ENTRAINMENT AND IMPINGEMENT AT IP2 AND IP3: A BIOLOGICAL IMPACT ASSESSMENT

Lawrence W. Barnthouse
LWB Environmental Services, Inc.

Douglas G. Heimbuch
AKRF, Inc.

Webster Van Winkle
Van Winkle Environmental Consulting
Boise, Idaho

John Young
ASA Analysis & Communications, Inc.

January 2008

TABLE OF CONTENTS

Executive Summary

This report evaluates whether entrainment and impingement by the respective cooling water intake structures ("CWIS") at Indian Point Unit 2 ("IP2") and Indian Point Unit 3 ("IP3") have caused an adverse environmental impact ("AEI"), using biologically-based definitions of AEI that are consistent with established definitions and standards of ecological risk assessment and fisheries management.

The approach involves three elements. First, we use the extensive Hudson River fisheries datasets to determine (1) whether changes in the status of species of interest identified by the New York State Department of Environmental Conservation ("NYSDEC") have occurred since IP2 and IP3 began commercial operation, (2) whether cooling-water withdrawals by IP2 and IP3 during this period could have been responsible for any such changes, or (3) whether alternative stressors including striped bass predation, zebra mussels, and harvesting are the more probable cause of perceived changes.

Second, we use a widely-accepted method for quantifying the impacts of harvesting on the sustainability of fish populations, termed the Spawning Stock Biomass per Recruit ("SSBPR") model, to determine whether entrainment and impingement at IP2 and IP3 could have adversely affected the sustainability of the Hudson River striped bass and American shad populations.

Third, we examine long-term trends in the abundance of all Hudson River fish species for which adequate trends data sets can be developed to determine whether species with high susceptibility to entrainment at IP2 and IP3 are more likely to have declined in abundance over the past 30 years than are species with low susceptibility to entrainment.

All three elements of the assessment support a conclusion that IP2 and IP3 have not caused an AEI. Evaluation of alternative hypotheses concerning the causes of changes in abundance of Hudson River fish populations found no evidence supporting the hypothesis that IP2 and IP3 contributed to these changes. Instead, the evaluation shows that overharvesting is the most likely cause of recent declines in abundance of American shad, with striped bass predation being a potentially significant contributing factor. Increased predation by the rapidly growing Hudson River striped bass population is the most likely cause of recent declines in the abundance of Atlantic tomcod, river herring and bay anchovy. Striped bass predation probably contributed to the decline in abundance of white perch, although other unknown causes were also involved.

Two additional lines of evidence support a conclusion that entrainment and impingement at IP2 and IP3 have not resulted in AEI. Application of the SSBPR model to stock assessment data for striped bass and American shad shows that mortality caused by entrainment at IP2 and IP3 is negligible, particularly compared to fishing mortality, and does not impair the ability of these populations to sustain themselves. Analysis of community-level trends data show that species with relatively high susceptibility to entrainment at IP2 and IP3 are no more likely to have declined in abundance since 1974 than are species with relatively low susceptibility to entrainment.

Considered together, the evidence evaluated in this report shows that the operation of IP2 and IP3 has not caused effects on early life stages of fish that reasonably would be considered "adverse" by fisheries scientists and/or managers. The operation of IP2 and IP3 has not destabilized or noticeably altered any important attribute of the resource.

Glossary

Ichthyoplankton: Eggs and larvae of fish with limited swimming abilities that float in the water-column and are passively transported by currents

Entrainment: The drawing of ichthyoplankton and other small aquatic organisms through a cooling water intake structure into the cooling system of a power plant

Impingement: The trapping of fish and other aquatic organisms against intake screens by the force of the water being drawn through a cooling water intake structure

Individual: A single organism

Population: A group of plants, animals, or other organisms, all of the same species, that live together and reproduce

Community: An assemblage of species populations that occur together in space and time

Yolk-sac larvae (YSL): Fish larvae that have recently hatched and are still receiving nutrition from yolk deposited in the eggs before they were spawned

Post yolk-sac larvae (PYSL): Fish larvae that have absorbed the yolk and obtain nutrition by feeding

Young-of-the-year (YOY): Fish that have completed the transformation from the larval to the juvenile stage and have grown large enough to be captured by the gear used in the generators' Beach Seine Survey and Fall Shoals Survey

Longitudinal River Survey (LRS): The Hudson River generators' annual riverwide ichthyoplankton survey

Beach Seine Survey (BSS): The Hudson River generators' annual survey of YOY and older fish abundance in the shorezone

Fall Shoals Survey (FSS): The Hudson River generators' annual survey of YOY and older fish abundance in the shoal zone

Early life stage: The collective term for the egg, YSL, PYSL, and early juvenile (juveniles too small to be captured by the gear used in the BSS and FSS) life stages

Conditional mortality rate (CMR): A measure of the mortality imposed on a population by a stressor such as a cooling water intake structure

Recruit: A fish that has grown large enough to be caught in gears used by agencies performing stock assessments for harvested fish species; as used in the spawning stock biomass per recruit model, a one-year-old fish

Spawning stock biomass per recruit (SSBPR): The expected lifetime reproduction of a typical female recruit, measured in terms of the expected future egg production or biomass

Density-dependence: A relationship between the abundance of a population and the growth rates or mortality rates of individuals belonging to that population

Stressor: An anthropogenic or environmental factor that increases mortality or decreases growth of organisms belonging to a population exposed to that factor

Stressor metric: A measure of the intensity of a stressor

Response metric: A measure of the response of an exposed population to one or more stressors

1. Introduction

This report evaluates whether entrainment and impingement by the respective cooling water intake structures ("CWIS") at Indian Point Unit 2 ("IP2") and Indian Point Unit 3 ("IP3") has caused an adverse environmental impact ("AEI"), as that term is employed in §316(b) of the Clean Water Act ("CWA") and 6 NYCRR §704.5 and reasonably may be interpreted by the scientific community.[1] Our evaluation of whether entrainment and impingement by the respective CWIS at IP2 and IP3 has caused AEI is based on biologically-based definitions of "adverse environmental impact" consistent with established definitions and standards of ecological risk assessment (USEPA 1998) and fisheries management (Restrepo et al. 1998, Quinn and Deriso 1999). Our approach involves three elements.

First, we use the extensive Hudson River fisheries datasets (prepared under the direction and oversight of the New York State Department of Environmental Conservation ("Department" or "NYSDEC")) to determine (1) whether changes in the status of species of interest identified by NYSDEC have occurred since IP2 and IP3 began commercial operation, (2) whether cooling-water withdrawals by IP2 and IP3 during this period could have been responsible for any such changes, or (3) whether alternative stressors including striped bass predation, zebra mussels, and harvesting are the more probable cause of perceived changes.

Second, we use a widely-accepted method for quantifying the impacts of harvesting on the sustainability of fish populations, termed the Spawning Stock Biomass per Recruit ("SSBPR") model, to determine whether entrainment and impingement at IP2 and IP3 could have adversely affected the sustainability of the Hudson River striped bass and American shad populations.

Third, we examine long-term trends in the abundance of all Hudson River fish species for which adequate trends data sets can be developed to determine whether species with high

---

[1] As applicable here, the CWIS for IP2 and IP3 extend from the point at which water is withdrawn from the Hudson River (the "River") up to, and including, the intake pumps. *See, e.g.*, In Re Matter of Bowline, LLC, 2001 WL 1587359 (N.Y. Dept. Env. Conserv.) (Nov. 30, 2001), at *6-7 (relying on USEPA definition, now codified at 40 C.F.R §125.93); 40 C.F.R. §125.93. The CWIS at IP2 and IP3 are shown schematically in Figures IV-12 through IV-15 of the Draft Environmental Impact Statement for State Pollutant Discharge Elimination System Permits for Bowline Point, Indian Point 2 & 3, and Roseton Steam Electric Generating Stations, dated December 1999 (the "DEIS"), subsequently incorporated into the Final Environmental Impact Statement by the New York State Department of Environmental Conservation, accepted June 25, 2003 (the "FEIS"). *See* FEIS, p. 12. These intake structures generally commence with bar racks and debris barriers at the point of entry, include modified Ristroph traveling screens and fish return systems upstream of the point of entry, and terminate with the circulating water pumps.

susceptibility to entrainment at IP2 and IP3 are more likely to have declined in abundance over the past 30 years than are species with low susceptibility to entrainment.

Although the technical analyses documented in this report emphasize entrainment, the conclusions reached apply to the combined impacts of entrainment and impingement. There are two reasons for this. First, the trends data that are the primary focus of this assessment reflect the combined effects of entrainment and impingement. Second, entrainment is the focus of the Department, as the existing retrofits (i.e., Ristroph screens and fish returns) have resolved the Department's concerns regarding impingement (Draft SPDES Permit, Special Condition 27).

2. Approach to Impact Assessment

Populations[2] and communities[3] are the proper focus for evaluating adverse impacts of cooling-water withdrawals on the Hudson River estuary. The fundamental reason for focusing on populations and communities is that, whereas all individual organisms have finite life spans, populations and communities can persist. Because populations and communities can persist in spite of the inevitable mortality of the individual organisms, populations and communities can be managed and restored. Most commonly, fisheries management agencies establish harvesting policies to manage populations of fish while allowing harvesting of individual fish to continue (Restrepo et al. 1998). The U.S. Environmental Protection Agency ("USEPA") develops biological assessment methods, based on measures of aquatic community composition, to help states, tribes, territories, and interstate commissions identify communities that are impaired and in need of restoration (USEPA 2002). Established principles of population and community ecology underly both fisheries management and biological assessment. These scientific disciplines also provide a sound foundation for assessing impacts of entrainment and impingement on the biological resources of the Hudson River.

Our evaluation is primarily based on an analysis of empirical data collected over the 30 years during which IP2 and IP3 have been operating, in a manner that appropriately accounts for other potential causes of changes in fish populations. This is because factors other than entrainment and impingement affect the abundance of fish populations, including short-term

[2] A population is a group of plants, animals, or other organisms, all of the same species, that live together and reproduce (Gotelli 1995).

[3] A community is an assemblage of species populations that occur together in space and time (Begon et al. 1996).

natural environmental fluctuations, long-term environmental change, introductions of exotic species, pollution, and over-harvesting (Pew Oceans Commission 2003). The preamble to USEPA's Phase II Rule, 69 Fed. Reg. 41588 (July 9, 2004), also acknowledges the potential influence of these factors on Hudson River fish populations. Where potentially adverse changes in Hudson River fish populations have occurred over the past 30 years, we attempt to determine whether those changes are reasonably attributable to entrainment and impingement, or whether they are more likely to have resulted from other factors.

This impact assessment focuses on eight of the ten species identified for quantitative assessment in NYSDEC's October 1, 1992 Scope of Work for the DEIS: (1) striped bass; (2) white perch; (3) American shad; (4) Atlantic tomcod; (5) alewife; (6) blueback herring; (7) bay anchovy; and (8) spottail shiner. All of these species have been included in §316(b) studies for Indian Point and other Hudson River power plants since the 1970s (TI 1980). Six of these species, striped bass, white perch, Atlantic tomcod, alewife, and bay anchovy, were listed by USEPA as Representative Important Species ("RIS") for the Hudson River (TI 1980). Although not officially listed as RIS, blueback herring was included in the list of species studied because of its abundance in impingement collections at Indian Point, and American shad was included because of its commercial importance (TI 1980).

NYSDEC finalized the Scope of Work for the DEIS following a public scoping meeting and the integration of comments received from the generators, state and federal agencies, and environmental organizations. Two of the species identified in the Scope of Work, blue crab and shortnose sturgeon, are not addressed in this report. These two species are not addressed here because there is broad consensus that the CWIS at IP2 and IP3 have no impact on these species. *See, e.g.*, DEIS, p. V-125, 126 (sturgeon); Technical Comments on the DEIS, Pisces Conservation, Ltd., June 2000 ("Pisces Comments"), p. 27 ("There seems no basis for suggesting that power plants are linked to [changes in Atlantic and shortnose sturgeon abundance]."); DEIS, p. V-157 (based on preferred habitat, blue crab eggs and larvae not entrained at IP2 and IP3; very high impingement survival); Pisces Comments, p. 28-29 (numbers of blue crab within the estuary have risen dramatically since 1980).

2.1 Definition of "adverse environmental impact"

Neither §316(b) of the CWA (including USEPA's Phase II Existing Facilities Rule), nor New York regulation provides a definition of the term "adverse environmental impact." *See, e.g.*, 6 NYCRR §704.5. However, both regulations governing fisheries management in the United States and other USEPA guidance provide a foundation for a scientifically appropriate definition of this term.

2.1.1 Definition of adverse environmental impact in the context of fishery management

In the context of fisheries management, mortality *per se* could not be considered an AEI, because the act of fishing necessarily causes mortality. To the contrary, fisheries management agencies, including NYSDEC, actively encourage the responsible harvesting of fish. For example, NYSDEC has issued a guide to saltwater fishing in the New York City area (http://www.dec.ny.gov/outdoor/8377.html) that discusses equipment, fish identification, and specific fishing locations in all five New York City boroughs.

Fishery policy in waters under the control of the U.S. federal government, including estuaries and rivers utilized by anadromous fish, is established in the Magnuson-Stevens Fishery Conservation and Management Act ("Magnuson-Stevens Act"). The amended Act states:

> Fishery resources are finite but renewable. If placed under sound management before over-fishing has caused irreversible effects, the fisheries can be conserved and maintained so as to provide optimal yields on a continuing basis.

16 U.S.C. §1801(a)(5).

Federal guidelines implementing the Magnuson-Stevens Act state that "[c]onservation and management measures shall prevent over-fishing while achieving on a continuing basis, the optimum yield ("OY") from each managed fishery for the U.S. fishing industry." 70 Fed. Reg. 36240, 36250 (June 22, 2005). Thus, a fish population is viewed by managers as a renewable resource for which mortality in the form of harvesting is permissible, provided that this mortality does not threaten the long-term productivity of the population. Over-fishing that threatens the long-term sustainability of harvests is considered to be adverse. The National Oceanic and Atmospheric Administration ("NOAA") guidelines and other related technical guidance documents (e.g., Restrepo et al. 1998) provide specific procedures for determining whether overfishing is occurring. Fishery management councils are required to take action to reduce harvest levels if over-fishing is found to exist. 70 Fed. Reg. 36240, 36257 (June 22, 2005).

The Magnuson-Stevens Act is often cited as the "Sustainable Fisheries Act." The term "sustainable" is often used in a wider environmental policy context to refer to an approach to economic development and resource utilization that meets the needs of the present without compromising the ability of future generations to meet their own needs (World Commission on Environment and Development 1987). Sustainable uses of resources preserve those resources for future use; non-sustainable uses degrade or destroy the resources so that they may be unavailable in the future (World Commission on Environment and Development 1987).

Applying the definition of sustainable use provided by the World Commission on Environment and Development, sustainable use in the context of a fish population refers to a resource-management approach that permits the population to persist indefinitely into the future, while continuing to perform its normal ecological function and support normal human use. Ecological function is included as part of the definition of sustainable use of fish populations because fish have a role in the maintenance of healthy aquatic systems that can be compromised by over-fishing (Dayton et al. 2002). Predatory fish, such as striped bass, control the abundance of other fish species upon which they prey, and forage fish, such as bay anchovy, serve as both food for other fish species and as controls on the abundance of smaller organisms at the base of the marine food chain (Dayton et al. 2002). Over-fishing has led to a wide variety of direct and indirect changes in the structure and function of fish communities throughout the world (Dayton et al. 2002).

The sustainability of a population is a function of the abundance and other characteristics of the population (e.g., age and size structure) and also of the ability of members of the population to reproduce and replace themselves. Thus, with respect to the harvest-related mortality imposed on a fish population, an adverse impact consists of harvest-related reductions in abundance, changes in age/size structure, increases in mortality rates, or reduction in reproduction rates that threaten the capacity of the population to persist, perform its normal ecological function, and support normal human uses.

2.1.2 Definition of AEI in the context of ecological risk assessment

USEPA's Guidelines for Ecological Risk Assessment (USEPA 1998) provide a general discussion of adverse ecological effects of environmental stressors, including criteria for evaluating whether or not observed or predicted changes should be considered adverse. These guidelines were expressly issued to "set forth current scientific thinking and approaches for conducting and evaluating ecological risk assessments" (USEPA 1998, p. 8). This guidance discusses adverse ecological effects of environmental stressors, including criteria for evaluating whether or not observed or predicted changes should be considered adverse. According to USEPA and the scientific community, adverse ecological effects are changes that "alter valued structural or functional attributes of the ecological entities under consideration" (USEPA 1998, p. 106). USEPA (1998, p. 106) further states that the following criteria should be considered when determining whether an observed or predicted effect is adverse:

- Nature and intensity of effects;
- Spatial and temporal scale; and
- Potential for recovery.

"Nature and intensity of effects" refers to the types of effects that have occurred (or are predicted to occur), and the magnitude of the measured or predicted effects, the statistical significance of measured effects, and the ecological significance of the effects. "Spatial and temporal scale" refers to the size and location of the area within which an effect occurs, and the duration of the period required for the effect to appear. "Potential for recovery" refers to the expected rate and extent of return of an affected population or community following elimination of the stressor responsible for an effect that has been determined to be ecologically significant.

USEPA's definition and criteria for determining ecological adversity are consistent both with accepted principles of fishery management and with the current scientific understanding of the potential effects of harvesting on fish populations and communities. As noted in the introduction to this Section, in the context of §316(b) and §704.5, the ecological entities of interest are the populations and communities potentially affected by entrainment at CWIS. A definition of AEI of CWIS consistent with the Guidelines for Ecological Risk Assessment (USEPA 1998) should be expressed in terms of undesirable alterations in the structural or functional attributes of these populations and communities. An assessment whether adverse impacts have occurred (or will occur) should address the three criteria provided in the Guidelines.

2.1.3 Definition of adverse environmental impact in the context of entrainment and impingement

The definition of sustainable use in the Magnuson-Stevens Act and the definition of ecological adversity in USEPA's Guidelines for Ecological Risk Assessment provide a reasoned basis for a definition of AEI applicable to entrainment and impingement at CWIS. A sustainable approach to managing a fishery would ensure the long-term persistence and productivity of the population being managed. A non-sustainable approach, in contrast, would cause harvest-related reductions in abundance, changes in age/size structure, increases in mortality, or reductions in reproduction that could threaten the capacity of a population to persist, perform its normal ecological function, and support normal human uses. Since the ecological function of a population is understood by scientists to include interactions with other populations, non-sustainable use of a population can affect an entire community.

Abundance, age/size structure, mortality, and reproduction are examples of the "structural and functional attributes" discussed in the USEPA Guidelines. Hence, non-sustainable management of a fishery would be an example of an AEI according to USEPA's definition. Entrainment mortality differs from mortality caused by harvesting only in that the mortality is imposed on early life stages of fish or shellfish rather than on adults. Excessive levels of entrainment mortality could potentially affect most of the same structural and functional attributes affected by harvesting.

In sum, the term AEI, as it relates to entrainment and impingement, is reasonably and appropriately defined as follows:

> An adverse environmental impact due to entrainment and impingement consists of adverse changes in important population or community characteristics sufficient to threaten the sustainability of susceptible populations or to cause significant or potentially irreversible changes in population or community structure and function.

Such a definition would be consistent with recognized principles of both natural resource management and ecological risk assessment, as discussed above.

2.2 Why entrainment losses alone are insufficient to demonstrate AEI

Context is essential to understanding what the term AEI reasonably may mean with respect to fisheries biology. As a matter of science and logic, losses, even large numbers of early life stage individuals do not necessarily equate to AEI. This is because fish species inhabiting the Hudson River exhibit either "periodic" or "opportunistic" life history traits (Winemiller and Rose 1992). From an ecological perspective, periodic fish species are characterized by high fecundity (i.e., they spawn a large number of eggs), large size, and long life spans during which a female fish may spawn many times (Winemiller and Rose 1992). Striped bass is an example of a periodic species (Winemiller and Rose 1992). Opportunistic species are characterized by small body size, short life spans, and the ability to disperse offspring widely throughout the environment (Winemiller and Rose 1992). Bay anchovy is an example of an opportunistic species. Periodic and opportunistic traits are advantageous to fish species that live in unstable or unpredictable environments, such as the Hudson River, which experiences significant within-year and between-year variation in environmental conditions (e.g., temperature, salinity, freshwater flow, etc.). In other words, the reproductive strategies of these fish in these unstable conditions, including the very large numbers of eggs produced, ensure that sufficient offspring will survive to sustain the populations, even in unstable environments characterized by the presence of multiple stressors.

Entrainment losses consist mainly of eggs and larvae. Only a small fraction of the entrained fish would survive to adulthood, even if IP2 and IP3 did not exist. For example, an 18-year-old Hudson River striped bass was found to contain more than 3 million eggs (Hoff et al. 1988). A 16-year-old female striped bass examined by Olsen and Rulifson (1992) was found to contain nearly 5 million eggs. Since striped bass can live for up to 30 years (Secor and Piccoli 1996), a single fish could potentially spawn tens of millions of eggs over her entire lifespan. According to early life stage survival estimates developed by Secor and Houde (1995), more than 99.99% of young striped bass eggs die from natural causes within 60 days following spawning. Less than one striped bass egg in 100,000 is likely to survive to become a one-year-old fish, and less than one in a million is likely to survive to reach six years of age, the median age at which female striped bass become sexually mature (EPRI 2005).

Because nearly all of the eggs and larvae entrained at IP2 and IP3 would have died in any case, counts of total numbers entrained reveal nothing meaningful about the potential impact of IP2 and IP3 on fish populations. What matters is whether or not entrainment significantly reduces the number of fish that survive the early period of high natural mortality. As discussed in the next sections, this fact was recognized more than 30 years ago by the scientists who performed the first entrainment impact assessments for IP2 and IP3, in conjunction with other Hudson River generating stations.

2.3 Role of the conditional mortality rate (CMR) in impact assessment

The first assessments of the effects of cooling-water withdrawals on Hudson River fish populations, conducted on behalf of the Consolidated Edison Company of New York and various federal regulatory agencies were based on mathematical models that predicted the potential effects of entrainment losses on the abundance and other characteristics of fish populations, especially striped bass (Barnthouse et al. 1984). Many of these models were developed to support U.S. Atomic Energy Commission licensing proceedings for IP2 and IP3, and were incorporated in environmental impact statements prepared to support these proceedings (Barnthouse et al. 1984). At the time they were first developed, in the early and mid-1970s, modeling was undertaken because no actual fisheries data were available to test whether cooling-water withdrawals would have adverse impacts on important fish populations. When data from riverwide ichthyoplankton sampling became available in the late 1970s, scientists studying entrainment impacts developed an empirical model, termed the Empirical Transport Model ("ETM", Boreman et al. 1981), and used it to estimate the impact of entrainment on the abundance of juvenile fish. The metric calculated using the ETM, which was termed the "conditional mortality rate" ("CMR"), provides an estimate of the fraction by which the abundance of young-of-the-year fish is reduced due to entrainment. A similar model, termed the Empirical Impingement Model ("EIM", Barnthouse and Van Winkle 1988), was used to estimate a CMR for impingement.

It was recognized at the time that the CMR could not be used to predict long-term impacts on populations, however, because neither the ETM, nor the EIM, accounts for the density-dependent processes that can partially offset mortality due to entrainment and impingement (Barnthouse et al. 1984). CMRs could, however, be used to compare the relative potential effectiveness of alternative technologies intended to reduce entrainment and impingement mortality. As discussed by Englert et al. (1988), CMRs calculated using the ETM also were used to develop the cross-plant outage credits that were included in the Hudson River Settlement Agreement ("HRSA"). CMRs were also used in the DEIS to compare alternative entrainment mitigation approaches. In all of these applications, CMRs were used usefully as measures of mortality caused by entrainment and impingement, not as measures of the impacts of that mortality on the long-term abundance or sustainability of susceptible populations.

Because it does not account for density-dependent effects, the CMR is not a valid measure of long-term entrainment impacts. Depending on the strength of density-dependence in a given population, a particular CMR value corresponds to either a negligible or a substantial impact on the sustainability of a population.[4] CMRs can, however, be used as a measure of the annual rate of mortality imposed by entrainment and as inputs to assessment models that estimate the combined impacts of entrainment mortality and fishing mortality on the sustainability of populations (Goodyear 1977, 1993). For this assessment, CMRs are used for both of these purposes. They are not, however, used as measures of AEI, because CMRs are not appropriately used in that fashion and superior methods for assessing adverse impacts are available. As discussed in the following sections, analysis of long-term trends in the abundance of important Hudson River fish populations, available from 30 years of intensive data collection, is the best method available for assessing impacts of IP2 and IP3 on Hudson River fish populations. The trends analysis is supplemented by an analysis of the impacts of IP2 and IP3 on the sustainability of the Hudson River striped bass and American shad populations, using the SSBPR model.

2.4 Role of long-term datasets in impact assessment

Today, nearly 30 years of data are available from both generator and agency-sponsored monitoring programs. Together, these overlapping datasets provide information concerning long-term trends in the abundance and distribution of eggs, larvae, and juveniles of all of the species addressed in this report. For some commercially harvested species, data on long-term

---

4 Although there can be substantial uncertainty concerning the strength of density-dependence in specific populations, there is strong theoretical and empirical evidence that the great majority of biological populations, including fish populations, are regulated in part by density-dependent mechanisms (Murdoch 1994, Turchin 1999, Rose et al. 2001, Brook and Bradshaw 2006).

trends in the abundance, age distribution, and mortality of adult fish are available. These datasets can be used both to assess trends in the status of important fish populations and to test alternative hypotheses concerning potential causes of adverse changes.

In this report, information concerning long-term trends on key population characteristics and on the intensities of potential stressors is used to test specific hypotheses concerning the expected impacts of cooling-water withdrawals, termed "risk hypotheses" in USEPA's Guidelines for Ecological Risk Assessment (USEPA 1998). These hypothesis tests are used to distinguish changes that could have been caused by cooling-water withdrawals from changes that are most likely related to other causes.

The following generator-sponsored long-term datasets are the primary datasets used in assessing the effects of the CWIS at IP2 and IP3:

- *Longitudinal River Ichthyoplankton Survey ("LRS").* This program samples eggs, larvae, and juvenile fish, weekly from April through July. The region between the George Washington Bridge and the Federal Dam at Troy (Figure 1) has been sampled with only minor changes in methodology since 1974. In 1988, the LRS was extended to sample the region between the Battery and the George Washington Bridge.

- *Beach Seine Survey ("BSS").* This program samples juvenile fish, also called "young-of-the-year" fish ("YOY") (i.e., fish spawned earlier in the year) on alternate weeks from June through October. Sampling is conducted from the George Washington Bridge to the Federal Dam. The BSS has been conducted annually with only minor changes in methodology since 1974.

- *Fall Shoals Survey ("FSS").* This program samples YOY and older fish in offshore habitats, on alternate weeks from the BSS. Approximately 200 samples are collected per week, from Manhattan to the Federal Dam. The FSS uses two different gears in order to sample as much of the Hudson River as possible: a 1-m^2 Tucker trawl and a 3-m beam trawl. This program was also initiated in 1974, however, the beam trawl was not used until 1985. From 1974 through 1984 an epibenthic sled was used to sample near the river bottom. To ensure comparability between years, only the data collected from 1985 onward are used in this assessment.

- *Atlantic Tomcod Mark-Recapture Program.* This program has been conducted in most years since 1974 to generate estimates of the number of tomcod in the winter spawning population.[5] Box traps and bottom trawls are used to collect fish for marking and recapture.

The above datasets were selected as the primary datasets for this assessment because they have been conducted continuously since the mid-1970s. They cover nearly all of the period of commercial operation of IP2 (1973 startup) and all of the period of commercial operation of IP3 (1976 startup). These four datasets provide the most comprehensive and consistent estimates of long-term trends in the abundance of multiple life stages of important Hudson River fish populations. More detailed descriptions of these datasets are provided in ASA (2007).

A variety of other programs, conducted by the generators, NYSDEC, and federal resource management agencies provide information that can be used to test the validity of the primary trends data. These programs include:

- *Striped Bass Mark-Recapture Program.* This program was initiated in 1984, to estimate the contribution of the Hudson River striped bass hatchery (established as a condition of the HRSA) to the Hudson River population. The program targets 1-year-old and 2-year-old striped bass, and is conducted from November through March. Data from this program are used to estimate the numbers of striped bass greater than 150 mm in length overwintering in the lower estuary. Growth and survival rate estimates are also obtained from this program.

[5] The program was not conducted in 1984 and 1986.

- *NYSDEC Beach Seine Survey.* Since 1976, the NYSDEC Division of Marine Resources has conducted a beach seine survey in the lower Hudson River estuary. The program focuses on the Tappan Zee and Haverstraw Bay. It samples juvenile fish using a method similar, but not identical to, the generators' BSS.

- *Juvenile Alosid Survey.* NYSDEC conducts a beach seine survey in the middle and upper regions of the estuary (above River Mile 55) to estimate the relative abundance of YOY American shad and other juvenile fishes. This program was initiated in 1980 and continues to the present.

- *Western Long Island Survey.* NYSDEC conducts a survey for subadult striped bass in the bays around western Long Island Sound. Sampling is conducted using a 200-ft. beach seine. The program was initiated in 1984 and is continuing, although it has been modified over time.

- *Spawning Stock Assessment.* NYSDEC conducts a haul seine survey in the Hudson River to provide information on length, age and sex distribution, and mortality rates for adult American shad and striped bass. The program was initiated in 1982 and continues to the present.

- *Commercial Fishery Monitoring.* NYSDEC monitors the commercial gill net fishery for American shad. The objective of the program is to determine the relative abundance and age structure of the commercial catch of American shad.

As shown in Appendix A, indices derived from these datasets are strongly correlated with indices derived from the primary datasets. These correlations support the use of the primary datasets in this assessment.

In addition to the Hudson River monitoring programs, information on population status and trends for important fish species is also available from the National Marine Fisheries Service

("NMFS") and the Atlantic States Marine Fisheries Commission ("ASMFC"). Quantitative stock assessments, which include estimates of age structure, natural mortality, and fishing mortality, are available for striped bass (ASMFC 2005) and American shad (ASMFC 2007a). These assessments provide additional information for determining whether these populations have been harmed by CWIS.

2.5 Indicators of adverse impacts potentially related to CWIS

As discussed above, an adverse impact of CWIS would consist of entrainment and impingement-related adverse changes in important population or community characteristics sufficient to threaten the sustainability of relevant populations, or to cause significant or potentially irreversible changes in community structure and function. Characteristics that influence the sustainability of a fish population include the total size of the population, the relative abundances of different life stages or age groups, the sizes and reproductive rates of the individual fish, and the rates of mortality of fish at different life stages or ages. Measures of any of these population characteristics could, at least in principle, be used as indicators of adverse impact. Some of these measures are not suitable as indicators of adverse impacts potentially caused by CWIS, however, because they measure changes that cannot be reasonably attributed to cooling-water withdrawals. For example, a reduction in fecundity could be an indicator of a potential impact caused by a toxic chemical but, because impingement and entrainment do not affect fecundity, this characteristic is not an appropriate indicator of impacts caused by CWIS. Similarly, some indicators of impact are not particularly useful in narrowing the potential causes of impacts. For example, a prolonged downward trend in the abundance of adult fish could be the result of any number of causes, including over-fishing or environmental factors.

CWIS may impose mortality on early life stages of fish (i.e., eggs, larvae, and YOY) in addition to the mortality that would have occurred naturally. Therefore, characteristics that are either directly or indirectly affected by increased mortality of these life stages are potentially useful as indicators of harm related to CWIS. Increased mortality imposed on a particular life stage would reduce the fraction of organisms in that stage that survive to the next stage. Accordingly, this assessment focuses on whether CWIS have had a measurable influence on the survival of early life stages of fish in the Hudson River.

As discussed in Section 2.1 of this report, however, mortality of early life stages as a result of CWIS is insufficient, of itself, to establish that an adverse impact has occurred. It is necessary, in addition, to evaluate whether the magnitude, spatial extent, and duration of this mortality are large enough to constitute an adverse impact (USEPA 1998). Fisheries scientists have developed metrics, termed "biological reference points," for determining whether harvested fish populations are being harmed by over-fishing (Restrepo et al. 1998). These reference points, expressed in terms of either the total spawning stock biomass ("SSB") or the SSBPR, are viewed as indicators of the risk that over-fishing will lead to future declines in abundance and harvest. The methods that fisheries scientists use to estimate effects of fishing mortality on SSB and SSBPR can also be used to estimate impacts of entrainment-related mortality on SSB and SSBPR (Goodyear 1993). Hence, the indicators used to determine whether fish populations are being adversely affected by fishing can also be used as indicators of whether these same populations are being adversely affected by cooling-water withdrawals. Accordingly, for species for which published agency stock assessment reports provide relevant information, this assessment addresses whether the magnitude of entrainment mortality (as measured using the CMR) is sufficient to produce an ecologically significant reduction in SSB or SSBPR.

Information needed to estimate SSBPR is available for both striped bass and American shad. A coastwide SSB estimate is available for striped bass.

The following indicators have been selected for this assessment:

1. Long-term declines in the abundance of YOY fish belonging to species with life stages susceptible to impingement and entrainment, *see, infra*, Section 3;
2. Reductions in the spawning potential of female fish below the sustainable level as estimated using the SSBPR approach, *see, infra*, Section 4; and
3. Long-term trends in the abundance of species with high susceptibility to entrainment at IP2 and IP3 as compared to species with low susceptibility to entrainment at IP2 and IP3, *see, infra*, Section 5.

The analyses documented in Sections 3, 4, and 5 of this report evaluate whether any such declines or reductions in spawning potential have occurred and, if so, whether they may reasonably be attributed to the CWIS of IP2 and IP3.

3. Evaluation of changes in abundance of fish populations with life stages susceptible to entrainment

In complex ecological systems, such as the Hudson River estuary, fish populations are influenced by many factors in addition to CWIS, including water quality impairment, introductions of non-native species, and overfishing (Pew Oceans Commission 2003). Many of these factors are discussed in the preamble to USEPA's Final Phase II Existing Facilities Rule. 69 Fed. Reg. 41575, 41588 (July 9, 2004). For this reason, investigations of the causes of changes in fish populations must consider multiple hypotheses, weighing the evidence for and against each hypothesis (Hilborn and Mangel 1997, Suter et al. 2007). This approach has been termed "ecological detection" by Hilborn and Mangel (1997) and "ecoepidemiology" by Suter et al. (2007).

Most environmental factors affecting Hudson River fish populations vary in intensity over time. Knowledge of these variations can be used to predict the change in each metric that should have occurred, if that stressor had been affecting a particular fish population. To test each hypothesis, this analysis utilizes rules for evaluating causal associations provided by Suter et al. (2007, p. 50). These authors identified five criteria that should guide analyses of potential causes of adverse environmental effects:

1. *Co-occurrence*: An effect occurs where and when its cause occurs and does not occur in the absence of its cause.
2. *Sufficiency*: The intensity or frequency of a cause should be adequate to produce the observed magnitude of effect.
3. *Temporality*: A cause must precede its effect.
4. *Manipulation*: Changing the cause must change its effect.
5. *Coherence*: The relationship between a cause and effect must be consistent with scientific knowledge and theory.

Evaluations of co-occurrence discussed in this sections rely on a commonly-used and relatively straightforward statistical method known as correlation analysis (Clarke and Kempson 1997). In simple terms, correlation is a measure of whether two different variables are related to one another and, if so, how strong that relationship is (Clarke and Kempson 1997). A positive correlation between two variables indicates that as the value of one variable increases, so does the other. For example, height and weight among people are positively correlated. Although some taller people weigh less than shorter people, on average the taller a person is, the more that person is likely to weigh. Conversely, a negative correlation indicates that, as the value of one variable increases, the other decreases (Clarke and Kempson 1997). For example, weight and fuel efficiency among automobiles are negatively correlated. Although some heavier cars get better gas mileage than some lighter cars, on average the heavier a car is, the lower its gas mileage will be.

The existence and strength of correlations between stressor metrics and response metrics provides evidence concerning the co-occurrence criterion. If, for example, entrainment mortality at IP2 and IP3 is reducing the survival of eggs and larvae of a particular fish species, then there should be a negative correlation between entrainment mortality and a measure of the fraction of eggs and larvae that survive to reach older life stages. This means that in years when mortality due to IP2 and IP3 is high, survival should be relatively low, and in years when mortality due to IP2 and IP3 is low, survival should be high. Data showing the presence of a negative correlation between early life stage survival and IP2 and IP3-related mortality would constitute evidence supporting this impact hypothesis; data showing the absence of a correlation would constitute evidence against this hypothesis.

Evaluations of sufficiency in this assessment rely on measures of the magnitude of the stressor, as compared to the magnitude required to cause the observed response. For example, the rate of fishing mortality imposed on the striped bass and American shad populations can be compared to overfishing thresholds established by the ASMFC.

Evaluations of temporality in this assessment rely on time trends of the various stressor and response metrics. For any stressor to be a potential cause of a decline in the survival or abundance of a fish population, the decline should be preceded by an increase in the intensity of the stressor. If the decline in survival or abundance precedes the increase in the stressor, then the stressor cannot have caused the decline.

Evaluations of manipulation in this assessment rely on observations of responses of populations to deliberate changes in the magnitudes of stressors, e.g., the harvesting restrictions imposed on the striped bass fishery in the 1980s.

Evaluations of coherence in this assessment rely on the consistency of the responses with all relevant scientific information.

Because the focus of the permit proceedings is on entrainment and impingement of age 0 fish, the analysis will focus primarily on age 0 response metrics. The steps in the analysis include:

1. Develop a conceptual model of each stressor, including (1) a description of the stressor itself, (2) the reasonably expected causal mechanisms through which fish populations would be affected, (3) the species that would likely be affected, (4) the life stages (e.g., juveniles) that would likely be affected, (5) the life history characteristics (e.g., survival and growth) that would likely be affected, and (6) the type of measurable effects that would likely occur (increase or decrease);
2. Identify appropriate sets of "stressor metrics" and "response metrics" that can be used to test the potential influence of the various stressors;
3. Summarize the expected effect of the stressor on each response metric;
4. Apply the five evaluation criteria discussed above to the available data for each fish species; and
5. Summarize conclusions regarding (1) whether changes in the response metrics could have been caused by entrainment by CWIS at IP2 or IP3, or (2) whether other stressors are more likely to be responsible for these changes.

3.1 Species addressed

The DEIS assessed entrainment and impingement impacts on striped bass (*Morone saxatilis*), white perch (*Morone Americana*), Atlantic tomcod (*Microgadus tomcod*), bay anchovy (*Anchoa mitchilli*), American shad (*Alosa sapidissima*), alewife (*Alosa *pseudoharengus*), blueback herring (*Alosa aestivalis*), and spottail shiner (*Notropis hudsonius*) (DEIS, Sections 5 and 6). This report assesses entrainment and impingement impacts on these same species, focusing on the most economically important species (striped bass) and on the three species (white perch, American shad, and Atlantic tomcod) identified in the draft permit fact sheet as being of potential concern with respect to IP2 and IP3. Fact Sheet, Draft SPDES Permit, Attachment B, at 1 of 8. The datasets used in these analyses are documented in the 2005 Year Class Report (ASA 2007). The stressor and response metrics are documented in Appendix B.

3.2 Impact hypotheses and stressor metrics

This section documents expected effects of CWIS and four other stressors that are widely regarded as potentially having affected Hudson River fish populations: fishing, invasion of the Hudson River by zebra mussels (*Dresseina polymorpha*), temperature (Atlantic tomcod only) and predation by striped bass.

3.2.1 CWIS

CWIS may cause mortality of fish due to entrainment and impingement. For most species, this mortality is largely limited to eggs, larvae, and YOY. Because most of the susceptible life stages are planktonic[6] and are widely dispersed throughout the estuary due to tidal and nontidal flows, cooling-water withdrawals would not be expected to alter the spatial distributions of the affected species. In addition, the CWIS would not be expected to reduce the survival of fish that have grown through the most susceptible life stages, or to reduce fish growth rates at any life stage.

As discussed in Section 2.3, the CMR is a direct estimate of the rate of mortality caused by entrainment and impingement, independent from natural mortality. Similar measures are used by fisheries scientists to estimate the rate of mortality imposed on adult fish by fishing. The CMR can have values ranging between 0.0 and 1.0. The higher the value of the CMR, the greater the mortality imposed on early life stages of fish.

---

[6] Planktonic organisms are small organisms such as fish larvae that have limited swimming capabilities and are passively transported up and downriver with tidal currents.

Expected effects of CWIS on the life stages potentially susceptible to entrainment and impingement (i.e., eggs, larvae, and YOY) are summarized in Figure 2. As shown in Figure 2, CWIS should affect the survival rates of the susceptible life stages, but should not affect the survival of stages that are not susceptible to entrainment or impingement. If entrainment or impingement were having a measurable impact on a fish population, then in years when the IP2 and IP3 CMR is high, the survival rates of susceptible life stages of that species should be lower than in years when the IP2 and IP3 CMR is low. As a consequence, long-term trends in IP2 and IP3 CMR values for that species should be negatively correlated with long-term trends in the survival rates of susceptible life stages.

Although entrainment would not affect the number of eggs spawned by females of susceptible species, it is still possible that entrainment could directly affect the abundance of early life stages. The reason for this is that the LRS is conducted during the period in which entrainment at IP2 and IP3 is occurring. Therefore, entrainment could affect the abundance estimates derived from LRS data. If entrainment at IP2 and IP3 is reducing early life stage abundance, then the IP2 and IP3 CMR values should also be negatively correlated with PYSL abundance estimates.

3.2.2 Fishing

Fishing imposes mortality primarily on harvestable-sized[7] fish.[8] For managed Hudson River fish species (i.e., striped bass and American shad), harvesting is largely limited to age 1 and older fish (ASMFC 1998, 2002). Fishing has predictable effects on the age distribution of adult fish and on the abundance (numbers and biomass) of the spawning stock (Dayton et al. 2002). Measures of age distribution and spawning stock abundance are used by fisheries managers as indicators of fishing (Restrepo et al. 1998). Fishing reduces the total reproductive output of a fish population (Goodyear 1993).

The most appropriate estimate of stress due to fishing is the annual rate of fishing mortality (*F*) imposed on the population. Estimates of *F* for two of the species addressed in this analysis, striped bass and American shad, are available from the ASMFC.

---

7 Harvestable-size fish are fish that fall within the size range for which harvesting is permitted.

8 Fish outside the permitted range are frequently caught by trawls and other fishing gear. Although they are returned to the ocean, substantial mortality may still occur. This mortality is termed "bycatch" mortality.

Expected effects of fishing on age 0 life stages are summarized in Figure 3. Over-harvesting reduces the size of the adult population and necessarily the total number of eggs produced per year. The reduction in egg production would be expected to reduce the number of eggs surviving to become one-year-old fish. Fishing should not reduce the survival or growth rate of any age 0 life stage, however, because early life stages of fish are not susceptible to harvesting.

3.2.3 Zebra mussels

Zebra mussels invaded the Hudson River in the early 1990s (Caraco et al. 1997). Zebra mussels form dense beds on the bottom of colonized water bodies. Because of their high filtering capacity, zebra mussels remove phytoplankton from the water column, thus reducing the food base that supports pelagic fish larvae, such as American shad, striped bass, and white perch (Strayer et al. 2004). Because less food is available to support fish species that feed in open water, the survival and growth of these species may decrease. The increased water clarity caused by zebra mussel filtration can result in improved growth of rooted vegetation. The survival and growth of species that inhabit vegetated areas may increase because of increased habitat availability (Strayer et al. 2004). Zebra mussels are limited to fresh water, and are not found in substantial numbers below approximately river kilometer ("RKM") 100 in the Hudson River. For this reason, zebra mussels could potentially alter the spatial distributions of some species, reducing their abundance above RKM 100 as compared to below RKM 100.

There is no readily available quantitative metric for zebra mussel abundance. Due to the discontinuous nature of the zebra mussel invasion (absent prior to 1992; highly abundant after 1992), however, the qualitative evaluation can use presence/absence to develop predicted effects, and the quantitative analysis can use a simple index to distinguish between these two periods (e.g., "0" for all years prior to 1993 and "1" for 1993 and later). Expected effects of zebra mussels on age 0 life stages are summarized in Figure 4. Zebra mussels would be expected to reduce the survival and growth rates of post yolk-sac larvae and YOY utilizing freshwater regions of the Hudson River. These changes in survival and growth could result in a shift in the relative abundance of YOY present in predominantly freshwater regions (Regions 6-12; Figure 1) as compared to marine and brackish regions (Regions 0-5; Figure 1). Specifically, if zebra mussel activity reduces the growth and survival of pelagic fish species in freshwater regions as compared to marine and brackish regions, then during the post-invasion period a greater fraction of the populations of pelagic species, such as striped bass, white perch, alewife, and river herring, should be found in marine and brackish regions than during the pre-invasion period.

3.2.4. Predation by striped bass

Increased abundance of yearling and older striped bass, which are piscivorous[9] (Gardinder and Hoff 1982, Walter et al. 2003), could lead to increased predation mortality. Savoy and Crecco (2004) have attributed a recent decline in American shad and blueback herring populations in the Connecticut River to predation by large adult striped bass on spawning adults of these species.

Because the abundance of striped bass early life stages has been found to be strongly correlated with the relative abundance of adults (Pace et al. 1993; Barnthouse et al. 2003), estimates of striped bass larval abundance from the LRS can be used as a surrogate for adult striped bass abundance.

Predation on adults would, like harvesting, reduce the number of spawning adults and, as a consequence, the number of eggs spawned. The reduction in egg production would be expected to reduce the number of eggs surviving to become one-year-old fish. Predation on YOY would directly reduce YOY abundance, over and above and reductions resulting from reduced egg production (Figure 5).

3.2.5 Temperature

Changes in temperature can cause either increases or decreases in the growth and survival of affected species, depending on species-specific temperature tolerances. Long-term trends in Riverwide temperatures could potentially lead to long-term changes in the abundance of sensitive species, such as Atlantic tomcod (FEIS, pp. 65-66). Expected effects of elevated summer temperatures on age 0 temperature sensitive species are summarized in Figure 6. Elevated summer temperatures would be expected to cause decreases in survival and growth of temperature-sensitive species during this period. Growth and survival of early life stages would

9 Piscivorous fish are fish that eat other fish.

not be depressed, however, because these life stages are present only during the winter and early spring, when temperatures would be well below adverse effects thresholds.

According to McLaren et al. (1988), the growth of juvenile Atlantic tomcod in the Hudson River ceases during the summer when river temperatures regularly exceed 25°C. The lethal temperature for juvenile Atlantic tomcod is 26.5°C (McLaren et al. 1988). Temperature records available from the Poughkeepsie Water Works (PWW) were used to develop a degree-day index for evaluating the potential effects of elevated summer temperatures on Atlantic tomcod. A degree-day is defined as the number of degrees by which the temperature measured at the PWW on that day exceeds 24°. If, for example, the temperature measured at the PWW on a given date was 27°C, then the degree-day value for that date would be 3. If the temperature on a date is 24° or less, then the degree-day value for that date is recorded as 0. The degree-day index for a years is calculated by summing the degree-days for all days during that year.

3.3 Response metrics

Because not all data sets are suitable for evaluating all species, the response metrics used in this assessment are not the same for all species.

3.3.1 Response metrics for striped bass, white perch, American shad, alewife, blueback herring, and bay anchovy

For species other than spottail shiner and Atlantic tomcod, the LRS and BSS provide the most reliable data concerning survival, growth, and spatial distribution. Because the durations of egg and YSL life stages are comparatively short, such that individuals can hatch and develop through one or both of these stages between survey dates, most of the fish captures in the LRS are PYSL. The PYSL stage is typically much longer, so that PYSL are susceptible to sampling for at least one and possibly two or more survey dates. For these reasons, estimates of total larval abundance from the LRS are best interpreted as estimates of the abundance of PYSL. Although the beach seine used in the BSS and the beam trawl used in the FSS do not capture larvae, they effectively sample YOY fish present in the sampled habitats (shore zone for the BSS and shoal zone for the FSS). The response variables that can be calculated from the generators' survey data are:

1. Abundance of PYSL, as measured in the LRS;
2. Survival from the PYSL to the YOY stage, as measured by the ratio of densities of larvae in the LRS dataset to densities of juveniles in the BSS or FSS,
3. Abundance of YOY, as measured in the BSS or FSS;
4. YOY growth, as measured by the average length of YOY fish from the BSS or FSS; and
5. Spatial distribution of PYSL and YOY relative to river regions with high zebra mussel densities, as measured by the per cent of the total population occurring downriver from RKM 100.

3.3.2 Response metrics for spottail shiner

Because the LRS does not adequately sample areas of the Hudson River inhabited by spottail shiner, for this species, no estimates of egg and larval abundance are available. However, the BSS provides estimates of both YOY abundance and adult abundance (age 1 and 2 adults) for this species. For the purpose of trends analysis, adult abundance is used as a surrogate for egg production.

3.3.3 Response metrics for Atlantic tomcod

Because a substantial fraction of Atlantic tomcod larvae and YOY occur downriver from the regions sampled by the generators' surveys, for Atlantic tomcod, the data provided by the Atlantic tomcod mark-recapture program should be more reliable than the LRS, BSS, or FSS data for estimating survival rates. The mark-recapture program provides annual estimates of age-1 abundance, spawning stock size, and total egg production that can be used to calculate the fraction of eggs produced during a given year that survive to become age-1 spawners the following year. The LRS data can be used to characterize both year-to-year variations in early life stage abundance and the distribution of Atlantic tomcod larvae and juveniles within the Hudson River.

For this species, the response variables include:

1. Abundance of PYSL and early juveniles, as estimated from the LRS;
2. Abundance of Age-1 and Age-2 fish, as estimated from the mark-recapture program;
3. Total age 0 survival, as measured by the ratio of total egg production each year to age 1 abundance during the following year;
4. Juvenile growth, as measured from growth rates of juveniles from the FSS; and
5. Spatial distribution of PYSL and early juveniles, as measured by the fraction of the total PYSL/juvenile population found in river regions 1-5 (LRS dataset).

3.4 Tests of impact hypotheses

The predicted impacts of the stressors on the response metrics are summarized below and in Tables 1 (striped bass, white perch, American shad, river herring, bay anchovy, and spottail shiner) and 2 (Atlantic tomcod):

- ***CWIS***: Entrainment at IP2 and IP3 would be expected to reduce survival from the PYSL to the YOY stage, and could also reduce the abundance of PYSL. Entrainment should have no effect on growth or spatial distribution.
- ***Fishing***: Fishing would be expected to reduce the abundance of eggs and early larvae because of reduced spawner abundance, but should not reduce the survival of any age 0 life stage.
- ***Zebra mussels***: Zebra mussel activity would be expected to decrease both PYSL survival and YOY growth, and also to shift the spatial distribution of juveniles toward the lower regions and away from the freshwater regions where zebra mussels are abundant.

- ***Temperature***: Since Atlantic tomcod are known to be sensitive to high summer water temperatures, increased summer temperatures would be expected to decrease the growth and survival rates of life stages of this species that are present in the Hudson during this season.
- ***Striped bass predation***: Predation by older striped bass would be expected to decrease juvenile abundance, if the juveniles are susceptible to predation, and early life stage abundance, if adults are susceptible to predation.

Appendix B documents the stressor and response metrics and statistical methods used in this analysis. The subsections below present the results of the analyses performed for each species, and evaluate the consistency of these results with the impact hypotheses.

3.4.1 Striped bass

Figure 7a depicts long-term trends in the abundance of striped bass PYSL and YOY in the Hudson. Figure 7b depicts long-term trends in striped bass PYSL to YOY survival. The abundance of juvenile striped bass in the Hudson has shown no trend, even though the abundance of striped bass early life stages has greatly increased. The increase in abundance of striped bass larvae has occurred concurrently with an increase in the abundance of the Hudson River spawning stock of striped bass (Barnthouse et al. 2003). The increase in spawning size has been attributed to coastwide restrictions on harvesting that were imposed to promote the recovery of the Chesapeake Bay striped bass stock (Young-Dubovsky et al. 1995). As first noted by Pace et al. (1993), and later confirmed by Barnthouse et al. (2003), there is no correlation between the abundance of striped bass PYSL and striped bass YOY (Figure 8a). There is a strong negative relationship between PYSL abundance and PYSL survival, however (Figure 8b). This negative correlation has been interpreted by both Pace et al. (1993) and Barnthouse et al. (2003) as evidence for density-dependent mortality of striped bass larvae. This density-dependent mortality is reflected in the long-term trend in PYSL to YOY survival (Figure 7b), which has declined through time as the size of the spawning population has increased.

3.4.1.1 *CWIS*

Co-occurrence

Appendix B (Tables B-11 and B-12) summarizes the results of the correlation analysis for striped bass. If entrainment at IP2 and IP3 were reducing the survival or abundance of early life stages of striped bass, then there should be a negative correlation between the CMR and striped bass PYSL survival, PYSL abundance, or both. However, as shown in Figure 9, there is no correlation between the IP2 and IP3 CMR and either PYSL survival (Figure 9a) or PYSL abundance (Figure 9b) for striped bass. Hence, the CWIS hypothesis fails the co-occurrence criterion for striped bass.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence, the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

If entrainment at IP2 and IP3 were reducing the survival or abundance of early life stages of striped bass, then a decline in PYSL survival, or PYSL abundance should have occurred after the startup of commercial operations of IP2 (1974) and IP3 (1976). However, as shown in Figure 7, no such declines occurred. PYSL abundance was relatively stable until 1985, and then rapidly increased. Striped bass PYSL survival has declined over time (Figure 7b), but the decline did not begin until several years after the startup of IP2 and IP3. Hence, the CWIS hypothesis fails the temporality criterion for striped bass.

Manipulation

No experimental manipulations of plant operations have been performed for the purpose of evaluating entrainment impacts on fish populations. However, outages, including refueling and maintenance outages mandated by the HRSA (Englert et al. 1988), have frequently occurred during the months when entrainable striped bass are present in the River. The peak abundance of striped bass eggs and larvae typically occurs during May and June (Boreman and Klauda, 1988). IP2 was offline during the entire months of May and June in 1976, 1989, 1991, 1997, 1998, and 2000. IP3 was offline during the entire months of May and June in 1975, 1982, 1993, and 1994. If entrainment at Indian Point were reducing the survival of striped bass PYSL, then PYSL survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 10a, the measured PYSL survival values are inconsistent with this expectation. Figure 10a shows the time series of annual PYSL survival indices from 1975 through 2002. The horizontal line in Figure 10a shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If striped bass PYSL survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 10a shows that the PYSL survival index was higher than the median for only 3 of the 11 years of one-unit operation. The PYSL index was lower than the median in 8 years of one-unit operation.

This result is confirmed by Figure 10b, which shows the relationship between the striped bass PYSL survival index and the May-June total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and striped bass PYSL survival. Hence, the CWIS hypothesis fails the manipulation criterion for striped bass.

Coherence

As noted above, the objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.1.2 Fishing

Co-occurrence

Fishing indirectly affects the abundance of early life stages of fish by reducing the abundance of spawning adults (Goodyear 1993). If a population is being overfished, then reducing the rate of fishing should cause the spawning population, and therefore the number of eggs spawned, to increase. As discussed by Young-Dubovsky et al. (1994), a coastwide ban on harvesting of striped bass was imposed in 1986. Estimates of fishing mortality and adult population abundance developed by the ASMFC (2005) show that the coastwide adult population has increased greatly since 1986. As shown in Figure 7a, the abundance of striped bass PYSL began increasing in 1988 and increased steadily throughout the 1990s. This is the same period during which the adult striped bass population was expanding. Hence, the overfishing hypothesis satisfies the co-occurrence criterion.

Sufficiency

Fishing mortality estimates for individual striped bass spawning stocks are not estimated by the ASMFC, because much of the fishing occurs along the Atlantic coast when fish from the individual spawning stocks are mixed (ASMFC 2003). Since the magnitude of fishing mortality imposed specifically on Hudson River striped bass has never been estimated, it is not possible to determine whether the fishing hypothesis satisfies the sufficiency criterion.

Temporality

The ban on striped bass harvesting preceded the increase in abundance of striped bass PYSL in the Hudson River by approximately 2 years. Hence, the fishing hypothesis satisfies the temporality criterion.

Manipulation

The 1986 ban on striped bass harvesting was described by Young-Dubovsky et al. (1996) as an "adaptive management experiment." In other words, fishing was deliberately reduced in order to observe the response of the striped bass population to reduced harvesting. The fact that the adult population of striped bass began to increase immediately following the ban was interpreted by Young-Dubovsky et al. (1994) as strong evidence that overfishing was, if not the only cause, at least the primary cause of the depressed abundance of Atlantic striped bass prior to the ban. Because the response of the population to this management was consistent with the expectations from the fishing hypothesis, the fishing hypothesis satisfies the manipulation criterion.

Coherence

Atlantic striped bass are managed as a single coastwide fishery because a large fraction of the harvest occurs when fish originating in Chesapeake Bay, the Delaware River, and the Hudson River are mixed and migrating along the Atlantic coast (ASMFC 2003, Waldman et al. 1990, Waldman and Fabrizio 1994). If reduced harvesting had been the cause of increases in the abundance of early life stages of striped bass in the Hudson River, then similar increases should have occurred in the Chesapeake Bay and the Delaware River as well. As shown in the ASMFC's 2003 stock assessment, the abundance of juvenile striped bass in both Chesapeake Bay and the Delaware River grew rapidly after the harvest ban. Hence, the overfishing hypothesis is consistent with the coherence criterion.

3.4.1.3 Zebra mussels

Co-occurrence

As documented in Appendix B (Table B-11), the zebra mussel index is negatively correlated with the striped bass PYSL survival index. This correlation is consistent with the zebra mussel hypothesis. Hence, the zebra mussel hypothesis satisfies the co-occurrence criterion.

Sufficiency

The potential effects of zebra mussel activity on early life stages of fish are indirect, and related to reductions in prey abundance and changes in habitat quality. No experiments have been performed that could quantify the relationship between zebra mussel activity and fish growth or survival, and no mathematical models that could be used to quantify the indirect effects of zebra mussel activity have been developed. Hence, whether or not the zebra mussel hypothesis satisfies the sufficiency criterion is unknown.

Temporality

Zebra mussels first became abundant in the Hudson River in 1992 (Caraco et al. 1997). However, as shown in Figure 7b, striped bass PYSL survival began declining in the 1980s and had already fallen to a very low level by 1990. Because the decline in striped bass PYSL survival preceded, rather than followed, the appearance of zebra mussels in the River, the zebra mussel hypothesis fails the temporality criterion.

Manipulation

No deliberate manipulations of zebra mussel populations in the Hudson River have been performed, therefore, this criterion is inapplicable to the zebra mussel hypothesis.

Coherence

Because the proposed mechanism through which zebra mussel activity could have affected striped bass in the Hudson River involves reducing food availability, the growth as well as the survival of striped bass PYSL and YOY should have been reduced. Although Strayer et al. (2004) found a negative relationship between the growth rate of YOY striped bass and the presence of zebra mussels, no significant correlation was found in the analyses performed to support this report (Appendix B, Table B-11). Zebra mussel activity should also have shifted the distribution of striped bass PYSL and YOY downriver, away from the freshwater zone in which zebra mussels are abundant. Strayer et al. (2004) found no downstream shift in the distribution of striped bass PYSL and YOY. In the analyses performed to support this report (Appendix B, Table B-11), no downstream shift in the distribution of PYSL was found, and an upstream shift (i.e., a shift in the opposite direction from the shift predicted by the zebra mussel hypothesis) in the distribution of YOY was found. The negative effect of zebra mussel activity on striped bass YOY growth that was reported by Strayer et al. (2004) conflicts with the findings in Appendix B, moreover, neither Strayer et al. (2004) nor the present analysis (Appendix B) found the predicted relationship between zebra mussel activity and striped bass PYSL and juvenile distribution. Hence, the zebra mussel hypothesis fails the coherence criterion for striped bass.

3.4.1.4 Summary evaluation of hypotheses

Table 3 summarizes the consistency of the striped bass trends data with the CWIS, overfishing, and zebra mussel hypotheses. Two of the five evaluation criteria – sufficiency and coherence – are inapplicable to the CWIS hypothesis. However, this hypothesis fails all three of the remaining criteria. Hence, the CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 striped bass in the Hudson River. The zebra mussel hypothesis passes the co-occurrence criterion, but fails the temporality and coherence criteria. Because striped bass PYSL survival declined several years prior to the invasion of the Hudson River by zebra mussels, and because predicted effects of zebra mussels on the growth and distribution of striped bass PYSL and YOY were not observed, the zebra mussel hypothesis also can be rejected as an explanation for long-term trends in the abundance of age 0 striped bass in the Hudson River.

The overfishing hypothesis, in contrast, passes four of the five evaluation criteria. The remaining criterion (sufficiency) is inapplicable to this hypothesis. The abundance of striped bass PYSL in the Hudson began increasing shortly following a reduction in striped bass harvesting. The reduction in harvest was specifically intended to promote striped bass reproduction, and was followed by simultaneous increases in striped bass reproductive success in all three of the major east coast spawning populations. It is reasonable to conclude, therefore, that elimination of overfishing is the most likely cause of trends in the abundance of early life stages of striped bass in the Hudson River.

3.4.2 White perch

Figure 11 depicts long-term trends in the abundance of white perch YOY and PYSL in the Hudson. As shown in Figure 11, the abundance of juvenile white perch declined steadily throughout the 1980s, but has increased since 1990. Despite the recent increase, over the entire time series, there is a statistically significant decline in YOY abundance (Appendix B, Table B-13 and Figure B-4). There is no long-term trend in the annual abundance of PYSL (Figure 11), however, which suggests that larval production is stable. There is no relationship between PYSL abundance and YOY abundance in white perch (Figure 12a). The survival rate of white perch from the PYSL to the juvenile stage has declined (Appendix B, Table B-13). Moreover, there is a strong positive relationship between PYSL survival and YOY abundance (Figure 12b, Appendix B, Table B-14). Because YOY abundance in white perch is closely related to PYSL survival but not to PYSL abundance, we can conclude that the decline in YOY abundance was due to a decline in PYSL survival rather than to a decline in white perch reproduction.

3.4.2.1 CWIS

Co-Occurrence

Appendix B, Table B-13 and B-14 summarize the results of the correlation analysis for white perch. If entrainment at Indian Point had caused the observed decline in white perch PYSL survival, there should be a negative relationship between the entrainment CMR for white perch and white perch PYSL survival. This means that in years when the CMR was high, white perch PYSL survival should have been low, and in years when the CMR was low, white perch PYSL survival should have been high. However, as shown in Figure 13a, the opposite relationship exists. The IP2 and IP3 CMR is *positively* correlated with PYSL to juvenile survival, meaning that the CMR was high in years when PYSL survival was high and the CMR was low in years when PYSL survival was low.

There is a negative relationship between the IP2 and IP3 CMR and white perch PYSL abundance (Figure 13b), but this correlation is significant only at the 10% level. Figure 14 plots time trends in both the CMR and in PYSL to juvenile survival for white perch. The two trend lines show similar patterns, with values decreasing from the mid-1970s to the mid-1980s, fluctuating until the mid-1990s, and then increasing. It is important to note that the recent increase in survival occurred during a period in which the capacity factors for IP2 and IP3 have been higher than in earlier years (Darla Gray, Entergy Corp., personal communication).

Although there is a weak negative relationship between the CMR for IP2 and IP3 and white perch PYSL abundance, the much stronger positive relationship between the CMR and PYSL to YOY survival must be accorded a higher weight. Because this positive correlation clearly conflicts with the CWIS hypothesis, the CWIS hypothesis fails the co-occurrence criterion for white perch.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence, the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

As shown in Figure 14, white perch PYSL survival began to decline in 1977, one year following the startup of commercial operation at IP3. Since the startup of 2-unit operation preceded the decline in white perch PYSL survival, the CWIS hypothesis satisfies the temporality criterion.

Manipulation

As discussed in Section 3.4.1.1, outages of IP2 or IP3 have frequently occurred during the entrainment season at Indian Point. The peak abundance of white perch eggs and larvae typically occurs during May and June (Klauda 1988). IP2 was offline during the entire months of May and June in 1976, 1989, 1991, 1997, 1998, and 2000. IP3 was offline during the entire months of May and June in 1975, 1982, 1993, and 1994. If entrainment at Indian Point were reducing the survival of white perch PYSL, then PYSL survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 15a, the measured PYSL survival values are inconsistent with this expectation. Figure 15a shows the time series of annual PYSL survival indices from 1975 through 2002, which are the years for which cooling water flow data were available. The horizontal line in Figure 15 shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If white perch PYSL survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 15a shows that the PYSL survival index was higher than the median for only 4 of the 11 years of one-unit operation. The PYSL index was equal to the median in one year (1989) of one-unit operation, and lower than the median in 6 years of one-unit operation.

This result is confirmed by Figure 15b, which shows the relationship between the white perch PYSL survival index and the May-June total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and white perch PYSL survival. Hence, the CWIS hypothesis fails the manipulation criterion for white perch.

Coherence

As noted above, the objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.2.2 Zebra mussels

Co-Occurrence

As shown in Appendix B, Table B-13, the zebra mussel index is negatively correlated with PYSL to YOY survival in white perch. Hence, the zebra mussel hypothesis satisfies the co-occurrence criterion.

Temporality

As shown in Figure 14, however, the decline in white perch PYSL to YOY survival occurred primarily between 1974 and 1986, prior to the zebra mussel invasion. PYSL to YOY survival has actually been increasing since 1993, the first year in which zebra mussels were abundant enough to potentially affect fish populations (Strayer et al. 2004). Hence, the zebra mussel hypothesis fails the temporality criterion.

Sufficiency

The potential effects of zebra mussel activity on early life stages of fish are indirect, and related to reductions in prey abundance and changes in habitat quality. No experiments have been performed that could quantify the relationship between zebra mussel activity and fish growth or survival, and no mathematical models that could be used to quantify the indirect effects of zebra mussel activity have been developed. Hence, whether or not the zebra mussel hypothesis satisfies the sufficiency criterion is unknown.

Manipulation

No deliberate manipulations of zebra mussel populations in the Hudson River have been performed, therefore, this criterion is inapplicable to the zebra mussel hypothesis.

Coherence

Because the proposed mechanism through which zebra mussel activity could have affected white perch in the Hudson River involves reducing food availability, the growth as well as the survival of white perch PYSL should have been reduced. Although Strayer et al. (2004) reported a negative relationship between zebra mussel activity and white perch growth, the analysis performed to support this assessment (Appendix B, Table B-13) found no significant relationship between zebra mussels and white perch growth. Moreover, the percent of white perch juveniles downriver from RKM 100 is negatively, instead of positively, correlated with the zebra mussel index (Appendix B, Table B-13). This negative correlation implies that over this same period of years, the percentage of the population present downriver from RKM 100 has declined, rather than increasing as predicted by the zebra mussel hypothesis. This result is also consistent with the findings of Strayer et al. (2004). Hence, the zebra mussel hypothesis partially, but not fully, satisfies the coherence criterion.

3.4.2.3 Striped bass predation

Co-occurrence

There is a weak negative correlation between the striped bass index and the white perch PYSL index (Appendix B, Table B-13). This relationship provides weak evidence supporting the hypothesis that striped bass are preying on adult white perch. There is much stronger negative correlation between the striped bass index and the YOY index (Figure 16a). This correlation is consistent with the hypothesis that striped bass are preying on juvenile white perch. There is also a strong negative correlation between the striped bass index and white perch PYSL to YOY survival, however, this relationship is difficult to interpret because striped bass would not be expected to prey on larval white perch. Overall, the striped bass hypothesis satisfies the co-occurrence criterion with respect to predation on YOY white perch.

Sufficiency

Striped bass larger than 200 mm in length have been shown to feed on white perch (Gardinier and Hoff 1982, Dunning et al. 1997). Appendix C to this report documents an analysis of prey consumption by Hudson River striped bass. This analysis compares the change in striped bass prey consumption requirements (August through October) between earlier (1983-1990) and more recent (1991-2004) periods to changes in abundance of YOY fish in the Hudson River between these same two periods. The analysis shows that the increase in prey consumption from the earlier to the later period would be sufficient to explain the decline in YOY white perch abundance between these two periods if 1% of the age 1 and age 2 striped bass seasonal predatory demand was satisfied by YOY white perch, or if 0.3% of the age 1 through age 13 striped bass seasonal predatory demand was satisfied by YOY white perch. Hence, the striped bass predation hypothesis satisfies the sufficiency criterion for white perch.

Temporality

A sustained decline in white perch YOY abundance began in 1989, at the same time the striped bass index began to increase (Figure 16b). However, the historic peak in YOY abundance occurred in 1980 (Figure 16b), and PYSL to YOY survival declined substantially between 1975 and 1985 (Figure 14). White perch PYSL to YOY survival and YOY abundance are strongly correlated (Figure 12b), implying that declining YOY abundance must have been at least in part caused by a decline in PYSL to YOY survival. The decline in PYSL to YOY survival that declined between 1975 and 1985 cannot be explained by striped bass predation. Hence, the striped bass predation hypothesis only partially satisfies the temporality criterion.

Manipulation

No deliberate manipulations of striped bass predation in the Hudson River have been performed, therefore, this criterion is inapplicable to the striped bass hypothesis.

Coherence

If predation by striped bass had caused the decline in abundance of YOY white perch in the Hudson River, then the YOY abundance of other known striped bass prey species, including river herring, American shad, bay anchovy, and Atlantic tomcod should also have declined. As shown in other Sections of this report, YOY abundance for all of these species has declined since the late 1980s, when striped bass abundance began to increase. Moreover, other published studies have concluded that striped bass predation is reducing the abundance of some prey species. Savoy and Crecco (2004) attributed recent declines in the abundance of both blueback herring and American shad in the Connecticut River to striped bass predation. Hartman (2003) estimated that the coastwide annual prey consumption by striped bass between 1 and 10 years of age increased by more than a factor of 8 between 1982 and 1995, from 17,900 metric tons (mt) to 147,900 mt. Uphoff (2003) calculated even larger estimates of striped bass consumption, and attributed a 90% decline in the abundance of Atlantic menhaden in upper Chesapeake Bay from 1980 through 1999 to predation by striped bass.

Because parallel declines in other susceptible species have occurred, and because the other published studies have documented the influence of striped bass predation on susceptible prey species, the striped bass predation hypothesis satisfies the coherence criterion.

3.4.2.4 Summary evaluation of hypotheses

Table 4 summarizes the consistency of the white perch trends data with the CWIS, zebra mussel, and striped bass predation hypotheses. Two of the five evaluation criteria – sufficiency and coherence – are inapplicable to the CWIS hypothesis. The CWIS hypothesis fails the co-occurrence and manipulation criteria. Although the CWIS hypothesis satisfies the temporality criterion because the observed decline in white perch PYSL survival followed the startup of IP2 and IP3, the inconsistency of this hypothesis with the co-occurrence and manipulation hypotheses means that the temporal correspondence between the beginning of the decline in survival and the startup of IP2 and IP3 is very likely a coincidence. Hence, the CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 white perch in the Hudson River.

The zebra mussel hypothesis passes the co-occurrence criterion and at least partially satisfies the coherence criterion. However, it fails the temporality criterion because the declines in white perch PYSL survival and YOY abundance began prior to the appearance of zebra mussels in the Hudson River. Although zebra mussel activity might have contributed to a decline in white perch PYSL to YOY survival and YOY abundance from 1993 to 2004, zebra mussels could not have been the primary explanation for long-term trends in white perch survival and abundance.

The striped bass predation hypothesis satisfies four of the five criteria. The fifth, manipulation, is inapplicable to this hypothesis. However, the strong relationship between white perch PYSL survival and YOY abundance over the entire period from 1974 to 2004 (Figure 12b) cannot be explained by the predation hypothesis, because striped bass abundance did not begin to increase until 1987. Hence, although striped bass predation likely contributed to the decline in white perch PYSL to YOY survival and YOY abundance, from 1987 onward, predation could not have been the primary cause of declines that took place between 1975 and 1985.

3.4.3 American shad

Figure 17 depicts long-term trends in the abundance of American shad YOY and PYSL in the Hudson. The abundance of both life stages has declined significantly since the initiation of the generators' monitoring program, with declines in the abundance of both life stages beginning in the late 1980s. As shown in Figure 18, there is a strong positive correlation between PYSL abundance and YOY abundance in American shad (Figure 18a), and no relationship between PYSL survival and YOY abundance (Figure 18b). Because YOY abundance is correlated with PYSL abundance but not with PYSL survival, we can conclude that the decline in YOY abundance is a consequence of reduced reproduction rather than reduced PYSL survival.

Four hypothetical causes for these changes are evaluated below: the Indian Point CWIS, overfishing, zebra mussels, and striped bass predation.

3.4.3.1 CWIS

Co-Occurrence

There is no correlation between PYSL survival and the entrainment CMR at IP2 and IP3 (Figure 19a). The IP2 and IP3 CMR is also uncorrelated with American shad PYSL abundance (Figure 19b). Hence, the CWIS hypothesis fails the co-occurrence criterion.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

American shad PYSL abundance grew from the mid-1970s, when IP2 and IP3 began commercial operations, until 1986 (Figure 17). The highest values for both PYSL and YOY abundance occurred in 1986, 10 years after the startup of commercial operations at IP3 and 12 years after the startup of IP2 (Figure 17). Hence, the CWIS hypothesis fails the temporality criterion.

Manipulation

As discussed in Section 3.4.1.1, outages of IP2 or IP3 have frequently occurred during the entrainment season at Indian Point. Although American shad eggs and larvae occur only at very low densities in the vicinity of Indian Point (DEIS, Figure V-68), the peak abundance of American shad eggs and larvae typically occurs during May and June (DEIS, Figure V-67). IP2 was offline during the entire months of May and June in 1976, 1989, 1991, 1997, 1998, and 2000. IP3 was offline during the entire months of May and June in 1975, 1982, 1993, and 1994. If entrainment at Indian Point were reducing the survival of American shad PYSL, then PYSL survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 20a, the measured PYSL survival values are inconsistent with this expectation. Figure 20a shows the time series of annual PYSL survival indices from 1985 through 2002. The horizontal line in Figure 20a shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If American shad PYSL survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 20a shows that the PYSL survival index was higher than the median for 5 of the 8 years of one-unit operation. The PYSL index was lower than the median in 3 years of one-unit operation. This difference could easily have arisen by chance. Moreover, 3 of the 5 years with the highest survival rates (1996, 1999, and 2002) were years of 2-unit operation.

This result is confirmed by Figure 20b, which shows the relationship between the American shad PYSL survival index and the May-June total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and American shad PYSL survival. Hence, the CWIS hypothesis fails the manipulation criterion for American shad.

Coherence

The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.3.2 Fishing

Co-Occurrence

If a population is being overfished to the point at which spawner abundance is reduced, then the number of eggs and larvae produced by those spawners should decline. Historically, American shad supported very large unregulated commercial fisheries along the east coast of both the United States and Canada (ASMFC 1999). These harvests have declined dramatically in recent years. In its most recent stock assessment for American shad (ASMFC 2007), the ASMFC found that the abundance of adult American shad in the Hudson River peaked in 1985 and 1986 and has since declined. This decline in adult abundance occurred during the same period in which the abundance of American shad PYSL and YOY in the Hudson River declined (Figure 17). Hence, the fishing hypothesis satisfies the co-occurrence criterion.

Sufficiency

There is conflicting information concerning whether the magnitude of fishing mortality imposed on Hudson River American shad has been sufficient to cause the declines in spawner abundance. According to the ASMFC (2007), many American shad stocks have declined in abundance in recent decades. Although the declines appear to be related to an increase in the mortality of adult shad, the contribution of fishing to the increase in mortality is unclear and probably differs between spawning populations. According to Hattala and Kahnle (2007), the Hudson River population of American shad is probably being overfished, however, other sources of mortality cannot be excluded as contributing causes. Although there is still substantial uncertainty concerning causes of decline in American shad population, this assessment accepts Hattala and Kahnle's (2007) results and concludes that the overfishing hypothesis satisfies the sufficiency criterion.

Temporality

The decline in American shad spawner abundance coincided with the decline in abundance of PYSL and YOY (Figure 17). Hence, the overfishing hypothesis satisfies the temporality criterion.

Manipulation

Amendment 1 to the Interstate Fisheries Management Plan for Shad and River Herring (ASMFC 1999) directed all states to phase out the coastal fishery for American shad over a five year period beginning in 2000. The phase-out should reduce fishing mortality on American shad. If the coastal fishery had been contributing to decreased abundance of Connecticut River American shad, then the abundance of this population should increase as a result of this action. Data on fishing mortality and population abundance from the post-closure period are not yet available, so it is not yet possible to evaluate whether the overfishing hypothesis satisfies the manipulation criterion.

Coherence

As noted above, there is still substantial uncertainty concerning the impact of fishing on the Hudson River American shad population. However, available data are consistent with a conclusion that fishing is at least a significant contributor to the recent decline in abundance of Hudson River American shad (Hattala and Kahnle 2007). Hence, the overfishing hypothesis satisfies the coherence criterion.

3.4.3.3 Zebra mussels

Co-occurrence

As shown in Appendix B, Table B-15, the American shad PYSL survival index is positively correlated with the zebra mussel index, rather than negatively correlated as predicted by the zebra mussel hypothesis. As can easily be seen from Figure 17, American shad PYSL to YOY survival has increased since the zebra mussel invasion. Hence, the zebra mussel hypothesis fails the co-occurrence criterion for American shad.

Sufficiency

The potential effects of zebra mussel activity on early life stages of fish are indirect, and related to reductions in prey abundance and changes in habitat quality. No experiments have been performed that could quantify the relationship between zebra mussel activity and fish growth or survival, and no mathematical models that could be used to quantify the indirect effects of zebra mussel activity have been developed. Hence, whether or not the zebra mussel hypothesis satisfies the sufficiency criterion is unknown.

Temporality

The decline in abundance of American shad PYSL and YOY began in the late 1980s (Figure 17), several years prior to the zebra mussel invasion. Hence, the zebra mussel hypothesis fails the temporality criterion.

Manipulation

No deliberate manipulations of zebra mussel populations in the Hudson River have been performed, therefore, this criterion is inapplicable to the zebra mussel hypothesis.

Coherence

Because the proposed mechanism through which zebra mussel activity could have affected American shad in the Hudson River involves reducing food availability, the growth as well as the survival of American shad PYSL and YOY should have been reduced. Although Strayer et al. (2004) found a decline in growth rate of American shad PYSL and YOY following the zebra mussel invasion, this relationship was not significant even at the 20% level (Strayer et al. 2004, Fig. 7). No relationship between American shad YOY growth and zebra mussel activity was found in the analysis performed to support this assessment (Appendix B, Table B-15). Zebra mussel activity should also have shifted the distribution of American shad PYSL and YOY downriver, away from the freshwater zone in which zebra mussels are abundant. Strayer et al. (2004) found a net downriver shift in the distribution of American shad YOY, but a net upriver shift in the distribution of PYSL. In the analysis performed to support this assessment (Appendix B, Table B-15), no significant shifts in the distribution of either life stage was found. The observed changes in growth and distribution predicted by the zebra mussel hypothesis were not observed. Hence, the zebra mussel hypothesis fails the coherence criterion for American shad.

3.4.3.4 Striped bass predation

Co-occurrence

American shad PYSL abundance, which reflects spawner abundance and reproduction, is negatively correlated with the striped bass index (Figure 21a), although this relationship is significant only at the 10% level. This correlation provides weak support for the hypothesis that striped bass are preying on adult American shad. There is a negative relationship between the striped bass index and the American shad YOY index, (Figure 21b), however, this relationship is not statistically significant. Hence, the striped bass predation hypothesis appears to marginally satisfy the co-occurrence criterion for predation .

Sufficiency

Striped bass larger than 200 mm in length have been shown to feed on alosids such as American shad (Gardinier and Hoff 1982, Dunning et al. 1997). However, the prey consumption analysis documented in Appendix C to this report did not address predation on YOY American shad. Hence, with respect to YOY American shad, whether or not striped bass predation satisfies the sufficiency criterion is unknown. Kahnle and Hattala (2007) have argued that the great majority of adult striped bass in the Hudson are feeding on river herring rather than shad, and the striped bass predation is insufficient to significantly affect the abundance of adult Hudson River American shad. This assessment accepts the conclusions of Kahnle and Hattala (2007) that striped bass predation on adult Hudson River American shad is probably low.

Temporality

As can be seen from Figure 22, the increase in striped bass spawner abundance that began in the late 1980s closely coincides with the decline in American shad PYSL abundance. As shown in Figure 17, American shad YOY abundance has declined over this same period. Hence, the striped bass predation hypothesis satisfies the temporality criterion with respect to predation on both adults and YOY.

Manipulation

No deliberate manipulations of striped bass predation in the Hudson River have been performed, therefore, this criterion is inapplicable to the striped bass hypothesis.

Coherence

If predation by striped bass had caused the decline in abundance of American shad PYSL and YOY in the Hudson River, then the PYSL and YOY abundance of other known striped bass prey species, including white perch, river herring, bay anchovy, and Atlantic tomcod should also have declined. As discussed in other Sections of this report, no declines in white perch or bay anchovy PYSL abundance have occurred. However, PYSL abundance for river herring and Atlantic tomcod declined over the same period in which PYSL abundance for American shad declined. YOY abundance for all of the above species has declined since the late 1980s, when striped bass abundance began to increase. Moreover, other published studies have concluded that striped bass predation is reducing the abundance of some prey species. Savoy and Crecco (2004) attributed recent declines in the abundance of both blueback herring and American shad in the Connecticut River to striped bass predation on spawning adults, however, Kahnle and Hattala (2007) concluded that predation of striped bass on adult American shad in the Hudson River is relatively low. On the other hand, Hattala and Kahnle (2007) acknowledged that predation by striped bass on young American shad could be substantial and could be contributing to a decline in recruitment of young shad to the adult population.

Hartman (2003) estimated that the coastwide annual prey consumption by striped bass between 1 and 10 years of age increased by more than a factor of 8 between 1982 and 1995, from 17,900 mt to 147,900 mt. Uphoff (2003) calculated even larger estimates of striped bass consumption, and attributed a 90% decline in the abundance of Atlantic menhaden in upper Chesapeake Bay from 1980 through 1999 to predation by striped bass.

Because parallel declines in YOY abundance of other susceptible species have occurred, and because the other published studies have documented the influence of striped bass predation on susceptible prey species, the striped bass predation hypothesis satisfies the coherence criterion with respect to predation on YOY American shad, but not with respect to predation on adults.

3.4.3.5 Summary evaluation of hypotheses

Table 5 summarizes the consistency of the American shad data with the CWIS, overfishing, zebra mussel, and striped bass predation hypotheses. Two of the five evaluation criteria – sufficiency and coherence – are inapplicable to the CWIS hypothesis. The CWIS hypothesis fails the co-occurrence, temporality, and manipulation criteria. Hence, the CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 American shad in the Hudson River.

The overfishing hypothesis satisfies the co-occurrence, sufficiency, temporality, and coherence criteria for American shad. The manipulation criterion is inapplicable at present, although applicable data may become available once the response of the population to the phase-out of the ocean intercept fishery has been observed.

The zebra mussel hypothesis fails the co-occurrence, temporality, and coherence criteria for American shad. Whether the sufficiency criterion is satisfied is unknown, and the manipulation criterion is inapplicable. Hence, the zebra mussel hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 American shad in the Hudson River.

The striped bass predation hypothesis satisfies two and possibly three of the five criteria. Because no estimates of potential striped bass predation on YOY American shad have been developed, whether this hypothesis satisfies the sufficiency criterion is unknown. The manipulation criterion, is inapplicable to this hypothesis. The simultaneous declines in abundance of susceptible life stages of other prey species in the Hudson River and the published studies documenting impacts of striped bass predation on prey species support for the predation hypothesis. However, substantial uncertainty remains concerning the fraction of the American shad YOY population that might be consumed.

It appears reasonable to conclude that the recent decline in abundance of Hudson River American shad is most likely a result of overfishing, but striped bass predation may be a contributing cause.

3.4.4. Atlantic tomcod

Figure 23 depicts long-term trends in the abundance of Atlantic tomcod as measured by the LRS and the Atlantic Tomcod mark-recapture program. The LRS index reflects the abundance of late PYSL and early juvenile fish. The mark-recapture index reflects the combined abundance of age 1 and older (predominantly age 2) fish. The abundance of Atlantic tomcod has declined since the initiation of the generators' monitoring programs, with the abundance of age 1 and older fish abundance showing an abrupt decline beginning in 1990. The trend in abundance in the LRS time series is less clear, but the LRS index also has declined since 1990. Using Atlantic tomcod survival rates derived from annual mark-recapture surveys, for each year, the total egg to age 1 survival rate is estimated by comparing the total egg production during that year to the number of age 1 fish estimated to be present in the Hudson River during the following year. As shown in Figure 24, there is no relationship between egg deposition and resulting age 1 abundance (Figure 24a). There is a positive relationship between egg to age 1 survival and age 1 abundance (Figure 24b). Hence, the decline in Atlantic tomcod abundance is related to a decrease in survival rather than a decrease in egg production.

Atlantic tomcod are uncommon in freshwater reaches of the Hudson River, therefore, they should not be susceptible to the effects of zebra mussel activity. This potential stressor is not evaluated as a cause of changes in the abundance of this species. Three hypothetical causes for these changes are evaluated below: the Indian Point CWIS, elevated summer temperatures, and striped bass predation.

3.4.4.1 CWIS

Co-occurrence

As shown in Figure 25a, there is no correlation between the IP2 and IP3 CMR and egg-to-age 1 survival. There is a negative correlation between the IP2 and IP3 CMR and the Atlantic tomcod LRS index (Figure 25b), but this correlation is significant only at the 10% level (Appendix B, Table B-17). There is no correlation between the IP2 and IP3 CMR and the mark-recapture index (Figure 25c). Because the IP2 and IP3 CMR are negatively correlated with only one of the three response metrics, and only at the 10% level, the CWIS hypothesis only weakly satisfies the co-occurrence criterion.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence, the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

As shown in Figure 23, the decline in abundance of Atlantic tomcod in the mark-recapture survey did not begin until the mid-1980s and the decline in the LRS survey did not begin until 1990. Hence, the CWIS hypothesis fails the temporality criterion.

Manipulation

Although American tomcod spawn in December and January, entrainable larvae and juveniles are still abundant in the lower estuary during May and June (DEIS, Figure 5-56). IP2 was offline during the entire months of May and June in 1976, 1989, 1991, 1997, 1998, and 2000. IP3 was offline during the entire months of May and June in 1975, 1982, 1993, and 1994. If entrainment at Indian Point were reducing the survival of Age 0 Atlantic tomcod, then egg to age 1 survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 26a, the measured PYSL survival values are inconsistent with this expectation. Figure 26a shows the time series of egg to age 1 indices from 1976 through 2001. The horizontal line in Figure 26a shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If Atlantic tomcod survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 26a shows that the PYSL survival index was higher than the median for 3 of the 7 years of one-unit operation. The PYSL index was lower than the median in 4 years of one-unit operation.

This result is confirmed by Figure 26b, which shows the relationship between the Atlantic tomcod egg to age 1 survival index and the May-June total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and Atlantic tomcod egg to age 1 survival. Hence, the CWIS hypothesis fails the manipulation criterion for Atlantic tomcod.

Coherence

The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.4.2 Elevated summer temperatures

Co-occurrence

As shown in Appendix B, Table B-17, egg to age 1 survival is negatively correlated with the PWW degree-day index. Egg to age 1 survival is not, however, correlated with the August cooling water flows at IP2 and IP3, which is an index of the thermal loading to the River from IP2 and IP3. Hence, the temperature hypothesis satisfies the co-occurrence criterion, although there is no evidence that IP2 and IP3 contribute to a temperature effect.

Sufficiency

As discussed by McLaren et al. (1988), summer temperatures in the Hudson River frequently exceed optimal levels for juvenile Atlantic tomcod, and occasionally can exceed the lethal tolerance temperature (26.5°C) for this species (McLaren et al. 1988). Although the temperature of the Hudson River is highly variable between locations, depth strata, and years, it can be concluded that the temperature hypothesis satisfies the sufficiency criterion.

Temporality

Figure 27 compares long-term trends in PWW degree-day index to long-term trends in the abundance of age 1 and age 2 Atlantic tomcod, for the period 1987-2001. For each year, the degree-day index is paired with the mark-recapture estimates generated during the following winter (e.g., the 1987 temperature value is paired with the mark-recapture value for the winter of 1987-1988). As shown in Figure 27, a decline in Atlantic tomcod occurred from 1990-2001. However, elevated temperatures that could have explained this decline did not occur. There is no long-term trend in the PWW degree-day index, and three of the four lowest values of the index have occurred since 1990. Hence, the temperature hypothesis fails the temporality criterion.

Manipulation

No deliberate manipulations of Hudson River water temperatures have been performed, therefore, this criterion is inapplicable to temperature hypothesis.

Coherence

If elevated temperatures were adversely affecting Atlantic tomcod in the Hudson River, then other temperature-sensitive species should also be declining. As noted in the FEIS (pp 66-67), the abundance of rainbow smelt in the Hudson River has also been declining. In addition, the temperature hypothesis is consistent with laboratory data on thermal tolerances in Atlantic tomcod and with the geographic distribution of this species. As noted by McLaren et al. (1988), the Hudson River is the southern-most reproducing Atlantic tomcod population. Hence, the temperature hypothesis satisfies the coherence criterion.

3.4.4.3 Striped bass predation

Co-occurrence

Both the Atlantic tomcod mark-recapture index and the LRS index are negatively correlated with the striped bass index (Figure 28). Hence, the striped bass predation hypothesis satisfies the co-occurrence criterion.

Sufficiency

Striped bass larger than 200 mm in length have been shown to feed on Atlantic tomcod (Gardinier and Hoff 1982, Dunning et al. 1997). Appendix C to this report documents an analysis of prey consumption by Hudson River striped bass. This analysis compares the change in striped bass prey consumption requirements (August through October) between earlier (1983-1990) and more recent (1991-2004) periods to changes in abundance of YOY fish in the Hudson River between these same two periods. The analysis shows that the increase in prey consumption from the earlier to the later period would be sufficient to explain the decline in YOY Atlantic tomcod abundance between these two periods if 1.4% of the age 1 and age 2 striped bass seasonal predatory demand was satisfied by YOY Atlantic tomcod, or if 0.4% of the age 1 through age 13 striped bass seasonal predatory demand was satisfied by YOY Atlantic tomcod. Hence, the striped bass predation hypothesis satisfies the sufficiency criterion.

Temporality

The increase in striped bass abundance coincides in time with the declines in both Atlantic tomcod abundance metrics (Figure 29). Hence, the striped bass predation hypothesis satisfies the temporality criterion.

Manipulation

No deliberate manipulations of striped bass predation in the Hudson River have been performed, therefore, this criterion is inapplicable to the striped bass hypothesis.

Coherence

If predation by striped bass had caused the decline in abundance of Atlantic tomcod in the Hudson River, then the YOY abundance of other known striped bass prey species, including white perch, river herring, American shad, and bay anchovy, should also have declined. As shown in other Sections of this report, YOY abundance for all of these species has declined since the late 1980s, when striped bass abundance began to increase. Moreover, other published studies have concluded that striped bass predation is reducing the abundance of some prey species. Savoy and Crecco (2004) attributed recent declines in the abundance of both blueback herring and American shad in the Connecticut River to striped bass predation. Hartman (2003) estimated that the coastwide annual prey consumption by striped bass between 1 and 10 years of age increased by more than a factor of 8 between 1982 and 1995, from 17,900 mt to 147,900 mt. Uphoff (2003) calculated even larger estimates of striped bass consumption, and attributed a 90% decline in the abundance of Atlantic menhaden in upper Chesapeake Bay from 1980 through 1999 to predation by striped bass.

Because parallel declines in other susceptible species have occurred, and because the other published studies have documented the influence of striped bass predation on susceptible prey species, the striped bass predation hypothesis satisfies the coherence criterion.

3.4.4.4 Summary evaluation of hypotheses

Table 6 summarizes the consistency of the Atlantic tomcod data with the CWIS, temperature, and striped bass predation hypotheses. Two of the five evaluation criteria –

sufficiency and coherence – are inapplicable to the CWIS hypothesis. The CWIS hypothesis weakly satisfies the co-occurrence criterion, but fails the temporality, and manipulation criteria. The CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 Atlantic tomcod in the Hudson River.

The temperature hypothesis satisfies the co-occurrence, sufficiency, and coherence criteria, but fails the temporality criterion. The manipulation criterion is inapplicable to this hypothesis. Hence, the temperature hypothesis cannot be rejected. However, failure to satisfy the temporality criterion indicates that factors other than temperature were responsible for the decline in abundance of Atlantic tomcod that occurred after 1990.

The striped bass predation hypothesis satisfies all of the applicable criteria. The correlations between striped bass abundance and Atlantic tomcod abundance, the temporal correspondence between the timing of the striped bass increase and the Atlantic tomcod decline, the estimates of striped bass prey consumption, the simultaneous declines in abundance of susceptible life stages of other prey species in the Hudson River, and the published studies documenting impacts of striped bass predation on prey species all provide relatively strong support for the predation hypothesis.

3.4.5 Alewife and blueback herring

Figure 30 depicts long-term trends in the abundance of alewife and blueback herring PYSL and YOY in the Hudson. These two species must be considered together for purposes of evaluating impacts of CWIS, because their larvae are indistinguishable. PYSL abundance for both species combined (Figure 30a) was stable until 1985, and has since declined. With respect to YOY abundance, these two species have tended to vary together (Figure 30b). YOY abundance in both species declined abruptly in the mid-1980s and has fluctuated without apparent trend since that time, but without returning to previous abundance levels.

3.4.5.1 CWIS

Co-occurrence

IP2 and IP3 entrainment CMR is uncorrelated with river herring PYSL survival (Figure 31a), river herring PYSL abundance (Figure 31b), alewife YOY abundance (Figure 32a), and blueback herring YOY abundance (Figure 32b). Hence, the CWIS hypothesis fails the co-occurrence criterion.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

As shown in Figures 30a and 30b, alewife and blueback herring PYSL and YOY abundance did not decline until the mid-1980s, nearly a decade after the startup of commercial operations at IP2 and IP3. Hence, the CWIS hypothesis fails the temporality criterion.

Manipulation

The peak abundance of river herring eggs and larvae typically occurs during May and June (DEIS, Figures V-71 and V-74). IP2 was offline during the entire months of May and June in 1976, 1989, 1991, 1997, 1998, and 2000. IP3 was offline during the entire months of May and June in 1975, 1982, 1993, and 1994. If entrainment at Indian Point were reducing the survival of river herring PYSL, then PYSL survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 33a, the measured PYSL survival values are inconsistent with this expectation. Figure 33a shows the time series of annual PYSL survival indices from 1974 through 2002. The horizontal line in Figure 33a shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If river herring PYSL survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 33a shows that the PYSL survival index was higher than the median for 4 of the 11 years of one-unit operation. The PYSL was index lower than the median in 7 years of one-unit operation.

This result is confirmed by Figure 33b, which shows the relationship between the river herring PYSL survival index and the May-June total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and river herring PYSL survival. Hence, the CWIS hypothesis fails the manipulation criterion for alewife and blueback herring.

Coherence

The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.5.2 Zebra mussels

Co-occurrence

As shown in Appendix B, Tables B-19 and B-21, there is no correlation between the zebra mussel index and any abundance index for either alewife or blueback herring. Hence, the zebra mussel hypothesis fails the co-occurrence criterion for both species.

Sufficiency

The potential effects of zebra mussel activity on early life stages of fish are indirect, and related to reductions in prey abundance and changes in habitat quality. No experiments have been performed that could quantify the relationship between zebra mussel activity and fish growth or survival, and no mathematical models that could be used to quantify the indirect effects of zebra mussel activity have been developed. Hence, whether or not the zebra mussel hypothesis satisfies the sufficiency criterion is unknown.

Temporality

The decline in abundance of alewife and blueback herring PYSL and YOY occurred during the mid-1980s, more than 5 years prior to the invasion of the river by zebra mussels (Figure 30). Hence, the zebra mussel hypothesis fails the temporality criterion.

Manipulation

No deliberate manipulations of zebra mussel populations in the Hudson River have been performed, therefore, this criterion is inapplicable to the zebra mussel hypothesis.

Coherence

Because the proposed mechanism through which zebra mussel activity could have affected river herring in the Hudson River involves reducing food availability, the growth as well as the survival of river herring PYSL and YOY should have been reduced. Strayer et al. (2004) found a decline in the growth rate of YOY alewife following the zebra mussel invasion using both the utility beach seine index and the NYSDEC beach seine index. Only the decline in the growth rate calculated from the NYSDEC index was statistically significant, and only at the 20% level. No relationship between alewife or blueback herring growth and zebra mussel activity was found in the analysis performed to support this assessment (Appendix B, Tables B-19 and B-21). Zebra mussel activity should also have shifted the distribution of river herring PYSL and YOY downriver, away from the freshwater zone in which zebra mussels are abundant. Strayer et al. (2004) found net downriver shifts in the distribution of alewife and blueback herring YOY, but a net upriver shift in the distribution of PYSL. None of these shifts was statistically significant, even at the 20% level. In the analysis performed to support this assessment (Appendix B, Tables B-19 and B-21), no significant shift in the distribution of blueback herring was found, but an upstream shift in the distribution of alewife YOY was found. Only one of the predicted effects of the zebra mussel invasion on river herring was observed, in only one out of three analyses, and at a significance level (20%) not usually accepted in scientific studies. Hence, the zebra mussel hypothesis fails the coherence criterion for alewife and blueback herring.

3.4.5.3 Striped bass predation

Co-occurrence

The river herring PYSL abundance index, which reflects spawner abundance and reproduction, is negatively correlated with the striped bass index (Figure 34a). The alewife YOY index, and the blueback herring YOY index are also negatively correlated with the striped bass index, although only at the 10% significance level (Appendix B, Tables B-19 and B-21). (Figures 34b and 34c). Hence, the striped bass predation hypothesis satisfies the co-occurrence criterion for predation, on both adults and YOY.

Sufficiency

Striped bass larger than 200 mm in length have been shown to feed on alosids, including alewife and blueback herring (Gardinier and Hoff 1982, Dunning et al. 1997). According to Savoy and Crecco (2004) and Davis et al. (2007), adult striped bass in the Connecticut River feed heavily on spawning blueback herring. Recently, Kahnle and Hattala (2007) reported that river herring were the most common prey item in the stomachs of adult striped bass captured in the Hudson River. Appendix C to this report documents an analysis of prey consumption by Hudson River striped bass. This analysis compares the change in striped bass prey consumption requirements (August through October) between earlier (1983-1990) and more recent (1991-2004) periods to changes in abundance of YOY fish in the Hudson River between these same two periods. The analysis shows that the increase in prey consumption from the earlier to the later period would be sufficient to explain the decline in YOY river herring abundance between these two periods if 3% of the age 1 and age 2 striped bass seasonal predatory demand was satisfied by YOY river herring, or if 0.9% of the age 1 through age 13 striped bass seasonal predatory demand was satisfied by YOY river herring. Hence, the striped bass predation hypothesis satisfies the sufficiency criterion with respect to predation on YOY river herring. No quantitative estimates of consumption of adult river herring by striped bass are available.

Temporality

The decline in river herring abundance coincides in time with the increase in the striped bass index (Figure 35). Hence, the trends analysis supports the hypothesis that predation by striped bass has contributed to the decline in alewife and blueback herring abundance. Alewife and blueback herring do not return to the Hudson as spawning adults until an age of at least four years (ASMFC 1998). Hence, if only juvenile river herring were susceptible to predation by striped bass, a four-year time lag would be expected between the increase in striped bass abundance and the decline in PYSL abundance. The fact that no such time lag is apparent over the substantial time series available (Figure 35a), is consistent with the hypothesis that spawning adults are also susceptible to predation. Hence, the predation hypothesis satisfies the temporality criterion for both predation on adults and predation on YOY.

Manipulation

No deliberate manipulations of striped bass predation in the Hudson River have been performed, therefore, this criterion is inapplicable to the striped bass hypothesis.

Coherence

If predation by striped bass had caused the decline in abundance of river herring in the Hudson River, then the YOY abundance of other known striped bass prey species, including white perch, American shad, Atlantic tomcod, and bay anchovy, should also have declined. As shown in other Sections of this report, YOY abundance for all of these species has declined since the late 1980s, when striped bass abundance began to increase. Moreover, other published studies have concluded that striped bass predation is reducing the abundance of some prey species. Savoy and Crecco (2004) attributed recent declines in the abundance of both blueback herring and American shad in the Connecticut River to striped bass predation. This conclusion is supported by a recent study of the diet composition of striped bass present in the Connecticut River during the spring shad and river herring spawning run (Davis et al. 2007). These authors found that striped bass between 600 and 800 mm in length feed predominantly on adult river herring. These results are consistent with the results published by Kahnle and Hattala (2007), who found that river herring were the most abundant of the identifiable prey items in the stomachs of adult striped bass captured in the Hudson River. Hartman (2003) estimated that the coastwide annual prey consumption by striped bass between 1 and 10 years of age increased by more than a factor of 8 between 1982 and 1995, from 17,900 metric tons (mt) to 147,900 mt. Uphoff (2003) calculated even larger estimates of striped bass consumption, and attributed a 90% decline in the abundance of Atlantic menhaden in upper Chesapeake Bay from 1980 through 1999 to predation by striped bass.

Because parallel declines in other susceptible species have occurred, because predation by striped bass on adult river herring has been demonstrated, and because the other published studies have documented the influence of striped bass predation on susceptible prey species, the striped bass predation hypothesis satisfies the coherence criterion.

3.4.5.4 Summary evaluation of hypotheses

Table 7 summarizes the consistency of the alewife and blueback herring data with the CWIS, temperature, and striped bass predation hypotheses. Two of the five evaluation criteria – sufficiency and coherence – are inapplicable to the CWIS hypothesis. The CWIS hypothesis fails the co-occurrence, temporality, and manipulation criteria. Hence, the CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 river herring in the Hudson River.

The zebra mussel hypothesis fails the co-occurrence, temporality, and coherence criteria for river herring. Whether the sufficiency criterion is satisfied is unknown, and the manipulation criterion is inapplicable. Hence, the zebra mussel hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 river herring in the Hudson River.

The striped bass predation hypothesis satisfies all of the applicable criteria. The correlations between striped bass abundance and river herring abundance, the temporal correspondence between the timing of the striped bass increase and the river herring decline, the estimates of striped bass prey consumption, the simultaneous declines in abundance of susceptible life stages of other prey species in the Hudson River, and the published studies documenting predation by striped bass on spawning adult river herring, and studies documenting impacts of striped bass predation on prey species all provide relatively strong support for the predation hypothesis.

3.4.6. Bay anchovy

Bay anchovy is a marine species and, because zebra mussels occur only in the freshwater zone of the Hudson River, bay anchovy should not be susceptible to the effects of zebra mussel activity. This potential stressor is not evaluated as a cause of changes in the abundance of this species. Two hypothetical causes for these changes are evaluated below: the Indian Point CWIS and striped bass predation.

Figure 36 depicts long-term trends in the abundance of bay anchovy YOY and PYSL in the Hudson. The abundance of juvenile bay anchovy, as measured by the FSS, has declined since 1985. There has been no trend in abundance of PYSL.

3.4.6.1 CWIS

Co-occurrence

As shown in Figure 37, the PYSL to YOY survival rate (Figure 37a) and the PYSL index (Figure 37b) are both uncorrelated with the IP2 and IP3 CMR. Hence, the CWIS hypothesis fails the co-occurrence criterion.

Sufficiency

There are no independent measures of sufficiency that can be applied to this hypothesis. The objective of this report is to determine, using all available and relevant evidence whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Hence, the sufficiency criterion is inapplicable to the CWIS hypothesis.

Temporality

There has been no decline in bay anchovy PYSL abundance, and bay anchovy YOY abundance did not decline until the late 1980s, more than 10 years following the startup of IP2 and IP3. Hence, the CWIS hypothesis fails the temporality criterion.

Manipulation

The peak abundance of bay anchovy eggs and larvae typically occurs during June and July (DEIS, Figures V-78). IP2 was offline during the entire months of June and July in 1976, 1998, and 2000. IP3 was offline during the entire months of June and July in 1975, 1982, 1987, 1993, 1994, and 1997. If entrainment at Indian Point were reducing the survival of bay anchovy PYSL, then PYSL survival should have been higher in years when one unit was offline than in years when both units were operating. As shown in Figure 38a, the measured PYSL survival values are inconsistent with this expectation. Figure 38a shows the time series of annual PYSL survival indices from 1985 through 2002. The horizontal line in Figure 38a shows the median survival index value for this time period. The median is defined as the midpoint of the entire distribution of survival index values, meaning that one-half of the survival indices are above the median and one-half are below the median. If bay anchovy PYSL survival were higher in years of one-unit operation than in years of 2-unit operation, then significantly more survival index values for years of one-year operation should be higher than the median than lower than the median. However, Figure 38a shows that the PYSL survival index was higher than the median for 4 of the 7 years of one-unit operation and lower than the median for the other 3 years. This difference could easily have arisen by chance.

This result is confirmed by Figure 38b, which shows the relationship between the bay anchovy PYSL survival index and the June-July total water withdrawals by IP2 and IP3 for the years 1975-2002. There is no correlation between withdrawals by IP2 and IP3 and bay anchovy PYSL survival. Hence, the CWIS hypothesis fails the manipulation criterion for bay anchovy.

Coherence

The objective of this report is to determine, using all available and relevant evidence, whether the magnitude of entrainment and impingement at Indian Point have been sufficient to cause a reduction in the abundance of important Hudson River fish species. Including "coherence" as an explicit evaluation criterion for CWIS would be redundant. Hence, the coherence criterion is inapplicable to the CWIS hypothesis.

3.4.6.2 Striped bass predation

Co-occurrence

Bay anchovy juvenile abundance is negatively correlated with the striped bass index (Figure 39a). Hence, the striped bass hypothesis satisfies the co-occurrence criterion.

Sufficiency

Striped bass larger than 200 mm in length have been shown to feed on clupeids such as bay anchovy (Gardinier and Hoff 1982, Dunning et al. 1997). However, the prey consumption analysis documented in Appendix C to this report did not address predation on bay anchovy. Hence, whether the striped bass predation hypothesis satisfies the sufficiency criterion for bay anchovy is unknown.

Temporality

The increase in striped bass abundance coincides in time with the decline in bay anchovy juvenile abundance (Figure 39b). Hence, the striped bass hypothesis satisfies the temporality criterion for bay anchovy.

Manipulation

No deliberate manipulations of striped bass predation in the Hudson River have been performed, therefore, this criterion is inapplicable to the striped bass hypothesis.

Coherence

If predation by striped bass had caused the decline in abundance of bay anchovy YOY in the Hudson River, then the YOY abundance of other known striped bass prey species, including white perch, American shad, river herring, and Atlantic tomcod should also have declined. As discussed in other Sections of this report, YOY abundance for all of the above species has declined since the late 1980s, when striped bass abundance began to increase. Moreover, other published studies have concluded that striped bass predation is reducing the abundance of some prey species.

Hartman (2003) estimated that the coastwide annual prey consumption by striped bass between 1 and 10 years of age increased by more than a factor of 8 between 1982 and 1995, from 17,900 mt to 147,900 mt. Uphoff (2003) calculated even larger estimates of striped bass consumption, and attributed a 90% decline in the abundance of Atlantic menhaden in upper Chesapeake Bay from 1980 through 1999 to predation by striped bass.

Because parallel declines in other susceptible species have occurred, and because the other published studies have documented the influence of striped bass predation on susceptible prey species, the striped bass predation hypothesis satisfies the coherence criterion with respect to predation on YOY bay anchovy.

3.4.6.3 Summary evaluation of hypotheses

Table 8 summarizes the consistency of the bay anchovy data with the CWIS and striped bass predation hypotheses. Two of the five evaluation criteria – sufficiency and coherence – are inapplicable to the CWIS hypothesis. The CWIS hypothesis fails the co-occurrence, temporality, and manipulation criteria. Hence, the CWIS hypothesis can be rejected as an explanation for long-term trends in the abundance of age 0 bay anchovy in the Hudson River.

The striped bass hypothesis satisfies three of the five criteria. The manipulation criterion is inapplicable to this hypothesis, and whether this hypothesis satisfies the sufficiency criterion is unknown. The simultaneous declines in abundance of susceptible life stages of other prey species in the Hudson River and the published studies documenting impacts of striped bass predation on prey species all provide relatively strong support for the predation hypothesis. However, substantial uncertainty remains concerning the fraction of the bay anchovy YOY population that might be consumed.

3.4.7. Spottail shiner

Figure 40 depicts long-term trends in the abundance of spottail shiners and YOY in the Hudson River. The abundance of shiners has significantly declined, while the abundance of YOY has significantly increased. The increase in abundance of YOY spottail shiner is inconsistent with all of the hypotheses evaluated in this report. Hence, there is no need to perform a formal evaluation using the criteria from Suter et al. (2007).

As shown in Figure 41, there is no correlation between the IP2 and IP3 CMR and either spottail shiner response metric. This result is not unexpected because, as discussed in the DEIS (Figure V-107), spottail shiner is a freshwater species that is uncommon in the vicinity of Indian Point. The causes of recent changes in the abundance of this species cannot be identified using the data available for this report; however, the CWIS hypothesis can be rejected.

3.5 Summary evaluation of trends analysis

The results of the trends analysis are inconsistent with the hypothesis that entrainment at IP2 and IP3 is reducing the survival or abundance of any of the eight Hudson River fish species considered in this assessment. Overfishing is the most likely cause of the recent decline in abundance of American shad, with striped bass predation being a potentially important contributing factor. For other species, the striped bass predation hypothesis is the most strongly supported hypothesis. This hypothesis satisfies the co-occurrence, sufficiency, temporality, and coherence criteria for many of the species evaluated. With respect to the co-occurrence criterion, the striped bass index is negatively correlated with abundance indices for white perch, American shad, Atlantic tomcod, river herring, and bay anchovy. With respect to sufficiency, the analyses documented in Appendix C show that the increase in prey consumption by Hudson River striped bass in recent years is sufficient to account for observed declines in the YOY abundance of white perch, Atlantic tomcod, and river herring. With respect to temporality, the increase in striped bass abundance that occurred following the imposition of harvest restrictions in the mid-1980s coincides in time with the declines in abundance of one or more life stages of all of these species. With respect to coherence, striped bass predation has been implicated in declines of susceptible species in other mid-Atlantic northeastern estuaries (Hartman 2003, Uphoff 2003, Savoy and Crecco 2004) and striped bass have been shown to prey on all of the species listed above (Gardinier and Hoff 1982, Dunning et al. 1997, Savoy and Crecco 2004, Kahnle and Hattala 2007).

The available evidence is sufficient to reject Indian Point CWIS as having a measurable effect on any of the species evaluated. Within the limits of the data available for this assessment, it can reasonably be concluded that striped bass predation is a far more likely cause of declines in the abundance of YOY white perch, American shad, Atlantic tomcod, river herring, and bay anchovy than are any of the other potential causes evaluated.

4. Evaluation of impacts of cooling-water withdrawals on spawning potential

Fisheries scientists have developed a variety of quantitative methods for determining whether the sustainability of a fish population is being harmed by excessive harvesting. From the perspective of population dynamics, entrainment and impingement have been characterized (somewhat over simplistically) as a type of "fishing," imposed on early life stages rather than on adult fish (Goodyear 1977). For this reason, these methods may be used to determine whether entrainment or impingement by IP2 and IP3's respective CWIS could have adversely affected Hudson River fish populations that support managed fisheries. The method to be used, the SSBPR model, has a long history of application both in power-plant impact assessment studies and in fisheries management (Goodyear 1993).

4.1 History of the SSBPR model

One of the critical questions in fisheries management is how much spawning stock (essentially, the number of adult fish) must be protected from harvesting to allow a population to replace itself and persist through time (i.e., a sustainable population) (Mace and Sissenwine 1993). The so-called spawning stock biomass per recruit or SSBPR model is the most widely used approach for answering this important question for fish populations subjected to commercial and recreational fishing (Sissenwine and Shepherd 1987, Gabriel et al. 1989, Goodyear 1993, Mace and Sissenwine 1993, Rosenberg et al. 1994). Further, since it was originally developed by Goodyear (1977) as a method for assessing whether entrainment and impingement of striped bass at Hudson River power plants could, in combination with fishing mortality, threaten the ability of the population to sustain itself, its application to entrainment and impingement is well-supported.

The SSBPR model uses information on age-specific mortality and fecundity (i.e., the number of eggs produced by a female fish of a given age) to calculate the expected lifetime reproduction of a one-year-old female fish (a "recruit," in fisheries terminology). Expected lifetime reproduction is a function both of the average fecundity of female fish at each age and the probability that the female will survive to reproduce at that age (Goodyear 1977). Mortality due to fishing, CWIS, or other causes reduces expected lifetime reproduction either by reducing the probability of survival (in the case of fishing), reducing the probability that spawned eggs will survive to become one-year-old recruits (in the case of CWIS), or reducing the fecundity of female fish (e.g., through adverse environmental conditions, such as toxic chemicals). For the population to persist, each one-year-old female fish must produce at least one female egg that survives to become a one-year-old female recruit (Mace and Sissenwine 1993, Goodyear 1993). An average female has the potential to produce far more eggs than are required to replace her (Mace and Sissenwine 1993). For example, a female striped bass can spawn 3 million or more eggs in a single year (Hoff et al. 1988; Olsen and Rulifson 1992) and can live for up to 30 years (Secor and Piccoli 1996). For the population to maintain itself at a stable level, only one of the female eggs produced by each fish over her lifetime must survive to adulthood. This massive surplus of eggs ensures that the population will be able to persist in spite of natural and potentially extreme fluctuations in environmental conditions. This massive surplus of eggs also ensures that even substantial harvesting by commercial and recreational fishermen will not adversely affect the population.

4.2 Explanation of the SSBPR concept

The use of SSBPR in fisheries management derives from recognition that the lifetime reproductive capacity of a typical recruit provides a useful measure of the replacement capability of a population (Goodyear 1977, 1993, Sissenwine and Shepherd 1987, Mace and Sissenwine 1993, Rosenberg et al. 1994). At low levels of fishing mortality, the lifetime reproductive capacity of a typical female recruit is far larger than is necessary to sustain the population. As fishing mortality increases, the expected life span of each fish decreases, resulting in a reduction in lifetime reproductive capacity. If fishing mortality exceeds a critical threshold, the number of eggs produced by a female over her lifetime will fall below the replacement level. Once egg production falls below this level, recruitment (the number of fish entering the population each year) will begin to decline, and will continue to decline unless fishing is reduced to a level that once again allows lifetime egg production to meet or exceed the replacement level (Sissenwine and Shepherd 1987, Mace and Sissenwine 1993).

In a review of over-fishing definitions used in the management of marine fish stocks, Rosenberg et al. (1994) found that most of these definitions were based on the SSBPR model, and used the SSBPR model to evaluate over-fishing definitions used to manage the marine fish stocks. NOAA guidelines (Restrepo et al. 1998) for implementing National Standard 1 of the Magnuson-Stevens Act identify the SSBPR model as one of the methods that can be used to establishing over-fishing reference points that comply with the Act.

SSBPR is estimated as:

$$SSBPR = \sum_i l_i m_i f_i$$

where

l_i = probability of survival from age 1 to age *i*

m_i = fraction of the population of age *i* which are mature females; and

f_i = average fecundity of a female fish at age *i* (average number of eggs/female of age *i*).

The probability of survival to age *i* is estimated by combining age-specific rates of natural mortality, fishing mortality, and entrainment/impingement mortality:

$$l_i = \prod_{a=1}^{a=i-1} e^{-(M_a+F_a+P_a)}$$

where

M_a = age-specific instantaneous natural mortality rate at age a;

F_a = instantaneous fishing mortality rate at age a; and

P_a = instantaneous power-plant mortality rate at age a.

The impact of fishing and power-plant mortality on expected lifetime egg production is expressed as the ratio of SSBPR including both sources of mortality to SSBPR without these sources of mortality. This ratio is often termed the "spawning potential ratio" ("SPR"):

$$SPR = \frac{SSBPR_{fished}}{SSBPR_{unfished}}$$

Rates of fishing mortality that would produce a given SPR value are used by fisheries management agencies to establish acceptable limits on fishing mortality. Historically, the two reference points most commonly used by fisheries managers are $F_{.35}$ and $F_{.20}$. $F_{.35}$ is the fishing mortality rate that will lead to an SPR value of 0.35. $F_{.35}$ has often been used as a default goal for achieving maximum sustained yield ("MSY"), i.e., the maximum amount of adult fish (in pounds or kilograms) that can be removed from the population each year by fishermen without affecting the sustainability of the population. Values of F greater than $F_{.35}$ would lead to harvests greater than could be sustained over time. $F_{.20}$ is the fishing mortality rate that will lead to SPR value of 0.2, a default value indicating over-fishing. If F consistently exceeds $F_{.20}$, then significant declines in the adult population may occur. Although some fish stocks may be able to maintain recruitment at $F_{.20}$, other stocks are more sensitive to fishing and cannot sustain exploitation at this level (Mace and Sissenwine 1993, Rosenberg et al. 1994).

4.3 Application to Hudson River fish populations

Quantitative stock assessments and biological reference points are available for two of the species addressed in this report: striped bass (ASMFC 2005) and American shad (ASMFC 2007). As long as mortality caused by entrainment and impingement is limited to fish that are younger than one year old (which is true for both striped bass and American shad), the CMR calculated using the generators' empirical entrainment and impingement models provides a direct measure of the reduction in SSBPR caused by IP2 and IP3 (Goodyear 1977). The likelihood that entrainment and impingement at IP2 and IP3 have adversely affected the sustainability of these two species is evaluated in two ways. First, estimates of reduction in SSBPR due IP2 and IP3 are compared to reductions caused by fishing mortality. Second, estimates of combined reductions in SSBPR due to both IP2 and IP3 and fishing are compared to the biological reference points that are currently used to manage these species.

4.3.1 Striped bass

As shown in Figure 42, the striped bass CMR for the 30 years for which data are available corresponds to an SPR of 0.92. In other words, IP2 and IP3 reduce the spawning potential of the Hudson River striped bass population to 92% of the value for an unfished population. Fishing for striped bass at the current target rate established by the ASMFC (F=0.30)[10] corresponds to an SPR of 0.13. This means that fishing for striped bass, under the current management approach, has reduced the reproductive potential of a typical 1-year-old female striped bass to only 13% of the value that would be expected in an unfished striped bass population. The threshold fishing rate for striped bass is currently set at F=0.41 (ASMFC 2003). This value corresponds to an SPR of 0.096. If the rate of fishing were to rise above F=0.41, the ASMFC would be required to declare the population to be over-fished and would take action to reduce harvesting.

As shown in Figure 42, even when effects of fishing are combined with effects of IP2 and IP3, the combined SPR is still above the threshold. Hence, either alone or in combination with fishing, entrainment and impingement at IP2 and IP3 have not jeopardized the sustainability of the Hudson River striped bass population as defined by ASMFC regulations. Further, as is clear from Figure 42, the impacts of fishing on the sustainability of the Hudson River striped bass population dwarf any impact of IP2 and IP3. Eliminating entrainment and impingement of striped bass at IP2 and IP3 would not have a measurable influence on the sustainability of the population.

4.3.2 American shad

The ASMFC (ASMFC 2007a, 2007b) recently used the SSBPR model to assess impacts of increased mortality on the sustainability of Atlantic coastal American shad populations, including the Hudson River American shad population. Because the relative contributions of fishing mortality and natural mortality to the increase are uncertain, the ASMFC expressed the maximum sustainable rate of mortality in terms of total mortality (*Z*) rather than fishing mortality. The ASMFC selected Z_{30}, the total mortality rate at which SSBPR would fall to 30%

[10] For assessment purposes, Atlantic striped bass are treated as a single mixed population, and the same fishing mortality rate is assumed to be applicable to all of the individual spawning populations that contribute to the mixed coastal fishery.

of an assumed baseline value, as an excess mortality threshold analogous to $F_{.30}$. Using alternative assumptions concerning the operation of the American shad fishery, the ASMFC developed a range of estimates of $Z_{.30}$ of Z=0.54 to Z=0.73 for the Hudson River American shad population.

Empirical estimates of total annual mortality in Hudson River American shad are available for the years 1984-2004 (ASMFC 2007a). Total mortality has exceeded $Z_{.30}$ in most years during this period. Hattala and Kahnle (2007) have contended that the excessive mortality imposed on Hudson River American shad is due primarily to overfishing. However, regardless of the actual cause, it is clear that entrainment at Indian Point is a negligible contributor to American shad mortality. Figure 43 compares reductions in spawning potential of American shad due to IP2 and IP3 to reductions due to other causes, including fishing. The calculations were performed using the Hudson-specific life history parameters from Tables 1.1.5.1-b (age-invariant natural mortality) and 1.1.5.2-b of ASMFC (2007a) and the revised Type 1 fishery model from ASMFC (2007b).

As shown in Figure 43, entrainment at IP2 and IP3 would reduce the spawning potential of Hudson River American shad by only 1% compared to the baseline value. According to the ASMFC (2007a), the current rate of total mortality on age 1 and older American shad (Z=0.87) corresponds to an SPR of 0.23, well below the threshold level. Because it was derived from an analysis of long-term trends in abundance and age structure of the Hudson River shad population, the total mortality rate estimate already includes the effects of entrainment at IP2 and IP3. If this contribution (as estimated using the CMR) is removed, the decrease in total mortality and increase in SPR level are negligibly small (Figure 43). Eliminating entrainment at IP2 and IP3 would result in less than a 1% increase in spawning potential, leaving the SPR still substantially below the threshold defined by the ASMFC.

5. Community-Level Trends Analysis

Cooling-water withdrawals impose some incremental additional mortality on species susceptible to entrainment. If entrainment at IP2 and IP3 were having an adverse impact on the Hudson River fish community, then species with high susceptibility to entrainment would be more likely to have declined in abundance over the past 30 years than would species with low susceptibility. Among those species that declined in abundance, the magnitude of the decline should have been greater for species with high susceptibility than for species with low susceptibility. Among species that increased in abundance, the magnitude of the increase should have been lower for species with high susceptibility than for species with low susceptibility.

This hypothesis can be tested using data available from the generators' riverwide survey programs, using data for all Hudson River fish species for which an adequate trends dataset could be developed. The method used to perform the test is analysis of correlations between indices of entrainment susceptibility, as calculated using distributional data obtained from the LRS, and indices of trends in age 0 abundance, obtained from the BSS and FSS.

Evaluating the correlation between entrainment susceptibility and change in YOY abundance requires selecting those species for which data are available for both variables. Entrainment susceptibility at IP2 and IP3 can be estimated by evaluating the distribution of entrainable life stages in the region from which IP2 and IP3 withdraws water in comparison to all the regions sampled. The generators' LRS program is designed to collect such data. The expected effect of continued annual entrainment losses of early life stages of a species, if losses are severe enough to reduce population size, is a decrease in YOY abundance. YOY is the best stage to look for the effect of entrainment losses because entrainment occurs prior to the YOY stage, and because most susceptible species are still in the river during the YOY stage and thus their abundance is measurable. The generators' BSS and FSS sampling programs are designed to monitor YOY abundance.

5.1 Methods

The evaluation involves three steps: (1) calculate a species-specific numeric index of entrainment susceptibility based on data from the LRS; (2) calculate a species-specific numeric index of change in YOY abundance based on data from the BSS and FSS; and (3) determine whether entrainment susceptibility is related to change in age 0 abundance.

Susceptibility to entrainment at IP2 and IP3 was evaluated using an index of standing crop estimated from the generators' LRS for the 31-year period 1974-2004 (Appendix D). Indian Point is located in Region 4 (Figure 1), but because of tidal and nontidal flows, can withdraw water originating in the two adjoining regions as well. Therefore, relative abundance of a species in Regions 3-5 (Figure 1), as compared to the riverwide abundance of that species, was used to define a susceptibility index termed *EntSus*. For each sampled year (and each seasonal period when possible), *EntSus* is estimated for each species as the ratio of standing crop in Regions 3-5 to standing crop in all sampled regions. For those species occurring in more than one of the three seasonal periods, annual *EntSus* values are calculated as an average across periods, p, weighted by abundance for each period:

$$EntSus_i = \frac{\sum_p SC_{ip} EntSus_{ip}}{\sum_p SC_{ip}}$$

where $EntSus_i$ = fraction of species in the Hudson River estuary in the IP2 and IP3 region in year i;

SC_{ip} = sum of abundance of the species within seasonal period p in year i; and

$EntSus_{ip}$ = value of *EntSus* for seasonal period p in year i.

Annual *EntSus* values for each species for each of 31 years (1974-2004) in which the yolk-sac or post yolk-sac stages appeared in the Hudson River are provided in Appendix D.

The BSS and FSS programs were selected as the best potential indicators of long-term relative abundance of fish in the estuary. These programs have sampled the estuary using similar gear and methodology since the early 1970s, although there have been variations in the regions sampled and in time of initiation and end of the sampling across the years. To maintain consistent sampling effort and maximize comparability of results, data are restricted to Regions 1-12, and weeks 31-42, approximately August through October.

As documented in Appendix D, abundance data by species are categorized into two salinity zones, three habitats, and two time periods. The two salinity zones are brackish (Regions 1-6; river miles 12-61) and freshwater (Regions 7-12; river miles 62-152). The three habitats sampled by these surveys are (a) shorezone (bottom area in water 10 ft or less in depth), (b) benthic (volume of water between river bottom and 3 ft above the bottom), and (c) water column (water volume not included in either the shorezone or benthic habitats). Time series of abundance data are divided into two equal periods: Period 1, covering the years 1974 through 1989, and Period 2, covering the years 1990-2005.

Because freshwater and marine species typically have strong salinity preferences, data from the non-preferred salinity zones (brackish zone for freshwater guild; freshwater zone for marine guild) were excluded when calculating overall relative change in abundance from Period 1 to Period 2 for species in these two guilds. So that species with greatly differing abundances could be compared in the same scale, the between-period changes were expressed as a relative change index (i.e., abundance in Period 2 divided by abundance in Period 1). Details concerning these calculations are provided in Appendix D.

The quantity and quality of abundance and distribution data vary greatly among species. The inclusion of species collected only rarely, or only in a small number of years, would weaken the analysis. Selection criteria are needed to eliminate species caught too infrequently to provide meaningful estimates of *EntSus* or meaningful abundance trends. However, any single choice of selection criteria can be questioned. For this reason, a sensitivity analysis was performed to evaluate influence of selection criteria on the outcome of the hypothesis test. The sensitivity analysis was performed by defining two cases, or sets of species, termed "Case A" and "Case B." Species included in both cases were selected based on the annual numbers of organisms collected in the LRS and BSS/FSS surveys. Species were included in the Case A analysis if (1) an average of at least 100 larvae per year of occurrence was collected in LRS samples during 1974-2005 and (2) at least 100 YOY were collected in BSS or FSS samples in at least one salinity zone-habitat combination in at least one of the two time periods. Species were included in the Case B analysis if (1) an average of at least 1000 larvae per year of occurrence was collected in LRS samples 1974-2005 and (2) at least 1000 YOY were collected in BSS or FSS samples in at least one salinity zone-habitat combination in at least one of the two time periods. The species included in Case B are a subset of the species included in Case A. The selection criteria and the species included in each case are more fully documented in Appendix D.

Three correlation metrics (Pearson, Spearman, and Kendall) were used to evaluate the association between entrainment susceptibility and YOY abundance change. There is no simple mathematical relation between any two of these three methods, and when the true correlation coefficient is not zero, it is likely that each coefficient is sensitive to different types of departures from independence (Sokal and Rohlf, 1995).

5.2 Results and Discussion

Table 9 shows the correlation coefficients and probability values, for both Case A and Case B, for all three correlation indices. None of the correlations are statistically significant. Figure 44 provide plots of mean entrainment susceptibility vs. the normalized index of relative change in YOY abundance from Period 1 to Period 2 for both Case A and Case B.

These figures illustrate the same two patterns. First, more species decreased in abundance than increased. For the 21 species in Case A, 71% decreased and 19% increased (Figure 44a). For the 11 species in Case B, 73% decreased and 17% increased (Figure 44b). Second, the regression of relative abundance change on E*ntSus* is not statistically significant for any case, even at the 20% level. This means that relative change from the earlier to the later period was the same for species with high susceptibility to entrainment (high E*ntSus*) as for species with low susceptibility to entrainment. This result is inconsistent with the hypothesis that the susceptibilities of species to entrainment at Indian Point influenced their rates of increase or decrease over the period 1974-2005. Although the number of taxa (19) included in this analysis is small compared to the total number of species present in the Hudson, these taxa represent approximately 94% (Case A) and 88% (Case B) of all age 0 fish captured in the BSS/FSS programs from 1974-2005.

The guild to which each of the 21 species in Case A belongs is indicated in Figure 44a. Although each guild is represented by only four to six species, at least one species in each guild increased in abundance. This pattern further reinforces the conclusion that the long-term trends in abundance of the fish species inhabiting the Hudson River estuary are similar across all guilds and are unrelated to entrainment at IP2 and IP3.

6. Conclusions

The FEIS and the Draft Permit for IP2 and IP3 stated that three fish species (Atlantic tomcod, American shad, and white perch) have declined in abundance in recent years, and attributed these declines to cooling-water withdrawals at IP2 and IP3. Analyses performed to test alternative hypotheses concerning the causes of these declines show that cooling water withdrawals by IP2 and IP3 did not cause these declines. Overharvesting is the most likely cause of recent declines in the abundance of American shad, with striped bass predation being a potentially significant contributing factor. Striped bass predation is the most likely cause of the decline in abundance of Atlantic tomcod (as well as river herring and bay anchovy). Striped bass predation probably contributed to the decline in abundance of YOY white perch, although other unknown causes were also involved. The striped bass hypothesis is supported not only by analysis of species abundance trends, but also by four recently-published studies of striped bass predation (Hartman 2003, Uphoff 2003, Savoy and Crecco 2004, Kahnle and Hattala 2007) and by an analysis of the increase in prey consumption needed to support the recent growth of the Hudson River striped bass population (Appendix C).

Two additional lines of evidence support a conclusion that entrainment and impingement at IP2 and IP3 have not resulted in AEI. Application of the SSBPR model to stock assessment data for striped bass and American shad (Section 4) shows that mortality caused by entrainment at IP2 and IP3 is negligible, particularly compared to fishing mortality, and does not impair the ability of these populations to sustain themselves. Analysis of community-level trends data (Section 5) shows that species with relatively high susceptibility to entrainment at IP2 and IP3 are no more likely to have declined in abundance since 1974 than are species with relatively low susceptibility to entrainment.

Considered together, the evidence evaluated in this report shows that the operation of IP2 and IP3 has not caused effects on early life stages of fish that reasonably would be considered "adverse" by fisheries scientists and/or managers. The effects of mortality at IP2 and IP3 on the survival and abundance of susceptible populations cannot be detected, even after 30 years of intensive monitoring. Those changes that have occurred are more likely attributable to predation by the Hudson River's rapidly growing striped bass population.

For all of the above reasons, from the perspective of a science-based definition of AEI, the available data demonstrate that entrainment and impingement associated with cooling-water withdrawals by IP2 and IP3 have not had an adverse impact on Hudson River fish populations and communities.

7. References

ASA. 2007. 2005 Year Class Report for the Hudson River Estuary monitoring program. ASA Analysis and Communication, Washingtonville, NY.

Atlantic States Marine Fisheries Commission (ASMFC). 1999. Amendment 1 to the Interstate Fishery Management Plan for shad and river herring. Fishery Management Report no. 35. ASMFC, Washington, D.C.

ASMFC 2003. Amendment 6 to the Interstate Fishery Management Plan for Atlantic striped bass. Fishery Management Report No. 41. ASMFC, Washington, D.C.

ASMFC 2004. 2004 review of the Interstate Fishery Management Plan for shad and river herring. ASMFC, Washington, D.C.

ASMFC 2005. 2005 stock assessment report for Atlantic striped bass. AMSFC, Washington, D.C.

ASMFC 2007a. American shad stock assessment for peer review. Stock Assessment Report No. 07-01 (supplement), ASMFC, Washington, D.C.

ASMFC 2007b. Terms of reference & advisory report to the American shad stock assessment peer review. Stock Assessment Report No. 07-01. ASMFC, Washington, D.C.

Barnthouse, L.W., J. Boreman, S.W. Christensen, C.P. Goodyear, W. Van Winkle, and D.S. Vaughan. 1984. Population biology in the courtroom: the Hudson River controversy. *Bioscience* **34:**14-19.

Barnthouse, L.W., D. Glaser, and J. Young. 2003. Effects of historic PCB exposures on the reproductive success of the Hudson River striped bass population. *Environmental Science and Technology* **37**:223-228.

Begon, M., J. L. Harper, and C. R. Townsend. 1996. *Ecology: Individuals, populations, and communities*. 3rd Edition. Blackwell Science, Oxford, UK.

Bigelow, H. B., and W. C. Schroeder. 1953. Fishes of the Gulf of Maine. *U.S. Wildlife Service Fishery Bulletin* **74**:1-576.

Boreman, J., C. P. Goodyear, and S. W. Christensen. 1981. An empirical methodology for estimating entrainment losses at power plants sited on estuaries. *Transactions of the American Fisheries Society* **110**:255-262.

Boreman, J. and R. J. Klauda. 1988. Distribution of early life stages of striped bass in the Hudson River estuary, 1974-1979. *American Fisheries Society Monograph* **4**:53-58.

Brook, B. W., and C. J. A. Bradshaw. 2006. Strength of evidence for density dependence in abundance time series of 1198 species. *Ecology* **87**:1445-1451.

Brosnan, T. M., A. Stoddard, and L. J. Hettling. 2006. Hudson River sewage inputs and impacts: past and present. pp. 335-348 in Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Caraco, N.F., Cole, J.J., Raymond, P.A., Strayer, D.L., Pace, M.L., Findlay, S.E.G., and Fischer, D.T. 1997. Zebra mussel invasion in a large, turbid river: phytoplankton response to increased grazing. *Ecology* **78:** 588-602.

Clarke, G. M., and R. E. Kempson. 1997. *Introduction to the Design and Analysis of Experiments.* John Wiley & Sons, New York.

Cole, J. J., and N. F. Caraco. 2006. Primary production and its regulation in the tidal-freshwater Hudson River. pp. 107-120 in Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Davis, J., E. Schultz, and J. Vokoun. 2007. Assessment of river herring and striped bass in the Connecticut River: Abundance, population structure, and predator/prey interactions. 2006 Progress Report. Submitted to the Connecticut Department of Environmental Protection.

Dayton, P. K., S. Thrush, and F. C. Coleman. 2002. Ecological effects of fishing in marine ecosystems of the United States. Prepared for the Pew Oceans Commission.

Dunning, D.J., J.R. Waldman, Q.E. Ross, and M.T. Mattson. 1997. Use of Atlantic tomcod and other prey by striped bass in the lower Hudson River estuary during winter. *Transactions of the American Fisheries Society* **126**:857-861.

Englert, T. L., J. Boreman, and H. Y. Chen. 1988. Plant flow reductions and outages as mitigative measures. *American Fisheries Society Monograph* **4**:274-279.

Findlay, S., C. Wigand, and W. C. Nieder. 2006. Submersed macrophyte distribution and function in the tidal freshwater Hudson River. Pp. 230-241 in Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Gabriel, W. L., M. P. Sissenwine and W. J. Overholtz. 1989. Analysis of spawning stock biomass per recruit: an example for Georges Bank Haddock. *North American Journal of Fisheries Management* **9**: 383-391.

Gardinier, M. and T.B. Hoff. 1982. Diet of striped bass in the Hudson River estuary. *New York Fish and Game Journal.* **29**:152-165.

Goodyear, C. P. 1977. Assessing the impact of power plant mortality on the compensatory reserve of fish population pp. 186-195. In: Proceedings of the Conference on Assessing the

Effects of Power Plant Induced Mortality on Fish Populations. Pergamon Press, N.Y. W. Van Winkle, [ed].

Goodyear, C.P. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use p. 67-81. In: S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. *Canadian Special Publication in Fisheries and Aquatic Sciences*. 120.

Gotelli NJ. 1995. *A primer of ecology*. Sunderland MA, USA: Sinauer Associates.

Hartman, K.J. 2003. Population-level consumption by Atlantic coastal striped bass and the influence of population recovery upon prey communities. *Fisheries Management and Ecology* **10**:281-288.

Hilborn, R., and M. Mangel. 1977. *The Ecological Detective*. Princeton University Press, Princeton, NJ.

Hattala, K. A., and A. W. Kahnle. 2007. Status of the Hudson River, New York, American shad stock. Pp 209-301 in American shad stock assessment for peer review, Assessment Report No. 07-01 (supplement), ASMFC, Washington, D.C.

Hoff, T. B., J. B. McLaren, and J. C. Cooper. 1988. Stock characteristics of Hudson River striped bass. *American Fisheries Society Monograph* **4**:59-68.

Howarth, R. W., R. Marino, D. P. Swaney, and E. W. Boyer. 2006. Wastewater and watershed influences on primary productivity and oxygen dynamics in the lower Hudson River estuary. pp. 121-139 in Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Kahnle, A. W., and K. A. Hattala. 2007. Striped bass predation on adult American shad: Occurrence and observed effects on American shad abundance in Atlantic coastal rivers and estuaries. Pp. 182-194 in: American shad stock assessment for peer review, Assessment Report No. 07-01 (supplement), ASMFC, Washington, D.C.

Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Luo, J. and J. A. Musick. 1991. Reproductive biology of the bay anchovy in Chesapeake Bay. *Transactions of the American Fisheries Society* **120**:701-710.

Mace, P. M. and M. P. Sissenwine. 1993. How much spawning pre recruit is enough? p. 101-118. In: S. J. Smith, J. J. Hunt and D. Rivard [ed.] Risk evaluation and biological reference points for fisheries management. *Canadian Special Publication in Fisheries and Aquatic Sciences* 120.

McLaren, J. B., T. H. Peck, W. P. Dey, and M. Gardinier. 1988. Biology of Atlantic tomcod in the Hudson River Estuary. *American Fisheries Society Monograph* **4**:102-112.

Murdoch, W. W. 1994. Population regulation in theory and practice. *Ecology* **75**: 271-287.

Olsen, E.J., and R.A. Rulifson. 1992. Maturation and fecundity of Roanoke River-Albermarle Sound striped bass. *Transactions of the American Fisheries Society* **121**:524-537.

Pace, . L., and D. J. Lonsdale. 2006. Ecology of the Hudson River zooplankton community. Pp. 217-229 in Levinton, J. S., and J. R. Waldman (eds.) *The Hudson River Estuary*. Cambridge University Press, New York, NY. 471 p.

Pace, M. L., S. B. Baines, H. Cyr, H, and J. A. Downing. 1993. Relationships among early life stages of *Morone americana* and *Morone saxatilis* from long-term monitoring of the Hudson River Estuary. *Canadian Journal of Fisheries and Aquatic Sci*ences. **50**:1976-1985.

Pew Oceans Commission. 2003. America's living oceans: Charting a course for change.
Quinn, T. J., II, and R. B. Deriso. 1999. *Quantitative Fish Dynamics*. Oxford University Press, New York, NY. 542 p.

Rosenberg, A., P. Mace, G. Thompson, G. Darcy, W. Clark, J. Collie, W. Gabriel, A. MacCall, R. Methot, J. Powers, V. Restrepo, T. Wainwright, L. Botsford, J. Hoenig, and K. Stokes. 1994. Scientific review of definitions of over-fishing in U.S. fishery management plans. NOAA Technical Memorandum NMFS/F-SPO-17. National Marine Fisheries Service, Silver Spring, MD.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry—*The Principles and Practice of Statistics in Biological Research*. 3rd Edition. W. H. Freeman and Company, San Francisco, CA.

Restrepo, V. R., G. G. Thompson, P. M. Mace, W. L. Gabriel, L. L. Low, A. D. MacCall, R. D. Mehot, J. E. Powers, B. L. Taylor, P. R. Wade, and J. F. Witzig. 1998. Technical guidance on the use of precautionary approaches to implementing the Magnuson-Stevens Fishery Conservation and Management Act. NOAA Technical Memorandum NMFS-F/SP-31. National Oceanic and Atmospheric Administration, Washington, D.C.

Rose, K. A., J. H. Cowan, Jr., K. O. Winemiller, R. A. Myers, and R. Hilborn. 2001. Compensatory density-dependence in fish populations: importance, controversy, understanding and prognosis. *Fish and Fisheries* **2**:293-327.

Savoy, T. F., and V. A. Crecco. 2004. Factors affecting the recent decline of blueback herring and American shad in the Connecticut River. *American Fisheries Society Monograph* **9**:361-378.

Secor, D.H. and P.M. Piccoli. 1996. Age- and sex-dependent migrations of striped bass in the Hudson River as determined by microanalysis of otoliths. *Estuaries* **19**:778-793.

Secor, D. H., J. R. Rooker, E. Zlokovitz, and V. S. Zdanowicz. 2001. Identification of riverine, estuarine, and coastal contingents of Hudson River striped bass based on otolith elemental fingerprints. *Marine Ecology Progress Series* **211**:245-253.

Sissenwine, M. P. and J. G. Shepherd. 1987. An alternative perspective on recruitment over-fishing and biological reference points. *Canadian Journal of. Fisheries and Aquatic Sciences* **44**:913-918.

Strayer, D. L., K. A. Hattala, and A. w. Kahnle. 2004. Effects of an invasive bivalve (Dreissena polymorpha) on fish in the Hudson River estuary. *Canadian Journal of Fisheries and Aquatic Sciences* **61**:924-941.

Suter, G. W. II, S. M. Cormier, and S. B. Norton. 2007. Ecological epidemiology and causal analysis. Ch. 4 in G. W. Suter II (ed.) *Ecological Risk Assessment*, 2nd Edition. Taylor & Francis, Boca Raton, FL.

Texas Instruments (TI) 1980. 1978 year class report for the multiplant impact study: Hudson River estuary. Prepared for Consolidated Edison Co. of New York, Inc., 4 Irving Place, New York, NY 10003.

Turchin, P. 1999. Population regulation: a synthetic view. *Oikos* **84**:153-159.

Uphoff, J.H. Jr. 2003. Predator-prey analysis of striped bass and Atlantic menhaden in upper Chesapeake Bay. *Fisheries Management and Ecology* **10**:313-322.

U.S. Environmental Protection Agency. 1998. Guidelines for ecological risk assessment. EPA/630/R-95/002F. U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency. 2002. Summary of biological assessment programs and biocriteria for states, tribes, territories, and interstate commissions: streams and wadeable rivers. EPA-822-R-02-048. U.S. Environmental Protection Agency, Washington, D.C.

Waldman, J. R., D. J. Dunning, Q. E. Ross, and M. T. Mattson. 1990. Range dynamics of Hudson River striped bass along the Atlantic coast. *Transactions of the American Fisheries Society* **119**:910-919.

Waldman, J. R., and M. C. Fabrizio. 1994. Problems of stock definition in estimating relative contributions of Atlantic striped bass to the coastal fishery. *Transactions of the American Fisheries Society* **123**:766-778.

Walter, J.F. III., A.S. Overton, K.H. Ferry, and M.E. Mather. 2003. Atlantic coast feeding habits of striped bass: a synthesis supporting a coast-wide understanding of trophic biology. *Fisheries Management and Ecology* **10**:349-360.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North American fishes: implications for population regulation. *Canadian Journal of Fisheries and Aquatic Sciences* **49**:2196-2218.

World Commission on Environment and Development. 1987. Our Common Future. Report to the United Nations General Assembly, August 4, 1987.

Young-Dubovsky, C., G. R. Shepherd, D. R. Smith, and J. Field. 1996. Striped Bass Research Study: Final Report. Jointly published by the U.S. Fish and Wildlife Service and the National Oceanic and the National Marine Fisheries Service, Washington, D.C.

Table 1. Expected effects of stressors on Hudson River fish populations (except Atlantic tomcod): age 0 growth, age 0 survival, and age 0 spatial distribution, and adult age structure.

Response metric	**CWIS**	**Fishing**	**Zebra mussels**	**Predation by striped bass**
PYSL Abundance	↓	↓	—	↓
PYSL→Juv survival	↓	—	↓	—
Juvenile abundance	—	—	—	↓
Juvenile growth	—	—	↓	—
Spatial distribution	—	—	↓	—

Table 2. Expected effects of stressors on Hudson River fish Atlantic tomcod population: Age 0 survival, age 1 survival, juvenile growth, and spatial distribution.

Response metric	**CWIS**	**Temperature**	**Striped bass predation**
PYSL/early juvenile abundance	↓	—	↓
Egg to age 1 survival	↓	↓	↓
Age 1 &2 abundance	—	—	↓
Age 1 to age 2 survival	—	↓	↓
Juvenile growth	—	↓	—
Spatial distribution	—	—	—

Table 3. Consistency of hypotheses with evaluation criteria: striped bass.

	CWIS	Fishing	Zebra Mussels
Co-occurrence	-	+	+
Sufficiency	N/A	unknown	unknown
Temporality	-	+	-
Manipulation	-	+	N/A
Coherence	N/A	+	-
Summary evaluation	CWIS and zebra mussel hypotheses rejected Most likely cause: fishing		

Table 4. Consistency of hypotheses with evaluation criteria: white perch.

	CWIS	Zebra mussels	Striped bass predation
Co-occurrence	-	+	+
Sufficiency	N/A	unknown	+
Temporality	+	-	+ (?)
Manipulation	-	N/A	N/A
Coherence	N/A	+(?)	+
Summary evaluation	CWIS hypothesis rejected. Zebra mussels and striped bass predation may have contributed declines occurring in later years, but other unknown causes were responsible for declines occurring between 1975 and 1985.		

Table 5. Consistency of hypotheses with evaluation criteria: American shad.

	CWIS	Overfishing	Zebra mussels	Striped bass predation
Co-occurrence	-	+	-	+ (?)
Sufficiency	N/A	+	unknown	unknown
Temporality	-	+	-	+
Manipulation	-	N/A	N/A	N/A
Coherence	N/A	+	-	+
Summary evaluation	CWIS and zebra mussel hypotheses rejected Most likely cause: fishing, with striped bass predation a potential contributing factor			

Table 6. Consistency of hypotheses with evaluation criteria: Atlantic tomcod.

	CWIS	Temperature	Striped bass predation
Co-occurrence	±	+	+
Sufficiency	N/A	+	+
Temporality	-	-	+
Manipulation	-	N/A	N/A
Coherence	N/A	+	+
Summary evaluation	CWIS hypothesis rejected Temperature a significant influence, but cannot explain post-1990 decline Most likely cause of decline: striped bass predation		

Table 7. Consistency of hypotheses with evaluation criteria: River herring.

	CWIS	Zebra mussels	Striped bass predation
Co-occurrence	-	-	+
Sufficiency	N/A	N/A	+
Temporality	-	-	+
Manipulation	-	N/A	N/A
Coherence	N/A	-	+
Summary evaluation	CWIS and zebra mussel hypotheses rejected Most likely cause: striped bass predation		

Table 8. Consistency of hypotheses with evaluation criteria: bay anchovy.

	CWIS	Striped bass predation
Co-occurrence	-	+
Sufficiency	N/A	Unknown
Temporality	-	+
Manipulation	-	N/A
Coherence	N/A	+
Summary evaluation	CWIS hypothesis rejected Striped bass predation most likely cause of change	

Table 9. Pearson, Spearman, and Kendall correlation coefficients for the association between *$Log_{10}(R)$* and mean *EntSus*. A value of *p* represents the probability of a sample correlation coefficient larger than the observed sample correlation coefficient, if the true correlation coefficient is zero.

Case	N		Pearson	Spearman	Kendall
A	19	*r*	0.225	0.182	0.129
		p	0.355	0.457	0.442
B	12	*r*	0.157	-0.042	-0.046
		p	0.625	0.897	0.837

Figure 1. Hudson River map, with sample regions

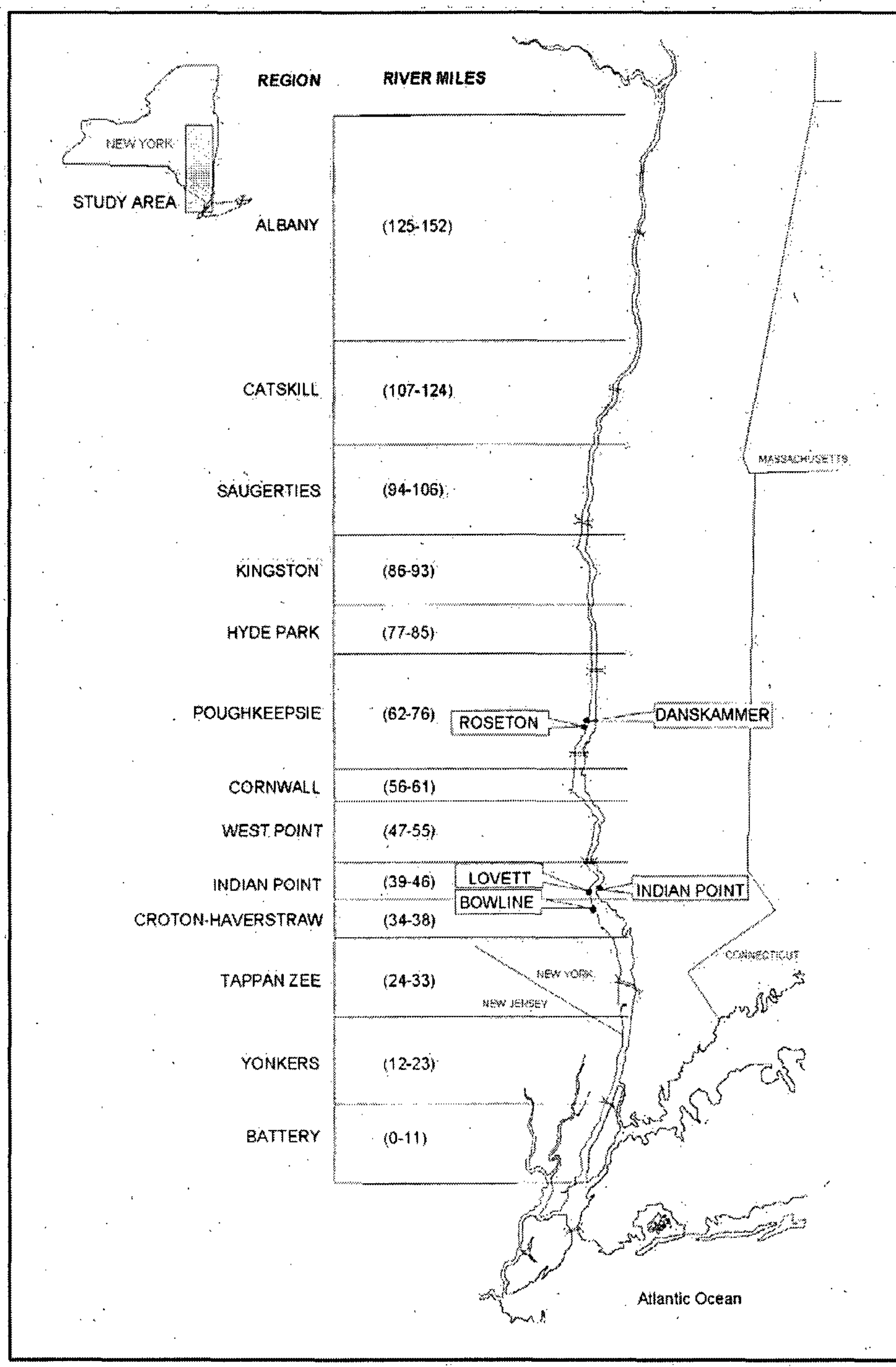


Figure 2. Impacts of CWIS on Age 0 life stages, partitioned between abundance of each life stage and survival between life stages.

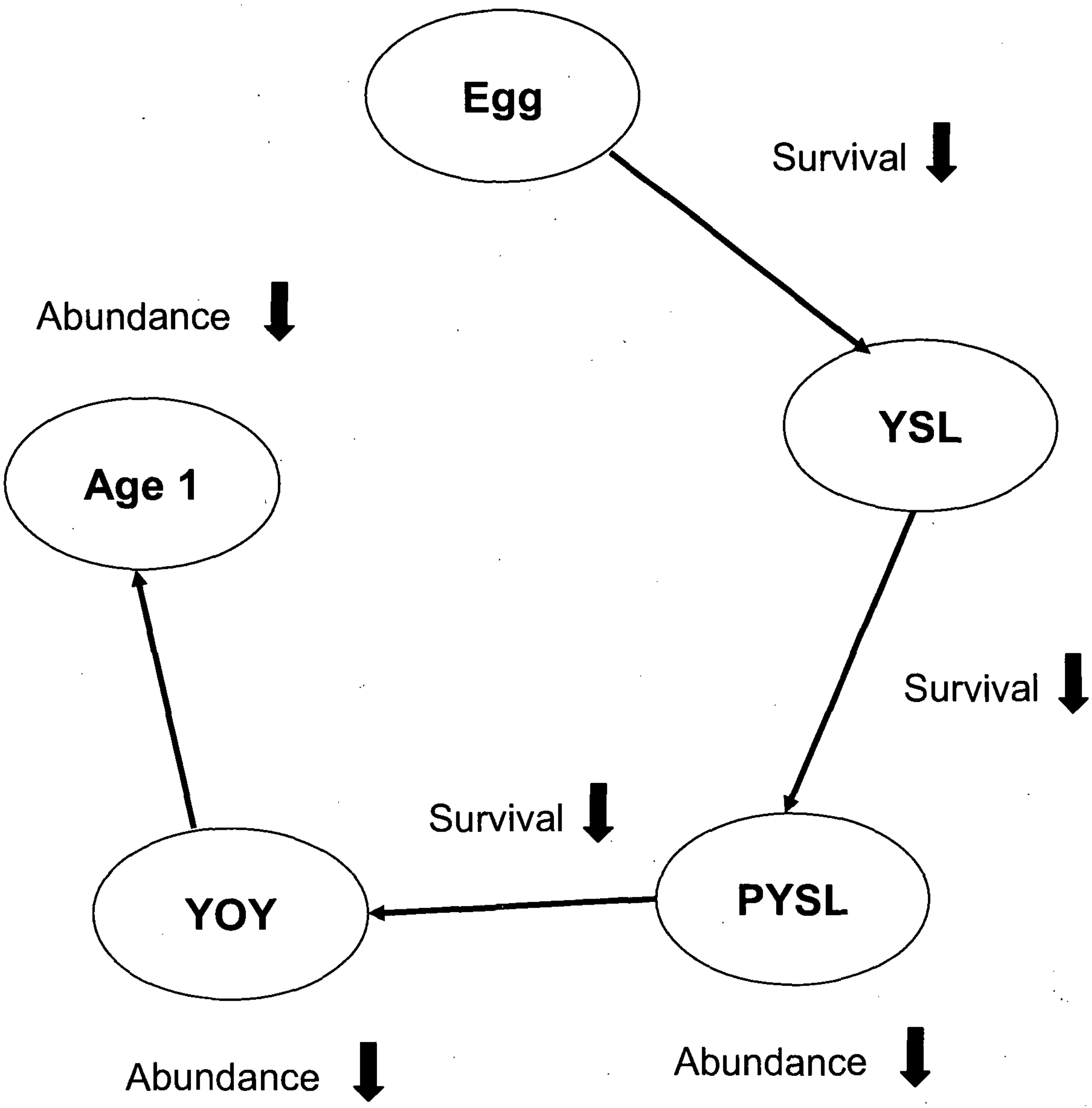


Figure 3. Impacts of fishing on Age 0 life stages.

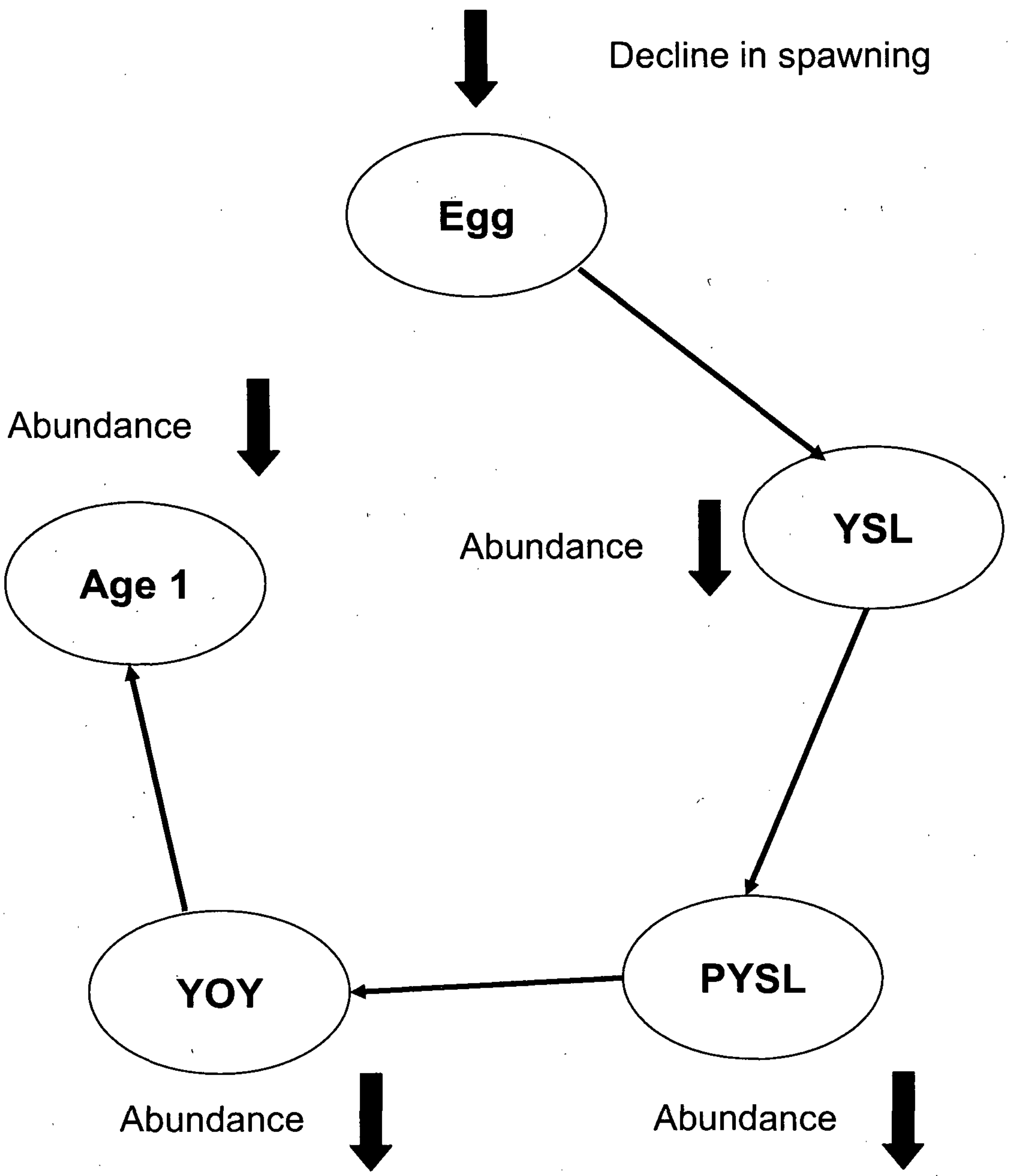


Figure 4. Impacts of zebra mussel activity on Age 0 life stages.

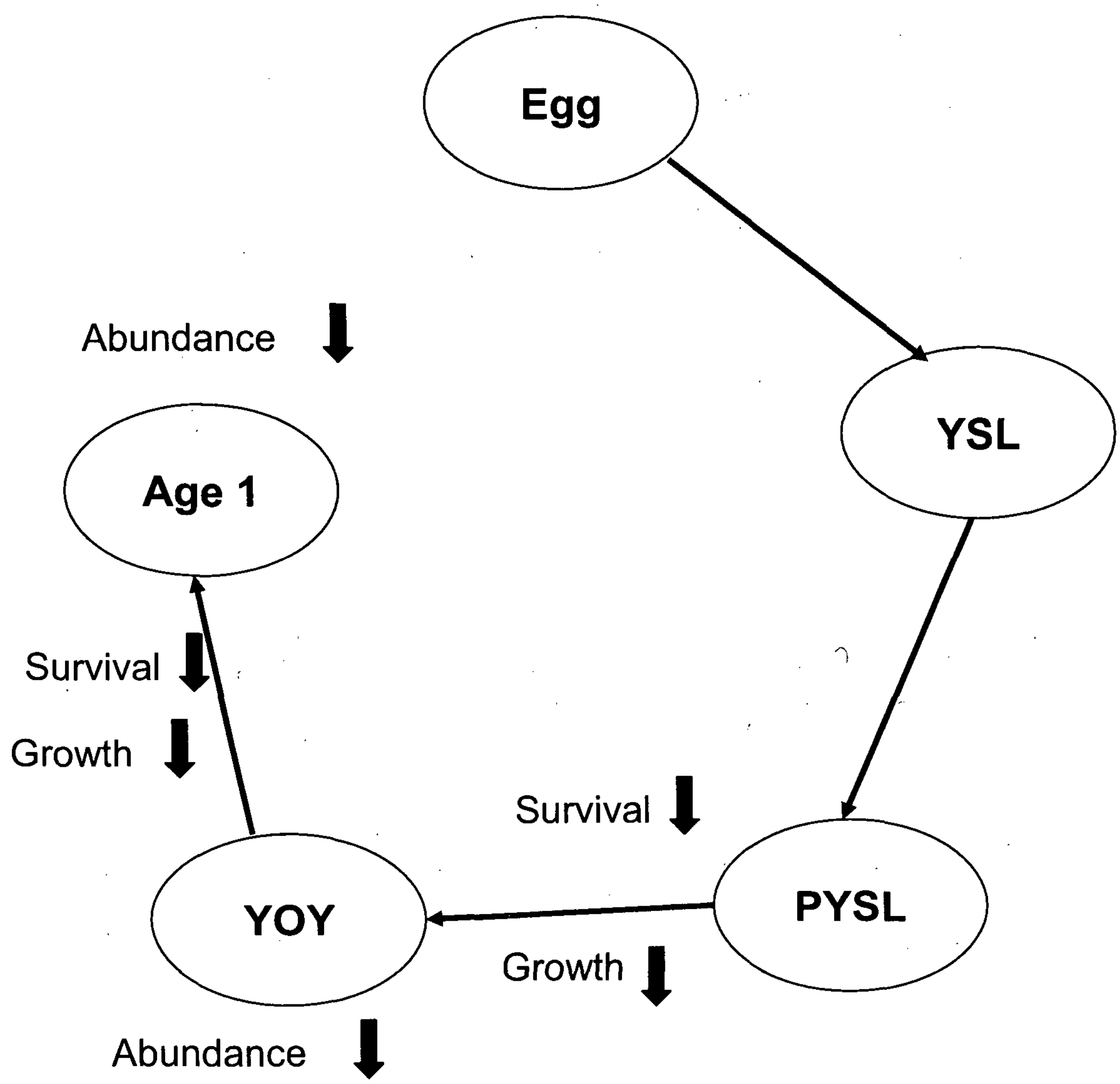


Figure 5. Impact of striped bass predation on Age 0 life stages.

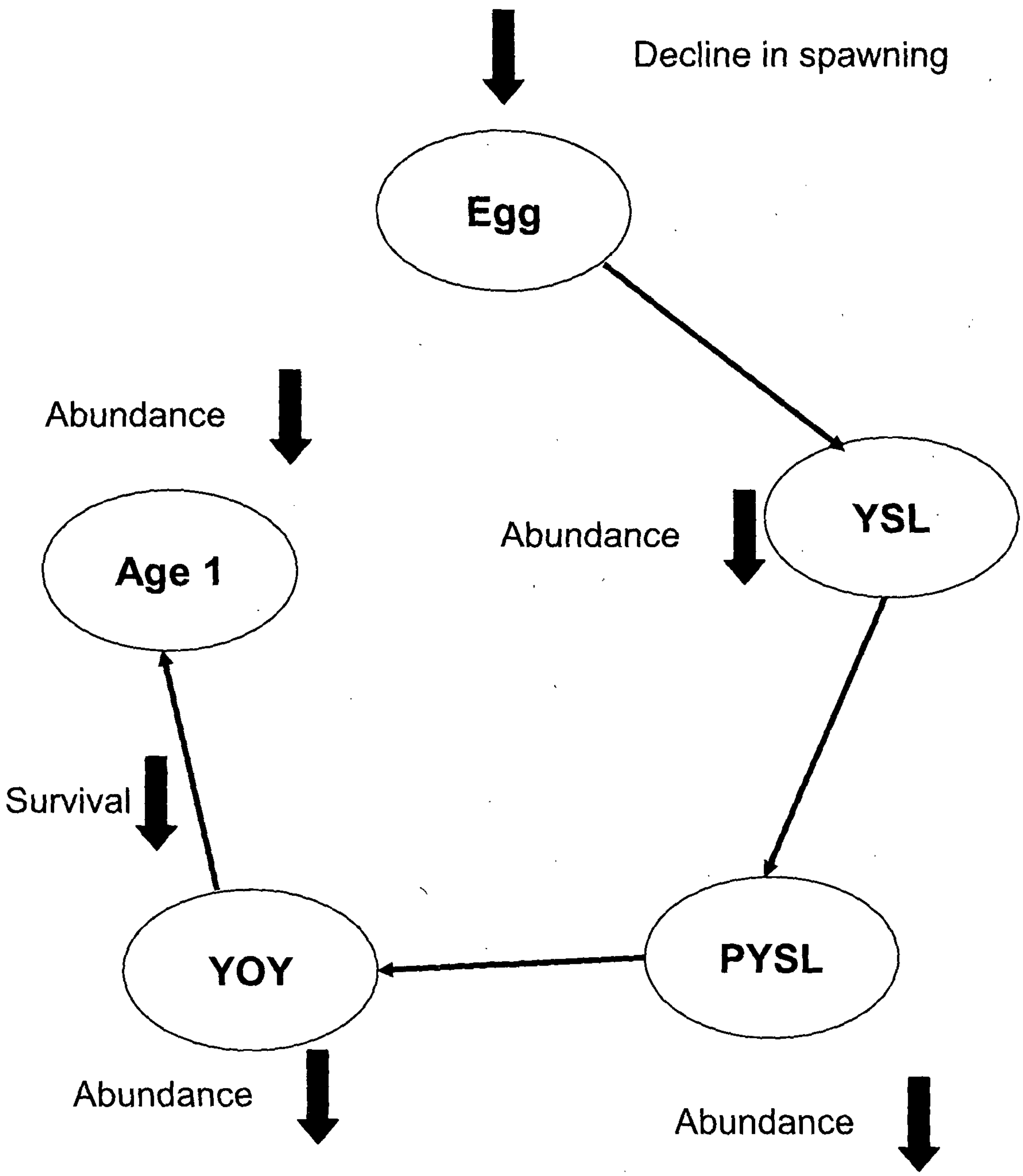


Figure 6. Impact of elevated summer temperatures on Age 0 Atlantic tomcod.

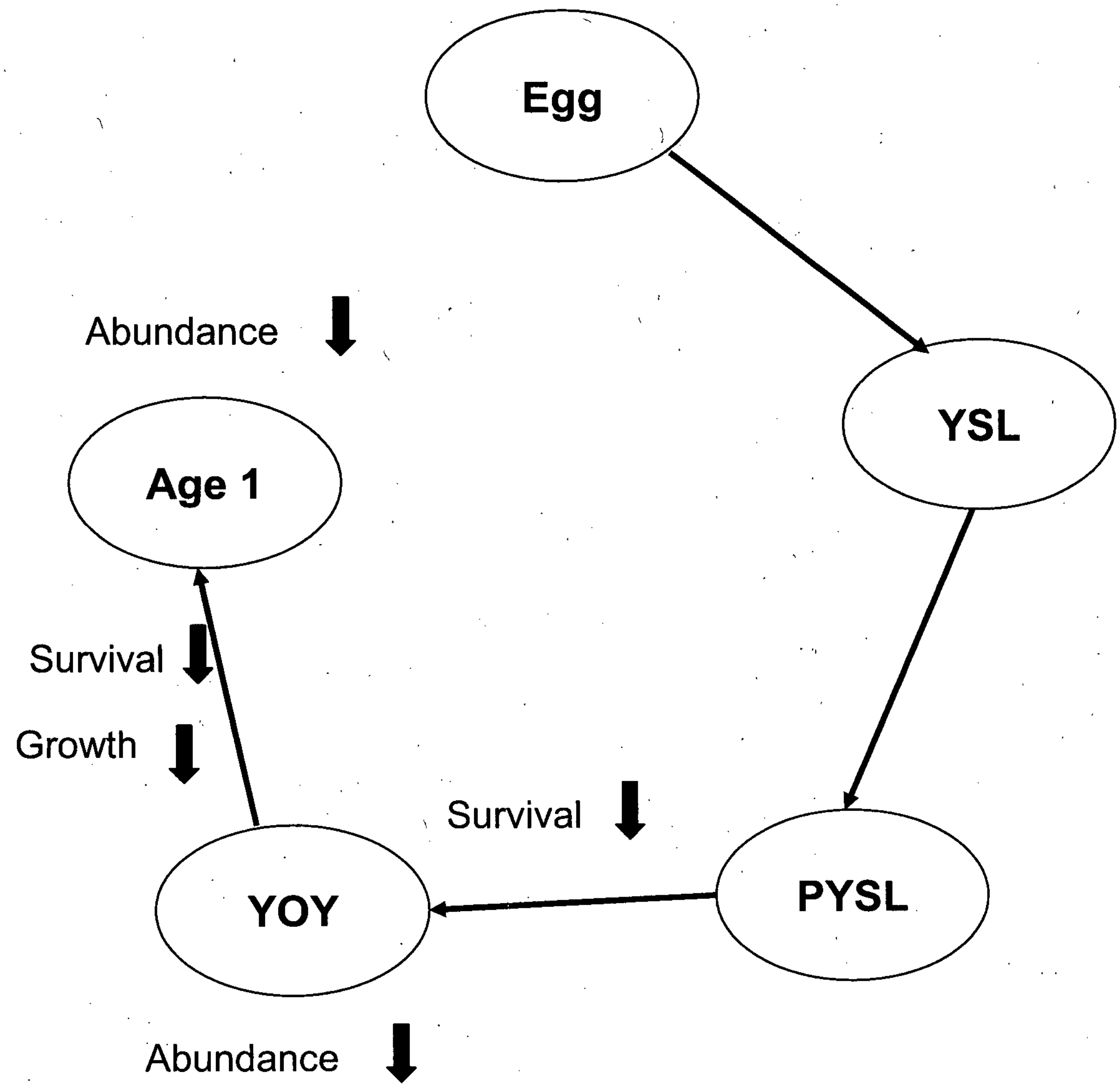


Figure 7a. Long-term trends in the abundance of striped bass PYSL and YOY.

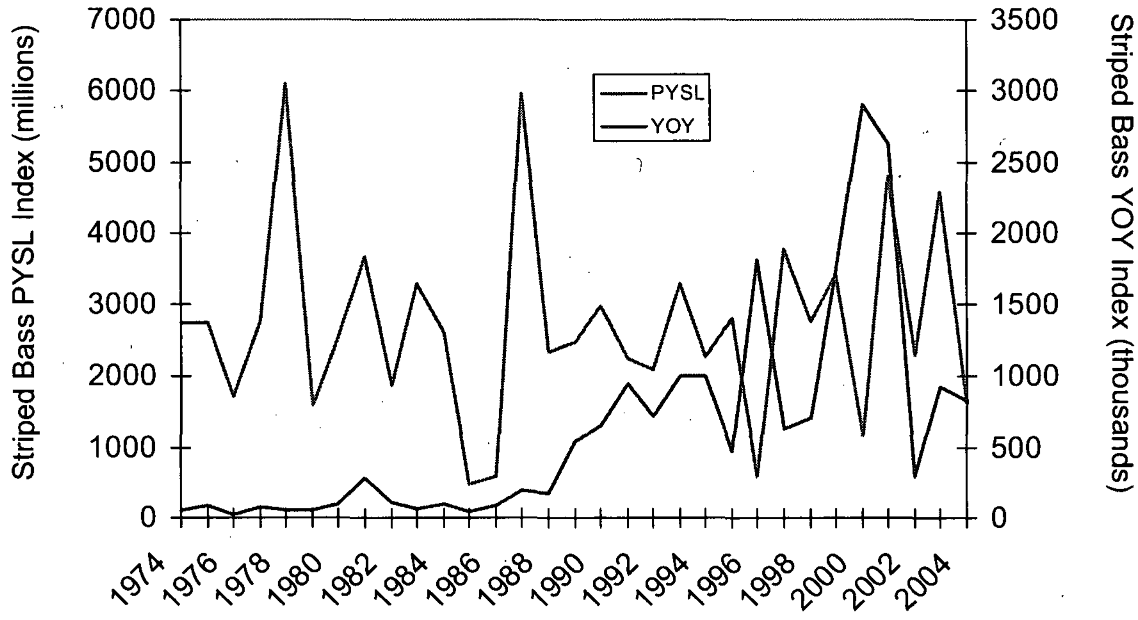


Figure 7b. Long-term trend in striped bass PYSL to YOY survival.

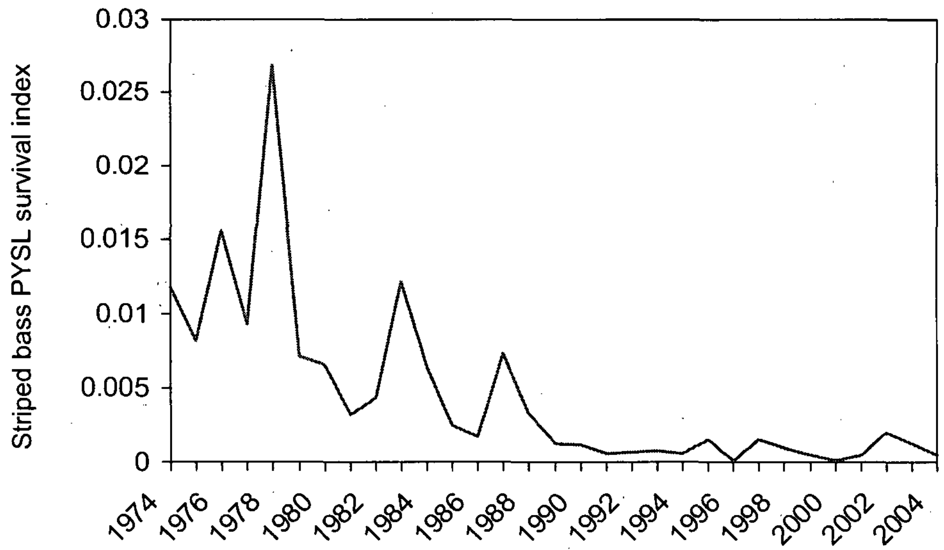


Figure 8a. Relationship between striped bass PYSL abundance and striped bass YOY abundance.

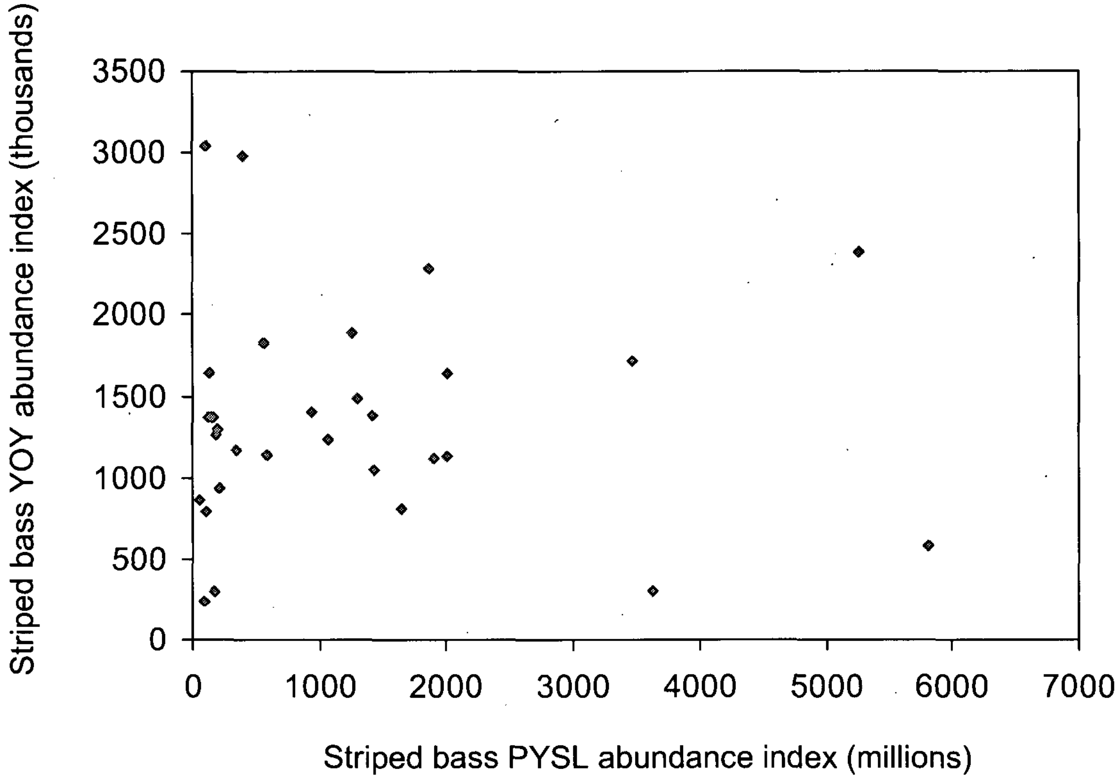


Figure 8b. Relationship between striped bass PYSL abundance and PYSL survival.

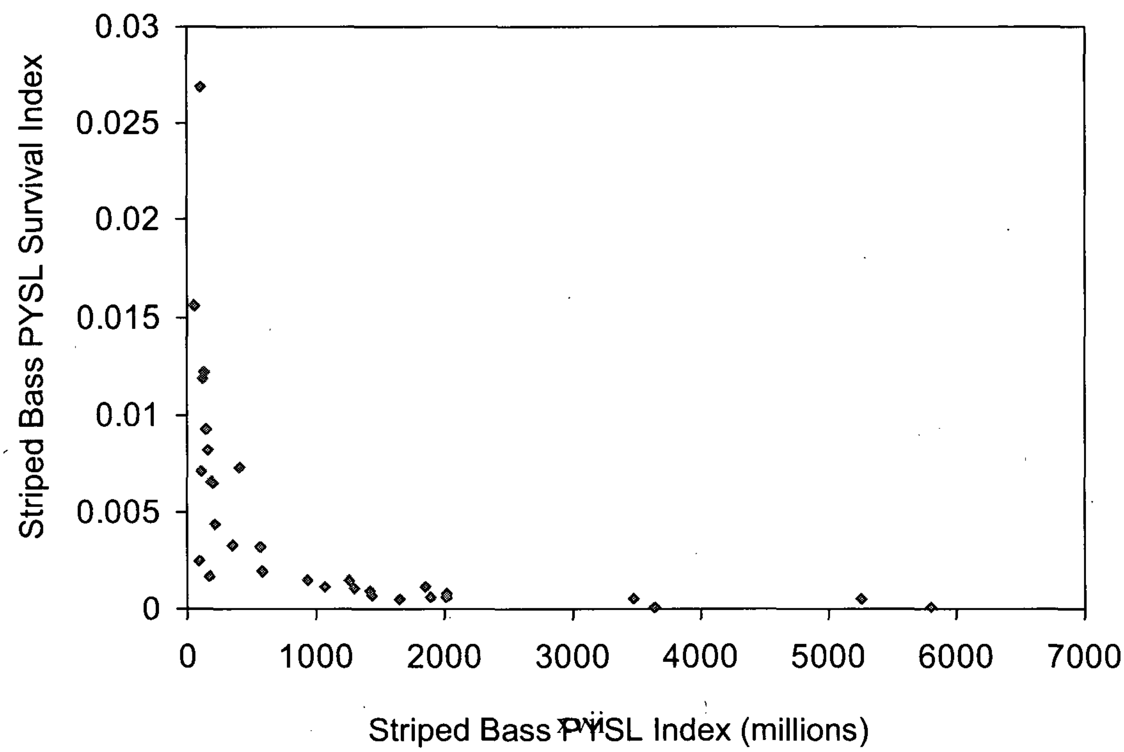


Figure 9a. Relationships between IP2 and IP3 CMR for striped bass and striped bass PYSL survival index.

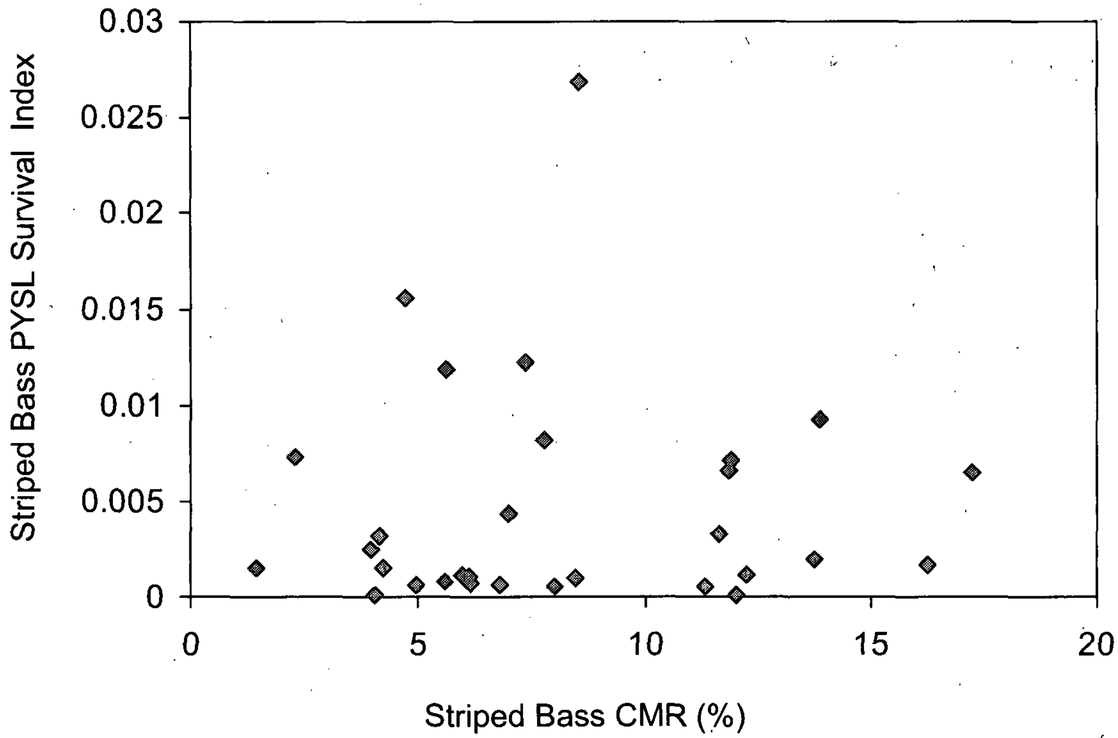


Figure 9b. Relationship between IP2 and IP3 CMR for striped bass and striped bass PYSL abundance index.

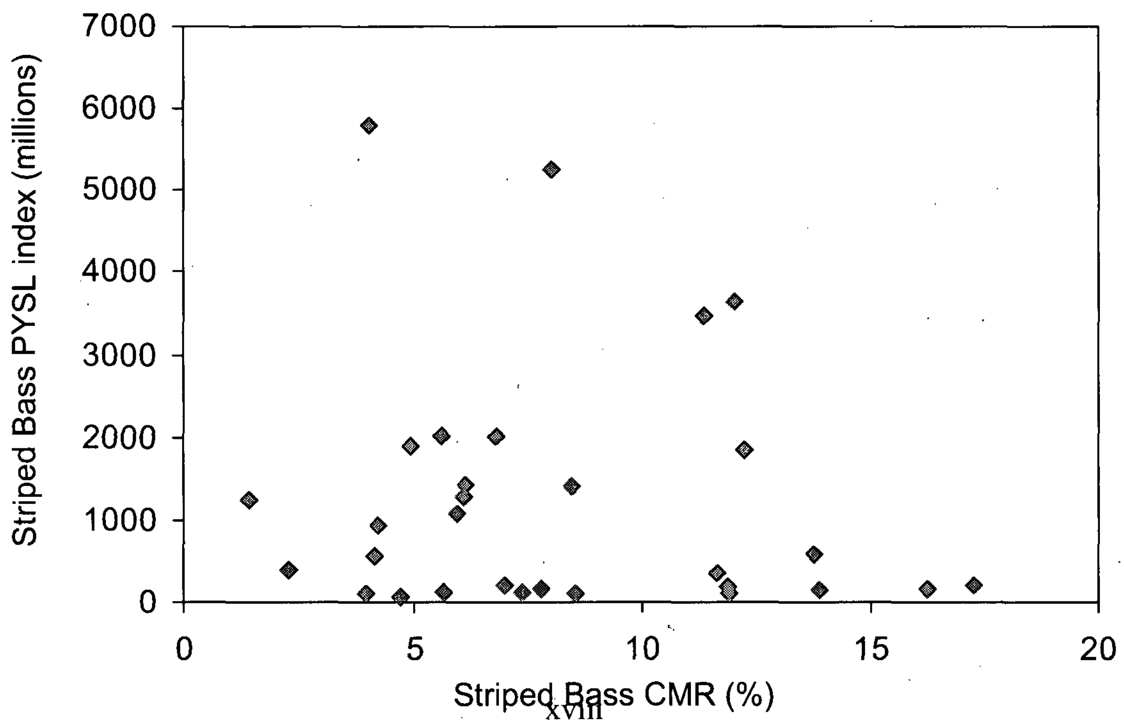
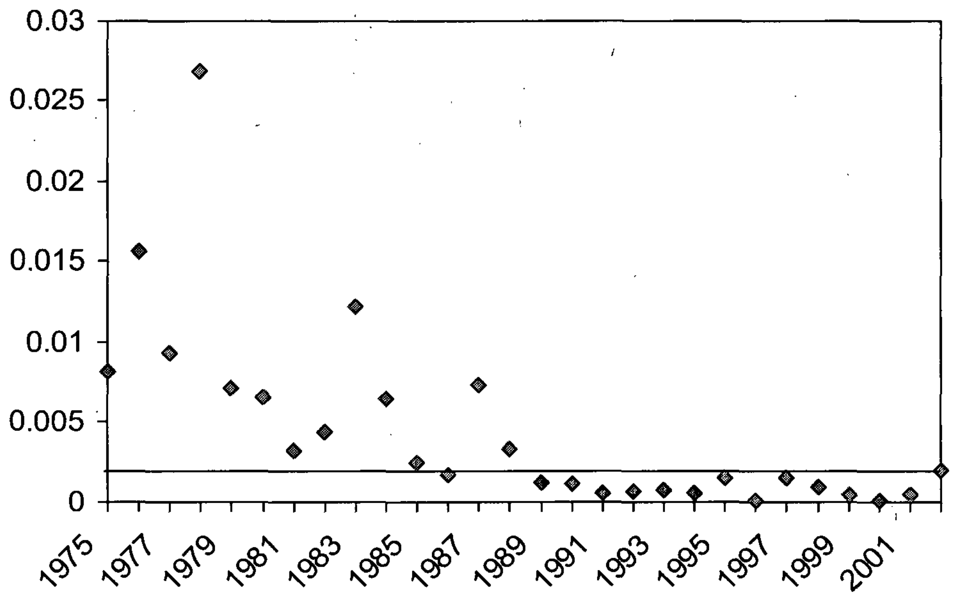


Figure 10. (a) Striped bass PYSL to YOY survival during years in which 1 unit (blue) and 2 units (red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of striped bass are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between total May-June withdrawals by IP2 and IP3 and striped bass PYSL survival.

(a)



(b)

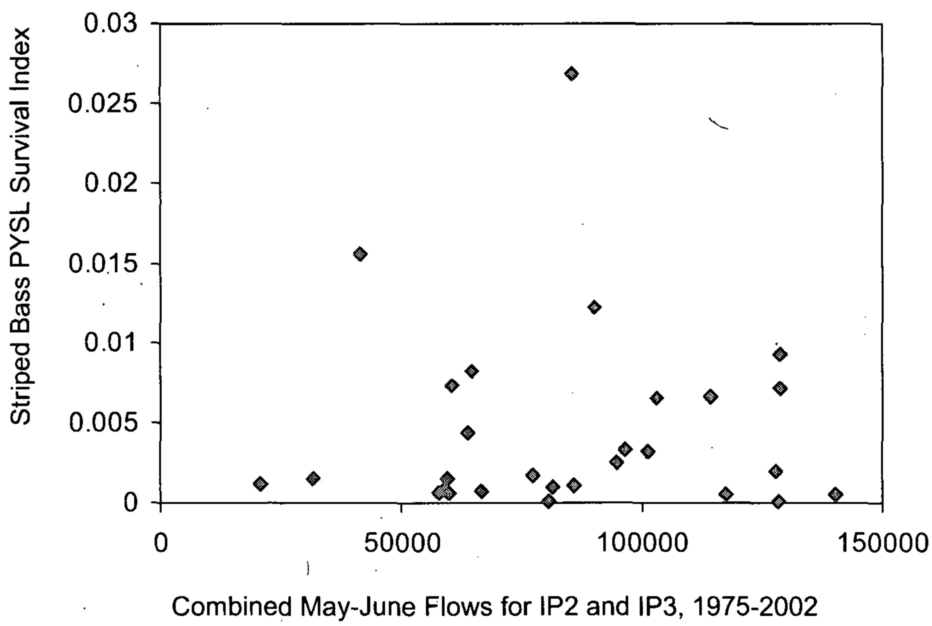


Figure 11. Long-term trends in the abundance of white perch PYSL and YOY in the Hudson River.

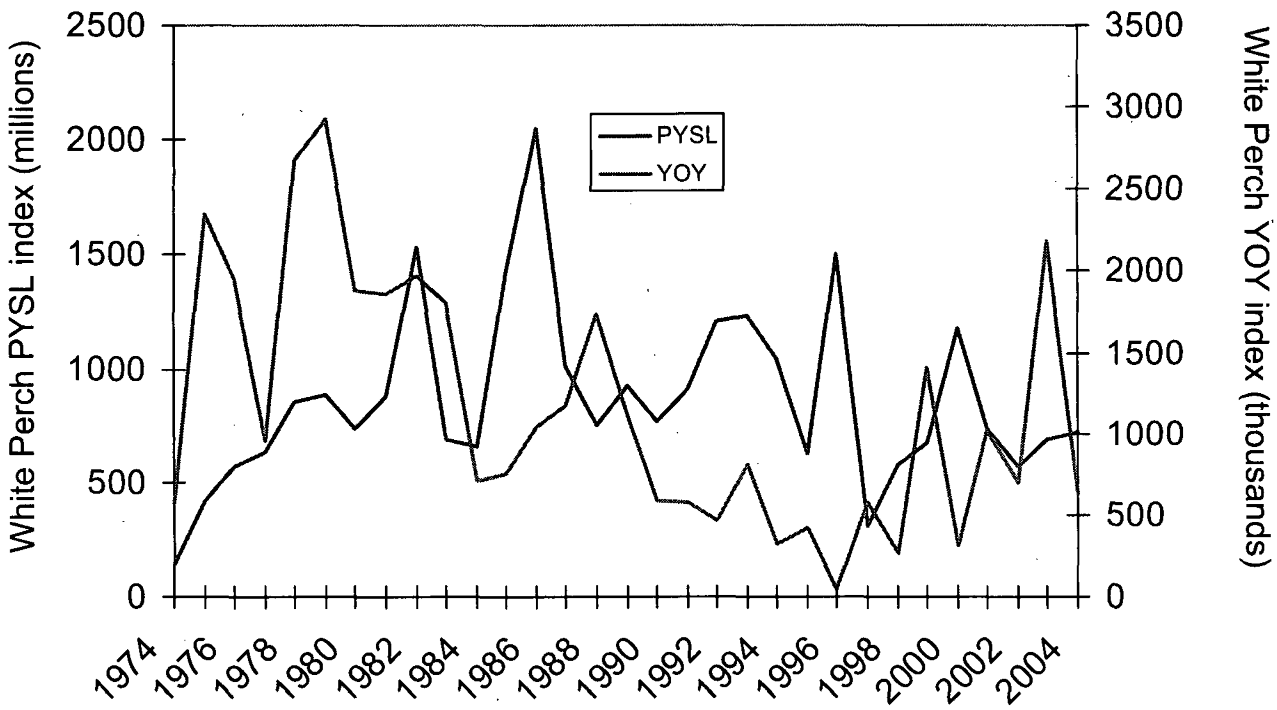


Figure 12a. Relationship between white perch PYSL abundance and YOY abundance.

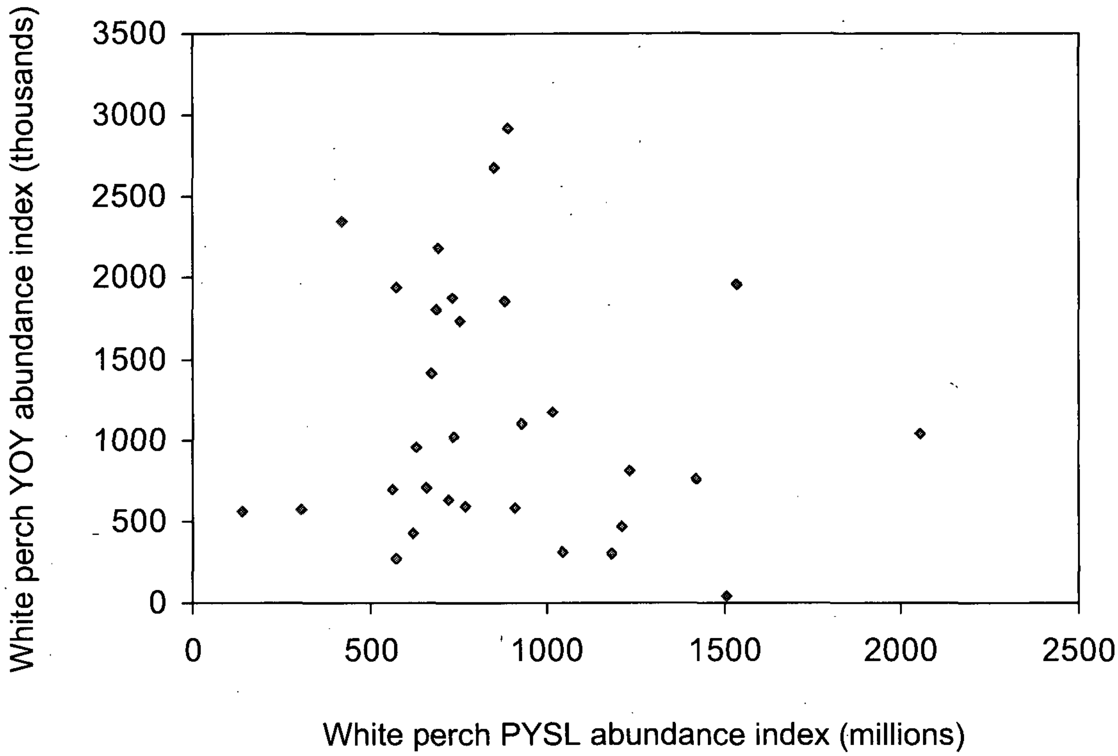


Figure 12b. Relationship between white perch PYSL survival and YOY abundance.

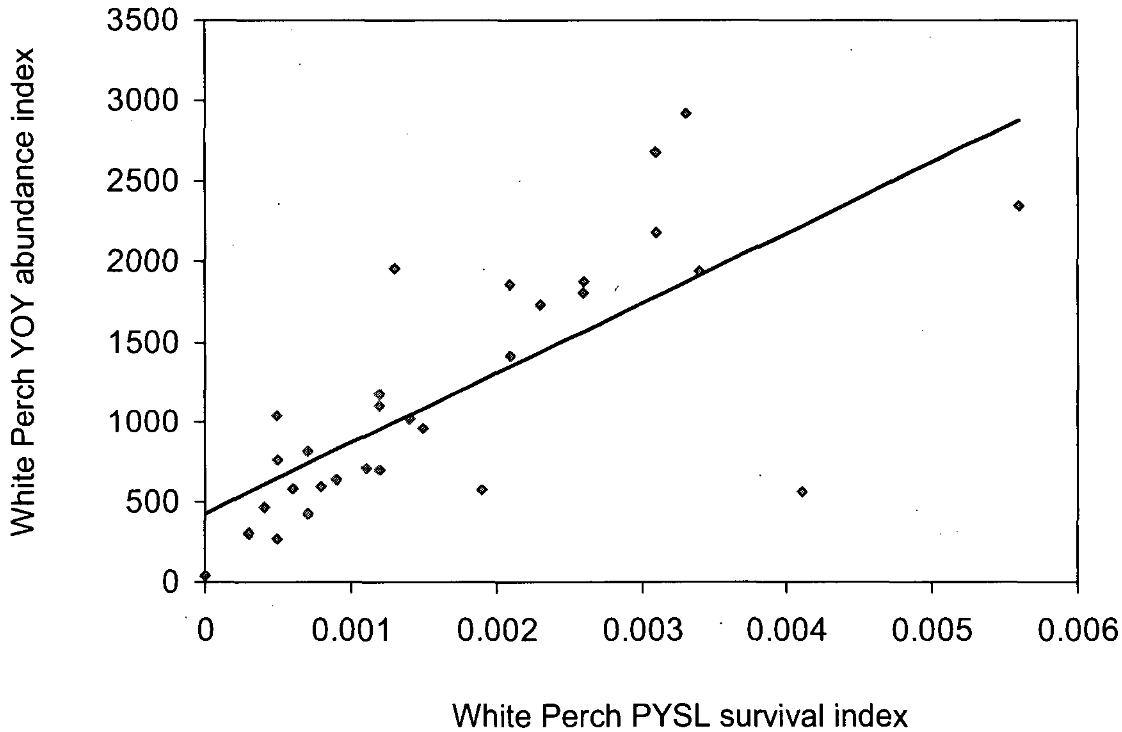


Figure 13a. Relationship between the IP2 and IP3 CMR for white perch and the white perch PYSL survival index.

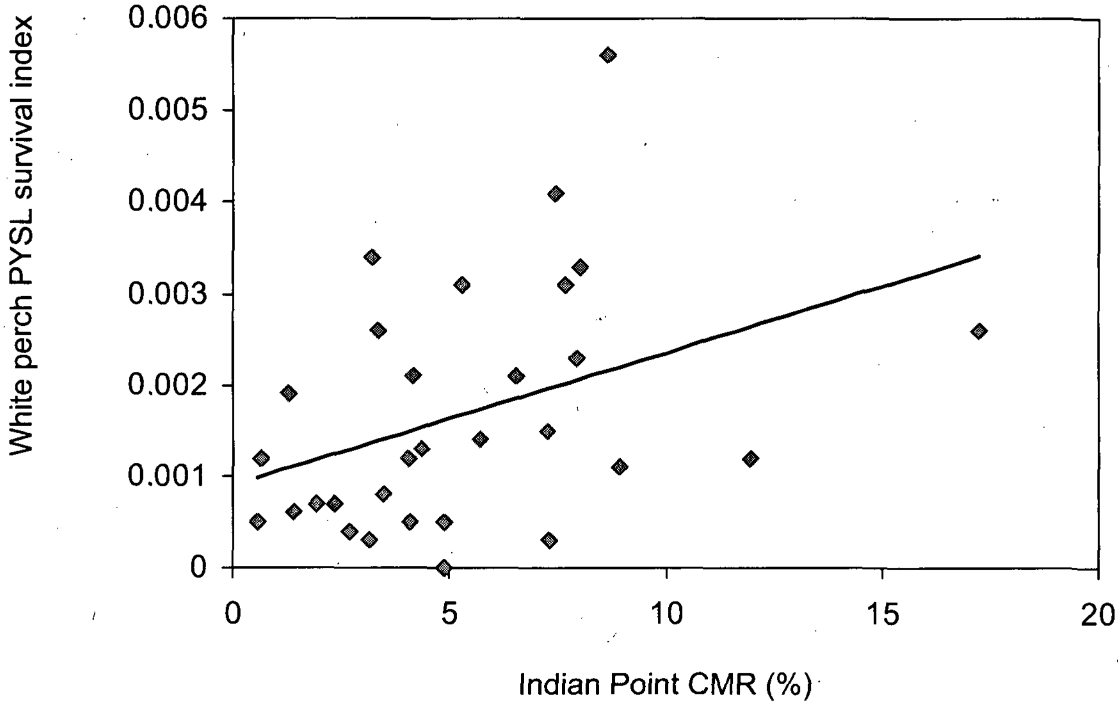


Figure 13b. Relationship between the IP2 and IP3 CMR for white perch and the white perch PYSL abundance index.

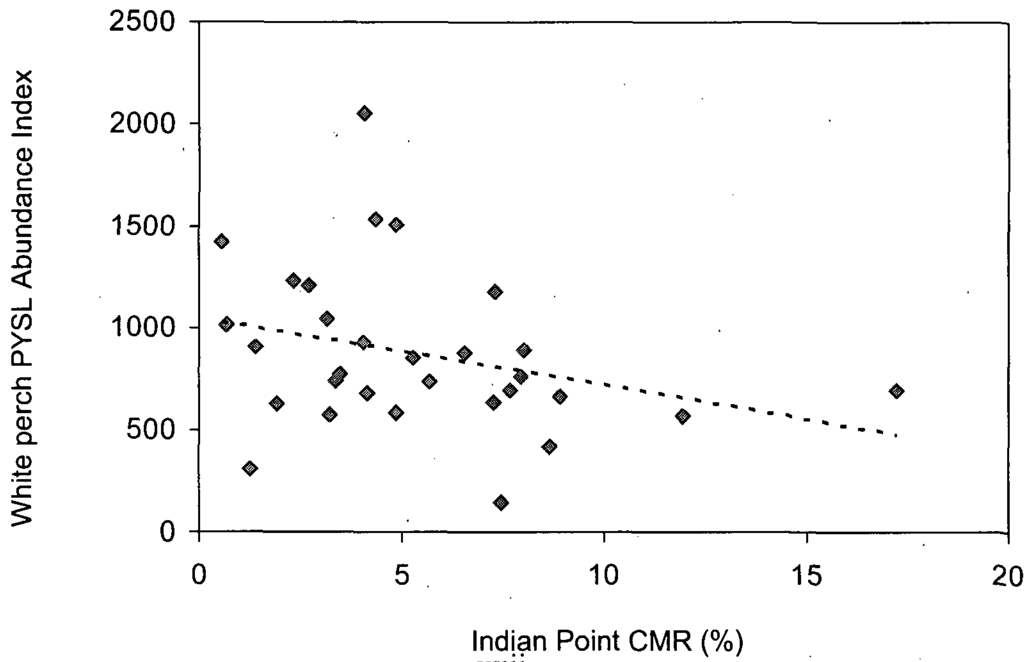


Figure 14. Long-term trends in IP2 and IP3 CMR for white perch and white perch PYSL survival.

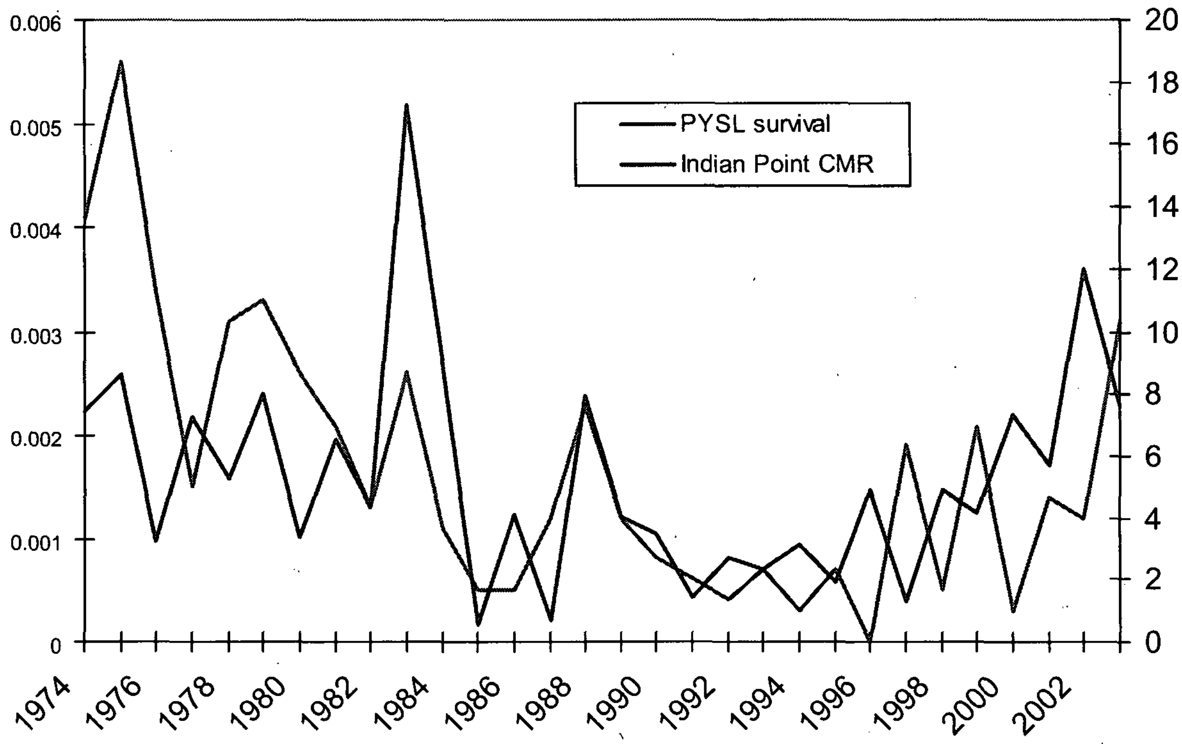
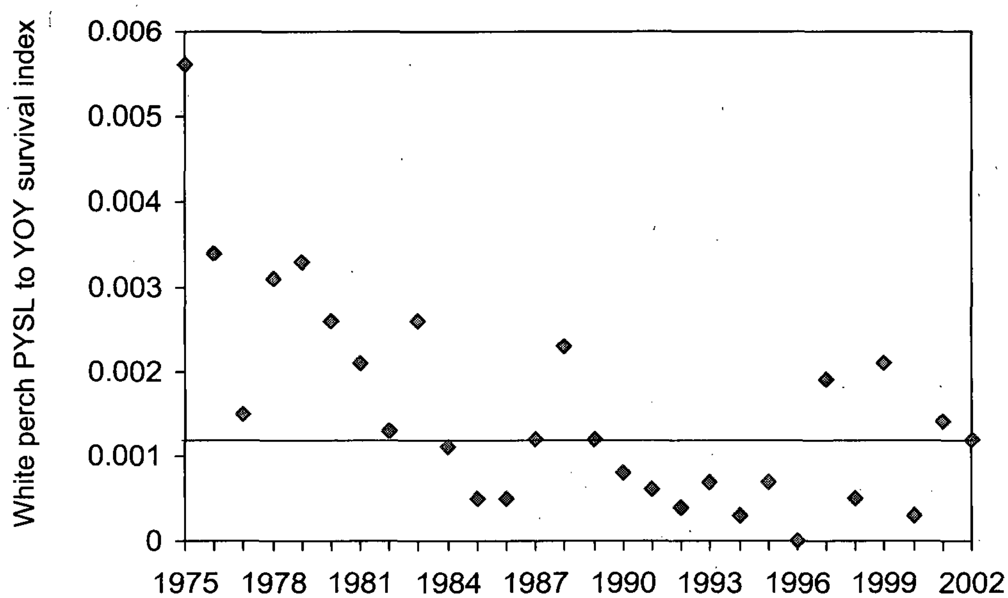


Figure 15. (a) White perch PYSL to YOY survival during years in which 1 unit (blue) and 2 units s(red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of white perch are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between total May-June withdrawals by IP2 and IP3 and white perch PYSL survival.

(a)



(b)

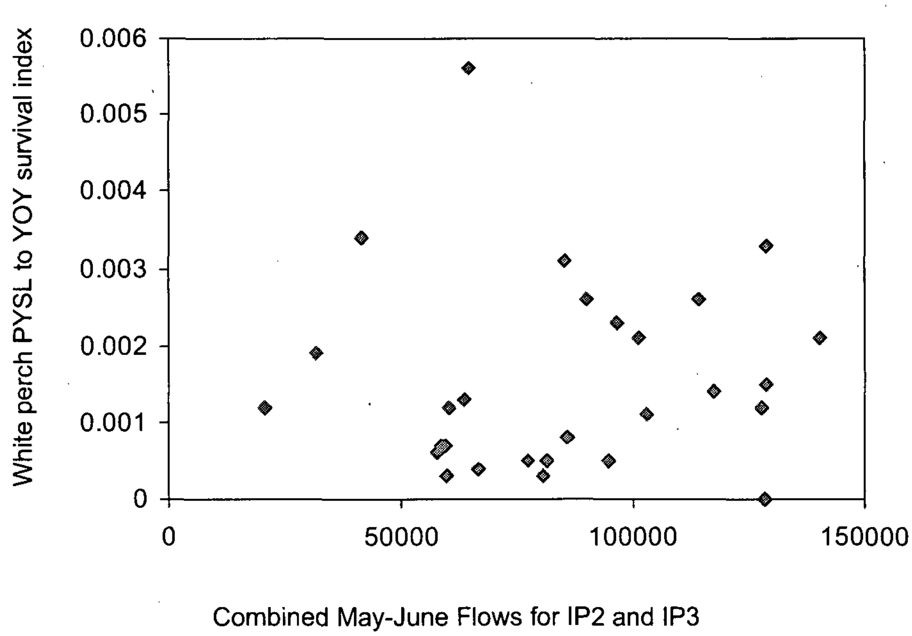


Figure 16a. Relationship between white perch YOY abundance and the striped bass predation index.

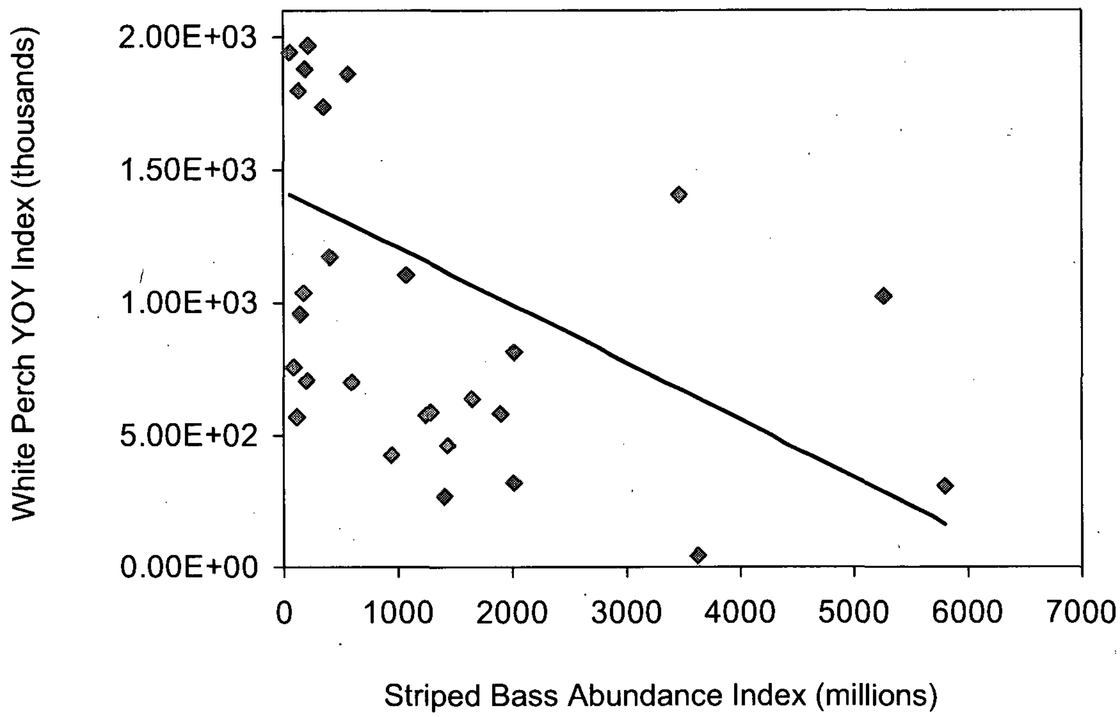


Figure 16b. Long-term trends in white perch YOY abundance and the striped bass predation index.

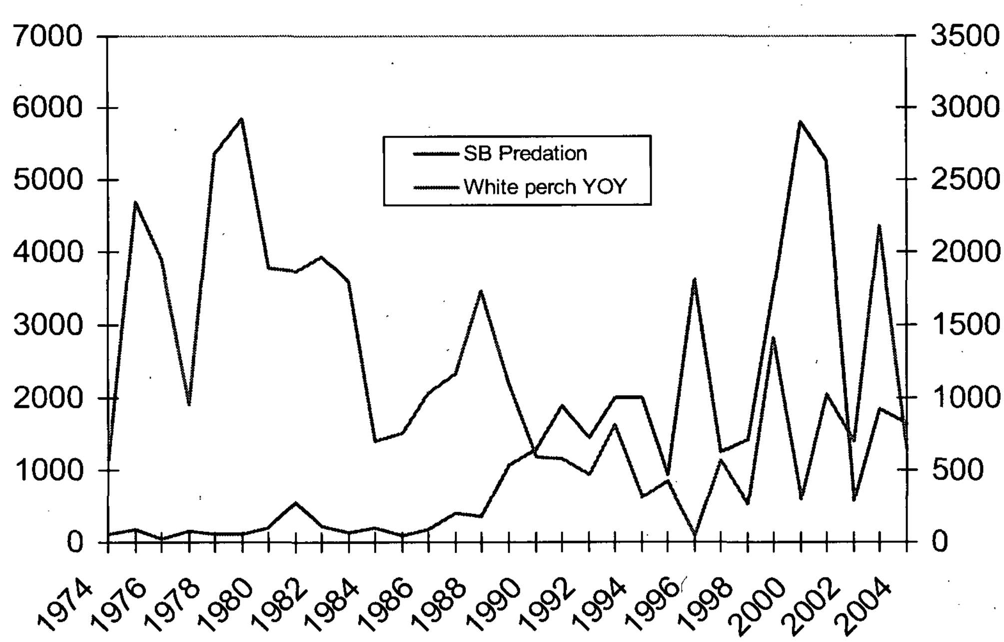


Figure 17. Long-term trends in abundance of American shad PYSL and YOY abundance in the Hudson River.

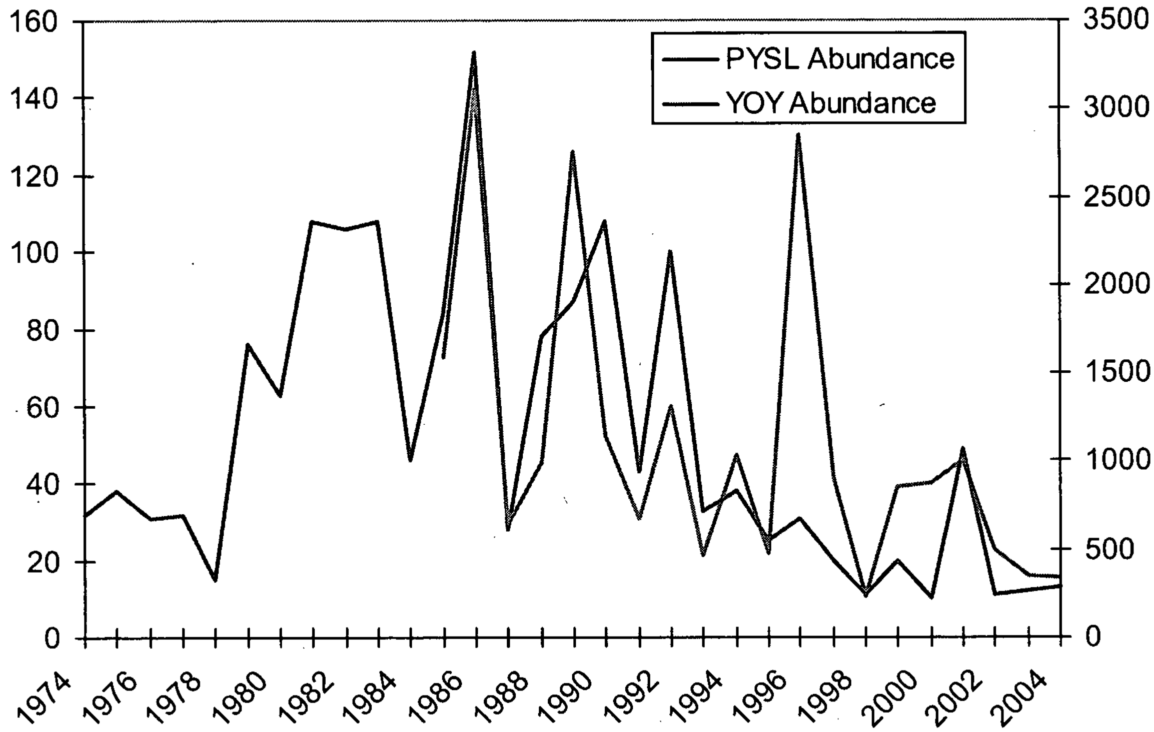


Figure 18a. Relationship between American shad PYSL abundance and YOY abundance in the Hudson River.

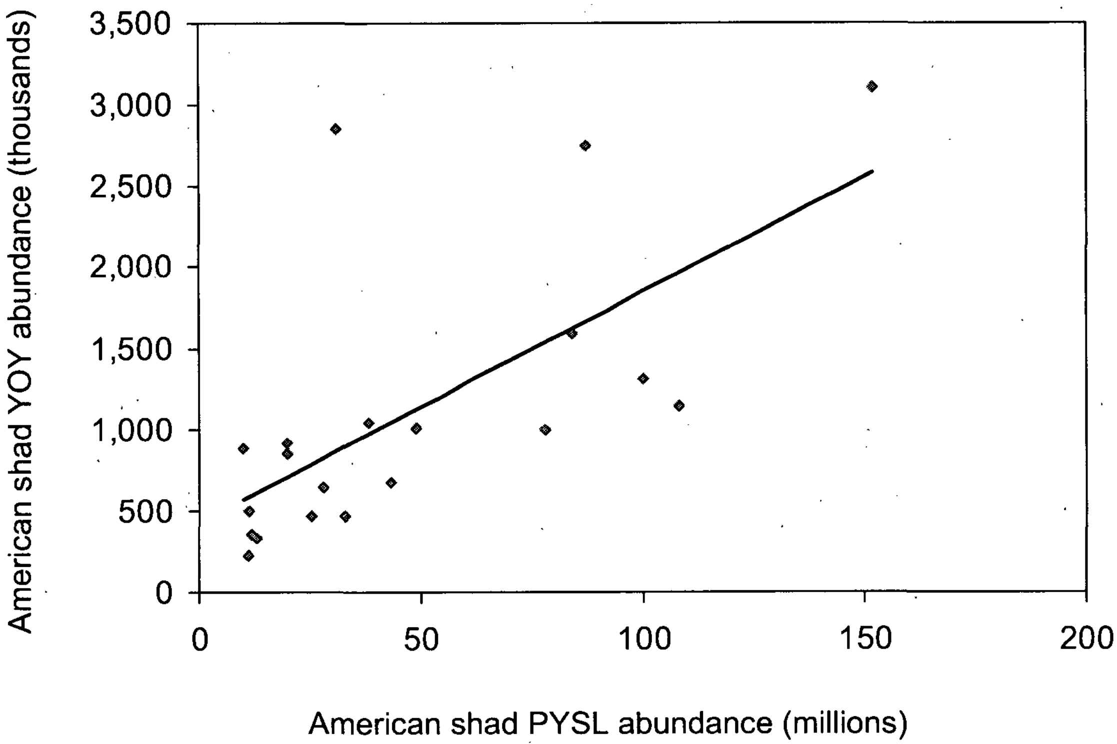


Figure 18b. Relationship between American shad PYSL survival and YOY abundance in the Hudson River.

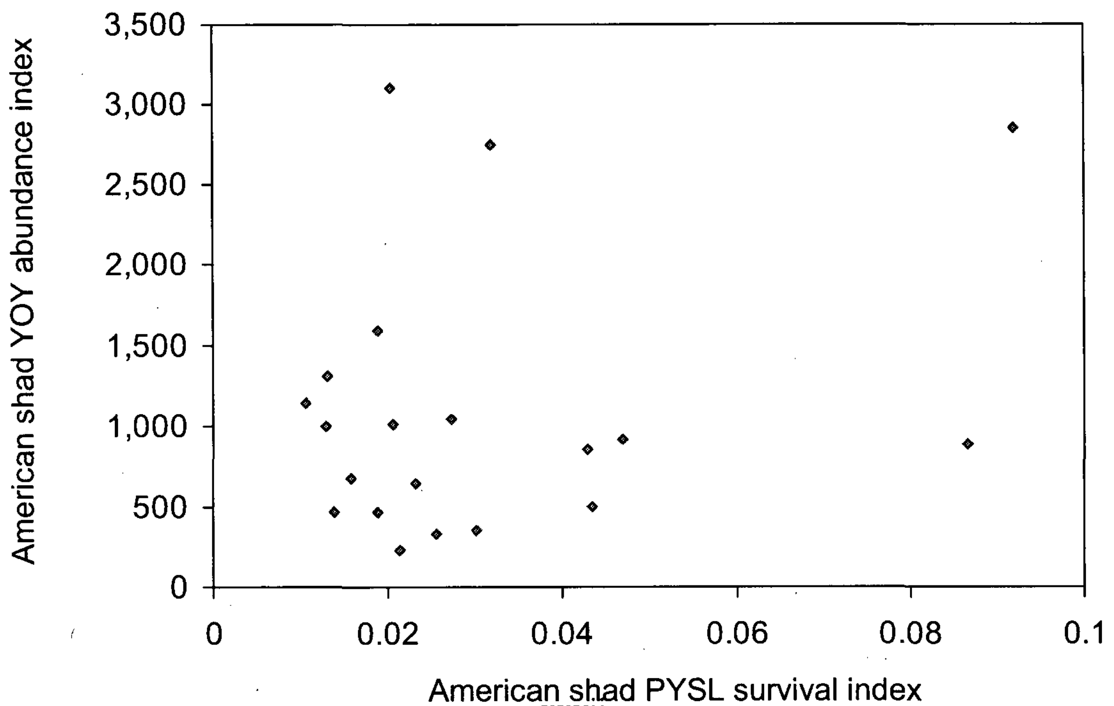


Figure 19a. Relationship between the IP2 and IP3 CMR for American shad and American shad PYSL survival.

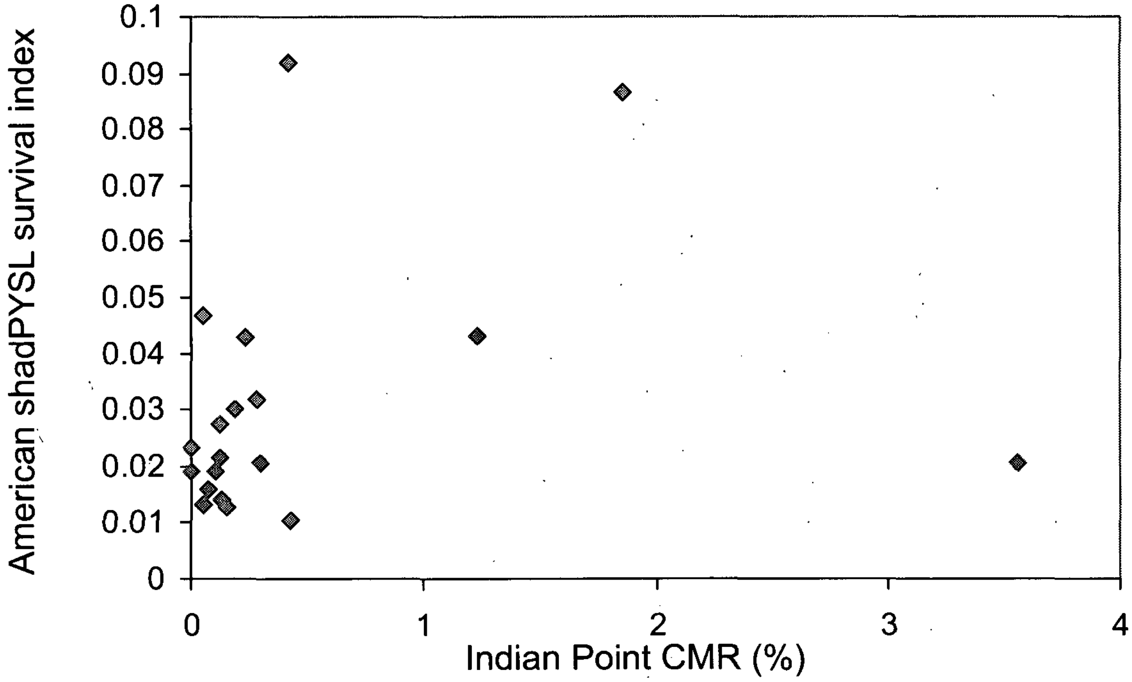


Figure 19b. Relationship between the IP2 and IP3 CMR for American shad and American shad PYSL abundance.

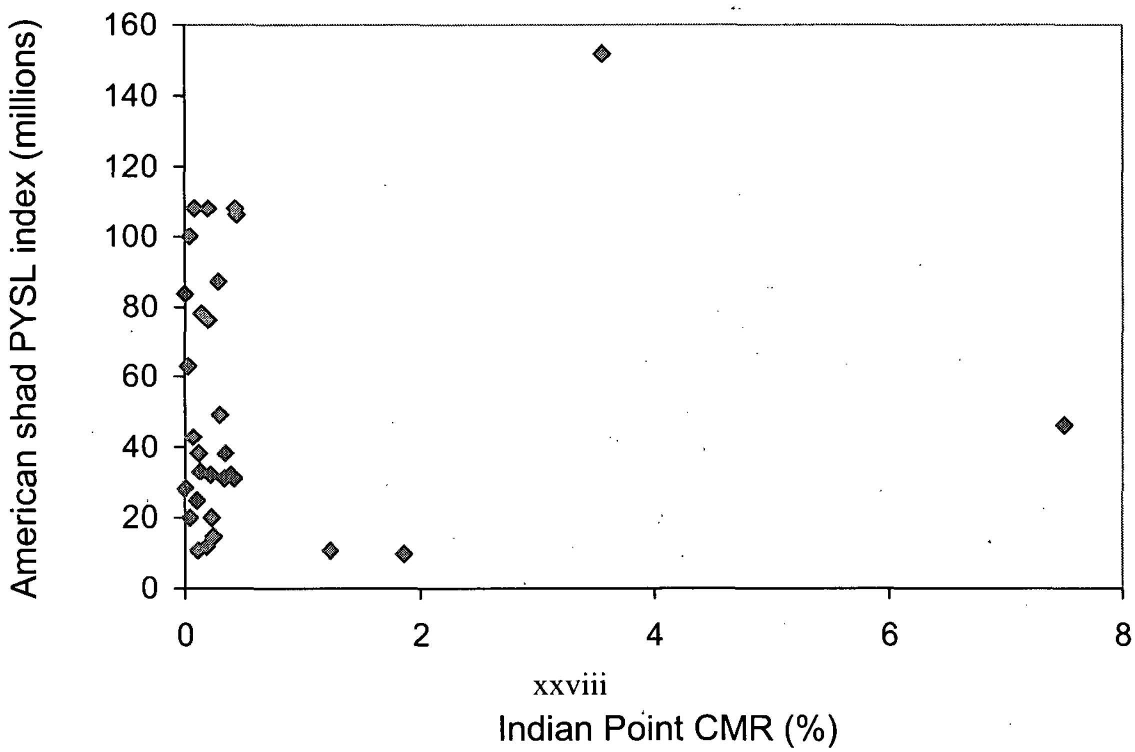
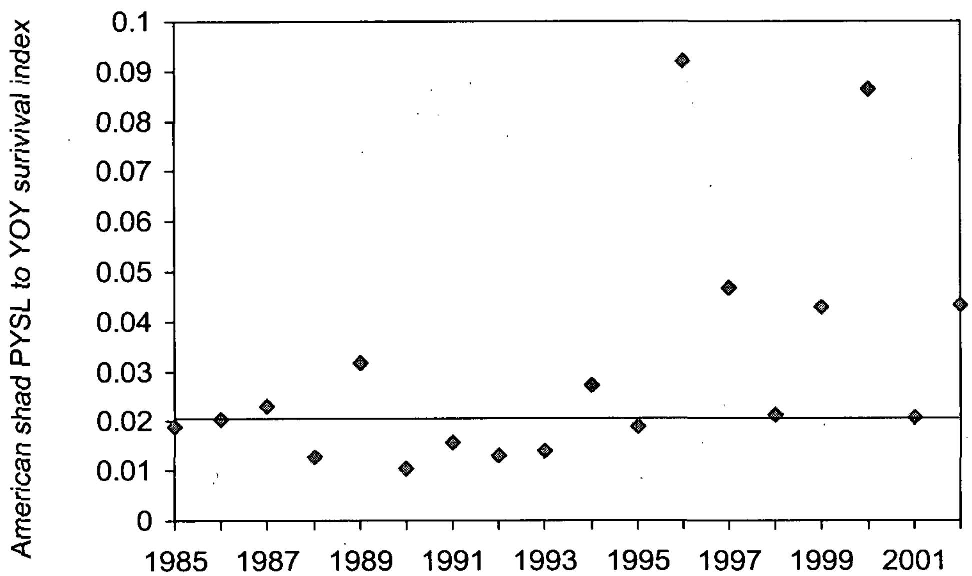


Figure 20. (a) American shad PYSL to YOY survival during years in which 1 unit (blue) and 2 units s(red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of American shad are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between total May-June withdrawals by IP2 and IP3 and American shad PYSL survival.

(a)



(b)

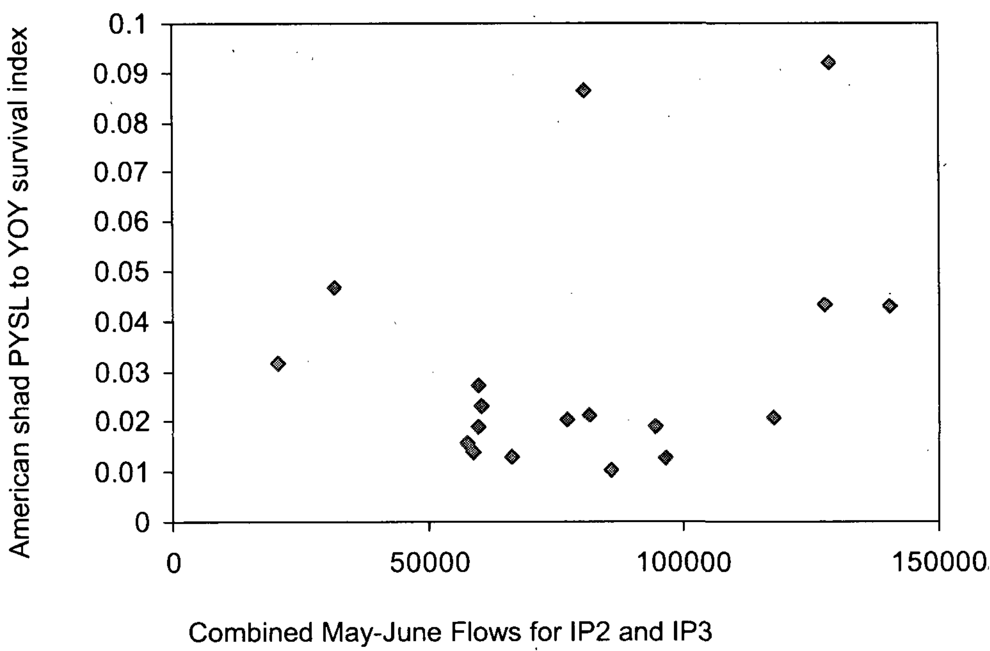


Figure 21a. Relationship between American shad PYSL abundance and the striped bass predation index.

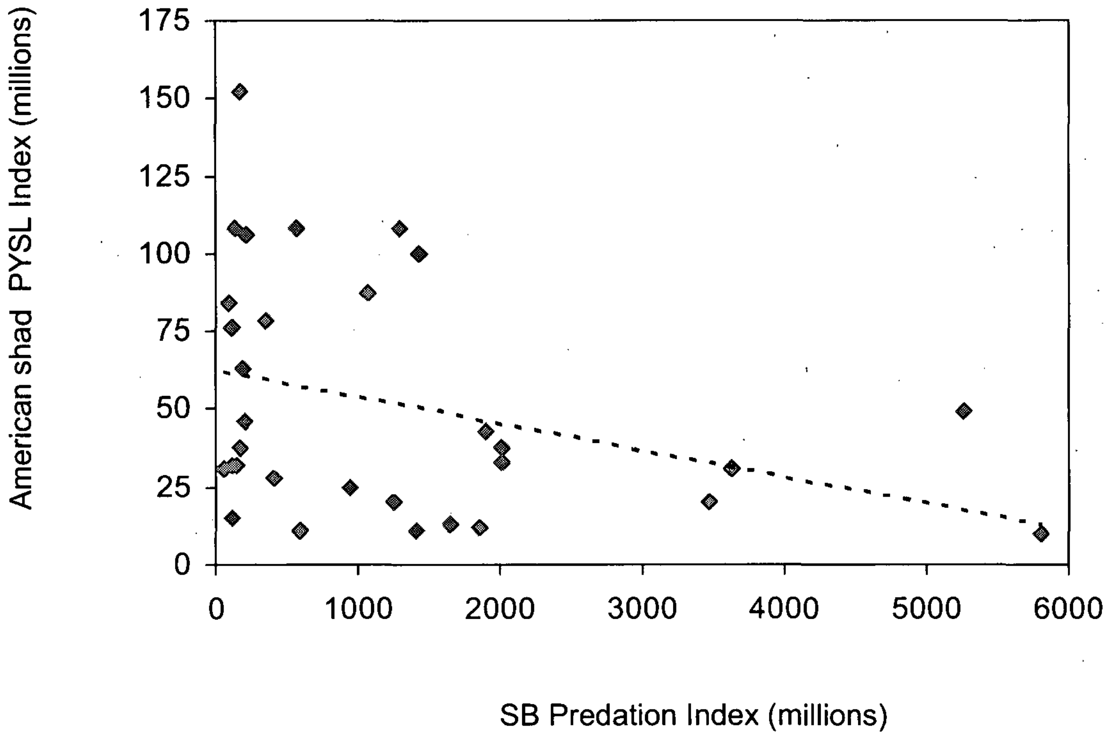


Figure 21b. Relationship between American shad YOY abundance and the striped bass predation index.

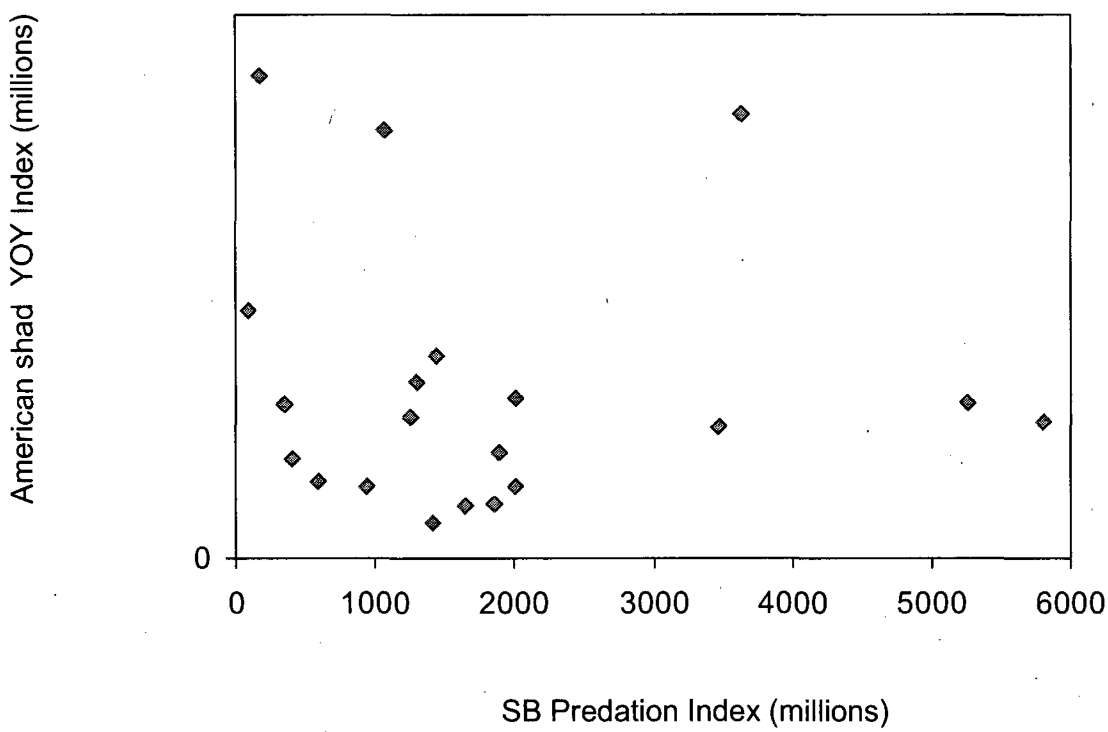


Figure 22. Long-term trends in American shad PYSL abundance and in the striped bass predation index.

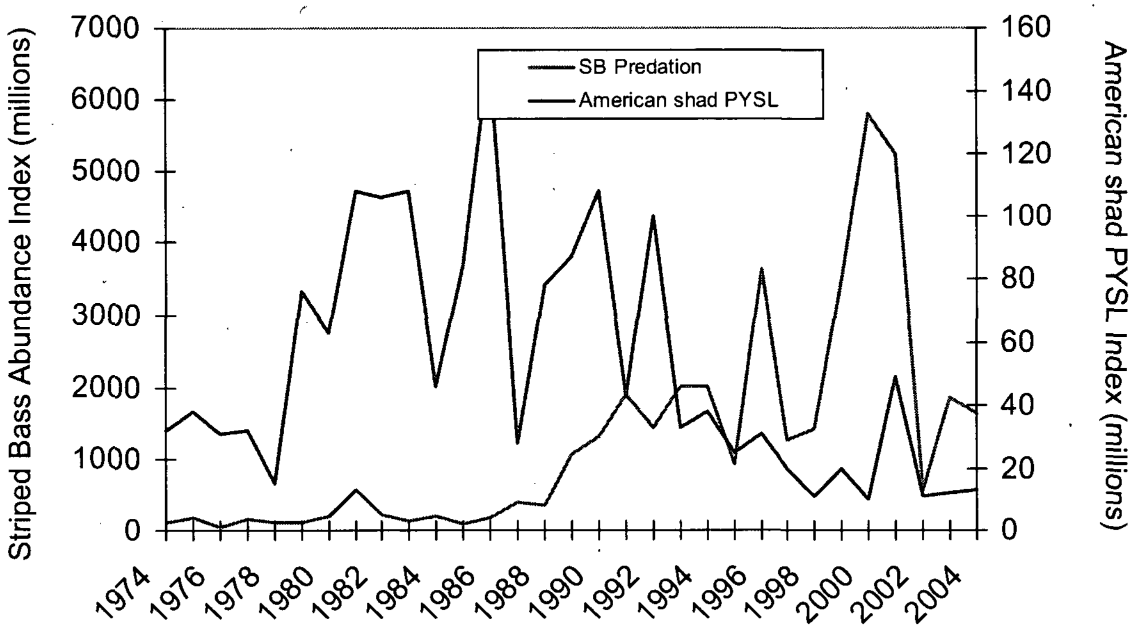


Figure 23. Long-term trends in the abundance of Atlantic tomcod in the Hudson River.

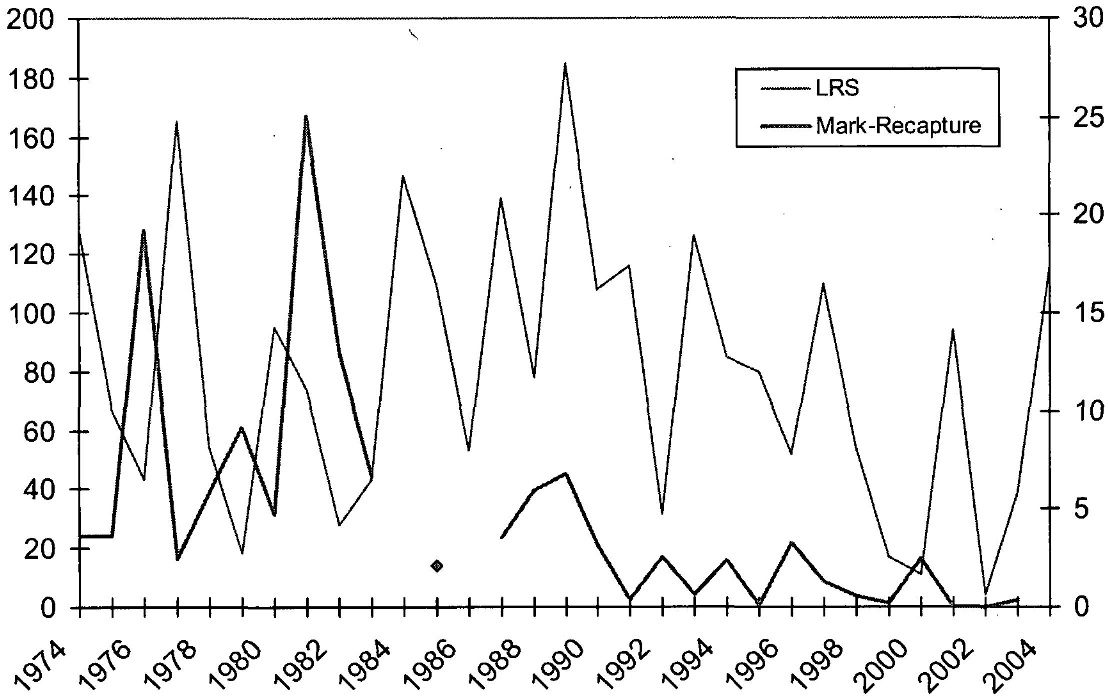


Figure 24a. Relationship between Atlantic tomcod egg deposition and resulting age 1 abundance.

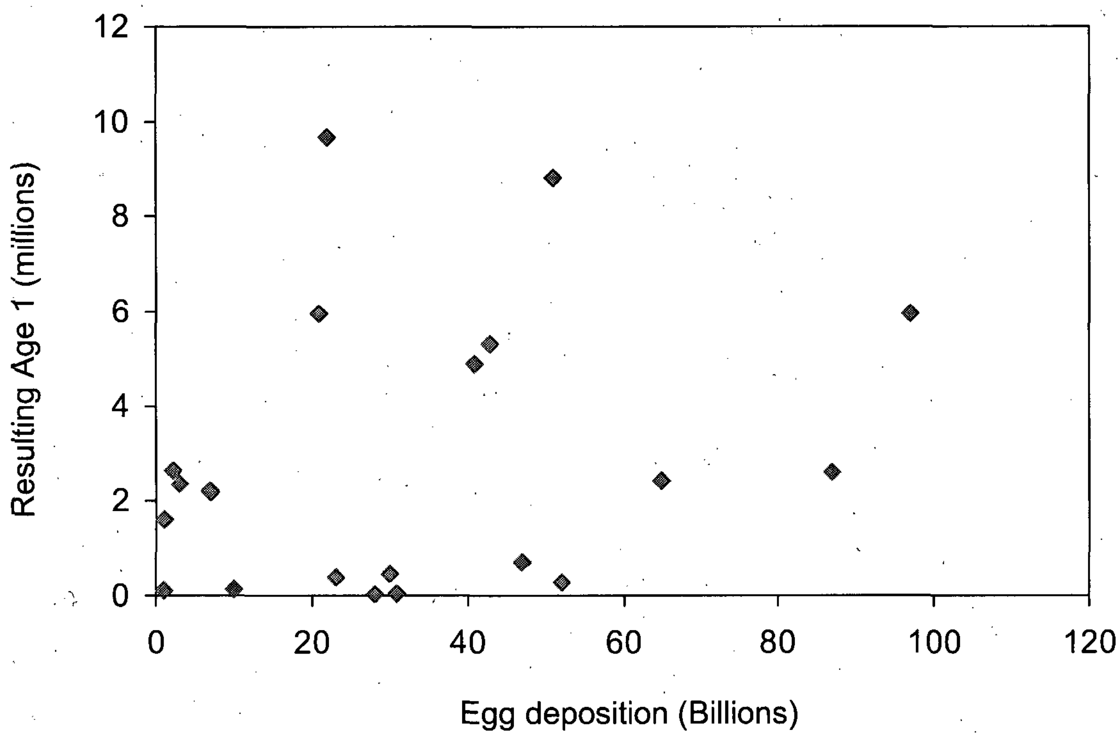


Figure 24b. Relationship between Atlantic tomcod egg to age 1 survival and age 1 abundance.

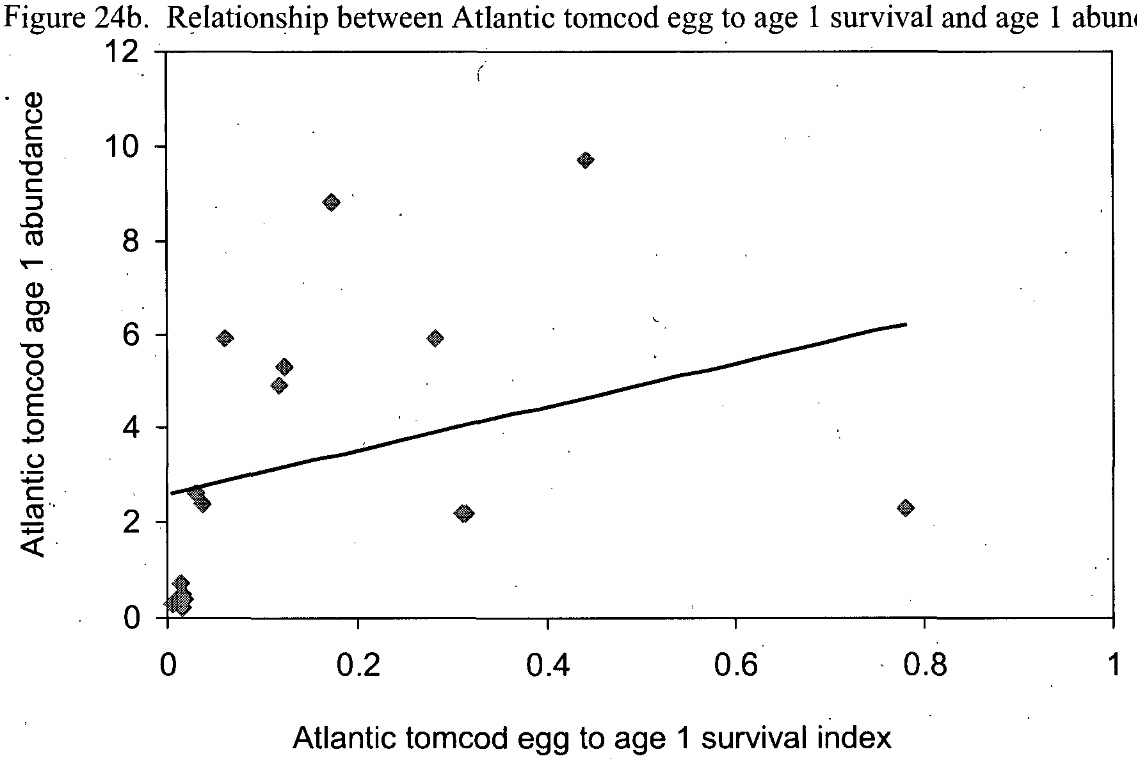


Figure 25a. Relationship between IP2 and IP3 CMR and Atlantic tomcod egg to age 1 survival.

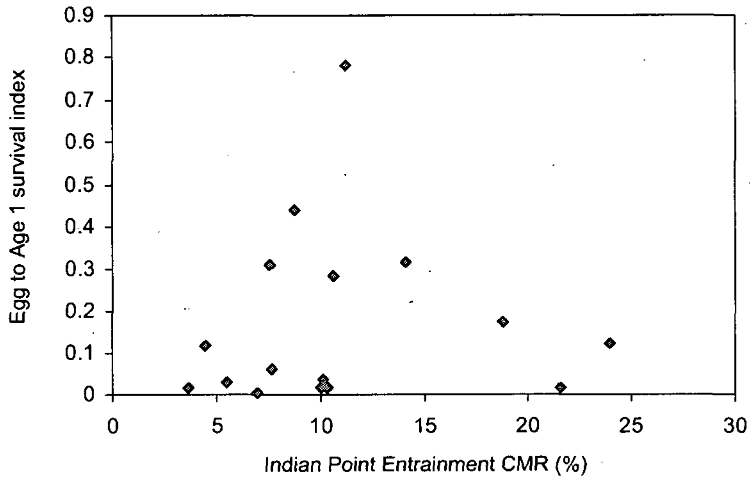


Figure 25b. Relationship between IP2 and IP3 CMR and Atlantic tomcod LRS index.

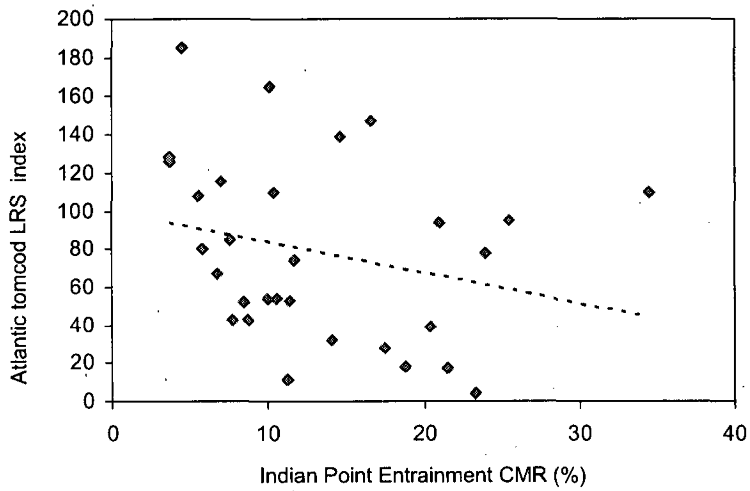


Figure 25c. Relationship between IP2 and IP3 CMR and Atlantic tomcod mark-recapture index

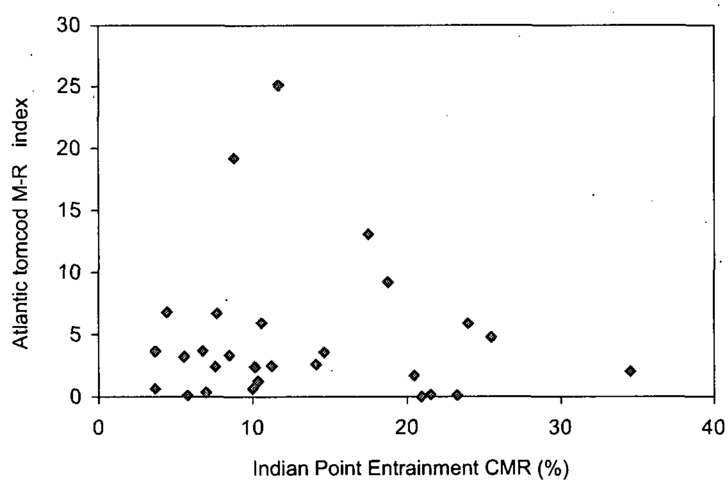
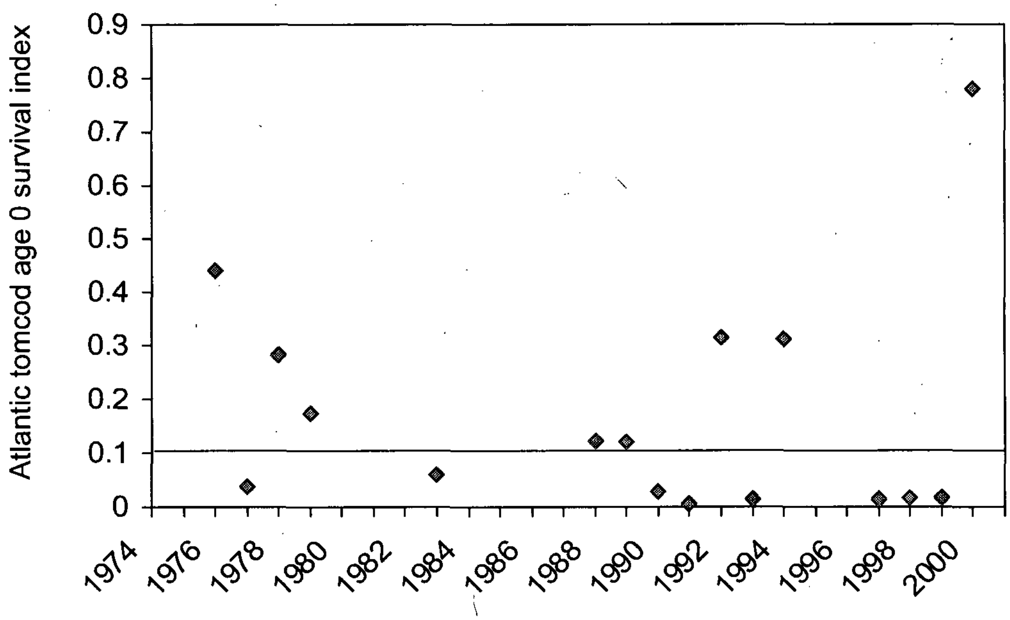


Figure 26. (a) Atlantic tomcod age 0 survival during years in which 1 unit (blue) and 2 units (red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of Atlantic tomcod are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between combined IP2 and IP3 May-June withdrawals and Atlantic tomcod egg to age 1 survival.

(a)



(b)

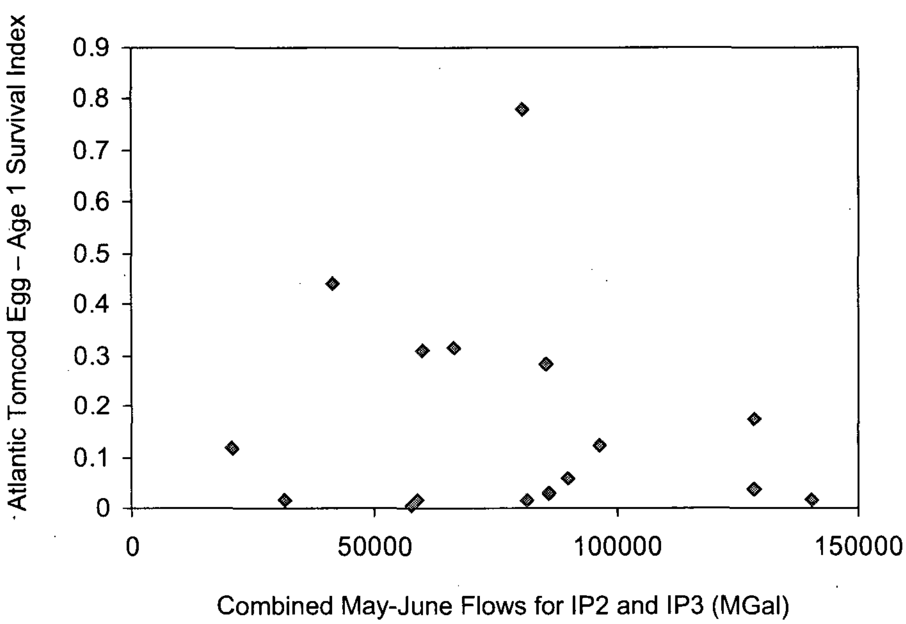


Figure 27. Comparison of long-term trends in the PWW degree-day index to long-term trends in the abundance of age 1 and age 2 Atlantic tomcod.

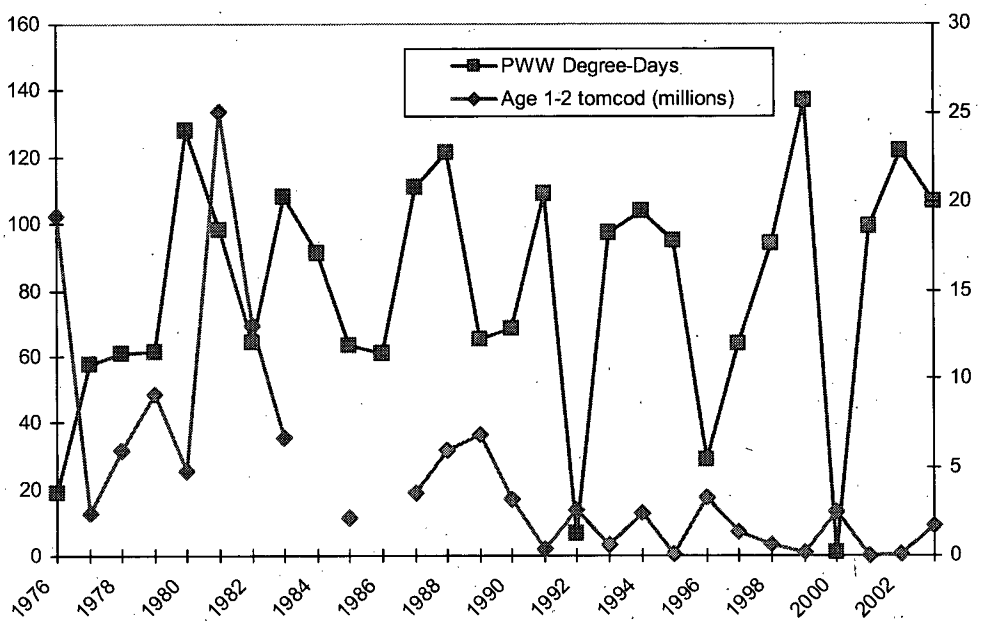


Figure 28a. Relationship between the striped bass predation index and the Atlantic tomcod LRS index.

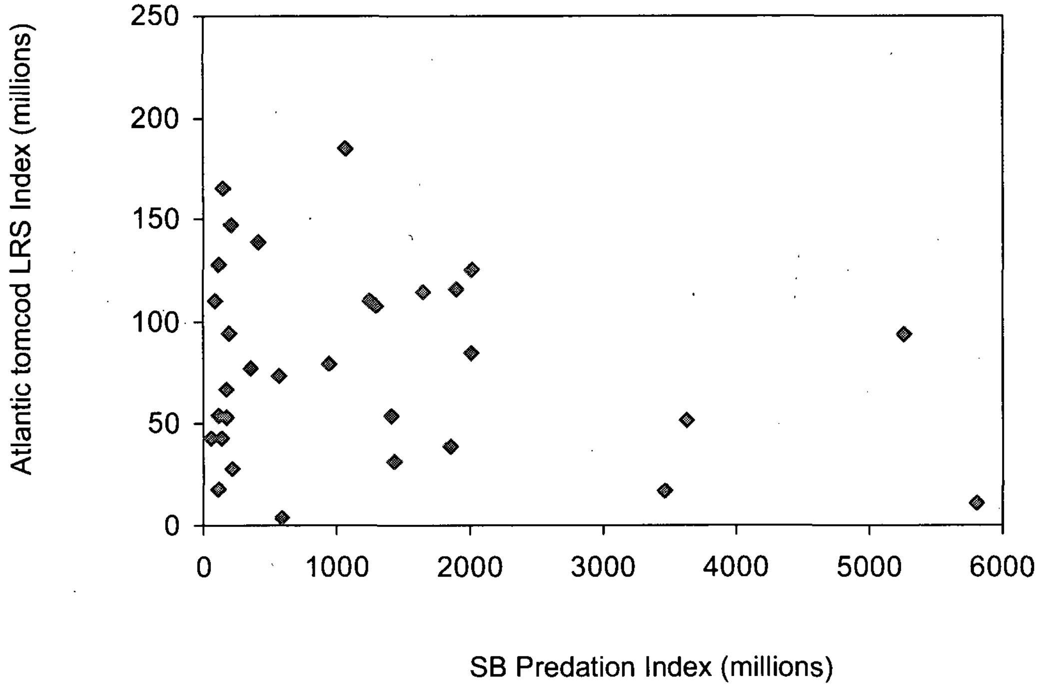


Figure 28b. Relationship between the striped bass predation index and the Atlantic tomcod mark-recapture index.

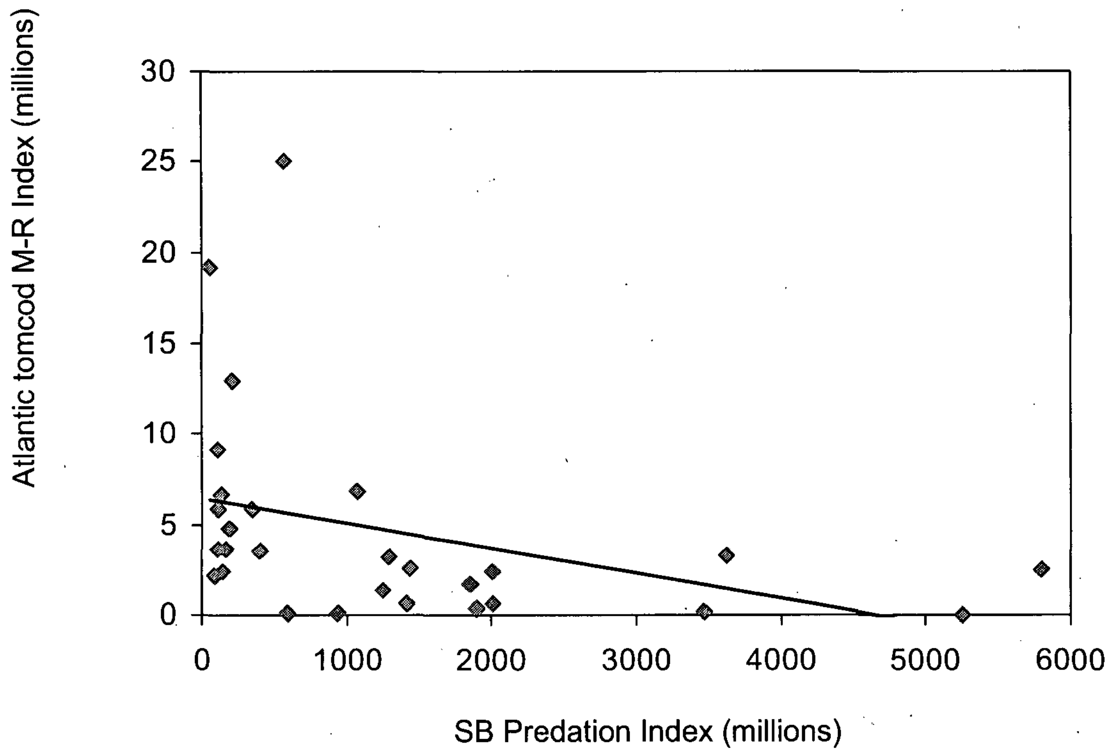


Figure 29a. Long-term trends in the Atlantic tomcod LRS index and the striped bass predation index.

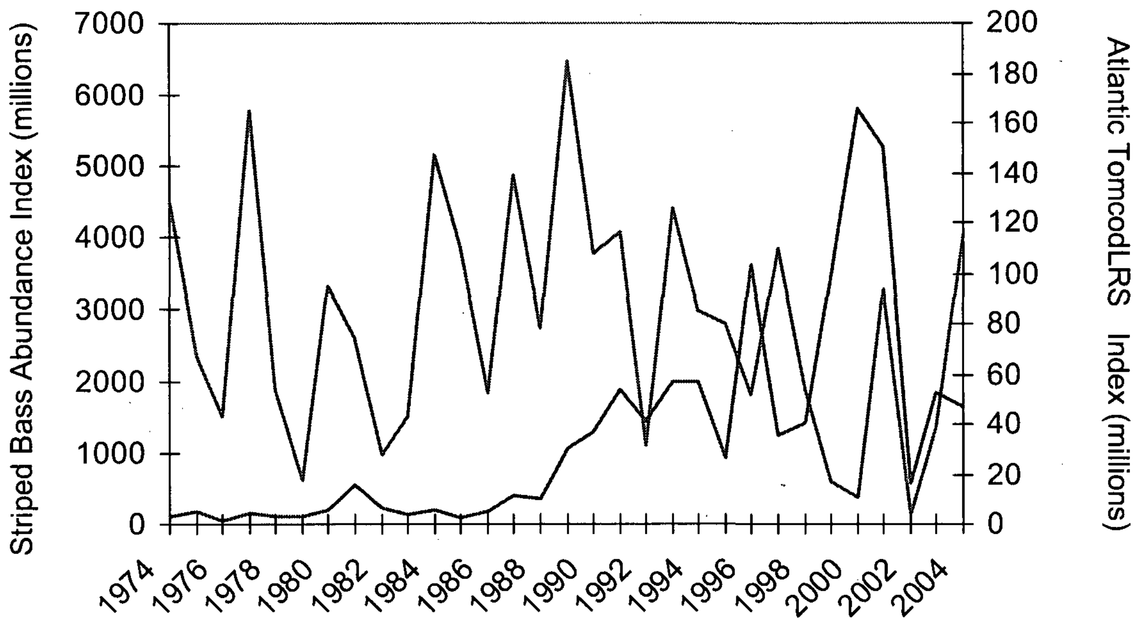


Figure 29b. Long-term trends in the Atlantic tomcod mark-recapture index and the striped bass predation index.

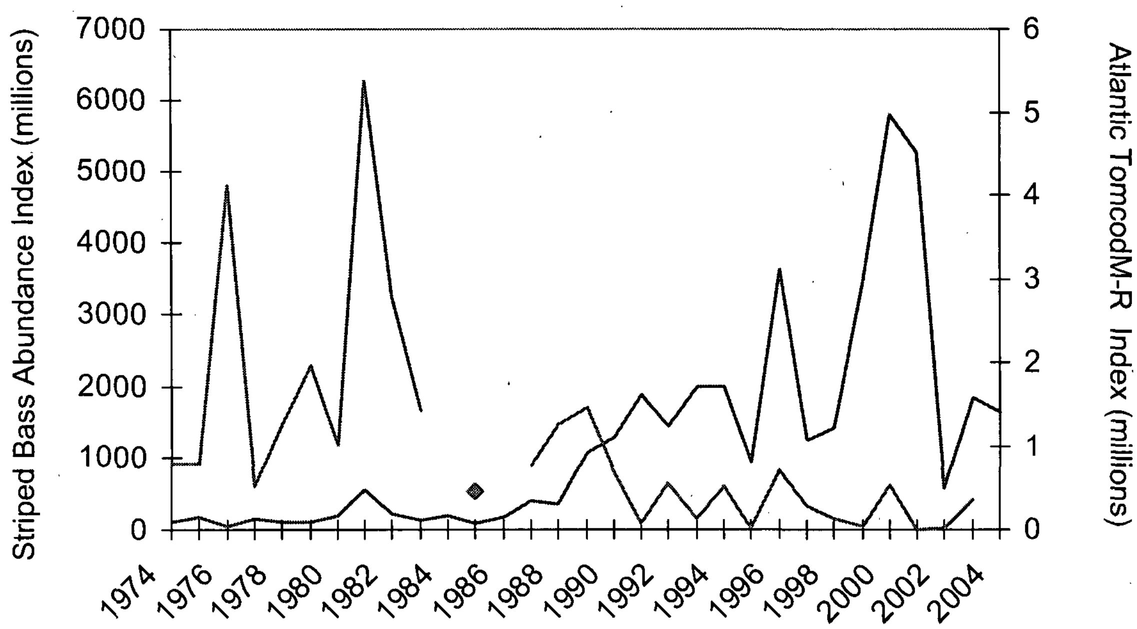


Figure 30a. Long-term trend in abundance of river herring PYSL in the Hudson River.

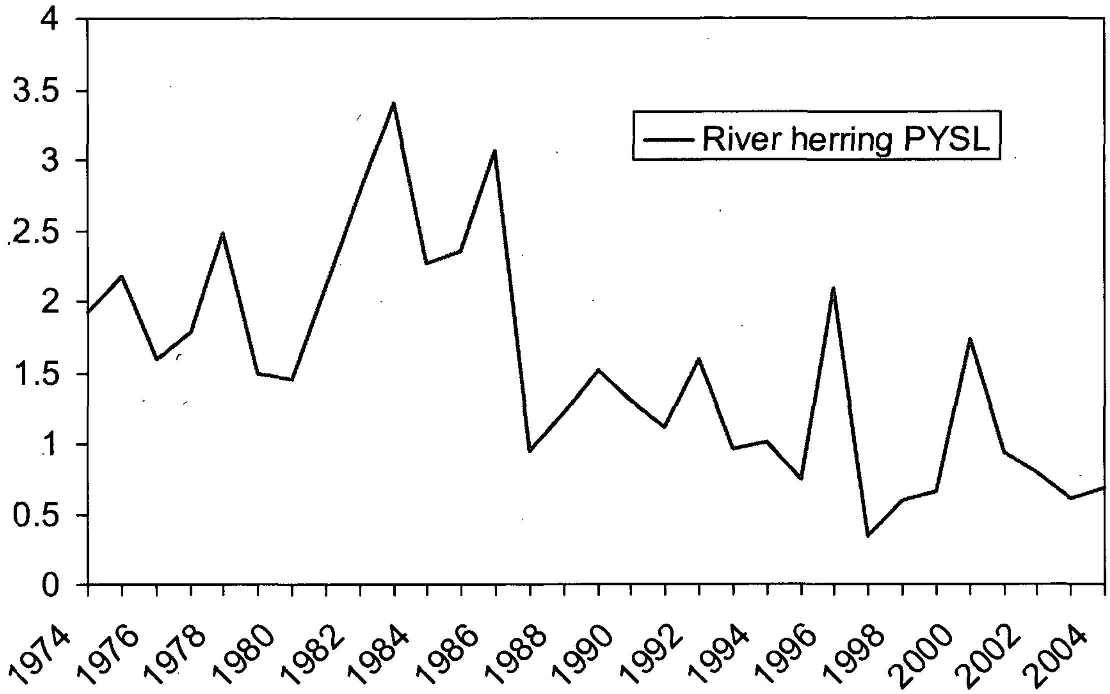


Figure 30b. Long-term trends in abundance of alewife and blueback herring YOY in the Hudson River.

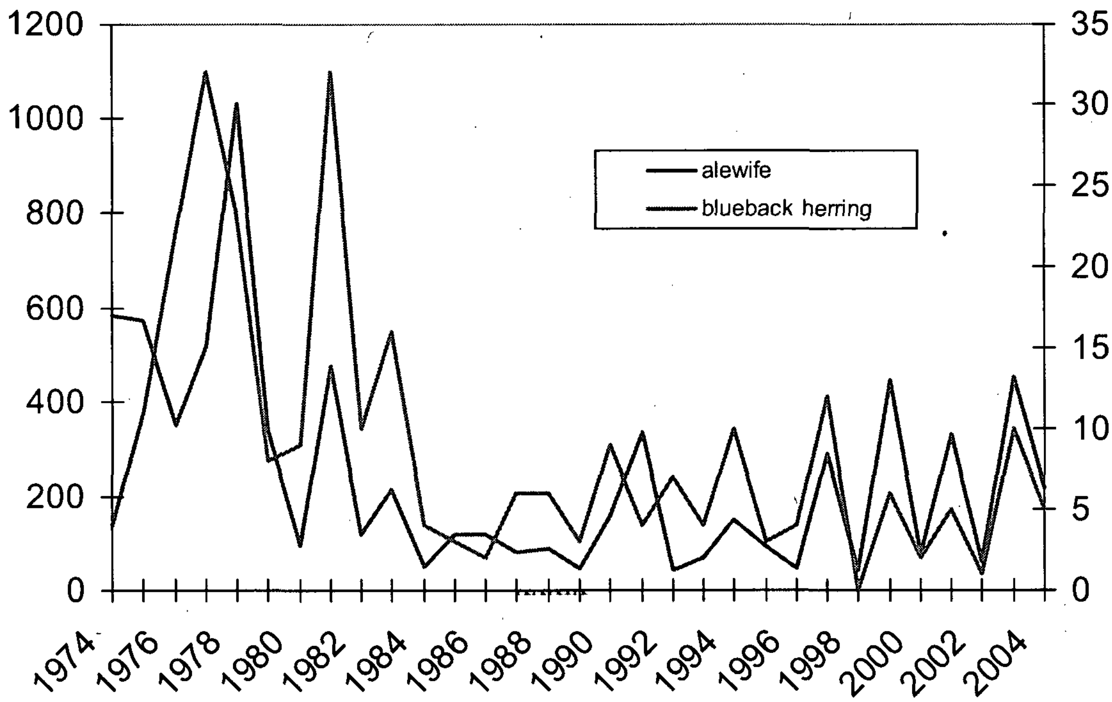


Figure 31a. Relationship between the IP2 and IP3 CMR and river herring PYSL survival.

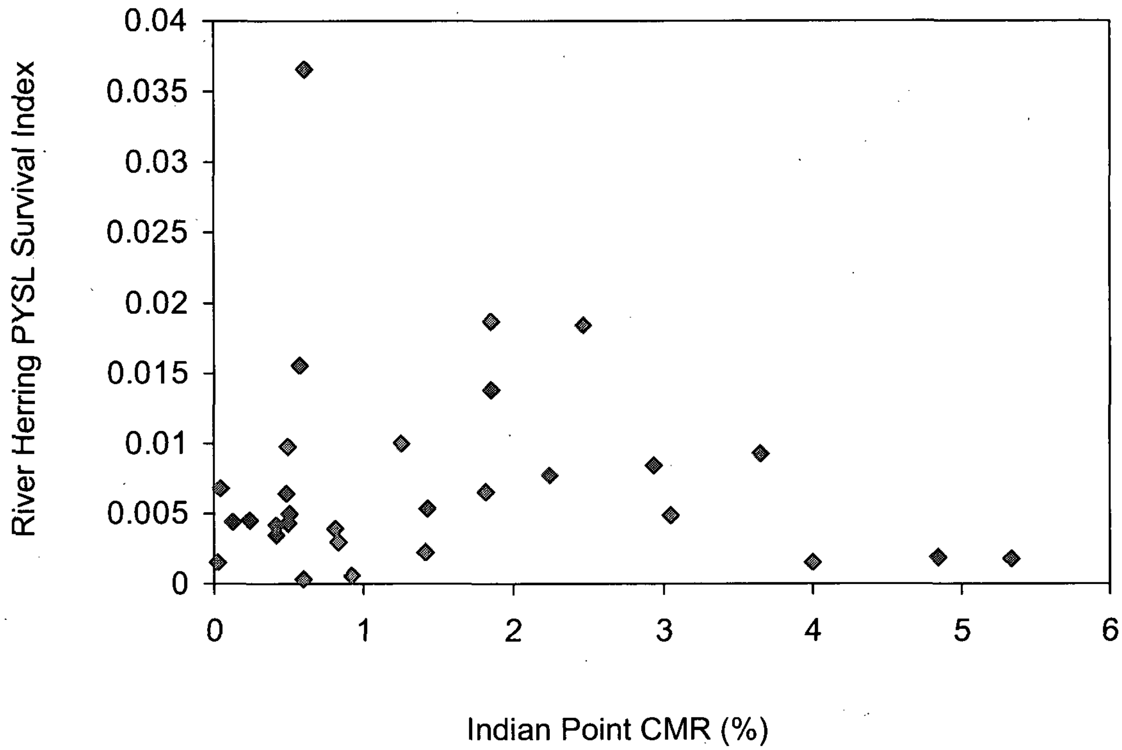


Figure 31b. Relationship between the IP2 and IP3 CMR and river herring PYSL abundance.

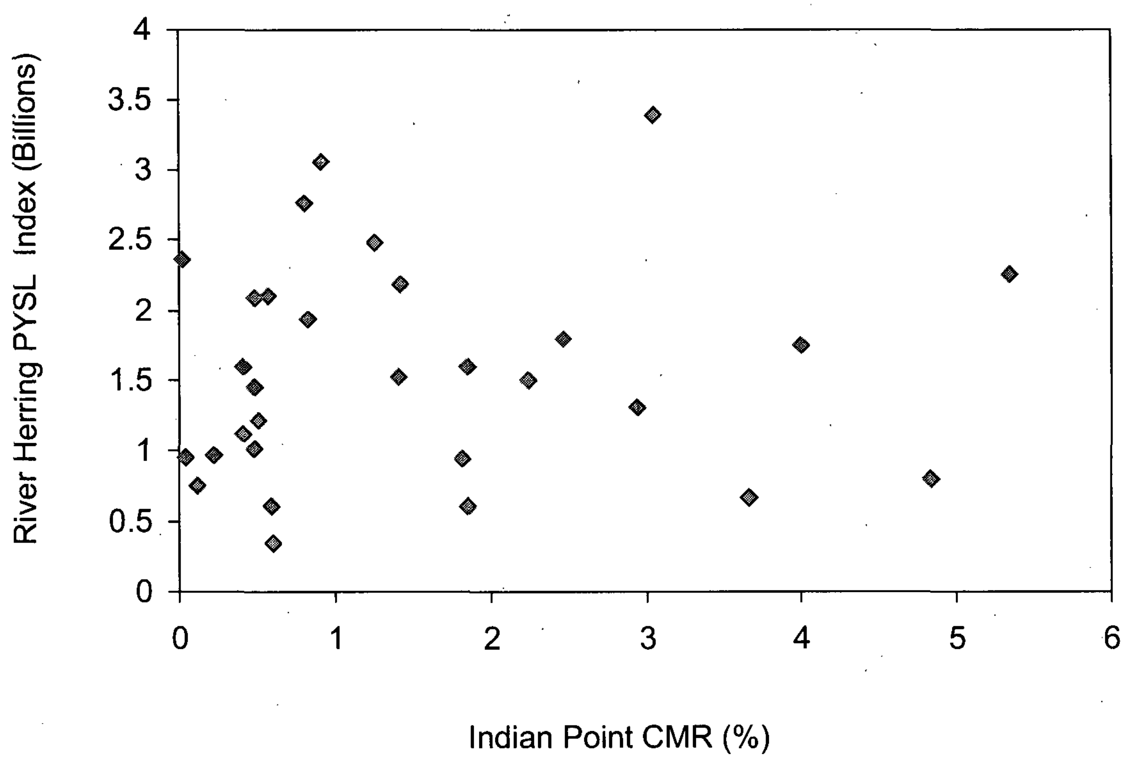


Figure 32a. Relationship between the IP2 and IP3 CMR and alewife YOY abundance.

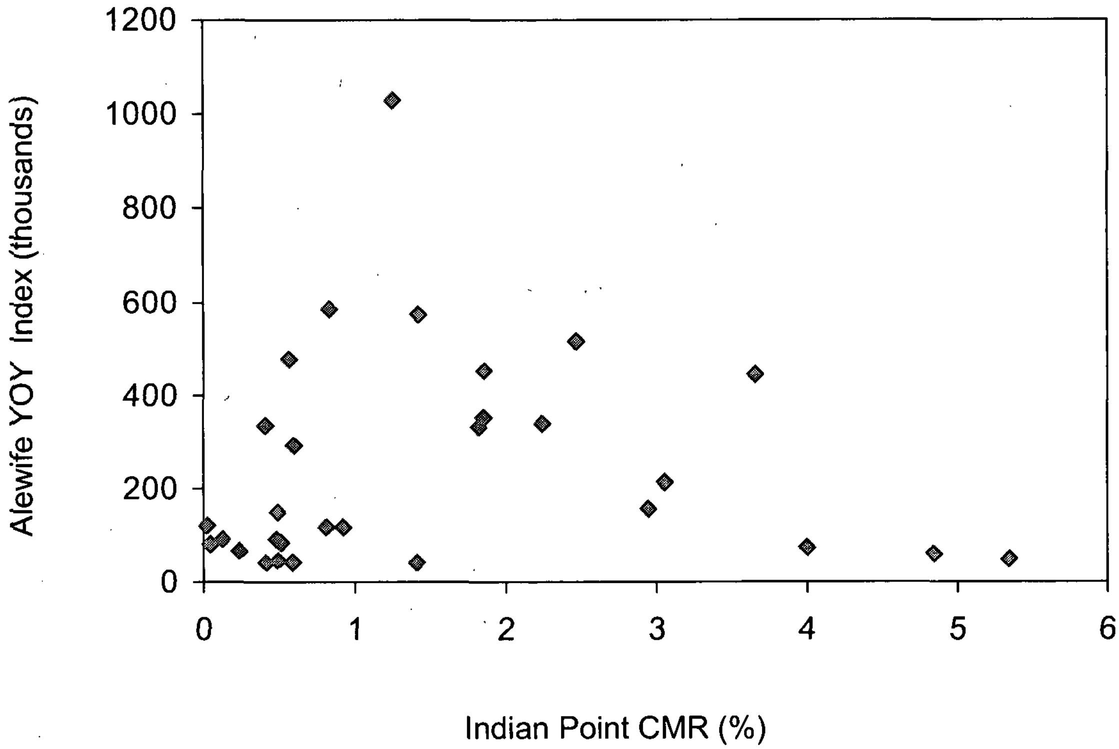


Figure32b. Relationship between the IP2 and IP3 CMR and blueback herring YOY abundance.

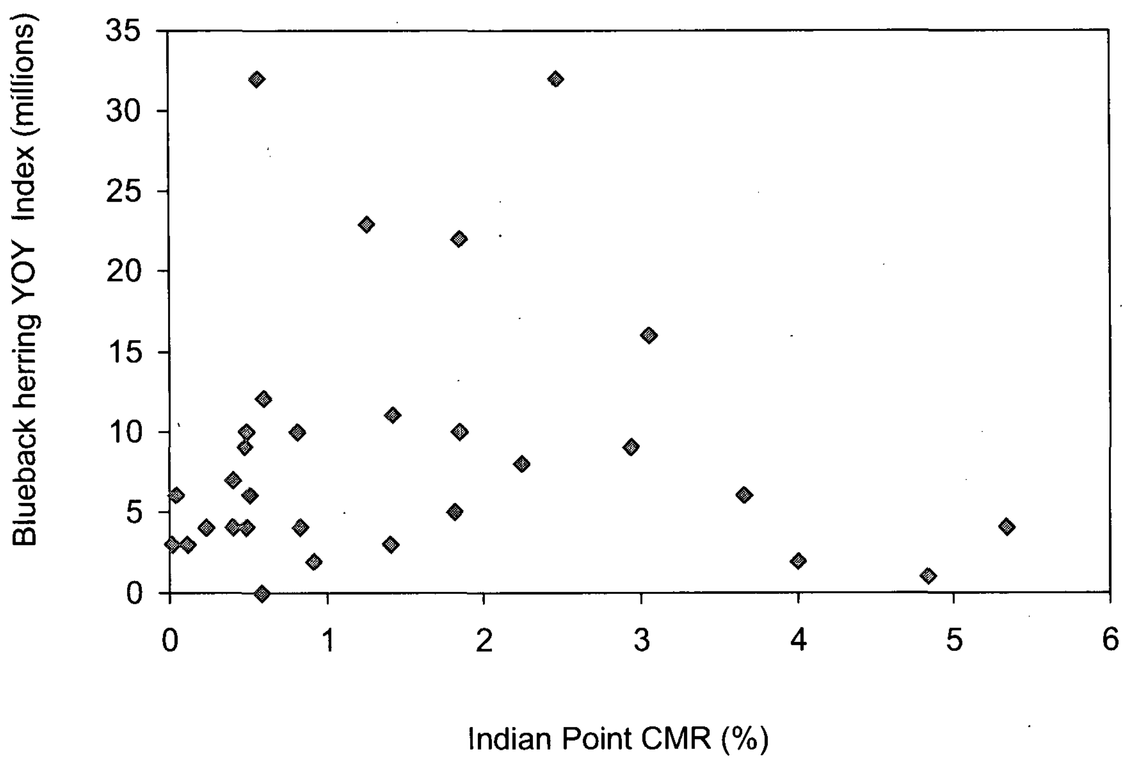
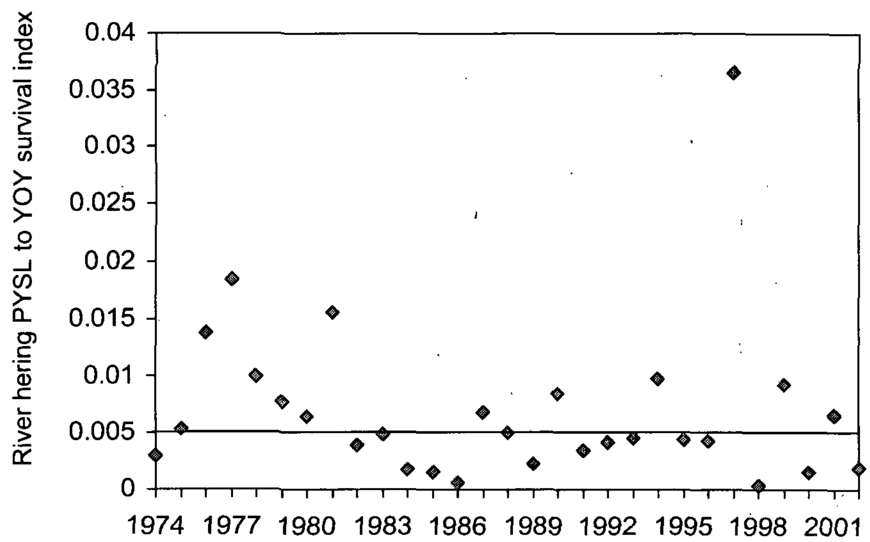


Figure 33. (a) River herring (alewife and blueback herring) PYSL to YOY survival during years in which 1 unit (blue) and 2 units (red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of river herring are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between IP2 and IP3 May-June water withdrawals and river herring PYSL survival.

(a)



(b)

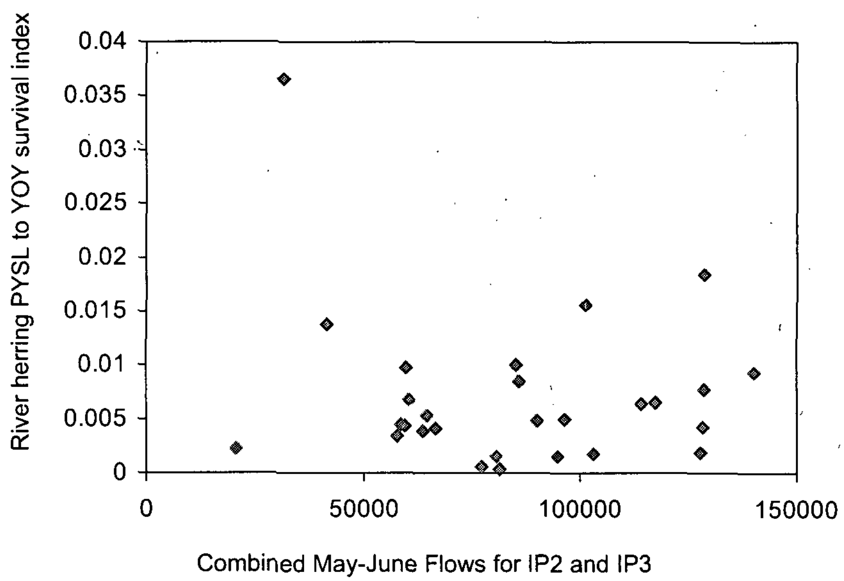


Figure 34a. Relationship between the striped bass predation index and river herring PYSL abundance.

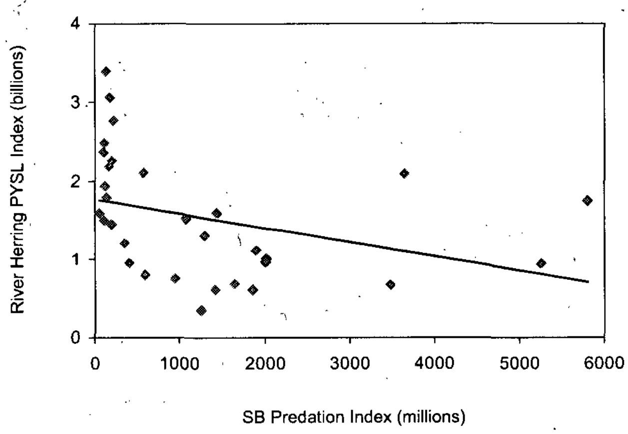


Figure 34b. Relationship between the striped bass predation index and alewife YOY abundance.

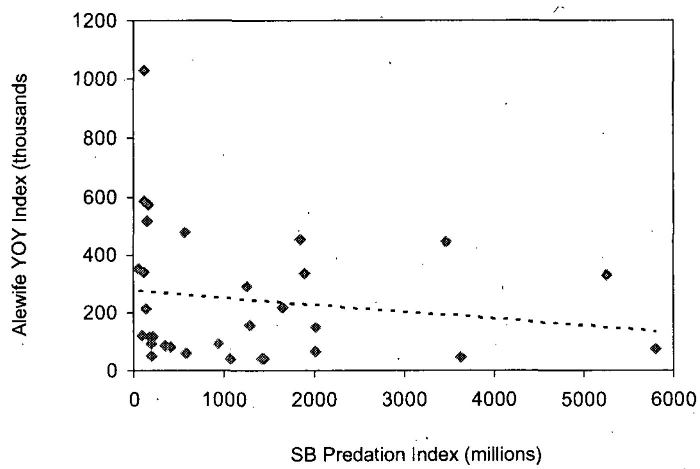


Figure 34c. Relationship between the striped bass predation index and blueback herring YOY abundance.

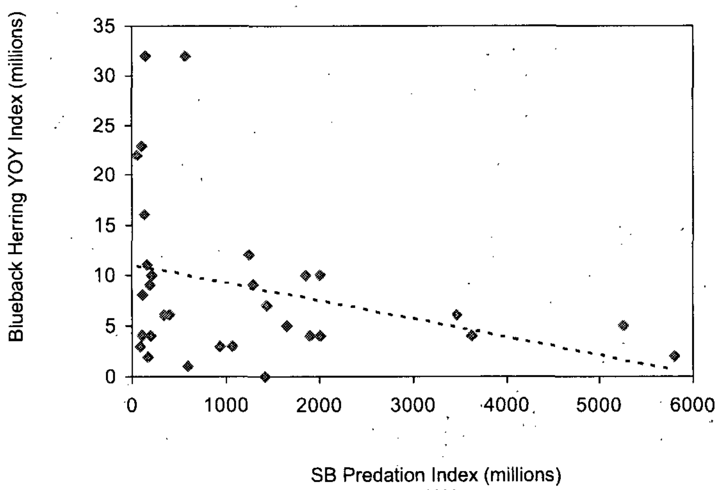


Figure 35a. Long-term trends in river herring PYSL abundance and in the striped bass predation index.

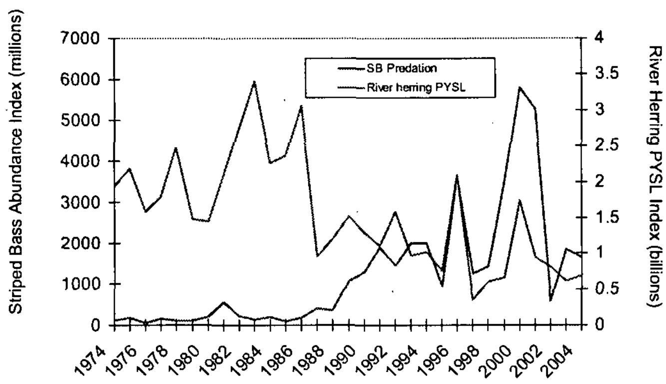


Figure 35b. Long-term trends in alewife YOY abundance and in the striped bass predation index.

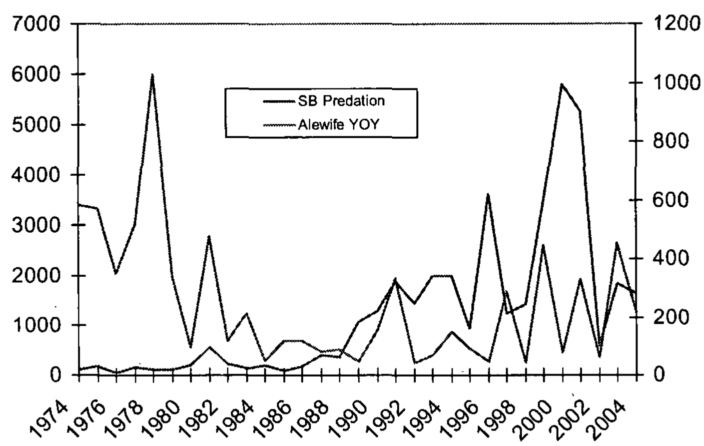


Figure 35c. Long-term trends in blueback herring YOY abundance and in the striped bass predation index.

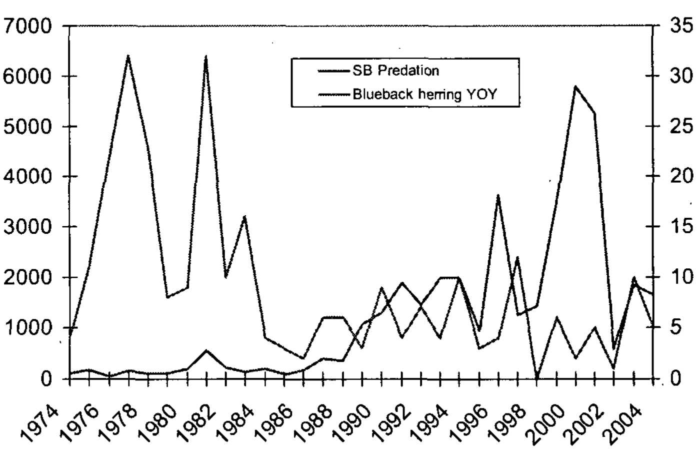


Figure 36. Long-term trends in abundance of bay anchovy PYSL and YOY.

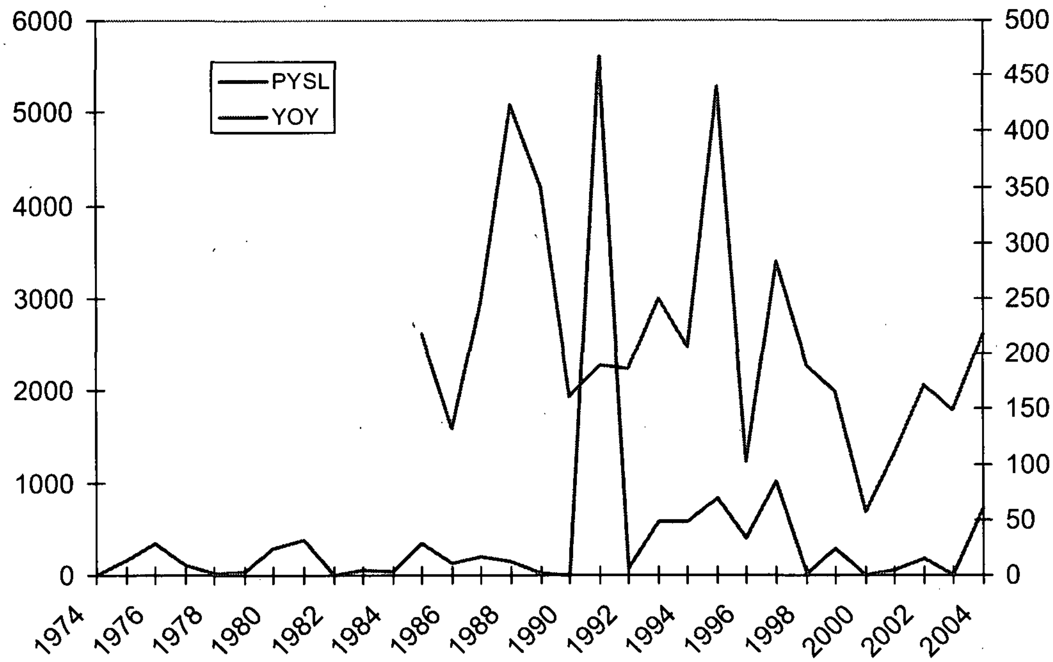


Figure 37a. Relationship between the IP2 and IP3 CMR and bay anchovy PYSL to YOY survival.

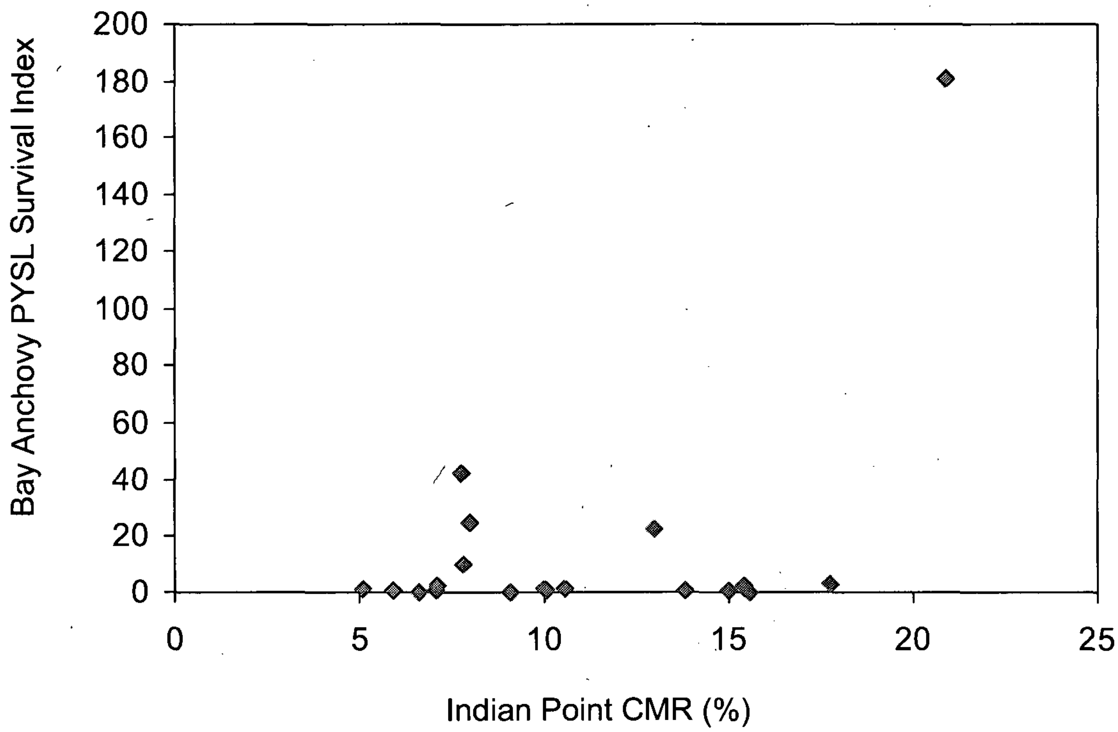


Figure 37b. Relationship between the IP2 and IP3 CMR and bay anchovy PYSL abundance.

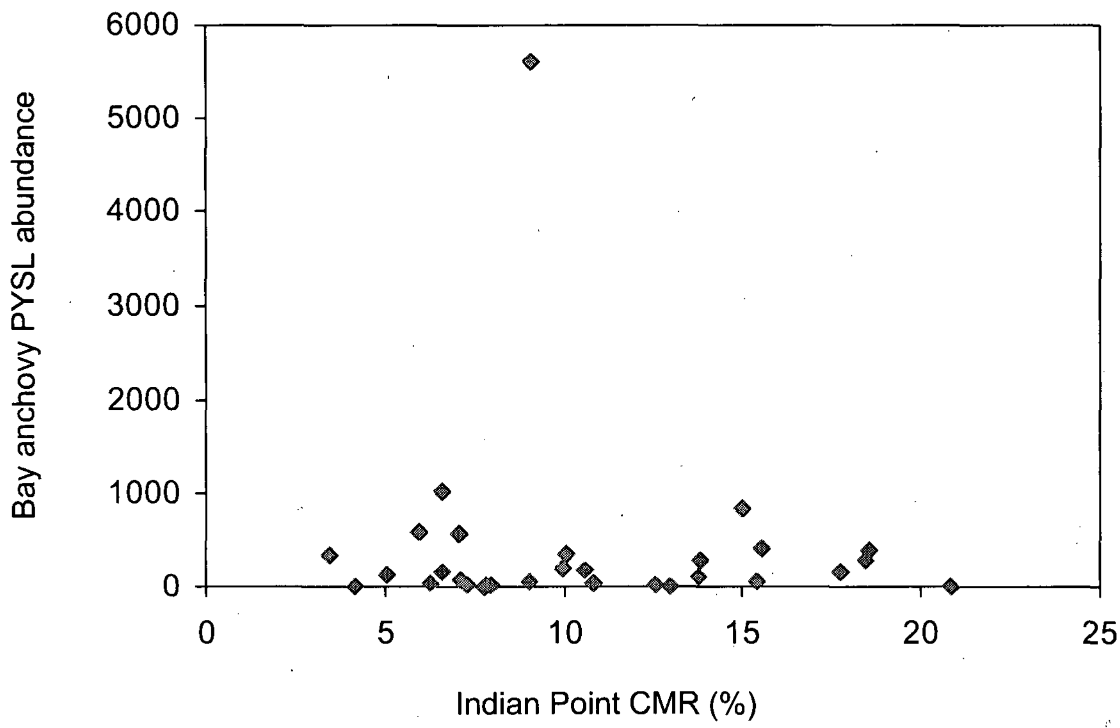
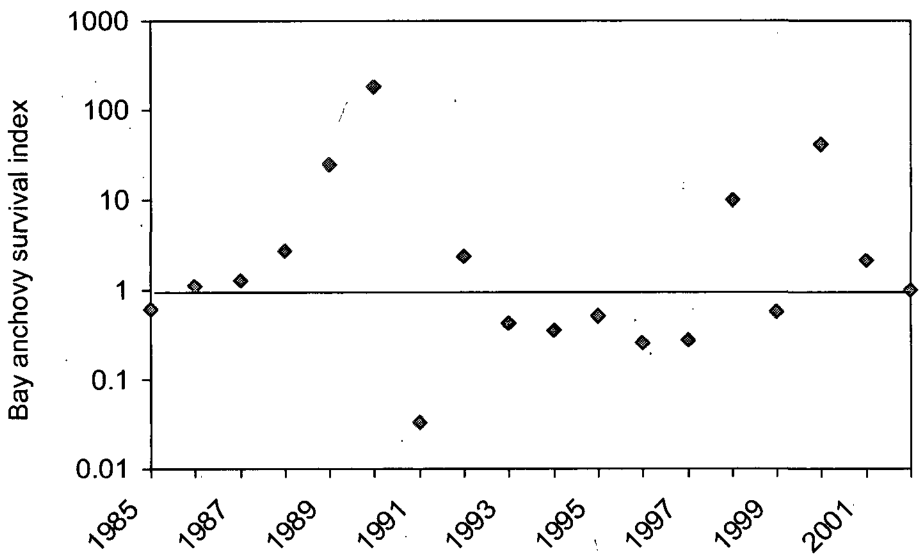


Figure 38. (a) Bay anchovy PYSL to YOY survival during years in which 1 unit (blue) and 2 units (red) at Indian Point were operating during May and June, the peak months during which entrainable life stages of river herring are present in the Hudson River. The horizontal line shows the median survival index value for the time series. (b) Relationship between total IP2 and IP3 June-July withdrawals and bay anchovy PYSL survival.

(a)



(b)

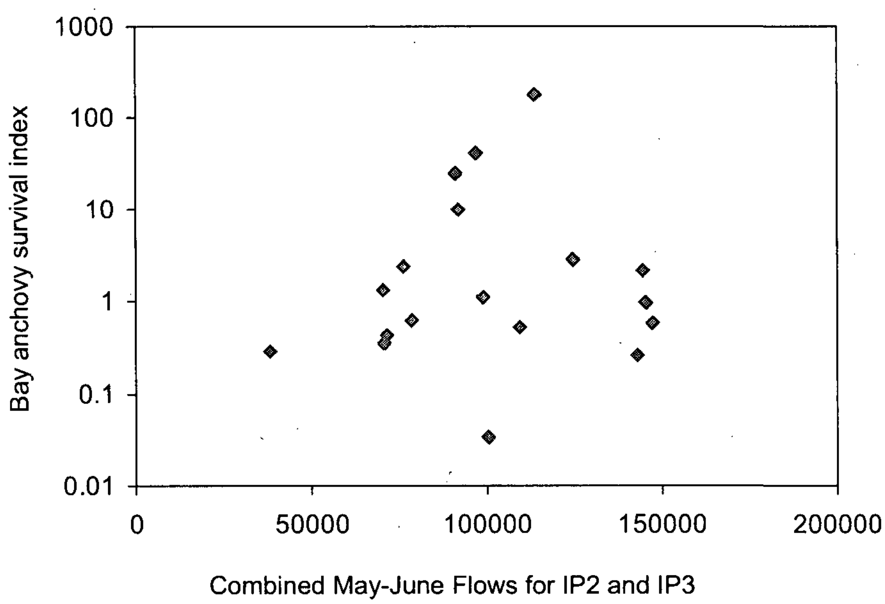


Figure 39a. Relationship between bay anchovy YOY abundance and the striped bass predation index.

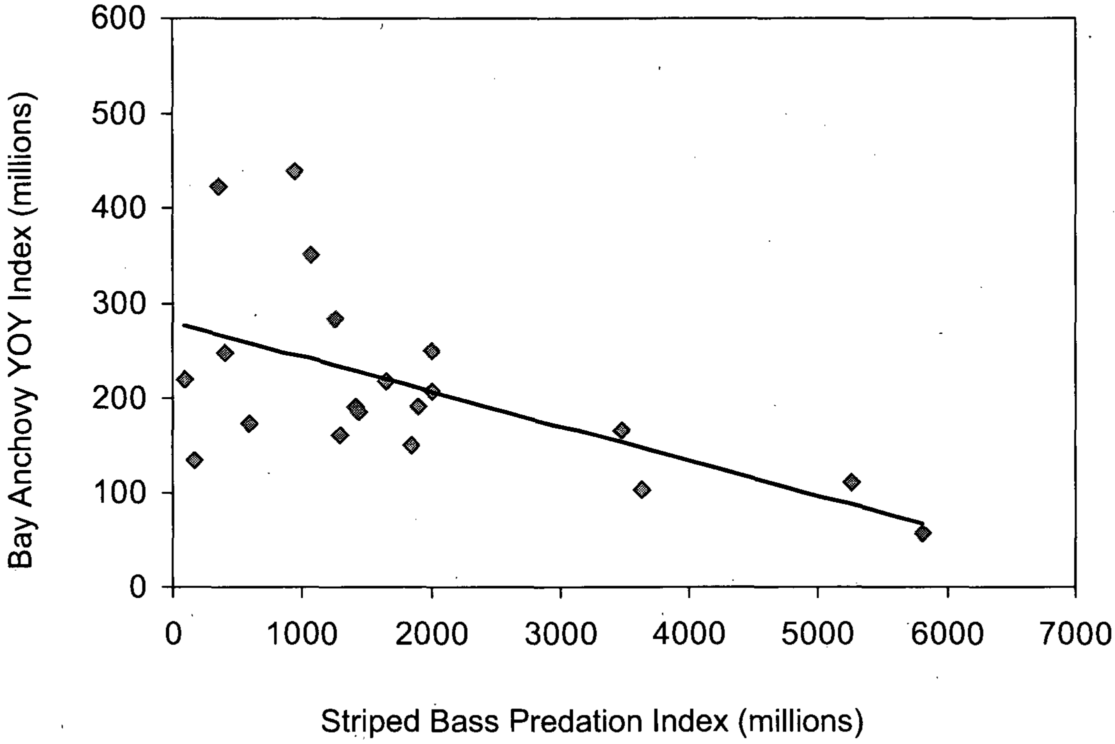


Figure 39b. Long-term trends in bay anchovy YOY abundance and the striped bass predation index.

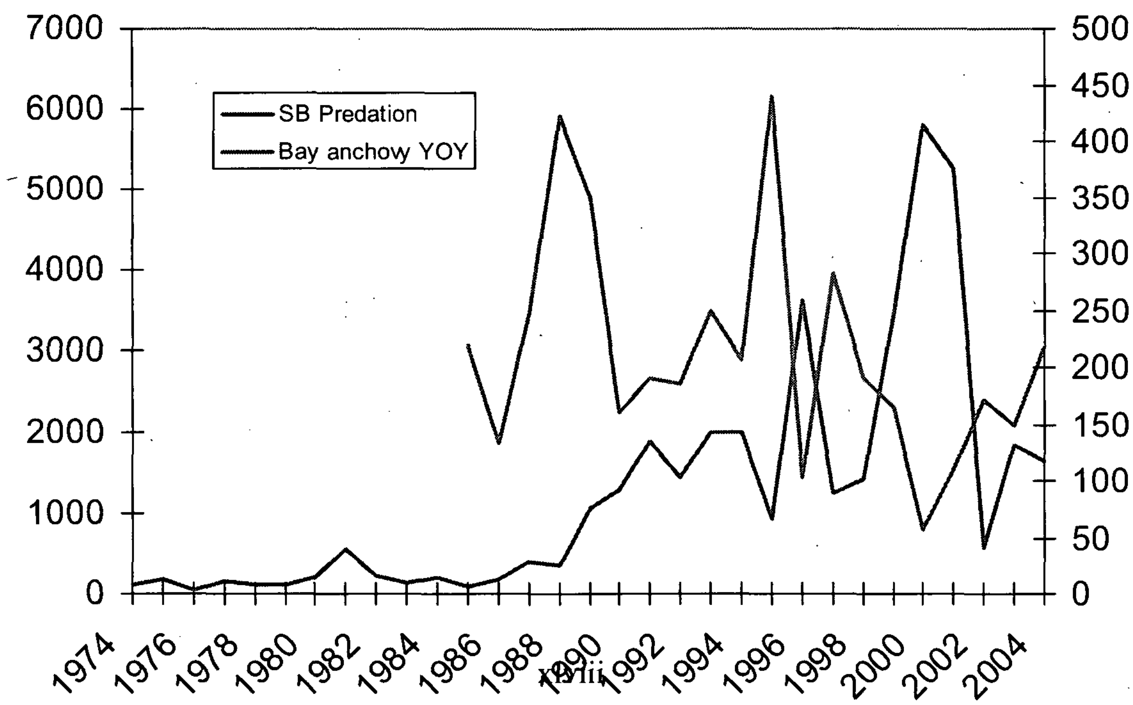


Figure 40. Long-term trends in the abundance of spottail shiner eggs and YOY.

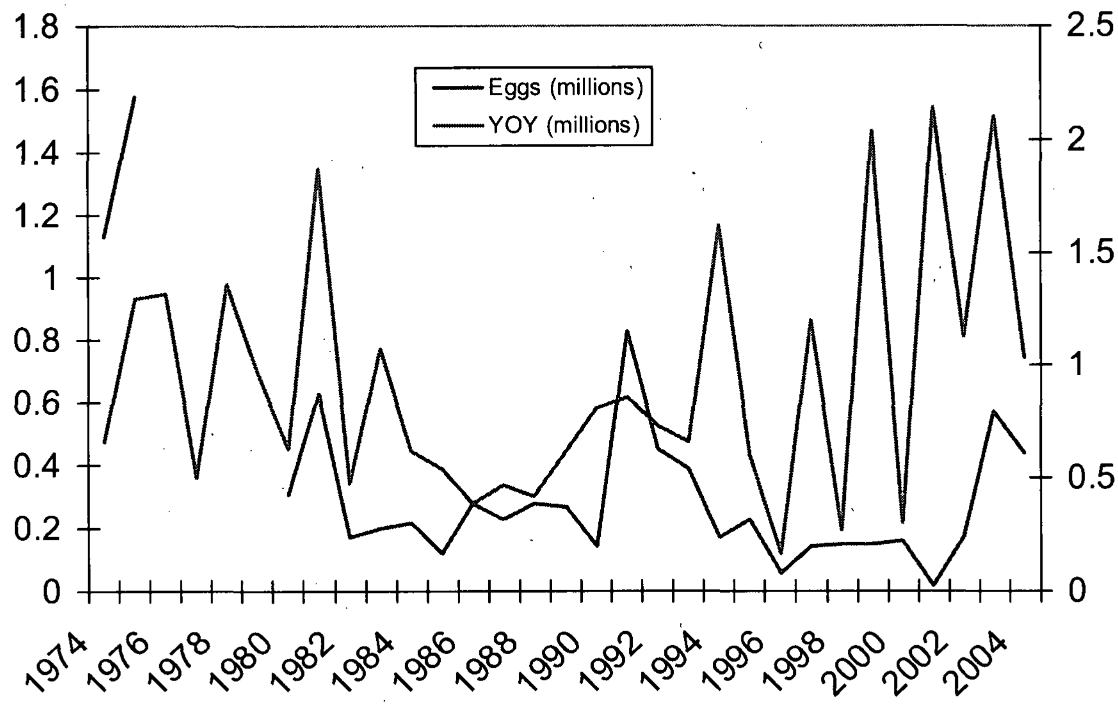


Figure 41a. Relationship between the IP2 and IP3 CMR and spottail shiner egg to YOY survival.

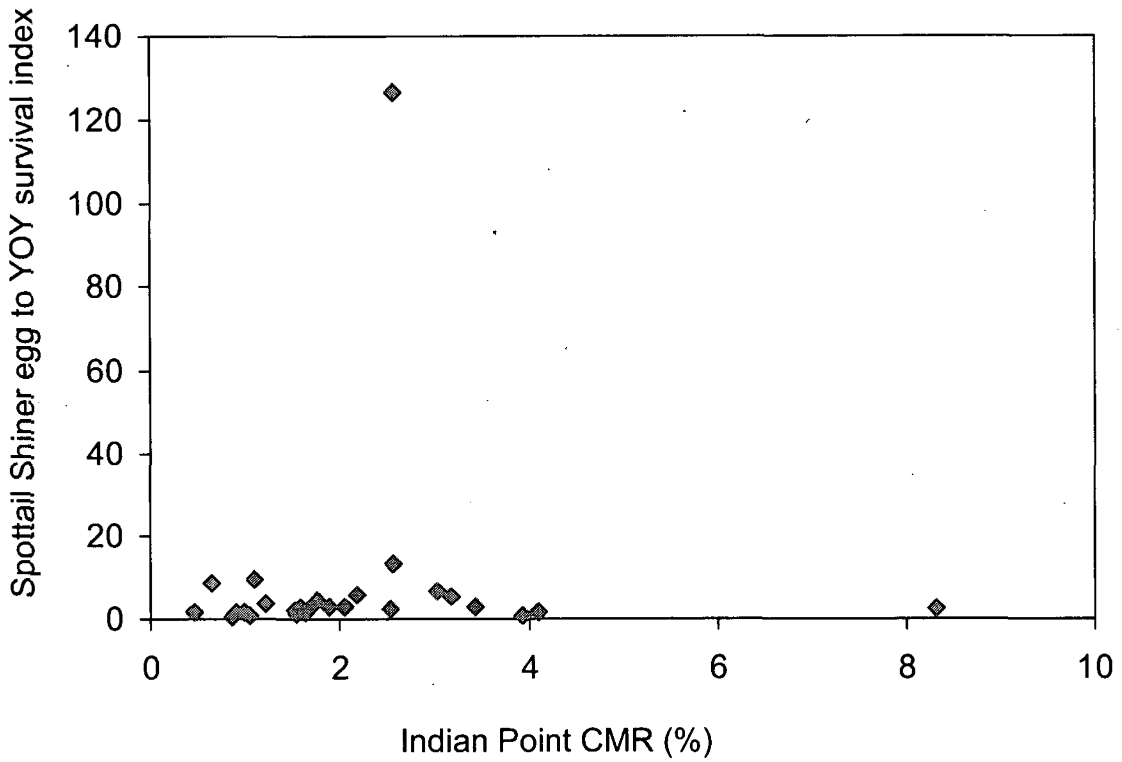


Figure 41b. Relationship between the IP2 and IP3 CMR and spottail shiner YOY abundance.

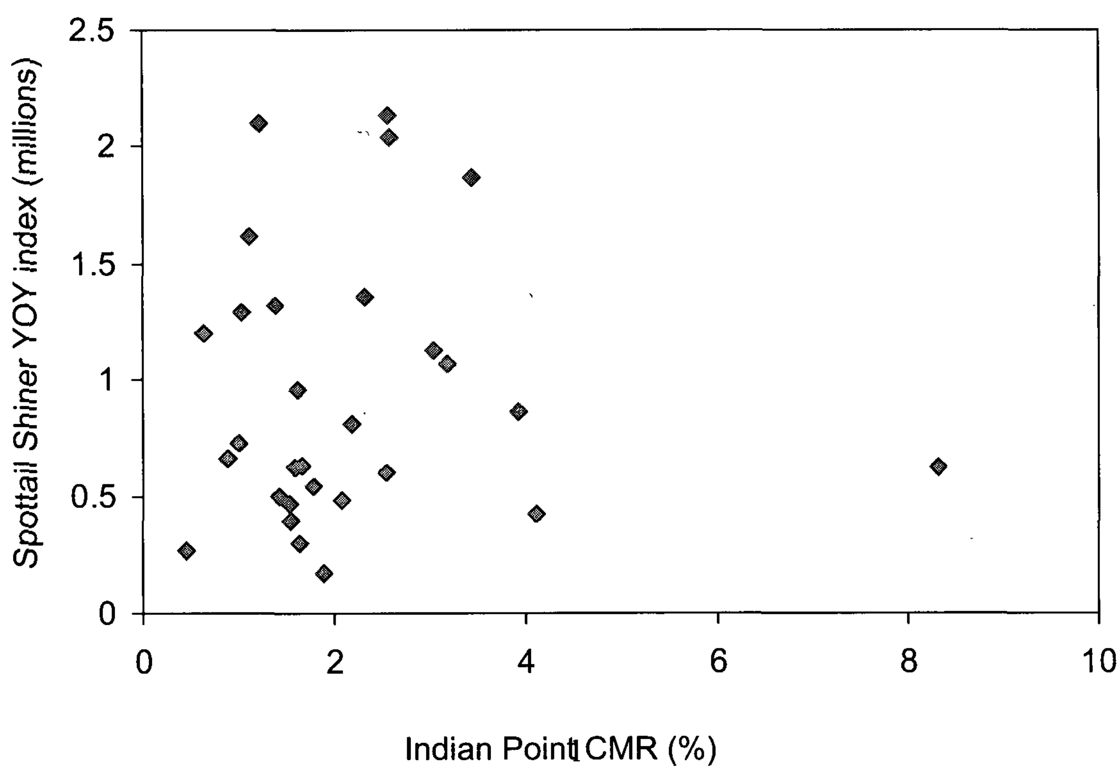


Figure 42. Relative influence of IP2 and IP3 vs. fishing on the spawning potential of Hudson River striped bass.

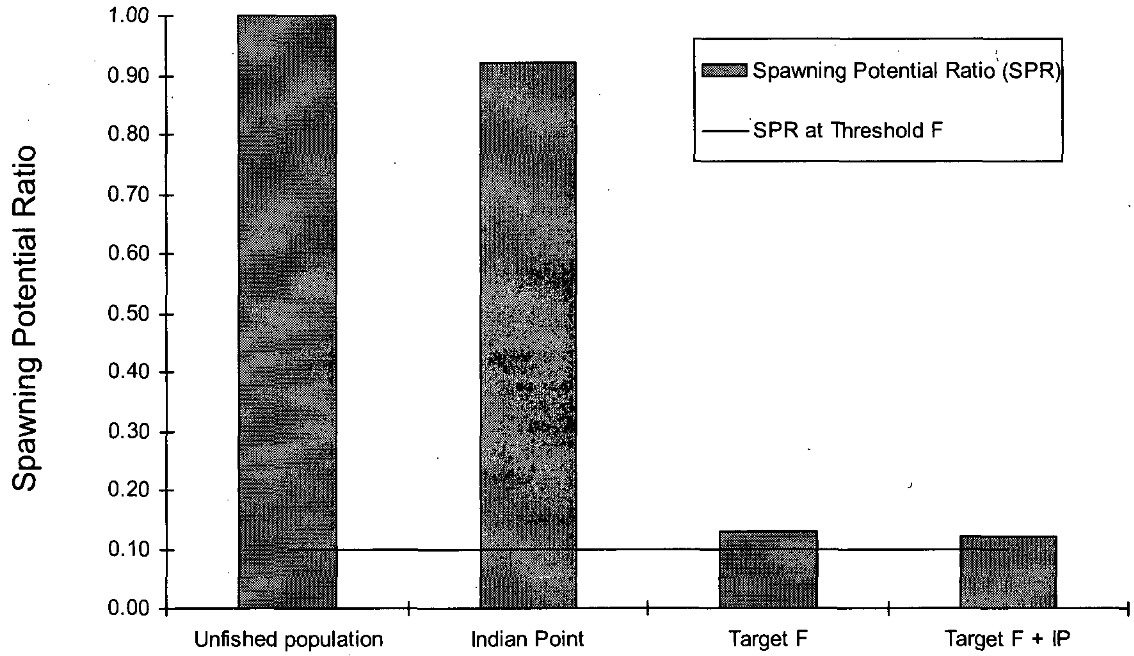


Figure 43. Comparative effects of Indian Point and fishing on Hudson River American shad SPR using data and modeling method from 2007 American shad stock assessment (ASMFC 2007a).

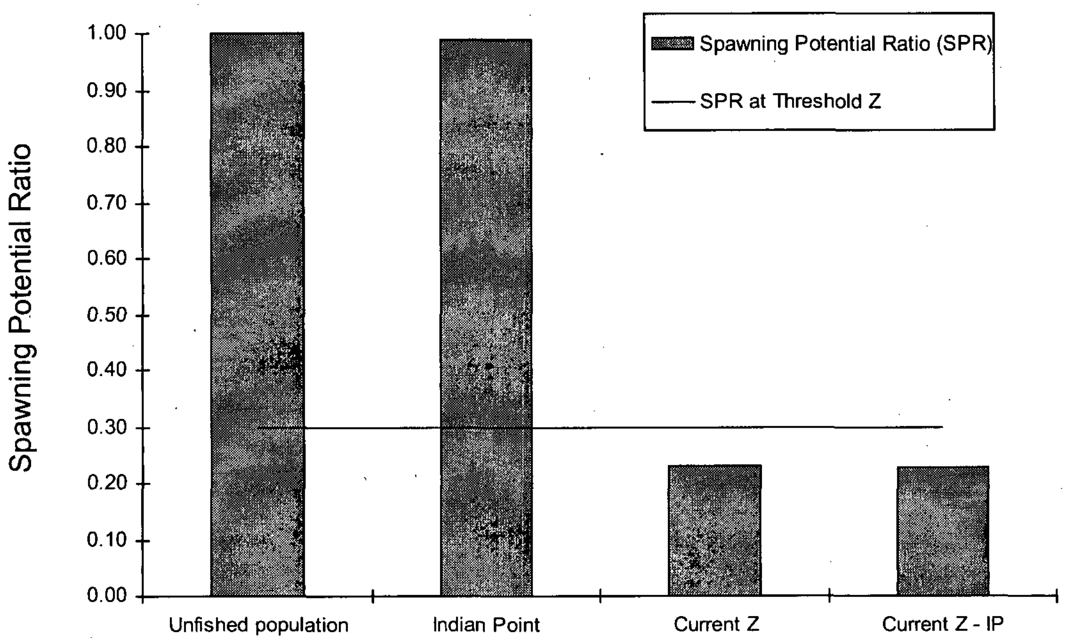


Figure 44a. Relationship between relative change in YOY abundance from Period 1 to Period 2 and entrainment susceptibility for the 21 fish species included in Case A. Zero on the logarithmic Y axis corresponds to no change in abundance from Period 1 to Period 2.

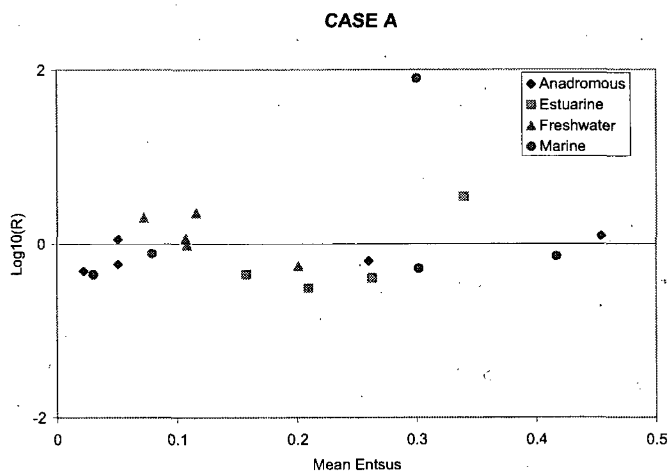
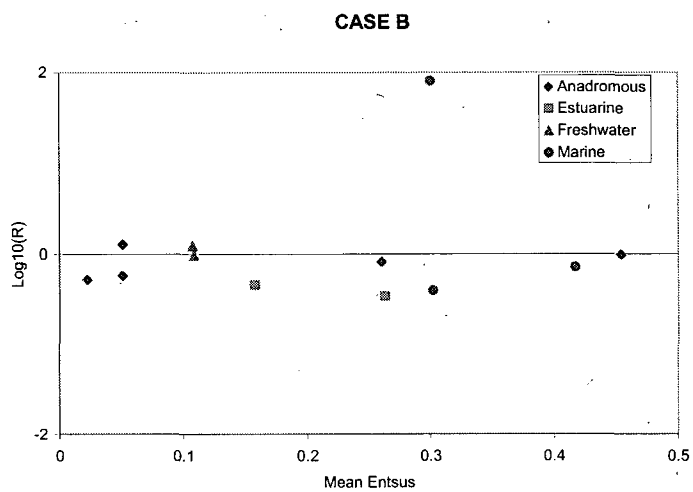


Figure 44b. Relationship between relative change in YOY abundance from Period 1 to Period 2 and entrainment susceptibility for the 11 fish species included in Case B. Zero on the logarithmic Y axis corresponds to no change in abundance from Period 1 to Period 2.



APPENDIX A

Prepared by:
AKRF, Inc.

TABLE OF CONTENTS

I. INTRODUCTION

Indices of relative abundance, derived from Hudson River Generator's Longitudinal River Ichthyoplankton Survey ("LRS"), Beach Seine Survey ("BSS"), and Fall Shoals Survey ("FSS") data, are used to analyze trends in abundance and to test the impact hypothesis for eight different species of finfish found in the Hudson River. These analyses are presented in Appendix B.

To confirm that the selection of relative abundance indices in Appendix B is valid, this document presents an examination of relationships that exist among LRS, BSS and FSS data. It also examines relationships that exist among LRS, BSS and FSS data and data from the Atlantic States Marine Fisheries Commission ("ASMFC"), as well as relationships that exist with the coast-wide striped bass abundance derived from its stock assessment (ASMFC 2005), the New York State Department of Environmental Conservation ("NYSDEC"), and the Hudson River Generators' mark-recapture studies of Atlantic tomcod ("ATMR") and striped bass. Correlation among these surveys validates the use of the LRS, BSS and FSS in Appendix B and demonstrates the robustness of the trends analysis and test of impact.

The strength of the correlation analysis can be evaluated using a power analysis. The power of a particular statistical test refers to the probability that the null hypothesis has been correctly rejected. In the case of a correlation analysis, the null hypothesis is defined as no significant correlation between surveys. The alternative hypothesis is defined as the presence of significant correlation between surveys. The power of a correlation analysis for different sample sizes is shown in Figure 1.

II. COMPARISON OF HUDSON RIVER GENERATORS' DATA

A correlation analysis was used to validate the use of the BSS and FSS surveys. The analysis demonstrates that the abundance index derived from the BSS follow the abundance index derived from the FSS.

A. Methods

Two datasets were compared in this analysis. Species-specific young-of-year indices based on the BSS were compared with species-specific FSS indices. See Appendix B for details on the development of these indices. The BSS and FSS indices are presented in Tables A-1 and A-2. The FSS indices were subset to the time period 1985 through 2004 to ensure that gear were comparable to the gear used in the BSS.

A Pearson correlation analysis was conducted, comparing the indices on a species-specific basis. A weighting factor based on the inverse of the variance was used, as described in the formula below:

$$WF = \frac{1}{(SE_{BSS})^2 + (SE_{FSS})^2}$$

where:

WF	=	weighting factor for Pearson Correlation Analysis
SE_{BSS}	=	standard error of BSS abundance estimate
SE_{FSS}	=	standard error of FSS abundance estimate

This analysis was conducted for white perch, striped bass, spottail shiner, bay anchovy, American shad, alewife, blueback herring, and Atlantic tomcod.

B. Results

The correlation analysis shows that that seven of the eight species of fish considered in this analysis are significantly and positively correlated (Table A-3). The correlation coefficients among the seven species range from 0.5 to 0.80. According to Figure A-1, the sample size of 20 in the present correlation analysis results in the power for the test ranging from about 60% to about 100%. Spottail shiner is the only species that does not show a significant correlation between the two indices. The lack of correlation is most likely attributable to large variation in the FSS data within individual years (Table A-2). The coefficient of variation for spottail shiner catch rates range between 0.17 and 1 in the FSS. Based on the overall results of the analysis, it can be concluded that species and life stages that share both habitats and are sampled by the two surveys exhibit the same interannual variation. This variation is reflected in the indices of the two surveys.

III. COMPARISON OF STRIPED BASS DATA WITH INDEPENDENT STUDIES

This analysis examines the relationship between the BSS striped bass data with independent studies conducted by the NYSDEC, the ASMFC and the Hudson River Generators.

Striped bass is sampled in a beach seine survey conducted by the NYSDEC. This survey is conducted in the Tappan Zee and Croton-Haverstraw region of the Hudson River. This is an area where a large proportion of the young-of-year ("YOY") striped bass found in the Hudson River are located in late summer and fall. The BSS and the NYSDEC beach seine survey overlap in this area, but the BSS samples a much larger area of the Hudson River, ranging from near the mouth of the river to Troy Dam. The two surveys have run concurrently since 1982. The size and the method of setting the beach seines vary between the two surveys. A correlation analysis was conducted to validate the use of the BSS in Appendix B.

The results from the NYSDEC beach seine survey are also used in the stock assessment of striped bass performed by the ASMFC (2005). An additional 61 age-specific and age-aggregated fishery-independent and fishery-dependent indices were used in the striped bass stock assessment (ASMFC 2005). A correlation analysis between the BSS and the coast-wide striped bass population abundance was conducted to show whether the Hudson River striped bass contribute significantly to the abundance of the coast-wide population.

Finally, the Hudson River Generators conducted a mark-recapture study of striped bass from 1984 through 1993. A correlation analysis was conducted to demonstrate the validity of the BSS when compared to these mark-recapture data.

A. Methods

Input data for this analysis included the ASMFC 2005 striped bass stock assessment – both total stock estimates as well as indices of abundance for different spawning regions, BSS YOY data, and striped bass mark-recapture data presented in the Draft Environmental Impact Statement ("DEIS") (Central Hudson Gas & Electric Corp. et al. 1999).

A linear regression was used to determine the fraction of the overall striped bass stock that could be attributed to the three major spawning stock regions: the Hudson River, the Delaware Estuary, and the Chesapeake Bay. The total estimated population of age-1 striped bass, as reported in the 2005 stock assessment (Table A-4), was compared with the indices of abundance for New York, New Jersey, Maryland, and Virginia (Table A-5) ("Model 1"). The index of New York abundance used by ASMFC was based on NYSDEC sampling data. A second linear regression was developed using BSS YOY data (Table A-1) to represent the New York component of the stock ("Model 2").

A correlation analysis using a Pearson model was used to compare the NYSDEC index, the BSS index, mark-recapture data collected by the Hudson River Generators (Table A-6), the estimate of the New York portion of the striped bass stock based on NYSDEC data (Table A-7), and the estimate of the New York portion of the striped bass stock based on BSS data (Table A-7).

B. Results

The correlation analysis between the BSS and the NYSDEC beach seine survey results in a significant positive correlation (Table A-8). This demonstrates that the two independent surveys of young-of-year striped bass in the Hudson River produce similar annual results. BSS and the coast-wide population abundance of striped bass are also significantly positively correlated. This positive correlation is not surprising, as the NYSDEC beach seine survey is one of many input parameters used in the coast-wide stock assessment of striped bass (ASMFC 2005). It has already been established that the NYSDEC beach seine survey and the BSS are positively correlated (See Section II.B). However, the results show that the many other input parameters in the striped bass stock assessment do not mask this relationship and confirm that striped bass associated with the Hudson River contribute significantly to the population dynamics of the coast-wide striped bass population. Another independent survey, a mark-recapture study, shows a significant linear relationship with the BSS. In summary, the BSS correlates significantly and positively with other existing independent surveys of striped bass YOY and older. This shows the robustness of the BSS in predicting young-of-year striped bass abundance.

IV. COMPARISON OF ATLANTIC TOMCOD DATASET WITH INDEPENDENT STUDIES

The ATMR study in the Hudson River has been conducted for 22 years, starting in 1974 (Normandeau Associates, Inc. 2006). Abundance indices of 1 and 2 year old Atlantic tomcod are calculated, using data from the ATMR program (Table A-9). Yearly egg production estimates are also provided in Normandeau (2006).

Atlantic tomcod data from the BSS, FSS, and the LRS were compared with data from the mark-recapture study conducted by the Hudson River Generators to validate the results of the ATMR program by determining if correlations among the datasets exist.

A. Methods

There were multiple inputs used to conduct further examinations of the Atlantic tomcod data used in earlier analyses. These data included the Atlantic tomcod index presented in Appendix A (based on mark-recapture surveys), BSS data, FSS data, and LRS data (Table A-10). Two different statistical methods were used to examine the Atlantic tomcod data.

- A correlation analysis, based on the Pearson model, was conducted comparing the mark-recapture data of age-1 Atlantic tomcod with young-of-year BSS and FSS data.
- A second correlation analysis, also based on the Pearson model, compared the estimated of eggs derived from the mark-recapture study with the post yolk-sac index based on LRS data.

B. Results

The relative abundance of Atlantic tomcod based on the FSS is significantly and positively correlated with their abundance based on the BSS (Table A-11). The mark-recapture program for Atlantic tomcod also correlates positively and significantly to the FSS and the BSS. The egg deposition is borderline positively correlated to the post yolk-sac larvae Atlantic tomcod estimated from the LRS (Table A-12).

V. LITERATURE CITED

Atlantic States Marine Fisheries Commission 2005. 2005 Stock Assessment Report for Atlantic striped bass: Catch-at-age VPA & tag release/recovery based survival estimation. A report prepared by the striped bass technical committee for the Atlantic striped bass management board. Atlantic States Marine Fisheries Center.

Central Hudson Gas & Electric Corp. et al. 1999. Draft Environmental Impact Statement for State Pollutant Discharge Elimination System Permits for Bowline Point, Indian Point 2 & 3, and Roseton Steam Electric Generating Stations.

Normandeau Associates, Inc. 2006. Abundance and stock characteristics of the Atlantic tomcod spawning population in the Hudson River, winter 2003-2004. Prepared for Entergy Nuclear Operations, Inc.

VI. TABLES

Table 1. Abundance indices and associated standard errors, based on BSS.

Year	WHITE PERCH Young-of-Year		STRIPED BASS Young-of-Year		SPOTTAIL SHINER Young-of-Year		BAY ANCHOVY Young-of-Year		AMERICAN SHAD Young-of-Year		ALEWIFE Young-of-Year	
	Index	SE	Index	SE	Index	SE	Index	SE	Index	SE	Index	SE
1974	566,346	61,280	1,373,138	264,598	658,945	87,448	2,999,066	973,844	2,123,265	232,509	583,238	74,805
1975	2,342,937	440,999	1,367,496	242,374	1,286,297	193,361	5,159,511	1,666,189	1,998,286	161,394	572,550	107,585
1976	1,944,220	255,910	864,743	70,734	1,324,434	203,989	5,234,482	2,595,405	2,354,807	125,450	352,263	96,375
1977	953,799	87,722	1,375,537	124,595	495,690	66,445	4,616,994	875,014	2,123,707	114,152	517,792	49,081
1978	2,675,700	402,374	3,042,920	614,048	1,363,313	148,541	329,478	57,321	4,021,203	251,047	1,027,891	174,698
1979	2,921,393	285,862	794,022	91,389	956,236	97,330	1,860,753	686,496	1,934,405	107,064	340,271	59,099
1980	1,884,895	231,650	1,265,254	147,121	633,323	72,196	3,445,878	818,900	1,632,041	117,820	93,783	17,894
1981	1,862,222	160,903	1,827,767	152,481	1,865,058	216,442	4,505,689	1,862,587	2,558,539	149,238	477,348	84,403
1982	1,967,754	287,490	934,550	97,768	477,090	62,605	2,740,240	1,735,314	1,768,839	150,312	116,606	24,817
1983	1,803,266	399,823	1,642,536	191,103	1,070,822	104,909	364,403	243,354	2,452,068	183,820	214,922	42,154
1984	703,959	145,133	1,300,754	173,872	616,182	128,367	1,887,240	963,767	1,060,902	74,374	49,776	10,864
1985	757,003	82,536	238,259	21,226	543,246	66,532	621,718	203,675	1,263,843	153,248	119,509	22,024
1986	1,036,321	97,303	298,745	31,415	388,736	69,297	975,435	779,300	2,207,907	125,447	119,468	48,899
1987	1,169,236	121,876	2,976,381	314,807	470,267	74,827	830,978	229,609	1,482,041	125,017	80,611	13,768
1988	1,738,310	255,364	1,172,303	68,239	419,874	49,588	546,894	225,975	997,414	59,920	87,080	15,727
1989	1,105,280	278,101	1,238,434	116,464	623,204	95,526	2,840,186	987,471	2,455,819	135,247	43,711	12,956
1990	588,162	75,727	1,486,911	89,409	808,662	101,694	208,541	65,810	2,004,620	162,122	157,159	25,580
1991	580,165	76,201	1,125,126	64,076	855,292	110,557	935,366	246,296	1,499,227	120,544	335,535	63,111
1992	463,555	53,444	1,046,654	53,265	726,888	124,009	1,629,973	1,184,246	1,886,715	101,469	40,507	9,371
1993	806,848	97,157	1,640,132	90,969	655,117	95,425	1,183,278	462,699	815,539	68,698	69,438	11,826
1994	315,662	39,618	1,136,106	63,179	1,624,997	289,784	2,255,731	478,603	1,963,731	124,116	148,030	30,079
1995	425,062	49,042	1,404,935	89,202	603,130	94,204	2,507,280	721,809	552,490	48,911	91,731	22,716
1996	44,925	10,283	299,997	30,506	174,026	39,053	720,000	151,968	1,743,007	125,007	47,371	14,912
1997	571,160	114,812	1,892,597	169,399	1,197,799	170,583	3,496,618	815,723	1,573,674	106,235	291,323	54,177
1998	270,835	51,992	1,384,364	85,327	273,165	53,055	2,675,549	670,172	319,702	47,834	40,865	30,194
1999	1,411,184	169,447	1,715,282	142,568	2,040,399	243,244	858,192	298,574	1,399,557	107,459	445,167	79,622
2000	304,950	52,787	580,006	52,449	303,081	52,956	769,133	427,827	941,909	105,935	76,445	37,606
2001	1,019,516	119,666	2,392,216	170,860	2,143,066	610,761	613,810	401,115	2,479,221	176,132	330,876	70,451
2002	699,145	80,612	1,145,686	60,295	1,132,479	146,862	3,826,181	1,061,795	721,680	72,203	60,954	13,491
2003	2,177,013	228,303	2,282,684	118,276	2,102,568	257,006	1,703,952	451,911	1,071,881	69,880	452,292	87,223
2004	632,961	89,075	807,661	70,743	1,031,399	152,802	404,497	145,762	444,880	31,585	218,118	35,902

Table A-1. Abundance indices and associated standard errors, based on BSS (continued).

Year	BLUEBACK HERRING Young-of-Year		ATLANTIC TOMCOD Young-of-Year	
	Index	SE	Index	SE
1974	3,647,758	502,857	18,536	4,046
1975	10,888,524	1,249,788	39,688	11,253
1976	21,621,271	3,075,761	41,196	12,039
1977	31,795,371	4,717,652	8,178	2,802
1978	22,993,451	4,200,939	37,401	11,147
1979	8,221,314	1,461,758	58,632	18,283
1980	8,892,467	2,207,337	17,337	6,016
1981	32,066,440	9,586,015	3,698	1,141
1982	10,164,307	1,750,817	70,051	14,120
1983	16,326,879	2,278,723	11,419	3,218
1984	3,577,323	786,742	50,486	12,104
1985	3,323,511	664,762	34,760	6,246
1986	1,555,182	357,032	28,125	5,369
1987	6,188,101	773,111	35,074	8,600
1988	5,887,963	1,008,925	21,020	5,249
1989	3,230,116	497,839	12,946	3,825
1990	9,436,487	1,274,900	16,941	5,709
1991	3,530,392	596,059	4,417	1,849
1992	6,642,282	1,599,250	43,740	10,403
1993	4,234,168	531,496	2,144	913
1994	9,584,696	1,308,960	1,198	579
1995	3,202,735	892,613	0	0
1996	4,044,353	890,186	9,182	5,836
1997	12,075,530	2,541,612	5,053	1,572
1998	155,761	32,365	1,384	616
1999	5,691,570	776,702	0	0
2000	2,342,499	572,561	9,823	3,892
2001	5,268,663	704,402	1,520	752
2002	1,438,577	299,230	0	0
2003	10,203,281	1,459,824	0	0
2004	5,091,421	620,888	5,928	1,647

Table A-2. Abundance indices and associated standard errors, based on FSS.

year	WHITE PERCH		STRIPED BASS		SPOTTAIL SHINER		BAY ANCHOVY		AMERICAN SHAD		ALEWIFE	
	Young-of-Year		Young-of-Year		Young-of-Year		Young-of-Year		Young-of-Year		Young-of-Year	
	Index	SE	Index	SE	Index	SE	Index	SE	Index	SE	Index	SE
1985	1,685,851	165,213	164,284	16,636	85,977	39,236	218,612,898	21,269,766	1,591,435	190,139	2,105,489	381,844
1986	1,759,522	207,644	651,049	49,859	49,745	11,399	132,925,173	13,133,411	3,104,605	640,844	595,155	115,129
1987	1,579,037	136,932	4,889,589	239,032	20,977	5,401	246,910,112	26,982,497	647,070	157,299	695,124	245,872
1988	3,777,521	297,018	9,569,544	497,548	83,429	20,121	422,678,791	38,213,532	997,871	144,252	624,702	142,344
1989	3,167,143	357,848	4,235,166	333,577	3,591	1,550	349,952,337	26,107,654	2,754,815	198,752	505,822	105,987
1990	548,583	167,722	2,883,805	200,426	17,347	5,614	161,039,442	14,450,450	1,139,272	235,276	807,620	138,564
1991	443,688	67,292	1,138,102	87,685	131,938	34,430	190,474,265	11,540,891	680,209	72,781	685,242	104,724
1992	1,064,922	136,793	1,186,233	113,756	23,041	8,964	185,902,303	13,738,226	1,306,732	147,744	746,514	158,432
1993	415,097	100,885	2,779,357	178,004	70,379	17,018	249,913,241	19,475,645	464,702	48,446	530,240	83,846
1994	566,404	53,440	3,439,449	209,768	34,772	5,983	206,642,043	14,141,476	1,036,782	88,932	571,174	82,018
1995	1,514,550	230,289	2,878,188	173,061	10,530	3,570	439,617,793	28,732,239	471,444	75,896	308,139	49,342
1996	414,924	60,068	2,396,874	172,968	73,863	15,117	102,941,191	5,959,974	2,859,373	451,439	1,076,096	124,312
1997	539,792	86,123	2,439,137	273,488	6,312	2,846	283,382,412	17,014,202	913,970	107,851	1,233,697	154,951
1998	357,696	35,390	580,977	65,746	2,367	2,367	189,541,611	9,166,785	232,260	56,459	112,261	28,629
1999	2,021,946	166,188	2,655,600	220,747	25,220	5,712	165,375,818	9,972,244	853,411	135,639	2,543,734	197,641
2000	433,794	60,439	1,634,254	228,331	2,010	1,496	57,208,944	3,577,181	878,405	100,807	913,399	108,152
2001	869,631	93,161	1,184,609	105,581	20,724	9,574	109,701,139	8,052,515	1,006,787	162,014	2,253,572	652,056
2002	401,209	46,026	982,555	156,264	14,619	4,774	171,692,430	10,652,063	497,537	57,524	255,519	37,190
2003	2,181,001	165,766	4,787,259	432,818	938	841	148,898,706	11,753,477	351,278	47,131	941,836	102,643
2004	543,243	159,067	991,181	119,540	40,935	8,459	218,178,981	17,899,774	336,973	63,105	249,944	43,269

Table A-2. Abundance indices and associated standard errors, based on FSS (continued).

year	BLUEBACK HERRING Young-of-Year		ATLANTIC TOMCOD Young-of-Year	
	Index	SE	Index	SE
1985	63,437,557	9,471,265	3,818,562	537,609
1986	15,577,561	2,395,825	6,935,212	588,195
1987	38,342,783	9,373,512	3,431,206	257,718
1988	61,946,416	6,136,684	3,731,674	370,666
1989	33,621,840	3,107,711	13,006,674	1,862,570
1990	63,121,526	6,836,956	1,377,747	247,070
1991	43,421,773	5,346,974	263,792	37,402
1992	46,987,241	6,744,931	3,846,993	297,928
1993	20,223,194	1,817,165	3,742,238	1,013,814
1994	17,568,127	1,521,183	604,300	55,493
1995	14,114,745	1,634,192	84,328	16,082
1996	67,981,601	8,013,906	3,543,737	380,726
1997	29,241,071	3,323,567	2,392,903	208,967
1998	927,634	153,551	507,900	73,503
1999	22,609,332	2,329,531	19,312	6,888
2000	11,400,882	1,150,959	2,262,871	196,166
2001	23,294,104	4,713,494	897,887	240,836
2002	10,219,873	969,053	80,565	17,597
2003	17,724,162	1,789,797	355,046	74,484
2004	6,347,406	606,675	2,100,531	318,419

Table A-3. Correlations between BSS and FSS data

Taxa	Number of Years	Inverse-Variance Weighted Correlation Factors	Significance Level
White Perch	20	0.69	0.0007
Striped Bass	20	0.69	0.0008
Spottail Shiner	20	-0.09	0.6969
Bay Anchovy	20	0.55	0.0122
American Shad	20	0.76	<0.0001
Alewife	20	0.50	0.0235
Blueback Herring	20	0.73	0.0002
Atlantic Tomcod	20	0.80	<0.0001

Table A-4. Estimated age-1 striped bass population.

Year	Striped Bass Age-1 Population (thousands)
1982	1,534
1983	3,181
1984	2,401
1985	3,579
1986	2,763
1987	3,944
1988	5,219
1989	5609
1990	8,419
1991	8,644
1992	8,706
1993	11,065
1994	16,562
1995	13,338
1996	12,932
1997	15,586
1998	10,625
1999	10,982
2000	8,261
2001	15,490
2002	18,024
2003	5,976
2004	22,275
2005	12,721

Source: ASMFC 2005

Table A-5. Indices of abundance for Atlantic striped bass adjusted to January 1st

Year	Young-of-Year New York Index	Young-of-Year New Jersey Index	Young-of-Year Maryland Index	Young-of-Year Virginia Index
1982	8.86		0.59	1.56
1983	14.17	0.12	3.57	2.71
1984	16.25	0.03	0.61	3.4
1985	15	0.29	1.64	4.47
1986	1.92	0.18	0.91	2.41
1987	2.92	0.28	1.34	4.74
1988	15.9	0.41	1.46	15.74
1989	33.46	0.35	0.73	7.64
1990	21.35	1.03	4.87	11.23
1991	19.08	1	1.03	7.34
1992	3.6	0.47	1.52	3.76
1993	11.43	1.19	2.34	7.35
1994	12.59	1.78	13.97	18.11
1995	17.64	0.96	6.4	10.48
1996	16.23	1.98	4.41	5.45
1997	8.93	1.7	17.61	23
1998	22.3	1.01	3.91	9.35
1999	13.39	1.31	5.5	13.25
2000	26.64	1.9	5.34	2.8
2001	3.16	1.77	7.42	16.18
2002	22.98	1.07	12.57	14.17
2003	12.32	0.51	2.2	3.98
2004	17.36	2.43	10.83	22.89
2005	8.81	1.13	4.85	12.7

Source: ASMFC 2005

Table A-6. Abundance estimate of Hudson River striped bass, based on mark-recapture data.

Year	Age-2+ Abundance
1984	213
1985	104
1986	108
1987	611
1988	560
1989	339
1990	344
1991	502
1992	238
1993	201

Source: Central Hudson Gas & Electric Corp. et al. 1999

Table A-7. Estimate of NY striped bass stock, based on NYSDEC and BSS data.

Year	Estimate of Hudson River age-1 striped bass	
	Based on NYSDEC Data	Based on BSS data
1974		1,510,636
1975		1,504,429
1976		951,333
1977		1,513,275
1978		3,347,621
1979		873,531
1980		1,391,949
1981	560,788	2,010,789
1982	896,882	1,028,131
1983	1,028,534	1,807,010
1984	949,416	1,431,004
1985	121,525	262,117
1986	184,820	328,660
1987	1,006,381	3,274,419
1988	2,117,831	1,289,691
1989	1,351,336	1,362,444
1990	1,207,657	1,635,802
1991	227,860	1,237,790
1992	723,455	1,151,460
1993	796,877	1,804,365
1994	1,116,513	1,249,869
1995	1,027,268	1,545,617
1996	565,219	330,037
1997	1,411,465	2,082,111
1998	847,512	1,522,986
1999	1,686,163	1,887,041
2000	200,010	638,085
2001	1,454,505	2,631,759
2002	779,787	1,260,408
2003	1,098,791	2,511,259
2004	557,624	888,536

Table A-8. Striped Bass correlation coefficients

	New York Index	BSS Index	Mark-recapture age-2 Abundance	New York Stock (based on NYDEC data)
New York Index		**0.53**	0.55	**1.00**
BSS Index	**0.53**		**0.68**	**0.53**
Mark-recapture age-2	0.55	**0.68**		0.55
New York Stock (based on NYDEC data)	**1.00**	**0.53**	0.55	

	New York Index	BSS Index	Mark-recapture age-2 Abundance	New York Stock (based on BSS data)
New York Index		**0.53**	0.55	**0.53**
BSS Index	**0.53**		**0.68**	**1.00**
Mark-recapture age-2	0.55	**0.68**		**0.68**
New York Stock (based on BSS data)	**0.53**	**1.00**	**0.68**	

Note: Correlation coefficients significant at the 10% level are shown.
Correlation coefficients significant at the 5% level are shown in bold.

Table A-9. Atlantic tomcod mark-recapture data

Year	Proportion Age-1	Proportion Age-2	Population Egg Deposition (billions)	Population Age-1 (millions)
1975				3.6
1976	0.98	0.02	22	9.7
1977	0.933	0.067	65	2.4
1978	0.965	0.035	21	5.9
1979	0.989	0.01	51	8.8
1980	0.97	0.03	57	
1981	0.943	0.056		
1982	0.968	0.032		10.5
1983	0.843	0.155	97	5.9
1984	0.887	0.113	75	
1985	.	.		2
1986	0.957	0.043	25	
1987	.	.		2.9
1988	0.837	0.163	43	5.3
1989	0.9	0.1	41	4.9
1990	0.715	0.285	87	2.6
1991	0.81	0.19	52	0.3
1992	0.715	0.285	7	2.2
1993	0.849	0.151	30	0.5
1994	0.662	0.338	7	2.2
1995	0.907	0.093	31	
1996	0.483	0.517		2.6
1997	0.8	0.2	47	0.7
1998	0.535	0.465	23	0.4
1999	0.664	0.336	10	0.2
2000	0.799	0.201	3	2.3
2001	0.935	0.065	28	
2002	0.827	0.173		
2003	0.95	0.05		1.6
2004	0.952	0.048	28	

Source: Normandeau Associates, Inc. 2006

Table A-10. Atlantic Tomcod abundance index and associated standard errors, based on LRS

year	ATLANTIC TOMCOD Post Yolk-Sac Larvae Index	SE
1974	128,306,743	19,426,263
1975	67,024,707	19,768,962
1976	42,777,042	13,470,065
1977	164,621,663	70,515,234
1978	54,313,088	10,307,482
1979	18,127,435	3,099,375
1980	95,402,234	13,128,146
1981	74,140,778	13,052,007
1982	28,419,800	7,665,326
1983	42,683,202	8,311,722
1984	147,133,069	25,916,525
1985	109,664,584	11,132,251
1986	53,404,268	4,770,519
1987	138,570,516	12,594,732
1988	78,376,300	10,680,903
1989	185,450,859	23,858,579
1990	107,915,374	25,158,013
1991	116,333,462	14,859,973
1992	32,021,214	4,889,565
1993	126,394,886	20,139,893
1994	85,456,373	22,227,930
1995	79,816,881	6,641,688
1996	51,571,386	5,696,759
1997	110,409,961	28,829,551
1998	53,594,909	8,409,591
1999	17,392,702	2,076,588
2000	11,120,807	1,442,773
2001	93,816,691	8,320,053
2002	4,382,650	649,979
2003	38,715,789	3,683,762
2004	115,401,578	16,005,570

Table A-11. Atlantic tomcod correlation coefficients

	Age-1 Mark-recapture data	Young-of-year: BSS data	Young-of-year: FSS data
Age-1: mark-recapture data		**0.77**	**0.65**
Young-of-year: BSS data	**0.77**		**0.45**
Young-of-year: FSS data	**0.65**	**0.45**	

Note: Only correlation coefficients significant at the 10% level are shown.
Correlation coefficients significant at the 5% level are shown in bold.

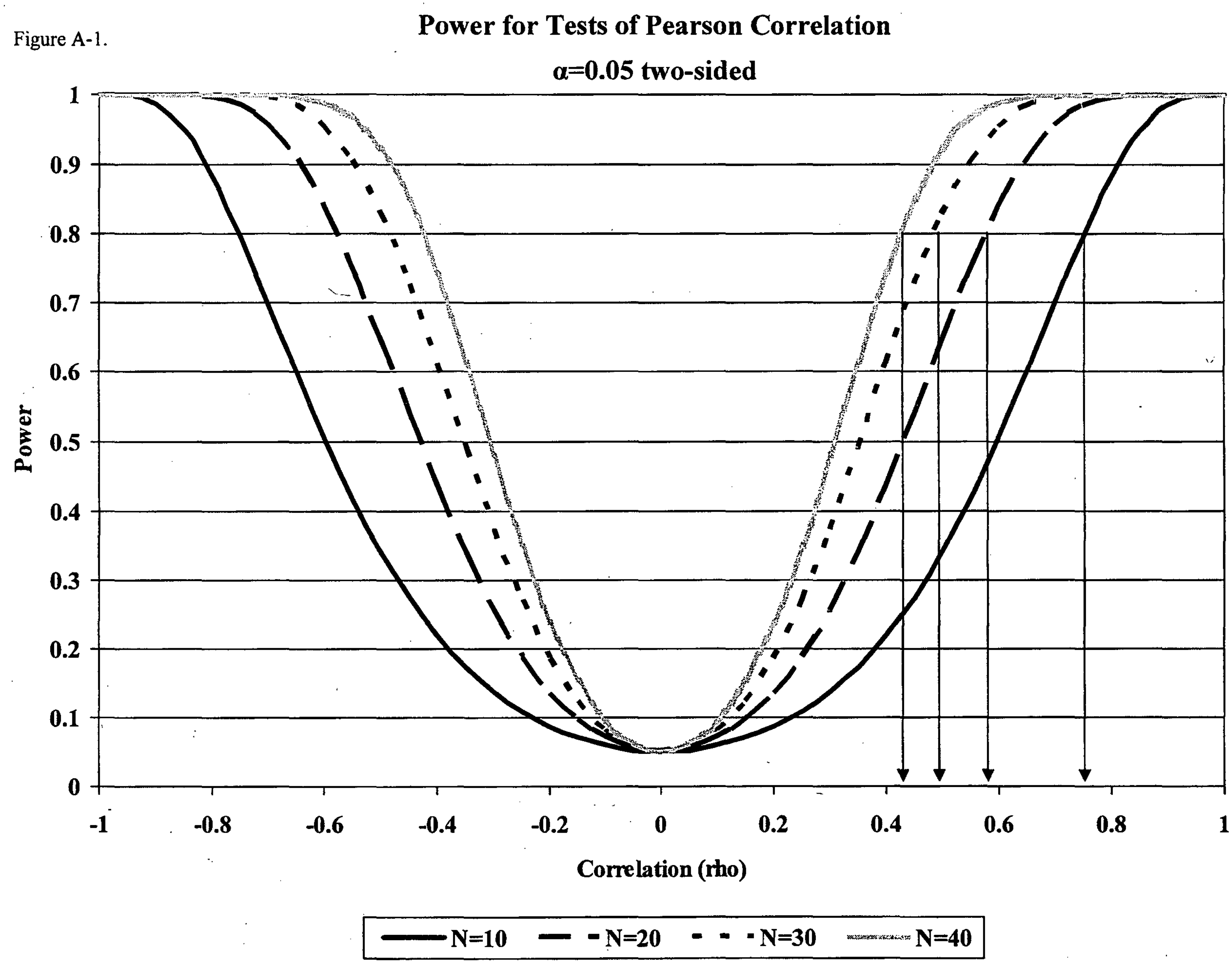
Table A-12. Atlantic tomcod correlation coefficients

	Eggs: Mark-recapture data	Post yolk-sac: LRS data
Eggs: mark-recapture data		0.41
Post yolk-sac: LRS data	0.41	

Note: Only correlation coefficients significant at the 10% level are shown.
Correlation coefficients significant at the 5% level are shown in bold.

VII. FIGURES

Figure A-1.



Power for Tests of Pearson Correlation
α=0.05 two-sided
Power
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-1
-0.8
-0.6
-0.4
-0.2
0
0.2
0.4
0.6
0.8
1
Correlation (rho)
N=10
N=20
N=30
N=40

APPENDIX B

Prepared by:

AKRF, Inc.

TABLE OF CONTENTS

I. INTRODUCTION

This Appendix documents the methods and data used in: (1) analyses of trends in fish population abundance; and (2) correlation analyses to address impact hypotheses. The rationale for and the results from the analyses of trends and the correlation analyses are discussed in the report titled: "Entrainment and Impingement at IP2 and IP3: A Biological Impact Assessment."

The analyses of trends in fish population abundance and the correlation analyses were based on indices developed from data collected by the Hudson River Generators' Longitudinal River Ichthyoplankton Survey ("LRS"), Beach Seine Survey ("BSS"), Fall Shoals Survey ("FSS"), and Atlantic Tomcod Mark-Recapture ("ATMR") Program. Three types of indices were defined for these analyses:

- indices of fish population abundance;
- indices of stressors of fish populations; and
- indices of fish population response to stressors.

The remainder of this Appendix is organized in three Sections. The first Section documents the three types of indices; the second Section documents the trend analysis methods and results; and the third Section documents the correlation analysis methods and results.

II. INDICES

A. Fish Population Abundance

Annual indices of fish population abundance were computed as the average of the weekly standing crop estimates presented in the Year Class Report for the Multiplant Impact Study of the Hudson River Estuary for the years 1974 through 1979 and the Hudson River Estuary Monitoring Program for the years 1980 through 2004 (collectively, ("Year Class Report") (Applied Science Associates, Inc. 2000, 2001; ASA Analysis & Communication, Inc. 2001, 2002, 2003, 2004a, 2004b, 2005, 2006; Batelle New England Marine Research Laboratory 1983; Consolidated Edison Company of New York, Inc. 1996, 1997a, 1997b; EA Engineering, Science, and Technology 1990, 1991, 1996; Lawler, Matusky & Skelly Engineers 1989, 1992, 1996; Martin Marietta Environmental Systems 1986; Normandeau Associates, Inc. 1985a, 1985b; Texas Instruments, Inc. 1977, 1978, 1979, 1980a, 1980b, 1981; Versar, Inc. 1987). A separate annual index value was computed for each species and life stage. Indices of abundance for age-1 and age-2 Atlantic tomcod and abundance of Atlantic tomcod eggs were based on abundance estimates from the ATMR Program (Normandeau Associates, Inc. 2006).

Weekly standing crop estimates for post yolk-sac larvae ("PYSL") were based on data collected by the LRS. Weekly standing crop estimates for young-of-year[1] ("YOY") fish inhabiting the beach zone of the Hudson River were based on data collected by the BSS. Weekly standing crop estimates for YOY fish inhabiting the shoals, bottom, and channel of the Hudson River were based on data collected by the FSS. These standing crop estimates, with associated standard errors, were provided in electronic format by ASA Analysis & Communication, Inc.

[1] Young-of-year fish are sometimes also referred to as juvenile fish.

("ASA"). Data collection methods for the LRS, BSS, and FSS, and methods for estimating weekly standing crops (and associated standard errors) are documented in the Year Class Reports. Annual estimates of the number of age-1 and age-2 Atlantic tomcod and the number of Atlantic tomcod eggs spawned were developed by the ATMR program, and were provided by Normandeau Associates, Inc. ("NAI"). Data collection methods for the ATMR program and methods for estimating Atlantic tomcod abundance are documented in annual ATMR Program Reports prepared by NAI for the Hudson River Generators. In addition, estimates of the variance of the estimate of the total number of age-1 and age-2 Atlantic tomcod were computed, as described below.

A set of regions and weeks that were consistently sampled among years was identified for each sampling program. Annual abundance indices based on LRS data were computed for 1974 through 2004, based on data from regions 1 through 12, and weeks 18 through 26. Annual abundance indices based on BSS data were computed for 1974 through 2004, based on data from regions 1 through 12, and weeks 31 through 42. Annual abundance indices based on FSS data were computed for 1979 through 2004, based on data from regions 1 through 12, and weeks 31 through 42. Data from the ATMR program were included for all years (1974 through 2004) in which the number of recaptured Atlantic tomcod exceeded one fish.

BSS data were used to develop YOY abundance indices for alewife, blueback herring, spottail shiner, striped bass, and white perch. FSS data were used to develop YOY abundance indices for American shad and bay anchovy. LRS data were used to develop the PYSL indices for striped bass, white perch, river herring (which included alewife, blueback herring and unidentified clupeids – three taxonomic groups that could not reliably be identified to species as PYSL), American shad, and bay anchovy. The LRS did not adequately sample areas of the river inhabited by spottail shiner larvae. To address the abundance of early life stages of spottail shiner, an index of egg abundance was developed based on spawning age spottail shiner (i.e., yearling and older) sampled by the BSS. The index of yearling and older spottail shiner was used as a surrogate index for spottail shiner egg abundance.

For each species, sampling program (LRS, BSS, and FSS), and year, the annual index of abundance ($\overline{A}_y$) was computed using the following formula:

$$\overline{A}_y = \left(\frac{1}{\sum_{W=W_{\min}}^{W_{\max}} \delta_{W,y}} \right) \sum_{W=W_{\min}}^{W_{\max}} SC_{W,y} \times \delta_{W,y}$$

where

$$SC_{W,y} = \sum_{R=1}^{12} SC_{R,W,y}$$

W_{min} = first week of the season,

W_{max} = last week of the season,

$SC_{R,W,y}$ = estimated standing crop in region R, week W and year y,

$\delta_{W,y}$ = 1 if all 12 standard regions were sampled in week W of year y, and

$\delta_{W,y}$ = 0 otherwise.

For Atlantic tomcod, approximately unbiased Peterson-type mark-recapture estimates of abundance were computed as (Seber 1982):

$$\tilde{A}_y = \frac{(C_y + 1)(M_y + 1)}{(m_y + 1)} - 1$$

and the variance of the estimated abundance was estimated as (Seber 1982):

$$v(\tilde{A}_y) = \frac{(C_y + 1)(M_y + 1)(C_y - m_y)(M_y - m_y)}{(m_y + 1)^2 (m_y + 2)}$$

where

C_y = number of fish (marked and unmarked) caught subsequent to marking,

M_y = number of fish marked, and

m_y = number of marked fish recaptured.

The abundance indices are presented in Tables B-1 through B-3.

B. Stressors of Fish Populations

Four potential stressors of fish populations in the Hudson River estuary were identified: (1) power plant mortality due to entrainment at Indian Point; (2) effects of the zebra mussel invasion on the Hudson River biota; (3) predation by increased abundance of striped bass in the Hudson River estuary; and (4) elevated late summer and fall bottom temperatures. For each stressor, an index was developed that was intended to track the intensity of the stressor.

1. Power Plant Mortality

The index of entrainment mortality at Indian Point was the conditional mortality rate ("CMR"). An annual CMR for entrainment can be interpreted as the fractional reduction in age-1 abundance of a year class of fish due to the effects of entrainment, assuming the absence of density-dependent mortality. Estimates of CMRs for entrainment at Indian Point from 1974 through 1997 were taken from the Draft Environmental Impact Statement ("DEIS") for State Pollution Discharge Elimination System Permits for Bowline Point, Indian Point 2 & 3, and Roseton Steam Electric Generating Stations (Central Hudson Gas & Electric Corp. et al. 1999). CMR estimates for entrainment at Indian Point for 1998 through 2003 were computed for this analysis using the same methods documented in the DEIS. CMR estimates were computed separately for striped bass, white perch, American shad, bay anchovy, spottail shiner, Atlantic tomcod, and river herring.

The indices of entrainment mortality are listed in Table B-4.

2. Zebra Mussels

The invasive zebra mussel (*Dreissena polymorpha*) first appeared in the Hudson in 1991 and became a dominant species in the Hudson River by September 1992 (Strayer et al. 1996). Strayer et al. (2004) reported that "(z)ebra mussels were quantitatively important only in freshwater parts of the Hudson, and their effects extend from the head of the estuary (rkm 248) down to approximately rkm 100 (Strayer et al. 1996; Caraco et al. 1997; Pace et al. 1998)." Based on this characterization, the indicator variable for zebra mussel effects was set to zero (i.e., no effect) for the period 1974 through 1992, and was set to one (i.e., effect was present) for the years 1993 through 2004. Also, an index of the spatial distribution of fish within the Hudson River was defined (see Section II.C.4, below), based on the relative abundance of fish downriver of rkm 100.

The index of zebra mussel effects is listed in Table B-5.

3. Striped Bass Predation

The index of striped bass predation was intended to represent the predatory pressure of adult striped bass on the fish community of the Hudson River estuary. Post yolk-sac larvae abundance was used as a surrogate for adult abundance under the assumption that PYSL abundance represented reproductive potential which, in turn, was roughly proportional to spawning abundance. Accordingly, the striped bass PYSL abundance index based on the LRS was used as the index of striped bass predation.

The index of striped bass predation is listed in Table B-6.

4. Temperature

For all species except Atlantic tomcod, the index of water temperature was based on water temperature in the bottom stratum of the river and was computed in two steps. First, a riverwide average temperature for each week within a season was computed. The weekly average value was computed as the weighted average, where the weighting factor for each region (1 through 12) was the volume of the bottom stratum in the region. The second step was to average the weekly values over all weeks (in which all 12 standard regions were sampled) within the season.

For Atlantic tomcod, an alternative index of water temperature was computed: a degree-day index based on data recorded at the Poughkeepsie Water Works ("PWW"). The annual PWW degree-day index was computed as the sum (January through December) of daily temperatures above 24°C. Days with water temperatures below 24°C did not contribute to the annual sum. The temperature of 24°C was chosen because growth in age-0 Atlantic tomcod from the Hudson River slows when water temperatures exceeded 20°C and ceased when water temperatures exceeded 24°C (Chambers and Witting, 2005).

The indices of water temperature are listed in Table B-7.

C. Fish Population Response Metrics

1. Survival Indices

Each survival index was defined as a ratio of abundance indices from two life stages: the denominator of the ratio was the earlier life stage and the numerator was a subsequent life stage. Therefore, the ratio was proportional to the fraction of the earlier life stage that survived to the subsequent life stage. Because the methods and data used for the abundance indices (see Section II.A, above) are species-specific, the definitions of the survival indices are also species-specific.

- The survival index for striped bass from PYSL to YOY was defined as the ratio of the YOY abundance index (based on BSS data) to the PYSL abundance index (based on LRS data).
- The survival index for white perch from PYSL to YOY was defined as the ratio of the YOY abundance index (based on BSS data) to the PYSL abundance index (based on LRS data).
- The survival index for alewife from PYSL to YOY was defined as the ratio of the alewife YOY abundance index (based on BSS data) to the river herring YOY abundance index (based on LRS data).
- The survival index for American shad from PYSL to YOY was defined as the ratio of the YOY abundance index (based on FSS data) to the PYSL abundance index (based on LRS data).
- The survival index for bay anchovy from PYSL to YOY was defined as the ratio of the YOY abundance index (based on FSS data) to the PYSL abundance index (based on LRS data).
- The survival index for spottail shiner from eggs to YOY was defined as the ratio of the YOY abundance index (based on BSS data) to the egg abundance index (based on BSS data).
- The survival index for Atlantic tomcod from age-1 to age-2 was defined as the ratio of the age-2 abundance index (based on ATMR data) to the age-1 abundance index (based on ATMR data).

- The survival index for Atlantic tomcod from eggs to age-1 was defined as the ratio of the egg abundance index (based on ATMR data) to the age-1 abundance index (based on ATMR data).

The survival indices are listed in Table B-8.

2. Abundance Indices

Because some stressors can act directly on the abundance of certain life stages, the abundance indices listed in Tables B-1 through B-3 were also used as response metrics.

3. Growth Indices

The growth index was intended to represent the relative amount of growth in juvenile fish that occurred during a standard set of weeks (31 through 42) in the fall of each year. Annual growth indices (1979 through 2004) were computed from BSS and FSS data.

The growth index for each species and year was computed in three steps. First, the average fish length was calculated for each week and region. Then, a weighted average length was computed for each week, where the weight for each region was the YOY abundances in the region. The third step was to conduct a log-linear regression analysis of the weighted-average length ($\overline{L}_W$) against week number (W):

$$\overline{L}_W = L_{W_{\min}} \times e^{\rho(W - W_{\min})}.$$

The slope estimate ($\hat{\rho}$) from that regression analysis represented the average growth rate during the fall season, and was used as the index of growth for the species in that year.

The growth indices are listed in Table B-9.

4. Spatial Distribution Indices

This index was intended to address the possible effects of zebra mussels on fish distribution patterns, and was defined as the portion of the total population that occurred downstream of rkm 100.

For American shad and bay anchovy, the spatial distribution indices for YOY were based on data from the FSS for weeks 31 through 42. For striped bass, white perch, blueback herring, alewife and spottail shiner, the spatial distribution indices for YOY were based on data from the BSS for weeks 31 through 42. The spatial distribution indices for PYSL were computed for striped bass, white perch, bay anchovy, American shad, river herring, and Atlantic tomcod based on data from the LRS from weeks 18 through 26. For Atlantic tomcod, which spawn in late winter/early spring, data from the LRS included juveniles in addition to PYSL. Annual spatial indices based on LRS data were computed for 1974 through 2004. Annual spatial indices based on BSS data were computed for 1974 through 2004. Annual spatial indices based on FSS data were computed for 1979 through 2004.

For each species, life stage, region (R), and year, the fraction of the riverwide abundance inhabiting areas within the region or downriver of the region ($\hat{F}_{R,y}$) was estimated using the following formula:

$$\hat{F}_{R,y} = \frac{\left(\sum_{r=1}^{R} \overline{SC}_{r,y}\right)}{\left(\sum_{r=1}^{12} \overline{SC}_{r,y}\right)}$$

where

$$\overline{SC}_{r,y} = \left(\frac{1}{\sum_{W=W_{\min}}^{W_{\max}} \delta_{W,y}}\right) \sum_{W=W_{\min}}^{W_{\max}} SC_{r,W,y} \times \delta_{W,y} .$$

The upper boundary of Region 6 is between rkm 99 and rkm 100. Therefore, the index of spatial distribution was defined as $\hat{F}_{6,y}$.

The spatial distribution indices are listed in Table B-10.

III. CORRELATION ANALYSES

A correlation analysis was conducted to identify significant correlations between (1) stressor indices and (2) indices of fish population response metrics. For each stressor, a set of relevant response variables was selected based on impact hypotheses and life history considerations. For example, zebra mussel effects were paired with the proportion of a population downriver of rkm 100, and temperature was paired with juvenile growth rate.

A correlation analysis was also conducted to identify significant correlations between (1) abundance indices and (2) indices of fish population response metrics. Relevant combinations of abundance and response metrics were selected based on impact hypotheses and life history considerations.

The correlation analyses were conducted using Spearman (rank) correlation coefficients to account for possible non-Normality of the indices. The correlation analyses were based on annual index values and were conducted separately for each species.

Results from the correlation analyses are summarized in Tables B-11 through B-26. Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

IV. LITERATURE CITED

Applied Science Associates, Inc. 2000. 1996 Year class report for the Hudson River estuary monitoring program ("Year Class Report"). Prepared for Central Hudson Gas & Electric Corporation.

Applied Science Associates, Inc. 2001. 1997 Year Class Report. Prepared for Central Hudson Gas & Electric Corporation.

ASA Analysis & Communication, Inc. 2001. 1998 Year Class Report. Prepared for Central Hudson Gas & Electric Corporation.

ASA Analysis & Communication, Inc. 2002. 1999 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

ASA Analysis & Communication, Inc. 2003. 2000 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

ASA Analysis & Communication, Inc. 2004a. 2001 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

ASA Analysis & Communication, Inc. 2004b. 2002 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

ASA Analysis & Communication, Inc. 2005. 2003 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

ASA Analysis & Communication, Inc. 2006. 2004 Year Class Report. Prepared for Dynegy Roseton L.L.C., Entergy Nuclear Indian Point 2 L.L.C., Entergy Nuclear Indian Point 3 L.L.C., and Mirant Bowline L.L.C.

Batelle New England Marine Research Laboratory. 1983. 1980 and 1981 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Caraco, N.F., Cole, J.J., Raymond, P.A., Strayer, D.L., Pace, M.L., Findlay, S.E.G., and Fischer, D.T. 1997. Zebra mussel invasion in a large, turbid river: phytoplankton response to increased grazing. Ecology 78: 588-602.

Central Hudson Gas & Electric Corp. et al. 1999. Draft Environmental Impact Statement for State Pollutant Discharge Elimination System Permits for Bowline Point, Indian Point 2 & 3, and Roseton Steam Electric Generating Stations.

Chambers, C and D Witting. 2005. Identifying environmental constraints on growth and survival of an anadromous gadid, Atlantic tomcod, at southern extreme of its geographic range. Ecological Society of America. 2005 Annual Meeting.

Consolidated Edison Company of New York, Inc. 1996. 1992 Year Class Report. New York, New York.

Consolidated Edison Company of New York, Inc. 1997a. 1993 Year Class Report. New York, New York.

Consolidated Edison Company of New York, Inc. 1997b. 1994 Year Class Report. New York, New York.

EA Engineering, Science, and Technology. 1990. 1988 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

EA Engineering, Science, and Technology. 1991. 1989 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

EA Engineering, Science, and Technology. 1996. 1995 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Lawler, Matusky & Skelly Engineers. 1989. 1986 and 1987 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Lawler, Matusky & Skelly Engineers. 1992. 1990 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Lawler, Matusky & Skelly Engineers. 1996. 1991 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Martin Marietta Environmental Systems. 1986. 1984 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Normandeau Associates, Inc. 1985a. 1982 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Normandeau Associates, Inc. 1985b. 1983 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Normandeau Associates, Inc. 2006. Abundance and stock characteristics of the Atlantic tomcod spawning population in the Hudson River, winter 2003-2004. Prepared for Entergy Nuclear Operations, Inc.

Pace, M.L., Findlay, S.E.G., and Fischer, D.T. 1998. Effects of an invasive bivalve on the zooplankton community of the Hudson River. Freshwat. Biol. 39: 103-116.

Seber, G.A.F. 1982. The estimation of animal abundance and related parameters, 2nd edition. Charles Griffin and Company Ltd., London.

Strayer, D.L., K.A. Hattala, and A.W. Kahnle. 2004. Effects of an invasive bivalve (*Dreissena polymorpha*) on fish in the Hudson River estuary. Can. J. Fish. Aquat. Sci. Vol. 61, 2004

Strayer, D.L., Powell, J., Ambrose, P., Smith, L.C., Pace, M.L., and Fischer, D.T. 1996. Arrival, spread, and early dynamics of a zebra mussel (*Dreissena polymorpha*) population in the Hudson River estuary. Can. J. Fish. Aquat. Sci. 53: 1143–1149.

Texas Instruments, Inc. 1977. 1974 Year Class Report for the multiplant impact study of the Hudson River Estuary ("Year Class Report"). Prepared for Consolidated Edison Company of New York, Inc.

Texas Instruments, Inc. 1978. 1975 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Texas Instruments, Inc. 1979. 1976 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Texas Instruments, Inc. 1980a. 1977 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Texas Instruments, Inc. 1980b. 1978 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Texas Instruments, Inc. 1981. 1979 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

Versar, Inc. 1987. 1985 Year Class Report. Prepared for Consolidated Edison Company of New York, Inc.

V. TABLES

Table B-1. Abundance Indices and Associated Standard Errors ("SE"), Based on Long River Survey Data.

Year Class	White Perch Post Yolk-Sac Larvae Index	SE	Striped Bass Post Yolk-Sac Larvae Index	SE	Bay Anchovy Post Yolk-Sac Larvae Index	SE	American Shad Post Yolk-Sac Larvae Index	SE
1974	139,139,531	9,461,494	116,793,360	14,525,520	9,111,556	2,155,940	32,149,174	5,436,351
1975	418,776,213	14,897,579	167,352,740	11,297,813	167,900,084	21,837,003	38,104,249	3,668,122
1976	571,765,805	26,442,918	55,463,017	3,014,531	341,602,306	88,340,964	30,532,518	4,411,773
1977	628,980,330	32,916,730	147,319,974	9,345,100	108,551,600	47,407,559	31,792,930	6,593,648
1978	852,286,248	54,375,932	113,088,409	9,188,267	13,499,413	2,574,305	14,808,830	1,725,494
1979	889,355,233	27,210,046	111,789,357	10,177,101	31,217,251	4,193,924	76,008,019	8,374,974
1980	731,972,701	29,071,443	193,067,215	15,374,877	282,472,131	47,526,524	62,624,636	6,850,621
1981	878,432,947	57,291,346	565,580,988	29,382,161	386,003,879	40,370,163	107,959,543	9,223,464
1982	1,533,952,669	63,678,126	214,574,357	17,311,853	7,721,685	1,434,887	105,866,404	11,668,608
1983	689,913,421	28,117,162	134,838,042	8,271,457	45,952,457	8,165,287	108,436,433	21,821,939
1984	659,480,715	40,337,372	200,167,635	28,656,262	39,045,805	11,944,143	46,171,178	7,590,296
1985	1,421,323,747	59,947,138	93,874,968	7,700,762	349,889,115	30,127,176	84,264,727	11,412,620
1986	2,052,461,814	98,317,198	171,163,020	8,998,325	118,354,834	10,883,362	152,128,084	17,215,544
1987	1,012,538,712	32,052,565	405,324,057	16,848,690	189,564,190	11,607,205	27,892,890	3,374,299
1988	754,305,782	42,580,552	351,072,816	35,669,346	152,035,433	30,786,324	78,027,604	11,883,534
1989	925,022,100	102,183,412	1,071,325,339	99,670,379	14,134,359	3,081,790	86,573,611	8,951,649
1990	768,296,570	79,095,729	1,295,596,696	153,298,294	890,027	256,957	108,278,134	14,347,189
1991	907,921,874	61,907,978	1,896,058,025	203,606,883	5,602,678,703	551,771,800	43,259,681	5,089,006
1992	1,211,029,021	53,752,949	1,436,836,717	103,392,955	77,338,304	10,339,754	99,755,719	15,257,291
1993	1,231,794,687	50,130,673	2,008,989,233	181,226,826	573,839,976	50,894,605	33,386,515	6,848,737
1994	1,043,697,036	46,808,643	2,009,527,814	204,188,984	583,968,501	47,054,442	37,913,769	3,901,481
1995	623,420,693	29,028,682	939,209,970	99,781,400	839,521,735	64,631,235	24,920,433	3,668,256
1996	1,505,193,548	83,865,093	3,629,518,187	365,724,596	405,338,653	43,811,932	31,112,517	3,986,134
1997	307,236,756	17,277,642	1,252,166,315	211,669,199	1,009,992,702	213,235,143	19,546,174	4,202,344
1998	575,146,100	35,729,754	1,413,117,919	122,712,647	18,860,574	3,243,002	10,840,582	1,389,788
1999	673,636,250	39,842,187	3,468,043,472	358,992,219	287,637,139	29,957,432	19,920,980	4,244,449
2000	1,180,789,474	133,501,704	5,803,754,734	715,393,543	1,355,732	345,802	10,158,022	1,432,512
2001	734,730,398	61,307,779	5,258,385,169	340,997,297	51,298,063	22,554,315	48,974,089	9,013,780
2002	566,273,447	39,302,719	587,019,561	40,128,197	173,651,942	21,508,231	11,487,215	2,321,455
2003	692,003,842	45,947,390	1,853,946,447	202,927,363	6,523,373	2,802,470	11,636,329	1,626,253
2004	721,129,750	39,776,443	1,646,077,551	106,676,037	717,812,470	71,311,509	13,196,538	1,966,124

Table 1. Abundance Indices and Associated Standard Errors ("SE"), Based on Long River Survey Data (continued).

Year Class	River Herring Post Yolk-Sac Larvae Index	SE	Atlantic Tomcod Post Yolk-Sac Larvae Index	SE
1974	1,925,093,580	1,073,772,004	128,306,743	19,426,263
1975	2,177,549,296	197,088,426	67,024,707	19,768,962
1976	1,590,931,203	156,327,051	42,777,042	13,470,065
1977	1,789,369,237	309,551,598	164,621,663	70,515,234
1978	2,483,545,195	230,530,412	54,313,088	10,307,482
1979	1,492,563,623	65,281,612	18,127,435	3,099,375
1980	1,451,864,997	82,238,743	95,402,234	13,128,146
1981	2,097,039,055	238,479,765	74,140,778	13,052,007
1982	2,761,588,726	248,286,854	28,419,800	7,665,326
1983	3,398,542,430	247,313,066	42,683,202	8,311,722
1984	2,263,857,937	168,138,864	147,133,069	25,916,525
1985	2,360,908,396	138,470,331	109,664,584	11,132,251
1986	3,060,453,736	212,481,475	53,404,268	4,770,519
1987	945,121,604	62,594,106	138,570,516	12,594,732
1988	1,205,794,912	101,740,608	78,376,300	10,680,903
1989	1,515,234,476	181,441,810	185,450,859	23,858,579
1990	1,296,493,803	106,557,985	107,915,374	25,158,013
1991	1,105,840,600	89,654,766	116,333,462	14,859,973
1992	1,592,451,980	119,021,893	32,021,214	4,889,565
1993	957,005,646	76,057,902	126,394,886	20,139,893
1994	1,006,699,048	57,426,960	85,456,373	22,227,930
1995	745,594,402	44,387,051	79,816,881	6,641,688
1996	2,092,537,070	119,641,340	51,571,386	5,696,759
1997	338,336,798	21,073,725	110,409,961	28,829,551
1998	599,669,094	37,989,853	53,594,909	8,409,591
1999	658,448,983	38,493,738	17,392,702	2,076,588
2000	1,736,751,090	110,473,230	11,120,807	1,442,773
2001	941,430,470	69,923,386	93,816,691	8,320,053
2002	798,010,496	43,842,607	4,382,650	649,979
2003	608,369,228	39,023,677	38,715,789	3,683,762
2004	681,555,090	40,476,571	115,401,578	16,005,570

Table B-2. Abundance Indices and Associated Standard Errors ("SE"), Based on Beach Seine Survey Data.

Year	White Perch Young-of-Year		Striped Bass Young-of-Year		Spottail Shiner Young-of-Year		Spottail Shiner Egg	
Class	Index	SE	Index	SE	Index	SE	Index	SE
1974	566,346	61,280	1,373,138	264,598	658,945	87,448	1,128,997	107,867
1975	2,342,937	440,999	1,367,496	242,374	1,286,297	193,361	1,578,455	195,841
1976	1,944,220	255,910	864,743	70,734	1,324,434	203,989	0	0
1977	953,799	87,722	1,375,537	124,595	495,690	66,445	0	0
1978	2,675,700	402,374	3,042,920	614,048	1,363,313	148,541	0	0
1979	2,921,393	285,862	794,022	91,389	956,236	97,330	0	0
1980	1,884,895	231,650	1,265,254	147,121	633,323	72,196	312,488	80,635
1981	1,862,222	160,903	1,827,767	152,481	1,865,058	216,442	627,176	96,220
1982	1,967,754	287,490	934,550	97,768	477,090	62,605	173,130	25,821
1983	1,803,266	399,823	1,642,536	191,103	1,070,822	104,909	197,639	51,127
1984	703,959	145,133	1,300,754	173,872	616,182	128,367	222,054	41,973
1985	757,003	82,536	238,259	21,226	543,246	66,532	116,419	17,690
1986	1,036,321	97,303	298,745	31,415	388,736	69,297	276,641	48,687
1987	1,169,236	121,876	2,976,381	314,807	470,267	74,827	234,226	45,133
1988	1,738,310	255,364	1,172,303	68,239	419,874	49,588	276,581	49,087
1989	1,105,280	278,101	1,238,434	116,464	623,204	95,526	272,136	61,641
1990	588,162	75,727	1,486,911	89,409	808,662	101,694	144,012	31,435
1991	580,165	76,201	1,125,126	64,076	855,292	110,557	833,354	126,276
1992	463,555	53,444	1,046,654	53,265	726,888	124,009	453,069	112,051
1993	806,848	97,157	1,640,132	90,969	655,117	95,425	391,317	97,925
1994	315,662	39,618	1,136,106	63,179	1,624,997	289,784	168,358	27,009
1995	425,062	49,042	1,404,935	89,202	603,130	94,204	229,394	41,809
1996	44,925	10,283	299,997	30,506	174,026	39,053	58,663	15,101
1997	571,160	114,812	1,892,597	169,399	1,197,799	170,583	140,490	33,758
1998	270,835	51,992	1,384,364	85,327	273,165	53,055	147,082	40,400
1999	1,411,184	169,447	1,715,282	142,568	2,040,399	243,244	154,889	21,463
2000	304,950	52,787	580,006	52,449	303,081	52,956	164,945	29,160
2001	1,019,516	119,666	2,392,216	170,860	2,143,066	610,761	16,919	5,028
2002	699,145	80,612	1,145,686	60,295	1,132,479	146,862	174,197	50,311
2003	2,177,013	228,303	2,282,684	118,276	2,102,568	257,006	565,369	131,279
2004	632,961	89,075	807,661	70,743	1,031,399	152,802	436,330	79,667

Table 2. Abundance Indices and Associated Standard Errors ("SE"), Based on Beach Seine Survey Data (continued).

Year Class	Alewife Young-of-Year Index	Alewife Young-of-Year SE	Blueback Herring Young-of-Year Index	Blueback Herring Young-of-Year SE
1974	583,238	74,805	3,647,758	502,857
1975	572,550	107,585	10,888,524	1,249,788
1976	352,263	96,375	21,621,271	3,075,761
1977	517,792	49,081	31,795,371	4,717,652
1978	1,027,891	174,698	22,993,451	4,200,939
1979	340,271	59,099	8,221,314	1,461,758
1980	93,783	17,894	8,892,467	2,207,337
1981	477,348	84,403	32,066,440	9,586,015
1982	116,606	24,817	10,164,307	1,750,817
1983	214,922	42,154	16,326,879	2,278,723
1984	49,776	10,864	3,577,323	786,742
1985	119,509	22,024	3,323,511	664,762
1986	119,468	48,899	1,555,182	357,032
1987	80,611	13,768	6,188,101	773,111
1988	87,080	15,727	5,887,963	1,008,925
1989	43,711	12,956	3,230,116	497,839
1990	157,159	25,580	9,436,487	1,274,900
1991	335,535	63,111	3,530,392	596,059
1992	40,507	9,371	6,642,282	1,599,250
1993	69,438	11,826	4,234,168	531,496
1994	148,030	30,079	9,584,696	1,308,960
1995	91,731	22,716	3,202,735	892,613
1996	47,371	14,912	4,044,353	890,186
1997	291,323	54,177	12,075,530	2,541,612
1998	40,865	30,194	155,761	32,365
1999	445,167	79,622	5,691,570	776,702
2000	76,445	37,606	2,342,499	572,561
2001	330,876	70,451	5,268,663	704,402
2002	60,954	13,491	1,438,577	299,230
2003	452,292	87,223	10,203,281	1,459,824
2004	218,118	35,902	5,091,421	620,888

Table B-3. Abundance Indices and Associated Standard Errors ("SE"), Based on Fall Shoals Survey and Atlantic Tomcod Mark Recapture Data.

Year Class	Bay Anchovy Young-of-Year (FSS) Index	SE	American Shad Young-of-Year (FSS) Index	SE	Atlantic Tomcod Ages 1 and 2 (ATMR) Index	SE
1974	-	-	-	-	3,666,156.2	667,339
1975	-	-	-	-	3,680,086.9	375,142
1976	-	-	-	-	19,210,329.2	2,767,571.7
1977	-	-	-	-	2,434,397.0	458,488.1
1978	-	-	-	-	5,894,583.8	917,687.4
1979	-	-	-	-	9,128,535	1,692,155.4
1980	-	-	-	-	4,747,440	3,355,405.2
1981	-	-	-	-	25,066,665.0	14,468,003
1982	-	-	-	-	12,983,676.9	2,899,705
1983	-	-	-	-	6,657,331.2	1,302,504.2
1984	-	-	-	-	-	-
1985	218,612,898	21,269,766	1,591,435	190,139	2,093,677	171,796
1986	132,925,173	13,133,411	3,104,605	640,844	-	-
1987	246,910,112	26,982,497	647,070	157,299	3,526,907.2	570,280
1988	422,678,791	38,213,532	997,871	144,252	5,897,656.7	524,801.4
1989	349,952,337	26,107,654	2,754,815	198,752	6,804,809.4	1,239,300.2
1990	161,039,442	14,450,450	1,139,272	235,276	3,208,815.0	615,208.4
1991	190,474,265	11,540,891	680,209	72,781	388,763.0	84,175.2
1992	185,902,303	13,738,226	1,306,732	147,744	2,553,778.3	319,857.2
1993	249,913,241	19,475,645	464,702	48,446	663,439.1	155,295.9
1994	206,642,043	14,141,476	1,036,782	88,932	2,384,183	659,618.4
1995	439,617,793	28,732,239	471,444	75,896	88,492.5	50,523.4
1996	102,941,191	5,959,974	2,859,373	451,439	3,277,909.3	1,637,090
1997	283,382,412	17,014,202	913,970	107,851	1,291,980.5	302,916.5
1998	189,541,611	9,166,785	232,260	56,459	592,891.0	241,105.3
1999	165,375,818	9,972,244	853,411	135,639	181,179.0	59,983.3
2000	57,208,944	3,577,181	878,405	100,807	2,504,266	624,327.3
2001	109,701,139	8,052,515	1,006,787	162,014	40,875	28,743.1
2002	171,692,430	10,652,063	497,537	57,524	108,528.0	76,363
2003	148,898,706	11,753,477	351,278	47,131	1,653,319	425,310
2004	218,178,981	17,899,774	336,973	63,105	-	-

Table B-4. Estimates of Indian Point Conditional Mortality Rate (CMR) for entrainment.

Year Class	White Perch CMR	Striped Bass CMR	Spottail Shiner CMR	Bay Anchovy CMR	American Shad CMR	River Herring CMR	Atlantic Tomcod CMR
1974	7.45	5.65	0.87	7.31	0.22	0.83	3.65
1975	8.65	7.78	1.04	6.61	0.35	1.42	6.75
1976	3.22	4.73	1.38	3.45	0.33	1.85	8.76
1977	7.27	13.89	1.41	13.78	0.38	2.47	10.15
1978	5.28	8.55	2.32	12.54	0.24	1.26	10.6
1979	8.02	11.92	1.62	10.8	0.2	2.24	18.8
1980	3.36	11.87	1.66	18.44	0.03	0.48	25.47
1981	6.54	4.17	3.43	18.56	0.2	0.57	11.68
1982	4.33	6.99	2.06	4.19	0.44	0.81	17.47
1983	17.23	7.36	3.17	9.04	0.09	3.05	7.69
1984	8.92	17.25	1.58	6.26	7.5	5.34	16.58
1985	0.55	3.97	1.77	10.06	0	0.02	34.5
1986	4.07	16.26	1.55	5.07	3.56	0.92	11.36
1987	0.66	2.3	1.53	9.99	0	0.04	14.61
1988	7.94	11.63	4.1	17.73	0.15	0.51	23.94
1989	4.03	5.96	8.32	7.96	0.28	1.41	4.49
1990	3.48	6.12	2.18	20.85	0.43	2.94	5.52
1991	1.4	4.95	3.92	9.09	0.07	0.41	6.99
1992	2.7	6.16	0.99	7.12	0.05	0.41	14.11
1993	2.34	5.6	0.89	7.08	0.13	0.23	3.67
1994	3.14	6.81	1.1	5.94	0.12	0.49	7.57
1995	1.92	4.22	2.54	14.99	0.1	0.12	5.77
1996	4.88	12.01	1.89	15.55	0.42	0.49	8.47
1997	1.29	1.42	0.64	6.62	0.05	0.6	10.35
1998	4.87	8.46	0.45	7.82	0.12	0.59	10.01
1999	4.16	11.35	2.57	13.81	0.23	3.66	21.54
2000	7.31	4.03	1.63	7.77	1.86	4	11.23
2001	5.69	8	2.56	15.4	0.3	1.82	20.97
2002	11.96	13.77	3.03	10.57	1.23	4.84	23.25
2003	7.67	12.26	1.21	12.97	0.19	1.85	20.43
2004	-	-	-	-	-	-	-

Table B-5. Zebra Mussel Index.

Year Class	Zebra Mussel Index
1974	0
1975	0
1976	0
1977	0
1978	0
1979	0
1980	0
1981	0
1982	0
1983	0
1984	0
1985	0
1986	0
1987	0
1988	0
1989	0
1990	0
1991	0
1992	0
1993	1
1994	1
1995	1
1996	1
1997	1
1998	1
1999	1
2000	1
2001	1
2002	1
2003	1
2004	1

Table B-6. Striped Bass Predation Index.

Year Class	Striped Bass PYSL Index
1974	116,793,360
1975	167,352,740
1976	55,463,017
1977	147,319,974
1978	113,088,409
1979	111,789,357
1980	193,067,215
1981	565,580,988
1982	214,574,357
1983	134,838,042
1984	200,167,635
1985	93,874,968
1986	171,163,020
1987	405,324,057
1988	351,072,816
1989	1,071,325,339
1990	1,295,596,696
1991	1,896,058,025
1992	1,436,836,717
1993	2,008,989,233
1994	2,009,527,814
1995	939,209,970
1996	3,629,518,187
1997	1,252,166,315
1998	1,413,117,919
1999	3,468,043,472
2000	5,803,754,734
2001	5,258,385,169
2002	587,019,561
2003	1,853,946,447
2004	1,646,077,551

Table B-7. Temperature Indices.

Year Class	FSS Temperature Index	PWW Degree-Day Index
1974	-	
1975	-	
1976	-	18.8
1977	-	57.7
1978	-	60.8
1979	22.5	61.3
1980	22.4	128.1
1981	19.8	98.0
1982	-	64.3
1983	24.0	107.9
1984	22.8	91.2
1985	21.5	63.1
1986	21.5	61.1
1987	19.9	111.1
1988	24.6	121.1
1989	22.2	65.2
1990	22.7	68.4
1991	21.5	108.9
1992	20.2	6.5
1993	22.2	97.1
1994	22.2	103.6
1995	22.6	94.9
1996	22.3	28.6
1997	22.4	63.7
1998	23.5	94.1
1999	23.2	136.8
2000	21.7	0.9
2001	23.1	98.9
2002	23.5	121.6
2003	22.6	106.8
2004	22.5	18.8

Table B-8. Survival Indices.

Year Class	White Perch PYSL to YOY Index	Striped Bass PYSL to YOY Index	Spottail Shiner Egg to YOY Index	Bay Anchovy PYSL to YOY Index	American Shad PYSL to YOY Index	River Herring PYSL to YOY Index	Atlantic Tomcod Egg to Age-1 Index	Atlantic Tomcod Age-1 to Age-2 Index
1974	0.0041	0.0118	0.5837	-	-	0.0030	-	-
1975	0.0056	0.0082	0.8149	-	-	0.0053	-	0.2008
1976	0.0034	0.0156	-	-	-	0.0138	0.4411	0.0103
1977	0.0015	0.0093	-	-	-	0.0184	0.0371	0.0249
1978	0.0031	0.0269	-	-	-	0.0100	0.2826	0.0460
1979	0.0033	0.0071	-	-	-	0.0077	0.1731	-
1980	0.0026	0.0066	2.0267	-	-	0.0064	-	-
1981	0.0021	0.0032	2.9737	-	-	0.0155	-	-
1982	0.0013	0.0044	2.7557	-	-	0.0039	-	0.0699
1983	0.0026	0.0122	5.4181	-	-	0.0049	0.0613	-
1984	0.0011	0.0065	2.7749	-	-	0.0018	-	-
1985	0.0005	0.0025	4.6663	0.6248	0.0189	0.0015	-	-
1986	0.0005	0.0017	1.4052	1.1231	0.0204	0.0006	-	-
1987	0.0012	0.0073	2.0077	1.3025	0.0232	0.0068	-	0.2014
1988	0.0023	0.0033	1.5181	2.7801	0.0128	0.0050	0.1235	0.3714
1989	0.0012	0.0012	2.2900	24.7590	0.0318	0.0023	0.1186	0.1251
1990	0.0008	0.0011	5.6152	180.9377	0.0105	0.0084	0.0298	0.0448
1991	0.0006	0.0006	1.0263	0.0340	0.0157	0.0035	0.0055	1.3636
1992	0.0004	0.0007	1.6044	2.4038	0.0131	0.0042	0.3153	0.1078
1993	0.0007	0.0008	1.6741	0.4355	0.0139	0.0045	0.0154	0.4661
1994	0.0003	0.0006	9.6520	0.3539	0.0273	0.0097	0.3110	-
1995	0.0007	0.0015	2.6292	0.5237	0.0189	0.0044	-	-
1996	0.0000	0.0001	2.9665	0.2540	0.0919	0.0043	-	0.2314
1997	0.0019	0.0015	8.5259	0.2806	0.0468	0.0366	0.0148	0.2933
1998	0.0005	0.0010	1.8572	10.0496	0.0214	0.0003	0.0173	0.1004
1999	0.0021	0.0005	13.1733	0.5749	0.0428	0.0093	0.0160	1.0951
2000	0.0003	0.0001	1.8375	42.1978	0.0865	0.0015	0.7792	-
2001	0.0014	0.0005	126.6690	2.1385	0.0206	0.0065	-	-
2002	0.0012	0.0020	6.5011	0.9887	0.0433	0.0019	-	-
2003	0.0031	0.0012	3.7189	22.8254	0.0302	0.0186	-	-
2004	0.0009	0.0005	2.3638	0.3039	0.0255	0.0079	-	-

Table B-9. Growth Rate Indices

Year Class	White Perch Index	Striped Bass Index	Spottail Shiner Index	Bay Anchovy Index	American Shad Index	Alewife Index	Blueback Herring Index
1974	0.0972	0.0727	0.0844	-	-	0.0265	0.0810
1975	0.0605	0.0495	0.0624	-	-	0.0420	0.0563
1976	0.0873	0.0542	-	-	-	-	-
1977	-	-	-	-	-	-	-
1978	-	-	-	-	-	-	-
1979	0.0725	0.0697	0.0768	-	-	0.0571	0.0894
1980	0.0790	0.0729	0.0742	-	-	0.0337	0.0658
1981	0.0578	0.0501	0.0651	-	-	0.0350	0.0632
1982	0.0769	0.0460	0.0733	-	-	0.0454	0.0591
1983	0.0845	0.0919	0.1417	-	-	0.0916	0.1037
1984	0.1142	0.0942	0.0824	-	-	0.0752	0.0669
1985	0.0611	0.1245	0.0520	0.0288	0.0234	0.0525	0.0304
1986	0.0640	0.0433	0.0534	0.0703	0.0716	0.0459	0.0604
1987	0.0750	0.0685	0.0864	0.0311	0.0466	0.0630	0.0555
1988	0.0589	0.0532	0.0691	0.0928	0.0813	0.0520	0.0573
1989	0.0973	0.0712	0.0788	0.0870	0.0661	0.0815	0.0858
1990	0.1081	0.0866	0.0998	0.1000	0.0711	0.0585	0.0603
1991	0.0620	0.0591	0.0552	0.0505	0.0572	0.0510	0.0808
1992	0.0933	0.0840	0.0616	0.0617	0.0759	0.0412	0.0581
1993	0.0732	0.0589	0.0621	0.0475	0.0346	0.0271	0.0200
1994	0.0362	0.0372	0.0502	0.0890	0.0546	0.0425	0.0204
1995	0.1088	0.0823	0.0793	0.0668	0.0460	0.0471	0.0845
1996	0.1073	0.1070	0.1168	0.0642	0.0853	0.0729	0.0384
1997	0.0764	0.0657	0.0716	0.0997	0.0756	0.0461	0.0322
1998	0.0813	0.0802	0.0603	0.0732	0.0520	0.0670	0.0454
1999	0.0457	0.0671	0.0414	0.0256	0.0320	0.0086	0.0316
2000	0.0813	0.0773	0.0732	0.0781	0.0824	0.0797	0.0610
2001	0.0961	0.0652	0.0978	0.0763	0.0637	0.0710	0.0686
2002	0.0624	0.0625	0.0637	0.0400	0.0445	0.0366	0.0982
2003	0.0732	0.0517	0.0863	0.0841	0.0493	0.0536	0.0465
2004	0.0515	0.0474	0.0592	0.1006	0.0601	0.0411	0.0715

Table B-10. Spatial Distribution Indices -- The Fraction of Standing Crop that is Downriver of rkm 100.

Year	White Perch		Striped Bass		Spottail Shiner	Bay Anchovy		American Shad	
Class	PYSL Index	YOY Index	PYSL Index	YOY Index	YOY Index	PYSL Index	YOY Index	PYSL Index	YOY Index
1974	0.4102	0.3501	0.6199	0.8947	0.0783	1.0000	-	0.0209	-
1975	0.4373	0.7000	0.7998	0.9192	0.0772	1.0000	-	0.1802	-
1976	0.1782	0.5473	0.7834	0.9109	0.1804	1.0000	-	0.0380	-
1977	0.2008	0.3872	0.7088	0.8765	0.0668	0.9999	-	0.0139	-
1978	0.2638	0.6703	0.8044	0.9554	0.1594	1.0000	-	0.0274	-
1979	0.3384	0.6210	0.8876	0.9027	0.2137	1.0000	-	0.0351	-
1980	0.2276	0.6592	0.7788	0.8260	0.0709	0.9998	-	0.0198	-
1981	0.2585	0.6813	0.5834	0.9247	0.0874	0.9998	-	0.0267	-
1982	0.3628	0.7975	0.8013	0.9668	0.2880	1.0000	-	0.0461	-
1983	0.4220	0.5556	0.8632	0.8634	0.1347	0.9997	-	0.0293	-
1984	0.2366	0.7919	0.8475	0.9402	0.0794	0.9997	-	0.3433	-
1985	0.1420	0.6204	0.6800	0.9004	0.0749	0.9982	0.8978	0.0015	0.3707
1986	0.2147	0.7541	0.8164	0.9115	0.0962	1.0000	0.9178	0.0104	0.1426
1987	0.0984	0.4309	0.4985	0.9110	0.0145	0.9964	0.9547	0.0012	0.1960
1988	0.3191	0.7514	0.7726	0.8233	0.1086	0.9249	0.8584	0.0032	0.3732
1989	0.4646	0.7267	0.7884	0.9188	0.1493	0.9557	0.8974	0.1272	0.1777
1990	0.3406	0.4131	0.5434	0.8682	0.0743	1.0000	0.9365	0.0539	0.3500
1991	0.2109	0.3581	0.7037	0.6287	0.0165	0.9835	0.6000	0.0036	0.2074
1992	0.2616	0.5105	0.8321	0.8619	0.0344	0.9964	0.8679	0.0154	0.3391
1993	0.1911	0.3349	0.7026	0.8189	0.0593	0.9966	0.7392	0.0029	0.2788
1994	0.2156	0.4619	0.8595	0.8084	0.0767	0.9995	0.9240	0.0077	0.3255
1995	0.2054	0.3869	0.7445	0.8986	0.0143	0.9888	0.7635	0.0049	0.3529
1996	0.1587	0.7707	0.7570	0.7614	0.1261	0.9978	0.9603	0.0062	0.2600
1997	0.2799	0.4857	0.8852	0.8555	0.0774	1.0000	0.8117	0.0078	0.1259
1998	0.2646	0.5741	0.8162	0.8603	0.0351	0.9986	0.8190	0.0202	0.0674
1999	0.1919	0.6035	0.7352	0.7392	0.0220	0.9987	0.8487	0.0235	0.2024
2000	0.6546	0.5040	0.9908	0.7759	0.1723	0.9797	0.8889	0.1399	0.2930
2001	0.1508	0.4677	0.7024	0.8177	0.0193	1.0000	0.9302	0.0438	0.2072
2002	0.2851	0.2743	0.8712	0.7682	0.0008	1.0000	0.7100	0.0879	0.0657
2003	0.3001	0.4981	0.8249	0.8803	0.0572	1.0000	0.9507	0.0132	0.1721
2004	0.2150	0.1672	0.8196	0.6875	0.0407	0.9997	0.9363	0.0364	0.1225

Table 10. Spatial Distribution Indices -- The Fraction of Standing Crop that is Downriver of rkm 100 (continued).

Year Class	Alewife PYSL Index*	Alewife YOY Index	Blueback Herring PYSL Index*	Blueback Herring YOY Index	Atlantic Tomcod PYSL Index
1974	0.0448	0.9065	0.0448	0.2928	0.9903
1975	0.0650	0.8709	0.0650	0.1996	0.9902
1976	0.1571	0.6064	0.1571	0.1818	0.9912
1977	0.0575	0.5622	0.0575	0.4164	0.9953
1978	0.0985	0.5909	0.0985	0.1202	0.9854
1979	0.1189	0.4444	0.1189	0.1452	0.9860
1980	0.0193	0.5528	0.0193	0.0663	0.9528
1981	0.0844	0.4460	0.0844	0.3646	0.9853
1982	0.0704	0.7575	0.0704	0.2143	0.9663
1983	0.1715	0.2247	0.1715	0.1088	0.9960
1984	0.2939	0.3330	0.2939	0.2982	0.9778
1985	0.0086	0.4559	0.0086	0.3012	0.9496
1986	0.0776	0.3842	0.0776	0.1475	0.9741
1987	0.0077	0.3363	0.0077	0.2725	0.8921
1988	0.0545	0.7762	0.0545	0.2218	0.9609
1989	0.0894	0.7374	0.0894	0.1058	0.9980
1990	0.1879	0.4526	0.1879	0.0988	0.9712
1991	0.0228	0.0304	0.0228	0.0101	0.9837
1992	0.0595	0.4622	0.0595	0.5121	0.9976
1993	0.0097	0.2508	0.0097	0.2744	0.9950
1994	0.0265	0.5730	0.0265	0.3236	0.9915
1995	0.0184	0.1994	0.0184	0.1357	0.9411
1996	0.0186	0.4721	0.0186	0.6749	0.9852
1997	0.1830	0.2906	0.1830	0.0769	0.9935
1998	0.0448	0.8889	0.0448	0.0846	0.9928
1999	0.1857	0.2304	0.1857	0.2034	0.9732
2000	0.2224	0.1696	0.2224	0.1666	0.9024
2001	0.0698	0.1830	0.0698	0.0800	0.9721
2002	0.2350	0.0914	0.2350	0.3404	0.9938
2003	0.1196	0.5519	0.1196	0.2539	0.9934
2004	0.1376	0.5527	0.1376	0.1861	0.9849

Table B-11. Striped Bass

Response Metric	Stressor: Indian Point Entrainment Mortality (CMR)	Stressor: Zebra Mussels	Stressor: Striped Bass Predation	Stressor: Temperature	Yearclass
PYSL-to-YOY Survival		**-0.69**			**-0.84**
PYSL Abundance					**+0.84**
YOY Abundance					
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100		**-0.63**			**-0.68**
Yearclass		**+0.84**	**+0.84**		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-12. Striped Bass

Response Metric	Response Metric: PYSL-to-YOY Survival	PYSL Abundance	YOY Abundance	Yearclass
PYSL-to-YOY Survival				**-0.84**
PYSL Abundance				**+0.84**
YOY Abundance				
Yearclass	**-0.84**	**+0.84**		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-13. White Perch

Response Metric	Stressor				Yearclass
	Indian Point Entrainment Mortality (CMR)	Zebra Mussels	Striped Bass Predation	Temperature	
PYSL-to-YOY Survival	**+0.44**	**-0.36**	**-0.57**	**+0.42**	**-0.53**
PYSL Abundance	-0.34				
YOY Abundance			**-0.54**		**-0.51**
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100		**-0.40**			**-0.37**
Yearclass		**+0.84**	**+0.84**		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-14. White Perch

Response Metric	Response Metric: PYSL-to-YOY Survival	Response Metric: PYSL Abundance	Response Metric: YOY Abundance	Yearclass
PYSL-to-YOY Survival			+0.76	-0.53
PYSL Abundance				
YOY Abundance	+0.76			-0.51
Yearclass	-0.53		-0.51	

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-15. American Shad

Response Metric	Stressor				Yearclass
	Indian Point Entrainment Mortality (CMR)	Zebra Mussels	Striped Bass Predation	Temperature	
PYSL-to-YOY Survival		**+0.58**			**+0.55**
PYSL Abundance			-0.31		**-0.46**
YOY Abundance					**-0.57**
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100					**-0.48**
Yearclass		**+0.84**	**+0.84**		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-16. American Shad

Response Metric	Response Metric: PYSL-to-YOY Survival	PYSL Abundance	YOY Abundance	Yearclass
PYSL-to-YOY Survival				**+0.55**
PYSL Abundance			**+0.75**	**-0.46**
YOY Abundance		**+0.75**		**-0.57**
Yearclass	**+0.55**	**-0.46**	**-0.57**	

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-17. Atlantic Tomcod

Response Metric	Stressor: Indian Point Entrainment Mortality (CMR)	Stressor: Zebra Mussels	Stressor: Striped Bass Predation	Stressor: Temperature (PWW degree-days)	Yearclass/ Year
Egg-to-Age1 Survival				-0.59	
Age1-to-Age2 Survival			+0.45		+0.56
Egg Abundance					-0.42
PYSL Abundance	-0.36				
Age1 Abundance			-0.65		-0.72
% PYSL Downriver of rkm 100					
Yearclass/ Year		+0.84	+0.84		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-18. Atlantic Tomcod

Response Metric	Response Metric				Yearclass/ Year
	Egg-to-Age1 Survival	Age1-to-Age2 Survival	Egg Abundance	Age1 Abundance	
Egg-to-Age1 Survival				**+0.61**	
Age1-to-Age2 Survival					**+0.56**
Egg Abundance					-0.42
Age1 Abundance	**+0.61**				**-0.72**
Yearclass/ Year		**+0.56**	-0.42	**-0.72**	

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-19. Alewife

Response Metric	Stressor: Indian Point Entrainment Mortality (CMR)	Stressor: Zebra Mussels	Stressor: Striped Bass Predation	Stressor: Temperature	Yearclass
PYSL-to-YOY Survival					
PYSL Abundance			-0.56		-0.70
YOY Abundance			-0.34		-0.40
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100		-0.33			-0.45
Yearclass		+0.84	+0.84		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-20. Alewife

Response Metric	Response Metric			Yearclass
	PYSL-to-YOY Survival	PYSL Abundance	YOY Abundance	
PYSL-to-YOY Survival				
PYSL Abundance				**-0.70**
YOY Abundance				**-0.40**
Yearclass		**-0.70**	**-0.40**	

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-21. Blueback Herring

Response Metric	Stressor				Yearclass
	Indian Point Entrainment Mortality (CMR)	Zebra Mussels	Striped Bass Predation	Temperature	
PYSL-to-YOY Survival					
PYSL Abundance			**-0.56**		**-0.70**
YOY Abundance			**-0.31**		**-0.45**
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100					
Yearclass		**+0.84**	**+0.84**		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-22. Blueback Herring

Response Metric	Response Metric: PYSL-to-YOY Survival	Response Metric: PYSL Abundance	Response Metric: YOY Abundance	Yearclass
PYSL-to-YOY Survival				
PYSL Abundance				**-0.70**
YOY Abundance				**-0.45**
Yearclass		**-0.70**	**-0.45**	

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-23. Bay Anchovy

Response Metric	Stressor				Yearclass
	Indian Point Entrainment Mortality (CMR)	Zebra Mussels	Striped Bass Predation	Temperature	
PYSL-to-YOY Survival					
PYSL Abundance					
YOY Abundance			-0.53		
YOY Growth Rate					
% PYSL Downriver of rkm 100					
% YOY Downriver of rkm 100					
Yearclass		+0.84	+0.84		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-24. Bay Anchovy

Response Metric	Response Metric			Yearclass
	PYSL-to-YOY Survival	PYSL Abundance	YOY Abundance	
PYSL-to-YOY Survival				
PYSL Abundance				
YOY Abundance				
Yearclass				

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-25. Spottail Shiner

Response Metric	Stressor				Yearclass
	Indian Point Entrainment Mortality (CMR)	Zebra Mussels	Striped Bass Predation	Temperature	
Egg-to-YOY Survival		+0.42		+0.38	+0.40
PYSL Abundance					
YOY Abundance					
YOY Growth Rate					
% YOY Downriver of rkm 100		-0.40			-0.51
Yearclass		+0.84	+0.84		

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

Table B-26. Spottail Shiner

Response Metric	Response Metric			Yearclass
	Egg-to-YOY Survival	PYSL Abundance	YOY Abundance	
Egg-to-YOY Survival				**+0.40**
PYSL Abundance				
YOY Abundance				
Yearclass	**+0.40**			

Correlation coefficients significant at the 0.05 level are printed in black and correlation coefficients significant at the 0.10 level are printed in gray. A blank cell in the table indicates that the correlation coefficient was not significant at a probability level of 0.10 or lower. Cells shaded gray indicate pairs of indices that were not considered relevant, based on impact hypothesis and/or life history considerations.

APPENDIX C

Potential Effects of Striped Bass Predation on

Juvenile Fish in the Hudson River

Douglas G. Heimbuch

AKRF, Inc.

Hanover, Maryland, U.S.A. 21076

dgheimbuch@verizon.net

Abstract

This study addressed the question of whether the increase in striped bass (*Morone saxatilis*) abundance in the Hudson River that began after 1990, and the associated increase in predatory demand, could have been responsible for observed declines in juvenile abundance of river herring (i.e., blueback herring (*Alosa aestivalis*) and alewife (*Alosa pseudoharengus*)), Atlantic tomcod (*Microgadus tomcod*) and white perch (*Morone americana*), and the apparent decline in juvenile survival of striped bass, in the Hudson River. Seasonal (August through October) predatory demand of Hudson River striped bass (ages 1 through 13) was estimated to have increased from an average of 3.4 million kg yr^{-1} for the period 1982-1990 to an average of 15.0 million kg yr^{-1} for the period 1991-2004. Juvenile river herring average abundance declined 60% since 1990, juvenile Atlantic tomcod average abundance declined 69%, juvenile white perch average abundance declined 59%, and juvenile striped bass survival declined 87%. It was estimated that the observed declines in juvenile abundance and the apparent decline in striped bass juvenile survival could be explained by the increase in striped bass predatory demand if: 1) 3.3% of the seasonal predatory demand of age 1 through age 13 Hudson River striped bass was satisfied by consumption of juveniles of the four taxa, or 2) 11.1% of the seasonal predatory demand of age 1 and age 2 Hudson River striped bass was satisfied by consumption of juveniles of the four taxa. Historical information on the fraction of the Hudson River striped bass stock that inhabits the Hudson River from August through October, combined with historical information on dietary preferences of Hudson River striped bass, appear consistent with these levels of consumption.

Introduction

Background

The Atlantic coast population of striped bass (*Morone saxatilis*) experienced a major increase in abundance over the past decade in response to changes in fishery regulation (Richards and Rago 1999). The average biomass of the population (age 1 and older) increased over five-fold from 16,800,000 kg to 87,900,000 kg for the period 1983-1990 to the period 1991-2004 (ASMFC 2005). The increase in abundance of the population raised concerns that the predatory demand of the restored stock might deplete stocks of some forage species (Hartman 2003, Uphoff 2003, and Savoy and Crecco 2004).

In the Hudson River, one of three major spawning estuaries of the Atlantic coast population of striped bass (ASMFC 2005), the abundances of juvenile blueback herring (*Alosa aestivalis*) and alewife (*Alosa pseudoharengus*), collectively referred to as river herring, and Atlantic tomcod (*Microgadus tomcod*) and white perch (*Morone americana*) have declined since about 1990 (Central Hudson Electric and Gas Corporation et al. 1999 and Hurst et al. 2004). During the same period, striped bass juvenile abundance has remained fairly stable while the abundance of larval striped bass abundance has increased substantially. White perch, river herring and striped bass spawning occurs in late May and June in the Hudson River. Juvenile striped bass, white perch and river herring are collected by beach seines from late July through October (Central Hudson Electric and Gas Corporation et al. 1999). Atlantic tomcod hatching occurs in late February and early March (Dew and Hecht 1994), and juveniles are present by late April (Central Hudson Electric and Gas Corporation et al. 1999). These five species comprised

85% of the average catch of estuarine and diadromous species collected by beach seines from 1980 through 2000 (Hurst et al. 2004).

Pre-spawning striped bass enter the lower Hudson River estuary in mid- to late fall and overwinter in the lower Hudson River (McLaren et al. 1981, and Clark 1968). In April, adult striped bass, including some immature fish, begin to migrate to the upriver spawning grounds (Bear Mountain Bridge (river km 74) to Newburgh-Beacon Bridge (river km 98)), often with immatures migrating first followed by older mature fish (McLaren et al. 1981). The peak period of spawing is typically between April and May After spawing, most adult striped bass migrate to the lower river and then out of the river to the Atlantic coast (McLaren et al. 1981). However, some portion of the adult population remains in the river, perhaps year-round (Secor and Piccoli 1996). Recaptures of tagged age 2 (immature) striped bass in the Hudson River have been reported in each month, April through November, and in each year, 1987 through 1992 (Dunning et al. 2006), providing positive evidence of their presence in the river through the fall.

The historical commercial fishery for striped bass in the Hudson River was open from May through November prior to its closure after 1975 over concerns of PCB contamination (McLaren et al. 1988). Commercial fishing generally was conducted with gill nets from the George Washington Bridge (river km 19) to Hudson, NY (river km 181). In 1976, 1977 and 1978, a study was conducted to simulate the commercial fishery from April through June with three commercial fishers fishing two days per week each week. The catch rate of striped bass greater than 250 mm declined each month from an average of 659 fish in April, to an average of 342 fish in May, to an average of 258 in June (Texas Instruments 1980), indicating that perhaps as much as 39% of the adult stock were still present in the river in June. A 2001 recreational fishery survey of the Hudson River (Normandeau Associates, Inc. 2003) estimated striped bass catch per unit effort (CPUE) for shore-based fishing of 13.5 (fish per 100 angling hours) in spring (mid-March through mid-June) and 3.3 in late summer (August through September), suggesting that late summer abundance could have been 24% of the spring abundance. Shore-based fishing was the predominant fishing mode in the portion of the Hudson River downriver of the striped bass spawning grounds. That study also estimated striped bass harvest (mean total length of 727 mm) per unit effort (HPUE) for shore-based fishing of 1.1 (fish per 100 angling hours) in spring and 0.2 in late summer, suggesting that late summer abundance of larger striped bass could have been 18% of the spring abundance. In fall (October through November) the shore-based fishing CPUE for striped bass increased to 29.9 (fish per 100 angling hours) and the HPUE increased to 1.1, possibly due to the arrival of over-wintering pre-spawners.

Hudson River striped bass in their first year of life are primarily consumers of invertebrates but become largely piscivorous during their second year of life (Walter et al. 2003, and Gardinier and Hoff 1982), at which time they grow to exceed 200 mm (Texas Instruments 1980). Stomach content studies of adult striped bass in the Hudson River were conducted in 1974, 1976 and 1977 (Gardinier and Hoff, 1982) and from 1990 through 2006 (Kahnle and Hattala, 2007). In 1976 and 1977, 380 striped bass from 200 mm to over 800 mm were collected with a 900 foot haul seine in April and May; 102 contained recognizable food items. In 1974, 317 striped bass (including 13 between 200 mm and 275 mm) were collected with beach seines and otter trawls from April through November. The only recognizable finfish present in stomachs of striped bass larger than 200 mm were Atlantic tomcod, white perch, striped bass, spottail shiner and unidentified clupeids (likely blueback herring, alewife and American shad, which are common in the Hudson River). From 1990 through 2006 stomach contents of 1859 mature striped bass (modal length 659-700 mm TL) were examined, 89% of which were collected in the spring. Approximately 15% of the stomachs from spring collected striped bass contained food items, and 33% of stomachs from the fall and summer collected striped bass contained food items. The dominant food items were unidentified fish (35.5%), crabs (16.1%), herring (18.1%), Atlantic menhaden (4.6%), isopods (4.3%) and white perch (3.6%). A stomach content study conducted in winter months of 1991-1992 (with water temperature less than 10°C) collected 137 striped bass larger than 200 mm (Dunning et al. 1997). The primary finfish identified were blueback herring, clupeids, white perch, and striped bass.

Objective and Analysis Approach

The objective of this study was to determine whether the increase in predatory demand of Hudson River striped bass, accompanying the increase in abundance of the recovered striped bass stock, could have been responsible for the observed changes in abundance of juvenile Atlantic tomcod, river herring, white perch and striped bass. The approach used to address this objective was developed in response to the availability of relevant historical data. Estimates of year- and age-specific abundances (age 1 through age 13+) and instantaneous mortality rates for the coastwide striped bass stock from 1982 through 2004, and an estimate of the fractional contribution of Hudson River striped bass to the coastwide stock, were available from the Atlantic States Marine Fisheries Commission ("ASMFC") stock assessment (ASMFC 2005). Estimates of the annual abundance of larval and juvenile life stages of the five species in the Hudson River for 1977 through 2004 were available from a series of annual reports referred to as Hudson River Year Class Reports (e.g., EA 1996), which document sampling results from the Hudson River Monitoring Program ("HRMP") funded by electric generators on the Hudson River. Season- and age-specific estimates of abundance of age 1 and older striped bass inhabiting the Hudson River were not available for the period of interest. Furthermore, with the exception of the studies cited in a previous paragraph, season- and age-specific characterizations of diets of Hudson River striped bass also were not available.

The analysis approach contained four steps. The first was the development of a method that would be supported by the available data for estimating instantaneous mortality rates that might be due to predation. Existing multispecies virtual population analysis methods and ecosystem balancing methods (Magnusson 1995, Whipple et al. 2000, and Christensen et al. 2005), which can generate separate estimates of mortality rate due to predation, were not selected due to their extensive data requirements. The second step was the estimation of the changes in juvenile abundances for two stanzas of years (1977 to 1991 was referred to as Period 1, and 1991 to 2004 was referred to as Period 2), and estimation of the changes in annual predatory demand of Hudson River striped bass for the two stanzas of years. Over the 28 years of interest, August through October has been the consistent sampling season for juvenile fish by the HRMP; therefore, estimates of juvenile abundance were restricted to that three month season. These estimates of change, expressed in terms of ratios, were used as the primary inputs to the analysis. The third step was estimation of the instantaneous mortality rates that might be due to predation. The final step was a comparison of the potential juvenile biomass consumed by striped bass predation (kg yr^{-1}), which was computed using the estimated mortality rates for possible predation, to the estimated predatory demand of Hudson River striped bass. The purpose of the final step was to confirm that the magnitude of predation required to produce the observed change in juvenile abundance was no greater than the predatory demand of Hudson River striped bass.

To address the possibility that different age classes of striped bass might exert different levels of predation on juvenile fish in the Hudson River, the assessment was conducted separately for two age groups of possible predators: ages 1 through 13 striped bass, and age 1 and age 2 striped bass only. Secor and Piccoli (1996) found evidence of size-dependent dispersion of striped bass from the Hudson River with male age 2 striped bass spending most of their year in mesohaline portions of the estuary.

Methods and Data

Underlying System of Equations

For the purpose of estimating instantaneous mortality rates that were possibly due to predation in the two periods, three ratios were defined. Ratios (of a variable in Period 2 to the same variable in Period 1) were selected as the basic inputs to the analysis because scaling factors that are common to the two periods (e.g., gear efficiency) would cancel out in ratios; this can help eliminate possible biases that otherwise could arise due to possible errors in specifying those scaling factors. Because the focus of the study was the overall change in predatory demand and juvenile abundance between the two periods, and not detailed inter-annual variability, the underlying system of equations was defined in terms of average conditions (rates) for each period.

The first ratio was the potential change in juvenile biomass consumed by striped bass, defined as a ratio of average biomass possibly consumed in Period 2 ($\overline{C}_2$) to the average possibly consumed in Period 1 ($\overline{C}_1$):

$$R_c = \frac{\overline{C}_2}{\overline{C}_1}. \tag{1}$$

The second was the change in average juvenile abundance, defined as the ratio of average abundance during the juvenile sampling season (i.e., August through October) in Period 2 ($\overline{N}_2$) to the average in Period 1 ($\overline{N}_1$):

$$R_n = \frac{\overline{N}_2}{\overline{N}_1}. \qquad (2)$$

The third was the change in the number of fish entering the juvenile life stage, defined as the ratio of the average number entering the juvenile life stage in Period 2 ($\overline{L}_2$) to the average in Period 1 ($\overline{L}_1$):

$$R_l = \frac{\overline{L}_2}{\overline{L}_1}. \qquad (3)$$

The three ratios were expressed in terms of the mortality rates of interest through the following standard equations from fishery science (Ricker 1975). For each period, the annual seasonal consumption of juvenile biomass by predation (which is directly analogous to the fishery yield) in period j was defined as:

$$C_j = \left(m_{p,j}t\right)B_j \frac{1-e^{\left(g_j - m_j - m_{p,j}\right)t}}{\left(m_j + m_{p,j} - g_j\right)t}, \qquad (4)$$

where g_j is the daily growth rate during the season; m_j is the background daily mortality rate (i.e. all mortality except mortality due predation); $m_{p,j}$ is the additional daily mortality rate due to predation during the season; and t is the duration of the season (days). The biomass at the beginning of the season, B_j, was defined as:

$$B_j = w_j \overline{N}_j \frac{\left(m_j + m_{p,j}\right)t}{1-e^{\left(m_j - m_{p,j}\right)t}} \qquad (5)$$

where w_j is the weight per fish at the beginning of the season. The average annual juvenile abundance during the sampling season was defined as:

$$\overline{N}_j = \overline{L}_j e^{-(m'_j + m_{p,j})t'} \frac{1 - e^{(m_j - m_{p,j})t}}{(m_j + m_{p,j})t} \tag{6}$$

where m'_j is the background daily mortality rate from the beginning of the juvenile life stage to the beginning of August, $\overline{L}_j$ is the average abundance at the beginning of the juvenile life stage during period j, and t' is the duration (days) from the beginning of the juvenile life stage to the beginning of the juvenile sampling season.

Combining equations (1) through (6) gives the following two equations which form the basis for the analysis:

$$\frac{R_c}{R_n} = \frac{\left((m_{p,2}t) w_2 \frac{(m_2 + m_{p,2})t}{1 - e^{(m_2 + m_{p,2})t}} \frac{1 - e^{(g_2 - m_2 - m_{p,2})t}}{(m_2 + m_{p,2} - g_2)t} \right)}{\left((m_{p,1}t) w_1 \frac{(m_1 + m_{p,1})t}{1 - e^{(m_1 + m_{p,1})t}} \frac{1 - e^{(g_1 - m_1 - m_{p,1})t}}{(m_1 + m_{p,1} - g_1)t} \right)} \tag{7}$$

and

$$\frac{R_n}{R_l} = \frac{e^{-(m'_2 + m_{p,2})t'} \frac{1 - e^{(m_2 - m_{p,2})t}}{(m_2 + m_{p,2})t}}{e^{-(m'_1 + m_{p,1})t'} \frac{1 - e^{(m_1 - m_{p,1})t}}{(m_1 + m_{p,1})t}} \tag{8}$$

The right hand sides of equations (7) and (8) contain only underlying rates (and initial weight per fish for equation (7)), and the left hand side of the equations contain the measurable quantities.

Approximations

Estimates of the instantaneous mortality rates due to possible predation for Period 1 and Period 2 can be identified through an exhaustive search (by computer) for values of $m_{p,1}$ and $m_{p,2}$ that satisfy the non-linear equations (7) and (8), given input values for the two ratios of ratios and estimates for the growth rates and background mortality rates. Alternatively, equations (7) and

(8) can be linearized, and approximate closed-form solutions for $m_{p,1}$ and $m_{p,2}$ can be derived (see Appendix A). The closed-form solutions provide a more convenient method for conducting the analysis and also provide a basis for developing variance estimates (see Appendix B).

The approximation for the ratio of ratios in equation (7) is:

$$\frac{R_c}{R_n} \doteq \left(\frac{m_{p,2}}{m_{p,1}}\right)\alpha \tag{9}$$

where α is the ratio (Period 2 to Period 1) of the average juvenile weight per fish at the mid-point of the season. The logarithm of the ratio of ratios in equation (8) is approximately:

$$\ln\left(\frac{R_n}{R_l}\right) \doteq \left(m_{p,1} - m_{p,2}\right)\left(t' + \frac{t}{2}\right) + \beta \tag{10}$$

where β is the difference between the juvenile background mortality rates for Period 1 and Period 2.

Combining equations (9) and (10) provides approximate solutions for the potential predation mortality rates in the two periods expressed in terms of functions of the two ratios of ratios:

$$\tilde{m}_{p,1} \doteq \frac{\ln\left(\frac{R_n}{R_l}\right) + \beta}{\left(1 - \frac{R_c}{\alpha R_n}\right)\left(t' + \frac{t}{2}\right)} \tag{11}$$

and

$$\tilde{m}_{p,2} \doteq \frac{\ln\left(\frac{R_n}{R_l}\right) + \beta}{\left(\frac{\alpha R_n}{R_c} - 1\right)\left(t' + \frac{t}{2}\right)} \tag{12}$$

Changes in Juvenile and Larval Abundances

The ratio of abundances of post yolk-sac-larvae (Table 2) was used as a surrogate for the ratio of abundance of fish entering the juvenile life stage (R_l) because field data on the number of fish entering the juvenile life stage were not available. Average abundance indices for post yolk-sac larvae were computed as the average of weekly standing crop estimates from Hudson River Year Class Reports. Weekly standing crop estimates for post yolk-sac larvae were based on data collected by the HRMP's Longitudinal River Survey ("LRS") which sampled with 1 m ichthyoplankton nets attached to epibenthic sleds (to sample the bottom stratum) and Tucker trawls (to sample the mid-water stratum). Annual abundance indices based on LRS data were computed for 1977 through 2004, based on data from stratified random sampling from the George Washington Bridge north to the Federal Dam at Troy, NY during May and June. Alewife and blueback herring were treated as a single taxonomic group (river herring) because they could not be reliably identified to species as post yolk-sac larvae.

The ratios of average abundances (R_n) of juvenile river herring, Atlantic tomcod, white perch and striped bass (Table 3) were based on annual indices of juvenile abundance. Annual juvenile abundance was computed as the average of weekly standing crop estimates from Hudson River Year Class Reports (e.g., EA 1996). Weekly standing crop estimates for juvenile fish inhabiting the beach zone of the Hudson River were based on data collected by the HRMP's Beach Seine Survey ("BSS"), which sampled with 100 ft beach seines from the George Washington Bridge to the Federal Dam at Troy, NY. Weekly standing crop estimates for juvenile fish inhabiting the shoals, bottom and channel of the Hudson River were based on data collected by the HRMP's Fall Shoals Survey ("FSS"), which sampled with beam trawls (to sample the bottom stratum) and Tucker trawls (to sample the mid-water stratum) from the George Washington Bridge to the Federal Dam at Troy, NY.

Annual abundance indices based on BSS data were computed for 1977 through 2004, using data from biweekly sampling in August through October. Annual abundance indices based on FSS data were computed for 1985 through 2004, using data from biweekly sampling in August though October. The FSS was conducted from 1979 to 1984; however, beam trawls replaced epibenthic sleds for sampling the bottom and shoal strata in 1985. To avoid possible confounding effects of the gear change, FSS data prior to 1985 were not included in the analysis. However, because BSS and FSS indices of abundance (1985-2004) were significantly correlated, juvenile abundance indices for a given species from the BSS from 1979 through 1984 were used to predict FSS abundance indices (as if beam trawl sampling had occurred in those years) for the years prior to 1985.

For each species, annual average (August through October) juvenile abundance estimates (Table 3) were computed by adjusting the annual average standing crop estimates from the BSS and FSS for gear efficiency and summing the resulting abundance estimates:

$$\overline{N}_y = \frac{\overline{A}_{BSS,y}}{q_{BSS}} + \frac{\overline{A}_{FSS,y}}{q_{FSS}} \tag{13}$$

where $\overline{A}_{BSS,y}$ and $\overline{A}_{FSS,y}$ are the reported average (August through October) standing crop estimates from the two programs for year y, and q_{BSS} and q_{FSS} are gear efficiencies for the two sampling programs. Gear efficiency estimates used for this computation are those reported in Central Hudson Electric and Gas Corporation et al. (1999), which were based on gear efficiency studies (Normandeau Associates Inc. 1984, Kjelson and Colby 1977, and Loesch 1976) and on comparisons of striped bass BSS catch rates to striped bass mark-recapture estimates of abundance. For the BSS, the gear efficiency was assumed to be 4%; and for the FSS, the gear efficiency was assumed to be 8.85, the average of the reported beam trawl gear efficiency (15%) and the reported Tucker trawl gear efficiency (2.7%).

The estimates of juvenile abundance computed as described above are generally consistent with other estimates reported in the literature. Young et al. (1988) reported estimates of juvenile white perch abundance in the Hudson River based on mark-recapture studies from 1974 through 1979. The estimates ranged from 13 million to 205 million with and average of 74 million. The estimated average juvenile white perch abundance for Period 1 of 65.5 million from this study is consistent with those mark-recapture estimates. McLaren et al. (1988) reported mark-recapture estimates of abundance for one year old (roughly mid-February) Hudson River Atlantic tomcod for 1975 to 1980 which ranged from 2.5 to 8.9 million, with an average of 5.8 million. To be consistent with the Period 1 estimate (Table 2) of 54 million juveniles, the mortality rate from mid-September to mid-February would have to be approximately Z=2.2 (5 months). Although estimates of survival rates for juvenile Hudson River Atlantic tomcod could not be found in the literature, McLaren et al. (1988) reported annual mortality rates from age 1 to age 2 for Atlantic tomcod. The average for 1975 through 1979 was Z=2.8 (12 months), which is not inconsistent if both the difference in age and the difference in duration are considered.

Changes in Predatory Demand

For the purpose of assessing whether the change in predatory demand could have been responsible for the observed changes in juvenile abundance, the ratio of potential consumption of juvenile biomass by striped bass (R_c) was assumed to be the same as the ratio (Period 2 to Period 1) of predatory demands of striped bass:

$$R_p = \frac{\overline{H}_2}{\overline{H}_1} \tag{14}$$

where $\overline{H}_j$ is the average of annual estimates of predatory demand during period j.

Estimates of the annual predatory demand exerted by the Hudson River stock were based on estimates of annual production by the Hudson River stock and an assumed trophic efficiency between striped bass and their prey. Age-specific estimates of annual production, $H_{a,y}$ (kg yr^{-1}), of age-1 and older striped bass were based on the production formulation from Ricker (1975):

$$H_{a,y} = G_{a,y}\overline{B}_{a,y} = \frac{G_{a,y} SB_{a,y} W_a \left(1 - e^{G_{a,y} - Z_{a,y}}\right)}{\left(Z_{a,y} - G_{a,y}\right)} \tag{15}$$

where $SB_{a,y}$ is the estimated abundance of age a striped bass in year y, W_a is the average weight of age a striped bass at the beginning of the year, $G_{a,y}$ is the annual growth rate for age a striped bass in year y, and $Z_{a,y}$ is the annual mortality rate for age a striped bass in year y. Annual predatory demand, $P_{a,y}$, was estimated by dividing annual production by trophic efficiency, assumed to be 10% (Pauly and Christensen 1995, Jennings and Mackinson 2003, and Jennings et al. 2002).

Estimates of the coastwide abundance of age 1 through age 13 striped bass for 1982 to 2004 ($SB_{a,y}$) were from the 2005 Stock Assessment (Table 18a, ASMFC 2005). Because striped bass post yolk-sac larval abundance (an indicator of spawning stock abundance) was relatively stable from 1977 through 1990, the average age-specific abundances from 1982 through 1990 were assumed to be representative of the averages for all years in Period 1 (1977 through 1990). For each age class (age 1 and older) and year the total striped bass mortality rate ($Z_{a,y}$) was computed as the sum of reported age- and year-specific fishing mortality rate (Table 16, ASMFC 2005) and a constant natural mortality rate of 0.15 (ASMFC 2005). The fraction of the coastwide abundance of striped bass that was of Hudson River origin was assumed to be 13%

(ASMFC 2005). Age- and year-specific annual growth rates ($G_{a,y}$) were estimated from reported average weights at age (Table 13, ASMFC 2005) assuming approximately exponential growth (Ricker 1975) over successive two-year intervals:

$$\hat{G}_{a,y} = 0.5\ln\left[\frac{\overline{W}_{a+1,y+1}}{\overline{W}_{a-1,y-1}}\right] \qquad (16)$$

where $\overline{W}_{a,y}$ is the reported average weight for age a striped bass in year y, and the initial weight for each age group and year, $\hat{W}_{a,y}$, was estimated as:

$$\hat{W}_{a,y} = \overline{W}_{a,y}\frac{-\hat{G}_{a,y}}{1-e^{\hat{G}_{a,y}}} \qquad (17)$$

Estimates of coastwide predatory demand of striped bass computed using these methods (Table 1) are consistent with other published estimates. Hartman (2003) estimated the annual coastwide predatory demand of the striped bass population to be 17.9 mt in 1982 and 147.9 mt in 1995. His estimates were based on age- and year-specific coastwide striped bass abundance and survival estimates from ASMFC (2000). Using those same inputs and the methods described above for this study, the estimates of coastwide predatory demand of striped bass are 17.3 mt in 1982 and 135.7 mt in 1995. The estimates listed in Table 1 used updated abundance and survival estimates from ASMFC (2005), which account for the difference in comparison to Hartman's estimates. Uphoff (2003), also using ASMFC abundance estimates from 2000, estimated the annual coastwide potential consumption of Atlantic menhaden by striped bass to be 26 mt in 1982-1983, and 190 to 200 mt from 1994 to 1998.

The seasonal pattern of predatory demand by striped bass was characterized based on average monthly water temperatures in the Hudson River and the consumption component of a bioengergetics model for striped bass (Hartman and Brandt 1995). The fraction of the annual consumption (τ) that occurred from August through October was approximated as:

$$\tau = \frac{\sum_{m=8}^{10} CR_m}{\sum_{m=1}^{12} CR_m} \qquad (18)$$

where CR_m is the predicted consumption rate (gm gm^{-1} day^{-1}) for the average water temperature in month m. This approximation does not account for possible month-specific variability in growth and mortality rates of striped bass. Estimates of month-specific water temperature, required for the bioenergetics model of the seasonal pattern of consumption, were from Poughkeepsie Water Works data (Table B-4, EA 1996). The consumption from August through October was estimated to be 41.8% of the annual total. The average seasonal predatory demand (Table 1) for each period was estimated as the product of the average annual predatory demand for the period and the fraction of the annual consumption that occurred from August through October.

Estimation of Instantaneous Mortality Rates Due to Possible Predation

Instantaneous mortality rates for possible predation, that were consistent with the estimated ratios (R_n, R_l, R_p), were identified through exhaustive search (by computer) of candidate values of $m_{p,1}$ and $m_{p,2}$ using equations (7) and (8). Because the question being addressed was whether the increase in striped bass predation could have caused the observed changes in juvenile abundance, all other things being equal, background mortality rates, growth rates, and initial weights were assumed to have remained the same for the two periods. Approximate estimates also were computed using the equations (11) and (12); and for the reason noted above, the parameter α was set equal to 1, and the parameter β was set equal to 0.

Variance estimates for the approximations were computed using the methods described in Appendix B.

Estimation of Potential Consumption of Juvenile Biomass

The potential juvenile biomass consumed by striped bass was computed using equation (4) with the estimates of instantaneous mortality due to potential predation and the estimates of average seasonal juvenile abundance. Also required for estimating potential juvenile biomass consumed by striped bass were estimates of daily background mortality rates and growth rates of the juvenile fish, and initial weights of the juvenile fish.

For each species, the background daily mortality rates (Table 4) for the three month sampling season (August through October) were estimated as a power function of dry weight (Peterson and Wroblewski, 1984):

$$m = \frac{1}{t}\sum_{i=1}^{t} 0.00525\left(0.2we^{gi}\right)^{-0.25} \tag{19}$$

where dry weight is assumed to be 20% of wet weight (Peterson and Wroblewski, 1984). Similarly, the background daily mortality rate for the interval from the start of the juvenile life stage to August was estimated as:

$$m' = \frac{1}{t'}\sum_{i=1}^{t'} 0.00525\left(0.2we^{gi}\right)^{-0.25} \tag{20}$$

The duration of the juvenile sampling season (t) was set to 90 days (August through October), and (based on life history considerations discussed in the Introduction) the interval from the beginning of the juvenile stage to the beginning of the juvenile sampling season (t') was set to 15 days for white perch, river herring and striped bass, and set to 90 days for Atlantic tomcod.

For each species, the daily juvenile growth rate through October (Table 4) was estimated from the beginning and ending weights, assuming approximate exponential growth during that interval, as (Ricker 1975):

$$g = \frac{\ln\left(\frac{w_{end}}{w_{start}}\right)}{t + t'} \tag{21}$$

and the weight of species s at the beginning of August (Table 4) was estimated as:

$$w = w_{start} e^{gt'} \tag{22}$$

Estimates of the average weight per fish at the beginning and end of the juvenile life stage were derived from reported lengths and length-weight relationships. For river herring, the lengths at the beginning and end of the juvenile stage were set to 25mm and 92mm (Mullen et al. 1986), respectively, and the length-weight relationship was from PSEG (2006). For Atlantic tomcod, the initial length (for mid-May) and the final length (for the end of October) were set to 25mm and 120mm, respectively, (McLaren et al. 1988); and the length-weight relationship was from Dew and Hecht (1994). For white perch, the lengths at the beginning and end of the juvenile stage were set to 25mm and 80mm, respectively (Texas Instruments 1980); and the length-weight relationship was from Klauda et al. (1988). For striped bass, lengths at the beginning and end of the juvenile stage were set to 30mm and 95mm, respectively (Dey 1981); and the length-weight relationship was from Fay et al. (1983).

Sensitivity Analysis to Address Assumptions

A sensitivity analysis was conducted to address: 1) the possible effects of density dependent mortality occurring between the larval and juvenile life stages, 2) the effects of possible errors in the estimation of background mortality rates on the predicted juvenile biomass to predation, and 3) an alternative assumption regarding the fraction of the coastwide stock that was from the Hudson River. Other input parameters, which did not require formal sensitivity analyses, but which could affect results are discussed at the end of this section.

For Atlantic tomcod, river herring and white perch, the historical data indicated a decline in larval abundance from Period 1 to Period 2, and for striped bass an increase was indicated. The results presented above assume the ratio of abundance (Period 2 to Period 1) of fish entering the juvenile life stage is the same as the ratio of larval abundance. However, if density dependent effects were present, the ratio of abundance of fish entering the juvenile stage could have been closer to unity. To address this possibility, the analyses were re-run with values for the ratio of abundance of fish entering the juvenile stage (R_l) ranging from the estimated value (r_l) based on post yolk-sac larval abundances to a value of R_l=1 (i.e. constant recruitment to the juvenile life stage). An index of the degree of density dependent effects (I) was defined as:

$$I = \frac{(R_l - r_l)}{(1 - r_l)} \tag{23}$$

with a range from 0 (for $R_l = r_l$) to 1 (for R_l=1).

The equation used to estimate the background mortality rate for juvenile fish (equations (19) and (20)) is a theoretically derived relationship for pelagic marine ecosystems (Peterson and Wroblewski 1984). Other authors (e.g. McGurk (1993), Lorenzen (1996) and Houde (1997)) have reported natural mortality rates of fish in marine and other ecosystems also as power functions of weight, but with empirical estimates for the coefficients that differ somewhat from those of Peterson and Wroblewski (1984). To address the effects of possible errors in the assumed background mortality rate, the analyses were re-run with the background mortality rates set to 0 and with the background mortality rates set to 2 times of the initial estimates.

Estimates of the coastwide abundance of age 1 striped bass, combined with indices of juvenile abundance from the major spawning areas of striped bass (ASMFC 2005) indicate that the proportion of the coastwide population of age 1 striped bass that is from the Hudson River has changed from Period 1 to Period 2 (see Appendix C). The average estimated contributions from the Hudson River for Periods 1 and 2 are 20.9% and 8.9% respectively. Assuming these proportions apply to age 1 and age 2 striped bass, then the ratio of predatory demands (R_p) for age 1 and age 2 striped bass would decline from 3.44 (Table 1) to 1.46. To address the effects of this alternative assumption regarding the contribution of Hudson River striped bass to the coastwide stock, the analyses were re-run the analysis with the alternative estimate for R_p for age 1 and age 2 striped bass.

Other input parameters of concern were the trophic conversion efficiency, the fraction of the annual predatory demand exerted during the three month fall season, and gear efficiencies. Selection of alternative values for these parameters would not affect estimates of instantaneous mortality rates possibly due to predation because, as noted above, the inputs to the analyses are ratios in which scaling factors that are common to both periods cancel out. However, if one of these factors varied substantially between the two periods, then the degree of change in that factor would determine the effect on estimates of instantaneous mortality rates possibly due to predation. The possible effects of changes in these factors between the two periods were viewed as second order considerations for this study; and therefore, sensitivity analyses of those possible changes were not undertaken.

Because the estimates of juvenile biomass possibly consumed by predation use these input parameters directly (not in ratios) estimates of juvenile biomass possibly consumed by predation would be affected by assumed gear efficiencies. A change of the assumed gear efficiency (e.g. doubling) would cause an inversely proportional change (i.e., halving) of the estimate of juvenile biomass possibly consumed. Similarly, a change of the assumed trophic conversion efficiency (e.g. doubling) would cause an inversely proportional change (i.e., halving) of the estimate of predatory demand. A change of the assumed fraction of the annual predatory demand exerted during the three month fall season (e.g., doubling) would cause a directly proportional change (i.e., doubling) of the estimate of predatory demand. Because the sensitivities of the estimates to these assumptions were clear, no additional analyses were conducted to address them.

Results

Estimates of Instantaneous Mortality Rates Possibly Due to Predation

Estimates of the seasonal instantaneous mortality rates possibly due to predation by striped bass (Tables 5 and 6) were higher for juvenile striped bass than for juveniles of the other three taxa. The estimated rates were slightly higher under the assumption that predation was by age 1 and age 2 striped bass only, than under the assumption that predation was by age 1 through age 13 striped bass. The estimated instantaneous mortality rates for Period 2 were 12 to 15 times higher than for Period 1 assuming predation was by all age classes; and were 10 to 12 times higher than Period 1 assuming predation by age 1 and age 2 striped bass only. For river herring, Atlantic tomcod and white perch, the estimates based on the approximations were very similar to the estimates based on exhaustive search; however, for striped bass the approximations underestimated the Period 2 rate and overestimated the Period 1 rate. The bias in the approximations for larger mortality rates was expected because the Paloheimo approximation works best with small mortality rates (Paloheimo 1961). Coefficients of variation for the estimates (based on the approximate standard errors) were 3-12% for striped bass, 31-39% for river herring, 10-13% for Atlantic tomcod, and 9-14% for white perch.

Comparison of Juvenile Biomass Possibly Consumed by Striped Bass to Hudson River Striped Bass Predatory Demand

The estimated juvenile biomass possibly consumed by striped bass during the three month season (Tables 7 and 8) was 148,000 kg in Period 1 and 509,000 kg in Period 2 assuming predation by age 1 and age 2 striped bass only, and was 112,000 kg in Period 1 and 498,000 kg in Period 2 assuming predation by age 1 through age 13 striped bass. Assuming predation by age 1 and age 2 striped bass only, the juvenile biomass possibly consumed by striped bass was 11.11% of the estimated seasonal predatory demand, and assuming predation by age 1 through age 13 striped bass, the juvenile biomass possibly consumed was 3.33%. Estimated consumption of juvenile striped bass was higher than the estimated consumption of the other three taxa, approximately 2 times higher than river herring, 4 times higher than Atlantic tomcod, and over 5 times higher than white perch.

Effects of Changes in Assumptions -- Sensitivity Analyses

Reducing the assumed background mortality rate had the effect of increasing the estimates of juvenile biomass possibly consumed by striped bass (Figures 2 and 3); increasing the assumed background mortality rate reduced the estimates of juvenile biomass possibly consumed by striped bass. Increases in the assumed degree of density dependent effects up to an index value between 0.5 and 0.75 caused the estimates of the juvenile biomass possibly consumed by striped bass to increase. Further increases in the assumed degree of density dependent effects, with the index increasing to 1, caused estimates of the juvenile biomass possibly consumed by striped bass to decrease (Figures 2 and 3). Changing the assumed proportion of the coastwide stock of age 1 and age 2 striped bass from 13% in both periods to 20.9% in Period 1 and 8.9% in Period 2 caused estimates of seasonal juvenile biomass possibly consumed by striped bass to increase. For Period 1 the estimate increased from 148,000 kg to 409,000 kg, and for Period 2 the estimate increased from 509,000 kg to 600,000 kg.

Considering the combined effects of alternative assumptions for background mortality rates and degree of density dependent effects, estimates of the percent of seasonal predatory demand potentially satisfied by consumption of juveniles of the four taxa were less than 18% for predation by age 1 and age 2 striped bass only, and were less than 6% for predation by age 1 through age 13 striped bass. Under the assumption that 20.9% (in Period 1) and 8.9% (in Period 2) of the coastwide stock of age 1 and age 2 striped bass were Hudson River fish, the maximum estimate of the percent of seasonal predatory demand potentially satisfied by consumption of juveniles of the four taxa increased from 18% to 28% (Figure 4).

Discussion

The percent of the seasonal predatory demand that could be satisfied by juvenile biomass consumed by striped bass has two components: 1) the fraction of the Hudson River striped bass population that inhabits the river from August through October, and 2) the contribution of the juvenile target species to the diet of striped bass in the river during those months. For example, if 75% of age 1 and age 2 striped bass from the Hudson River stock were present in the river from August through October, and 40% of their diet while in the river was satisfied by juveniles of the target species, then 30% of the predatory demand would be satisfied by those juvenile fish. The estimated percents of seasonal predatory demand that would be needed to explain the observed declines in juvenile abundance appear consistent with what is known about the fraction of the stock that inhabits the river in fall, and with what is known about Hudson River striped bass dietary preferences. The findings of Secor and Piccoli (1996) demonstrated that some fraction of the adult stock inhabits the river year-round; the simulated commercial fishery study indicated that more than one third of the spawning stock may have remained in the river in June; and the 2001 recreational fishery survey indicated that as much as 18%-24% of the striped bass abundance present in the river during the spring was present in the river by late summer. The available stomach content studies (Gardinier and Hoff 1982, Dunning et al 1997, and Kahnle and Hattala 2007) found clupeids, Atlantic tomcod, white perch, and striped bass among the dominant identifiable food items in age 1 and older in the Hudson River.

This study focused on the decline in juvenile abundance of four forage taxa as measured by sampling that occurred from August through October, and did not explicitly address possible reductions in spawning stock biomass that could have been caused by the reductions in juvenile abundance. However, the data on post yolk-sac larvae river herring, Atlantic tomcod and white perch abundance suggest that a reduction in spawning has occurred for these taxa, which may be due, in part, to the increased mortality during the juvenile stage. The reduction in spawning might also be due to increased mortality in older life stages of these taxa – possibly due, in part, to striped bass predation on age 1 or older fish. For striped bass, estimates of post yolk-sac larval abundance suggest a six-fold increase in larval abundance from Period 1 to Period 2, which is consistent with the apparent increase in adult abundance. However, the data on striped bass juveniles shows no corresponding increase in juvenile abundance. The analysis presented in this paper demonstrated that striped bass predation alone could have kept the juvenile abundance from increasing. Other possible explanations include a drastic reduction in the juvenile background mortality rate, or density dependent out-migration of juveniles.

The results from this study indicate that the increase in predatory demand of Hudson River striped bass could have been responsible for the decline in juvenile abundance of river herring, Atlantic tomcod and white perch, and responsible for the apparent decline in survival of striped bass from the post yolk-sac larvae to juveniles. The required magnitude of consumption of juvenile biomass to account for the declines in juvenile abundance appears to be well below the estimated predatory demand of Hudson River striped bass, whether considering all ages, or only age 1 and age 2 striped bass. The sensitivity analyses suggest this result is fairly robust to possible violations in assumptions and to possible errors in input parameter values. However, a field survey to estimate the biomass of juvenile fish consumed by Hudson River striped bass in the fall would be needed to confirm the proposition that Hudson River striped bass, in fact, were responsible for the declines in juvenile abundance.

Acknowledgements

I gratefully acknowledge the very thoughtful comments and suggestions on the first draft of this manuscript from four anonymous reviewers and an associate editor of the Transactions of the American Fisheries Society. I also acknowledge the many field crews and scientists who, since the mid 1970's, collected the data on age-0 Hudson River fishes that I relied on for this study; and I acknowledge the scientists of the ASMFC its member states for their many years of collecting and analyzing data on the Atlantic coastal striped bass stock, the results of which were key inputs to this study.

Literature cited

Atlantic States Marine Fisheries Commission (ASMFC). 2005. 2005 Stock Assessment Report for Atlantic Striped Bass: Catch-at-Age Based VPA & Tag Release/Recovery Based Survival Estimation. Washington, D.C.

Central Hudson Gas and Electric Corporation, Consolidated Edison Company of New York, Inc., New York Power Authority, and Southern Energy New York. 1999. Draft environmental impact statement – state pollutant discharge elimination system permits for Bowline Point, Indian Point 2 & 3, and Roseton steam electric generating stations. Pearl River, New York.

Clark, J. 1968. Seasonal movements of striped bass contingents of Long Island Sound and the New York Bight. Transaction of the American Fisheries Society. 97:320-343.

Cristensen, V., C.J. Walters, and D. Pauly. 2005. Ecopath with Ecosim: a user's guide. Fisheries Centre, University of British Columbia, Vancouver. November 2005 edition. 154 p. Vancouver, British Columbia.

Dew, B.C. and J.H Hecht. 1994a. Hatching, estuarine transport, and distribution of larval and early juvenile Atlantic tomcod, *Microgadus tomcod*, in the Hudson River. Estuaries Vol. 17, No. 2, p. 472-488.

Dew, B.C. and J.H. Hecht. 1994b. Recruitment, growth, mortality, and biomass production of larval and early juvenile Atlantic tomcod in the Hudson River estuary. Transaction of the American Fisheries Society. 123:681-702.

Dunning, D.J., J.R. Waldman, Q.E. Ross, and M.T. Mattson. 1997. Use of Atlantic tomcod and other prey by striped bass in the lower Hudson River estuary during winter. Transactions of the American Fisheries Society. 126:857-861.

Dunning, D.J., J.R. Waldman, Q.E. Ross, and M.T. Mattson. 2006. Dispersal of age-2+ striped bass out of the Hudson River. Pages 287-294 *in* J.R. Waldman, K.E. Limburg, and D.L. Strayer, editors. Hudson River fishes and their environment. American Fisheries Society, Symposium 51, Bethesda, Maryland.

EA Engineering, Science and Technology (EA). 1996. 1995 year class report for the Hudson River estuary monitoring program. Prepared for Consolidated Edison Company of New York, Inc. New York, New York.

Fay, C.W., R.J. Neves, and G.B. Pardue. 1983. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic) – striped bass. U.S. Fish and Wildlife Service, Division of Biological Services, FWS/OBS-82/11.8. U.S. Army Corps of Engineers, TR EL-82-4. 36 pp. U.S. Fish and Wildlife Service, National Wetlands Research Center, Washington, D.C., and U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Gardinier, M. and T.B. Hoff. 1982. Diet of striped bass in the Hudson River estuary. New York Fish and Game Journal. Vol. 29, No. 2. p152-165.

Hartman K.J. and S.B. Brandt. 1995. Comparative energetics and the development of bioenergetics models for sympatric estuarine piscivores. Canadian Journal of Fisheries and Aquatic Sciences 52:1647-1666.

Hartman, K.J. 2003. Population-level consumption by Atlantic coastal striped bass and the influence of population recovery upon prey communities. Fisheries Management and Ecology. 10:281-288.

Hartman, K.J., and S.B. Brandt. 1995. Predatory demand and impact of striped bass, bluefish and weakfish in the Chesapeake Bay: applications of bioenergetics models. Canadian Journal of Fisheries and Aquatic Science. Vol 52. p.1667-1687.

Hurst, T.P., K.A. McKown, and D.O. Conover. 2004. Interannual and long-term variation in the nearshore fish community of the mesohaline Hudson River estuary. Estuaries. Vol 27, No. 4. p. 659-669.

Jennings, S., K.J. Ware, and S. Mackinson. 2002. Use of size-based production and stable isotope analyses to predict trophic transfer efficiencies and predator-prey body mass ratios in food webs. Marine Ecology Progress Series. Vol. 240:11-20.

Kahnle A.W., and K.A. Hattala. 2007. Striped bass predation on adult American shad: occurrence and observed effects on American shad abundance in Atlantic coastal rivers and estuaries. In American shad stock assessment report for peer review. Volume I. Stock Assessment Report No. 07-01 (Supplement) of the Atlantic States Marine Fisheries Commission. August 2007. Washington, D.C.

Kendall, M. and A. Stuart. 1977. The advanced theory of statistics. Volume 1. Distribution Theory. 4th Edition. Macmillan Publishing Co., Inc. New York, New York. 472 p.

Kjelson, M.A., and D.R. Colby. 1977. The evaluation and use of gear efficiencies in the estimation of estuarine fish abundance. In M. Wiley (ed.) Esuarine Processes, Vol. II, Circulation, Sediments, and Transfer of Material in the Estuary, p. 416-424. Academic Press, New York.

Klauda, R.J., J.B. McLaren, R.E. Schmidt, and W.P. Dey. 1988. Life History of White Perch in the Hudson River estuary. American Fisheries Society Monograph 4:69-88.

Loesch, H., J. Bishop, A. Crowe, R. Kuckyr, and P. Wagner. 1976. Technique for estimating trawl efficiency in catching brown shrimp (*Penaeus aztecus*), Atlantic croaker (*Micropogon undulates*), and spot (*Leiostomus xanthurus*). Gulf Research Reports 2:29-33.

Magnusson, K.G. 1995. An overview of the multispecies VPA – theory and applications. Reviews in Fish Biology and Fisheries, 5:195-212.

McLaren, J.B., J.C. Cooper, T.B. Hoff, and V. Lander. 1981. Movements of Hudson River striped bass. Transaction of the American Fisheries Society. 110:681-702.

McLaren, J.B., Klauda, R.J., Hoff, T.B., and Gardinier, M. 1988. Commercial fishery for striped bass in the Hudson River, 1931-80. In C.L. Smith (ed.) Fisheries Research in the Hudson River. State University of New York Press. Pages 89-123. Albany, New York.

McLaren, J.B., T.H Peck, W.P. Dey and M. Gardinier. 1988. Biology of Atlantic tomcod in the Hudson River estuary. American Fisheries Society Monograph 4:102-112.

Mullen, D.M., C.W. Fay, and J.R. Moring. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (North Atlantic) – alewife/blueback herring. U.S. Fish and Wildlife Service. Biological Report (82)(11.56). U.S. Army Corps of Engineers, TR EL-82-4. 21 pp. U.S. Fish and Wildlife Service, National Wetlands Research Center, Washington, D.C., and U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi.

Normandeau Associates, Inc. 1984. Relative catch efficiency of a 3 m beam trawl and a 6.2 m high-rise trawl for sampling young of the year fishes in the Hudson River estuary. Prepared for New York Power Authority, White Plains, New York.

Normandeau Associates, Inc. 2003. Assessment of Hudson River recreational fisheries. Prepared for New York State Department of Environmental Conservation and Bureau of Marine Resources Hudson River Fisheries Unit. 69 pp. New Paltz, New York.

Paloheimo, J.E. 1961. Studies on estimation of mortalities: I. Comparison of a method described by Beverton and Holt to a new linear formula. Journal of the Fisheries Research Board of Canada 18:645-662.

Pauly, D, and V. Christensen. 1995. Primary production required to sustain global fisheries. Nature. Vol. 374:255-257.

Peterson, I., and J. S. Wroblewski. 1984. Mortality rate of fishes in the pelagic ecosystem. Canadian Journal of Fisheries and Aquatic Sciences 41(77):1117-1120.

PSEG. 2006. New Jersey Pollution Discharge Elimination System Permit Application for Salem Generating Station.

Richards, R.A., and P.J. Rago. 1999. A case history of effective fishery management: Chespeake Bay striped bass. North American Journal of Fisheries Management. 19:356-375.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada. Bulletin 191. 382 p. Department of the Environment, Fisheries and Marine Service. Ottawa.

Savoy, T.F., and V.A. Crecco. 2004. Factors affecting the recent decline of blueback herring and American shad in the Connecticut River. Pages 361-378. *In* P.M Jacobson, D.A. Dixon, W.C. Leggett, B.C. Marcy, Jr., and R.R. Massengill, editors. The Connecticut River ecological study (1965-1973) revisited: ecology of the lower Connecticut River 1973-2003. American Fisheries Society, Monograph 9, Bethesda, Maryland.

Secor, D.H. and P.M. Piccoli. 1996. Age- and sex- dependent migrations of striped bass in the Hudson River as determined by chemical microanalysis of otoliths. Estuaries. Vol. 19, No. 4, p. 778-793.

Texas Instruments. 1980. 1978 Year class report for the multiplant impact study of the Hudson River estuary. Appendix B. Prepared for Consolidated Edison of New York, Inc. 846 pp. New York, New York.

Uphoff, J.H. Jr. 2003. Predator-prey analysis of striped bass and Atlantic menhaden in upper Chesapeake Bay. Fisheries Management and Ecology. 10:313-322.

Walter, J.F. III., A.S. Overton, K.H. Ferry, and M.E. Mather. 2003. Atlantic coast feeding habits of striped bass: a synthesis supporting a coast-wide understanding of trophic biology. Fisheries Management and Ecology. 10:349-360.

Whipple, S.J., J.S. Link, L.P. Garrison, and M.J. Fogarty. 2000. Models of predation and fishing mortality in aquatic ecosystems. Fish and Fisheries, 1:22-40.

Young, J.R., R.J. Klauda, and W.P. Dey. 1988. Population estimates for juvenile striped bass and white perch in the Hudson River. American Fisheries Society Monograph 4:89-101.

554

Table 1. Estimates of average predatory demand ($\hat{\bar{P}}_j$) of striped bass populations for the two periods of years (j) with estimated standard errors (in parentheses) and ratios of estimated predatory demands (Period 2 to Period 1).

Stock	Season	Ages	$\hat{\bar{P}}_1$ (kg)	$\hat{\bar{P}}_2$ (kg)	Ratio of Average Predatory Demands (R_p)
Atlantic Coastwide	January-December	1 – 13+	61,829,229 (2,031,616)	274,937,594 (5,853,828)	
Hudson River	August-October	1 - 13+	3,363,749 (110,398)	14,957,667 (318,097)	4.45
Hudson River	August-October	1 and 2 Only	1,332,950 (89,354)	4,583,020 (246,129)	3.44

Table 2. Average index values for post yolk-sac larval ("PYSL") abundance ($\hat{\bar{L}}_j$) for the two periods of years (j) with estimated standard errors (in parentheses) and ratios of average PYSL abundances (Period 2 to Period 1).

Taxon	$\hat{\bar{L}}_1$	$\hat{\bar{L}}_2$	Ratio of Average PYSL Abundance (R_l)
Striped Bass	362,055,919	2,371,617,937	6.55
	(13,868,061)	(76,310,566)	
River Herring	2,008,741,295	990,192,857	0.49
	(50,041,415)	(19,314,422)	
Atlantic Tomcod	92,730,226	66,887,806	0.72
	(6,329,898)	(3,548,958)	
White Perch	985,594,499	855,285,920	0.87
	(15,678,812)	(15,857,150)	

Table 3. Estimates of average seasonal (August through October) juvenile abundance ($\hat{\bar{N}}_j$) for the two periods of years (j) with estimated standard errors (in parentheses) and ratios of average estimated juvenile abundances (Period 2 to Period 1).

Taxon	$\hat{\bar{N}}_1$	$\hat{\bar{N}}_2$	Ratio of Average Juvenile Abundances (R_n)
Striped Bass	68,372,839	57,132,380	0.84
	(1,312,794)	(903,684)	
River Herring	1,118,600,941	448,416,556	0.40
	(30,380,270)	(14,130,644)	
Atlantic Tomcod	54,150,749	16,859,655	0.31
	(2,671,040)	(995,044)	
White Perch	65,493,845	26,860,369	0.41
	(1,782,169)	(779,541)	

Table 4. Life history parameter estimates for juvenile striped bass, river herring, Atlantic tomcod and white perch.

Parameter	Taxon			
	Striped Bass	River Herring	Atlantic Tomcod	White Perch
Juvenile Growth Rate, g (day^{-1})	0.032	0.047	0.030	0.034
Initial Weight of Juvenile Fish, w (gm)	0.286	0.034	0.095	0.179
Background Mortality Rate -- August through October, m	0.606	0.847	0.470	0.669
Background Mortality Rate -- Beginning of Juvenile Life Stage to the Beginning of August, m' (duration in parentheses)	0.151 (15 days)	0.250 (15 days)	0.922 (90 days)	0.169 (15 days)

Table 5. Estimates of average seasonal (August through October) instantaneous mortality rates possibly due to predation by age 1 and age 2 striped bass for Period 1 (1977-1990) and Period 2 (1991-2004). For estimates based on approximation, estimated standard errors are listed (in parentheses).

Prey Taxon	Estimates Based on Exhaustive Search		Estimates Based on Approximation	
	$\hat{m}_{p,1}$	$\hat{m}_{p,2}$	$\hat{\tilde{m}}_{p,1}$	$\hat{\tilde{m}}_{p,2}$
Striped Bass	0.611	6.172	1.157	4.760
			(0.137)	(0.199)
River Herring	0.049	0.475	0.048	0.410
			(0.015)	(0.159)
Atlantic Tomcod	0.103	1.287	0.112	1.232
			(0.014)	(0.150)
White Perch	0.149	1.737	0.178	1.489
			(0.018)	(0.134)

Table 6. Estimates of average seasonal (August through October) instantaneous mortality rates possibly due to predation by age 1 through age 13 striped bass for Period 1 (1977-1990) and Period 2 (1991-2004). For estimates based on approximation, estimated standard errors are listed (in parentheses).

Prey Taxon	Estimates Based on Exhaustive Search		Estimates Based on Approximation	
	$\hat{m}_{p,1}$	$\hat{m}_{p,2}$	$\hat{\tilde{m}}_{p,1}$	$\hat{\tilde{m}}_{p,2}$
Striped Bass	0.449	5.894	0.834	4.437
			(0.048)	(0.141)
River Herring	0.037	0.457	0.036	0.398
			(0.011)	(0.154)
Atlantic Tomcod	0.077	1.255	0.084	1.205
			(0.008)	(0.146)
White Perch	0.113	1.712	0.133	1.445
			(0.018)	(0.134)

Table 7. Estimates of average seasonal (August through October) juvenile biomass possibly consumed by predation by age 1 and age 2 striped bass ($\hat{\bar{C}}_j$) for Period 1 (1977-1990) and Period 2 (1991-2004), and corresponding percent of seasonal predatory demand of age 1 and age 2 Hudson River striped bass.

Prey Taxon	$\hat{\bar{C}}_1$ (kg)	$\hat{\bar{C}}_2$ (kg)	Percent of Seasonal Predatory Demand
Striped Bass	76,652	263,547	5.75%
River Herring	39,804	136,821	2.99%
Atlantic Tomcod	18,073	62,137	1.36%
White Perch	13,420	46,147	1.01%
Total	147,949	508,652	11.11%

Table 8. Estimates of average seasonal (August through October) juvenile biomass possibly consumed by predation by age 1 through age 13 striped bass ($\hat{\bar{C}}_j$) for Period 1 (1977-1990) and Period 2 (1991-2004), and corresponding percent of seasonal predatory demand of age 1 through age 13 Hudson River striped bass.

Prey Taxon	$\hat{\bar{C}}_1$ (kg)	$\hat{\bar{C}}_2$ (kg)	Percent of Seasonal Predatory Demand
Striped Bass	58,215	258,862	1.73%
River Herring	29,741	132,319	0.88%
Atlantic Tomcod	13,671	60,787	0.41%
White Perch	10,281	45,713	0.31%
Total	111,908	497,681	3.33%

Figure 1. Estimates of annual predatory demand of the Atlantic coast striped bass stock, ages 1 through 13.

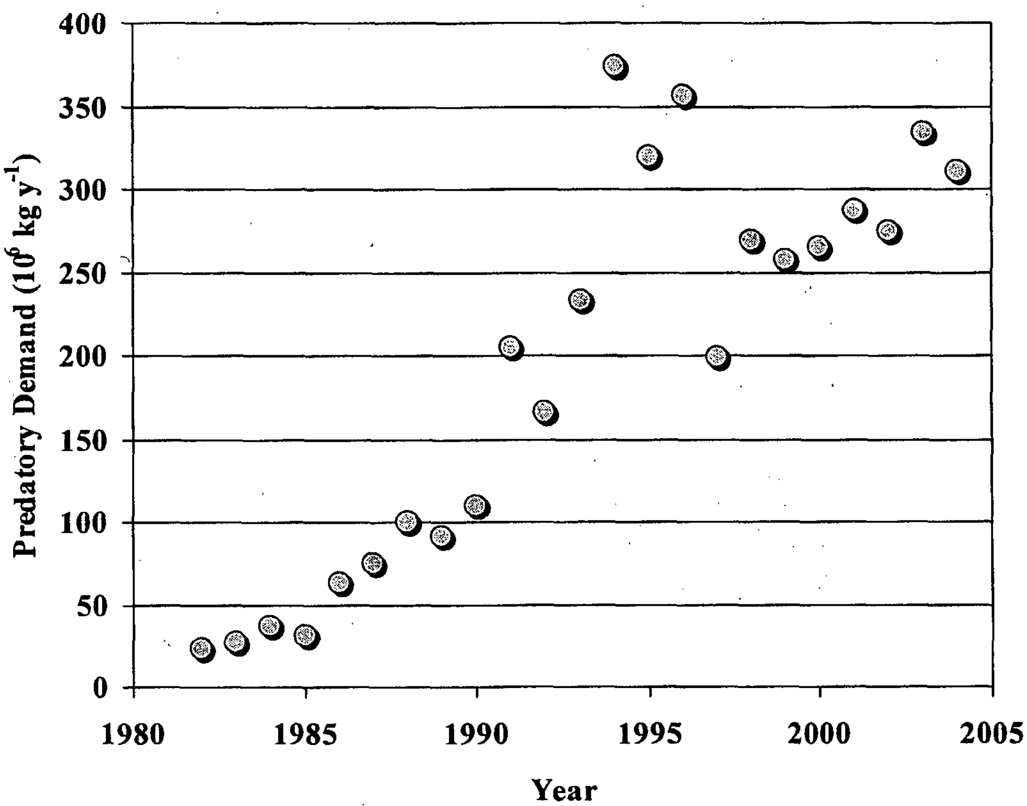


Figure 2. Estimates of the percent of seasonal predatory demand of age 1 and age 2 Hudson River striped bass potentially satisfied by consumption of juveniles of the four taxa, as functions of the index of density dependent effects (see text) and assumed background mortality rates. Curve A is for the estimated background mortality rates (see text), curve B is for background mortality rates of zero, and curve C is for two times the estimated background mortality rates. The proportion of the coastwide population of age 1 and age 2 striped bass that were Hudson River fish was assumed to be 13% in Period 1 and Period 2.

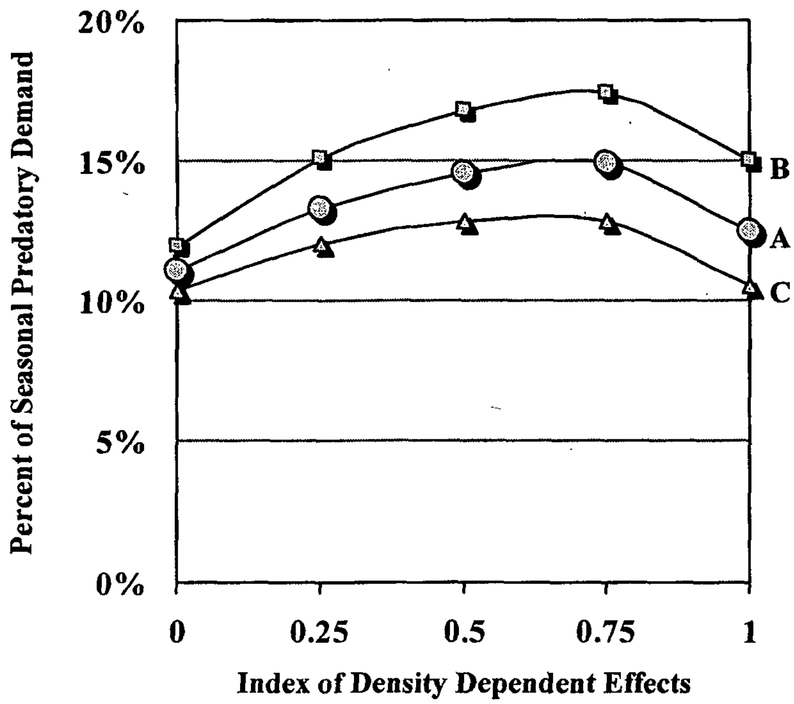


Figure 3. Estimates of the percent of seasonal predatory demand of age 1 and age 2 Hudson River striped bass potentially satisfied by consumption of juveniles of the four taxa, as functions of the index of density dependent effects (see text) and assumed background mortality rates. Curve A is for the estimated background mortality rates (see text), curve B is for background mortality rates of zero, and curve C is for two times the estimated background mortality rates. The proportion of the coastwide population of age 1 and age 2 striped bass that were Hudson River fish was assumed to be 20.9% in Period 1 and 8.9% in Period 2.

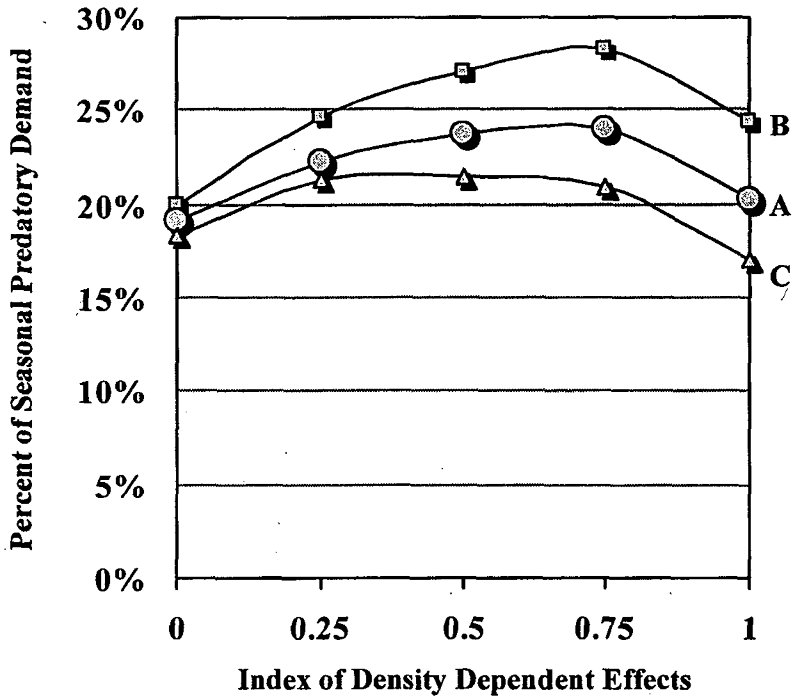
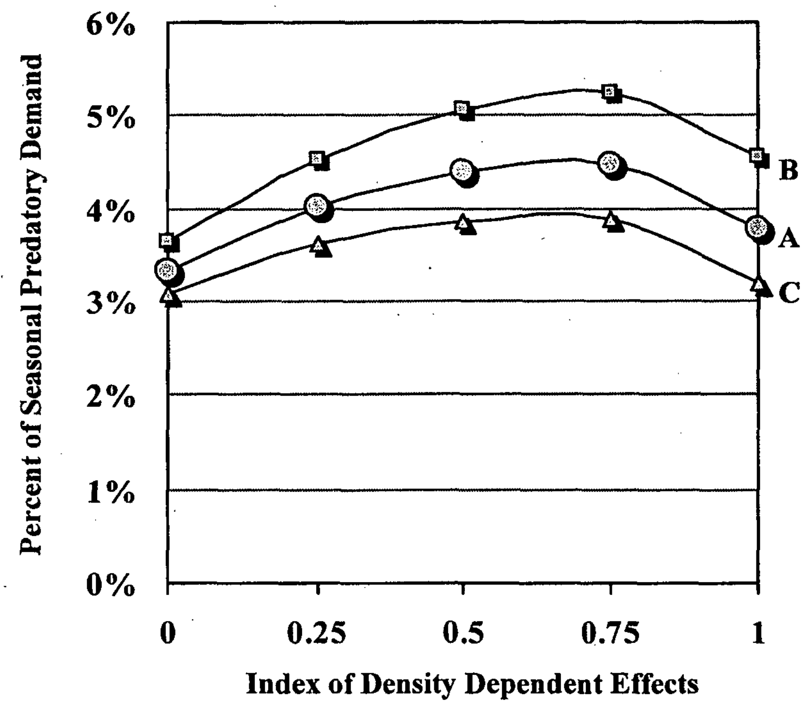


Figure 4. Estimates of the percent of seasonal predatory demand of age 1 through age 13 Hudson River striped bass potentially satisfied by consumption of juveniles of the four taxa, as functions of the index of density dependent effects (see text) and assumed background mortality rates. Curve A is for the estimated background mortality rates (see text), curve B is for background mortality rates of zero, and curve C is for two times the estimated background mortality rates. The proportion of the coastwide population of age 1 through age 13 striped bass that were Hudson River fish was assumed to be 13% in Period 1 and Period 2.



Appendices

Appendix A: Derivation of Approximations

The approximations were based on the following equivalences:

$$\frac{R_c}{R_n}=\frac{\left(\dfrac{\overline{C}_2}{\overline{C}_1}\right)}{\left(\dfrac{\overline{N}_2}{\overline{N}_1}\right)}=\frac{\left(\dfrac{\overline{C}_2}{\overline{N}_2}\right)}{\left(\dfrac{\overline{C}_1}{\overline{N}_1}\right)} \qquad \text{(A1)}$$

and

$$\frac{R_n}{R_l}=\frac{\left(\dfrac{\overline{N}_2}{\overline{N}_1}\right)}{\left(\dfrac{\overline{L}_2}{\overline{L}_1}\right)}=\frac{\left(\dfrac{\overline{N}_2}{\overline{L}_2}\right)}{\left(\dfrac{\overline{N}_1}{\overline{L}_1}\right)} \qquad \text{(A2)}$$

The first order Taylor series approximation (evaluated at $m_{P,j}=0$) for the numerator (with $j=2$) or denominator (with $j=1$) of equation (A1), that expresses that term as a function of the mortality rate for predation, is:

$$\frac{\overline{C}_j}{\overline{N}_j} \doteq m_{p,j} t w_j \left(\frac{1-e^{(-m_j t)}}{m_j t}\right)^{-1}\left(\frac{1-e^{-(m_j-g_j)t}}{(m_j-g_j)t}\right) \qquad \text{(A3)}$$

which, using the approximation from Paloheimo (1961) can be written as:

$$\frac{\overline{C}_j}{\overline{N}_j} \doteq m_{p,j} t w_j \left(e^{-\frac{m_j t}{2}}\right)^{-1}\left(e^{-\frac{(m_j-g_j)t}{2}}\right) \doteq m_{p,j} t w\, e^{\left(\frac{gt}{2}\right)} \qquad \text{(A4)}$$

Therefore, an approximation for the ratio of ratios in equation (A1) is:

$$\frac{R_c}{R_n} \doteq \left(\frac{m_{p,2}}{m_{p,1}}\right)^{\prime} \alpha \tag{A5}$$

where α is the ratio (Period 2 to Period 1) of the average juvenile weight per fish at the mid-point of the season.

Again using the approximation from Paloheimo (1961), the numerator (with j=2) or denominator (with j=1) of equation (A2) was approximated as:

$$\frac{\overline{N}_j}{\overline{L}_j} \doteq e^{-(m'_j + m_{p,j})t'} e^{-\frac{(m_j + m_{p,j})t}{2}} \tag{A6}$$

Therefore, the logarithm of the ratio of ratios in equation (A2) is approximately:

$$\ln\left(\frac{R_n}{R_l}\right) \doteq \left(m_{p,1} - m_{p,2}\right)\left(t' + \frac{t}{2}\right) + \beta \tag{A7}$$

where β is the difference between the juvenile background mortality rates for Period 1 and Period 2.

Appendix B: Formulae for Variance Estimates

Formulae for variance estimates for the approximate estimates of instantaneous mortality rates due to possible predation were derived using a Taylor series approximation (Kendall and Stuart 1977). Because the variances were intended to represent imprecision due to sampling error, and data for the three component ratios are from independent sampling programs, all covariance terms were set to zero. Lower case symbols (e.g. r_n) indicate estimates of corresponding paramters (e.g. R_n).

For the approximate estimate of the instantaneous mortality rate for Period 1:

$$\hat{\tilde{m}}_{p,1} = \frac{\ln\left(\dfrac{r_n}{r_l}\right)}{\left(1-\dfrac{r_n}{r_p}\right)\left(t'+\dfrac{t}{2}\right)}, \tag{B1}$$

the formula for the variance estimate is:

$$\operatorname{var}(\hat{\tilde{m}}_{p,1}) \doteq \left(\frac{dm_{p1}}{dr_n}\right)^2 \operatorname{var}(r_n) + \left(\frac{dm_{p1}}{dr_l}\right)^2 \operatorname{var}(r_l) + \left(\frac{dm_{p1}}{dr_p}\right)^2 \operatorname{var}(r_p) \tag{B2}$$

where

$$\frac{dm_{p1}}{dr_n} = \left(\left(r_n^{-1}\left(1-\frac{r_p}{r_n}\right)^{-1}\right) - \left(\left(1-\frac{r_p}{r_n}\right)^{-2} r_n^{-2}\, r_p\right)\right)\left(t'+\frac{t}{2}\right)^{-1} \tag{B3}$$

$$\frac{dm_{p1}}{dr_l} = -\left(r_l^{-1}\left(1-\frac{r_p}{r_n}\right)^{-1}\right)\left(t'+\frac{t}{2}\right)^{-1} \tag{B4}$$

and

$$\frac{dm_{p1}}{dr_p} = \left(r_n^{-1}\left(1-\frac{r_p}{r_n}\right)^{-2}\right)\ln\left(\frac{r_n}{r_l}\right)\left(t'+\frac{t}{2}\right)^{-1}. \tag{B5}$$

For the approximate estimate of the instantaneous mortality rate for Period 2:

$$\hat{\tilde{m}}_{p,2} = \frac{\ln\left(\dfrac{r_n}{r_l}\right)}{\left(\dfrac{r_n}{r_p}-1\right)\left(t'+\dfrac{t}{2}\right)}, \tag{B6}$$

the formula for the variance estimate is:

$$\operatorname{var}(m_{P2}) \doteq \left(\frac{dm_{p2}}{dr_n}\right)^2 \operatorname{var}(r_n) + \left(\frac{dm_{p2}}{dr_l}\right)^2 \operatorname{var}(r_l) + \left(\frac{dm_{p2}}{dr_p}\right)^2 \operatorname{var}(r_p) \tag{B7}$$

where

$$\frac{dm_{p2}}{dr_n} = \left(\left(r_n^{-1} \left(\frac{r_n}{r_p} - 1 \right)^{-1} \right) - \left(\left(\frac{r_n}{r_p} - 1 \right)^{-2} r_p^{-1} \ln\left(\frac{r_n}{r_l} \right) \right) \right) \left(t' + \frac{t}{2} \right)^{-1} \qquad \text{(B8)}$$

$$\frac{dm_{p2}}{dr_l} = -\left(r_l^{-1} \left(\frac{r_p}{r_n} - 1 \right)^{-1} \right) \left(t' + \frac{t}{2} \right)^{-1} \qquad \text{(B9)}$$

and

$$\frac{dm_{p2}}{dr_p} = \left(r_p^{-2} r_n \left(\frac{r_n}{r_p} - 1 \right)^{-2} \right) \ln\left(\frac{r_n}{r_l} \right) \left(t' + \frac{t}{2} \right)^{-1} . \qquad \text{(B10)}$$

Estimated variances for the component ratios (r_n, r_p, and r_l) were computed using the following formulation (using r_n as an example):

$$\text{var}(r_n) \doteq \left(\frac{\bar{n}_2}{\bar{n}_1} \right)^2 \left(\frac{\text{var}(\bar{n}_2)}{\bar{n}_2^2} + \frac{\text{var}(\bar{n}_1)}{\bar{n}_1^2} \right) \qquad \text{(B11)}$$

where

$$\text{var}(\bar{n}_j) = \frac{1}{k^2} \sum_{i=1}^{k} \left(se(n_{ji}) \right)^2 \qquad \text{(B12)}$$

$$\bar{n}_j = \frac{1}{k} \sum_{i=1}^{k} n_{ji} \qquad \text{(B13)}$$

for year i within period j; and

$$r_n = \frac{\bar{n}_2}{\bar{n}_1} \qquad \text{(B14)}$$

Estimates of standard errors (for equation (B12)) for estimates of juvenile and post yolk-sac larval abundance were from the annual Year Class Reports (e.g. EA 1996). For estimates of predatory demand, estimates of standard errors were based on reported coefficients of variation for estimates of age-specific abundance of Atlantic coast striped bass (ASMFC 2005). Parameters other than abundance were treated as constants in the variance estimates.

Appendix C: Estimates of the Proportion of the Coastwide Population of Age 1 Striped Bass from the Hudson River

The proportion of the coastwide population of age-1 striped bass that was of Hudson River origin was estimated from: 1) the time series of estimates of age-1 abundance ($N_{1,y}$), and 2) the time series of juvenile abundance indices for four major spawning areas: Chesapeake Bay Maryland (CBM), Chesapeake Bay Virginia (CMV), Hudson River (HR), and Delaware River (DR). For each year, y, the proportion was estimated as:

$$\rho_y = \frac{\hat{\beta}_{HR} X_{HR,y}}{\hat{\beta}_{CBM} X_{CBM,y} + \hat{\beta}_{CBV} X_{CBV,y} + \hat{\beta}_{HR} X_{HR,y} + \hat{\beta}_{DR} X_{DR,y}} \qquad \text{(C1)}$$

where the $\hat{\beta}'s$ are the estimated regression coefficients from a multiple regression of age-1 coastwide abundance against the year-specific juvenile indices ($X_{CBM,y}$, $X_{CBV,y}$, $X_{HR,y}$, $X_{DR,y}$) from the four spawning areas (ASMFC, 2005):

$$N_{1,y} = \beta_{CBM} X_{CBM,y} + \beta_{CBV} X_{CBV,y} + \beta_{HR} X_{HR,y} + \beta_{DR} X_{DR,y} \qquad \text{(C2)}$$

The R^2 for this multiple regression was 0.96 ($p<0.0001$).

APPENDIX D

Appendix D

Prepared by:

Webster Van Winkle
Van Winkle Environmental Consulting Co.

John Young
ASA Analysis & Communication, Inc.

Entrainment Susceptibility at Indian Point and Change in YOY Abundance

Cooling water withdrawals impose some incremental mortality on species susceptible to entrainment. The effect of this incremental mortality may be inconsequential to the populations and communities in the water body, or, if the increment is large enough, could potentially lead to either a decrease or a reduced rate of increase in the affected populations. However, in addition to cooling water withdrawals, there are many other factors that can affect population trends, including changes in prey and predator populations, climatic effects, harvesting intensity, habitat modification, invasive species, and water quality. Thus, over any given time period, populations of some species can be expected to increase, while others decrease, regardless of cooling water withdrawals.

If entrainment at IP2 and IP3 were having an adverse impact on the Hudson River fish community, then species with high susceptibility to entrainment would be expected to have decreased, or increased less in abundance, over the past 32 years than would species with low susceptibility. This possibility can be evaluated by examining the relationship between a measure of entrainment susceptibility and a measure of population change derived by comparing the mean abundance of young-of-year ("YOY") fish belonging to various species from 1974-1989 to the mean abundance of the same species of fish from 1990-2005. YOY is selected for the metric because the effects of entrainment have been realized by the time fish reach the YOY stage, and this age group is still within the estuary and can be sampled for most species. The periods 1974-1989 and 1990-2005 were selected so that the two periods of comparison would include equal numbers of years.

Evaluating the relationship between entrainment susceptibility and change in YOY abundance requires selecting those species for which adequate data are available for both variables. Entrainment susceptibility can be characterized quantitatively by evaluating the distribution of entrainable life stages in the Regions from which IP2 and IP3 withdraw water in comparison to all the Regions sampled. The expected effect of continued annual entrainment losses of early life stages, if losses are severe enough to affect population size, is a negative relationship between entrainment susceptibility and the ratio of YOY abundance from the early part of the time series (1974-1989) to the latter part (1990-2005).

METHODS

The process for evaluating the relationship between entrainment susceptibility and changes in YOY abundance is summarized in Figure D-1. The process involves three steps:

(1) Calculate a species-specific metric of entrainment susceptibility based on larval abundance data from the LRS;

(2) Calculate a species-specific metric of change in YOY abundance based on data from the BSS/FSS; and

(3) Determine if entrainment susceptibility is negatively related to change in YOY abundance.

Step 1. Entrainment Susceptibility Based on Larval Distribution (*EntSus*)

A species-specific metric of entrainment susceptibility is calculated from the utilities' LRS for the 32-year period 1974-2005.[1] Species using the Hudson River estuary as a spawning and nursery area vary by season within a year. In addition, the geographic and temporal extent of the LRS sampling varies among years, and some species occur in two or three seasonal periods. These realities are addressed by dividing the LRS database into three seasonal periods and considering only those weeks that were sampled:

- Winter & early spring: Years 1975-1980 and 1995-2005; Weeks 8-16; Regions 1-6
- Late spring: Years 1974-2005; Weeks 17-27; Regions 1-12
- Summer: Years 1991-2005; Weeks 28-41; Regions 1-7

Identification of larvae to species level is not always practical, in which case larvae are classified by genus or family. Differences in taxonomic level of *EntSus* and YOY abundance data are resolved in one of two ways: (a) if BSS/FSS data are adequate at species level but LRS data are not, then use the same genus or family *EntSus* value for each species, or (b) if BSS/FSS

---

1 An index of standing crop (the number of fish in an area or volume at a particular time) is estimated by life stage and species. Standing crop indices are calculated for each habitat (shorezone, benthic, water column) in each region and each week by taking the product of the average density in a habitat during that week and the area (shorezone habitat) or volume (benthic and water column habitats) contained in that region. The standing crop index for each region and week is then estimated as the sum of the habitat index values. This value is an index rather than an absolute standing crop value because no adjustment is applied for differences in collection efficiency between sampling gears (ASA, 2005; Chapter 2, Materials and Methods, 2004 Year Class Report).

data are not adequate at species level but LRS data are, then pool species-level LRS abundance data to the genus or family taxonomic level.

Relative abundance of larvae in Regions 3-5, *EntSus*, is the index of entrainment susceptibility. For each sampled year (and each seasonal period when possible), *EntSus* is estimated for each species as the ratio of standing crop in Regions 3-5 to standing crop in all sampled regions. For those species occurring in more than one of the three seasonal periods, annual *EntSus* values are calculated as an average across periods, *p*, weighted by abundance for each period:

$$EntSus_i = \frac{\sum_p SC_{ip} EntSus_{ip}}{\sum_p SC_{ip}}$$

where $EntSus_i$ = fraction of species in the Hudson River estuary in Regions 3-5 in year i
SC_{ip} = sum of abundance of the species within seasonal period p in year i
$EntSus_{ip}$ = value of *EntSus* for seasonal period p in year i

Annual *EntSus* values are estimated for each species for each year in which the species occurred during 1974-2005. Mean entrainment susceptibility and its variance are calculated for each species based on its annual *EntSus* values.[2]

Step 2. Change in YOY Abundance (*R*)

The utilities' Beach Seine Survey (BSS) and Fall Shoals Survey (FSS) programs are selected as the best measures of change in abundance of YOY fish. These programs have sampled the estuary using similar gear and methodology since 1974, although there have been variations in the Regions sampled and in time of initiation and end of the sampling across the years. To maintain consistent sampling effort and maximize comparability of results, data are restricted to Regions 1-12 and weeks 31-42, approximately corresponding to August through October.

Abundance data by species are categorized into two salinity zones, three habitats, and two time periods. The two salinity zones are brackish (Regions 1-6; river miles 12-61) and freshwater (Regions 7-12; river miles 62-152). The three habitats sampled by these surveys are:

2 Entrainment susceptibility at Indian Point will change during extreme water years. In wet years some freshwater and anadromous species will be more at risk, while in dry years some marine species will be more at risk.

(a) shorezone (bottom area in water 10 ft or less in depth), sampled with the 100-ft beach seine in the BSS from 1974-2005; (b) benthic (volume of water between river bottom and 3 ft above the bottom), sampled with the beam trawl in the FSS from 1985-2005; and (c) water column (water volume not included in either the shorezone or benthic habitats), sampled with the Tucker trawl in the FSS from 1979-2005. Except for weekly BSS sampling in the 1970s, all of the sampling was done on an alternate week basis.

Time series of abundance data are divided into two periods: Period 1 = 1974-1989; Period 2 = 1990-2005. This division results in equal number of years in the two periods for shorezone habitat (16 years), but unequal number of years for benthic habitat (five years and 16 years) and water column habitat (11 years and 16 years).

The available data for measuring change in abundance provide the potential for six independent estimates of relative abundance change for each species (two salinity zones and three habitats). However, some species may be concentrated in particular habitats or salinity zones. Due to the strong salinity preferences of freshwater and marine fish, only sampling from their preferred salinity zone (freshwater zone for freshwater fish, brackish zone for marine fish) was used. In addition, it is difficult to accurately measure abundance changes for species that occur only occasionally. Thus, species data from a salinity zone-habitat combination are included in the analysis only if the total catch meets a minimum level of catch in at least one of the two periods (see Step 3 below). To adjust for the unequal number of years for benthic and water column habitats mentioned above, the Period 1 catch is adjusted upward by a factor based on the number of years sampled, i.e., 3.20 (=16 yr/5 yr) for benthic and 1.45 (=16 yr/11 yr) for water column.

For each selected salinity zone-habitat, the weighted mean YOY abundance for Period 1, Period 2, and Periods 1 and 2 combined are calculated with the GLM procedure in SAS. Mean abundance for each of these three time intervals is calculated as the weighted mean abundance across the sampling Regions within a salinity zone, where the weight is the proportion of the total amount of a habitat in that salinity zone that occurs within each of its six Regions.

Relative change in YOY abundance for each species, R_i, and its standard error, $se(R_i)$, are calculated based on (Cochran 1977, pp. 30-34)[3]. Since R_i is bounded on the lower side by 0 for

[3] Let:

$\bar{x}_{1ijk}$ = weighted mean cpue in Period 1 for species i in habitat j in salinity zone k decreases in abundance, is 1 if mean abundance is unchanged, and is unbounded above 1 for increases in abundance, a log_{10} transformation is used to normalize the distribution of R values.[4]

$$R_i = \frac{\sum_{jk} \bar{y}_{2ijk} / n_i}{\sum_{jk} \bar{y}_{1ijk} / n_i} = \frac{\bar{y}_{2i}}{\bar{y}_{1i}} = \text{relative change in species i abundance from Period 1 to Period 2}$$

$$se(R_i) = \frac{1}{\sqrt{n_i}\,\bar{y}_{1i}} \sqrt{\frac{(\sum_{jk} \bar{y}_{2ijk}^2 + R_i^2 \sum_{jk} \bar{y}_{1ijk}^2 - 2R_i \sum_{jk} \bar{y}_{1ijk}\bar{y}_{2ijk})}{n_i - 1}}$$

Step 3. Association between Entrainment Susceptibility and Change in YOY Abundance

Three correlation methods (Pearson, Spearman, and Kendall) are used to evaluate the association between *EntSus* and YOY abundance change using the CORR procedure in SAS. There is no simple mathematical relation between any two of these three methods. When the true correlation coefficient is not zero, it is likely that each coefficient is sensitive to different types of departures from independence (Sokal and Rohlf, 1995).

Availability of data varies among species, and results of correlation analysis could be sensitive to how many species are included in the analysis. Thus a limited sensitivity analysis is performed to evaluate to what extent the correlation results depend on selection criteria. The approach to this sensitivity analysis is to define two cases, Case A and Case B. The species in

---

$\bar{x}_{2ijk}$ = weighted mean cpue in Period 2 for species i in habitat j in salinity zone k

$\bar{x}_{\bullet ijk}$ = weighted mean cpue over both Periods for species i in habitat j in salinity zone k

$\bar{y}_{1ijk} = \bar{x}_{1ijk} / \bar{x}_{\bullet ijk}$ = relative mean cpue in Period 1 for species i in habitat j in salinity zone k

$\bar{y}_{2ijk} = \bar{x}_{2ijk} / \bar{x}_{\bullet ijk}$ = relative mean cpue in Period 2 for species i in habitat j in salinity zone k

n_{1i} = number of salinity zone-habitat combinations selected for species i in Period 1.

n_{2i} = number of salinity zone-habitat combinations selected for species i in Period 2.

4 The effectiveness of estimating change in YOY abundance from Period 1 to Period 2 based on BSS/FSS data is limited for some species because these surveys do not sample some habitats that are primary habitats for YOY (i.e., tributaries, bays, wetlands, or shorezone habitat with structure). Although R integrates BSS/FSS YOY abundance data from benthic, water column, and shorezone habitats, the growth and survival of larvae and YOY fish that are most common in these unsampled habitats may be determined by factors that are largely irrelevant for species in the sampled habitats. Examples of such factors are micro-habitats suitable for parental nest building and guarding of young, protection from predators, and availability of food not present in open water habitats. Although species that frequent these habitats exclusively or primarily are not adequately sampled compared to other Hudson River species, there is a relatively small amount of such unsampled habitats in the estuary, and these species are not likely to be affected by IP entrainment because of their preference for these unsampled habitats.

Case B are a subset of the species in Case A. Species in Case A are selected based on LRS data criteria for *EntSus* and on BSS/FSS data criteria for YOY abundance. Species are excluded from Case A to create Case B based on more restrictive criteria for both larval and YOY abundance data. Species selection decisions are made independently for each of these two variables. Thus, a species can be excluded from this evaluation even if data are adequate for one variable but not the other variable.

Species selection criteria for entrainment susceptibility based on larval abundance

Cases A and B. *EntSus* > 0, i.e., minimum of one larva in LRS samples from Regions 3-5 during 1974-2005.

Case A. Minimum average of 100 larvae per year of occurrence collected in LRS samples from Regions 1-12 during 1974-2005.

Case B. Minimum average of 1,000 larvae per year of occurrence collected in LRS samples from Regions 1-12 during 1974-2005.

Species and salinity-zone habitat selection criteria for change in YOY abundance[5]

Case A. Minimum of 100 YOY collected in BSS/FSS samples in at least one SZ-habitat in at least one of the two time periods.

Case B. Minimum of 1,000 YOY collected in BSS/FSS samples in at least one SZ-habitat in at least one of the two time periods.

RESULTS

Entrainment Susceptibility (*EntSus*)

EntSus is a measure of the proportion of larvae in those habitats sampled by the LRS that were collected in Regions 3-5 compared to Regions 1-12.[6] Twenty four (24) species meet the Case A selection criterion for *EntSus*.[7] For these 24 species, mean *EntSus* scores range from 0.45 for striped bass to 0.02 for American shad.[8]

---

5 Number of SZ-habitats selected can vary from 1 to 6 for anadromous and estuarine species and from 1 to 3 for freshwater and marine species. If a SZ-habit is selected for Period 1 (or 2), Period 2 (or 1) is included also.

6 The LRS does not sample in some habitats that are critical for many Hudson River fish species for spawning and larval life stages, e.g., tributaries, bays, wetlands, and shorezone habitat with structure.

7 Five of these 24 species are not selected for correlation analysis because they do not meet the Case A selection criterion for YOY abundance.

8 The list of species collected during the intensive entrainment study at Indian Point (1983-1987) was compared with the list of species collected during the 1974-2005 LRS in Regions 1-12. Four species, all marine, were collected only in the Indian Point entrainment study and not in the LRS. These species are not selected for the

Mean annual *EntSus* values for the representative species varied by more than an order of magnitude: striped bass (0.45), bay anchovy (0.42), Atlantic tomcod (0.26), white perch (0.16), alewife and blueback herring (0.05), and American shad (0.02). Most of these seven species were collected as larvae every year, although the average number of larvae collected per year of occurrence varied by two orders of magnitude from alewife/blueback herring (3×10^5) to American shad (2×10^3). Spottail shiner had fewer than 100 larvae/yr occurrence, and no *EntSus* value is calculated.

Change in YOY Abundance

Forty-six (46) species are selected based on the Case A criterion for YOY abundance. However, only 19 of these species are also selected based on the Case A criterion for larval abundance, and thus only these 19 species are selected for the *EntSus-R* correlation analysis.

Correlation Analysis

Table D-1 shows the correlation coefficients and probability values, for both Case A and Case B, for all three correlation indices. Figures D-2 and D-3 provide plots of mean entrainment susceptibility vs. the normalized index of relative change in YOY abundance from Period 1 to Period 2 for both Case A and Case B. For both Cases A and B, all three estimates of the correlation between $Log_{10}(R)$ and *EntSus* are not statistically significantly different from zero (Table D-1). This result is opposite the expected significant negative correlation if Indian Point entrainment were adversely affecting the population trends of susceptible species. Therefore, the effect of Indian Point entrainment on abundance patterns of the fish community, if there is one, is not large enough to be statistically detectable in the 32 years of monitoring data.

Nineteen (19) taxa, representing 31 species, four of the five guilds, 13 taxonomic families, and a broad range of both *EntSus* and *R* values (Table D-2, Figures D-1 and D-2) are selected for Case A. Eleven (11) of these taxa, representing 17 species, are retained in Case B.[9] Plots of *EntSus* vs. $Log_{10}(R)$ illustrate that more species decreased than increased in YOY

EntSus- R analysis. The species (and number of larvae collected) are Atlantic needlefish (3), smallmouth flounder (1), striped searobin (1), and northern searobin (1).

9 Eight taxa are excluded from Case A in creating Case B. The eight taxa are: Atherinid spp., banded killifish, gizzard shad, centrarchid spp, northern pipefish, rainbow smelt, winter flounder, and yellow perch. These taxa are excluded because of not meeting the more restrictive Case B selection criterion for larvae, YOY, or both.

abundance for both cases (Figures D-2 and D-3), but the change in abundance values (R) was only weakly associated with the magnitude of *EntSus* values.

DISCUSSION AND CONCLUSIONS

EntSus is a quantitative index bounded by 0.00 and 1.00. It is based on LRS data for larval abundance in water column and benthic habitats sampled in Regions 3-5 relative to larval abundance in these habitats sampled in Regions 1-12 of the Hudson River estuary. Thus, *EntSus* is an index of risk of entrainment of larvae at Indian Point. It is not an index of impact on the population.

The low correlations observed between *EntSus* and $Log_{10}(R)$ are counter to the expected more negative correlations if Indian Point entrainment were a significant factor influencing population dynamics of the fish community. Although the number of taxa (19) for which both variables could be measured is small, these taxa represent approximately 94% (Case A) and 88% (Case B) of all YOY fish captured in the BSS/FSS programs from 1974-2005.

In conclusion, 32 years of monitoring data do not support the hypothesis that entrainment at Indian Point has caused substantial harm to the fish community of the Hudson River estuary. Although more species have decreased than increased in YOY abundance over this time period, changes in abundance are unrelated to species susceptibility to entrainment at IP2 and IP3.

REFERENCES

ASA Analysis & Communication. 2005. 2003 Year Class Report for the Hudson River Estuary Monitoring Program. Prepared for Dynegy Northeast Generation, Entergy Nuclear Indian Point 2 LLP, Entergy Nuclear Indian Point 3 LLC, Mirant Bowline.

Cochran, W. G. 1977. Sampling Techniques. John Wiley & Sons, New York, New York

Sokal, R.R., and F.J. Rohlf. 1995. Biometry—The Principles and Practice of Statistics in Biological Research. 3rd Edition. W. H. Freeman and Company, San Francisco, CA.

Table D-1. Pearson, Spearman, and Kendall correlation coefficients for the association between $Log_{10}(R)$ and mean *EntSus*. A value of p represents the probability of a sample correlation coefficient larger than the observed sample correlation coefficient, if the true correlation coefficient is zero.

Case	N		Pearson	Spearman	Kendall
A	19	r	0.225	0.182	0.129
		p	0.355	0.457	0.442
B	12	r	0.157	-0.042	-0.046
		p	0.625	0.897	0.837

Table D-2. EntSus and $Log_{10}R$ values for Figures 1 and 2, including standard errors. Case A, 19 taxa; Case B, 12 taxa. Sorted by EntSus, low to high, for each case.

Case	Family	Guild	Taxon/Species	EntSus	SE EntSus	R	SE R	Log_{10} R
A	CLUP	A	American shad	0.023	0.009	0.480	0.091	-0.318
A	PLEU	M	Winter flounder	0.030	0.007	0.440	0.374	-0.357
A	CLUP	A	Alewife	0.051	0.008	1.133	0.337	0.054
A	CLUP	A	Blueback herring	0.051	0.008	0.582	0.101	-0.235
A	CLUP	F	Gizzard shad	0.072	0.049	2.011	0.671	0.303
A	SYNG	M	Northern pipefish	0.079	0.024	0.774	0.058	-0.111
A	CYPR	F	Cyprinid unid	0.107	0.013	1.154	0.076	0.062
A	PERC	F	Tesselated darter	0.109	0.012	0.971	0.149	-0.013
A	CENT	F	Centrarchid unid	0.116	0.015	2.271	1.609	0.356
A	MORO	E	White perch	0.158	0.013	0.440	0.072	-0.357
A	PERC	F	Yellow perch	0.201	0.024	0.551	0.197	-0.259
A	CYPD	E	Banded killifish	0.210	0.096	0.306	0.242	-0.515
A	OSME	A	Rainbow smelt	0.260	0.030	0.633	0.087	-0.198
A	GADI	E	Atlantic tomcod	0.263	0.042	0.400	0.134	-0.398
A	CLUP	M	Atlantic menhaden	0.300	0.046	80.026	35.284	1.903
A	SCIA	M	Weakfish	0.302	0.050	0.516	0.265	-0.287
A	ATHE	E	Atherinid sp.	0.339	0.032	3.509	2.487	0.545
A	ENGR	M	Bay anchovy	0.417	0.032	0.720	0.200	-0.142
A	MORO	A	Striped bass	0.454	0.020	1.236	0.380	0.092
B	CLUP	A	American shad	0.023	0.009	0.527	0.109	-0.278
B	CLUP	A	Alewife	0.051	0.008	1.267	0.574	0.103
B	CLUP	A	Blueback herring	0.051	0.008	0.582	0.101	-0.235
B	CYPR	F	Cyprinid unid	0.107	0.013	1.233	1.432	0.091
B	PERC	F	Tesselated darter	0.109	0.012	0.971	0.149	-0.013
B	MORO	E	White perch	0.158	0.013	0.459	0.094	-0.338
B	OSME	A	Rainbow smelt	0.260	0.030	0.821	0.129	-0.086
B	GADI	E	Atlantic tomcod	0.263	0.042	0.346	0.157	-0.461
B	CLUP	M	Atlantic menhaden	0.300	0.046	80.026	35.284	1.903
B	SCIA	M	Weakfish	0.302	0.050	0.398	0.294	-0.400
B	ENGR	M	Bay anchovy	0.417	0.032	0.720	0.200	-0.142
B	MORO	A	Striped bass	0.454	0.020	0.976	0.364	-0.011

Figure D-1. Analysis Flow Chart for Entrainment Susceptibility

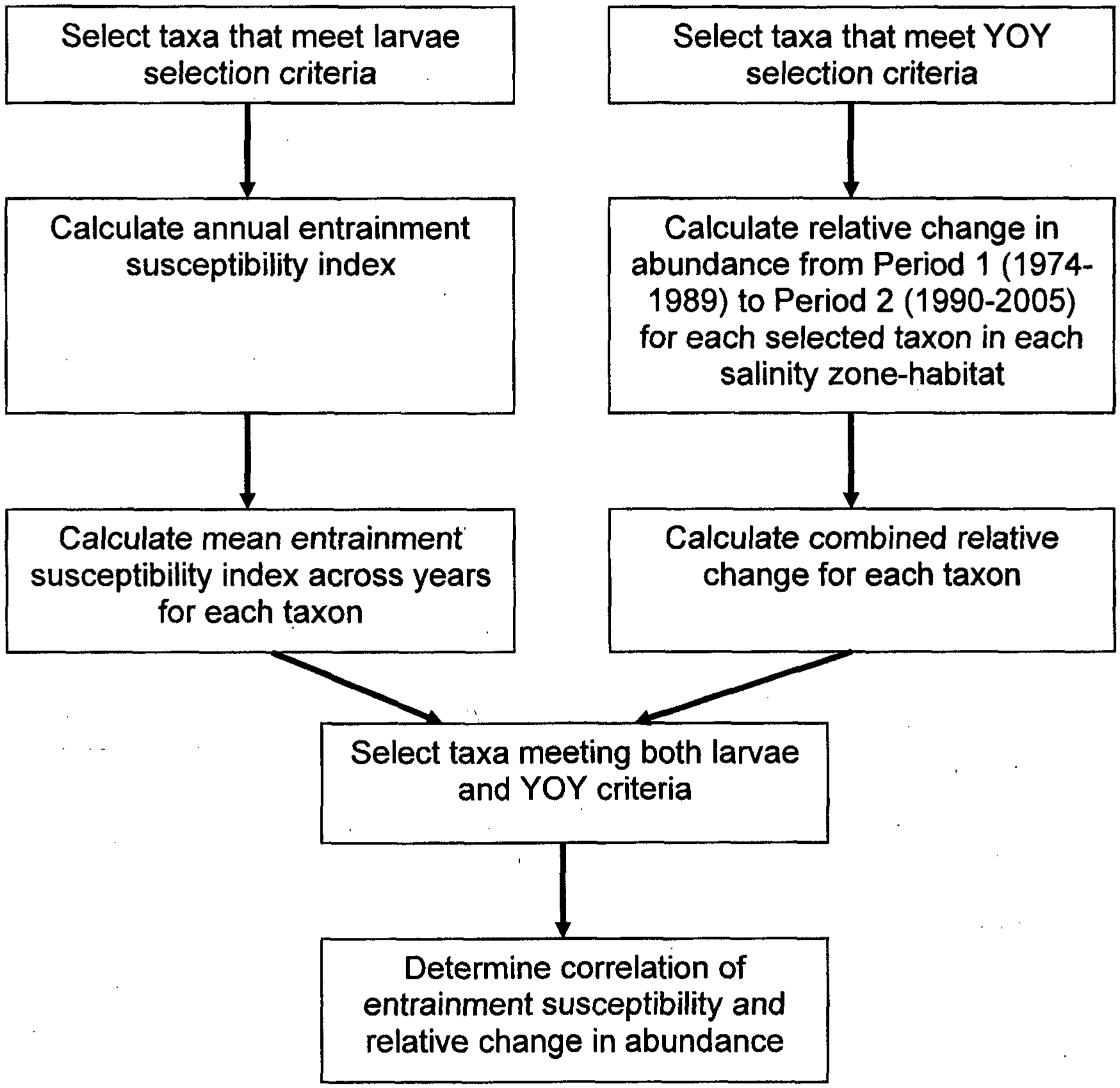


Figure D-2. Association between change in YOY abundance from Period 1 to Period 2, *Log₁₀(R)*, and entrainment susceptibility, *EntSus*, for the 19 taxa selected for Case A. Zero on the logarithmic Y axis corresponds to no change in YOY abundance. Use Table 2 as an aid in determining which species is associated with which point in the figure. N = 19; r = 0.16; P = 0.51

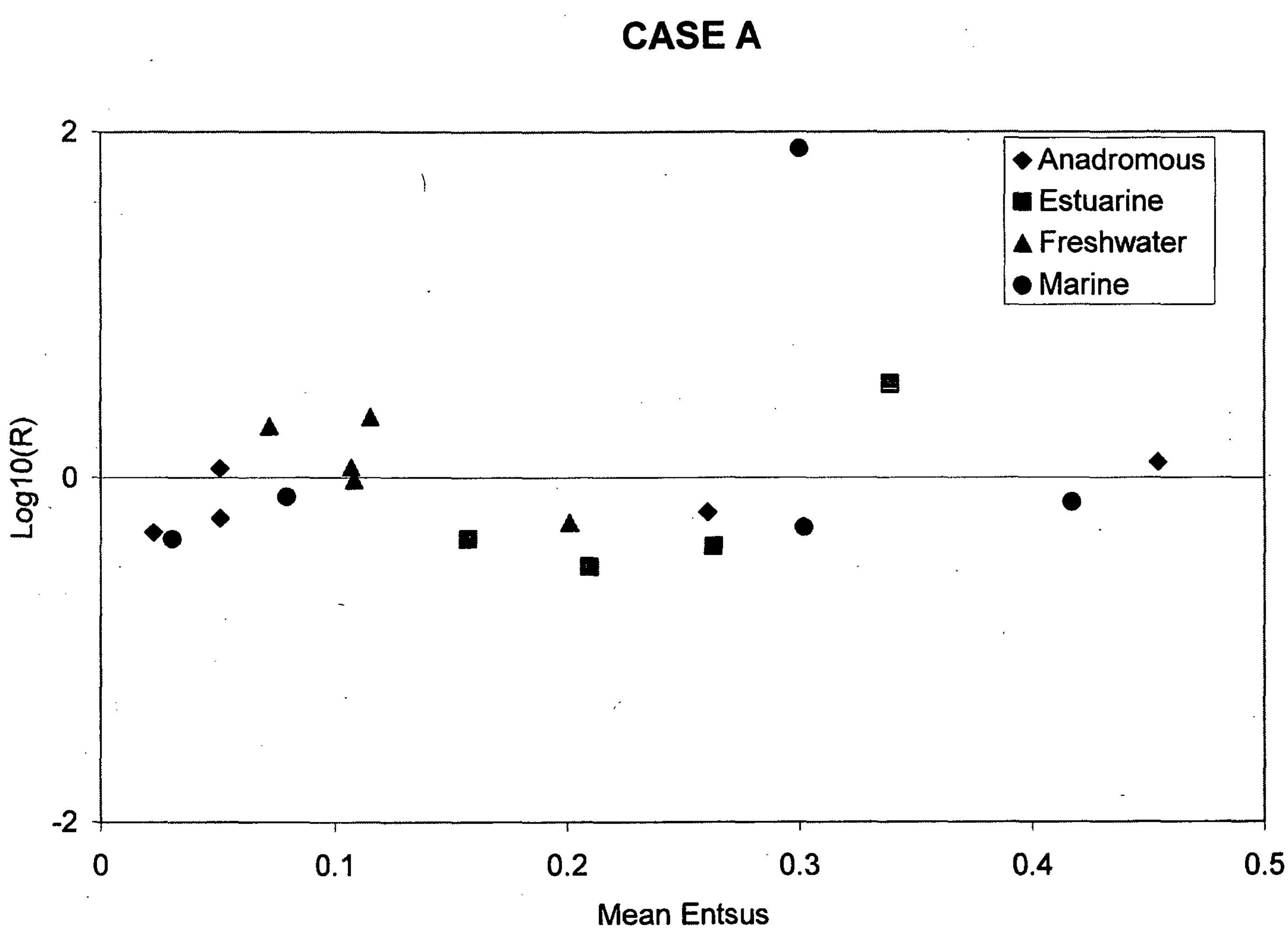


Figure D-3. Association between change in YOY abundance from Period 1 to Period 2, $Log_{10}(R)$, and entrainment susceptibility, *EntSus,* for the 11 fish taxa selected for Case B. Zero on the logarithmic Y axis corresponds to no change in YOY abundance. Use Table 2 as an aid in determining which species is associated with which point in the figure.