upstream edge usually is mounted about 1-inch above the top of the outlet. In some cases, an outlet may span the trailing edge of the turbine bucket and obstruct the upstream edge of the baffle. In these cases, the horizontal plate is notched to allow the trailing edge to bisect the baffle.

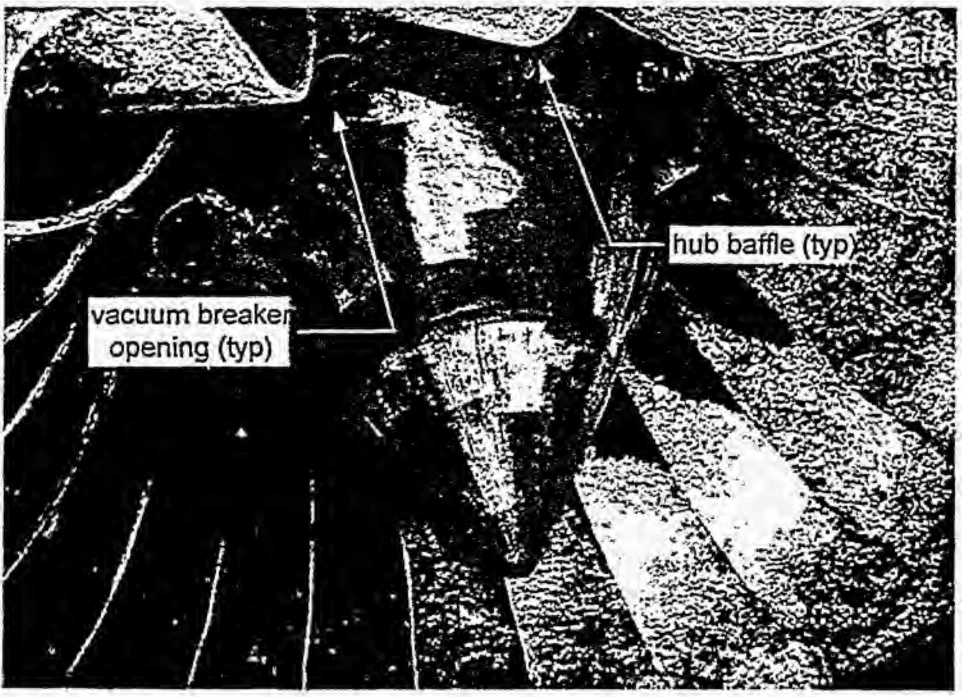


Figure 5.5-3 Streamlined Hub Baffles For TVA's Norris Dam

Bypass Conduits
Bypass conduits usually include a butterfly valve, a check valve, a debris strainer, and a bellmouth inlet. A typical arrangement is shown in Figure 5.5-4. The butterfly valve is used to control airflow, and the check valve is used to prevent the backflow of water from the turbine. The strainer is required to prevent loose objects from entering the bypass, which can damage turbine seals or obstruct vacuum breaker openings. The bellmouth inlet reduces both noise and pressure losses at the entrance of the conduit.

Two common locations for bypass conduits are illustrated in Figure 5.5-1. In the first, the bypass is attached to an opening cut in the turbine headcover. This arrangement provides the most direct and unobstructed flowpath for air entering the turbine. In the second location, the bypass is attached to an opening cut in the existing vacuum breaker air supply pipe, usually at a point downstream of the vacuum breaker valve. In most cases the conduits can be positioned at locations unobstructive to surrounding equipment and include a handwheel operator to open and close the bypass butterfly valve.

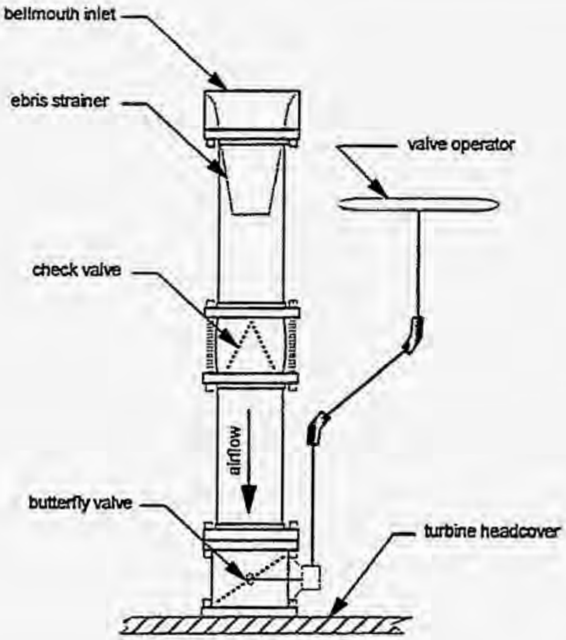


Figure 5.5-4 Typical Arrangement for Bypass Conduit

Experience
Table 5.5-2 provides a sample of projects containing experience for retrofitting existing turbines for aeration. This data was collected from published literature and personal contacts. The table includes information only for original equipment (e.g., some units have been upgraded with new turbines). At TVA, retrofit arrangements have been used on 23 units at 10 projects. Twenty-two of the units, all Frances turbines, include auto-venting capabilities provided by hub baffles and/or bypass conduits. The remaining unit, a mixed flow turbine at the Tims Ford Dam, uses forced air injection through the headcover (both in the original and new turbine). The DO uptake by the TVA retrofit arrangements typically is 2.0 mg/L or more. For this amount of oxygenation, the accompanying aeration-induced turbine efficiency loss usually is less than 2 percent (see March et al., 1992). In general, based on TVA experience with retrofit arrangements in low to medium-head Francis units (80 to 200 feet), the amount of aeration-induced turbine efficiency loss varies between 0.5 and 1.0 percentage points per mg/L of uptake depending on the operating conditions of the unit (i.e., head, discharge, and incoming DO). This efficiency loss is experienced only when the air supply conduits are open to provide oxygenation of the turbine discharge. The efficiency loss due solely to the presence of the baffles is typically 0.5 percent or less.

Although aeration is not needed year round, TVA experience has found that the energy loss attributable to hub baffles is small compared to the cost to temporarily install these devices for the low DO season. As such, hub baffles for TVA projects are attached as permanent equipment. Also, cavitation damage has not been significantly increased by baffles, and in several cases enhanced airflow has reduced adverse surging, load swings, and turbine vibrations.

Project	Owner	Turbines			Aeration Features	DO Uptake (mg/L)
		Type	No.Units	Power (MW)		
American Falls	Idaho Power	*	*	*	forced aeration thru draft tube	2.0
Apalachia	TVA	Francis	2	39.5	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Bagnell	Union Electric	Francis	8	25.0	vacuum breaker	*
Baldeney	*	*	*	*	forced aeration	*
Bankhead	Alabama Power	Kaplan	1	*	auto-venting thru air distribution conduit with draft tube deflectors	1.7
Bartletts Ferry	Georgia Power	*	*	*	auto-venting thru air distribution conduit with draft tube deflectors	*
Blanchard	Minnesota Power & Light	*	3	*	auto-venting thru air distribution conduit with draft tube deflectors	*
Boone	TVA	Francis	3	25.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Box Canyon	*	Francis	*	*	vacuum breaker	*
Bull Shoals 1-4	USACE	Francis	4	38.8	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Bull Shoals 5-8	USACE	Francis	4	41.8	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Cascade	Idaho Power	Kaplan	2	6.4	forced aeration thru draft tube	*
Cherokee	TVA	Francis	4	30.9	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.5
Clarks Hill	USACE	*	*	*	auto-venting thru draft tube air distribution conduit	1.5
Conowingo	Susquehanna Power	*	*	*	auto-venting thru vacuum breaker	*
Coyote	*	Francis	*	*	auto-venting thru vacuum breaker	*
Deer Creek	Bureau of Reclamation	Francis	2	2.5	auto-venting thru vacuum breaker and snorkel tube	1.2
Douglas	TVA	Francis	4	26.5	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Fontana	TVA	Francis	3	68.2	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.5
Hiwassee	TVA	Francis	1	59.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	1.0
Holt	Alabama Power	*	1	*	*	*
Kaw	KAMO Electric Co-op.	*	*	*	forced aeration thru draft tube	*
Kimberly	*	Propeller	4	0.9	*	*
Logan Martin	Alabama Power	Propeller	3	42.0	auto-venting thru air distribution conduit with draft tube deflectors	0.5
Martin	Alabama Power	*	4	*	*	*
Mitchell	Alabama Power	Propeller	4	50.0	auto-venting thru air distribution conduit with draft tube deflectors	1.0
Monticello	*	Francis	3	*	vacuum breaker	*
Norfork	USACE	Francis	2	31.3	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Norris**	TVA	Francis	2	64.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	5.5
Pixley	Flambeau Paper	Francis	2	0.6	auto-venting thru draft tube air distribution conduit	1.0
Poppenweller	*	Kaplan	2	1.8	auto-venting thru draft tube air distribution conduit	*
Rapide Croche	*	Propeller	1	0.8	auto-venting thru vacuum breaker	1.5
Rat Rapids	*	Francis	4	0.32	auto-venting thru vacuum breaker	0.8
R.L. Harris	Alabama Power	Kaplan	2	67.5	auto-venting thru air distribution conduit with draft tube deflectors	2.0
Rothschild	*	Francis	6	0.8	auto-venting thru vacuum breaker	1.5
Safe Harbor	Pennsylvania Power & Light	*	*	*	draft tube	*
Shepaug	Connecticut Light & Power	Kaplan	1	42.0	forced aeration thru draft tube	2.0
South Holston	TVA	Francis	1	36.2	auto-venting thru vacuum breaker and bypass conduit	2.0
Tenkiller Ferry	USACE	Francis	2	28.3	auto-venting thru vacuum breaker with hub baffles and bypass conduit	*
Tims Ford**	TVA	Mixed Flow	1	*	forced aeration thru vacuum breaker	*
Watauga	TVA	Francis	2	25.7	auto-venting thru vacuum breaker with hub baffles and bypass conduit	2.0
Winfield	American Electric Power Service	Kaplan	3	6.15	auto-venting thru vacuum breaker	3.0
Wylie	Duke Power	Francis	4	15.0	auto-venting thru vacuum breaker	0 - 2
Youghiogheny	*	*	1	*	forced aeration thru draft tube	2-5

*Missing Information

**Original Turbines

Table 5.5-2 Aeration Experience In Retrofitting Existing Turbines

Outside of TVA, a variety of retrofit experience is found. Of particular note is that for Kaplan turbines (e.g., see Miller and Sheppard, 1983). There presently are no feasible alternatives to retrofit Kaplan runners with devices such as hub baffles, and aeration is induced by adding deflectors in the draft tube. To offset lower velocities and higher pressures found at this location, such arrangements usually require a deflector of larger size than that of hub baffles. Favorable performance is reported for both the amount of turbine efficiency loss and DO uptake. However, based on the indicated volume of air entrainment, the reported turbine efficiency loss appears to be unrealistically low. Other Kaplan experience is found in the original work by Wagner (1958). One such installation included an aeration ring immediately below the runner plane. In this case, tests showed that airflow was highly sensitive to operation of the unit and that the efficiency losses were substantial.

Advanced Technology

In existing turbines, work to develop advanced aeration technology needs focus in several areas, including the following:

- Evaluation tools. Advanced numerical models using computational fluid dynamics (CFD) are needed to determine the detailed hydraulic characteristics of retrofit arrangements. In particular, the velocity, Reynolds stress, and pressure fields associated with the three-dimensional viscous flow around baffles and deflectors are needed to examine designs for these devices. The techniques need to be coupled with models for the overall flow through the unit to properly evaluate the effects of the turbine boundaries, runner rotation, draft tube swirl, tailwater, and so on. Numerical models for the single phase flow of water would provide abundant help in design efforts. However, to estimate the extent of cavity formation and entrainment in the wake of baffles and deflectors, models capable of determining the characteristics of two-phase air/water flow are needed. Models for multiphase flow also are required to compute the overall effect of aeration on turbine performance (i.e., efficiency loss).
- Validation data. Hand-in-hand with evaluation tools, model and prototype data for retrofit arrangements will be needed to validate computational techniques. Depending on the phenomenon being examined, validation experiments can range from simple water tunnel studies to full-scale turbine tests.
- Retrofit optimization. Although the experience summarized in Table 5.5-2 includes many successful applications of retrofit technology, it is unknown whether or not these designs are optimal. The arrangements used by TVA have evolved over a number of "trial-and-error" experiments. Using evaluation tools as discussed above, variations of these designs could be examined to see if enhancements can be obtained by increasing the amount of DO uptake and decreasing the aeration-induced turbine efficiency loss. Deflector and baffle arrangements for projects with high tailwater are especially in need of improvement. For these cases, investigations should examine larger baffles and deflectors positioned at different locations in the turbine runner and draft tube.
- Airflow passageway improvements. Data from tests at TVA indicate that sealwater interference may be obstructing airflow through passageways feeding aeration outlets located in the turbine runner. Advanced retrofit technologies should consider turbine modifications that better isolate airflow passageways from seal leakage or backflow from the aeration outlets.

5.5.1.2 **New Turbines**

Background

Discussions of including aeration as an integral part of the design of new turbines began in the early 1980's. Formal action to implement ideas, however, did not emerge until 1987. At that time, TVA initiated the Hydraulic Turbine Aeration Research Program to develop physical and numerical modeling techniques to aid in the design of aerating units and to demonstrate the feasibility of this technology in a full-scale installation. The primary goal of the program was to develop methods of aeration in new turbines that would provide up to 6 mg/L of DO uptake in hydro releases while minimizing adverse side-effects on turbine efficiency, capacity, and reliability.

Based on the success with hub baffles in the original units, TVA's Norris Dam was selected as the experimental site to demonstrate the application of aeration technology for new turbines. Norris Dam also was among the first projects containing discharges of low dissolved oxygen scheduled for runner replacement in TVA's Hydro Modernization Program. This program includes an aggressive plan to upgrade hydro facilities at twenty-four TVA projects, with the replacement of eighty-eight turbine runners. In 1988, TVA contracted two hydroturbine manufacturers to use existing numerical models in evaluating the performance of the turbines at Norris Dam. The manufacturers were asked to propose and evaluate alternatives for increasing aeration and to estimate the performance for the upgraded aerating turbines. The recommended alternatives received by this process ranged from a continuation of hub baffle technology to a variety of innovative methods to provide air at a number of sites inside and around the turbine. These sites, shown in Figure 5.5-5, include the headcover, bottom ring, runner crown, deflector, hub snorkel, discharge ring, runner band, runner entrance edge, runner discharge edge, draft tube cone, and draft tube coaxial diffuser (March et al., 1991).

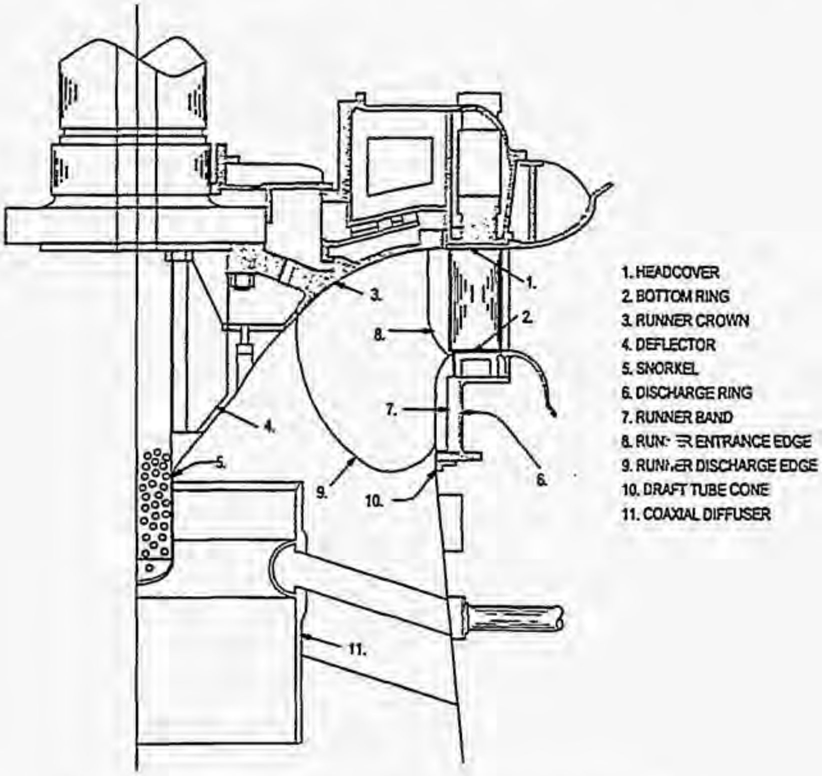


Figure 5.5-5 Aeration Alternatives for New Hydroturbines

Based on results from the 1988 study, TVA and Voith Hydro, Inc., initiated, in 1991, an extensive physical model program to investigate performance of the turbine aeration alternatives for the new turbines at Norris Dam. The tests included measurements for efficiency, cavitation, runaway speed, pressures and pressure pulsations, airflow, and DO uptake (Cybularz et al., 1992). Procedures for measuring and scaling the aeration performance of turbines were developed during the course of the model test. To obtain a qualified set of data by which scaling relationships could be developed, a model test was performed of the original turbines. Prototype data for these units was available from field test previously conducted to evaluate the performance of hub baffles. The original turbines at Norris had a capacity of about 53 MW at a net head of about 170 feet (Dodson, 1995). Major aspects of the testing and scaling procedures developed in the Norris model studies are presented later in this report. The major finding from this work is related to scale effects that occur in modeling air/water mixtures. Due to these scale effects, different model operating conditions are required to evaluate the environmental performance (i.e., DO uptake/gas transfer characteristics) than those needed to evaluate hydraulic performance (i.e., aeration-induced efficiency loss/energy dissipation characteristics) of aerating turbines. For non-aerating conditions, performance was determined using conventional procedures for modeling homologous hydraulic turbines.

Based on the model studies, the resulting designs chosen for the prototype turbines at Norris contain five aeration alternatives. Those for unit 2 (Figure 5.5-6 and 5.5-7) include the runner deflector (DEF), bucket discharge edge (DE), thrust relief (TR), and draft tube cone (DTC). Unit 1 is similar, but contains an airflow passageway to the runner discharge ring (DR) rather than the draft tube cone.

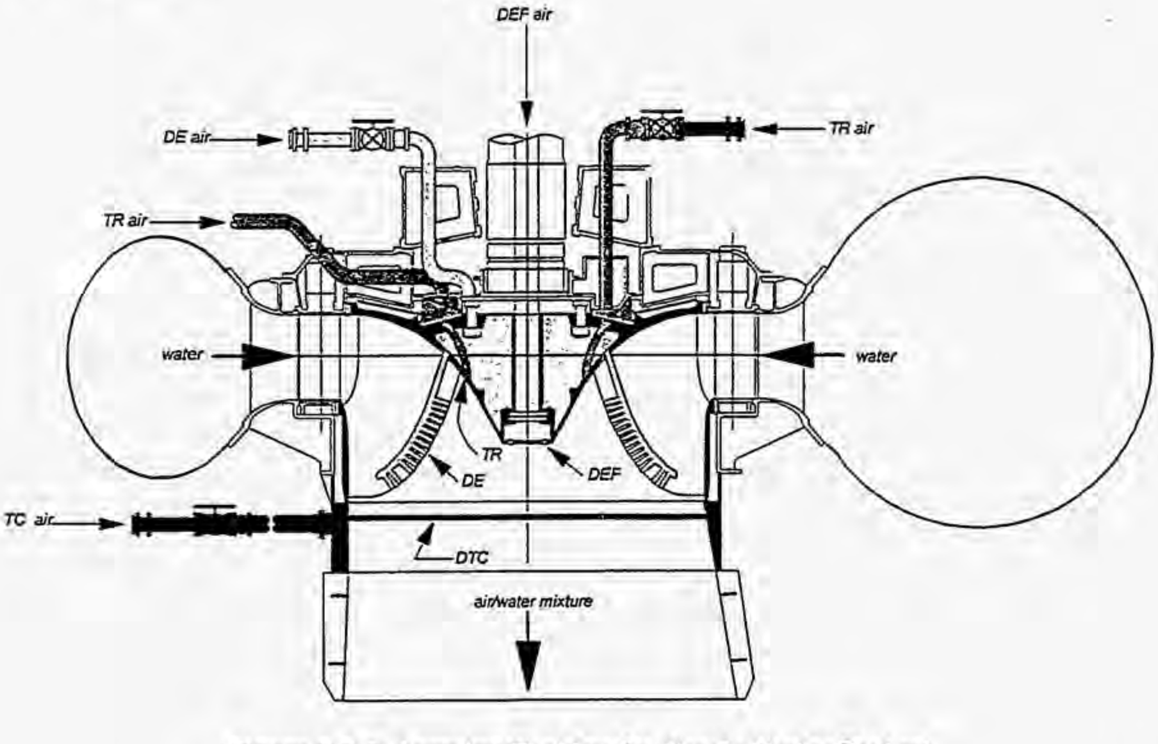


Figure 5.5-6 Aerating Turbine for Norris Hydro Project

The new turbines for Norris, installed in 1995 and 1996, have undergone extensive testing to evaluate the environmental and hydraulic performance of the various aeration alternatives (Hopping et al., 1996). The testing included single and combined operation of the alternatives over a wide range of turbine flow conditions. For environmental performance, results show that up to 5.5 mg/L of DO uptake can be obtained for single unit operation with all aeration options in service. In this case, the amount of air aspirated by the turbine is more than twice that obtained in the original turbines with hub baffles. The resulting bubble plume in the turbine discharge is shown in Figure 5.5-8. To meet the 6.0 mg/L target established for the project tailwater, an additional 0.5 mg/L of DO improvement is obtained by flow over a re-regulation weir downstream of the powerhouse. For hydraulic performance, efficiency losses ranging from 0 to 4 percent are obtained, depending on the operating condition and the aeration options. Compared to the original turbines, the Norris units provide overall efficiency and capacity improvements of 3.5 and 10 percent, respectively (March and Fisher, 1996). The new runners also have demonstrated significant reductions in both cavitation and vibration.

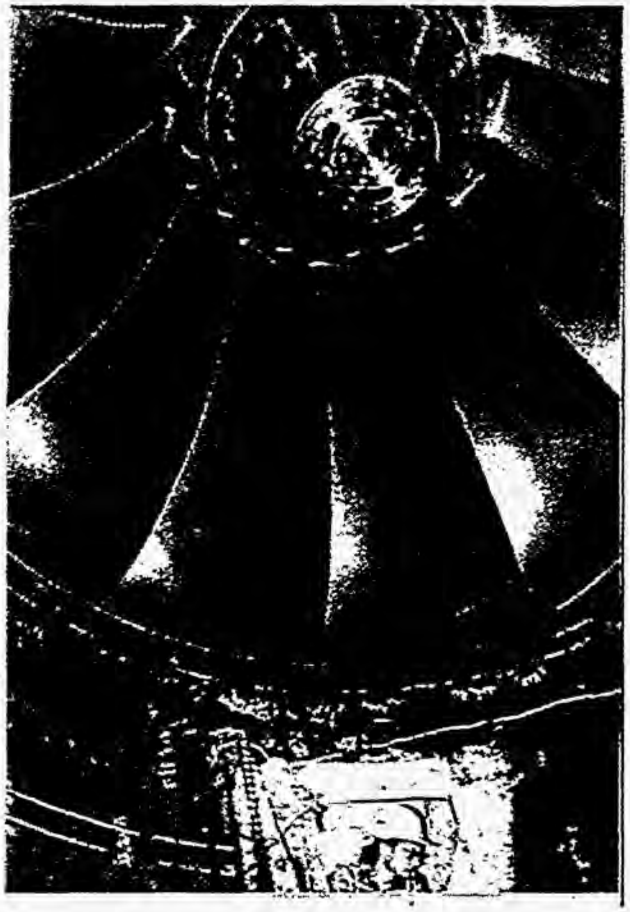
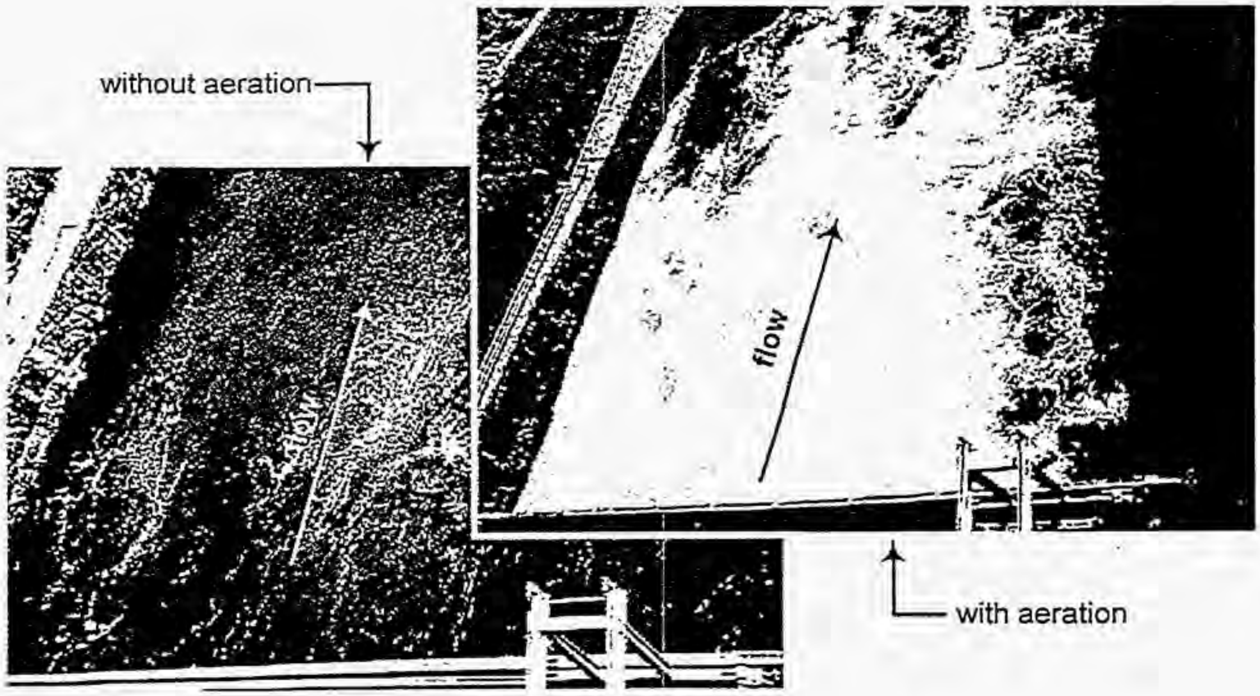


Figure 5.5-7 New Auto-Venting Turbine for Norris Dam Unit 2



• **Figure 5.5-8** Bubble Plume in Tailwater at Norris Dam

Design and Applicability

The fundamental requirement in the design of an aerating turbine is the ability to supply air to outlets inside the turbine in an efficient and cost-effective manner. In this, the shape and position of the turbine components and surrounding equipment need to be determined so that no degradation occurs in the non-aerating performance of the unit. In general, auto-venting alternatives that aspirate air as a natural consequence of the turbine geometry provide the best aeration alternatives. Auto-venting technology is applicable only for turbines containing locations where subatmospheric pressures can be found or created. Aeration sites without baffles or other obstructive devices are desired.

Compared to other alternatives to improve dissolved oxygen (i.e., Table 5.5-1), turbine aeration in new units usually is optimal at projects where the aeration-induced head loss is small compared to the overall turbine head. Such projects tend to include Francis units. As previously emphasized, aeration-induced losses in the draft tube can limit the feasibility of aeration in low head projects containing propeller-type units. At this time, no attempts to include objectives for turbine aeration as an integral part of the design of new Kaplan turbines (i.e., that is for DO improvement in reservoir releases) are underway.

In supplying AVT technology for new turbines, a wide range of design factors and, consequently, potential air supply arrangements exist. The environmental performance and hydraulic performance of a given aeration alternative are sensitive to the detailed layout of the turbine components, including the size, shape, and orientation of the airflow passageways, scrollcase, runner, and draft tube. The basic requirements for these components vary from project to project depending on the head, flow, and other local conditions. The design of an aeration alternative tends to be site-specific – a single arrangement with fixed dimensions will not apply at all projects. However, within the geometrical limits of most installations, enough flexibility usually exists in the size, shape, and orientation of the turbine components to find workable arrangements for aeration alternatives. The complex aspects of the flow through turbines and airflow passageways underscore the importance of a detailed hydraulic analysis to obtain functional aeration solutions.

The range of applicability of turbine aeration in new units can be demonstrated in terms of TVA's Hydro Modernization (HMOD) Program. Sixteen of the twenty-four projects in the HMOD program presently experience problems with seasonally low dissolved oxygen. Of these, current plans call for the use of twenty-seven new aerating turbines at thirteen of the projects. A summary is given in Table 5.5-3. Except for Tims Ford, all of the projects contain Francis turbines. Tims Ford is a diagonal flow unit and includes a forced air system. Overall, turbine aeration is expected to supply roughly 75 percent of the total required median DO uptake at these sites, including 100 percent at six of the projects.

Project	Turbines			Target DO in Reservoir Release	Median DO Improvement Required	DO Improvement by Turbine Aeration	
	No. Units	Head (m)	Power (MW)	(mg/L)	(mg/L)	(mg/L)	% of Median DO Required
Apalachia	2	110	40	6.0	0.8	2.0	100%
Blue Ridge 1	1	45	22	6.0	2.6	3.0	100%
Blue Ridge 2 (1)	1	45	1.5				
Boone	3	27	26	4.0	0.0	2.0	100%
Chatuge	1	30	10	4.0	2.9	1.0	34%
Cherokee	4	30	31	4.0	3.8	2.5	65%
Douglas 1&3	2	30	31	4.0	3.3	2.0	60%
Douglas 2&4	2	24	26				
Fontana	3	100	68	6.0	1.5	2.5	100%
Hiwassee 1	1	58	60	6.0	2.1	1.0	47%
Norris (2)	2	58	65	6.0	5.3	5.5	100%
Nottely	1	38	16	4.0	2.9	1.0	34%
South Holston	1	55	36	6.0	4.2	2.0	47%
Tims Ford (3)	1	41	41	6.0	5.6	4.0	71%
Watauga	2	66	26	6.0	2.0	2.0	100%

Notes: (1) New small turbine for minimum flow requirements.
(2) Upgrade with new turbines complete.
(3) Upgrade with new diagonal flow turbine complete. Aeration provided by forced air system.

Table 5.5-3 TVA Plans for New Auto-Venting Turbines

Advanced Technology

Experience at TVA's Norris Dam has demonstrated the feasibility of integrating objectives for DO improvement in the design of new turbines. At projects containing low dissolved oxygen where upgrades or new construction of Francis units are planned, turbine specifications should include performance requirements for the combined environmental and hydraulic performance of the units.

Despite progress made in recent years, continued work is needed for the development of advanced aeration technology for new turbines. As emphasized for existing units, work to improve evaluation tools, validation data, and airflow passageways also are needed for new aerating turbines. Enhanced methods to supply and entrain air need to be examined to expand the range of applicability in new turbines and to reduce costs. New options should consider aeration at projects where the total dissolved gas or dissolved nitrogen in the may be too high, as well as the unique aeration problems of propeller-type units. Other needs for continued work related to improvements in analysis, specification requirements, testing, and operational support are provided in the ensuing sections of this report.

5.5.2 ANALYSIS OF AERATING TURBINES

The analysis of aerating turbines focuses on the estimation of three parameters – airflow, DO uptake, and performance effects (Greenplate and Cybularz, 1993). In general, airflow is the additional fluid parameter that distinguishes aerating turbines from conventional turbines. If Q_A and Q_W are, respectively, the air and water flow through a turbine, the ratio Q_A/Q_W emerges from a dimensional analysis as the additional parameter that must be considered in the dynamic similitude of the unit. Most studies express the air and water contributions in terms of the mean void fraction (or mean air concentration), given by

$$\phi = \frac{Q_A}{Q_A + Q_W}. \tag{5.5-1}$$

In the analysis of aerating turbines, DO uptake and performance effects typically are expressed in terms of ϕ.

Prediction of airflow requires a balance between the pressure loss for flow through the turbine air supply passageways and the pressure at the aeration outlets based on flow through the runner and draft tube. The latter involves the turbulent, two-phase flow of an air/water mixture and cannot be reliably evaluated using current state-of-the-art computational tools. Due to these complexities, the prediction of airflow at this time is based primarily on dimensionless parameters (e.g., pressure coefficients) derived from testing model and prototype turbines.

The focus of work summarized herein is for DO uptake and performance effects. The DO uptake provides a measure of the environmental performance of an aerating turbine. In some cases, other water quality parameters also may contribute to the turbine environmental performance, such as total dissolved gas in the tailwater. Dissolved oxygen uptake is given by

$$\Delta DO = DO_{TW} - DO_{SC}, \tag{5.5-2}$$

where DO_{SC} and DO_{TW} are the dissolved oxygen concentrations in the scrollcase and tailwater, respectively. Depending on the site-specific spatial distribution of flow, the location of dissolved gas samples taken from the scrollcase and tailwater can have a significant impact on the measured ΔDO (see Section 5.5.3). To be complete, the values of DO_{TW} and DO_{SC} in Eq. 5.5.-2 should be defined as mean values based on the total flux of dissolved oxygen upstream (above scrollcase inlet) and downstream (below tailwater bubble zone) of the turbine.

Performance effects refer to the impact of aeration-related changes in the turbine on the efficiency of the unit. This impact, called the hydraulic performance of an aerating turbine, is measured by the aeration-induced efficiency loss, given by

$$\Delta\eta = \eta_o - \eta_a, \tag{5.5-3}$$

where η_a and η_o are the turbine efficiency with and without aeration, respectively. Discussions of methods to evaluate the environmental and hydraulic performance of aerating turbines follow.

5.5.2.1 **Environmental Performance**

The fundamental relationship for estimating the DO uptake in turbine aeration is the familiar first order differential equation of oxygen transfer across the surfaces of bubbles dispersed in a flow, given by

$$\frac{dC}{dt} = K_L a(C - C_S). \tag{5.5-4}$$

In Eq. 5.5-4 C is the DO concentration, K_L is the liquid film mass transfer coefficient, a is the interfacial bubble surface area density, and the subscript s refers to saturation conditions. Equation 5.5-4 describes the gas transfer occurring between a parcel of water passing through a draft tube and the bubbles encountered along its journey. In general, the simplicity of this equation masks the complexity of oxygen transfer in aerating turbines. The liquid film coefficient is a spatially varying parameter that describes the effect of local turbulence on the transfer of oxygen across the bubble surfaces. To account for the breakup and coalescence of bubbles in the flow, the interfacial density also is spatially dependent. As written, use of Eq. 5.5-4 requires detailed knowledge of the distributions of water and bubble velocities, bubble sizes, temperature, and pressure throughout the draft tube.

Because the detailed characteristics of the flow are difficult to obtain, integration of Eq. 5.5-4 over the ensemble of water parcel trajectories in the draft tube is seldom practical. At this time, the only efforts to account for local effects are based on one-dimensional analyses that use the average pressure and velocity along the center streamline of the flow (Wilhelms et al., 1987; Buck et al., 1980). In this model, Eq. 5.5-4 is integrated in combination with the pressure-time history of the flow to account for local variation in saturation concentration. Also, the liquid film coefficient is assumed to be proportional to the air/water ratio, with the constant of proportionality occurring as an empirical coefficient to be determined from field data. In applications to the U.S. Army Corps of Engineers J.S. Thurmond Project (formerly Clarks Hill), the standard error in the predicted tailwater DO is of magnitude 0.5 mg/L whereas the maximum error is of magnitude 1.0 mg/L. In practice, errors of this size can have significant impact on the cost of meeting environmental limits, hence the need to improve these models is obvious.

Other efforts to predict the environmental performance of aerating turbines have considered the problem of scaling oxygen transfer using test data from existing geometrically similar prototypes or models. For models, the classical problem of how to properly account for scale effects in the behavior of air/water mixtures arises. The relative characteristics of air and water are not easily controlled in the model or prototype. Not all of the pertinent forces occur in the same fraction in the model and the prototype. Although geometric similarity is achieved (except perhaps in the tailwater), differences in the dominant forces will degrade dynamic and kinematic similarity. Despite these difficulties, scaling relationships are often the tools of choice for predicting environmental performance of aerating turbines. As discussed below, the primary focus in this approach is in how to properly consider the liquid film coefficient and the interfacial surface area density.

Mass Transfer Scaling

Liquid Film Mass Transfer Coefficient

Schroeder (1977) summarizes surface renewal models of mass transfer across an air and water interface. The liquid film mass transfer coefficient in such models is scaled by the relationship

$$K_L \propto (D\,r)^{1/2}, \tag{5.5-5}$$

in which r is the surface renewal rate, or the frequency of arrival of turbulent eddies at the interface (the bubble surface). The molecular diffusivity, D, is a property of the oxygen and water mixture and depends primarily upon the local pressure and temperature.

Bubble Interfacial Density

If the bubbles in an aerated flow are assumed spherical and possessed of a single common diameter, d_b, one may show that the interfacial density scales according to the relationship

$$a \propto \frac{1}{d_b}\frac{\phi}{1-\phi}. \tag{5.5-6}$$

Gulliver et al. (1990) determined from analysis of self-aerated flows an alternative relationship given by

$$a \propto \frac{1}{d_b} \phi . \qquad (5.5\text{-}7)$$

In aerating turbines, for which air void fractions are low (typically $\phi < 0.06$), the difference between these two relationships is negligible. Scaling relationships for the bubble diameter usually employ the energy balance between turbulent shear and bubble surface tension forces, expressed by Hinze (1955) as

$$d_m \propto \left(\frac{\sigma}{\rho}\right)^{3/5} \varepsilon^{-2/5} , \qquad (5.5\text{-}8)$$

where d_m is the bubble diameter for which 95% of the air is contained in bubbles of size d_m or smaller, σ and ρ are the surface tension and density for water, respectively, and ε is the rate of mixture energy dissipation per unit mass. A typical assumption is that d_b is proportional to d_m, with the constant of proportionality dependent on the nature of the aerated flow (e.g., open channel versus internal flows). With this assumption, d_m may be replaced by d_b in the above equation.

Scaling of the K_La *Product*

Using the above yields the scaling relationship for K_La, given by

$$K_La \propto (Dr)^{1/2} \left(\frac{\rho}{\sigma}\right)^{3/5} \varepsilon^{2/5} \phi . \qquad (5.5\text{-}9)$$

Further development of Eq. 5.5-9 requires scaling relationships for the surface renewal rate, r, and the dissipation rate, ε. Alternatives for these relationships are discussed below. It is typical to assume that both r and ε are related to the "integral" velocity and length scales, u_t and L_t, respectively, of the turbulent flow. The appropriate relationships are

$$r \propto u_t / L_t \qquad (5.5\text{-}10)$$

and

$$\varepsilon \propto \frac{u_t^3}{L_t} . \qquad (5.5\text{-}11)$$

Tennekes and Lumley (1972) provide a discussion of integral velocity and length scales in turbulent flows.

Aeration Scaling Relationship of Thompson and Gulliver

Thompson and Gulliver (1997) develop a scaling relationship for the liquid film mass transfer coefficient by extending work originally presented by Azbel (1981). Azbel combined an equation for the rate of interfacial gas mass transfer (moles/area/time) in a dispersed bubbly flow developed Levich (1962),

$$\frac{dm}{dt} = 2\left(\frac{\pi D v_r}{d_b}\right)^{1/2} d_b^2 (C_S - C) , \qquad (5.5\text{-}12)$$

with a relationship for the relative velocity of air in a bubble swarm, given by

$$v_r \propto \frac{u_t^{3/2} d_b}{(\nu L_t)^{1/2}} F(\phi) , \qquad (5.5\text{-}13)$$

where ν is the kinematic viscosity of water and $F(\phi)$ is defined by

$$F(\phi)=\frac{(1-\phi)}{\left(1-\phi^{5/3}\right)^{1/2}}. \tag{5.5-14}$$

Thompson and Gulliver compare Eqs. 5.5-12 and 5.5-13 with Eq. 5.5-4 to deduce a scaling relationship for the liquid film mass transfer coefficient. One may use the comparison to obtain an equivalent scaling relationship for the surface renewal rate,

$$r\propto\frac{u_t}{L_t}Re_t^{\beta-1}, \tag{5.5-15}$$

where $Re_t=u_t L_t/\nu$ is a turbulent Reynolds number. The ϕ dependence in Eq. 5.5-14 has been ignored, since $F(\phi)\approx 1$ for small ϕ. Eq. 5.5-15 can be compared to Eq. 5.5-10, in which Re_t does not appear. Thompson and Gulliver suggest an augmentation of the turbulent Reynolds number exponent (β in Eq. 5.5-15) to account for differences in the motion of bubble swarms in a draft tube relative to the buoyancy-driven turbulent motion of bubble swarms in an otherwise quiescent fluid. A value of $\beta = 1.5$ is theorized for the latter case. The value of β for turbine draft tubes must be determined empirically.

Substitution of Eqs. 5.5-11 and 5.5-15 into Eq. 5.5-9 yields a scaling relationship for K_La given by

$$Tr\propto We_t^{3/5}Pe_t^{-1/2}Re_t^{(\beta-1)/2}\frac{d_r}{L_t}\phi, \tag{5.5-16}$$

in which $Tr\equiv K_La/(u_t/d_r)$ is a non-dimensional mass transfer rate, $We_t\equiv\rho u_t^2L_t/\sigma$ is the turbulent Weber number, $Pe_t\equiv u_tL_t/D$ is the turbulent Peclet number, and d_r is the turbine runner diameter. To enable application of the scaling relationship to actual aerated flows through model and prototype turbines, Thompson and Gulliver relate the velocity and length scales of the turbulent flow to the velocity and length scales of the turbine runner. Stated explicitly,

$$u_t\propto Nd_r \text{ and } L_t\propto d_r, \tag{5.5-17}$$

where N is the rotational speed of the runner. According to Eq. 5.5-17, the runner diameter and rotational speed determine completely the structure of the turbulent flow field, the evolution of gas bubble sizes within the draft tube, and the gas transfer across the surface of the bubbles.

An Aeration Scaling Relationship Incorporating Draft Tube Losses

Draft tube losses can be incorporated into a scaling relationship for the K_La by relating the integral length scale, velocity scale, and dissipation rate to the draft tube loss coefficient or Darcy-Weisbach friction factor f. Jun and Jain (1993) express this relationship by

$$\frac{L_t}{d_r}\propto f^{-1/4}Re^{-3/4}, \tag{5.5-18}$$

where $Re = Ud_r/\nu$ is the bulk flow Reynolds number for the draft tube and U is the average water velocity through the draft tube. Falvey (1980) suggests modeling turbulent dissipation as $\varepsilon = gS_fU$, where g is the acceleration of gravity and S_f is the slope of the hydraulic grade line. Use of the Darcy-Weisbach equation to estimate S_f gives the result

$$\varepsilon=f\frac{U^3}{2d_r}. \tag{5.5-19}$$

The turbulent velocity scale may be related to the friction factor as follows:

$$u_t \propto u_\tau = \left(\frac{f}{8}\right)^{1/2} U, \qquad (5.5\text{-}20)$$

in which u_τ is the shear velocity of the draft tube flow. Substitution of Eqs. 5.5-18 through 5.5-20 into Eqs. 5.5-9 and 5.5-10 gives

$$Tr \propto We_t^{3/5} Pe_t^{-1/2} \left(\frac{d_r}{L_t}\right)^{3/5} f^{3/5} \phi, \qquad (5.5\text{-}21)$$

which may be compared to Eq. 5.5-16.

Analysis of Turbine Aeration Data
TVA has employed both of the similitude relationships (i.e., Eqs. 5.5-16 and 5.5-21) to investigate the aeration performance of hub baffle turbines at its Cherokee, Fontana, and Norris Dam sites. Because draft tube energy loss data is not available for the Fontana and Cherokee sites, the second procedure was not applied to those sites.

Figure 5.5-9 shows aeration data from the aforementioned projects scaled according to the procedure of Thompson and Gulliver (1997), Eq. 5.5-16. Values of $K_L a$ are computed from a linearized, integrated form of Eq. 5.5-4,

$$K_L a = t \, ln\left(\frac{C_S - C_{SC}}{C_S - C_{TW}}\right), \qquad (5.5\text{-}22)$$

where C_{SC} and C_{TW} are the dissolved oxygen concentrations measured in the scrollcase and tailrace, respectively. The draft tube transit time is computed as $t = 1/N$ (Thompson and Gulliver, 1997). The value of $\beta = 1.1$, determined by Thompson and Gulliver in their analysis of the Norris model and prototype data, is used also for the Fontana and Cherokee data.

Figure 5.5-10 shows aeration data for the Norris model and prototype, scaled using the draft tube loss procedure, Eq. 5.5-21. Draft tube transit times are computed by integrating the mean velocity of the air/water mixture (via the one-dimensional energy equation) along the draft tube centerline.

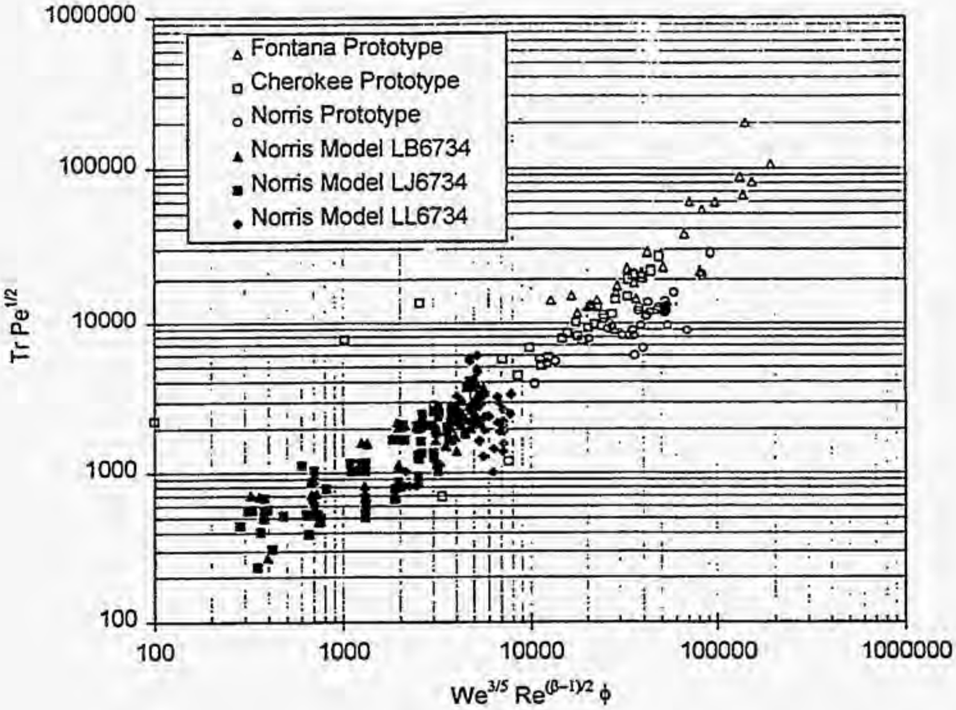


Figure 5.5-9 Analysis of Hub Baffle Aeration Data from TVA's Norris, Fontana, and Cherokee Dams Using the Procedure of Thompson and Gulliver (1997)

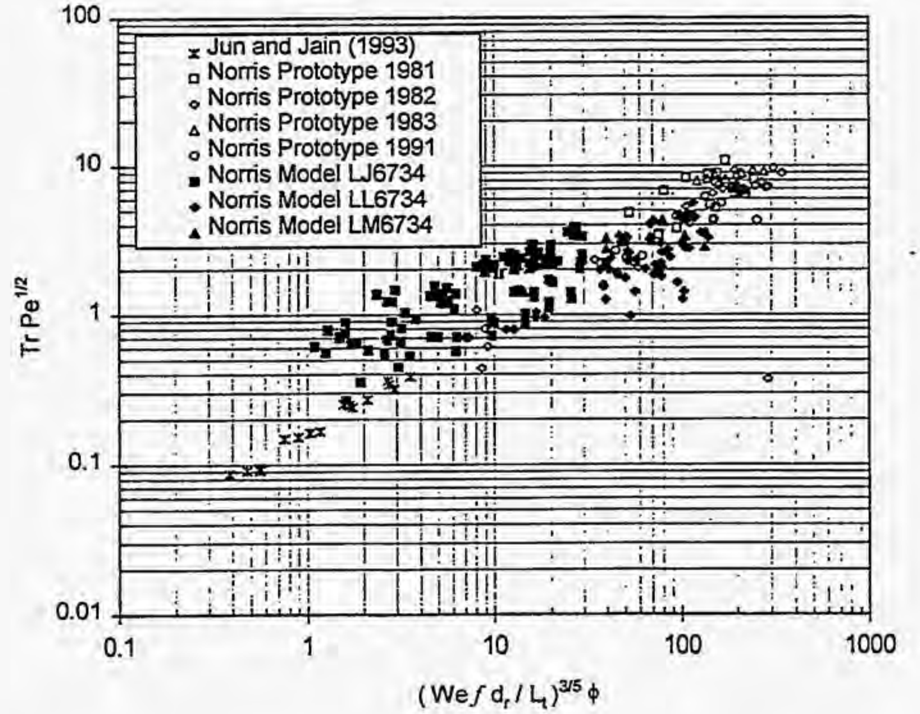


Figure 5.5-10 Analysis of Hub Baffle Aeration Data from Laboratory Experiments and TVA's Norris Dam Using the Draft Tube Loss Procedure

Although the data are scattered widely in Figures 5.5-9 and 5.5-10, the linearity of the data suggests a power-law relationship for Eqs 5.5-16 and 5.5-21. Both procedures appear to account for the essential mechanisms of aeration. In particular,

- Increased turbulence in the draft tube promotes more frequent surface renewal, increased bubble splitting, and higher interfacial density, with a resulting increase in oxygen transfer.
- Increasing the air/water ratio increases the interfacial density, with a resulting increase in oxygen transfer.

In general, mechanisms that increase turbulence and airflow are undesirable because they tend to degrade hydraulic performance. Thus, one must balance the requirements for oxygen transfer against hydraulic efficiency loss. For this reason, there is need to minimize the uncertainty present in the data, and in the empirical relations embodied by Figures 5.5-9 and 5.5-10, if one is to use them in conjunction with hydraulic efficiency relations for economic analysis and design of aerating turbines. Possible causes of the uncertainty exhibited by the data include:

- *Additional site and operating parameters.* Parameters not included in the scaling procedures (for example, net head or net positive suction head), which affect the hydraulic performance of turbines, may cause systematic deviations from relationships expressed by Equations 5.5-16 and 5.5-21. It is unlikely that turbulent flows produced by adverse or off-peak operating conditions are adequately scaled by turbine runner geometry or a single-valued draft tube loss parameter. As Thompson and Gulliver acknowledge, there is a need for further research of these issues.
- *Oxygen transfer in the tailrace.* The aeration scaling procedures consider only oxygen transfer occurring in the draft tube. Because measurement of the dissolved oxygen concentration at the draft tube exit is difficult and unreliable, concentrations at a point downstream in the tailrace are used to compute transfer. Thus, tailrace hydrodynamics, which vary among prototype sites and model test facilities, influence the aeration data in a way that neither of the scaling procedures considers.
- *Measurement uncertainty.* Uncertainties in the measurement of dissolved oxygen contribute directly to the uncertainty of the liquid film mass transfer coefficient. TVA's experience with aerating turbine testing has shown that standardization of dissolved oxygen meters is difficult, but not impossible, to realize in the field. Measurement of air flowrates into the turbine and draft tube requires the use of nozzle and Venturi meters in less than ideal settings, which differ from site to site and from model to prototype. Uncertainties of measurements of air flowrates contribute directly to the uncertainties of calculated air void fraction values. Figure 5.5-10 also shows the data of Jun and Jain (1993), collected in a downward aerated flow in a pipe. This data is included here to illustrate the measurement quality attainable under ideal (laboratory) conditions. Development and application of useful turbine aeration similitude theory will depend upon standardization of testing procedures for aerating turbines. (These issues are discussed in more detail in Section 5.5.3).
- *Evaluation tools.* As has been emphasized in previous discussions, advanced numerical models using CFD techniques need to be considered to evaluate and scale oxygen transfer in aerating turbines. Such models have already been employed to compute the detailed spatial characteristics for the single phase flow of water in draft tubes (e.g., see Ventikos et al., 1996). These models need to be expanded to include physics for the two-phase flow of an air/water mixture and gas transfer. To account

for the entire domain of air/water flow, numerical models need to include both the draft tube and tailwater in a coupled fashion.

5.5.2.2 Hydraulic Performance

The design objective for hydraulic performance of conventional hydroturbines is to maximize efficiency over as wide a range of the expected operating conditions as possible. Without aeration, manufacturers currently can produce turbines with peak efficiencies near 95 percent. The design of aerating units poses a new challenge – that of providing highly efficient turbines which produce as much dissolved oxygen as possible while minimizing aeration-induced efficiency losses. A key issue in analyzing the hydraulic performance of aerating turbines is understanding the extra mechanisms of energy loss produced by the two-phase, air/water flow created by this process. Another related issue is how to predict the resulting aeration-induced loss. Brief discussions of these items follow.

Energy Loss Mechanisms
According to Almquist et al. (1991), the energy loss associated with turbine aeration may be due to either reduced runner efficiency or reduced draft tube efficiency. Due to the complexity of the flow, the potential for reduced runner efficiency is difficult to assess. In general, however, most aeration alternatives introduce air at locations near the exit of the turbine. It seems unlikely that this practice, which typically provides air in amounts ranging as high as four to six percent by volume, would affect characteristics of flow in the runner to an extent sufficient to account for observed efficiency losses of two to four percent. A more likely explanation is that performance degradation occurs in the turbine component directly affected by the presence of air (i.e., the draft tube). According to Mosonyi (1987), draft tubes typically account for between ten and fifty percent of the total energy available to a hydroturbine, depending on the operating head.

Major mechanisms by which the admission of air at the exit of the runner may lower draft tube efficiency include the following.

- Increased hydraulic losses (e.g., expansion, bend, friction, and exit losses). Assuming the air/water mixture behaves as a homogenous fluid, analyses show that the average velocity of flow in the draft tube due to air admission will increase by a factor of $1+\phi$. Subsequently, the increase in draft tube head loss ΔH_{hl} can be scaled by

  $$\Delta H_{hl} \propto \phi k_{hl} \frac{U_{DT}^2}{2g}, \tag{5.5-23}$$

  where k_{hl} is the loss coefficient for the draft tube hydraulic losses, U_{DT} is the average velocity at the entrance of the draft tube, g is the gravitational acceleration, and ϕ is as previously defined.

- Air Transport losses. These losses are incurred by the expenditure of energy to transport entrained air against the hydrostatic gradient in the vertical portion of the draft tube. In this case, the resulting draft tube transport loss ΔH_{tl} can be scaled by

  $$\Delta H_{tl} \propto \phi h_v, \tag{5.5-24}$$

  where h_v is the vertical distance between the air admission location and the minimum centerline elevation of the draft tube. Note that the energy to transport air in this manner is assumed to be non-conservative. That is, the energy imparted to the air, which is manifested as high pressure at the low point of the draft tube, does not impart additional potential or kinetic energy to the flow as the bubbles emerge from the draft tube and rise to the water surface in the tailrace.

- Bubble losses. In general, due to the difference in density and fluid immiscibility, slippage will occur between entrained air bubbles and water. Energy dissipation subsequently occurs due to non-conservative forces arising from drag on the bubbles traveling through the draft tube. Based on the flow around entrained spherical particles, the resulting draft tube bubble loss can be scaled by

$$\Delta H_{bll} \propto \phi \frac{V_{DT}}{Q_W} \left(\frac{g d_b}{C_d} \right)^{1/2} , \qquad (5.5\text{-}25)$$

where V_{DT} is the volume of the draft tube, C_d is a bubble drag coefficient, and Q_W and d_b are as previously defined.

Using field data for a medium-head Francis turbine, Almquist et al. (1991) compared the relative magnitude of each of the above mechanisms. The results, given in Table 5.5-4, were obtained from data in a test where the actual measured efficiency loss of the turbine was about 0.56 percentage points. In this case, the three mechanisms account for about 85 percent of the total observed efficiency loss. Although many simplifying assumptions are made, the analysis provides reasonable agreement between the total computed and measured losses.

Mechanism	Efficiency Loss (percentage points)
Hydraulic Losses	0.04
Air Transport Losses	0.30
Bubble Losses	0.14
Total	0.48

Table 5.5-4 Efficiency Losses for Turbine Aeration

Before it can be concluded that the mechanisms given by Eqs. 5.5-23 to 5.5-25 encompass all the major sources of aeration-induced energy loss for an aerating turbine, analyses need to be performed using additional data from a broad range of turbine design and operation conditions, including information at both model and prototype scales. Refinement of scaling relationships is needed to evaluate applicability to other sites and to account for the remaining 15 percent of energy loss shown in Table 5.5-4. Further analyses may define other mechanisms for losses and/or improved methods of scaling. The greatest promise for obtaining a reliable procedure for analyzing and scaling the hydraulic performance of an aerating turbine lies in the development of advanced numerical models. Formulations currently are available for predicting the characteristics of two-phase air/water flow, including mechanism accounting for the additional dissipation of energy resulting from the increased flux, buoyancy, and relative velocity of the air phase.

Predicting Efficiency Loss

In the absence of model-prototype confirmation of the above energy loss mechanisms, the prediction of performance effects for prototype turbines has been limited to an examination of the relative change in turbine efficiency derived from model tests conducted over a range of operating conditions, with and without air admitted at the various aeration locations (Greenplate and Cybularz, 1993). In particular, for the aeration-induced change in turbine efficiency, Cybularz et al. (1992) found $\Delta\eta$ in the prototype to be roughly the same as that in the model, if the model is operated at a speed which yields the same relative change in pressure through the unit. Based on the theory of homologous units, this requirement leads to Froude scaling for the discharge in the model turbine (Mobley and Brice, 1991). Results of prototype tests at Norris show that this procedure tends to overpredict the amount of aeration-induced efficiency loss. Although this leads to a conservative estimate of the hydraulic performance (i.e., the actual hydraulic performance will not be as severe), improved procedures are needed. This is true not only for assessing the quality of the turbine design, but also for effectively evaluating the cost of turbine aeration compared to that of other methods for enhancing dissolved oxygen.

5.5.2.3 **Economic Considerations**

In comparing options for increasing the dissolved oxygen for a hydro project, the cost of different oxygenation methods, such as those in Table 5.5-1, must be evaluated. The optimal method will be that which reliably provides the desired DO uptake at the lowest cost. Depending on the DO requirements and operational limits of the different alternatives, two or more oxygenation methods may be required.

Likewise, for turbine aeration, the proper selection of one or more aeration alternatives will depend on the cost of each. Optimal designs, both in new and retrofit situations, will be those providing the lowest cost per mg/L of DO uptake. In general, costs include capital and operation and maintenance (O&M) expenses. Capital costs include that for the design, fabrication, installation, and testing of the aeration-related features of the turbine. Operation and maintenance costs are those for the routine preventive maintenance of the aeration-related equipment and the aeration-induced efficiency loss of the turbine. Preventive maintenance includes that for valves, piping, controls, instrumentation, and blowers or compressors (for a forced injection system). In turbines where aeration-related modifications cause significant permanent efficiency losses, these should also be included.

Operation costs due to the aeration-induced efficiency loss should be determined based on the seasonal variation of required DO uptake, the aeration performance and operating procedures for the aeration alternative, and expected generation schedule. Note that for turbines with two or more alternatives (e.g., see Figures 5.5-5 and 5.5-6), the aeration performance and operating procedures may include that for combined operation of the alternatives. Depending on DO requirements and seasonal flow conditions, aeration may not be required in all turbines for a project containing multiple units.

5.5.3 TESTING OF AERATING TURBINES

Performance testing, both at model and prototype scales, will play an important role in the development and commercial acceptance of advanced aerating turbine technology. Owners already have begun to issue specifications containing requirements for environmental and hydraulic performance of aerating turbines. This especially is true for new equipment. Similar requirements undoubtedly will emerge for those involved in upgrading exiting units with retrofit options for aeration.

In response to these trends, there is an obvious need for the development of formal test procedures for aerating turbines to evaluate conformance to contractual guarantees and, perhaps, to assess bonuses or penalties. In addition, to provide meaningful guarantees, suppliers also need a thorough understanding of how to analyze and predict the environmental and hydraulic performance of aerating turbines (i.e., DO uptake and aeration-induced turbine efficiency loss). The development of dependable analysis and prediction techniques, in turn, requires valid test data. It is also important to obtain data on a regular basis to monitor and update daily operations of aerating turbines. Updates are required, in general, because performance of an aerating turbine, both environmental and hydraulic, varies with changes in operating conditions (e.g., turbine head, tailwater elevation, power output, incoming DO concentration, etc.).

The following discussions are given to summarize the major issues involved in testing aerating turbines. Comments are provided regarding specification requirements and parameters required in testing. Recommendations focus on establishing a comprehensive test code for aerating turbines.

5.5.3.1 **Specification Requirements**

The major test codes used to define specification requirements for hydroturbines are PTC-18 (ASME, 1992) and IEC 41 (IEC, 1991). Because these codes address performance only of conventional non-aerating units, specification requirements for existing retrofit and new turbines containing arrangements to aerate hydro releases are not well defined. Until appropriate test codes emerge, the following general guidelines are given to help formulate specification requirements:

- Specifications for aerating turbines should include target requirements for DO improvement to be met by the aeration options. Depending on the aquatic community in the tailrace, specifications for aerating turbines also may include upper limits for total dissolved gas (TDG). Levels of TDG above about 110 percent can cause "gas bubble disease," which is fatal in sensitive species (see EPA, 1986). Depending on site-specific conditions, other water quality parameters that are influenced by turbine operation also could be included (e.g., temperature). The requirements for DO, TDG, and other water quality parameters comprise the guarantees for the aerating turbine's environmental performance. The conditions under which environmental guarantees are to be satisfied should be identified (e.g., head, gate, tailwater elevation, incoming DO concentration, flow).
- Specifications for aerating turbines should include maximum acceptable levels for the aeration-induced efficiency loss, $\Delta\eta$. These requirements comprise the guarantees for the aerating turbine's hydraulic performance. The conditions under which $\Delta\eta$ guarantees are to be satisfied should be identified (e.g., head, gate, tailwater elevation, incoming DO concentration, DO uptake, flow).
- Specifications for aerating turbines should place upon the manufacturer the burden of proposing the detailed design arrangements by which their turbine will supply the target DO and TDG guarantees (i.e., the environmental performance guarantees). The manufacturer should clearly identify the technology to be used, the conditions

under which the aeration guarantees will be met, and the accompanying maximum aeration-induced efficiency loss (i.e., the hydraulic performance guarantees).

- Because there is not yet an industry-approved test code for evaluating the performance of aerating turbines, the exact methods by which DO, TDG, and $\Delta\eta$ will be measured and evaluated in the model and prototype units should perhaps be left open as negotiable during selection of the turbine contractor. Recommendations for a test code, provided below, should help in this process.
- Successful implementation of advanced aeration technology for hydroturbines requires expertise in site analysis, engineering, fabrication, installation, and testing. Without this, there is substantial risk of obtaining an aerating turbine that fails to meet target environmental and hydraulic guarantees. As such, specifications for aerating units should require the turbine contractor to demonstrate their ability to supply aerating units. This can be accomplished by requiring statements of aeration expertise, such as staff capabilities (e.g., resumes and publications) and a list of projects where the contractor has successfully implemented aerating turbine technology. The methods of analyses used to determine predicted environmental and hydraulic performance should be provided (i.e., assumptions, data, computations, and references). Bid specifications should include "bonus points" for contractors with aeration experience. The owner also could ensure that qualified and experienced personnel carefully evaluate the bids and monitor the design, fabrication, installation, and testing of the aerating turbines.

5.5.3.2 **Test Parameters**

Testing of aerating turbines can be broadly divided into two categories, aeration and non-aeration performance (Figure 5.5-11). Non-aeration testing is performed with all aeration systems turned off and is the same as that for the mechanical performance of conventional turbines. Key parameters include turbine efficiency, maximum power output, cavitation level, vibration, shaft runout, and thrust load. Test codes PTC-18 and IEC 41 apply and include procedures to measure discharge, head, and power output to calculate the turbine efficiency.

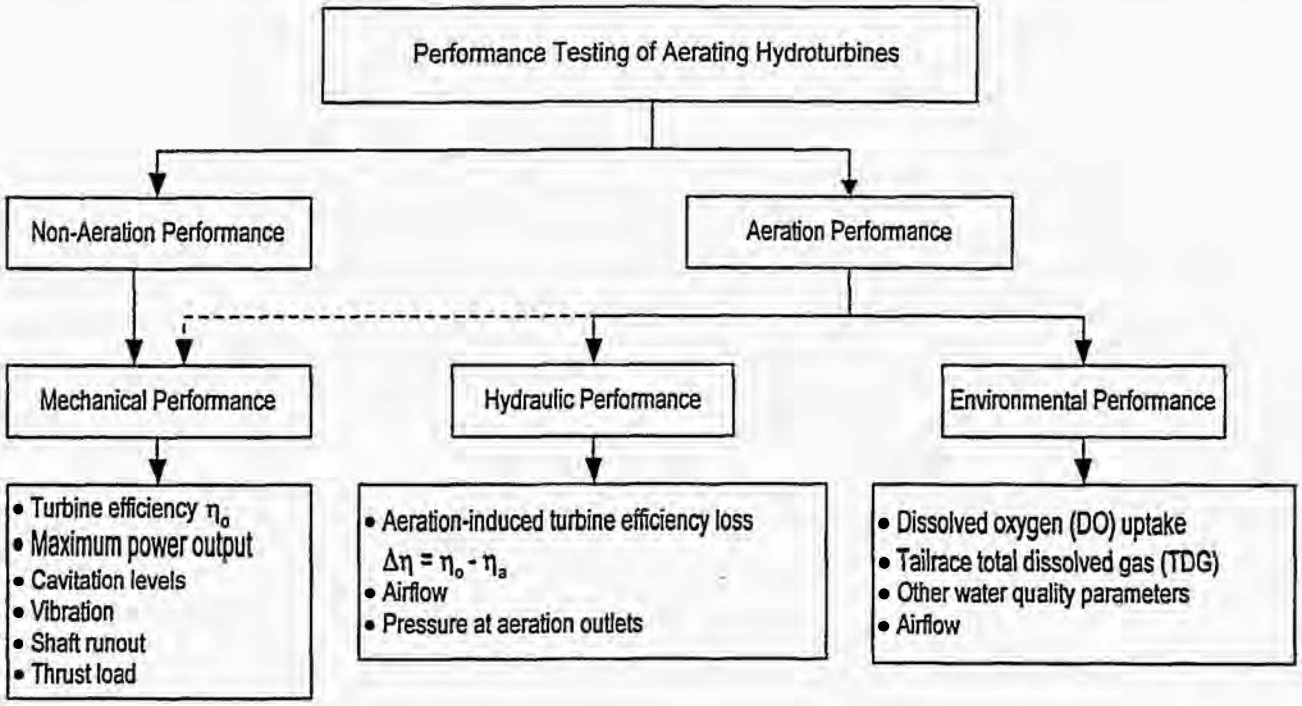


Figure 5.5-11 Flowchart for Testing Aerating Hydroturbines

Testing of an aerating unit focuses on evaluating the turbine hydraulic performance and turbine environmental performance. During aeration, hydraulic performance is measured by the aeration-induced efficiency loss $\Delta\eta = \eta_o - \eta_a$, where η_o is the turbine efficiency with all aeration systems turned off and η_a is the turbine efficiency with one or more aeration systems turned on. Both η_o and η_a are found using procedures of PTC-18 and IEC 41. Airflow and aeration outlet pressure are needed to verify hydraulic characteristics of individual aeration options, if desired. The environmental performance of an aerating turbine is measured by the dissolved oxygen uptake and level of total dissolved gas in the tailrace. Depending on site-specific conditions, other water quality parameters also may be required. Airflow is needed to verify gas transfer characteristics of individual aeration options, if desired. Because air entrainment can affect other mechanical aspects of turbine operation, measurements for cavitation and vibration should also be a part of aeration testing.

It is emphasized that test codes PTC-18 and IEC 41 provide detailed guidelines for both absolute performance testing and index testing. Because changes in performance rather than absolute performance are of primary interest, index testing is often adequate for many aeration performance tests.

In general, compared to testing of conventional units, much less knowledge and experience currently exist for testing aerating turbines. Changes in the concentration of dissolved oxygen of only a few tenths of a mg/L are significant from the standpoint of cost and meeting environmental commitments. However, measuring DO to this level of accuracy is difficult for several reasons. First, the spatial distribution of DO at the inlet and exit of the turbine can be highly nonuniform, often requiring a strategic deployment of multiple sensors. Secondly, DO sensors can drift significantly during the course of a test, requiring careful pre, post, and perhaps intermediate calibration experiments. Thirdly, DO levels in the tailrace are highly dependent on the distance downstream from the draft tube exit, requiring additional analyses to account for the effects of surface aeration and perhaps biological activity. TVA has gained substantial knowledge in addressing these issues. However, more study and experience are necessary to gain a better understanding of the achievable accuracy for DO, TDG, and other water quality measurements in testing aerating turbines.

Part of the difficulties in measuring dissolved oxygen could be overcome by improvements in sensor technology. Nearly all DO probes currently used for turbine testing rely on oxygen permeable membrane electrodes with verification by the Winkler method (see APHA, 1990). Despite many improvements, this forty-year-old technology remains troublesome. Common deficiencies include:

- The accuracy can be low--While the specified accuracy of most DO probes is 0.2 mg/L, field experience has shown that measurements at times are good only to 0.5 mg/L.
- The sensors drift out of calibration--Even under clean water conditions, sensors can drift out of calibration within a day.
- The sensors foul easily--Microscopic organisms, algae, and silt collect on the sensor membranes, causing inaccurate, readings.

With this type of DO sensor, it often is difficult to obtain good data, even in closely monitored performance tests. A greater impact perhaps occurs in the routine DO measurements used to establish operating conditions for hydropower oxygenation systems. The less-stringent DO probe maintenance requirements that typically accompany these measurements can easily result in data of poor quality. In many cases this leads to overuse of oxygenation equipment, thereby creating needless costs for operation and maintenance. Along with the development of advanced technology for turbine aeration, work also is needed for new technology and/or methods to measure and obtain accurate DO data.

5.5.3.3 **Test Code Recommendations**

A preliminary version for an aerating turbine test code is presented in Appendix 10.6. This document describes guidelines, procedures, measurements and instrumentation, and calculations required for an aerating turbine test. The purpose of the document is to provide guidelines for both performance and acceptance testing that will eventually lead to a comprehensive test code. This code is generally modeled after the American Society of Mechanical Engineers (ASME) Performance Test Code (PTC) series, although it is in no way sponsored or endorsed by ASME at this time.

5.5.4 OPERATION OF AERATING TURBINES

The operation of aerating turbines requires an understanding of basic requirements for water quality, monitoring and control of dissolved oxygen and aeration systems, and biological impact. The following includes brief discussions of each of these items.

5.5.4.1 **Water Quality Requirements**

Dissolved oxygen is usually the most critical water quality factor affecting survival and growth of fish and other aquatic organisms. In addition to damaging aquatic wildlife, low DO can contribute to problems related to the dissolution of trace metals, release of nutrients, formation of hydrogen sulfide, depression of pH, and low assimilative capacity. Other water quality parameters affecting aquatic activity which can be influenced by turbine operation include minimum flow, temperature, and total dissolved gas.

Many studies have been performed to investigate the impact of DO on fish mortality, growth rates, and survival. The results of these studies, which are summarized by EPRI (1990) and EPA (1986), provide the basis for establishing guidelines for DO in reservoirs, rivers, and streams. In general, guidelines include the effect of dissolved oxygen at five different levels of impairment for aquatic wildlife – no production impairment, slight production impairment, moderate production impairment, severe production impairment, and a limit to avoid acute mortality. These levels further have been determined for two life stages of fish – "early life stages" and "other life stages". The DO levels also depend on the type of aquatic wildlife in three broad categories – salmonid/cold water species (trout, salmon, whitefish), non-salmonid/warm water species (bass, bluegill, walleye), and invertebrates. The guidelines are presented in Table 5.5-5 through Table 5.5-7. For most projects, the low DO season typically occurs during "other life stages" where the guidelines for no production impairment varies between 6 and 8 mg/L. Most states have adopted minimum levels, however, that are aligned with the requirements for slight production impairment – 5 and 6 mg/L, respectively, for non-salmonid and salmonid waters.

Note that these DO levels should be considered subjective because they are based on short-term laboratory tests that do not include combined water quality effects as found in the natural environment. More field experience and testing is required to obtain a better understanding of conditions that are well-suited for aquatic life and achievable with turbine aeration technology.

Salmonid Waters (trout, salmon, whitefish)	
Life Stage & Impairment Level	DO Level (mg/L)
Early Life Stages	
No production impairment	11
Slight production impairment	9
Moderate production impairment	8
Severe production impairment	7
Limit to avoid acute mortality	6
Other Life Stages	
No production impairment	8
Slight production impairment	6
Moderate production impairment	5
Severe production impairment	4
Limit to avoid acute mortality	3

Table 5.5-5 DO Guidelines for Salmonid Waters (EPA, 1986)

Non-Salmonid Waters (bass, bluegill, walleye)	
Life Stage & Impairment Level	DO Level (mg/L)
Early Life Stages	
No production impairment	6.5
Slight production impairment	5.5
Moderate production impairment	5
Severe production impairment	4.5
Limit to avoid acute mortality	4
Other Life Stages	
No production impairment	6
Slight production impairment	5
Moderate production impairment	4
Severe production impairment	3.5
Limit to avoid acute mortality	3

Table 5.5-6 DO Guidelines for Non-Salmonid Waters (EPA, 1986)

Invertebrates	
Impairment Level	DO Level (mg/l)
No production impairment	8
Some production impairment	5
Acute mortality limit	4

Table 5.5-7 DO Guidelines for Invertebrates (EPA, 1986)

5.5.4.2 **Monitoring and Control**

To effectively operate air entrainment systems in an aerating turbine, monitoring is required. The primary parameter to be measured is the dissolved oxygen level in the turbine discharge. Effective control of the turbine will require monitoring of the DO uptake and efficiency (i.e., environmental and hydraulic performance). Depending on site-specific conditions, measurements also may be needed for discharge, temperature, total dissolved gas, or other water quality parameters.

For any parameters, the detailed monitoring arrangements will depend on several factors, including the frequency and magnitude of variations, available types of instrumentation, costs and O&M requirements for instrumentation, project location, and operating patterns for the turbine (e.g., base load, peaking, load regulation). For projects where the DO varies rapidly, continuous monitoring may be desired, while for those where DO varies gradually, weekly grab samples may be adequate. The placement and number of permanent sensors or location of grab samples ultimately depends on site-specific characteristics. Considerations can include placing DO sensors at a distance downstream of the plant to include tailrace aeration, placing sensors away from sluggish eddy zones, and placing sensors so they can be easily accessed for reading and maintenance.

Aerating turbines can be equipped with a variety of airflow alternatives (recall Figure 5.5-6), each possessing distinct operating characteristics with respect to DO uptake (i.e., environmental performance), aeration-induced efficiency loss (i.e., hydraulic performance), and O&M cost. For aerating turbines, O&M costs will include primarily that for efficiency loss. If the turbine is equipped with a forced injection system, the cost to operate air compressors or blowers also will be incurred. Other costs may stem from increased wear due to cavitation or vibration, if caused by an aeration alternative. However, based on TVA experience, aeration tends to reduce, not exacerbate, cavitation and vibration.

In the context of this report, control refers to decisions as to when to operate the turbine aeration alternatives, and if multiple alternatives are available, which should be operated. Other issues related to the specific equipment used to open and close aeration alternatives, including automatic controls, are not addressed herein.

Operation of turbine aeration alternatives should commence when the tailwater DO falls and approaches the target minimum level established for the project. Operating the turbine in the most cost-effective manner will involve determining the best alternative, or combination of alternatives, to meet the DO and power requirements. Other operating requirements may include maintaining minimum flows, temperature, and total dissolved gas. Because the environmental and hydraulic performance of a given alternative varies with the head and power, the options used to meet the desired DO can be strategically chosen to minimize the aeration-induced efficiency loss and other costs. As an example, consider the 1996 DO data for the new units at TVA's Norris Dam, shown in Figure 5.5-12. Turbine aeration was initiated in July when the scrollcase DO began to decrease. Throughout the low DO season, a mix of aeration options was used, based on the head, power output, and required DO uptake, to limit the turbine efficiency loss. Aeration was discontinued in November following reservoir turnover. On the average, the DO downstream of the project was maintained near the 6.0 mg/L target (except for a period when aeration was disrupted for an extreme series of performance tests on the new units). During the same period, the average aeration-induced turbine efficiency loss was about 1.9 percent.

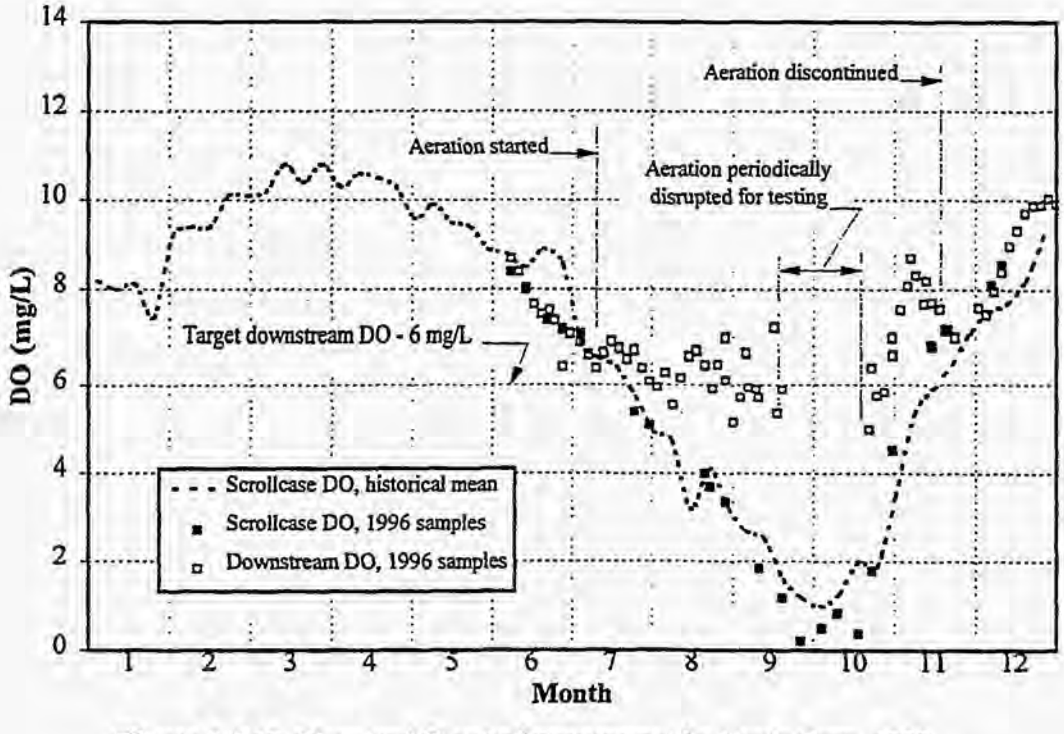
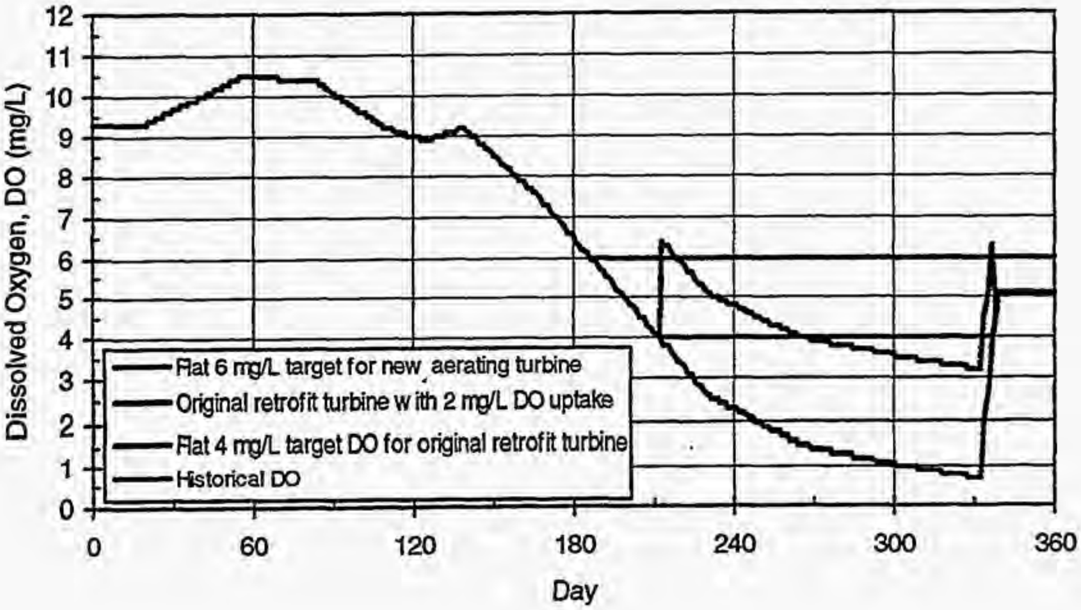
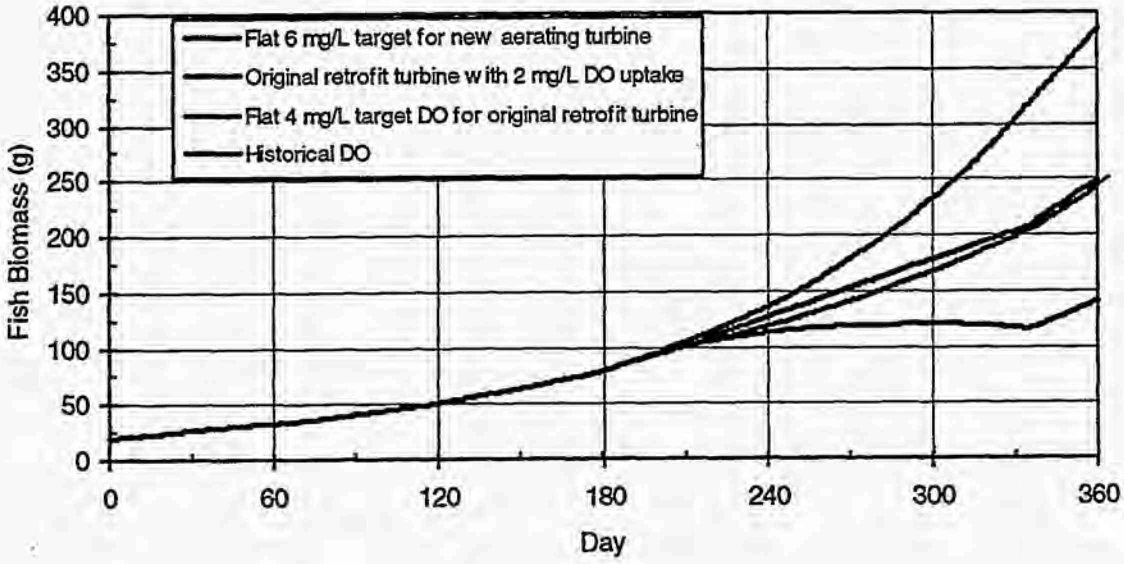


Figure 5.5-12 Dissolved Oxygen Improvement for Norris Dam, 1996.

[redacted]



Plot (a) Assumed Annual Variation of Tailwater Dissolved Oxygen



Plot (b) Computed Biomass for Rainbow Trout

Figure 5.5-13. Bioenergetics Model Simulation for Tailwater of Norris Dam

5.5.5 SUMMARY AND CONCLUSIONS

For a given hydropower facility, the amount of dissolved oxygen in the turbine discharges will vary depending on a variety of factors. Included are the design, environmental, watershed, and operational characteristics of the project. The likelihood of occurrence of low dissolved oxygen has been evaluated in several studies based on project geographic location, season, and size of turbines. Data collected in the tailwater at numerous sites confirm that dissolved oxygen enhancements are needed at many hydropower facilities.

Based on experience of the Tennessee Valley Authority, over seventy-five percent of the required DO enhancements in the United States could be achieved using turbine aeration. The greatest opportunities occur in situations involving new units (i.e., new construction or upgrades). In these cases, objectives for DO enhancement can be included as an integral part of the turbine design. Within the physical limits of most installations, the shape and orientation of the turbine components can often be selected to provide aeration. Based on knowledge obtained at TVA's Norris Dam, a wide range of design factors and, consequently, air supply alternatives can usually be found. The challenge in designing advanced hydroturbines is to determine the number and type of alternatives providing the most cost effective dissolved oxygen uptake. Since turbine and water quality characteristics vary from project to project, the optimal aeration arrangement for a given installation is site specific.

In considering aeration technologies for advanced hydroturbines, attention also must be given to existing units. This is because for most projects it is not economically feasible to install new turbines for the sole purpose of providing aeration. Although the number of design options are fewer, innovative retrofit arrangements usually can be used to provide turbine aeration in existing projects. The most common include a combination of hub baffles and bypass conduits.

The analysis of aerating hydroturbines focuses on the prediction of environmental and hydraulic performance. The first is measured by the dissolved oxygen uptake, the second is measured by the aeration-induced turbine efficiency loss. In general, procedures to perform these evaluations fall far behind the level of accuracy needed to obtain reliable aerating turbine designs. An extensive amount of study is needed to improve predictions. These should consider several factors, including additional site and turbine operating parameters, tailrace oxygen transfer, measurement uncertainty (primarily DO), and state-of-the-art CFD evaluation techniques.

An obvious need currently exists for test procedures for aerating turbines. Owners have begun to issue specifications containing requirements for environmental and hydraulic performance. Verification of this performance, as well as compliance with dissolved oxygen requirements imposed by regulatory agencies, will rely on well defined test procedures. The major test codes currently available, PTC-18 and IEC 41, address only the performance of conventional, non-aerating units. To facilitate development of procedures for aerating units, a preliminary test code is presented in this study. Until industry-accepted procedures emerge, specifications for aerating turbines should include requirements for DO uptake, aeration-induced efficiency loss, and statements of qualifications for providing this technology.

Dissolved oxygen is perhaps the most critical water quality parameter affecting the survival and growth of fish and other aquatic organisms. Higher levels of DO will promote fish growth in all life stages. Over a period of several years, the use of turbine aeration will create an aquatic community dependent on higher DO. As a result, it becomes increasingly important to provide aeration as the diversity and size of the aquatic habitat develops. To avoid fish kills or other biological damage, close attention must be given to the measured tailwater DO and reliable operation of aeration alternatives. Optimal operation will be achieved by the aeration alternative, or combination of alternatives, that meets the required DO uptake with minimum impact on power production.

5.5.6 REFERENCES

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6.0 TASK 4 REPORT -- PRESENTATION OF DESIGN CONCEPTS

6.0 TASK 4 REPORT – PRESENTATION OF DESIGN CONCEPTS

6.1 INTRODUCTION

Design practices of the past have not typically included specific fish friendly considerations, primarily as a consequence of economics. However this need not be the case, and when the need is present for design changes to provide for improved conditions for fish passage, alternative designs are possible. This section presents advanced design concepts that have potential for improving fish passage survival and environmental compatibility for axial flow and Francis type units based on discussions previously presented in Sections 4 and 5. They include mechanical design concepts, operational concepts, lubrication schemes, electrical concepts and control concepts. A description of turbine types is located in appendix 10.2.

While the thrust of this work has been to address the existing 92000 MW of installed turbine capacity, many of these advanced design concepts are as applicable to new units as they are for the rehabilitation of existing units. The differences between rehabilitation and new unit design considerations will be discussed in each section as applicable. More detailed analyses are required to determine the merit and cost tradeoffs of each concept. The fish friendly concepts deal primarily with direct injury to fish as they pass through the unit or with water quality. Only in a secondary manner do they address the potential problem of disorientation and associated predation. Due to the fact that the ratio between fish size and turbine size is significant for fish passage survival, there is some turbine size below which the best solution for fish passage survival may be to keep fish out of the unit.

Due to the fact that every project site has its own unique features, each site has a customized turbine design. Because of this, this deliverable is provided as a series of generalized design concept elements which will need to be assembled and customized to each specific turbine site. Based on the studies of Task 1 and the recommendations of the project review board, the three design concepts chosen for presentation are:

- ***An advanced environmentally friendly Kaplan turbine*** featuring a high efficiency level, cavitation free operation and the possibility for reduced backroll; a gapless design for wicket gates and for the hub and discharge ring with the blades; a hub filled with an environmentally compatible fluid; greaseless bushings for the wicket gates; an upgraded smooth surface for stay vanes, wicket gates and draft tube cone; an adjustable speed generator; and an advanced control system for speed adjustment, cam optimization and optimized energy generation with considerations for improved fish passage survival.

- ***An advanced environmentally friendly Francis turbine*** featuring a high efficiency level, cavitation free operation to powers beyond previous generations of turbines and the possibility for reduced backroll; a reduced number of blades; appropriate clearance between wicket gates and the runner blade entrance edge; greaseless wicket gate bushings; a gapless design for the wicket gates; an upgraded smooth surface for stay vanes, wicket gates and draft tube cone; an adjustable speed generator; and an advanced control system for speed adjustment and optimized energy generation with considerations for improved fish passage survival.

- ***An advanced environmentally friendly aerating Francis turbine*** with the above features and with design features for increasing the quantities of dissolved oxygen in the water discharging from the turbine. This Section will appear in a supplement to this report to be issued at a later date.

6.2 ENVIRONMENTALLY ADVANCED KAPLAN DESIGN FOR IMPROVED FISH SURVIVABILITY

6.2.1 PRIMARY ISSUES

6.2.1.1 **Runner Gaps**

A Kaplan runner has three to seven adjustable tilt blades attached to a central hub. The blade and hub assembly (runner) rotates inside a stationary discharge ring. Adjustable blade tilt allows the machine to maintain a relatively high efficiency over a wide range of operation. The ratio of hub diameter to runner diameter is a compromise of hydraulic and mechanical concerns. Hydraulically it is preferable for this ratio to be small. This creates lower velocities through the runner and improves overall performance. Mechanically it is preferable for this ratio to be large in order to accommodate the blade tilt adjusting mechanism.

The conventional hub and discharge ring are combinations of spherical, cylindrical and conical sections in the areas where they are in close proximity to the blades. This combination of surfaces is a compromise between hydraulic, mechanical and economic concerns. The advantages of cylindrical and conical sections on the hub relate to lower cost of manufacture and more space in which to house the blade tilt operating mechanism. Conventional Kaplans use cylindrical sections on the upper half of the discharge ring for lower costs and ease of assembly and disassembly in the field. In the gapless design, the hub and discharge ring geometry is spherical in the region swept by the blades requiring a more complex blade actuation mechanism and a removable upper portion of the discharge ring. Figure 6.2.1-1 shows a Kaplan blade at minimum and maximum tilts. Figure 6.2.1-2 shows the conventional design gaps created by cylindrical and conical sections on the hub and periphery. See Section 4.3.2.2 for descriptions of clearances and gaps.

Certain features discussed in this material relating to gapless Kaplan turbine designs are based on technology developed by Voith outside the scope of this DOE program and are the subject of one or more patent applications.

Rehabilitation Design
Figure 6.2.1-3 shows a gapless Kaplan runner design with spherical surfaces on the hub, blade, and discharge ring wherever the surfaces are in close proximity to each other. The gapless design will result in increased efficiency, improved cavitation, minimized leakage vortices and eliminated mechanical gaps through which fish could pass. Associated with the reduced mortality mechanisms and improved performance will be an increase in manufacturing cost and a more complicated assembly and disassembly of the unit in the field. Aspects of these design features have already been incorporated into the upgraded designs for Rocky Reach, Bonneville and Wanapum projects.

New Unit Design
A new unit design incorporates the same contours as the rehabilitation design. Spherical discharge ring shapes will be easier to achieve in a design of a new unit.

6.2.1.2 **Wicket Gate Overhang**

In a conventional design, the diameter of the wicket gate shaft location is often minimized in order to reduce the overall size and cost of the machine. Reducing the gate pin circle reduces the size of the gate operating mechanism, stay ring, head cover, bottom ring, spiral case, and possibly the power house. The runner diameter is maximized to improve hydraulic performance. The combination of minimum gate pin circle and maximum runner diameter can lead to wicket gates which overhang the bottom ring at higher gate openings. The resulting leakage flow through the overhung gap disrupts smooth flow and induces strong vortices and additional energy losses, i.e. avoidable losses.

Rehabilitation Design
The overhang can be reduced or eliminated by changing the shape of the discharge ring as shown in Figure 6.2.1-4 and/or moving the trailing edge of the wicket gate to a larger diameter. Reducing this overhang will increase efficiency and reduce avoidable losses.

New Unit Design
A new unit design will incorporate the same contours as the rehabilitation design. Spherical discharge ring shapes will be easier to achieve in the design of a new unit.

6.2.1.3 Optimized Hydraulic Design

An optimized hydraulic Kaplan design contains the design elements mentioned above: no gaps between the blades and the hub or periphery, and no wicket gate overhang. The design of rehabilitated, upgraded or new turbine components is optimized by cutting edge flow analysis (CFD) tools at all operating conditions to maximize efficiency, avoid cavitation and provide for maximum flow smoothness and minimized fluid mechanisms for mortality. Through use of the CFD tools, even backroll associated with draft tube exit velocity distributions can be addressed. New unit designs can benefit to a greater extent from the cutting edge technology as there is more flexibility in developing advanced geometrical shapes for the turbine components. Design changes for upgrade or rehabilitation are limited by, existing structures, the difficulty in alterations to concrete structures, and in many cases by component strength issues.

The use of cutting edge technologies has provided improved efficiency and cavitation performance for recent and "in progress" upgrades at Rocky Reach, Bonneville, Dardanelle, and Wanapum.

6.2.1.4 Lubrication

In a conventional Kaplan design the blade operating linkage mechanism inside the hub is submerged in lubrication oil. The oil can be either normal oil or one of the newer special biodegradable oils. Seals are designed to prevent oil leakage from the hub to the water and to prevent water leakage into the hub. Despite the seals, there is the potential for oil leaking into the water. The vast majority of existing units also use grease lubricated wicket gate bushings which have the potential of grease leaking into the water.

Certain concepts developed in connection with the use of an environmentally friendly medium to fill the hub were developed by Voith outside the scope or this DOE program and are the subject of a patent application.

Rehabilitation Design
Environmentally friendly designs will use an environmentally friendly medium inside the hub. The medium could be an biodegradable oil or, for example, water. The use of water would require the use of corrosion resistant materials and coatings that may be difficult to apply in an existing hub and use oil free bushings in place of the normal oil lubricated bushings for the blade adjustment mechanism. For the wicket gate mechanisms, the use of greaseless self-lubricated wicket gate bushings is foreseen. There are a variety of these bushings on the market today where the lubricant is an integral part of the bushing.

New Unit Design
The design concept will be the same as for a rehabilitation design but implementation will be simpler.

6.2.1.5 **Surface Roughness**

Roughness refers to turbine component surfaces. For older turbines, surface smoothness may have deteriorated over a period of time. Rough surfaces and surface roughness hot spots associated with cavitation damage or rough structural welds in the water passage (there may be weld joints in an as-welded condition in spiral cases, draft tubes, and some stay rings) can cause damage to the fish as it scrapes along or is carried against a surface as a result of the turbulent flow field within the turbine. As more biological data becomes available, it may be possible to evaluate fish injury for different values of roughness. The water velocity in some of these parts is relatively low. Therefore, the cost to "smooth" these surfaces and maintain them may not be justified from a performance standpoint.

Rehabilitation Design
Providing for a surface upgrade for the stay vane, wicket gate, and upper draft tube cone, as a minimum, may decrease descaling associated with scraping. Figure 6.2.1-5 shows a reduction in weld roughness. The surface upgrade can be accomplished by a combination of surface restoration, smoothing and use of special friction reducing coatings. Even compliant coatings may provide value on components where fish may impact the surface with high energy. Some older designs used rivets to join steel plates in some of the larger water passages such as in the spiral case and draft tube. For structural reasons these rivets can not be reduced in size.

New Unit Design
Selection of initial base material will have an effect on the ease with which surface smoothness can be maintained.

6.2.1.6 **Fixed vs Adjustable Rotational Speed (RPM)**

In a conventional design all operation is at a fixed rpm because generators have a fixed number of poles and must output a fixed frequency. Running at a fixed rpm restricts the machine to being at peak efficiency at only one combination of head and discharge. As head or discharge change, the point of operation moves away from the peak as shown in Figure 6.2.1-6 The plant operator has control of the discharge by controlling the gate opening. The plant operator has no direct control over the head. As head changes, the unit must move away from the point of optimum performance. This will cause a drop in efficiency and move the operating point closer to cavitation and/or pressure pulsation operation limits. Moving to different locations on the operating hill curve will also have an effect on fish friendliness as discussed in Section 4. The disadvantages of fixed rpm increase as the head range increases.

Rehabilitation Design
Installation of electrical conversion equipment enables the turbine to operate with adjustable speeds. To take full advantage of this equipment, a new runner design should be part of this upgrade. Adjustable rpm compensates for head variations. The combination of adjustable rpm and the conventional adjustable gate opening allows the machine to stay near the optimum operating point as shown in Figure 6.2.1-7. The definition of optimum is derived by considering:

- Strike probability
- Size of shear zones

- Efficiency
- Cavitation
- Pressure pulsations

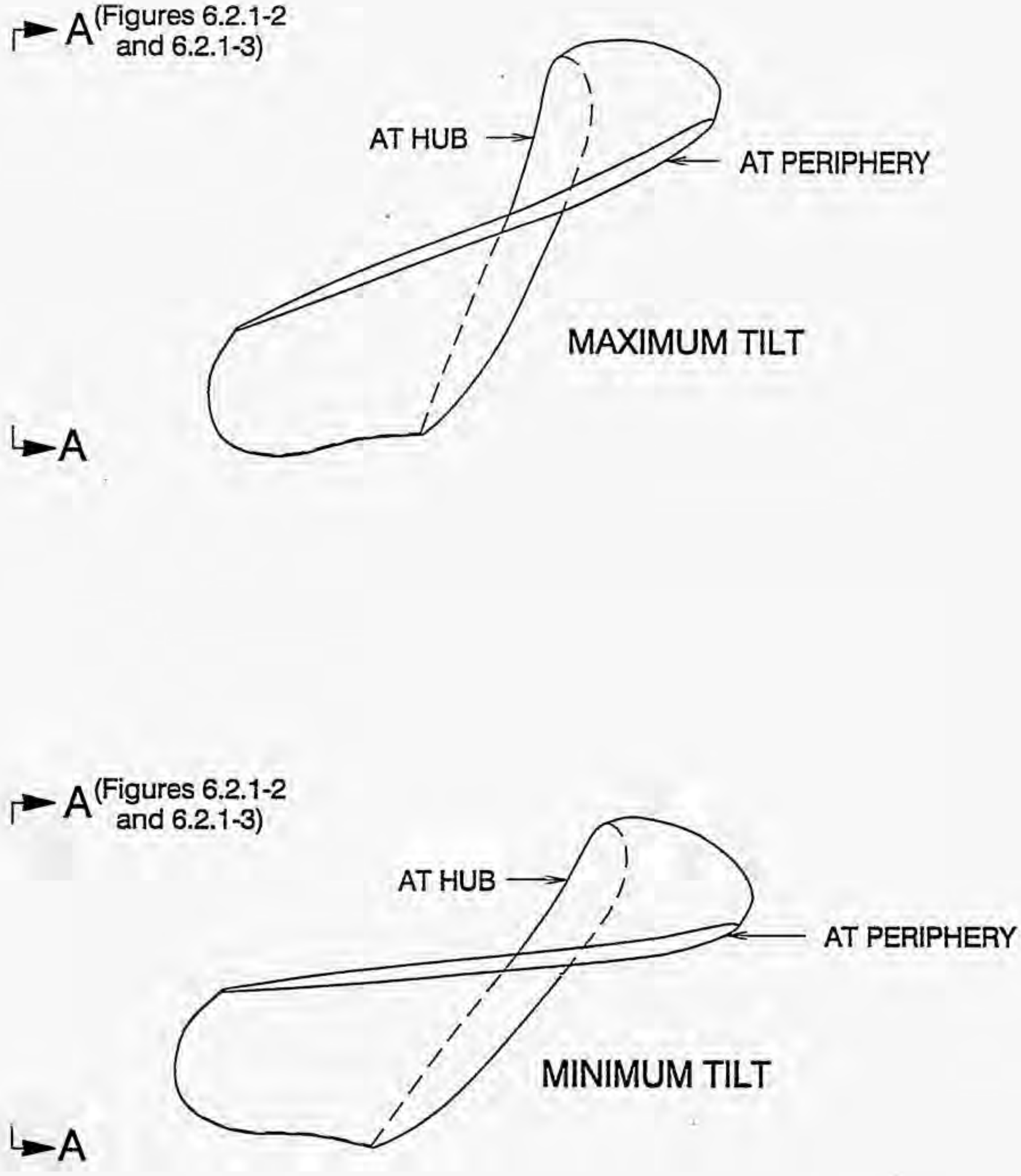
There is an inefficiency associated with the electrical conversion equipment. This will reduce the overall plant efficiency but has no impact on the hydraulic efficiency of the turbine. This conversion equipment is becoming more efficient and less expensive as research develops more cost effective solutions and a higher sales volume leads to lower costs. The conversion equipment is relatively large and must be housed indoors.

New Unit Design
The advantages of adjustable speed for a new unit is the same as for a rehabilitated unit. In addition, several of the hydraulic components would be designed to take full advantage of adjustable speed.

6.2.1.7 **Advanced Control System**

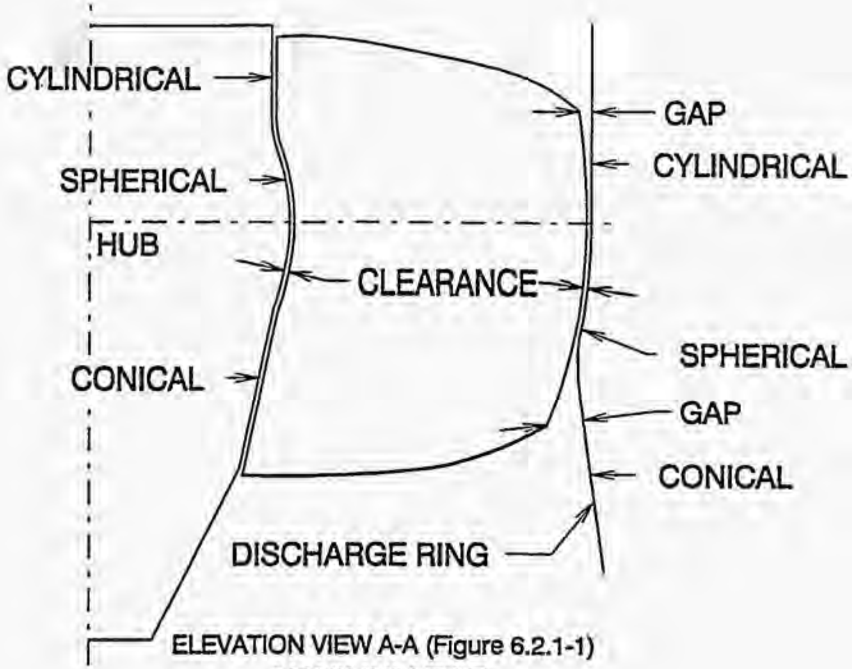
Utilizing advanced sensors and digital technology, an advanced control system can provide significant advantages in turbine operation to improve fish survival. Several features are discusses below.

- Adjustable speed control in conjunction with an adjustable speed generator can adjust the operation of the turbine to provide the maximum fish survival at the discharge of operation independent of the head at the power plant. At all heads of operation, the turbine characteristics can be those of the most fish friendly point of operation at the required discharge.
- It is a fact that most Kaplan turbines are not operated at the optimum cam position because of the difficulty in determining the optimum position and because of the dependence of the optimum position of a wide number of variables. Automated cam optimization can be incorporated into the turbine governor to assure that the turbine blade and wicket gates are in the positions to provide the maximum efficiency at a given head and discharge point of operation. Operation at the on cam blade and gate positions minimizes fluid injury mechanisms associated with off cam operation within the turbine. Certain features discussed in this material relating to Kaplan cam optimization are based on technology developed by Voith outside the scope of this DOE program and are the subject of pending patent applications.
- Kaplan turbines often operate with significant trash on the trash racks. This results in disturbed flow entering the intake and can move fish from the upper intake to the lower intake exposing them to greater injury associated with blade tip strike. Use of an automated system to announce that trash racks need cleaning can minimize this problem. Certain features discussed in this material relating to trash rack cleaning are based on technology developed outside the scope of this DOE program and are the subject of one or more patent applications.
- For multiunit plants, the units are often not operated based on optimized plant efficiency for the given output, but rather based on other criteria. An automated multiunit optimization system can aid the operator in determining how to operate the plant to optimize the plant efficiency for the required output. While this will provide better conditions for fish passage, it is still not the best that can be done. When fish are present in the unit, a system to operate the unit at optimum plant fish passage survival discharge would provide conditions within the turbine having lower mechanical and fluid mechanisms for injury. Such advanced sensing and control system is based on technology developed by Voith outside the scope of this DOE program and is the subject of pending patent applications.

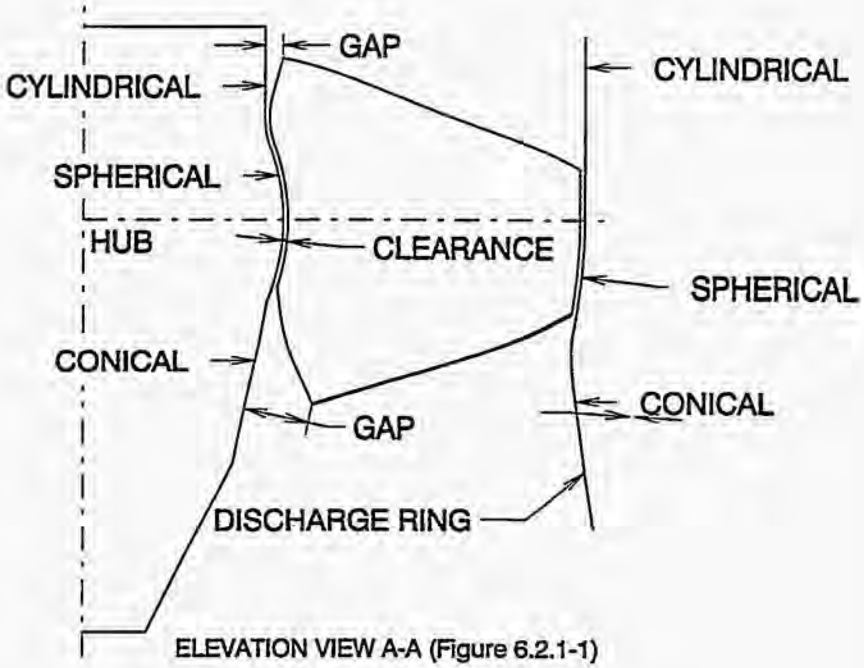


END VIEW OF BLADE WITHOUT HUB

Figure 6.2.1-1 Explanation of Blade Tilt

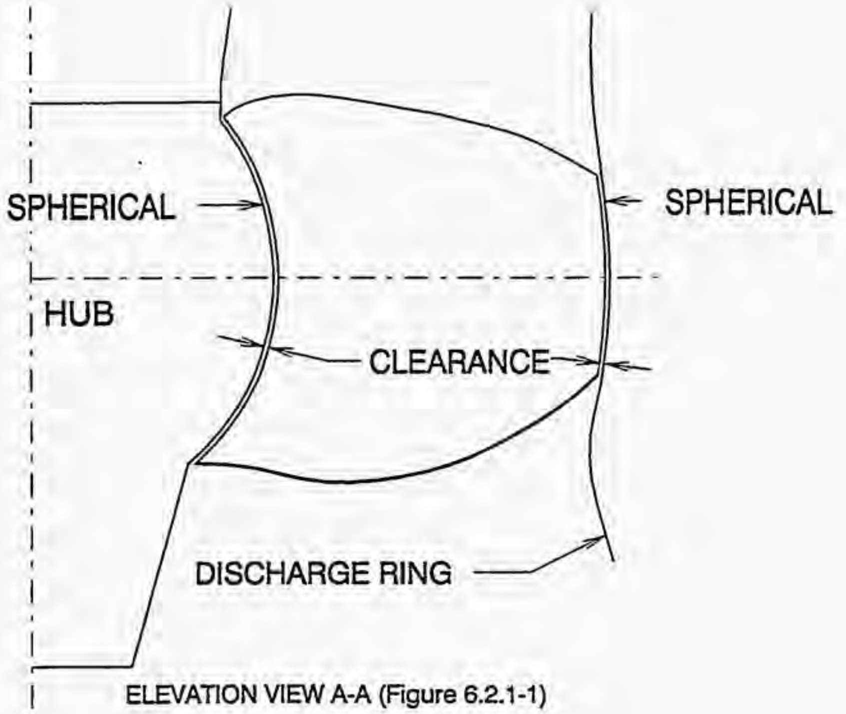


ELEVATION VIEW A-A (Figure 6.2.1-1)
MAXIMUM TILT

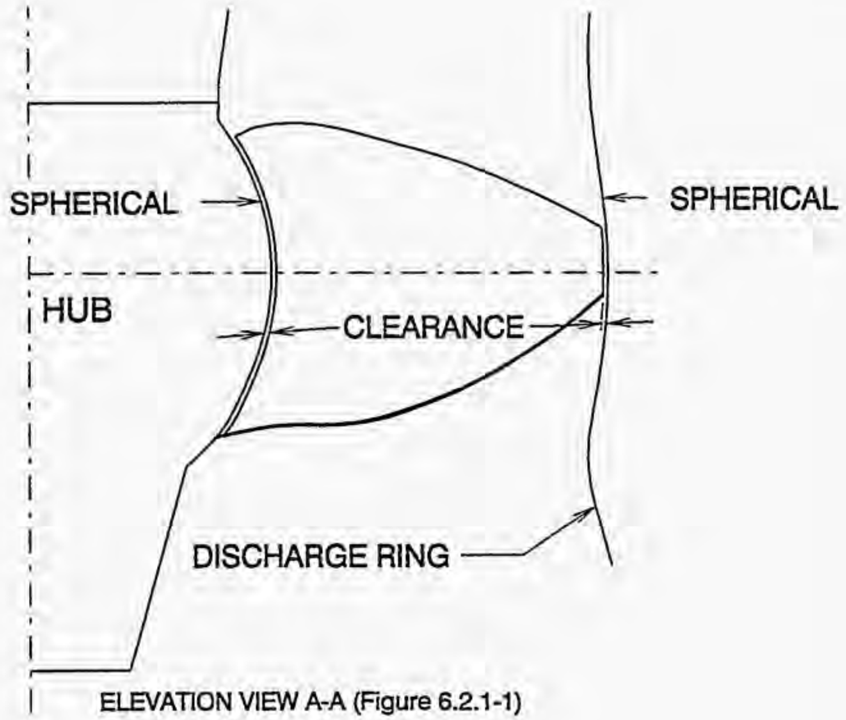


ELEVATION VIEW A-A (Figure 6.2.1-1)
MINIMUM TILT

Figure 6.2.1-2 Typical Runner with Combination of Cylindrical, Spherical, and Conical Hub and Discharge Ring Profiles



MAXIMUM TILT



MINIMUM TILT

Figure 6.2.1-3 Typical Runner with Spherical Hub and Discharge Ring Profiles

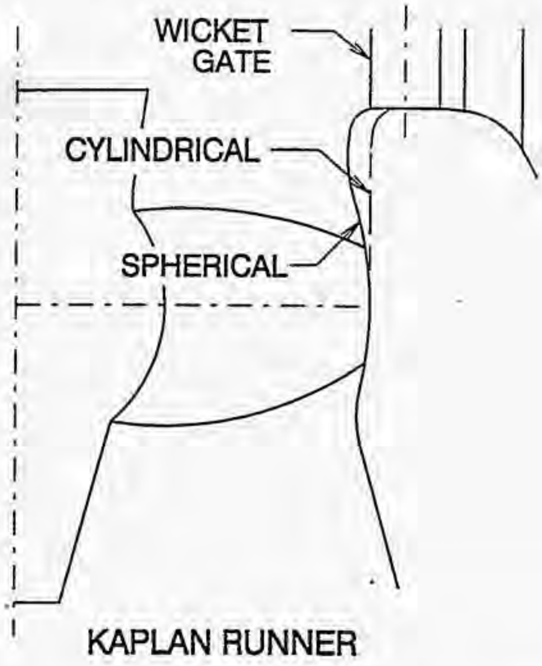


Figure 6.2.1-4 Elimination of Wicket Gate Overhang

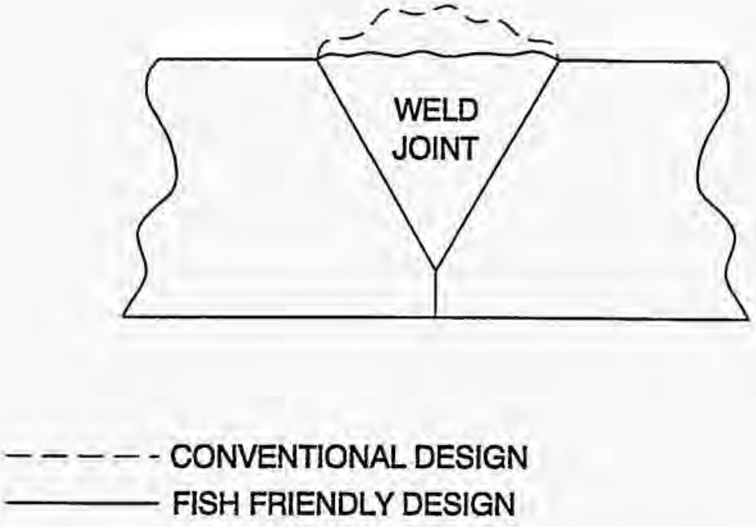


Figure 6.2.1-5 Reduction of Weld Joint Roughness

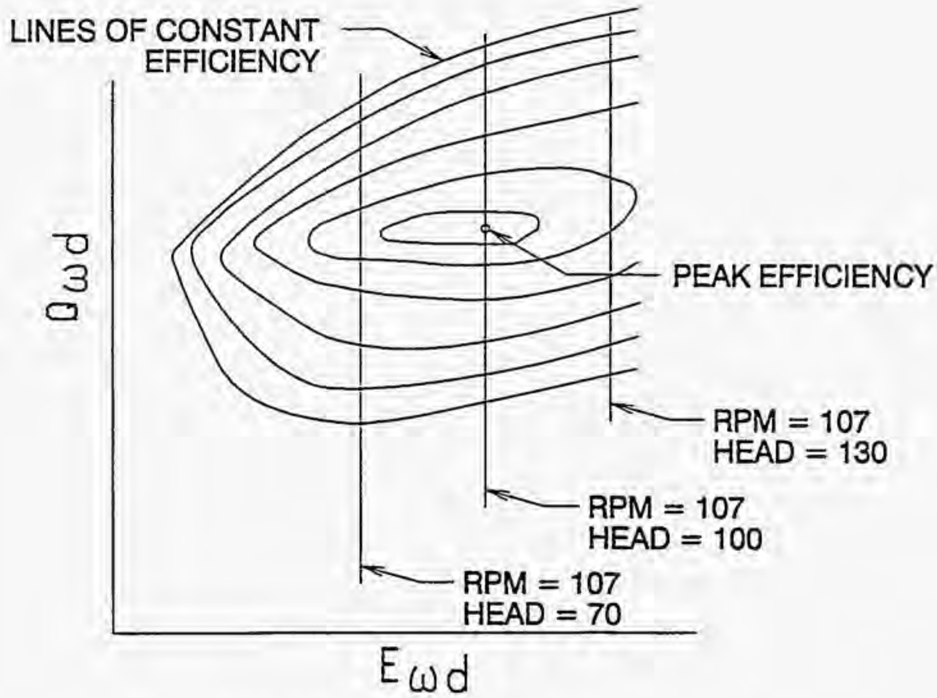


Figure 6.2.1-6 Operation Range with Constant RPM

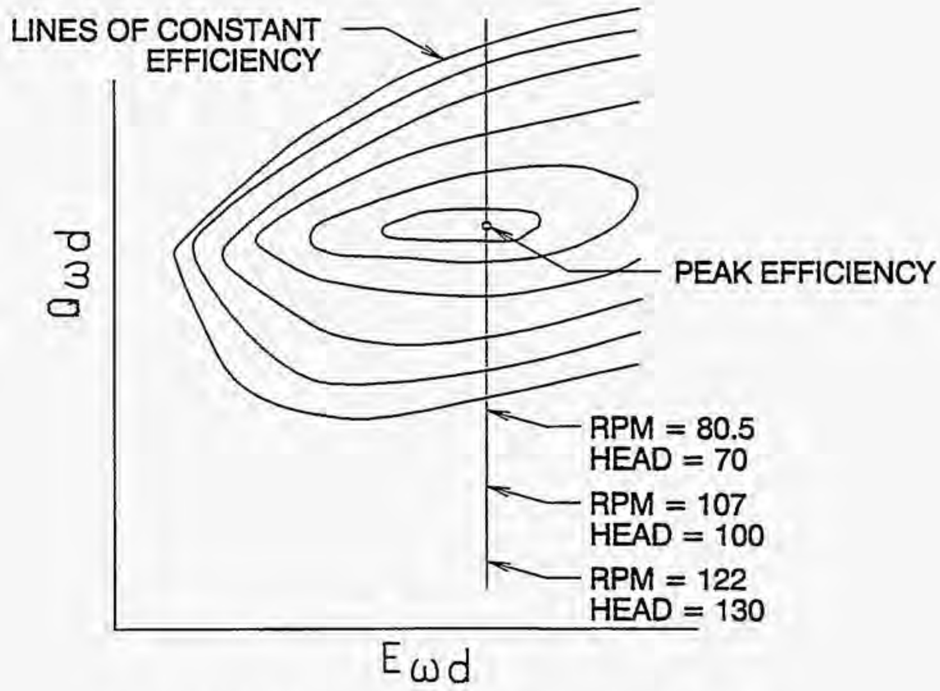


Figure 6.2.1-7 Operation Range with Adjustable RPM

6.2.2 SECONDARY ISSUES

6.2.2.1 **Interaction Of Wicket Gates and Stay Vanes**

Wicket gate overhang covered in Section 6.2.1.2 is one factor discussed in this section. The other factors are the number of wicket gates and the relative position of wicket gates to stay vanes. Each of these factors may have an interaction with one or more of the others.

In a conventional design the number of wicket gates and stay vanes typically varies from 16 to 32. Some existing designs use one stay vane for each two wicket gates. The wicket gates serve two purposes: 1) They shut off the flow when the unit is idle, and 2) They control the flow to the unit to achieve different levels of output. The shape of the wicket gate has a significant influence on efficiency. The stay vane is a very important structural member linking the top and bottom portions of the unit. The shape of the stay vane also has a significant influence on efficiency. The relative position of the stay vanes and wicket gates was often determined principally by mechanical considerations. Fluid analysis tools of the past were non existent or inadequate to address the effect of design shapes and relative position of the gates and stay vanes on efficiency. In some of the these designs the wicket gates and stay vanes were not aligned to minimize the potential for mechanical strike such as the wicket gate being located in the "shadow" of a stay vane as shown in Figure 6.2.2-1. CFD tools can help optimize the fluid induced disturbances and minimize mechanical strike for a selected range of gate openings.

Rehabilitation Design

Wicket Gate Overhang: As mentioned in section 6.2.1.2. Reducing this overhang will increase efficiency and reduce fluid and mechanical mechanisms for fish injury. The overhang can be reduced or eliminated by changing the shape of the discharge ring as shown in Figure 6.2.1-4 and/or moving the trailing edge of the wicket gate to a larger diameter. The trailing edge of the wicket gate can be moved to a larger diameter by moving the gate pin circle to a larger diameter and/or by shortening the wicket gate. Since the gates have to touch each other in the closed position, a shorter gate may require a larger number of gates. The above mentioned modification to the discharge ring will complicate the removal of the runner. An additional, less important, advantage of a larger gate pin circle is the lower velocities through the stay vanes and wicket gates which result from having these parts at a larger diameter and therefore at a larger flow through area.

Relative Position of Wicket Gates and Stay Vanes: Rotating the wicket gate pin circle can allow a wicket gate (at least at one gate opening) to be the "shadow" of a stay vane, thus reducing strike probability (Figure 6.2.2-1). The rotation must be evaluated by CFD methods to determine the effectiveness. Making this modification on an existing unit will require changes to the gate operating and servomotor system. Since this could be accomplished by rebolting components, the shift may be limited to the existing bolt spacing in the head cover and bottom ring.

Number of Wicket Gates: In general, lowering the number of wicket gates will decrease strike probability if it could be done without producing a gate overhang gap. However, it must be recognized that, for a rehabilitation, a larger number of wicket gates may allow a better alignment of wicket gates behind existing stay vanes and may be needed to eliminate gate overhang gaps. When more of the wicket gates are in the "shadow" of a stay vane the strike probability can be lower. For new units the number of wicket gates should equal one or two times the number of stay vanes with proper relative position. If the number of wicket gates and stay vanes are equal and uniformly spaced, each gate can be in the shadow of a stay vane. If the number of stay vanes can be one half the number of wicket gates, (for lower head units) every other gate can be in the shadow of a stay vane.

Achieving these changes in an existing plant will require careful review of the mechanical and civil structure.

New Unit Design
Accommodating the above mentioned geometries will be more easily done in a new unit design since civil, mechanical and hydraulic requirements can be coordinated at the initial design stage. Unconventional design shapes may be investigated using cutting edge CFD tools to minimize strike.

6.2.2.2 Rotational Speed (RPM)

The selection of rotational speed deals primarily with new units. The ongoing trend in the industry is to use higher fixed rpm units for a given site condition and output requirement. The size and cost of the turbine and generator decrease as the rpm increase. Since the smaller unit must produce the same output, the discharge must be the same as it would be for a larger unit. Therefore, the smaller unit would have higher water velocities. For the smaller unit with the same sized fish, the probability of strike will be higher. The energy concentrated in the fluid injury mechanisms will be higher. Therefore the trend to higher speeds for a given head turbine is going in the wrong direction for improved survivability.

New Unit Design
Use of a lower rpm machine for the design conditions will reduce the water velocities and increase the size of the water passages for a given site condition and output requirement. The lower rpm machine will have more margin against cavitation for a given submergence. If there is sufficient margin against cavitation, the lower rpm machine can be installed at a higher elevation. This will decrease installation costs. Larger units will, of course, have higher hardware costs and may require a larger power house.

6.2.2.3 Draft Tube Piers

The number and size of draft tube piers are determined by the civil design of the power house. Typical designs have from 0 to 2 vertical piers. A few designs also have horizontal piers. The velocities at the inlet to the pier are relatively low but the flow pattern can be complex at off-design conditions - especially for fixed blade axial flow machines.

Rehabilitation Design
For structural reasons elimination of piers in an existing powerhouse would not be practical. The shape of the pier nose should be reviewed using cutting edge CFD analysis tools for possible improvements.

New Unit Design
Eliminating the pier(s) removes a strike source. This would have to be incorporated into the initial powerhouse design and may be impractical for larger draft tubes. Changing the draft tube cross-sectional area will influence the structural requirements. Increasing the height while decreasing the width may allow a pier(s) to be eliminated. See Figure 6.2.2-2 for examples. If the pier(s) cannot be removed, the shape of the pier nose should be reviewed for possible improvements.

6.2.2.4 Runner Cones

In a conventional design axial flow hubs are typically extended beyond the outlet edge of the blades with a conical extension (Figure 10.2-7).

Rehabilitation Design
Further investigation is required to determine if the shape and length of these extensions influence the flow patterns in a manner that affects fish friendliness.

New Unit Design
Same as rehabilitation design.

6.2.2.5 **Inlet Valves**

Higher head axial flow machines sometimes have a penstock and inlet pipe instead of the more conventional rectangular inlet section. For these units there is a valve near the inlet to the spiral case. These valves have a disk that remains in the water passage when it is in its full-open position. This obstruction in the water passage causes some loss of efficiency and creates some strike probability.

Rehabilitation Design
Option 1: Use a spherical valve that will have no obstruction in the water passage. This is an expensive solution, but it is the best valve for fish friendliness and efficiency. A spherical valve housing is relatively large. Retrofitting will require a detailed review of the plant layout.

Option 2: Install trunnion fairings and blend any sharp corners that have an obstruction in the water passage. This is a relatively inexpensive solution. These modifications will reduce the head loss and increase the efficiency of the valve.

New Unit Design
Use of a spherical valve or a ring gate will eliminate the obstruction from the water passageway.

6.2.2.6 **Sharp Corners**

In a conventional design some of the corners at inlet edges are manufactured with sharp corners. The cost to round these corners is not justified from a performance standpoint.

Rehabilitation Design
Rounding of these corners may decrease injury when strike does occur. See Figure 6.2.2-3

New Unit Design
Same as rehabilitation design.

6.2.2.7 **Gate Slots**

Sliding gates are used as closure devices in the draft tube and/or inlet sections of turbines. They ride in slots that are located on the perimeter of the sections. When the units are operating, these gates are always fully opened which exposes the gate slots to the flow. These slots typically have relatively sharp corners and can cause vortices.

Rehabilitation Design
Filling these slots with a removable filler strip will minimize the possibility of a strike. The time needed to install and remove these fillers must be weighed against the frequency of use. See Figure 6.2.2-4.

New Unit Design
In addition to slot filling, rounding the corners of these slots could decrease the probability of injury if a strike occurs.

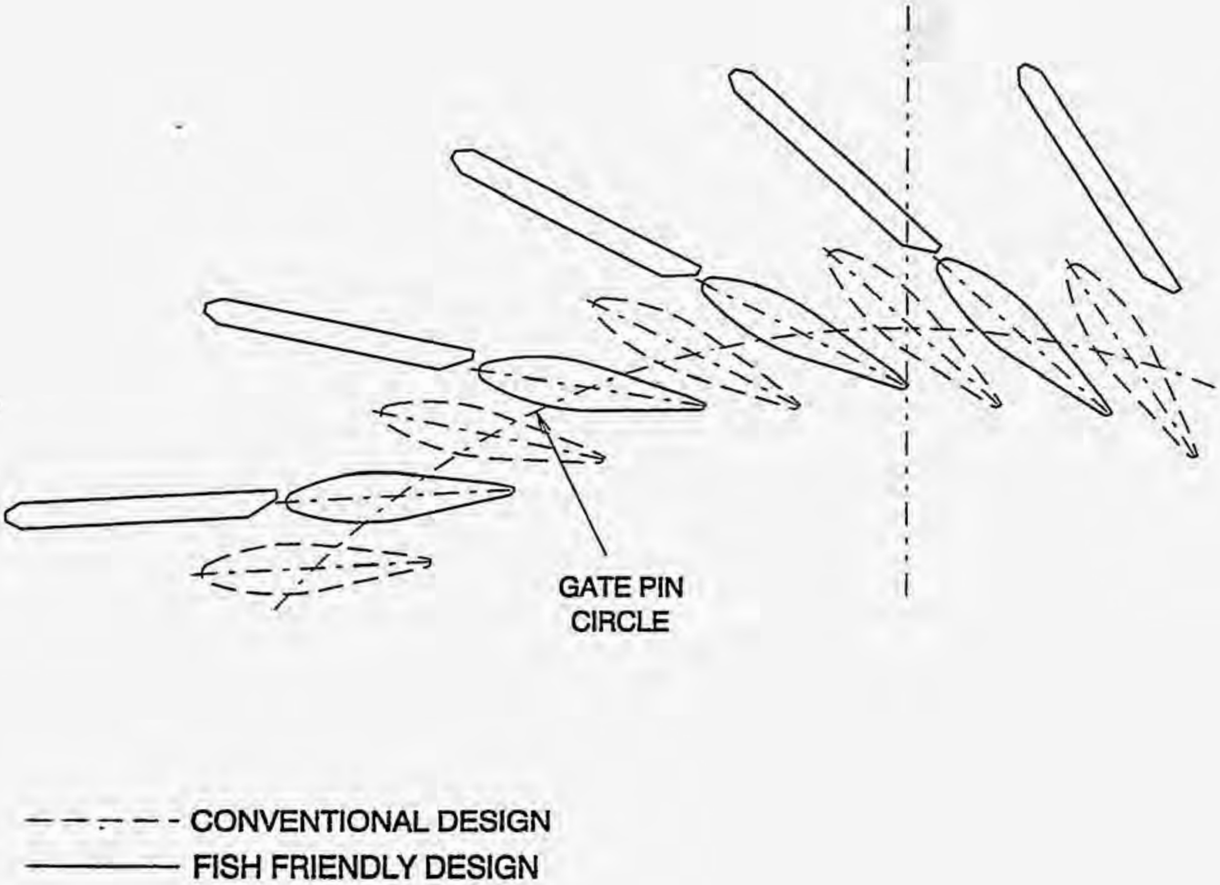
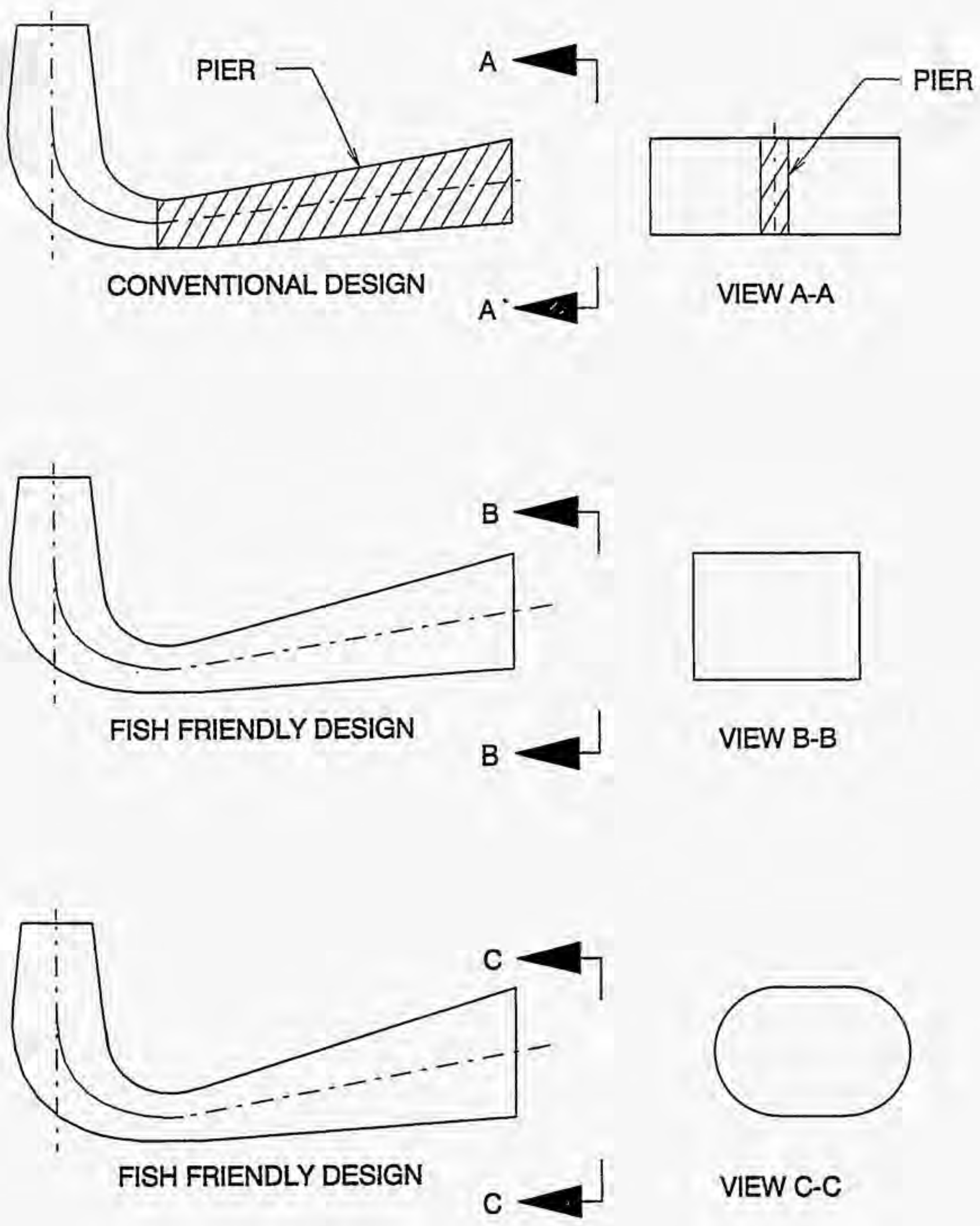


Figure 6.2.2-1 Wicket Gate / Stay Vane Relative Position



NOTE: OUTLET AREA IS SAME ON ALL VIEWS

Figure 6.2.2-2 Elimination of Draft Tube Piers

- - - - - CONVENTIONAL DESIGN

——— FISH FRIENDLY DESIGN

Figure 6.2.2-3 Elimination of Sharp Corners

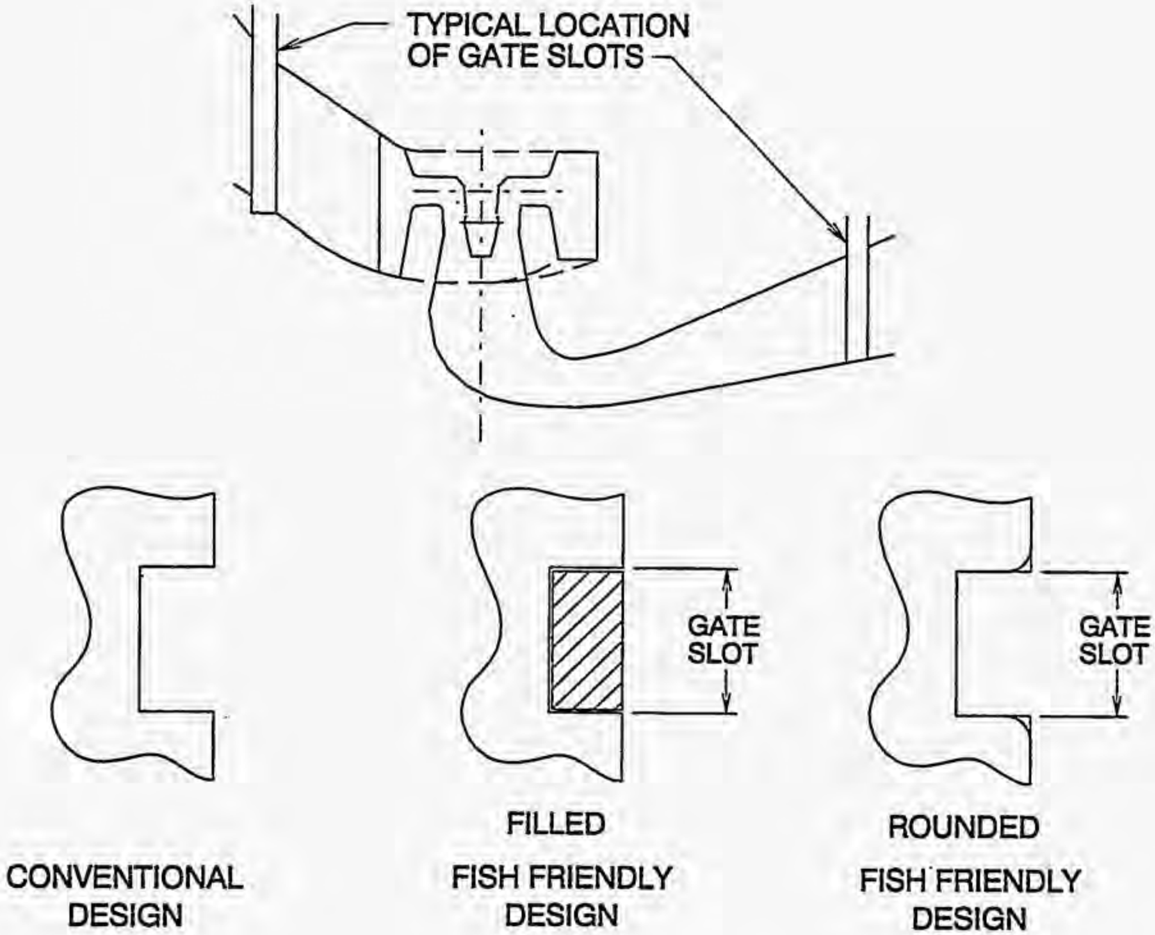


Figure 6.2.2-4 Treatment of Gate Slots

6.3 ENVIRONMENTALLY ADVANCED FRANCIS DESIGN FOR IMPROVED FISH SURVIVABILITY

6.3.1 PRIMARY ISSUES

6.3.1.1 **Number of Runner Blades**

A Francis runner has multiple fixed position blades attached to a central hub (crown) and an outer shroud (band). The runner rotates inside a stationary headcover and bottom ring. The shape of the runner for a low head site is much different from one for a high head site. See Figures 10.2-9 and 10.2-10. Traditional designs have runners with the number of blades varying from 13 to 30.

Rehabilitation Design

In comparison to a conventional design, a more fish friendly Francis runner design would have a lower number of blades which would decrease probability of strike and increase the space between adjacent blades minimizing the probability for abrasion. Lowering the number of blades while maintaining the same design power rating will usually result in longer (from top to bottom) blades in order to maintain capacity and minimize cavitation. See Figure 6.3.1-1. To implement a lower number of blades design for a turbine rehabilitation or upgrade will require some mechanical and civil changes to the bottom ring and draft tube cone. Advanced CDF flow analysis tools and FEM structural analysis tools are suitable for comparing alternative designs. Section 4.3.5 discusses the results of a series of design studies to evaluate the tradeoffs in lowering the number of blades for a typical Francis turbine project where an upgrade has take place. As can be seen from that section, the effect of the number of blades on strike probability is especially significant for smaller runners

New Unit Design

An environmentally enhanced runner design for new turbines would have similar characteristics as that for the rehabilitation design but the implementation may be less complicated because the longer blades could be more easily accommodated by the initial design of the mechanical and civil structure. Because of the effect of the rotational speed selection on the runner size and corresponding strike probability, a different set of geometries for the runner may result. See Section 6.3.2.1.

6.3.1.2 **Inlet Edge Thickness**

The effect of the inlet edge thickness on injury mechanism is two fold. The first is on the fluid injury mechanisms discussed in Section 4.3.3. The angle of the water with respect to the runner blade varies with the operating head and discharge. When the unit must operate over a wide range of heads there will be a wide variation of water inflow angle of attack to the blade. This range of inflow angles creates an unfavorable flow field with large shear zones at the blade entrance edge and can give rise to large efficiency losses at operating points at the extremes of the operating head range. If there is too much mismatch between the water angle and the blade angle (too high an attack angle) cavitation can originate at the inlet edge. For many Francis sites the head range is relatively small with (Hmax-Hmin)/Hoptimum < 0.1. Runners at these sites are acceptable with standard inlet edge thickness. For the sites that have large head ranges, (Hmax-Hmin)/Hoptimum > 0.4, a thicker inlet edge is clearly beneficial in minimizing the fluid injury mechanisms at heads of operation near the extremes of the head range. The second effect relates to mechanical mechanisms of injury. While not validated with CFD tools it is believed that a thicker entrance edge than that normally required for classical design reasons will help carry small fish around the entrance edge, thus minimizing the probability of strike.

Rehabilitation Design

In developing a design for rehabilitation of upgrade of an existing runner, using a thicker blade entrance edge than that required by conventional design reasons will result in a runner design having somewhat flatter efficiency characteristics as a function of head. The improved efficiency and reduced susceptibility for entrance edge cavitation at heads at the extremes of the head range will result in less intense fluid related injury mechanisms. As a result of the thicker edge, the enhanced tendency of the flow to carry fish around the entrance edge rather than against it will lower the probability of strike. Runner costs will increase as a result of using a thicker blade design.

New Unit Design

Same as rehabilitation design.

6.3.1.3 Interaction Of Runner Blades, Wicket Gates, And Stay Vanes

In a conventional design the gate pin circle is minimized in order to reduce the overall size of the machine. Reducing the gate pin circle reduces the size of the gate operating mechanism, stay ring, head cover, bottom ring, spiral case, and possibly the power house. The runner diameter is often maximized to improve hydraulic performance. The combination of minimum gate pin circle and maximum runner diameter leads to wicket gates which overhang the bottom ring. The resulting leakage flow through the overhung gap disrupts smooth flow and induces additional energy losses, i.e. avoidable losses. Wicket gates with small gate pin circle can also result in a situation where the discharge of the wicket gates can be near the Francis runner blades when operating near full gate opening. This close proximity of the wicket gates to Francis runner blades can result in the chopping of fish between the stationary wicket gate and rotating Francis runner blades if this distance is comparable to fish length.

In conventional designs, the number of wicket gates and stay vanes typically varies from 16 to 32. Some designs use one stay vane for each two wicket gates. The wicket gates serve two purposes: 1) They shut off the flow when the unit is idle, and 2) They control the flow to the unit to achieve different levels of output. The shape of the wicket gate has a significant influence on efficiency. The stay vane is a very important structural member linking the top and bottom portions of the unit. The shape of the stay vane also has a significant influence on efficiency. The relative position of the stay vanes and wicket gates was often determined principally by mechanical considerations. Fluid analysis tools of the past were non existent or inadequate to address the effect of design shapes and the relative position of the gates and stay vanes on efficiency. In some of the these designs the wicket gates and stay vanes were not aligned to minimize the potential for mechanical strike such as the wicket gate being located in the "shadow" of a stay vane as shown in Figure 6.2.2-1. CFD tools can help optimize the fluid induced disturbances and minimize mechanical strike for a selected range of gate openings.

Rehabilitation Design

Wicket Gate Overhang: As mentioned above, reducing this overhang will increase efficiency and reduce fluid and mechanical mechanisms for fish injury. The overhang can be reduced or eliminated by changing the shape of the discharge ring as shown in Figure 6.3.1-2 and/or moving the trailing edge of the wicket gate to a larger diameter. The trailing edge of the wicket gate can be moved to a larger diameter by moving the gate pin circle to a larger diameter and/or by shortening the wicket gate. Since the gates have to touch each other in the closed position, a shorter gate may require a larger number of gates. The above mentioned modification to the discharge ring will complicate the removal of the runner. An additional, less important, advantage of a larger gate pin circle is the lower velocities through the stay vanes and wicket gates which result from having these parts at a larger diameter and therefore at a larger flow through area.

Space between Francis Runner and Wicket Gates: Moving the gate pin circle to a larger diameter will increase the space between the outlet edge of the wicket gates and the rotating runner. For smaller turbines, this should reduce the probability of a fish being trapped and chopped by the action of the runner blade passing by the wicket gates. An additional, less important, advantage in doing this will be the lower velocities through the stay vanes and wicket gates resulting from having these parts at a large diameter and therefore a larger flow through area. See Figure 6.3.1-3. Alternatively, a runner design with an unconventional blade shape may be explored to increase the space between the gate trailing edge and the blade entrance edge.

Relative Position of Wicket Gates and Stay Vanes: While an issue of secondary effect, shifting the gate pin circle can allow a wicket gate (at least at one gate opening) to be the "shadow" of a stay vane, thus reducing strike probability. See Figure 6.2.2-1. Making this modification on an existing unit will require changes to the gate operating and servomotor system. Since this could be accomplished by rebolting components, the shift may be limited to the existing bolt spacing in the head cover and bottom ring.

Achieving these changes in an existing plant will require careful review of the mechanical and civil structure.

New Unit Design

Accommodating the above mentioned geometries will be more easily done in a new unit design since civil, mechanical and hydraulic requirements can be coordinated at the initial design stage. In addition to the above, the number of wicket gates should be evaluated for reducing mechanical injury effects.

Number of Wicket Gates: In general, lowering the number of wicket gates will decrease strike probability. The number of wicket gates should equal one or two times the number of stay vanes with proper relative position. If the number of wicket gates and stay vanes are equal, each gate can be in the shadow of a stay vane. If the number of stay vanes can be one half the number of wicket gates, (for lower head units) every other gate can be in the shadow of a stay vane.

6.3.1.4 Optimized Hydraulic Design

An optimized Francis design contains the design elements mentioned above. The runner consists of fewer blades than a conventional design, with greater blade thickness at the entrance edges. Fewer blades will reduce the leading edge strike probability and the thicker (and carefully profiled) blade entrance edge shapes will reduce shear zones when the turbine is operated away from the best efficiency condition. Stay vanes and wicket gate shapes can be optimized to minimize fluid and mechanical mechanisms for injury. The design of rehabilitated, upgraded or new turbine components is optimized by cutting edge flow analysis (CFD) tools at all operating conditions to maximize efficiency, avoid cavitation and provide for maximum flow smoothness and minimized fluid mechanisms for mortality. Through use of the CFD tools, even backroll associated with draft tube exit velocity distributions can be addresses. New unit designs can benefit to a greater extent from the cutting edge technology as there is more flexibility in developing advanced geometrical shapes for the turbine components. Design changes for upgrade or rehabilitation are limited by, existing structures, the difficulty in alterations to concrete structures, and in many cases by component strength issues.

The use of cutting edge technologies has provided improved efficiency and cavitation performance for recent and "in progress" upgrades. Some recently redesigned Francis turbine runners incorporating 11 blades are operating at Flatrock and West Buxton plants and are being manufactured or installed for Sartell, McDougal, Watertown, Rock Island Arsenal and Nottely plants.

6.3.1.5 Lubrication

The vast majority of units use grease lubricated wicket gate bushings which have the potential of grease leaking into the water.

Rehabilitation Design

An environmentally advanced Francis turbine would use greaseless self-lubricated wicket gate bushings. There are a variety of these bushings on the market today where the lubricant is an integral part of the bushing.

New Unit Design

The goals will be the same as a rehabilitation design but implementation will be simpler.

6.3.1.6 Surface Roughness

Roughness refers to turbine component surfaces. For older turbines, surface smoothness may have deteriorated over a period of time. Rough surfaces and surface roughness hot spots associated with cavitation damage or rough structural welds in the water passage (there may be weld joints in an as-welded condition in spiral cases, draft tubes, and some stay rings) can cause damage to the fish as it scrapes along or is carried against a surface as a result of the turbulent flow field within the turbine. As more biological data becomes available, it may be possible to evaluate fish injury for different values of roughness. The water velocity in some of these parts is relatively low. Therefore, the cost to "smooth" these surfaces and maintain them may not be justified from a performance standpoint.

Rehabilitation Design

Providing for a surface upgrade for the stay vane, wicket gate, and upper draft tube cone, as a minimum, may decrease descaling associated with scraping. Figure 6.2.1-5 shows a reduction in weld roughness. The surface upgrade can be accomplished by a combination of surface restoration, smoothing and use of special friction reducing coatings. Even compliant coatings may provide value on components where fish may impact the surface with high energy. Some older designs used rivets to join steel plates in some of the larger water passages such as in the spiral case and draft tube. For structural reasons these rivets can not be reduced in size.

New Unit Design

Selection of initial base material will have an effect on the ease with which surface smoothness can be maintained.

6.3.1.7 Fixed vs Adjustable Rotational Speed (RPM)

In a conventional design all operation is at a fixed rpm because generators have a fixed number of poles and must output a fixed frequency. Running at a fixed rpm restricts the machine to being at peak efficiency at only one combination of head and discharge. As head or discharge change, the point of operation moves away from the peak as shown in Figure 6.2.1-6. The plant operator has control of the discharge by controlling the gate opening. The plant operator has no direct control over the head. As head changes, the unit must move away from the point of optimum performance. This will cause a drop in efficiency and move the operating point closer to cavitation and/or pressure pulsation operation limits. Moving to different locations on the operating hill curve will also have an effect on fish friendliness as discussed in Section 4. The disadvantages of fixed rpm increase as the head range increases.

Rehabilitation Design

Installation of electrical conversion equipment enables the turbine to operate with adjustable speeds. To take full advantage of this equipment, a new runner design should be part of this upgrade. Adjustable rpm compensates for head variations. The combination of adjustable rpm and the conventional adjustable gate opening allows the machine to stay near the optimum operating point as shown in Figure 6.2.1-7. The definition of optimum is derived by considering:

- Strike probability
- Size of shear zones
- Efficiency
- Cavitation
- Pressure pulsations

There is an inefficiency associated with the electrical conversion equipment. This will reduce the overall plant efficiency but has no impact on the hydraulic efficiency of the turbine. This conversion equipment is becoming more efficient and less expensive as research develops more cost effective solutions and a higher sales volume leads to lower costs. The conversion equipment is relatively large and must be housed indoors.

New Unit Design

The advantages of adjustable speed for a new unit is the same as for a rehabilitated unit. In addition, several of the hydraulic components would be designed to take full advantage of adjustable speed.

6.3.1.8 Advanced Control System

Utilizing advanced sensors and digital technology, an advanced control system can provide significant advantages in turbine operation to improve fish survival. Several features are discusses below.

- Adjustable speed control in conjunction with an adjustable speed generator can adjust the operation of the turbine to provide the maximum fish survival at the discharge of operation independent of the head at the power plant. At all heads of operation, the turbine characteristics can be those of the most fish friendly point of operation at the required discharge.
- Francis turbines often operate with significant trash on the trash racks. This results in disturbed flow entering the intake and can move fish from the upper intake to the lower intake exposing them to greater injury associated with blade tip strike. Use of an automated system to announce when trash racks need cleaned can minimize this problem. Certain features discussed in this material relating to trash rack cleaning are based on technology developed outside the scope of this DOE program and are the subject of one or more patent applications.
- For multiunit plants, the units are often not operated based on optimized plant efficiency for the given output, but rather based on other criteria. An automated multiunit optimization system can aid the operator in determining how to operate the plant to optimize the plant efficiency for the required output. While this will provide better conditions for fish passage, it is still not the best that can be done. When fish are present in the unit, a system to operate the unit at optimum plant fish passage survival discharge would provide conditions within the turbine having lower mechanical and fluid mechanisms for injury. Such an advanced sensing and control system is based on technology developed by Voith outside the scope of this DOE program and is the subject of pending patent applications.

6.3.1.9 **Pressure Change**

Fish mortality due to sudden reductions of pressure is of primary importance at sites with operating heads above approximately 35 m (120 ft). Section 4.3.4 addresses the mechanisms of these pressure changes.

Rehabilitation Design

Since the design concepts described in this section involve the civil structure of the dam and / or penstock system, they are primarily of interest for new sites and will be impractical to implement at most existing plants.

New Unit Design

Figure 6.3.1-4 shows plant layout possibilities for plants with short penstocks. Option (a) shows a dam with a penstock entrance at a relatively low elevation. This inlet will draw fish which have become acclimated to high pressures. They will then be transported quickly to the low pressure of the tailrace, with the possibility of decompression trauma. Option (b) is more fish friendly because it will draw fish which are acclimated to low pressures and transport them quickly through the high pressure region at the turbine inlet and back to the low pressure in the tailrace. Design concepts like that of Figure 6.3.1-4b are preferred to minimize decompression trauma.

Figure 6.3.1-5 shows plant layout possibilities for plants with long penstocks. Option (a) shows a penstock which draws fish at low to moderate pressure and then exposes them to a high pressure and retains them at this high pressure for a period long enough for acclimation and then transports them quickly to the low pressure of the tailrace, with the possibility of decompression trauma. Option (b) is more fish friendly because it allows the fish to remain at low pressures and transports them quickly through the high pressure region and back to the low pressure in the tailrace. Design concepts like that of Figure 6.3.1-5b are preferred to minimize decompression trauma.

Obviously, civil constraints may dictate some compromise between the (a) and (b) solutions. If a long penstock of type (a) can not be avoided, a smaller diameter penstock could be considered. Smaller penstock diameters will increase water velocity and reduce both the transport time and the time for acclimation to high pressures. Smaller penstocks will reduce installation cost but also plant efficiency as a consequence of higher head losses.

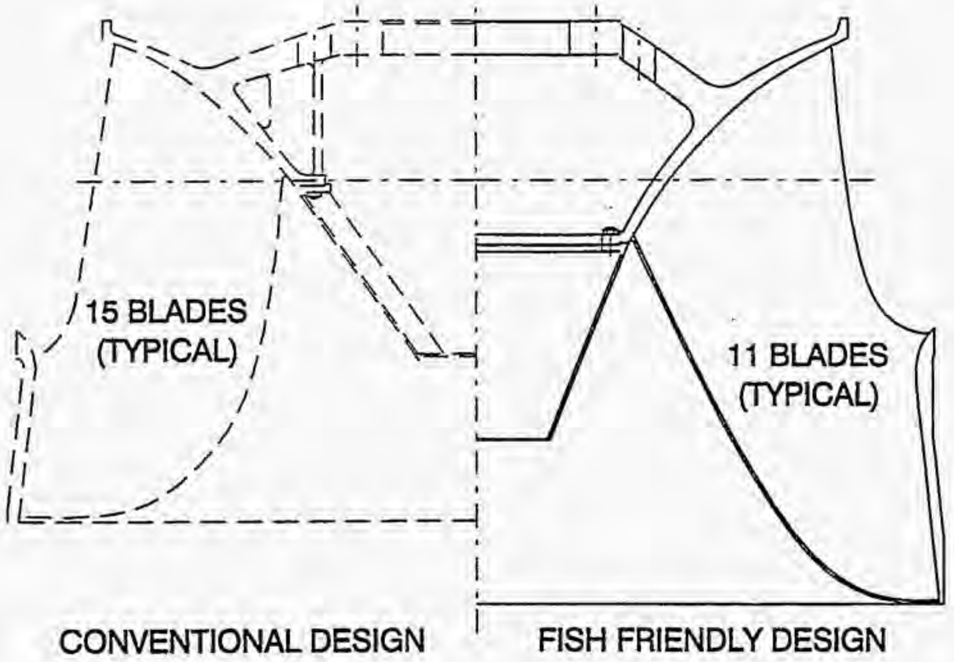


Figure 6.3.1-1 Reduction of Number of Francis Runner Blades

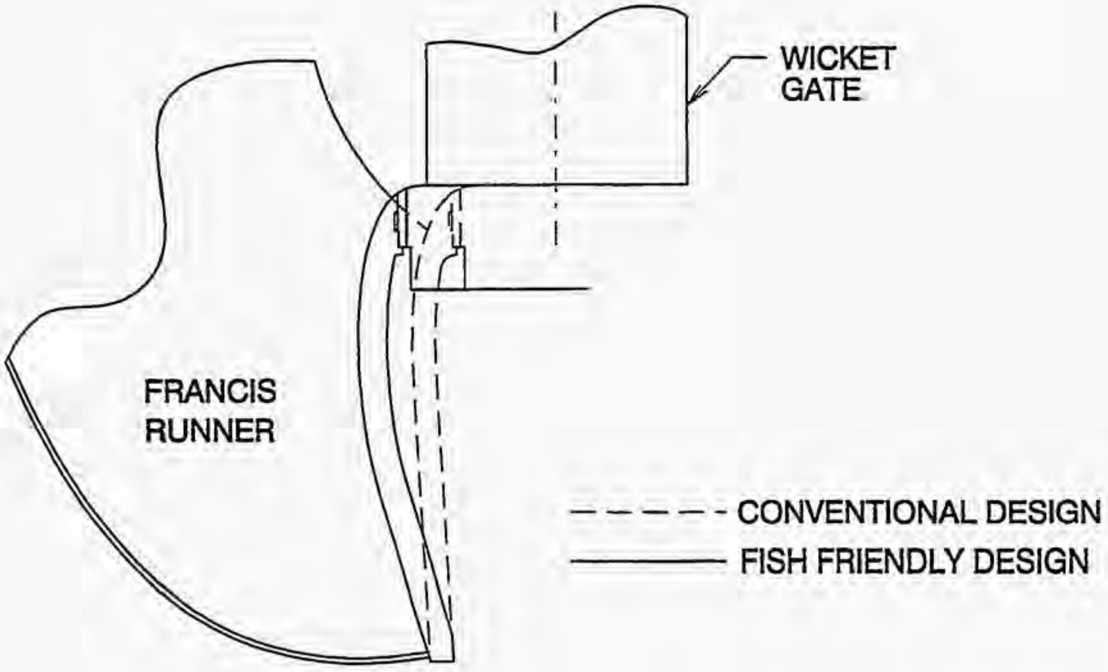
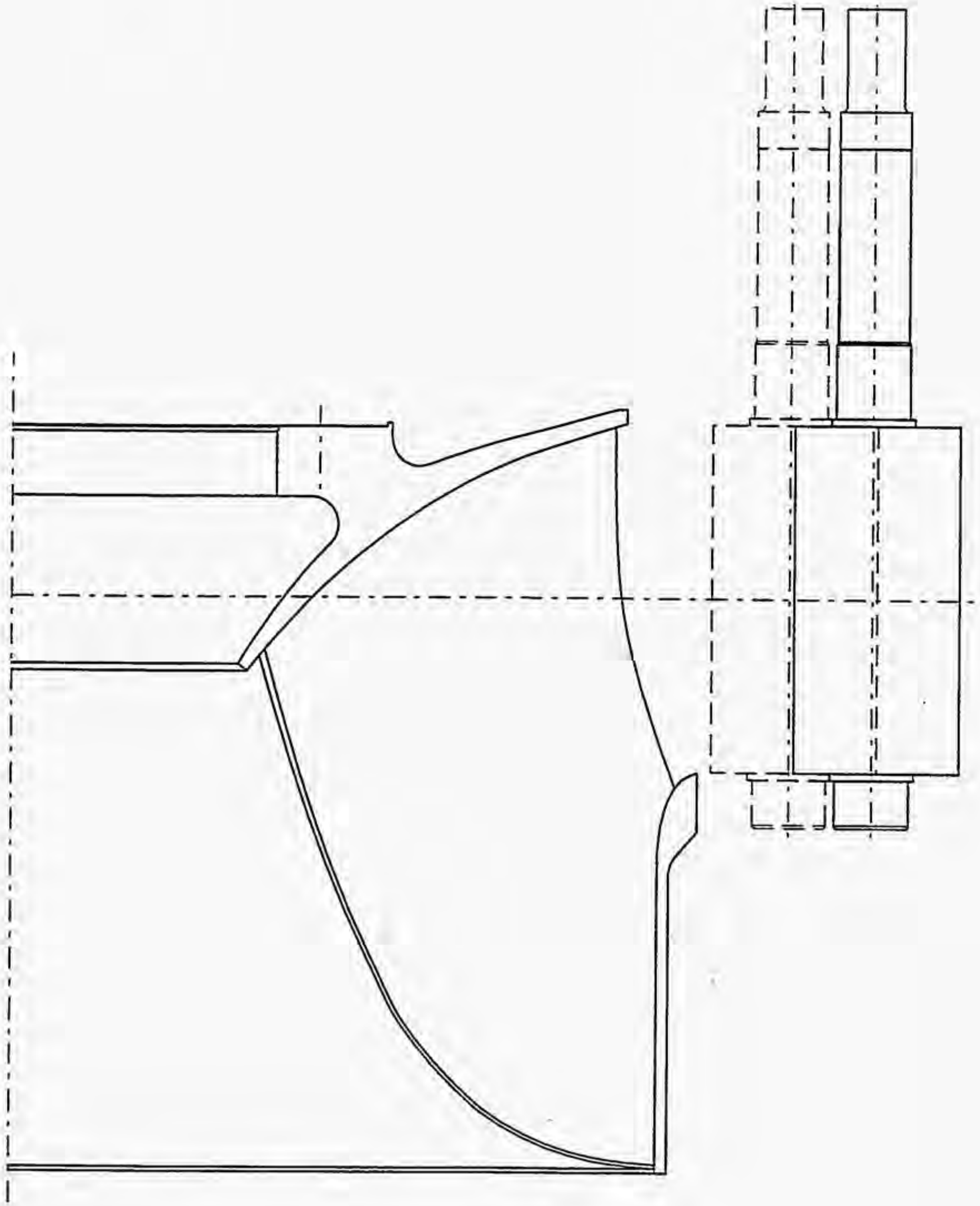


Figure 6.3.1-2 Elimination of Wicket Gate Overhang



- - - - - - CONVENTIONAL DESIGN

———— FISH FRIENDLY DESIGN

Figure 6.3.1-3 Increased Space Between Wicket Gates and Runner

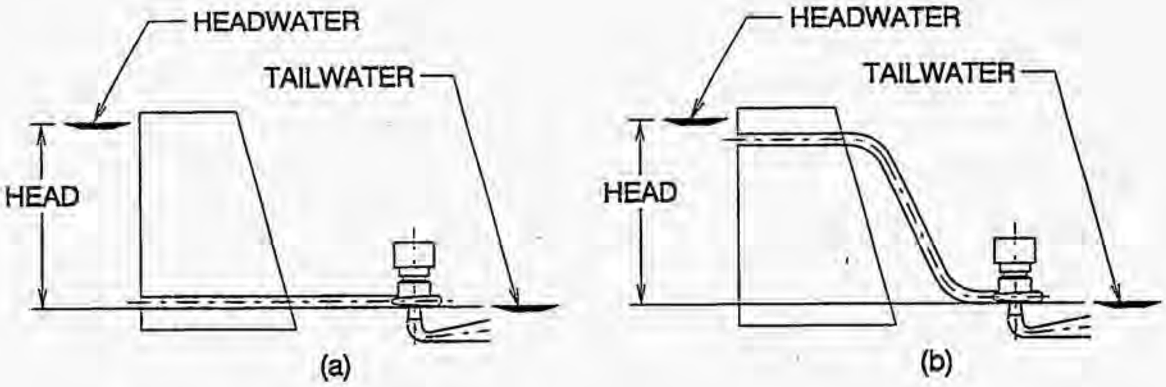


Figure 6.3.1-4 Pressure Change in Short Penstocks
(a) Will draw fish acclimated to high pressure and transport quickly to low pressure
(b) Will draw fish acclimated to low pressure and transport quickly through high pressure and back to low pressure

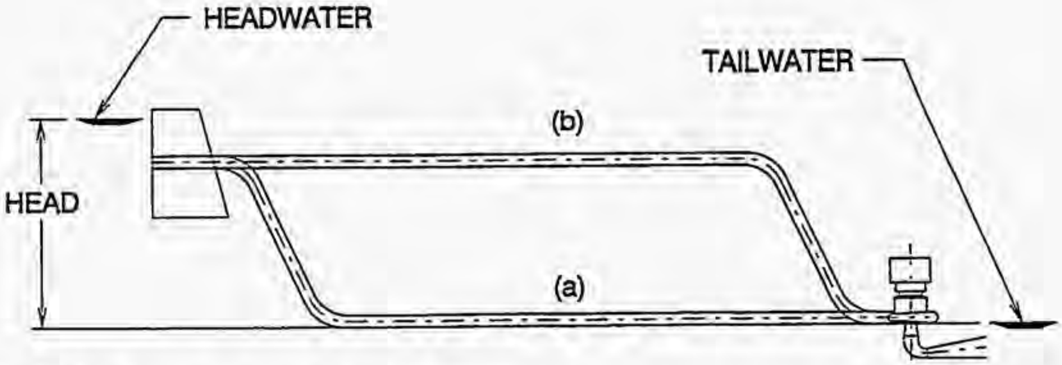


Figure 6.3.1-5 Pressure Change in Long Penstocks
(a) Will retain fish at high pressure long enough for acclimation and then transport quickly to low pressure
(b) Will retain fish at low pressure and transport quickly through high pressure and back to low pressure

6.3.2 SECONDARY ISSUES

6.3.2.1 Rotational Speed (RPM)

The selection of rotational speed deals primarily with new units. The ongoing trend in the industry is to use higher fixed rpm units for a given site condition and output requirement. The size and cost of the turbine and generator decrease as the rpm increase. Since the smaller unit must produce the same output, the discharge must be the same as it would be for a larger unit. Therefore, the smaller unit would have higher water velocities. For the smaller unit with the same sized fish, the probability of strike will be higher. The energy concentrated in the fluid injury mechanisms will be higher. Therefore the trend to higher speeds for a given head turbine is going in the wrong direction for improved survivability.

New Unit Design
Use of a lower rpm machine for the design conditions will reduce the water velocities and increase the size of the water passages for a given site condition and output requirement. The lower rpm machine will have more margin against cavitation for a given submergence. If there is sufficient margin against cavitation, the lower rpm machine can be installed at a higher elevation. This will decrease installation costs. Larger units will, of course, have higher hardware costs and may require a larger power house.

6.3.2.2 Draft Tube Piers

The number and size of draft tube piers are determined by the civil design of the power house. Typical designs have from 0 to 2 vertical piers. A few designs also have horizontal piers. The velocities at the inlet to the pier are relatively low but the flow pattern can be complex at off-design conditions - especially for Francis machines.

Rehabilitation Design
For structural reasons elimination of piers in an existing powerhouse would not be practical. The shape of the pier nose should be reviewed using cutting edge CFD analysis tools for possible improvements.

New Unit Design
Eliminating the pier(s) removes a strike source. This would have to be incorporated into the initial powerhouse design and may be impractical for larger draft tubes. Changing the draft tube cross-sectional area will influence the structural requirements. Increasing the height while decreasing the width may allow a pier(s) to be eliminated. See Figure 6.2.2-2 for examples. If the pier(s) cannot be removed, the shape of the pier nose should be reviewed for possible improvements.

6.3.2.3 Runner Cones

In a conventional designs, Francis runner crowns are typically extended beyond the outlet edge of the blades with a conical extension (Figure 10.2-8).

Rehabilitation Design
Further investigation is required to determine if the shape and length of these extensions influence the flow patterns in a manner that affects fish friendliness.

New Unit Design
Same as rehabilitation design.

6.3.2.4 **Inlet Valves**

In a conventional design of a unit having a penstock or inlet pipe, there is usually a valve near the inlet to the spiral case. For low to medium head units these valves have a disk that remains in the water passage when it is in its full-open position. This obstruction in the water passage causes some loss of efficiency and creates some strike probability. High head units tend to use spherical valves that have no obstruction in the water passage when opened.

Rehabilitation Design
Option 1: Use a spherical valve that will have no obstruction in the water passage. This is an expensive solution, but it is the best valve for fish friendliness and efficiency. A spherical valve housing is relatively large. Retrofitting will require a detailed review of the plant layout.

Option 2: For butterfly valves, install trunnion fairings and blend any sharp corners that have an obstruction in the water passage. This is a relatively inexpensive solution. These modifications will reduce the head loss and increase the efficiency of the valve.

New Unit Design
Use of a spherical valve or a ring gate will eliminate the obstruction from the water passageway.

6.3.2.5 **Sharp Corners**

In a conventional design some of the corners at inlet edges are manufactured with sharp corners. The cost to round these corners is not justified from a performance standpoint.

Rehabilitation Design
Rounding of these corners may decrease injury when strike does occur. See Figure 6.2.2-3

New Unit Design
Same as rehabilitation design.

6.3.2.6 **Gate Slots**

Sliding gates are used as closure devices in the draft tube and/or inlet sections of turbines. They ride in slots that are located on the perimeter of the sections. When the units are operating, these gates are always fully opened which exposes the gate slots to the flow. These slots typically have relatively sharp corners and can cause vortices.

Rehabilitation Design
Filling these slots with a removable filler strip will minimize the possibility of a strike. The time needed to install and remove these fillers must be weighed against the frequency of use. See Figure 6.2.2-4.

New Unit Design
In addition to slot filling, rounding the corners of these slots could decrease the probability of injury if a strike occurs.

6.4 AERATING FRANCIS DESIGN FOR INCREASING DISSOLVED OXYGEN CONTENT

This section will be delivered at a later date as a supplement to this report in conjunction with the delivery of a report on the use of advanced CFD and a "virtual fish" to evaluate the four conditions tested experimentally by fish injection at Wanapum dam (described in Section 4.4.6).

6.5 SUMMARY AND CONCLUSIONS

This study was conducted to define three families of environmentally advanced hydro turbine design concepts to improve hydropower's impact on the environment and to improve the understanding of the technical and environmental issues that especially effect fish survival. The study conclusively demonstrates that enhanced environmental designs can improve fish survival during turbine passage. In addition, enhanced design concepts can improve operating efficiency, reduce operations and maintenance expenses, and improve water quality. Based on the results of the studies conducted, the team members feel that the environmentally friendly design features presented can be incorporated into existing and new designs in a cost effective manner. Several rehabilitation projects currently underway already include some of the design concepts presented providing evidence of market acceptance.

To improve the environmental compatibility of hydro projects, existing hydropower turbine designs can be upgraded using some of the design concepts presented. Each plant is unique and the presented design concepts will need to be adapted to the plant requirements and the project constraints. In addition to design improvements, plant operation can be coordinated with the presence of fish to result in fish passage survival improvements. Many of the design concepts can be implemented with a minimal impact on cost when combined with an upgrade initiated to address mechanical problems or to take advantage of efficiency and/or capacity gains arising from improved turbine design technology. Other of the design concepts can be implemented providing further improvements, but with a higher marginal cost. For new turbines, many of the design concepts can be integrated into the design with a minor cost impact.

The implementation of the presented design concepts will have a positive impact on the environment. Each of the concepts will have an incrementally different contribution to the improvements. Estimation of the incremental cost of an improvement can be made. This will be done and be included in the supplemental report which will be issued at a later date. Estimation of the incremental environmental improvement is more difficult to do and despite progress made during this study is subjective. However an attempt to quantify the improvement will be included in the supplemental report also.

Project specific designs may be developed incorporating elements of the three families of design concepts presented. However, more than three families of design concepts are needed to address all of the relevant issues. Design concepts to address minimum stream flows, for example, were not addressed.

7.0 OVERALL CONCLUSIONS

7.0 OVERALL CONCLUSIONS

This study was conducted to define three families of environmentally advanced hydro turbine design concepts to improve hydropower's impact on the environment and to improve the understanding of the technical and environmental issues involved, in particular, on fish survival. The study conclusively demonstrates that enhanced environmental designs can improve fish survival during turbine passage. In addition, enhanced design concepts can improve operating efficiency, reduce operations and maintenance expenses, and improve water quality. Based on the results of the studies conducted, the team members feel that environmentally friendly design features presented can be incorporated into new and existing designs in a cost effective manner. Several rehabilitation projects currently underway already include some of the design concepts presented providing evidence of market acceptance.

Specific results and conclusions are:

- Project specific designs may be developed incorporating elements of the families of design concepts presented which will improve the environmental compatibility of hydropower installations. Concept family one addresses Kaplan turbines for improved fish passage survival. Concept family two addresses Francis turbines. This family of concepts is applicable to many low-head Francis units which are important for fish passage at older projects in the eastern and upper mid west states. Concept family three addresses Francis turbines for improving levels of dissolved oxygen in their discharge. However, more than three families of design concepts are needed to address all of the relevant issues. Design concepts to address minimum stream flows, for example, were not addressed.

- Turbine operation has a significant effect on fish survival during turbine passage. Field test results and CFD analyses were used to characterize the effects of turbine operation. When fish are present, limitation of turbine operation to the range of discharges providing the best survival will improve fish passage survival compared to conventional operation which does not recognize the effect.

- Improvements in the level of understanding of the physics governing fish damage due to leading edge strike were achieved. Revised equations to estimate strike mortality were presented. The concept of blade effective zone (BEZ) was developed to expand the concept of the interaction of fish with the blade and its near by fluid characteristics induced by the blade presence.

- The utility of advanced turbulence models in computational fluid dynamics (CFD) investigations of fish passage issues is demonstrated. More accurate prediction of flow characteristics results.

- Controlled field test experiments and CFD calculations demonstrate that different zones of the turbine have significantly different effects on fish during passage. Zonal geometry and its associated flow conditions are important. In planning tests to evaluate fish passage, zonal effect determination must be considered to adequately develop a survival estimate for the turbine.

- In the absence of cavitation, pressure effects on fish during turbine passage are not significant. Effects related to the state of pressure acclimation are significant. These effects relate more to project planning than to turbine design or operation.

- Incorporating capabilities for aeration in the turbine design can alleviate water quality problems stemming from low dissolved oxygen in hydropower releases. Depending on design conditions, aerating turbines can increase the level of dissolved oxygen by over 5 mg/L.

Furthermore a review of existing test results and testing procedures pointed out that:

- Only a few tests conducted under controlled conditions have provided some understanding of mechanisms of fish injury and mortality and are deemed useful from the standpoint of turbine design modifications. Data from these tests show that fish survival and injury types differ depending on entrainment depths, point of turbine operation, intake configuration (e.g., whether fish guidance screens are present), and site-specific characteristics. However, these test results have also demonstrated large knowledge voids with respect to the actual path of entrained fish in passage through turbines, flow patterns within turbines, etc. Integration of information on these parameters can lead to further refinement of turbine designs.
- Risks of injury and mortality due to mechanical causes, fluid mechanisms (shear-turbulence and cavitation) and pressure changes are not the same at all sites nor at all operating conditions at the same site. At low head dams, mechanically related injuries and mortality predominate while at high head dams, whether equipped with hydro turbines or not, pressure changes can inflict a high level of injury (rupture of air bladder and other internal organs, decompression trauma, etc.) and mortality
- Specific biological criteria for turbine redesign are difficult to develop from the existing database due to lack of replication in experimental data and variability in tests between similar type turbines at a site.
- Most studies have been conducted with small fish (<200 mm (<7.9 in.)), particularly anadromous fishes. Because few survival data exist on large sized fish, either resident or migratory, considerations for turbine design modifications for these fish cannot be fully concluded.
- Recent experimental data and reanalysis of historical data **do not** support certain historical hypotheses. Instead, they show that (1) survival is not necessarily maximized at peak turbine operating efficiency, (2) survival is not necessarily higher for fish entrained near the hub, and (3) survival is not necessarily lower for unguided fish at turbines equipped with fish guidance screens. The report demonstrates that complex interacting mechanisms are in effect within the turbine and the fish passage survival depends of the turbine geometry, its operation and on the fish and its location in the water column.
- None of the passage routes is 100% safe for fish. Survival through alternative passage routes (e.g., sluices, spillway, etc.) at some sites is comparable to that in passage through turbines. Pressure related and impact injuries can occur in passage through these routes depending upon the physical configuration, head, turbulence, characteristics of plunge pool, etc.
- The effectiveness of turbine designs should be evaluated against "best of class" benchmarks. This would help in setting realistic, achievable goals in fish survival improvement for each turbine type. Effects of turbine modifications on fish survival can be evaluated using tests and "comparative" benchmarking.

An objective of the work was to improve the understanding of the technical and environmental issues involved in improving fish passage survival and to point out needs for additional research. While improvements were made, insights developed led to the following conclusions which indicate areas needing additional investigation:

- Fish paths within intakes and turbines are not well understood. The effects of both the dynamics of three dimensional bodies as well as fish volitional movement (FVM) will need to be investigated fully to provide additional improvements in fish survival.

- Additional testing is required to develop accurate indices of forces, pressure differentials, or other deterministic quantities that can be related to fish damage mechanisms in more detail.

- Calculation of flow fields can be performed. However a means of calculating the resulting loads on the fish and the effect of the loads on fish survival is needed to advance the state of the art.

8.0 RECOMMENDATIONS FOR FUTURE TESTING AND INVESTIGATION

8.0 RECOMMENDATIONS FOR FUTURE TESTING AND INVESTIGATION

This work has shown that there are attractive design concepts for improving fish passage survival. The attractiveness has been perceived based on theories derived from a combination of analytic studies (CFD) and knowledge of fluid flow fields and from correlation of the theories with site survival studies. Testing coupled with more research is required to sharpen the theories. The characteristics of fluid and mechanical mechanisms which injured fish need to be quantified. The Advanced Hydro Turbine Program can benefit significantly from additional research and tests that will fill the knowledge voids in the existing data and improve the probability of success for developing further improvements to the presented design and operational concepts and incorporating them into project designs for more fish friendly turbines and hydro plant systems and operating those plants more successfully. Some of these types of experiments are identified below.

- Design and conduct tests to determine the effect of fluid and mechanical mechanisms which may injure fish. In the field, this should include the release of fish into multiple locations (zones) determined to have significant mechanical or fluid mechanisms to cause injury which warrant evaluation. CFD analysis tools and scale model testing should be used to identify the zones within the turbine where the mechanical or fluid mechanisms are significant and quantify the intensity of the mechanism. Areas that are benign and others which are not should be identified. Temporary modifications to turbine component geometry could be made to accommodate testing to separate multiple effects taking place within a zone. As an example, the higher mortality observed at Wanapum dam for fish that were presumed to enter the runner near the hub is presumed to be related to hub gaps, and the hub surface geometry. Wanapum has a three-dimensional localized contour on the hub downstream of the spherical portion of the hub to minimize the gap between the blade inner edge and the hub. These contours are an obstruction to the flow when the blades are not near minimum blade position. Repeating the tests at a blade position with the gaps temporarily closed would provide the insight to the effect related to the gaps alone.

- Design and conduct tests to quantify the effects of fluid shear to obtain sufficient data for 3-D "virtual fish" CFD correlation.

- Test multiple turbine operating points in all field experiments.

- Include a wider range of fish size and species (of interest) in the tests than in the past. Data on larger sized fish (e.g., post-spawned migratory fish, upstream migrant fall backs) are scant. Inclusion of strong and weak swimmers in tests will provide comparative data.

- Conduct a test where fish are injected at a location to enter the runner in the blade periphery zone. Note that most plants have some wicket gate overhang when the blades are at maximum blade position. It may be difficult to separate the effect of blade periphery gap and wicket gate overhang.

- Design and conduct field and laboratory tests subjecting fish to turbine like cavitation mechanisms. Zonal field testing is appropriate, but the zone of cavitation related fluid mechanisms lies in the vicinity of the blade surface and rotates with the blades. In the majority of the space between blades there may not be any cavitation. Therefore zonal testing will not be able to identify which fish get into the cavitation zone and which pass in the non cavitating zone. A laboratory experiment would need to be carefully designed to simulate the characteristics of cavitation and the related fluid effects (bubble sizes compared to fish size, velocity gradients etc.) present in turbines.

- The strike equations predict an increased tendency for strike as the turbine head is decreased. Tests could be designed and conducted to verify this effect.

- Testing designed to clarify a number of unknown factors relating to fish behavior would add to the knowledge base. It would be valuable to know if fish actively swim inside the turbine, and if so, where and why and if in their passage through the higher velocity part of the turbine (through wicket gates and runner blades) they maintain an orientation to the local flow field. How fish respond to changes in the local pressure field is another point for further clarification. Tests to delineate the actual flow path entrained fish traverse in passage from the point of entry into the turbine to the exit point would be instructive. This information will validate or invalidate the assumption that a fish is moving involuntarily with the flow and not "behaving". Release of dead fish (see if injury types are the same) along with the alive fish (same size and species) at the same location may shed some light on this if technology does not exist to track fish passage by unobtrusive means. This would give insight into behavior.

- Controlled tests for turbines equipped with fish guidance screens or bypass structures, and for turbines without guidance screens or bypass are needed to understand the effect of the bypass structure and fish screens on fish redistribution from the intake to the turbine runner.

- Through testing, evaluate synergistic effects of gas supersaturation and turbine passage (mostly a Pacific Northwest concern).

- Conduct field tests to quantify spatial distribution and orientation of fish in the water column prior to entry into the turbine intakes.

- Additional testing of the pressure reduction effect is desired to understand the 100% variation in mortality that has occurred in different investigations.

- Evaluate civil engineering design modifications of plant intakes and draft tubes for pressure and boil effects. This could be done by numerical investigation and be complimented by laboratory tests and field testing.

9.0 SUPPLEMENTAL REPORT

9.0 **SUPPLEMENTAL REPORT**

This section is reserved for a supplemental report covering the Task 4 family of design concepts relating to aerating Francis design for increasing oxygen content (Section 6.4) and a report on the use of advanced CFD and a "virtual fish" to evaluate the four conditions tested experimentally by fish injection at Wanapum Dam (described in Section 4.4.6).

10.0 APPENDICES

APPENDIX 10.1

COMPILATION OF SURVIVAL DATA

Table 10.1-1

Comparison of physical and hydraulic characteristics of low-head hydroelectric dams equipped with propeller/Kaplan type turbines at which turbine passage survival was estimated.

Station	Designed Turbine Discharge (cms)	Number of Blades	Runner Speed (rpm)	Head (m)	Runner Diameter (m)	Peripheral Velocity (m/s)
Annapolis, NS	404.6	-	50.0	6.7	6.78	17.7
Big Cliff, OR	71.1	6	163.6	27.7	3.76	32.2
Bonneville, OR-WA	498.4	5	69.2	18.3	8.38	30.4
Chalk Hill, MI-WI	37.7	4	150.0	8.8	2.59	20.3
Conowingo, MD	283.2	6	120.0	27.4	5.72	35.9
Craggy Dam, NC	17.0	4	229.0	6.4	1.75	21.0
Crescent, NY	43.0	5	144.0	8.2	2.74	20.7
Feeder Dam, NY	29.5	6	120.0	4.7	2.92	18.3
Foster Dam, OR	22.7	6	257.0	33.5	2.54	34.2
Fourth Lake, NS	15.0	6	360.0	22.9	-	NA
Greenup Dam, OH (Vanceburg)	336.7	5	-	9.1	7.19	NA
Hadley Falls, MA	118.9	5	128.0	15.8	4.32	28.9
Herrings, NY	34.0	4	138.0	5.8	2.87	20.7
Kleber Dam, MI	5.7	-	450.0	13.4	-	NA
la centrale Beauhamois, Quebec	262.7	6	94.7	24.1	6.32	31.3
Lawrence, MA (Essex)	124.6	3	128.6	8.8	4.00	26.9
Little Goose, WA	509.8	6	90.0	28.3	7.92	37.3
Lowell, MA	127.4	5	120.0	11.9	3.86	24.2
Lower Granite, WA	538.1	6	90.0	29.9	7.92	37.3
Lower Monumental, WA	509.8	6	90.0	28.7	7.92	37.3
Marshall, NC	35.4	4	212.0	9.4	3.78	42.0
McNary, WA	348.3	6	87.5	24.4	7.11	32.6
Morrow, MI	6.7	-	175.0	3.7	1.37	12.6
Racine, WI	226.6	4	62.1	6.7	7.71	25.1
Raymondville, NY	46.4	6	120.0	6.4	3.33	20.9
Rock Island, WA (Powerhouse 1)	498.4	6	100.0	13.7	7.01	36.7
Rock Island, WA (Powerhouse 2)	509.8	4	85.7	12.2	7.01	31.4
Rocky Reach, WA (Unit 3)	453.1	6	90.0	28.0	7.11	33.5
Rocky Reach, WA (Unit 5)	453.1	6	90.0	28.0	7.11	33.5
Rocky Reach, WA (Unit 6)	453.1	6	90.0	28.0	7.11	33.5
Rocky Reach, WA (Unit 8)	594.7	5	85.7	26.4	7.90	35.4
Safe Harbor, PA (Unit 7)	235.1	5	109.0	16.8	5.64	32.2
Safe Harbor, PA (Unit 8)	260.5	7	75.0	16.8	6.15	24.1
T. W. Sullivan, OR	11.0	6	240.0	12.8	1.78	22.3
Thornapple, WI	19.8	6	120.0	4.9	1.09	6.9
Townsend Dam, PA	42.5	3	152.0	4.9	2.87	22.8
Tusket, NS	-	-	225.0	8.2	-	NA
Twin Branch, IN	11.6	4	241.0	6.5	-	NA
Walterville, OR	70.8	5	-	16.8	3.07	NA
Wanapum, WA	481.5	5	85.7	24.4	7.24	32.5
Wells, WA	566.4	6	85.7	19.8	7.43	33.3
West Enfield, ME	150.1	3	89.0	6.4	4.88	22.7
Wilder, VT-NH	127.4	5	112.5	15.5	4.57	26.9
Range	*5.7-595*	*3-7*	*50-450*	*3.7-33.5*	*1.09-8.38*	*6.9-42*

Table 10.1-2

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test Sample Size	Control Sample Size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Control Survival (%)	Percent Recapture Test	Percent Recapture Control	% Survival 1 hr	Source
Annapolis, NS	Radio telemetry	American shad	20	29	-	404.6	-	50	6.7	6.78	NA	NA	NA	53.7	Hogan (1986)
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	3,500	3,500	100	52.5	6	163.6	27.7	3.76	NA	98.1	97.0	91.1	Oligher & Donaldson (1966)
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	2,750	2,750	100	71.1	6	163.6	24.7	3.76	NA	98.1	97.0	94.5	Oligher & Donaldson (1966)
Big Cliff, OR (1964)	Full discharge netting	Chinook Salmon	3,500	3,500	100	71.1	6	163.6	21.6	3.76	NA	98.1	97.0	89.7	Oligher & Donaldson (1966)
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	2,750	2,750	100	52.5	6	163.6	27.7	3.76	NA	93.2	98.9	92.2	Oligher & Donaldson (1966)
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	3,750	3,750	100	71.1	6	163.6	24.7	3.76	NA	93.2	98.9	89.8	Oligher & Donaldson (1966)
Big Cliff, OR (1966)	Full discharge netting	Chinook Salmon	2,500	2,500	100	71.1	6	163.6	21.6	3.76	NA	93.2	98.9	90.6	Oligher & Donaldson (1966)
Big Cliff, OR (1967)	Full discharge netting	Steelhead	-	-	152	71.1	6	163.6	21.6	3.76	NA	-	-	90.4	Oligher & Donaldson (1966)
Bonneville, OR/WA	Brand, CWT, Seine	Chinook salmon	850,406	435,099	91	498.4	5	69.2	18.3	8.38	NA	<1.0	<1.0	97.5	EPRI (1992)
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	Bluegill	60	43	103	37.7	4	150	8.8	2.59	95.3	86.7	97.7	97.0	RMC (1994)
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	Bluegill	50	67	153	37.7	4	150	8.8	2.59	95.5	94.0	97.0	98.0	RMC (1994)
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	W. Sucker/R. Trout	77	70	119	37.7	4	150	8.8	2.59	94.3	80.5	94.3	91.0	RMC (1994)
Chalk Hill, MI-WI	Hi-Z Turb'N Tag	W. Sucker/R. Trout	38	45	261	37.7	4	150	8.8	2.59	100.0	97.4	100.0	97.0	RMC (1994)
Conowingo, MD	Hi-Z Turb'N Tag	American Shad	108	108	125	226.6	6	120	27.4	5.72	91.7	88.0	97.6	94.9	RMC (1994)
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	43	28	180	17.0	4	229	6.4	1.75	100.0	93.0	100.0	93.0	Mathur et al. (1993)
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	63	28	180	5.7	4	229	6.4	1.75	100.0	90.0	100.0	90.0	Mathur et al. (1993)
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	39	22	277	5.7	4	229	6.4	1.75	100.0	90.0	100.0	81.0	Mathur et al. (1993)
Craggy Dam, NC	Hi-Z Turb'N Tag	Bluegill	33	40	100	5.7	4	229	6.4	1.75	90.0	85.0	90.0	96.0	Mathur et al. (1993)
Craggy Dam, NC	Hi-Z Turb'N Tag	Channel Catfish	32	22	277	17.0	4	229	6.4	1.75	100.0	88.0	100.0	93.0	Mathur et al. (1993)
Craggy Dam, NC	Hi-Z Turb'N Tag	Bluegill	72	54	155	5.7	4	229	6.4	1.75	96.0	90.0	96.0	86.0	Mathur et al. (1993)
Crescent, NY	Hi-Z Turb'N Tag	Blueback Herring	125	125	91	43.0	5	144	8.2	2.74	90.0	84.0	86.0	96.0	Mathur et al. (1996)
Essex, MA (bulb turbine)	Radio telemetry	Atlantic Salmon	50	0	288	124.6	3	128.6	8.8	4.00	NA	50.0	NA	98.0	Knight (1982)
Foster, OR (tests combined)	Full discharge netting	Chinook Salmon	-	-	120	22.7	6	257	30.8	2.54	NA	NA	NA	82.1	Bell (1981)
Foster, OR (tests combined)	Full discharge netting	Chinook Salmon	-	-	120	22.7	6	257	33.5	2.54	NA	NA	NA	93.9	Bell (1981)
Foster, OR (tests combined)	Full discharge netting	Chinook Salmon	-	-	120	22.7	6	257	33.5	2.54	NA	NA	NA	88.8	Bell (1981)
Feeder Dam, NY	Full discharge netting	Bluegill	-	-	91.6	29.5	6	120	4.7	2.92	100.0	-	-	97.3	Acres (1995)
Feeder Dam, NY	Full discharge netting	Bluegill	-	-	128.6	29.5	6	120	5.2	2.92	97.7	-	-	92.3	Acres (1995)
Feeder Dam, NY	Full discharge netting	Largemouth bass	-	-	87.7	29.5	6	120	5.5	2.92	90.1	-	-	98.0	Acres (1995)
Feeder Dam, NY	Full discharge netting	Largemouth bass	-	-	190	29.5	6	120	5.8	2.92	96.3	-	-	90.0	Acres (1995)
Feeder Dam, NY	Full discharge netting	Largemouth bass	-	-	292.1	29.5	6	120	6.1	2.92	99.2	-	-	86.8	Acres (1995)
Feeder Dam, NY	Full discharge netting	Brown trout	-	-	205.5	29.5	6	120	6.4	2.92	93.1	80.5	93.1	86.4	Acres (1995)
Feeder Dam, NY	Full discharge netting	Golden shiner	-	-	88	29.5	6	120	6.7	2.92	95.0	92.8	95.8	96.8	Acres (1995)
Fourth Lake, NS	Full dschrg/dye or brand	Atlantic Salmon	503	494	163	15.0	6	360	22.9	-	99.4	74.4	74.3	83.7	Ruggles et al. (1990)
Fourth Lake, NS	Full dschrg/dye or brand	Brook trout	1,908	NA	105.5	15.0	6	360	22.9	-	96.5	24.5	-	87.1	Ruggles et al. (1990)
Fourth Lake, NS	Full dschrg/dye or brand	Alewife	675	627	96	15.0	6	360	22.9	-	83.1	70.8	83.1	89.0	Ruggles et al. (1990)

Table 10.1-2

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test Sample Size	Control Sample Size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Control Survival (%)	Percent Recapture		% Survival 1 hr	Source
												Test	Control		
Greenup Dam, OH (Vanceburg)	Radio telemetry	Sauger	48	NA	231	336.1	5	90	9.1	6.10	NA	85.4	NA	85.4	Olson (1990)
Hadley Falls, MA	Radio telemetry	American Shad	36	69	560	118.9	5	128	15.6	4.32	98.6	-	-	78.2	Bell and Kynard (1988)
Hadley Falls, MA	HI-Z Turb'N Tag	American Shad	100	·100	82	118.9	5	128	15.8	4.32	75.0	76.0	76.0	97.3	Mathur et al. (1994)
Hadley Falls, MA	Radio telemetry	Atlantic Salmon	108	89	285	118.9	5	128	15.8	4.32	92.5	100.0	100.0	93.7	Kynard et al. (1982)
Hadley Falls, MA	HI-Z Turb'N Tag	American Shad	100	100	82	43.9	5	128	15.8	4.32	77.0	81.0	78.0	100.0	Mathur et al. (1994)
Herrings, NY	Full discharge netting	Centrarchid	74	65	100	34.0	4	138	5.8	2.87	98.3	74.3	90.8	98.3	KA (1996)
Herrings, NY	Full discharge netting	Centrarchid	77	63	175	34.0	4	138	5.8	2.87	100.0	96.0	100.0	97.3	KA (1996)
Herrings, NY	Full discharge netting	Centrarchid	80	65	250	34.0	4	138	5.8	2.87	100.0	91.3	70.8	93.2	KA (1996)
Herrings, NY	Full discharge netting	Percid	46	51	100	34.0	4	138	5.8	2.87	99.1	84.8	88.2	91.1	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	31	57	100	34.0	4	138	5.8	2.87	100.0	32.3	22.8	90.0	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	74	63	175	34.0	4	138	5.8	2.87	100.0	32.4	1.6	87.5	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	82	72	250	34.0	4	138	5.8	2.87	-	96.2	0.0	96.2	KA (1996)
Herrings, NY	Full discharge netting	Centrarchid	90	65	100	34.0	4	138	5.8	2.87	98.2	98.7	95.4	95.0	KA (1996)
Herrings, NY	Full discharge netting	Centrarchid	90	69	175	34.0	4	138	5.8	2.87	100.0	92.2	97.1	96.4	KA (1996)
Herrings, NY	Full discharge netting	Centrarchid	90	77	250	34.0	4	138	5.8	2.87	100.0	88.9	97.4	92.5	KA (1996)
Herrings, NY	Full discharge netting	Percid	185	78	100	34.0	4	138	5.8	2.87	100.0	83.8	84.6	94.9	KA (1996)
Herrings, NY	Full discharge netting	Percid	179	139	175	34.0	4	138	5.8	2.87	100.0	91.1	94.2	98.2	KA (1996)
Herrings, NY	Full discharge netting	Percid	138	137	250	34.0	4	138	5.8	2.87	100.0	84.8	94.2	96.2	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	91	74	100	34.0	4	138	5.8	2.87	100.0	24.2	18.9	95.5	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	95	72	175	34.0	4	138	5.8	2.87	100.0	78.9	73.6	98.7	KA (1996)
Herrings, NY	Full discharge netting	Salmonids	111	77	250	34.0	4	138	5.8	2.87	100.0	64.0	72.7	98.6	KA (1996)
Herrings, NY	Full discharge netting	Soft ray	188	144	100	34.0	4	138	5.8	2.87	100.0	63.3	85.4	97.5	KA (1996)
Herrings, NY	Full discharge netting	Soft ray	201	159	175	34.0	4	138	5.8	2.87	100.0	74.1	94.7	91.7	KA (1996)
Herrings, NY	Full discharge netting	Soft ray	175	125	250	34.0	4	138	5.8	2.87	100.0	95.4	99.2	85.1	KA (1996)
Herrings, NY	Full discharge netting	Clupeids	196	166	100	34.0	4	138	5.8	2.87	100.0	90.3	90.4	92.8	KA (1996)
Kleber Dam, MI	Full discharge netting	Mixed resident fish	-	-	Adults	5.7	-	450	13.4	-	-	-	-	59.0	EPRI (1992)
la centrale de Beauharnois, Quebec, Canada	Float tag	American eel	122	-	881	262.7	6	94.7	24.1	6.32	NA	95.9	-	76.1	Desrochers (1995)
Little Goose, WA	PIT tag	Chinook salmon	-	-	-	509.8	6	90	28.3	7.92	-	-	-	92.0	Muir et al. (1995)
Lowell, MA	Radio telemetry	Atlantic Salmon	50	0	265	127.4	5	120	11.0	3.86	NA	100.0	NA	88.5	Nelson et al. (1989)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	820	821	134	594.7	6	90	29.9	7.92	97.8	94.5	98.8	94.6	RMC et al. (1994)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	320	320	151	509.8	6	90	29.9	7.92	98.4	96.8	98.7	94.9	Normandeau et al. (1995)
Lower Granite, WA	PIT tagging	Chinook Salmon	3,200	1,600	151	509.8	6	90	29.9	7.92	NA	-	-	92.7	Muir et al. (1995)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	320	320	150	509.8	6	90	29.9	7.92	98.4	96.6	98.7	95.3	Normandeau et al. (1995)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	250	250	148	382.3	6	90	29.9	7.92	98.4	96.4	99.6	97.2	Normandeau et al. (1995)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	300	300	148	538.1	6	90	29.9	7.92	99.3	96.7	99.3	94.6	Normandeau et al. (1995)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	250	250	151	509.8	6	90	29.9	7.92	99.6	98.1	98.1	97.5	Normandeau et al. (1995)
Lower Granite, WA	HI-Z Turb'N Tag	Chinook Salmon	320	320	150	509.8	6	90	29.9	7.92	98.1	98.2	98.1	97.5	Normandeau et al. (1995)

Table 10.1-2

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test Sample Size	Control Sample Size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Control Survival (%)	Percent Recapture		% Survival 1 hr	Source
												Test	Control		
Lower Monumental, WA	PIT tag	Chinook Salmon	-	-	-	509.8	6	90	28.7	7.92	-	-	-	86.5	Muir et al. (1995)
Marshall, NC	Partial netting	Resident spp.	2,544	2,544	-	35.4	4	212	9.6	3.79	-	-	39.0	92.3	EPRI (1992)
McNary, WA	Brand/partial netting	Chinook salmon	120,000	120,000	52	348.3	6	82.5	24.4	7.11	-	<5.0	<5.0	89.0	Schoeneman et al. (1961)
Morrow, MI	Full discharge netting	Brown Bullhead	117	39	Adult	6.7	-	175	3.7	1.37	100.0	75.2	84.6	97.0	EPRI (1992)
Morrow, MI	Full discharge netting	Pumpkinseed	88	22	Adult\YOY	6.7	-	175	3.7	1.37	100.0	86.4	100.0	90.0	EPRI (1992)
Morrow, MI	Full discharge netting	Black Crappie	90	33	Adult\YOY	6.7	-	175	3.7	1.37	93.0	67.8	90.9	74.0	EPRI (1992)
Morrow, MI	Full discharge netting	White Sucker	64	29	Adult\YOY	6.7	-	175	3.7	1.37	100.0	79.7	100.0	67.0	EPRI (1992)
Morrow, MI	Full discharge netting	Yellow Perch	39	5	Adult	6.7	-	175	3.7	1.37	100.0	82.1	100.0	78.0	EPRI (1992)
Morrow, MI	Full discharge netting	Redhorse	31	10	Adult	6.7	-	175	3.7	1.37	100.0	87.1	100.0	71.0	EPRI (1992)
Morrow, MI	Full discharge netting	Largemouth Bass	24	5	Adult	6.7	-	175	3.7	1.37	100.0	87.5	100.0	81.0	EPRI (1992)
Morrow, MI	Full discharge netting	Northern Pike	21	1	Adult	6.7	-	175	3.7	1.37	0.0	95.2	100.0	45.0	EPRI (1992)
Morrow, MI	Full discharge netting	Yellow Bullhead	39	5	Adult	6.7	-	175	3.7	1.37	100.0	82.1	100.0	92.0	EPRI (1992)
Racine, WI	Partial netting	Gizzard shad	-	-	-	226.6	4	62.1	6.7	7.71	NA	-	-	93.5*	EPRI (1992)
Racine, WI	Partial netting	Drum	-	-	-	226.6	4	62.1	6.7	7.71	NA	-	-	94.0*	EPRI (1992)
Racine, WI	Partial netting	Game species	-	-	-	226.6	4	62.1	6.7	7.71	NA	-	-	94.0*	EPRI (1992)
Raymondville, NY	Full discharge netting	Eel	-	-	625	46.4	6	120	6.4	3.33	-	85.0	90.0	63.0	KA (1996)
Rock Island, WA (bulb turbine)	Brand/partial netting	Coho salmon	203,338	203,843	115	509.8	4	85.7	12.2	7.01	NA	18.4	19.5	93.0	Olson & Kaczynski (1980)
Rock Island, WA (bulb turbine)	Brand/partial netting	Steelhead	58,571	57,864	166	509.8	4	85.7	12.2	7.01	NA	17.9	18.5	96.9	Olson & Kaczynski (1980)
Rock Island, WA (bulb turbine)	Hi-Z Turb'N Tag	Chinook Salmon	280	140	162	4984.4	4	85.7	12.2	7.01	100.0	100.0	100.0	96.1	Normandeau & Skalski (1997)
Rock Island, WA (PH 1, Unit 4)	Hi-Z Turb'N Tag	Chinook Salmon	280	140	162	481.5	6	100	13.7	5.74	100.0	100.0	100.0	95.0	Normandeau & Skalski (1997)
Rock Island, WA (PH 1, Unit 6)	Hi-Z Turb'N Tag	Chinook Salmon	280	140	162	481.5	6	100	13.7	5.74	100.0	100.0	100.0	96.1	Normandeau & Skalski (1997)
Rocky Reach, WA (30',U. 3)	Hi-Z Turb'N Tag	Chinook Salmon	250	250	161	453.1	6	90	28.0	7.11	98.9	96.4	98.8	94.7	Mathur et al. (1996)
Rocky Reach, WA (10',U. 3)	Hi-Z Turb'N Tag	Chinook Salmon	350	350	161	453.1	6	90	28.0	7.11	98.9	95.0	96.0	93.9	Mathur et al. (1996)
Rocky Reach, WA (10',U. 5)	Hi-Z Turb'N Tag	Chinook Salmon	235	300	184	396.5	6	90	28.0	7.11	99.0	98.3	99.0	97.3	Normandeau & Skalski (1996)
Rocky Reach, WA (30',U. 5)	Hi-Z Turb'N Tag	Chinook Salmon	241	220	184	396.5	6	90	28.0	7.11	97.3	98.3	97.7	94.4	Normandeau & Skalski (1996)
Rocky Reach, WA (10',U. 6)	Hi-Z Turb'N Tag	Chinook Salmon	420	300	184	396.5	6	90	28.0	7.11	99.0	97.6	99.0	94.2	Normandeau & Skalski (1996)
Rocky Reach, WA (30',U. 6)	Hi-Z Turb'N Tag	Chinook Salmon	235	220	184	396.5	6	90	28.0	7.11	97.3	97.1	97.7	95.8	Normandeau & Skalski (1996)
Safe Harbor, PA (Unit 7)	Hi-Z Turb'N Tag	American Shad	100	100	118	235.1	5	109	16.8	5.64	99.0	99.0	99.0	98.0	Heisey et al. (1992)
Thornapple, WI	Full discharge netting	Indigenous spp.	3,378	-	-	19.8	6	120	4.6	2.79	-	-	-	95.3	EPRI (1992)
Thornapple, WI	Full discharge netting	Bullheads/Catfish	-	-	-	-	6	120	-	2.79	-	-	-	91.9	EPRI (1992)
Thornapple, WI	Full discharge netting	Suckers/Redhorse	-	-	-	-	6	120	-	2.79	-	-	-	93.4	EPRI (1992)
Thornapple, WI	Full discharge netting	Panfish/Y. Perch	-	-	-	-	6	120	-	2.79	-	-	-	93.5	EPRI (1992)
Thornapple, WI	Full discharge netting	N. Pike/Muskellunge	-	-	-	-	6	120	-	2.79	-	-	-	94.1	EPRI (1992)
Thornapple, WI	Full discharge netting	Burbot	-	-	-	-	6	120	-	2.79	-	-	-	96.9	EPRI (1992)

Table 10.1-2

Physical and hydraulic characteristics of all hydroelectric dams equipped with Kaplan type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test Sample Size	Control Sample Size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Control Survival (%)	Percent Recapture		% Survival 1 hr	Source
												Test	Control		
Thornapple, WI	Full discharge netting	Mnnw/Dace/Drtr	-	-	-	-	6	120	-	2.79	-	-	-	97.1	EPRI (1992)
Thornapple, WI	Full discharge netting	Sm/Lgmth Bass	-	-	-	-	6	120	-	2.79	-	-	-	97.4	EPRI (1992)
Thornapple, WI	Full discharge netting	Walleye	-	-	-	-	6	120	-	2.79	-	-	-	97.6	EPRI (1992)
Townsend Dam, PA	HI-Z Turb'N Tag	Largemouth Bass	31	NA	217	42.5	3	152	4.9	2.87	NA	96.8	NA	96.8	RMC (1994)
Townsend Dam, PA	HI-Z Turb'N Tag	Rainbow Trout	54	52	139	22.7	3	152	4.9	2.87	100.0	96.3	100.0	94.4	RMC (1994)
Townsend Dam, PA	HI-Z Turb'N Tag	Rainbow Trout	52	51	344	22.7	3	152	4.9	2.87	100.0	92.3	94.1	86.5	RMC (1994)
Townsend Dam, PA	HI-Z Turb'N Tag	Largemouth Bass	51	50	102	22.7	3	152	4.9	2.87	98.0	98.0	98.0	100.0	RMC (1994)
Townsend Dam, PA	HI-Z Turb'N Tag	Largemouth Bass	50	50	217	22.7	3	152	4.9	2.87	100.0	100.0	100.0	86.0	RMC (1994)
Townsend Dam, PA	HI-Z Turb'N Tag	Rainbow Trout	21	NA	139	42.5	3	152	4.9	2.87	NA	100.0	NA	100.0	RMC (1994)
Tusket, NS	Draft tube net	Atlantic salmon	-	-	-	-	-	225	8.2	-	-	-	-	84.5	Ruggles et al. (1990)
Twin Branch, IN	Full discharge netting	Steelhead Trout	300	300	186	11.6	4	241	6.5	NA	-	65.0	79.7	93.2	RMC (1994)
Twin Branch, IN	Full discharge netting	Chinook Salmon	600	450	121	11.6	4	241	6.5	NA	-	97.5	99.3	99.3	RMC (1994)
Twin Branch, IN	Full discharge netting	Bluegill	300	300	126	11.6	4	241	6.5	NA	-	73.0	57.7	94.7	RMC (1994)
T. W. Sullivan, OR (Unit 7)	Full discharge netting	Steelhead	1,800	500	128	11.0	6	240	12.8	1.78	92.8	6.6	44.6	92.3	Massen (1967)
T. W. Sullivan, OR (Unit 7)	Full discharge netting	Chinook salmon	1,800	500	112	11.0	6	240	12.8	1.78	97.3	23.6	89.6	88.2	Massen (1967)
T. W. Sullivan, OR (Unit 8)	Full discharge netting	Steelhead	1,800	500	128	7.4	6	240	12.8	1.78	99.0	43.2	97.0	90.1	Massen (1967)
T. W. Sullivan, OR (Unit 8)	Full discharge netting	Chinook salmon	1,800	500	112	7.4	6	240	12.8	1.78	98.4	61.1	100.0	89.5	Massen (1967)
Walterville, OR(61% wckt)	Brand, full dschrg netting	Rainbow Trout	991	631	fingerling	56.9	-	-	16.8	3.05	-	63.0	94.9	97.5	Eicher Associates (1987)
Walterville, OR(77% wckt)	Brand, full dschrg netting	Rainbow Trout	991	631	fingerling	56.9	-	-	16.8	3.05	-	36.4	68.3	92.5	Eicher Associates (1987)
Walterville, OR	Brand,dwnstr bypass trap	Chinook Salmon	30,000	30,000	135	56.6	-	-	16.8	3.05	-	-	-	87.0	Eicher Associates (1987)
Wanapum, WA (10ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	158	160	154	254.9	5	85.7	22.9	7.24	98.8	92.4	98.8	89.7	Normandeau et al. (1996)
Wanapum, WA (10ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	311.5	5	85.7	22.9	7.24	96.9	93.1	96.9	92.4	Normandeau et al. (1996)
Wanapum, WA (10ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	424.8	5	85.7	22.9	7.24	97.5	93.8	97.5	94.8	Normandeau et al. (1996)
Wanapum, WA (10ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	481.5	5	85.7	22.9	7.24	99.4	88.2	99.4	88.5	Normandeau et al. (1996)
Wanapum, WA (30ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	254.9	5	85.7	22.9	7.24	98.8	95.7	98.8	94.9	Normandeau et al. (1996)
Wanapum, WA (30ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	311.5	5	85.7	22.9	7.24	96.9	95.6	96.9	96.8	Normandeau et al. (1996)
Wanapum, WA (30ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	424.8	5	85.7	22.9	7.24	97.4	98.1	97.4	100.0	Normandeau et al. (1996)
Wanapum, WA (30ft, Unit 9)	HI-Z Turb'N Tag	Coho Salmon	160	160	154	481.5	5	85.7	22.9	7.24	97.4	96.2	99.4	96.8	Normandeau et al. (1996)
Wells, WA(Unit 1)	Brand, Partial netting	Steelhead	-	-	smolts	566.4	6	85.7	19.8	7.43	NA	-	-	84.0	Parametrix (1986)
West Enfield, ME	Radio telemetry	Atlantic Salmon	148	NA	212	150.1	3	89	6.4	4.88	NA	100.0	NA	96.0	Shepard (1988)
Wilder, VT-NH	HI-Z Turb'N Tag	Atlantic Salmon	125	125	191	127.4	5	112.5	15.5	4.57	100.0	99.2	100.0	96.0	RMC (1994)

NA = Not Available, Not Applicable
NE = Naturally Entrained
* Agency agreed upon estimates
** Release numbers unavailable at present but the study appears valid and meets the screening criteria. When release numbers become available they will be included

Table 10.1-3

Physical and hydraulic characteristics of all hydroelectric dams equipped with propeller type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test Sample Size	Control Sample Size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Blades	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Control Survival (%)	Percent Recapture		Est. Percent Survival	Source
												Test	Control		
Hadley Falls, MA	HI-Z Turb'N Tag	American Shad	120	120	82	118.9	5	150	15.8	4.32	83.3	74.2	83.3	89.1	RMC (1992)
Rocky Reach, WA (10',U. 8)	HI-Z Turb'N Tag	Chinook Salmon	265	265	114	566.4	5	85.7	26.4	7.90	88.7	85.7	88.7	96.9	RMC & Skalski (1994)
Safe Harbor, PA (Unit 8)	HI-Z Turb'N Tag	American Shad	100	100	118	260.5	7	75	16.8	6.15	92.0	92.0	92.0	97.8	Helsey et al. (1992)
Safe Harbor, PA (Unit 8)	HI-Z Turb'N Tag	American Shad	100	100	118	260.5	7	75	16.8	6.15	98.0	96.0	98.0	98.9	Helsey et al. (1992)

Table 10.1-4

Comparison of physical and hydraulic characteristics of hydroelectric dams equipped with Francis type turbines.

Station	Designed Turbine Flow (cms)	Number of Buckets	Runner Speed (rpm)	Head (m)	Runner Diameter (m)	Peripheral Velocity (m/s)
Alcona, MI	17.4	16	90	13.1	2.54	12.0
Alcona, MI	47.0	16	90	13.1	2.54	12.0
Baker, WA	-	-	300	76.2	1.52	23.9
Bond Falls, MI	12.7	16	300	64.0	1.84	28.9
Buchanan, MI	2.8	-	-	-	-	NA
Caldron Falls, WI (Unit 1)	18.4	15	226	24.4	1.83	21.6
Centralia, WI (Unit 1)	14.4	15	-	6.1	-	NA
Centralia, WI (Unit 2)	14.4	15	90	6.1	0.71	3.3
Colton, NY	14.1	19	360	80.8	1.50	28.2
Crown Zellerback, OR	11.5	-	277	11.9	-	NA
Cushman Plant 2, WA (1960)	22.7	17	300	137.2	2.11	33.1
Cushman Plant 2, WA	22.7	17	300	137.2	2.11	33.1
E. J. West, NY	76.5	15	113	19.2	3.33	19.7
Elwha, WA	14.2	-	300	31.7	1.49	23.4
Faraday,OR	-	-	360	36.6	1.01	19.0
Finch Pruyn, NY (Unit 4)	20.1	15	225	14.0	1.04	12.3
Finch Pruyn, NY (Unit 5)	23.7	15	225	14.0	1.04	12.3
Five Channels, MI	19.1	16	150	11.0	1.40	11.0
Five Channels, MI	33.1	16	150	11.0	1.40	11.0
Glines, WA	44.7	-	225	59.1	2.35	27.6
Grand Rapids, WI (U 1,2,4 comb)	18.3	15	90	8.5	1.47	6.9
Grand Rapids, WI (Unit 2)	18.3	15	150	8.5	1.47	11.6
Grand Rapids, WI (Unit 4)	26.2	6	180	8.5	1.83	17.2
Hardy, MI (Unit 2)	14.4	16	163.6	30.5	2.13	18.2
High Falls, WI (Unit 5)	7.8	12	358	25.3	0.99	18.6
Highley, NY	19.1	13	257	14.0	1.22	16.4
Hoist, MI	8.5	19	360	43.3	1.83	34.5
Holtwood, PA	99.1	16	95	16.8	4.17	20.7
Holtwood, PA (U3/double runner)	99.1	17	102.8	18.9	2.84	15.3
Holtwood, PA(U10/single runner)	99.1	16	94.7	18.9	3.80	18.8
Leaburg, OR	31.2	-	225	27.1	2.29	26.9
Lequille, NS	9.9	13	519	118.0	1.37	37.3
Luray, VA	10.5	12	164	5.5	1.59	13.7
McClure, MI	4.4	-	600	129.2	-	NA
Minetto, NY	42.5	16	72	5.2	3.53	13.3
North Fork, OR	70.8	-	139	41.5	2.96	21.5
Peshtigo, WI (Unit 4)	13.0	15	100	4.0	2.03	10.6
Potato Rapids, WI (Unit 1)	14.2	15	123	5.2	2.13	13.7
Potato Rapids, WI (Unit 2)	12.5	15	135	5.2	2.03	14.4
Pricket, MI	9.2	15	257	16.5	1.36	18.3
Publishers, OR	7.9	-	300	12.8	0.91	14.4
Puntledge, BC	-	-	277	103.6	2.16	31.4
Rogers, MI (units 1 & 2)	10.8	15	150	11.9	1.52	12.0
Ruskin, BC	113.3	-	120	39.6	3.78	23.8
Sandstone Rapids,WI	18.4	15	150	12.8	2.21	17.3
Schaghticoke, NY (Unit 4)	11.6	17	300	46.6	2.03	31.9
Seton Creek, BC	127.4	-	120	45.7	2.90	18.2
Shasta, CA	90.6	15	138.5	115.8	4.67	33.9
Shasta, CA	90.6	15	138.5	-	4.67	33.9
Stevens Creek, SC	28.3	14	75	8.5	3.43	13.5
T. W. Sullivan, OR	11.8	-	240	12.8	1.88	23.6
Vernon, VT/NH	51.9	15	74	10.4	3.96	15.3
White Rapids, WI	43.6	14	100	8.8	3.40	17.8
Youghiogheny, PA	21.2	-	-	36.6	-	NA

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture		% Survival 1 hr	Source
											Test	Control		
Alcona, MI	Full dschrg netting	Bluegill	97	-	118	47.0	16	90	13.1	2.54	97.0	-	90.2	LMS (1991)
Alcona, MI	Full dschrg netting	Bluegill	102	-	170	47.0	16	90	13.1	2.54	86.0	-	84.1	LMS (1991)
Alcona, MI	Full dschrg netting	Gold./Common Shiner	51	-	114	47.1	16	90	13.1	2.54	96.0	-	80.9	LMS (1991)
Alcona, MI	Full dschrg netting	Gold./Common Shiner	58	-	154	47.1	16	90	13.1	2.54	90.0	-	84.7	LMS (1991)
Alcona, MI	Full dschrg netting	Grass Pickerel	30	-	235	47.1	16	90	13.1	2.54	100.0	-	86.7	LMS (1991)
Alcona, MI	Full dschrg netting	Northern Pike	44	-	352	47.2	16	90	13.1	2.54	98.0	-	51.2	LMS (1991)
Alcona, MI	Full dschrg netting	Rainbow Trout	40	-	108	47.2	16	90	13.1	2.54	70.0	-	100	LMS (1991)
Alcona, MI	Full dschrg netting	Rainbow Trout	40	-	317	47.2	16	90	13.1	2.54	70.0	-	89.4	LMS (1991)
Alcona, MI	Full dschrg netting	Spottail Shiner	40	-	116	47.2	16	90	13.1	2.54	88.0	-	59.5	LMS (1991)
Alcona, MI	Full dschrg netting	Walleye	47	-	162	47.3	16	90	13.1	2.54	100.0	-	16.4	LMS (1991)
Alcona, MI	Full dschrg netting	Walleye	45	-	385	47.3	16	90	13.1	2.54	100.0	-	38.7	LMS (1991)
Alcona, MI	Full dschrg netting	White Sucker	60	-	180	47.3	16	90	13.1	2.54	100.0	-	94.4	LMS (1991)
Alcona, MI	Full dschrg netting	White Sucker	54	-	290	47.4	16	90	13.1	2.54	100.0	-	90.4	LMS (1991)
Alcona, MI	Full dschrg netting	Yellow Perch	55	-	107	47.4	16	90	13.1	2.54	100.0	-	65.1	LMS (1991)
Alcona, MI	Full dschrg netting	Yellow Perch	45	-	186	47.4	16	90	13.1	2.54	89.0	-	55.1	LMS (1991)
Baker, WA	Fyke net	Sockeye salmon	-	-	-	15.6	19	300	76.2	1.52	-	-	64.0	Eicher Associates (1987)
Baker, WA	Fyke net	Coho salmon	-	-	-	15.6	19	300	76.2	1.52	-	-	72.0	Eicher Associates (1987)
Buchanan, MI	Full dschrg netting	Chinook salmon	600	400	420	2.8	-	-	-	-	79.7	98.3	79.6	RMC (1992)
Buchanan, MI	Full dschrg netting	Steelhead trout	600	400	420	6.2	-	-	-	-	75.3	87.8	79.4	RMC (1992)
Bond Falls, MI	Full dschrg netting	Rainbow Trout	350	225	210	12.7	-	300	64.0	-	82.0	97.8	83.8	RMC (1996)
Bond Falls, MI	Full dschrg netting	Yellow Perch	360	225	102	12.7	-	300	64.0	-	82.5	98.7	79.5	RMC (1996)
Bond Falls, MI	Full dschrg netting	Golden Shiner	405	225	70	12.7	-	300	64.0	-	70.4	93.3	77.9	RMC (1996)
Bond Falls, MI	Full dschrg netting	Bluegill	660	450	115	12.7	-	300	64.0	-	82.1	97.3	81.7	RMC (1996)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	144	94	76	18.4	15	226	24.4	1.83	99.3	87.2	100.0	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	141	90	127	18.4	15	226	24.4	1.83	87.2	92.2	98.2	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Centrarchiforms	76	35	178	18.4	15	226	24.4	1.83	100.0	100.0	86.8	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	145	86	76	18.4	15	226	24.4	1.83	86.9	95.3	80.3	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	139	92	127	18.4	15	226	24.4	1.83	95.7	91.3	84.8	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	125	58	178	18.4	15	226	24.4	1.83	95.2	100.0	70.3	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	136	63	229	18.4	15	226	24.4	1.83	100.0	98.4	64.3	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	146	94	292	18.4	15	226	24.4	1.83	97.9	85.0	59.5	Harza (1995)
Caldron Falls, WI (Unit 1)	Full dschrg netting	Fusiforms	153	76	>292	18.4	15	226	24.4	1.83	95.4	81.6	35.5	Harza (1995)
Centralia, WI (Unit 2)	Full dschrg netting	White Sucker	-	-	125	14.4	15	90	6.1	0.71	-	-	97.9	Harza (1995)
Centralia, WI (Unit 1)	Full dschrg netting	Bluegill	-	-	125	14.4	15	90	6.1	0.71	-	-	98.2	Harza (1995)
Centralia, WI (Unit 1)	Full dschrg netting	Bluegill	-	-	175	14.4	15	90	6.1	0.71	-	-	86.8	Harza (1995)
Centralia, WI	Full dschrg netting	resident	-	-	< 100	variable	15	90	4.7	0.71	-	-	64.0	BVMCA, (1991)
Colton, NY	Full dschrg netting	Centrarchid	-	-	< 100	14.1	19	360	80.8	1.50	-	-	3.0	KA (1996)
Colton, NY	Full dschrg netting	Centrarchid	-	-	175	14.1	19	360	80.8	1.50	-	-	1.0	KA (1996)
Colton, NY	Full dschrg netting	Centrarchid	-	-	> 250	14.1	19	360	80.8	1.50	-	-	0.0	KA (1996)
Colton, NY	Full dschrg netting	Percid	-	-	< 100	14.1	19	360	80.8	1.50	-	-	65.0	KA (1996)

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture		% Survival 1 hr	Source
											Test	Control		
Colton, NY	Full dschrg netting	Percid	-	-	175	14.1	19	360	80.8	1.50	-	-	14.0	KA (1996)
Colton, NY	Full dschrg netting	Percid	-	-	> 250	14.1	19	360	80.8	1.50	-	-	17.0	KA (1996)
Colton, NY	Full dschrg netting	Salmonid	-	-	< 100	14.1	19	360	80.8	1.50	-	-	68.0	KA (1996)
Colton, NY	Full dschrg netting	Salmonid	-	-	175	14.1	19	360	80.8	1.50	-	-	31.0	KA (1996)
Colton, NY	Full dschrg netting	Salmonid	-	-	> 250	14.1	19	360	80.8	1.50	-	-	7.0	KA (1996)
Colton, NY	Full dschrg netting	Soft Ray	-	-	100	14.1	19	360	80.8	1.50	-	-	75.0	KA (1996)
Colton, NY	Full dschrg netting	Soft Ray	-	-	175	14.1	19	360	80.8	1.50	-	-	47.0	KA (1996)
Colton, NY	Full dschrg netting	Soft Ray	-	-	> 250	14.1	19	360	80.8	1.50	-	-	17.0	KA (1996)
Crown Zellerback, OR (Unit 20)	Full dschrg netting	Steelhead trout	1,777	500	-	11.6	-	277	11.9	-	52.1	96.0	69.4	Eicher Associates (1987)
Crown Zellerback, OR (Unit 20)	Full dschrg netting	Chinook salmon	1,800	500	-	11.6	-	277	11.9	-	52.2	98.6	71.6	Eicher Associates (1987)
Crown Zellerback, OR (Unit 21)	Full dschrg netting	Steelhead trout	17,999	500	-	14.8	-	255	13.0	-	51.0	70.8	80.0	Eicher Associates (1987)
Crown Zellerback, OR (Unit 21)	Full dschrg netting	Chinook salmon	1,798	500	-	14.8	-	255	13.0	-	74.3	91.8	81.2	Eicher Associates (1987)
Cushman Plant 2 (1960)	Full dschrg netting	Salmonids	25,108	-	58	22.7	17	300	137.2	2.11	85.8-99.7	70.4-90.9	44.6-77.3	Cramer/Oligher (1964)
Cushman Plant 2 (1961)	Full dschrg netting	Silver Salmon	7,923	4,000	89	22.7	17	300	137.2	2.11	94.0	82.0	34.5 - 72	Cramer/Oligher (1964)
Cushman Plant 2 (1961)	Full dschrg netting	Steelhead	1,590	800	127	22.7	17	300	137.2	2.11	77.0	46.0	33.8 - 51.9	Cramer/Oligher (1964)
E. J. West, NY	Full dschrg netting	Centrarchid	320	320	< 100	76.5	15	113	19.2	3.33	62.5	79.1	71.7	KA (1996)
E. J. West, NY	Full dschrg netting	Centrarchid	159	160	175	76.5	15	113	19.2	3.33	73.0	62.8	85.5	KA (1996)
E. J. West, NY	Full dschrg netting	Centrarchid	128	128	> 250	76.5	15	113	19.2	3.33	86.7	94.9	59.8	KA (1996)
E. J. West, NY	Full dschrg netting	Percid	240	240	< 100	76.5	15	113	19.2	3.33	69.6	62.0	56.1	KA (1996)
E. J. West, NY	Full dschrg netting	Soft Ray	157	159	< 100	76.5	15	113	19.2	3.33	54.8	49.7	32.3	KA (1996)
E. J. West, NY	Full dschrg netting	Soft Ray	160	159	175	76.5	15	113	19.2	3.33	67.5	79..9	71.3	KA (1996)
E. J. West, NY	Full dschrg netting	Soft Ray	160	160	> 250	76.5	15	113	19.2	3.33	71.3	58.1	67.5	KA (1996)
E. J. West, NY	Full dschrg netting	Salmonid	280	280	< 100	76.5	15	113	19.2	3.33	41.1	31.8	65.2	KA (1996)
E. J. West, NY	Full dschrg netting	Salmonid	160	160	175	76.5	15	113	19.2	3.33	72.5	53.8	90.6	KA (1996)
E. J. West, NY	Full dschrg netting	Salmonid	160	160	> 250	76.5	15	113	19.2	3.33	99.4	49.4	95.6	KA (1996)
Elwha, WA	Partial netting	Chinook salmon	42,168	20,030	-	14.2	-	300	31.7	1.49	13.1	9.9	100.0	Eicher Associates (1987)
Faraday, OR	Partial netting	Chinook salmon	1,700	0	-	14.2	-	360	36.6	1.01	-	-	50.0	Eicher Associates (1987)
Finch Pruyn, NY (Unit 4)	Balloon tag	Smallmouth Bass	61	44	191	20.1	15	225	14.0	0.91	96.7	97.8	95.0	RMC (1990a)
Finch Pruyn, NY (Unit 4)	Balloon tag	Smallmouth Bass	49	37	210	20.1	15	225	14.0	0.91	89.8	97.4	91.0	RMC (1990a)
Finch Pruyn, NY (Unit 4)	Balloon tag	Smallmouth Bass	28	44	271	20.1	15	225	14.0	0.91	96.4	93.6	93.0	RMC (1990a)
Finch Pruyn, NY (Unit 5)	Balloon tag	Smallmouth Bass	25	44	191	23.7	15	225	14.0	0.91	68.0	97.8	94.0	RMC (1990a)
Finch Pruyn, NY (Unit 5)	Balloon tag	Smallmouth Bass	32	37	210	23.7	15	225	14.0	0.91	84.4	97.5	91.0	RMC (1990a)
Finch Pruyn, NY (Unit 5)	Balloon tag	Smallmouth Bass	43	44	271	23.7	15	225	14.0	0.91	95.3	93.6	71.0	RMC (1990a)
Five Channels, MI	Full dschrg netting	Bluegill	95	-	118	33.1	16	150	11.0	1.40	99.0	-	93.6	LMS (1991)
Five Channels, MI	Full dschrg netting	Bluegill	91	-	170	33.1	16	150	11.0	1.40	86.0	-	89.2	LMS (1991)
Five Channels, MI	Full dschrg netting	Gold./Common Shiner	59	-	114	33.1	16	150	11.0	1.40	86.0	-	81.8	LMS (1991)
Five Channels, MI	Full dschrg netting	Gold./Common Shiner	60	-	154	33.1	16	150	11.0	1.40	87.0	-	85.5	LMS (1991)
Five Channels, MI	Full dschrg netting	Northern Pike	31	-	352	33.1	16	150	11.0	1.40	97.0	-	91.3	LMS (1991)
Five Channels, MI	Full dschrg netting	Rainbow Trout	40	-	108	33.1	16	150	11.0	1.40	60.0	-	95.8	LMS (1991)

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Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture		% Survival 1 hr	Source
											Test	Control		
Five Channels, MI	Full dschrg netting	Rainbow Trout	46	-	317	33.1	16	150	11.0	1.40	20.0	-	70.0	LMS (1991)
Five Channels, MI	Full dschrg netting	Spottail Shiner	30	-	116	33.1	16	150	11.0	1.40	37.0	-	36.4	LMS (1991)
Five Channels, MI	Full dschrg netting	Walleye	55	-	162	33.1	16	150	11.0	1.40	100.0	-	71.2	LMS (1991)
Five Channels, MI	Full dschrg netting	Walleye	60	-	385	33.1	16	150	11.0	1.40	100.0	-	76.7	LMS (1991)
Five Channels, MI	Full dschrg netting	White Sucker	56	-	180	33.1	16	150	11.0	1.40	86.0	-	88.6	LMS (1991)
Five Channels, MI	Full dschrg netting	White Sucker	60	-	290	33.1	16	150	11.0	1.40	82.0	-	71.4	LMS (1991)
Five Channels, MI	Full dschrg netting	Yellow Perch	25	-	107	33.1	16	150	11.0	1.40	88.0	-	72.7	LMS (1991)
Five Channels, MI	Full dschrg netting	Yellow Perch	30	-	186	33.1	16	150	11.0	1.40	93.0	-	77.1	LMS (1991)
Glines, WA	Partial netting	Silver salmon	31,256	23,442	-	42.5	-	225	59.1	2.35	5.0	49.3	69.6	Eicher Associates (1987)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	76	18.3	15	90	8.5	1.47	-	-	96.7	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	127	18.3	15	90	8.5	1.47	-	-	100.0	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	Bluegill	-	-	178	18.3	15	90	8.5	1.47	-	-	94.9	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	76	18.3	15	90	8.5	1.47	-	-	100.0	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	127	18.3	15	90	8.5	1.47	-	-	100.0	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	178	18.3	15	90	8.5	1.47	-	-	94.9	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	229	18.3	15	90	8.5	1.47	-	-	93.7	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	292	18.3	15	90	8.5	1.47	-	-	90.4	NAI (1994)
Grand Rapids, WI (U 1,2,4 comb)	Full dschrg netting	White Sucker	-	-	>292	18.3	15	90	8.5	1.47	-	-	80.5	NAI (1994)
Hardy, MI (Unit 2)	Full dschrg netting	Bluegill	63	-	118	14.4	16	163.6	30.5	2.13	56.0	-	89.5	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Bluegill	30	-	170	14.4	16	163.6	30.5	2.13	80.0	-	91.5	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Gold./Common Shiner	30	-	114	14.4	16	163.6	30.5	2.13	82.0	-	85.5	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Gold./Common Shiner	59	-	154	14.4	16	163.6	30.5	2.13	81.0	-	88.7	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Largemouth Bass	60	-	118	14.4	16	163.6	30.5	2.13	65.0	-	76.2	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Northern Pike	58	-	352	14.4	16	163.6	30.5	2.13	86.0	-	76.0	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Rainbow Trout	59	-	108	14.4	16	163.6	30.5	2.13	44.0	-	71.4	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Rainbow Trout	60	-	317	14.4	16	163.6	30.5	2.13	60.0	-	68.6	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Walleye	60	-	385	14.4	16	163.6	30.5	2.13	95.0	-	77.3	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	White Sucker	59	-	180	14.4	16	163.6	30.5	2.13	65.0	-	76.9	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	White Sucker	60	-	290	14.4	16	163.6	30.5	2.13	76.0	-	64.5	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Yellow Perch	60	-	107	14.4	16	163.6	30.5	2.13	63.0	-	83.1	LMS (1991)
Hardy, MI (Unit 2)	Full dschrg netting	Yellow Perch	-	-	188	14.4	16	163.6	30.5	2.13	82.0	-	95.5	LMS (1991)
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	154	88	76	7.8	12	358	25.3	0.99	90.9	84.1	85.5	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	90	48	127	7.8	12	358	25.3	0.99	90.0	81.3	78.1	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Centrarchiforms	111	70	178	7.8	12	358	25.3	0.99	90.9	84.0	58.9	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	146	95	76	7.8	12	358	25.3	0.99	80.1	82.1	87.8	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	81	49	127	7.8	12	358	25.3	0.99	-	-	67.9	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	184	79	178	7.8	12	358	25.3	0.99	-	-	48.4	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	96	66	229	7.8	12	358	25.3	0.99	-	-	46.2	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	160	58	292	7.8	12	358	25.3	0.99	-	-	20.1	Harza (1995)
High Falls (Unit 5)	Full dschrg netting	Fusiforms	71	41	>292	7.8	12	358	25.3	0.99	-	-	2.7	Harza (1995)
Highley, NY	Full dschrg netting	Centrarchid	-	-	< 100	19.1	13	257	14.0	1.22	-	-	81.0	KA (1996)

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture Test	Percent Recapture Control	% Survival 1 hr	Source
Highley, NY	Full dschrg netting	Centrarchid	-	-	175	19.1	13	257	14.0	1.22	-	-	14.0	KA (1996)
Highley, NY	Full dschrg netting	Centrarchid	-	-	> 250	19.1	13	257	14.0	1.22	-	-	17.0	KA (1996)
Highley, NY	Full dschrg netting	Percid	-	-	< 100	19.1	13	257	14.0	1.22	-	-	59.0	KA (1996)
Highley, NY	Full dschrg netting	Percid	-	-	> 250	19.1	13	257	14.0	1.22	-	-	40.0	KA (1996)
Highley, NY	Full dschrg netting	Salmonid	-	-	< 100	19.1	13	257	14.0	1.22	-	-	70.0	KA (1996)
Highley, NY	Full dschrg netting	Salmonid	-	-	175	19.1	13	257	14.0	1.22	-	-	44.0	KA (1996)
Highley, NY	Full dschrg netting	Salmonid	-	-	> 250	19.1	13	257	14.0	1.22	-	-	61.0	KA (1996)
Highley, NY	Full dschrg netting	Soft Ray	-	-	< 100	19.1	13	257	14.0	1.22	-	-	60.0	KA (1996)
Highley, NY	Full dschrg netting	Soft Ray	-	-	175	19.1	13	257	14.0	1.22	-	-	72.0	KA (1996)
Highley, NY	Full dschrg netting	Soft Ray	-	-	> 250	19.1	13	257	14.0	1.22	-	-	40.0	KA (1996)
Holst, MI	Full dschrg netting	Brown Trout	150	150	85	8.5	-	360	43.3	-	56.0	99.3	45.1	RMC (1993c)
Holst, MI	Full dschrg netting	Brook Trout	150	150	135	8.5	-	360	43.3	-	73.3	1.0	43.0	RMC (1993c)
Holst, MI	Full dschrg netting	Brown Trout	150	150	220	8.5	-	360	43.3	-	90.7	1.0	22.8	RMC (1993c)
Holst, MI	Full dschrg netting	Bluegill	150	150	65	8.5	-	360	43.3	-	44.0	98.7	19.7	RMC (1993c)
Holst, MI	Full dschrg netting	Bluegill	150	150	115	8.5	-	360	43.3	-	65.3	1.0	75.0	RMC (1993c)
Holtwood, PA(U10/single runner)	Balloon tag	American Shad	100	100	125	99.1	16	94.7	18.9	3.80	81.0	90.0	89.4	RMC (1992d)
Holtwood, PA (U3/double runner)	Balloon tag	American Shad	100	80	125	99.1	17	102.8	18.9	2.84	78.0	93.8	83.5	RMC (1992d)
la centrale Beauharnois, QE	Float tag	American eel	100	-	888	198.2	13	75	24.1	5.38	97.1	-	84.2	Desrochers (1995)
Leaburg, OR	Full dschrg netting	Rainbow trout	1,249	624	-	31.2	-	225	27.1	2.29	67.0	96.2	95.2	Eicher Associates (1987)
Lequille, NS	Full dschrg netting	Atlantic salmon	-	-	-	9.9	13	519	118.0	1.37	-	-	52.0	Eicher Associates (1987)
Luray, VA	Full dschrg netting	American Eel	393	-	853	10.5	12	164	4.9	1.59	-	-	99.0	RMC (1995)
McClure, MI	Full dschrg netting	Resident spp.	NA	NA	-	4.4	-	600	129.2	-	-	NA	-	RMC (1993b)
Minetto, NY	Full dschrg netting	Centrarchid	164	104	< 100	42.5	16	72	5.2	3.53	64.0	88.5	62.0	KA (1996)
Minetto, NY	Full dschrg netting	Centrarchid	236	110	175	42.5	16	72	5.2	3.53	90.7	91.3	83.0	KA (1996)
Minetto, NY	Full dschrg netting	Centrarchid	165	120	> 250	42.5	16	72	5.2	3.53	85.5	91.7	84.0	KA (1996)
Minetto, NY	Full dschrg netting	Percid	133	117	< 100	42.5	16	72	5.2	3.53	44.4	47.0	80.0	KA (1996)
Minetto, NY	Full dschrg netting	Percid	243	142	175	42.5	16	72	5.2	3.53	68.7	85.2	86.0	KA (1996)
Minetto, NY	Full dschrg netting	Soft Ray	348	220	< 100	42.5	16	72	5.2	3.53	49.7	42.3	82.0	KA (1996)
Minetto, NY	Full dschrg netting	Soft Ray	214	133	175	42.5	16	72	5.2	3.53	72.9	98.5	94.0	KA (1996)
Minetto, NY	Full dschrg netting	Soft Ray	177	160	> 250	42.5	16	72	5.2	3.53	94.4	90.0	84.0	KA (1996)
Minetto, NY	Full dschrg netting	Salmonids	237	160	< 100	42.5	16	72	5.2	3.53	62.5	83.3	92.0	KA (1996)
Minetto, NY	Full dschrg netting	Salmonids	184	107	175	42.5	16	72	5.2	3.53	81.5	84.1	91.0	KA (1996)
Minetto, NY	Full dschrg netting	Salmonids	178	159	> 250	42.5	16	72	5.2	3.53	78.1	67.9	92.0	KA (1996)
Minetto, NY	Full dschrg netting	American Eel	107	92	625	42.5	16	72	5.2	3.53	43.9	66.3	94.0	KA (1996)
Minetto, NY	Full dschrg netting	Alewife	189	140	<100	-	-	-	-	-	74.1	90.0	80.0	KA (1996)
North Fork, OR	Partial netting	Coho salmon	4,076	5,158	-	70.8	-	139	41.5	2.95	18.2	23.1	74.0	Eicher Associates (1987)

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture		% Survival 1 hr	Source
											Test	Control		
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	146	84	76	13.0	15	100	4.0	2.03	88.4	91.7	100.0	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	140	77	127	13.0	15	100	4.0	2.03	79.3	79.2	98.9	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Centrarchiforms	121	75	178	13.0	15	100	4.0	2.03	71.9	69.3	100.0	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	158	103	76	13.0	15	100	4.0	2.03	85.4	97.1	94.0	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	141	90	127	13.0	15	100	4.0	2.03	86.5	95.6	93.7	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	166	109	178	13.0	15	100	4.0	2.03	92.2	93.6	96.6	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	158	93	229	13.0	15	100	4.0	2.03	94.9	91.4	95.4	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	166	105	292	13.0	15	100	4.0	2.03	85.5	84.8	85.5	Harza (1995)
Peshtigo, WI (Unit 4)	Full dschrg netting	Fusiforms	128	79	>292	13.0	15	100	4.0	2.03	83.6	79.7	82.8	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	134	94	76	14.2	15	123	5.2	2.13	94.0	93.6	100.0	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	154	93	127	14.2	15	123	5.2	2.13	75.3	96.8	84.7	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Centrarchiforms	111	70	178	14.2	15	123	5.2	2.13	49.5	98.6	83.0	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	168	104	76	14.2	15	123	5.2	2.13	87.5	92.3	89.2	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	104	69	127	14.2	15	123	5.2	2.13	93.3	98.6	76.5	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	150	91	178	14.2	15	123	5.2	2.13	98.0	93.4	68.4	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	160	96	229	14.2	15	123	5.2	2.13	75.6	96.9	61.1	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	136	83	292	14.2	15	123	5.2	2.13	89.0	100.0	53.3	Harza (1995)
Potato Rapids, WI (Unit 1)	Full dschrg netting	Fusiforms	145	112	>292	14.2	15	123	5.2	2.13	89.7	94.6	34.5	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	166	105	76	14.2	15	123	5.2	2.13	89.2	97.1	93.4	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	137	104	127	12.5	15	135	5.2	2.03	74.5	98.1	83.7	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Centrarchiforms	58	28	178	12.5	15	135	5.2	2.03	100.0	96.4	91.4	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	179	123	76	12.5	15	135	5.2	2.03	74.3	67.5	84.5	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	134	93	127	12.5	15	135	5.2	2.03	90.3	100.0	61.7	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	138	92	178	12.5	15	135	5.2	2.03	97.8	98.9	75.1	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	158	98	229	12.5	15	135	5.2	2.03	91.8	99.0	61.0	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	156	91	292	12.5	15	135	5.2	2.03	89.7	97.8	57.8	Harza (1995)
Potato Rapids, WI (Unit 2)	Full dschrg netting	Fusiforms	149	85	>292	12.5	15	135	5.2	2.03	92.3	94.1	48.2	Harza (1995)
Pricket, MI	Full dschrg netting	Bluegill	256	150	52	9.2	15	257	16.5	1.36	57.0	62.7	97.7	RMC (1991c)
Pricket, MI	Full dschrg netting	Golden Shiner	182	120	< 100	9.2	15	257	16.5	1.36	93.3	70.0	93.9	RMC (1991c)
Pricket, MI	Full dschrg netting	Bluegill	131	90	102	9.2	15	257	16.5	1.36	80.9	80.0	92.5	RMC (1991c)
Pricket, MI	Full dschrg netting	Bluegill	21	21	> 127	9.2	15	257	16.5	1.36	100.0	90.5	85.7	RMC (1991c)
Pricket, MI	Full dschrg netting	Mixed resident	-	-	-	9.2	15	257	16.5	1.36			97.8	RMC (1991c)
Pricket, MI	Full dschrg netting	White Sucker	201	119	165	9.2	15	257	16.5	1.36	81.6	80.7	70.8	RMC (1991c)
Pricket, MI	Full dschrg netting	White Sucker	15	10	> 254	9.2	15	257	16.5	1.36	93.3	70.0	35.7	RMC (1991c)
Publishers, OR (1960)	Full dschrg netting	Steelhead trout	1,768	500	-	7.8	-	255	12.2	-	36.2	58.0	87.9	Eicher Associates (1987)
Publishers, OR (1960)	Full dschrg netting	Chinook salmon	1,798	503	-	7.8	-	255	12.2	-	51.2	100.0	87.4	Eicher Associates (1987)
Publishers, OR (1961)	Full dschrg netting	Steelhead trout	1,800	500	-	7.8	-	255	12.2	-	24.9	36.0	84.5	Eicher Associates (1987)
Publishers, OR (1961)	Full dschrg netting	Chinook salmon	1,800	500	-	7.8	-	255	12.2	-	43.5	69.6	87.1	Eicher Associates (1987)
Puntledge, BC	Floating net	Steelhead trout	1,500	-	124	-	-	277	103.6	2.16	3.5	-	58.1	Eicher Associates (1987)
Puntledge, BC	Floating net	Kamploops	1,500	-	69	-	-	277	103.6	2.16	3.4	-	72.5	Eicher Associates (1987)
Puntledge, BC	Floating net	Kamploops	1,500	-	46	-	-	277	103.6	2.16	4.9	-	71.2	Eicher Associates (1987)
Puntledge, BC	Floating net	Salmon	1,500	-	36	-	-	277	103.6	2.16	2.5	-	67.4	Eicher Associates (1987)

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture Test	Percent Recapture Control	% Survival 1 hr	Source
Rogers, MI (Units 1 & 2)	Full dschrg netting	Bluegill	90	-	118	10.8	15	150	11.9	1.52	96.0	-	96.0	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Bluegill	92	-	170	10.8	15	150	11.9	1.52	96.0	-	85.2	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Gold./Common Shiner	60	-	114	10.8	15	150	11.9	1.52	98.0	-		LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Gold./Common Shiner	34	-	154	10.8	15	150	11.9	1.52	53.0	-	92.5	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Largemouth Bass	60	-	118	10.8	15	150	11.9	1.52	92.0	-	77.4	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Northern Pike	47	-	352	10.8	15	150	11.9	1.52	89.0	-	83.4	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Rainbow Trout	30	-	108	10.8	15	150	11.9	1.52	100.0	-	89.9	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Rainbow Trout	30	-	317	10.8	15	150	11.9	1.52	83.0	-	61.2	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Spottail Shiner	31	-	116	10.8	15	150	11.9	1.52	100.0	-	73.5	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Walleye	40	-	385	10.8	15	150	11.9	1.52	95.0	-	86.2	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	White Sucker	55	-	180	10.8	15	150	11.9	1.52	73.0	-	91.2	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	White Sucker	57	-	290	10.8	15	150	11.9	1.52	68.0	-	88.1	LMS (1991)
Rogers, MI (Units 1 & 2)	Full dschrg netting	Yellow Perch	78	-	107	10.8	15	150	11.9	1.52	98.0	-	91.8	LMS (1991)
Ruskin, BC	Fyke netting dwnstrm	Sockeye Salmon	12,125	12,159	86	113.3	-	120	39.6	3.78	-	-	89.5	Eicher (1987)
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	165	99	76	18.4	15	150	12.8	2.21	89.1	94.9	97.0	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	141	90	127	18.4	15	150	12.8	2.21	97.8	100.0	80.7	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Centrarchiforms	61	53	178	18.4	15	150	12.8	2.21	100.0	98.1	79.9	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	169	100	76	18.4	15	150	12.8	2.21	62.1	94.0	64.9	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	132	96	127	18.4	15	150	12.8	2.21	86.4	99.0	75.0	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	145	97	178	18.4	15	150	12.8	2.21	97.2	100.0	76.0	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	127	78	229	18.4	15	150	12.8	2.21	91.3	91.0	69.8	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	119	71	292	18.4	15	150	12.8	2.21	87.3	98.6	58.4	Harza (1995)
Sandstone Rapids,WI	Full dschrg netting	Fusiforms	144	92	>292	18.4	15	150	12.8	2.21	93.1	89.1	47.1	Harza (1995)
Schaghticoke, NY	Full dschrg netting	Centrarchid	149	144	<100	11.6	17	300	43.6	2.03	84.3	95.1	27.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Centrarchid	160	160	175	11.6	17	300	43.6	2.03	92.8	84.4	59.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Centrarchid	200	200	>250	11.6	17	300	43.6	2.03	80.5	89.0	7.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Percid	239	237	<100	11.6	17	300	43.6	2.03	74.4	79.7	68.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Percid	80	80	175	11.6	17	300	43.6	2.03	100.0	87.5	39.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Soft ray	160	160	<100	11.6	17	300	43.6	2.03	67.5	85.6	60.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Soft ray	241	240	175	11.6	17	300	43.6	2.03	92.9	92.5	17.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Soft ray	149	150	>250	11.6	17	300	43.6	2.03	66.4	74.0	22.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Salmonid	159	160	<100	11.6	17	300	43.6	2.03	86.2	76.3	56.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Salmonid	240	240	175	11.6	17	300	43.6	2.03	92.5	89.2	27.0	KA (1996)
Schaghticoke, NY	Full dschrg netting	Salmonid	162	160	>250	11.6	17	300	43.6	2.03	80.2	60.6	11.0	KA (1996)
Seton Creek, BC	Fyke net in tailrace	Sockeye Salmon	-	-	86	127.4	-	120	43.3	3.66	-	-	90.8	Andrew & Geen (1958)
Shasta, CA (January)	Full dschrg netting	Chinook Salmon	4,800	*	102	90.6	15	138.5	115.8	4.67	72.0*	66.0*	54.8 - 72.1	Oligher/Cramer (1964)
Shasta, CA (January)	Full dschrg netting	Rainbow Trout	1,000	*	254	90.6	15	138.5	115.8	4.67	*	*	53.1 - 71.2	Oligher/Cramer (1964)
Shasta, CA (January)	Full dschrg netting	Steelhead	3,200	*	152	90.6	15	138.5	115.8	4.67	*	*	75.4 - 89.3	Oligher/Cramer (1964)
Shasta, CA (November)	Full dschrg netting	Chinook Salmon	11,500	*	102	90.6	15	138.5	115.8	4.67	93.0*	73.8*	61.7 - 84.5	Oligher/Cramer (1964)
Shasta, CA (November)	Full dschrg netting	Steelhead	4,400	*	152	90.6	15	138.5	115.8	4.67	*	*	50.5 - 69.2	Oligher/Cramer (1964)

Table 10.1-5

Physical and hydraulic characteristics of all hydroelectric dams equipped with Francis type turbines for which survival data are available.

Station	Sampling Method	Species Tested	Test sample size	Control sample size	Avg. Fish Length (mm)	Turbine Discharge (cms)	No. of Buckets	Runner Speed (rpm)	Head (m)	Runner Dia. (m)	Percent Recapture		% Survival 1 hr	Source
											Test	Control		
Shasta, CA (November)	Full dschrg netting	Rainbow Trout	1,025	*	254	90.6	15	138.5	115.8	4.67	*	*	39.6 - 90.5	Oligher/Cramer (1964)
Stevens Creek, SC	Balloon tag	Bluegill	110	110	122	28.3	14	75	8.5	3.43	95.5	99.1	95.4	RMC (1994d)
Stevens Creek, SC	Balloon tag	Blueback Herring	131	120	203	28.3	14	75	8.5	3.43	90.8	89.2	95.3	RMC (1994d)
Stevens Creek, SC	Balloon tag	Spotted Sucker/Y. Perch	120	120	165	28.3	14	75	8.5	3.43	96.7	98.3	98.3	RMC (1994d)
T. W. Sullivan, OR	Discharge netting	Steelhead trout	-	-	-	-	-	242	12.5	-	-	-	74.1	Eicher Associates (1987)
T. W. Sullivan, OR	Discharge netting	Chinook salmon	-	-	-	7.4	-	242	12.5	-	-	-	85.7	Eicher Associates (1987)
Vernon, VT/NH	Balloon tag	American Shad	153	150	95	51.9	15	74	10.4	3.98	93.5	98.7	94.7	NAI (1996b)
White Rapids, WI	Balloon tag	White Sucker	42	38	204	25.5	14	100	8.8	3.40	90.5	91.7	93.0	RMC (1993)
White Rapids, WI	Balloon tag	White Sucker	58	64	112	25.5	14	100	8.8	3.40	96.6	98.4	100.0	RMC (1993)
White Rapids, WI	Balloon tag	Bluegill	56	62	90	25.5	14	100	8.8	3.40	92.9	98.4	95.0	RMC (1993)
White Rapids, WI	Balloon tag	Bluegill	44	38	155	25.5	14	100	8.8	3.40	93.2	97.4	100.0	RMC (1993)
Youghiogheny, PA	Full dschrg netting	Alewife	Naturally entrained		51	21.2	-	-	36.6	-	-	-	0.1	RMC (1992a)
Youghiogheny, PA	Full dschrg netting	Walleye	Naturally entrained		376	21.2	-	-	36.6	-	-	-	39.5	RMC (1992a)
Youghiogheny, PA	Full dschrg netting	Rock bass	Naturally entrained		-	21.2	-	-	36.6	-	-	-	4	RMC (1992a)
Youghiogheny, PA	Full dschrg netting	Yellow perch	Naturally entrained		-	21.2	-	-	36.6	-	-	-	7	RMC (1992a)
Youghiogheny, PA	Full dschrg netting	Crappies	Naturally entrained		-	21.2	-	-	36.6	-	-	-	0.2	RMC (1992a)
Youghiogheny, PA	Full dschrg netting	White sucker	Naturally entrained		-	21.2	-	-	36.6	-	-	-	9.5	RMC (1992a)

* Composite number of fish introduced and their recapture rates; November tests - test=91.0% and control=73.8%, January tests - test=72% and control=66%.

APPENDIX 10.2

TYPES OF TURBINES

APPENDIX 10.2
TYPES OF TURBINES

There are two major families of turbines: reaction and impulse. The axial flow and Francis turbines covered in this report are in the reaction family. Low head sites utilize axial flow type units. There are two categories of axial flow units. Axial flow Kaplan units have adjustable tilt blades as shown in Figures 10.2-1 and 10.2-2. Axial flow fixed blade or propeller units have blades which are welded to the hub at a fixed tilt as shown in Figure 10.2-3 and 10.2-4. Medium and high head sites utilize Francis type units as shown in Figure 10.2-5 and 10.2-6. These figures illustrate the major turbine and generator components. Figures 10.2-7 and 10.2-8. show components of runner assemblies for Kaplan and Francis units respectively.

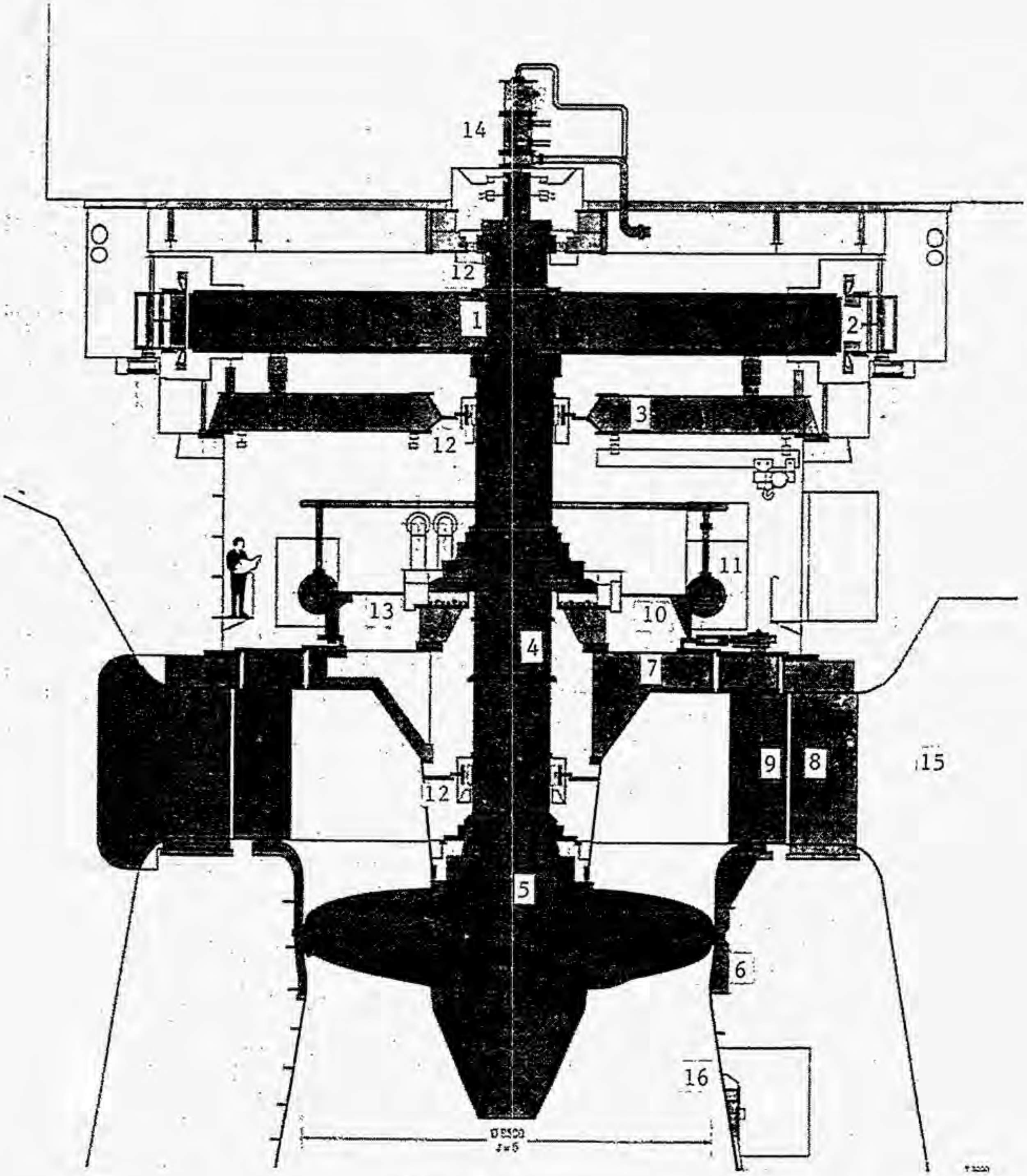
Turbines are typically classified according to a characteristic called specific speed. Specific speed is typically defined as follows:

$$n_q = \frac{N * \sqrt{Q}}{H^{.75}}$$

Where:
N = rpm
Q = Flow (cms)
H = Head (m)

Figure 10.2-9 shows various turbines as a function of specific speed. Specific speed is the rpm at which a unit would operate under 1 meter of head at a discharge of 1 m^3/s. The numerical value of this number varies with the system of units and the operating point at which it is defined. It is typically defined at the peak efficiency point. It does not vary with the size of the unit. Low specific speed designs are applied to high head sites and high specific speed designs are applied to low head sites. As specific speed increases within the axial flow type the number of blades decreases. As specific speed increases within the Francis type the ratios of runner outlet diameter to inlet diameter and wicket gate height to inlet diameter increase as shown in Figure 10.2-10.

Kaplan Turbine with Generator



1 Generator rotor
2 Generator stator
3 Spider
4 Turbine Shaft
5 Runner
6 Discharge ring
7 Turbine cover
8 Stay ring

9 Guide vane
10 Operating ring
11 Guide vane servomotor
12 Guide bearing
13 Thrust bearing
14 Oil supply head
15 Concrete semi-spiral case
16 Draft tube cone

■ Rotating parts
■ Stationary parts
■ Regulating parts
■ Bearings
☐ Bearing oil
▤ Water
▨ Masonry

Figure 10.2-1 Axial Flow Kaplan Unit

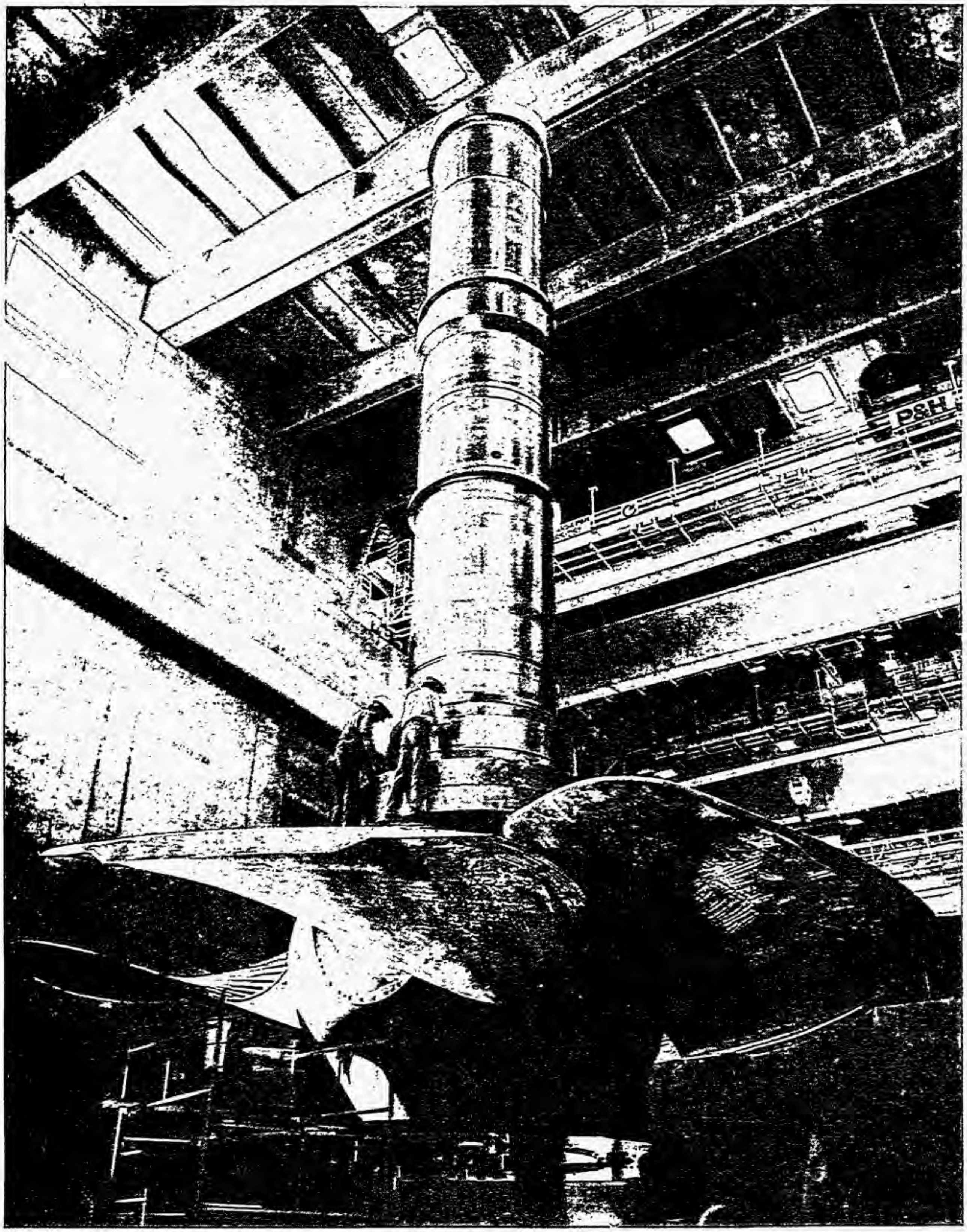


Figure 10.2-2 Axial Flow Kaplan Runner

Fixed Blade with Generator

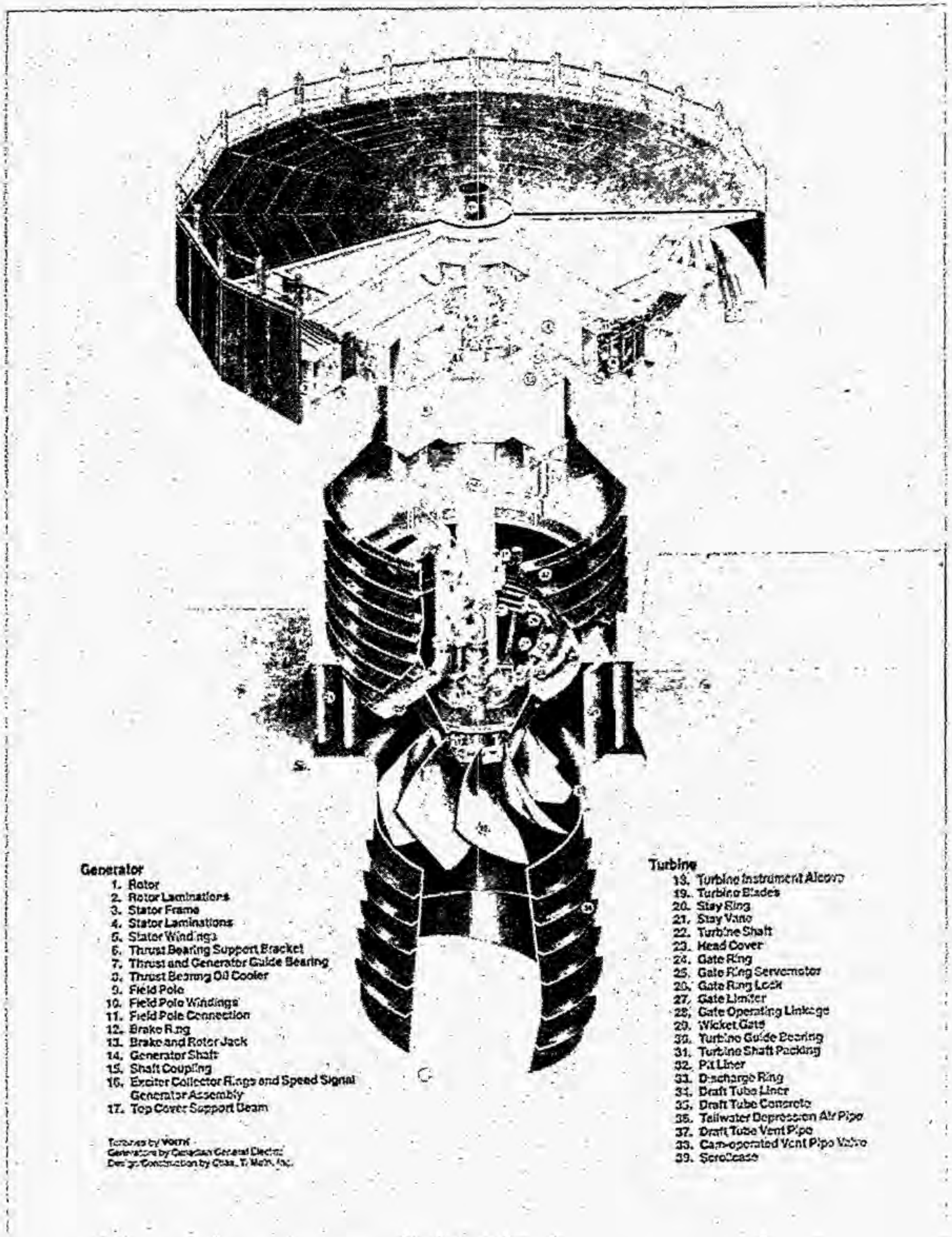
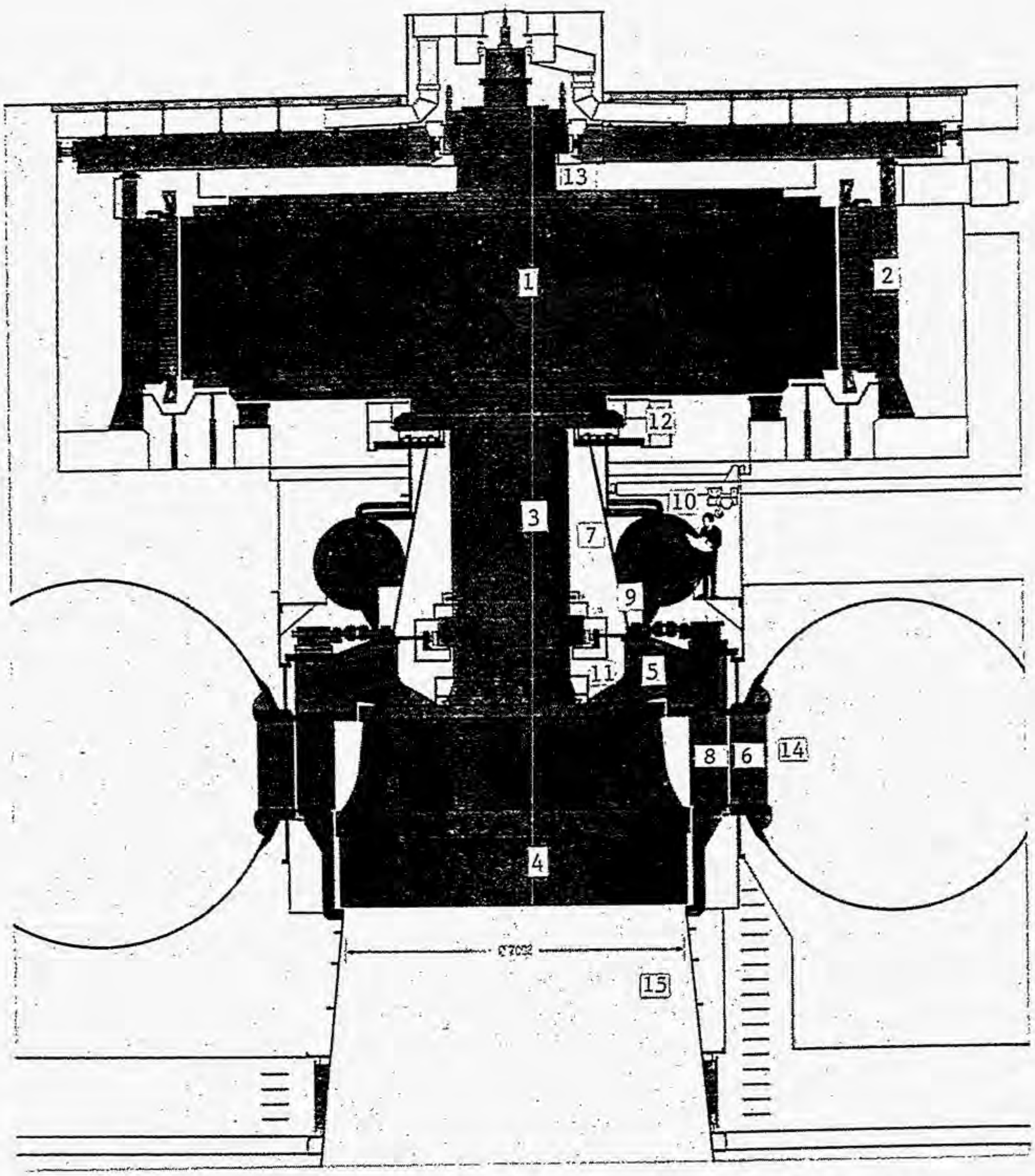


Figure 10.2-3 Fixed Blade Unit

Figure 10.2-4 Fixed Blade Runner

Francis Turbine with Generator



1 Generator rotor
2 Generator stator
3 Turbine shaft
4 Runner
5 Turbine head cover
6 Stay ring
7 Supporting cone
8 Guide vane

9 Operating ring
10 Guide vane servomotor
11 Lower guide bearing
12 Thrust bearing
13 Upper guide bearing
14 Spiral case
15 Draft tube cone

■ Rotating parrs
■ Stationary parts
■ Regulating parts
■ Bearings
▨ Bearing oil
☐ Water
☐ Masonry

Figure 10.2-5 Francis Unit

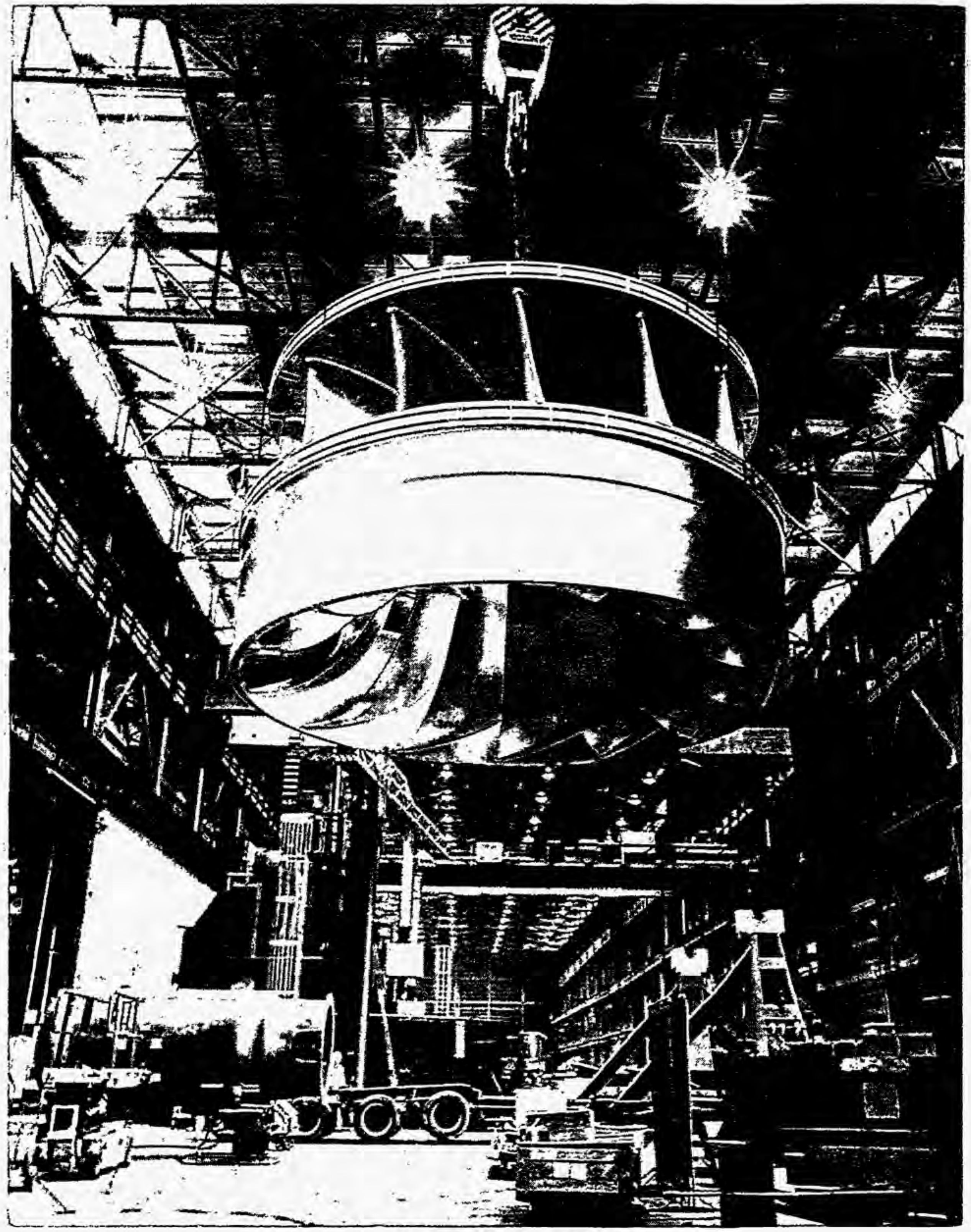


Figure 10.2-6 Francis Runner

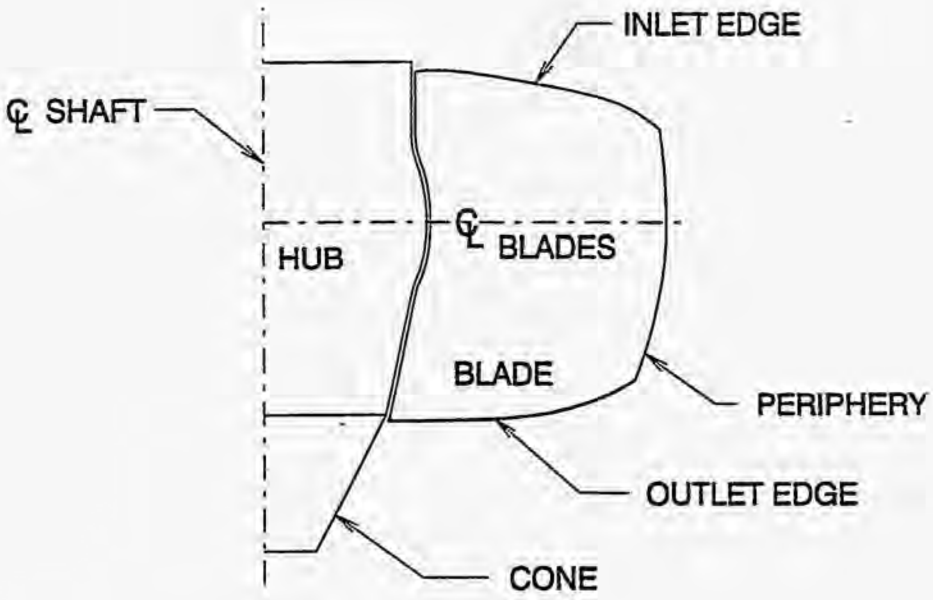


Figure 10.2-7 Major Parts of Kaplan Runner

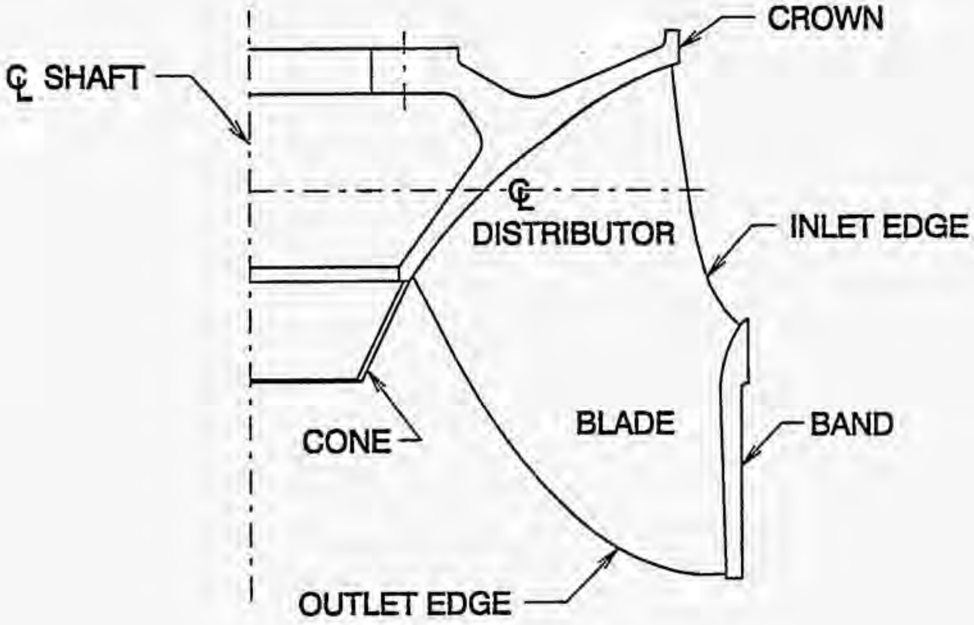


Figure 10.2-8 Major Parts of Francis Runner

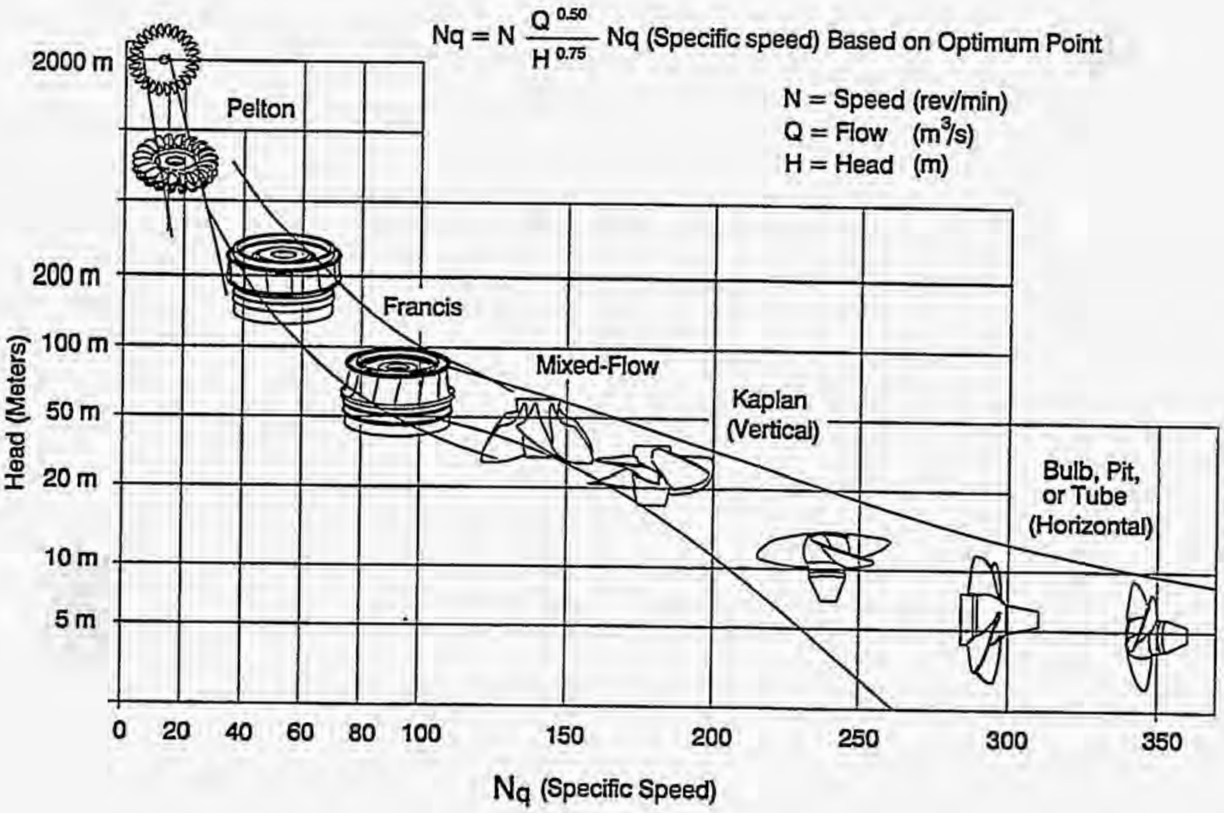


Figure 10.2-9 Turbine Type as a Function of Specific Speed

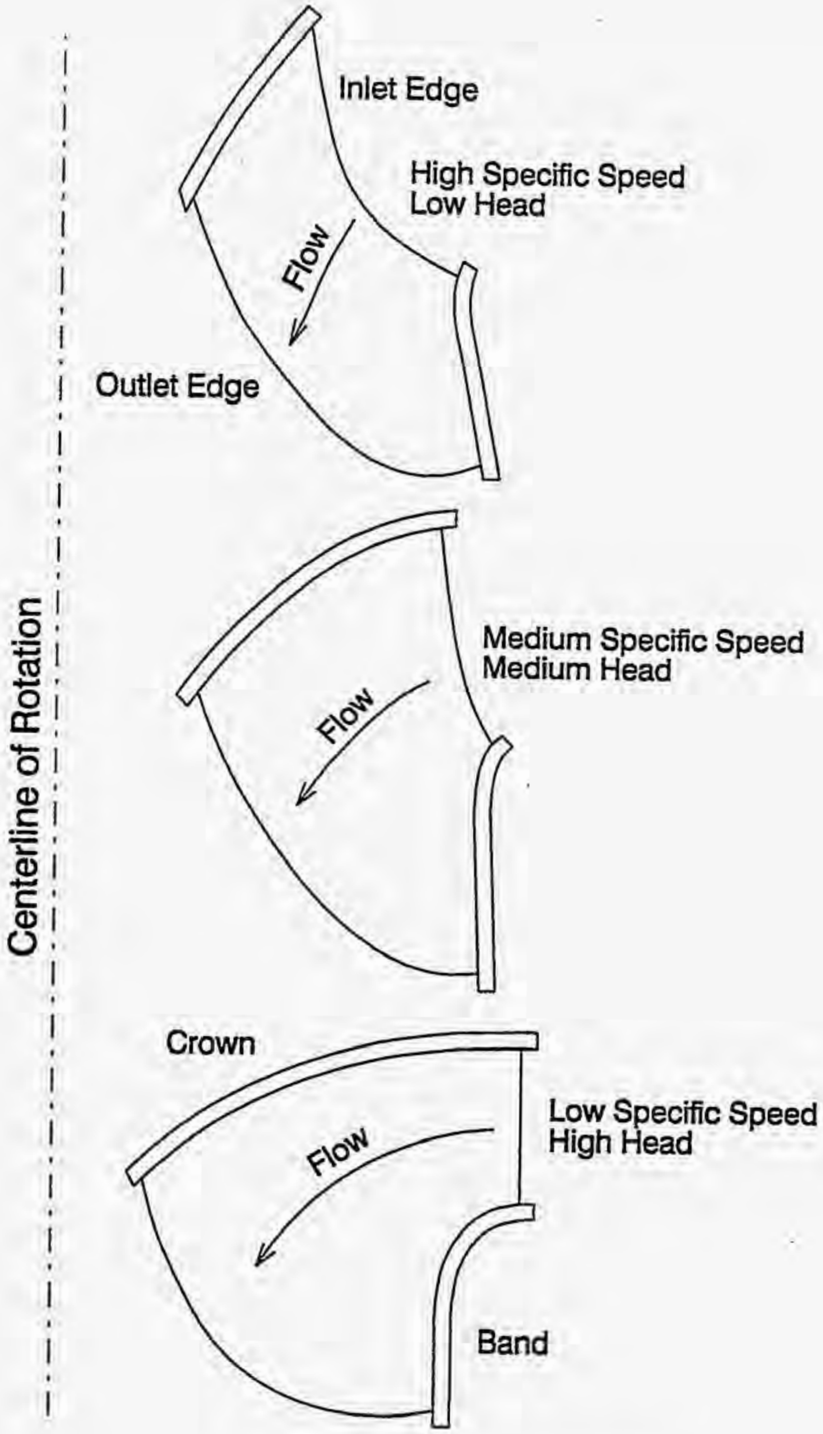


Figure 10.2-10 Variation of Francis Runner Shape with Specific Speed

APPENDIX 10.3

DERIVATION OF SHEAR PROBABILITY EQUATION

APPENDIX 10.3
DERIVATION OF SHEAR PROBABILITY EQUATION

The calculation of the probability that a fish will enter a critical shear zone is calculated in a manner very similar to the strike calculation. This calculation method is called the shear probability equation. Rather than considering the blade to be a point, as is done for the strike probability equation, the blade entrance edge has a finite dimension. The time for the passage of successive blades is based on the blade spacing minus the critical shear distance.

$$t_{runner} = \frac{\frac{\pi D}{N} - d}{\frac{\omega D}{2}}$$

This time is the same as was used in the leading edge strike equations if the value of d is zero. To use the non-dimensional value of critical shear distance, this equation may be rearranged slightly:

$$t_{runner} = \frac{2\pi}{N\omega}(1 - D^*)$$

This equation is also identical to the strike equation if the value of d is zero.

The resulting shear probability equations are:

Francis Turbine

$$P_{shear} = \lambda_{shear} \frac{N \cdot L}{D}\left(\frac{1}{1 - D^*}\right)\left[\frac{\sin\alpha_t \cdot \frac{B}{D_1}}{2Q\omega d} + \frac{\cos\alpha_t}{\pi}\right]$$

Kaplan and Propeller Turbine

$$P_{shear} = \lambda_{shear} \frac{N \cdot L}{D}\left(\frac{1}{1 - D^*}\right)\left[\frac{\cos\alpha_a}{8Q_{\omega d}} + \frac{\sin\alpha_a}{\pi \frac{r}{R}}\right]$$

From these equations, it is clear that the non-dimensional shear distance has a linear effect on the shear probability.

A correlating function, λ_{shear} has been introduced. It's value is unknown, but could be significantly different than the correlating function for leading edge strike. A CFD analysis with a virtual fish might shed light on this question.

Based on the previous evaluation of flow over airfoils, the non-dimensional shear distances have values up to .03, when the angle of attack becomes very large. As compared to the strike calculation, and ignoring any difference in correlating functions, a non-dimensional shear value of 3% will increase the strike / shear probability by a factor of (1 / (1- .03)) or by 1.03. This increase in the strike probability will have a small effect when the strike probability is low. For example, if the strike probability is 10%, a factor of 1.03 will increase it to 10.3%. If the strike probability is 90%, a factor of 1.03 will increase it to 92.7%. These effects seem too small to account for any observed fish survival effects.

The shear probability equations may be a useful analysis, but the lack of information regarding the correlating function, and the simplified shear analysis does not give results that support observed fish survival effects.

APPENDIX 10.4

EVALUATION OF ACCURACY OF FLOW ANGLE CALCULATIONS

APPENDIX 10.4
EVALUATION OF ACCURACY OF FLOW ANGLE CALCULATIONS

KAPLAN TURBINE

A CFD analysis of a Kaplan runner was used to evaluate the approximate method used to calculate the flow angle upstream of the runner. Figure 10.4-1 shows that the grid used. Two comparisons were made. Figure 10.4-2 uses a surface at the inlet of the computational model to compare the flow angle calculated by CFD with the approximate analysis. Generally, the angles agree within 4 to 10 degrees. A second surface closer to the runner was also used. This surface has more nearly axial flow than the inlet surface, but has more non-uniformity due to the near influence of the runner blade. Figure 10.4-3 shows the comparison the flow angle calculated by CFD with the approximate analysis. The angles agree within 3 degrees at the periphery, and within 8 to 10 degrees at the midspan.

FRANCIS TURBINE

A CFD analysis was performed on stay vanes and wicket gates of a Francis turbine. Two gate openings were analyzed. A location downstream of the wicket gates was used as a comparison plane. Figures 10.4-4 and 10.4-5 show a comparison of flow angles determined by the CFD analysis and by the approximate method of Section 4.3. The velocity profiles according to the CFD analysis are shown as well. The velocities and angles vary significantly from crown to band, but the approximate angle calculation is a good average value.

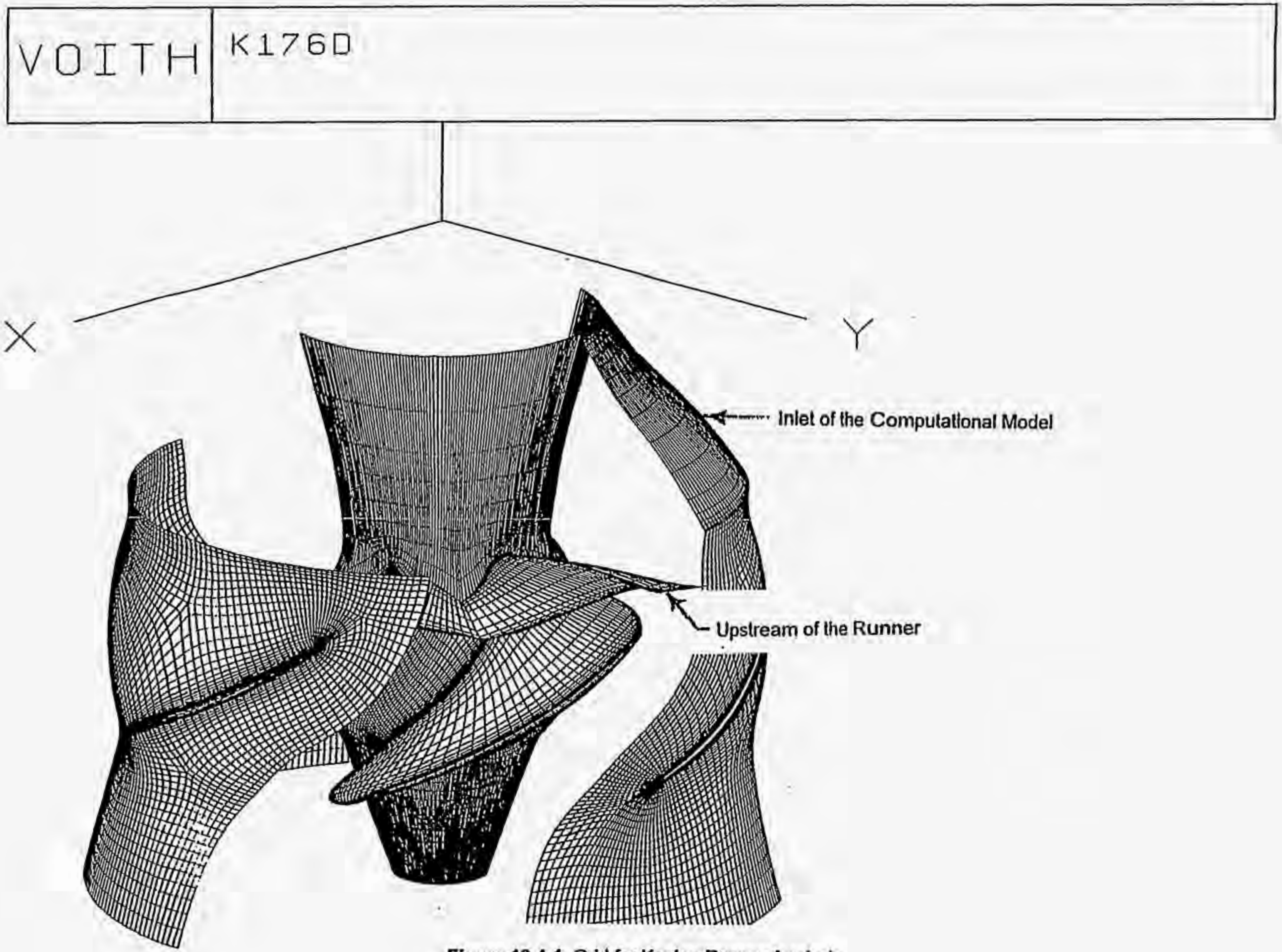
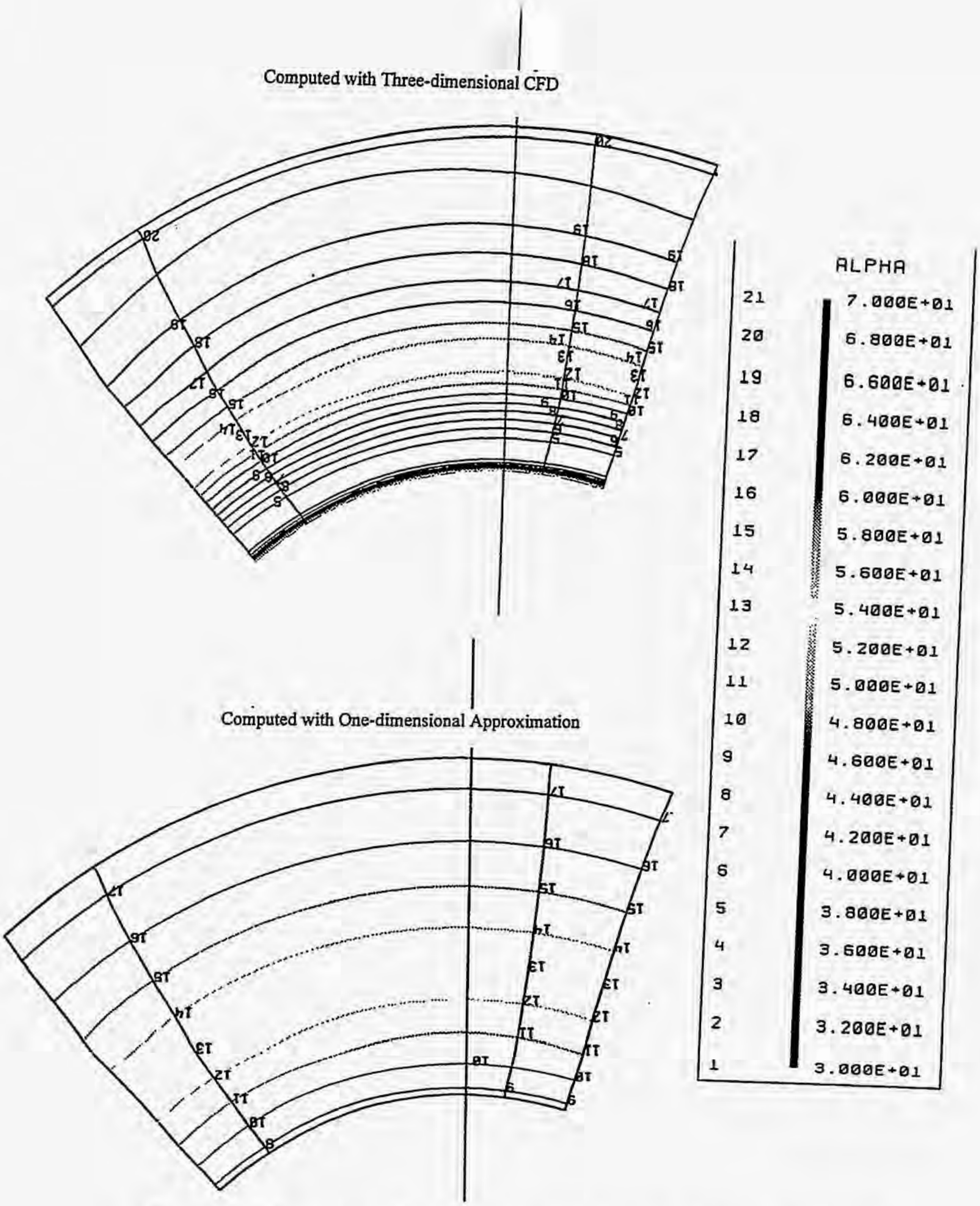


Figure 10.4-1 Grid for Kaplan Runner Analysis



Computed with Three-dimensional CFD

Computed with One-dimensional Approximation

Figure 10.4-2 Comparison of Kaplan Flow Angle at the Inlet of the Computational Model

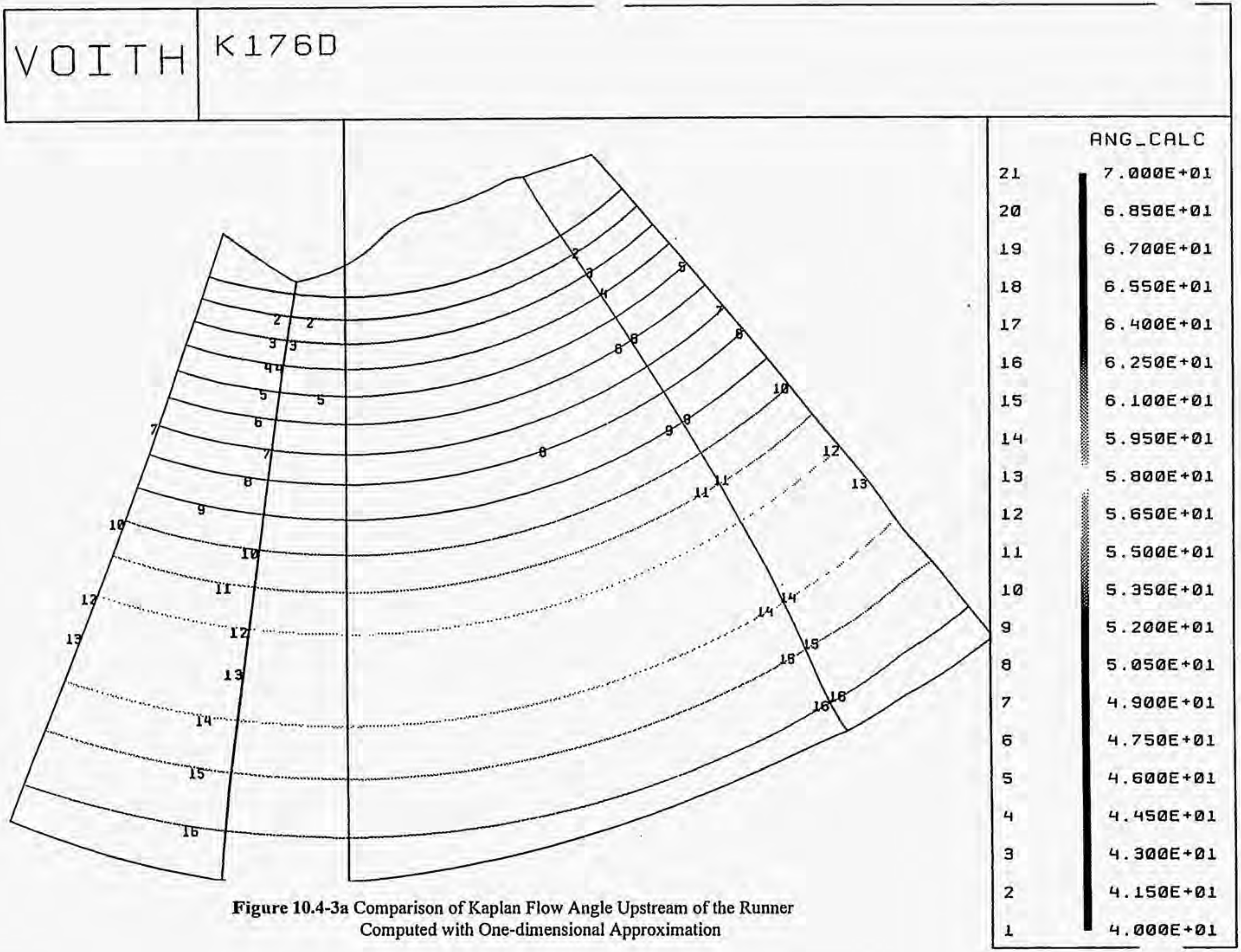


Figure 10.4-3a Comparison of Kaplan Flow Angle Upstream of the Runner Computed with One-dimensional Approximation

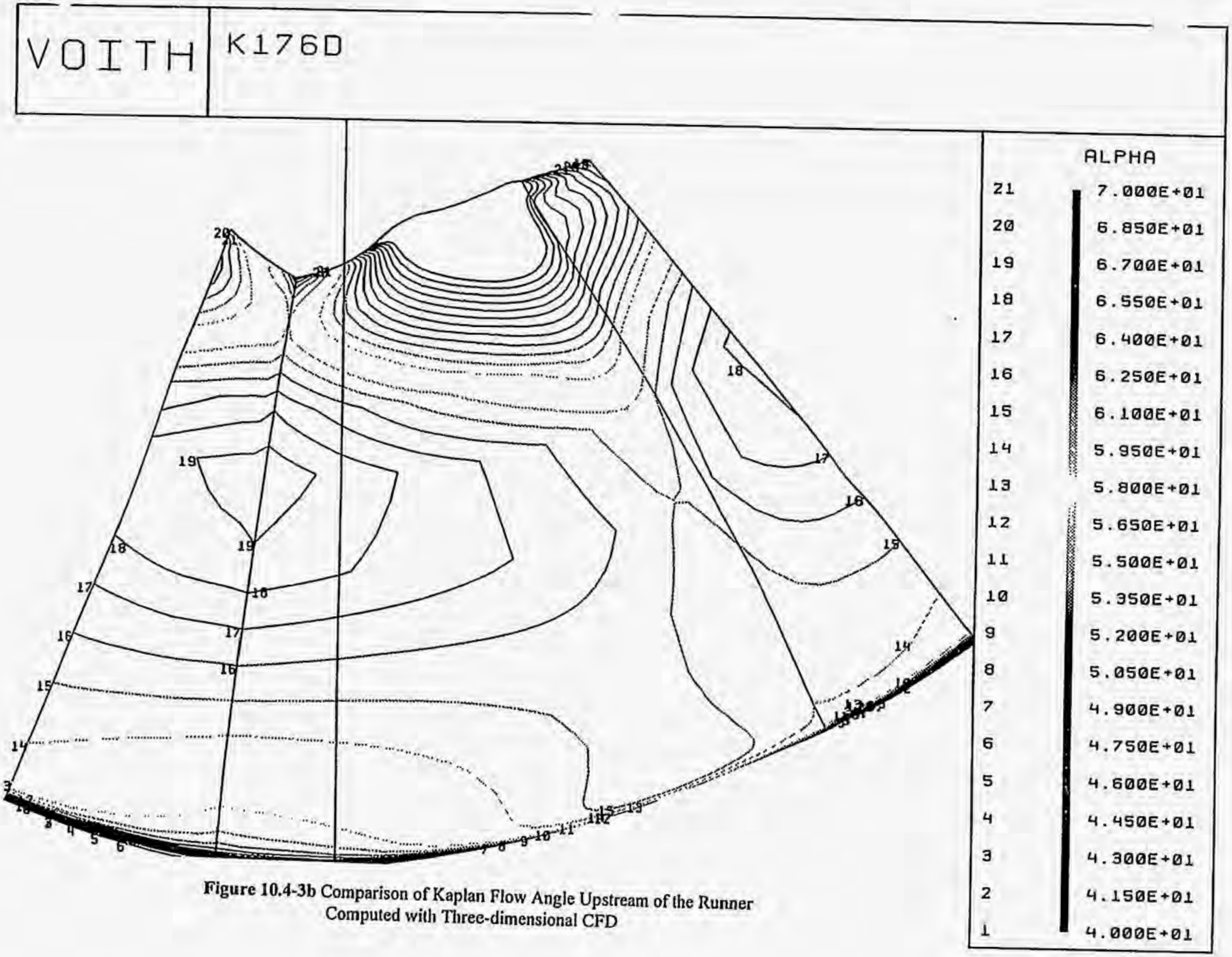


Figure 10.4-3b Comparison of Kaplan Flow Angle Upstream of the Runner Computed with Three-dimensional CFD

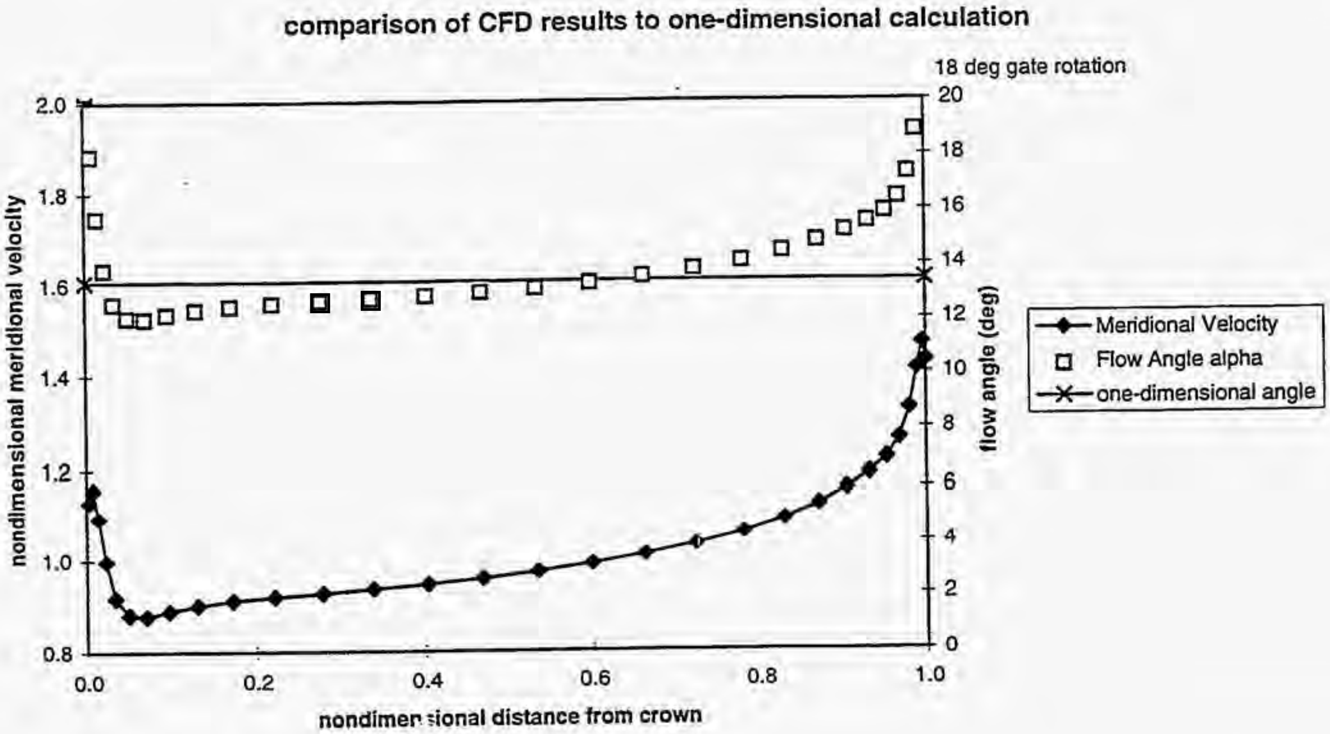


Figure 10.4-4 Comparison of Francis Flow Angle Upstream of the Runner, Smaller Gate Opening

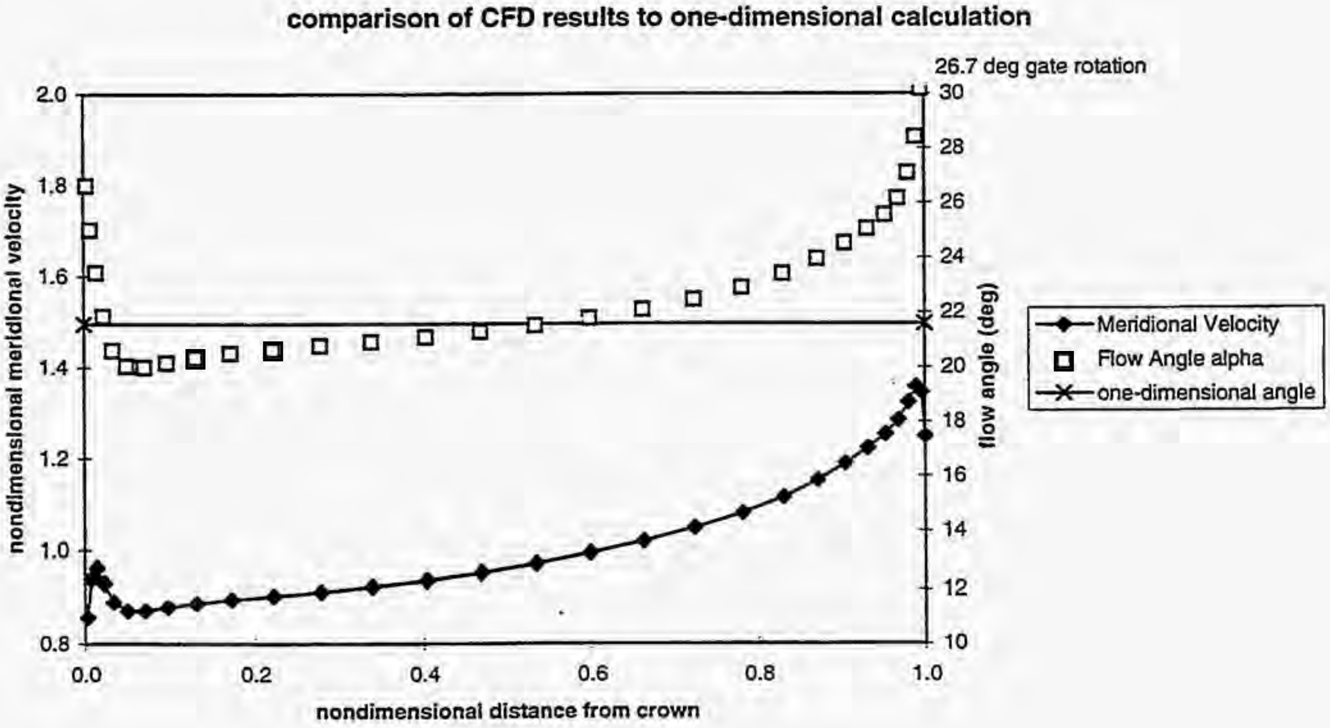


Figure 10.4-5 Comparison of Francis Flow Angle Upstream of the Runner, Larger Gate Opening

APPENDIX 10.5

ADVANCED TURBULENCE MODELLING PUBLICATIONS

APPENDIX 10.6
PERFORMANCE TESTING OF AERATING HYDROTURBINES

OBJECT AND SCOPE

The objective of this test code is to present procedures for testing (including acceptance testing) turbines which have been designed to improve the dissolved oxygen content of water in hydropower operations. These include both new turbines with integrated aeration systems and existing turbines containing retrofit aeration systems. The procedures are applicable to testing of the aeration capabilities of all types of turbines, although Francis and fixed-blade propeller turbines form the majority of the experience base. This procedure applies to any size of hydroturbine.

GUIDING PRINCIPLES

Aeration Acceptance/Guarantee Criteria

The following parameters may be the basis for guarantees and acceptance criteria for aerating hydroturbines:

- Dissolved oxygen (DO) uptake of specified mg/L,
- Total dissolved gas (TDG) in tailwater of specified percent,
- Efficiency loss of less than specified percent,
- Max power output loss less than specified hp/kW,
- Thrust increase limitation,
- Shaft runout increase limitation,
- Amount of compressor assist required (SCFM at specified pressure), and
- Aeration-induced cavitation guarantee.

Other parameters may exist based on site-specific conditions. Guarantee/acceptance criteria may be specified at multiple power outputs and water quality conditions.

Unit Efficiency Determination

Turbine efficiency testing will follow the general procedure as outlined in ASME PTC 18 - Hydraulic Turbines. For new units, aeration guarantee testing will be most effectively accomplished if performed in close coordination with acceptance testing. Absent a companion acceptance test, the general procedures of PTC 18 will be followed, with the following exceptions:

- The determination of unit efficiency will generally be sufficient, and
- Absolute flow measurement will not generally be required.

These exceptions of usual code procedures apply only for the determination of efficiency change due to operation of aeration systems.

Uncertainty Analysis

Pre-test and post-test uncertainty analyses are considered essential in this test procedure. All parties to the test are urged to consider the effects of potential measurement error on the overall uncertainty of the parameters being evaluated when choices are made concerning:

- Selection of instruments,
- Accuracy requirements,
- Agreement to exceptions from the applicable test codes.

Test Conditions

Guarantees will generally be made for specified conditions, including nominal values for the following:

- Net head on the unit,
- Power output (at peak efficiency and max gate),

- Tailwater elevation,
- Water temperature, and
- Inlet DO saturation ratio.

Test Limits

Allowable deviations from nominal values will be specified for the following:

- Head,
- Temperature,
- Tailwater elevation,
- Inlet DO saturation ratio,
- Turbine discharge, and
- Turbine speed.

Specified deviations are deviations from nominal test conditions, and deviations from average conditions during a test. PTC18 will be the starting point for this determination for the head deviations.

TEST PROCEDURES

A typical test sequence is presented below. In general, at each operating point a test run will be performed with: (1) each aeration system operating alone, and (2) with all desired combinations of aeration systems operating.

The sequence for performing an aeration test typically is as follows:

1. Perform zeroing runs in which all instruments are read with the unit off line and the wicket gates closed. Inspect instruments to ensure that all readings are reasonable. A zeroing run should be performed at the beginning and end of each test day.
2. Set wicket gate to desired position.
3. Set plant reactive power to zero.
4. Open air inlets of desired aeration alternatives.
5. Allow tailwater, power output, discharge, and dissolved oxygen to stabilize.
6. Record the electronic and manual measurements for three minutes.
7. Repeat steps 3 through 6 for each combination of alternatives.
8. Return to step 2 for the next wicket operating point.

Test runs are generally performed over a range of gate settings.

INSTRUMENTS AND METHODS OF MEASUREMENT

Mechanical Performance Measurement

Power Output

Because the determination of small changes in efficiency is especially important in aeration testing, the stability and repeatability of the power measuring equipment is especially important.

- Plant meter
- Watt-transducer
- Rotating standard
- Digital power test set

Discharge

For determination of the effect on efficiency of aerating devices, index test methods will generally be sufficient. Previous calibration or indexing to model tests or to absolute field tests is necessary.

- Winter-Kennedy taps
- Velocity traverse methods

- Dye-dilution methods
- Ultrasonic flow meters
- Gibson testing
- Thermodynamic method

Wicket Gate Position
Wicket gate position can be measured with either of the following techniques.

- Scale
- Linear displacement transducer

Inlet Head
The inlet head can be measured with either of the following techniques.

- Inlet pressure taps
- Headwater elevation + friction loss calculation

Tailwater Elevation/Discharge Head
The tailwater elevation can be measured with either of the following techniques.

- Installed stilling well
- Submersible pressure cell
- Ultrasonic ranger
- Draft tube pressure taps

Machine Dynamics/Vibration
The following measurements can be used to evaluate vibration effects produced by aeration systems.

- Generator guide bearing acceleration
- Turbine guide bearing acceleration
- Thrust bridge deflection
- Shaft runout at turbine guide bearing

Instrumentation to perform these measurements include:

- Accelerometers with appropriate anti-aliasing filter and sample frequency,
- LVDT,
- Proximity probes

Temperature and Dissolved Oxygen Concentration
A key difficulty in measuring dissolved oxygen concentration is the potential for non-uniform distribution of DO, both in the turbine intake and in the discharge. Non-uniformity in the DO and temperature distributions in the intake usually are caused by thermal stratification in the reservoir. In the tailwater, non-uniformity can be expected due to operating conditions that create a non-uniform distribution of air and water in the turbine discharge.

Obtaining a DO profile at the entrance to the scrollcase is essential for accurate testing. The DO should be sampled at multiple points around the circumference of the penstock. The sampling interval will depend on the magnitude rate of variation of DO stratification in the intake. If the magnitude of the DO stratification is consistently lower than the accuracy of the DO sensors, only a few readings will be required. For large levels of DO stratification containing frequent variations, multiple measurements may be required.

Temperature and DO profiles measured in the powerhouse forebay will determine the extent of reservoir stratification and potential magnitude of intake temperature and DO variations. If stratification occurs within the withdrawal zone for the turbine intake, frequent measurement of the temperature and DO profiles at the inlet to the scrollcase may be required.

In the discharge, sampling should be downstream of the zone where: (1) entrained and undissolved gasses escape through the water surface, and (2) surface aeration from the draft tube boil is complete. To minimize non-homogeneity, DO and temperature measurements should be far enough downstream to ensure good mixing of the turbine discharge, subject to the constraints that no mixing with other flows occurs (e.g., from adjacent units or tributary flow), and that no significant free-stream surface aeration occurs. The location of downstream sampling stations also may be influenced by time constraints. Samples located further downstream will require a longer time for measurements to stabilize, increasing both test duration and cost. If sampling for the discharge is located in a zone with incomplete mixing, multiple stations may be required. These should be strategically located based on the flow distribution of the turbine discharge.

Inlet Temperature and DO Concentration Sampling Locations
The magnitude of inlet temperature and DO stratification must be verified prior to testing. Possible measurement points include the following:

- Inlet head pressure taps,
- Gibson pressure taps, and
- Reservoir profiles.

Discharge Temperature and DO Concentration Sampling Locations
The location of the discharge temperature and DO measurements is critical for obtaining accurate, repeatable results. Some factors important for determining this location include:

- Location of the boil,
- Number of discharge bays,
- Flow patterns (e.g., avoid recirculation zones and flows from other units or DO sources),
- The magnitude of lateral and vertical mixing, and
- The amount of boil-induced surface aeration.

Temperature and Dissolved Oxygen Concentration Measurement
Various options exist for sampling and measuring temperature and DO concentration. Sampling methods include:

- Grab sample,
- In-situ, and
- Pumped sample.

Factors affecting the selected sampling method include the sampling interval, setup difficulty, and the effect of the sampling method on the sample quality and accuracy. For example, methods to collect pumped and grab samples can add additional air and DO by improper setup or handling. Pumps can alter the temperature and pressure of the sample, which can affect the DO concentration significantly.

The two primary methods to measure DO include:

- Winkler titration, and
- Membrane probe.

Because Winkler titration measurements are relatively labor intensive, these are typically used as calibrations checks for automated membrane probe readings.

Air Flow
The following primary elements are appropriate for air flow measurements.

- Bell-mouth inlet
- Venturi meter
- Orifice Plate

- Velocity traverse
  - Pitot tube
  - Hot-film anemometer
- Calibrated elbow meter
  - Calibrate in lab, including upstream piping
  - Calibrate in place
- Calibrated single point velocity measurement
  - Calibrate in lab, including upstream piping
  - Calibrate in place

The following instruments also will be required for air flow measurement.

- Differential pressure (for differential producing primary elements)
  - Electronic DP cell (preferred)
  - Manometer
  - Differential pressure gage
- Air temperature at primary element
  - Thermometer
  - Thermistor
  - RTD
  - Thermocouple
- Air pressure at primary element
- Relative humidity at primary element

Water Temperature

Since the change is usually small, water temperature can be measured at either the inlet or discharge. Most membrane-type DO probes include instrumentation for simultaneous temperature measurements.

Water Quality Parameters

Water quality parameters other than DO may be important for various sites. These parameters include:

- Total Dissolved Gas (TDG),
- Salinity,
- MBAS/Surfactants,
- Tannins, and
- Other constituents which may affect the measurement of DO concentration.

Total Dissolved Gas

Total dissolved gas (TDG) concentration can be determined by measuring total gas pressure in conjunction with barometric pressure. Instruments are commercially available that can perform this measurement automatically or manually.

ANALYSIS

Averaging of Data

Computations will be based on averages obtained from the individual readings after outliers have been removed, for each test run.

Mechanical Performance

The following computations can be used to evaluate the mechanical performance of a turbine. A key parameter resulting from these computations is the change in turbine efficiency produced by operating the aeration systems. For turbines with multiple systems, this parameter will be evaluated as a function of the system in service, and as a function of gate position. Note that some relationships given below will differ depending on the instrumentation used for a particular test. For example, computation of the turbine discharge will change if measured by a method other than with a Winter-Kennedy differential pressure.

Gravitational Acceleration

PTC18-1992 gives for local gravitational acceleration

$$g = \left(9.80616 / 0.3048 \right)\left(1 - 0.0026373\cos 2\theta + 0.0000059\cos^2 2\theta\right) - 3.086\times10^{-6} z, \tag{1}$$

where

g = gravitational acceleration in ft/s^2.
θ = latitude of the unit in degrees, and
z = altitude of the distributor centerline in feet.

Water Density

Water density is a function of water temperature. The computation used herein is that of PTC18-1992, which requires specification of T_W in °C and ρ_W in kg/m^3, given by

$$\rho_w = b_0 + b_1 T_w + b_2 T_w^2 + b_3 T_w^3 + b_4 T_w^4 + b_5 T_w^5, \tag{2}$$

where

ρ_w = water density in kg/m^3.
T_w = water temperature in °C,
b_0 = 999.8394,
b_1 = 0.06862162,
b_2 = - 0.009270732,
b_3 = 1.155160 x 10^{-4},
b_4 = - 1.626299 x 10^{-6}, and
b_5 = 1.211919 x 10^{-8}.

Specific Weight of Water

The computation used herein for the specific weight of water is that of PTC18-1992, given by

$$\rho_A(z) = 1.225\left(1 - \frac{0.0065}{288.16} z\right)^{4.2561}, \text{ and} \tag{3}$$

$$\gamma_W = \frac{g}{g_c}\left[\rho_W - \rho_A(z)\right]\frac{(0.3048)^3}{0.45359237}, \tag{4}$$

where

ρ_A = air density in kg/m^3,
γ_W = specific weight of water in lbf/ft^3, and
g_c = gravitational constant - 32.17405 ft/s^2.

Water Discharge

Water discharge is indexed to the differential pressure across the Winter-Kennedy taps. The formula for the discharge is

$$Q_w = C_{WK}\sqrt{\Delta H_{WK}}, \tag{5}$$

where

C_{wk} = Winter-Kennedy discharge coefficient in cfs/(inch of water), and
ΔH_{wk} = pressure measured across Winter-Kennedy pressure taps in inches of water.

Turbine Power

Turbine power output is the sum of generator power output and generator power losses, given by

$$P_t = P_G + P_{GL}, \tag{6}$$

where

P_G = Generator power in MW, and
P_{GL} = Generator power losses in MW.

Generator power losses can be modeled in many cases by quadratic polynomial,

$$P_{GL} = c_0 + c_1 P_G + c_2 P_G^2, \tag{7}$$

where

c_0 = 0.66337,
c_1 = 0.00054371, and
c_2 = 0.000096853.

Net Head

Net head is the difference between total specific energy at the scrollcase inlet section and total specific energy at the draft tube exit section. The net head computation will ignore the added volume of air flow at the draft tube exit and is given by

$$H = \left[Z_{SC} + H_{SC} + \frac{1}{2g}\left(\frac{Q_W}{A_{SC}}\right)^2 \right] - \left[TW_{DT} + \frac{1}{2g}\left(\frac{Q_W}{A_{DT}}\right)^2 \right], \tag{8}$$

where

Z_{SC} = elevation of inlet head measurement point in feet,
H_{SC} = Inlet head pressure in feet of water,
A_{SC} = Cross section area of penstock at inlet head measurement point,
TW_{DT} = tailwater elevation in feet, and
A_{DT} = draft tube exit area.

Turbine Efficiency

Turbine efficiency is the ratio of shaft power (turbine output power) to water power, given by

$$\eta = 737562.1 \frac{P_t}{\gamma_W Q_W H} \tag{9}$$

where

η = turbine efficiency,
P_t = turbine power output in MW, and
γ_W = specific weight of water in lbf/ft^3.

Turbine Efficiency Change (Aeration Hydraulic Performance)

The effect of each aeration alternative will be evaluated by the change in turbine efficiency at constant gate setting, with and without aeration, given by

$$\Delta\eta = \eta_o - \eta_a, \tag{10}$$

where

$\Delta\eta$ = change in turbine efficiency
η_o = turbine efficiency with aeration systems off, and
η_a = turbine efficiency with aeration systems on.

Analysis of Vibration Data
Vibration data can be analyzed by various methods. Possible techniques include:

- Peaks on vibration spectrum,
- Shaft runout limits, and
- Thrust bridge deflection (thrust increase/decrease).

Aeration Performance
The following computations can be used to evaluate the aeration performance of a turbine. Key parameters resulting from these computations include the dissolved oxygen uptake and the oxygen transfer efficiency. Others, such as total dissolved gas, also can be added. These parameters could be evaluated as a function of aeration option and gate position. The functions presented below will change depending on the instruments used for a particular test.

Net Positive Suction Head
Net positive suction head (NPSH) is the minimum head required for cavitation-free operation at a reference elevation located near the exit of the runner and is given by

$$NPSH = \left(\frac{144 p_{bar}}{\gamma_W}\right) - \left(Z_{NPSH} - TW_{DT}\right) - \left(\frac{144 p_v}{\gamma_W}\right), \tag{11}$$

where

$NPSH$ = net positive suction head in feet of water,
γ_W = specific weight of water in lbf/ft^3,
Z_{NPSH} = reference elevation for net positive suction head in feet,
Z_{TW} = tailwater elevation in feet, and
p_v = vapor pressure of water in psi.

Saturation Vapor Pressure Over Liquid Water
The saturation vapor pressure over liquid water is given by (ASHRAE, 1989)

$$p_{ws} = exp\left(\frac{C_8}{T_{wb}} + C_9 + C_{10} T_{wb} + C_{11} T_{wb}^2 + C_{12} T_{wb}^3 + C_{13} \ln T_{wb}\right) \tag{12}$$

where

p_{ws} = saturation vapor pressure over liquid water in psi,
T_{wb} = wet bulb temperature in °R,
$C_8 = -1.044039708 \times 10^4$,
$C_9 = -11.2946496$,
$C_{10} = -2.7022355 \times 10^{-2}$,
$C_{11} = 1.2890360 \times 10^{-5}$,
$C_{12} = -2.478068 \times 10^{-9}$, and
$C_{13} = 6.5459673$.

Humidity Ratio
The humidity ratio is given by (ASHRAE, 1989)

$$w_s^* = 0.62198 \frac{p_{ws}}{p_{bar} - p_{ws}}, \text{ and} \tag{13}$$

$$w = \frac{(1093 - 0.556T_{wb})w_s^{*} - 0.240(T_{db} - T_{wb})}{1093 + 0.444T_{db} - T_{wb}}, \tag{14}$$

where

w_s^{*} = saturation vapor pressure,
p_{bar} = barometric pressure in psi,
w = humidity ratio,
T_{db} = dry bulb temperature in °F, and
T_{wb} = wet bulb temperature in °F.

Air Density
Air density is given by (ASHRAE, 1989)

$$v = \frac{R_a T_{db}}{p_{bar}}(1 + 1.6078\,w), \text{ and} \tag{15}$$

$$\rho_A = \frac{1 + w}{v}, \tag{16}$$

where

v = specific volume of wet air in ft³/lbm,
R_a = 0.3705 psia-ft³/(lbm-R),
ρ_A = air density in lbm/ft³, and
T_{db} = dry bulb temperature in °R.

Air Viscosity
The dynamic, or absolute, viscosity of air is computed from a quadratic curve fit to air viscosity data published in NBS Circular 564 (1955), given by

$$\mu_A = 1.09472\times10^{-5} + 1.87925\times10^{-8}T_{db} - 7.057778\times10^{-12}T_{db}^2, \tag{17}$$

where

μ_a = dynamic viscosity in lbm/ft-s, and
T_{db} = dry bulb temperature in °F.

Mass Flowrate of Air (Nozzles)
The computation of the mass flowrates of air through inlet nozzles is based on the procedure presented in ASME MFC-3M-1989. The following computations are given by

$$q_m = 0.09970190\,C_D\,Y\,d_t^2\sqrt{\rho_A \Delta H}, \tag{18}$$

$$C_D = 0.9975 - 0.00653\left(\frac{10^6}{R_d}\right)^a, \quad a = \begin{cases} 0.5, & R_d < 10^6 \\ 0.2, & R_d \geq 10^6 \end{cases}, \tag{19}$$

$$R_d = \frac{48q_m}{\pi\mu_A d_t}, \tag{20}$$

$$Y = \sqrt{\left(\frac{\kappa\tau^{2/\kappa}}{\kappa - 1}\right)\left(\frac{1}{1-\tau^{2/\kappa}}\right)\left(\frac{1-\tau^{(\kappa-1)/\kappa}}{1-\tau}\right)}, \ (\kappa = 1.4 \text{ for air}), \text{ and} \tag{21}$$

$$\tau = \frac{p_{bar} + \Delta p_i}{p_{bar}}, \;\; \Delta p_i = 0.03606 \Delta H_i \,, \tag{22}$$

where

q_m = mass flowrate of air in lbm/s,
C_D = discharge coefficient,
Y = expansion factor,
d_t = nozzle throat diameter in inches,
ρ_a = air density in lbm/ft^3,
ΔH = differential pressure in inches of water,
R_d = Reynolds number based on nozzle throat diameter,
μ_a = dynamic viscosity in lbm/ft·s,
κ = specific heat ratio,
τ = pressure ratio,
p_{bar} = barometric pressure in psi, and
Δp = pressure differential in psi.

Volumetric Air Flowrate of Air

The mass flowrate of air through a nozzle is converted to volumetric flowrate of dry air at standard temperature (68 °F) and to a pressure equal to the net positive suction head. Based on the ideal gas law, the air flowrate is given by

$$Q = \frac{q_m}{\rho_A} \frac{(p_{bar} - p_v)}{p_{NPSH}} \frac{527.67}{T_{db}}, \tag{23}$$

$$p_v = \frac{p_{bar} w}{(0.62198 + w)}, \text{ and} \tag{24}$$

$$p_{NPSH} = \frac{\gamma_W}{144} (NPSH)\,, \tag{25}$$

where

Q_A = volumetric flowrate of dry air at the reference conditions in ft^3/s,
p_v = partial pressure of water vapor in air in psi, and
T_{db} = dry bulb temperature.

Air Void Ratio

The air void ratio is volumetric ratio of total air flow to total air/water mixture flow, given by

$$\phi = \frac{Q_A}{Q_W + Q_A}. \tag{26}$$

Dissolved Oxygen Saturation Concentration

The dissolved oxygen saturation concentration is computed by (Hua, 1990)

$$DO_{sat,p0} = 14.562 - 0.41022 T_W + 0.0079910 T_W^2 - 0.000077774 T_W^3, \text{ and} \tag{27}$$

$$DO_{sat} = \left(\frac{p}{p_0}\right) DO_{sat,p0}\,, \tag{28}$$

where

$DO_{sat,p0}$ = dissolved oxygen saturation concentration at standard atmospheric pressure,

T_W = water temperature in °C,
DO_{sat} = dissolved oxygen saturation concentration at measured atmospheric pressure in mg/L,
p = atmospheric pressure in psia, and
p_0 = standard atmospheric pressure (14.696 psia).

Dissolved Oxygen Uptake (Aeration Environmental Performance)

The dissolved oxygen uptake is the increase in DO concentration between the turbine scrollcase and tailwater, given by

$$\Delta DO = DO_{TW} - DO_{SC}, \tag{29}$$

where

DO_{TW} = dissolved oxygen concentration of tailwater in mg/L, and
DO_{SC} = dissolved oxygen concentration of water entering turbine scrollcase in mg/L.

Oxygen Transfer Efficiency

Oxygen transfer efficiency is the fraction of the influent dissolved oxygen deficit the turbine is able to replace. This parameter is computed by (Gulliver et al., 1990)

$$E = \frac{DO_{TW} - DO_{SC}}{DO_{sat} - DO_{SC}}, \tag{30}$$

$$E_{20} = 1 - (1 - E)^f \text{, and} \tag{31}$$

$$f = 1 + 0.02103(T_W - 20) + 8.261\text{x}10^{-5}(T_W - 20)^2, \tag{32}$$

where

E = oxygen transfer efficiency, and
E_{20} = oxygen transfer efficiency referenced to a water temperature of 20 °C.

Uncertainty Analysis

Uncertainly calculations will be based on PTC 19-1.

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