Monitoring the Marine Environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut

Annual Report 1994

Northeast
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Northeast Utilities Service Company
Corporate and Environmental Affairs
Aquatic Services Branch
N U Environmental Laboratory
April 1995

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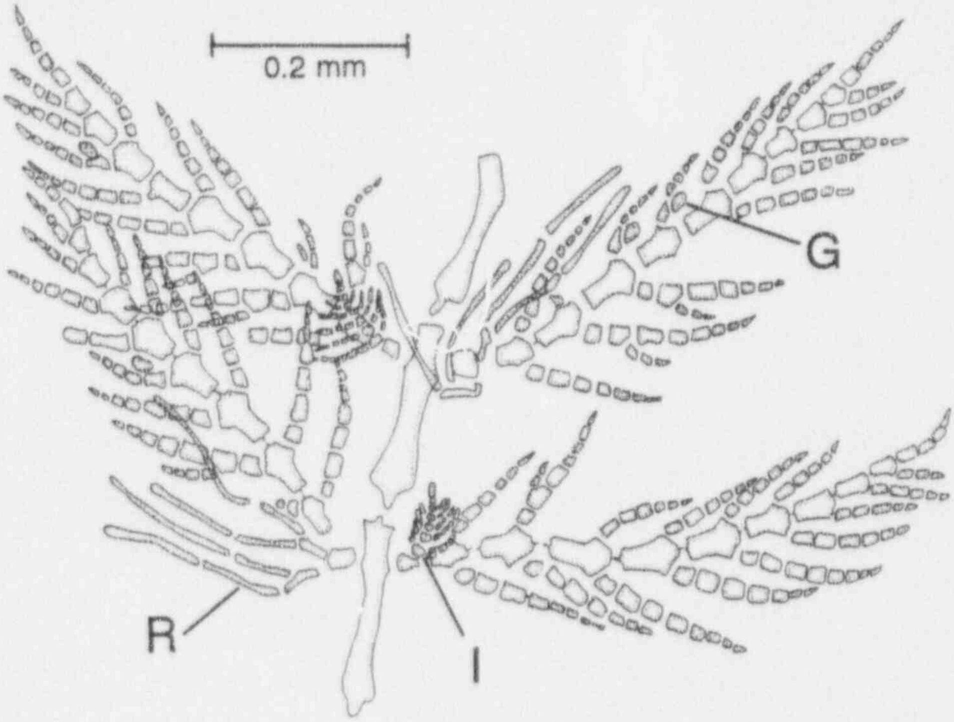

The drawing on the front cover represents a small piece of *Antithamnion pectinatum*, a small epiphytic red alga common on local rocky shores. Characteristic features include oppositely attached branches, gland cells (G) adjacent to at least two cells of the branchlets, and rhizoids (R) and undeveloped indeterminate branches (I) arising from the basal cells of the determinate branches. The latter two features permit fragments of *A. pectinatum* to reattach to intertidal macroalgae, and grow into entire new plants.

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1994 Annual Report

Prepared by: Staff of
Northeast Utilities Service Company
Corporate and Environmental Affairs
Aquatic Services Branch
NU Environmental Laboratory

Approved by: Milan Keser
Dr. Milan Keser

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Dr. Milan Keser, Manager
Ronald C. Nichols, Supervisor

John A. Castleman
Donald J. Danila
David G. Dodge
Christine P. Gauthier
Donald F. Landers
J. Dale Miller
John T. Swenarton
David P. Colby
Gregory C. Decker
James F. Foertch
Raymond O. Heller
Dr. Ernest Lorda
Douglas E. Morgan
Joseph M. Vozarik

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Executive Summary

Winter Flounder Studies

The local Niantic River population of the winter flounder (*Pleuronectes americanus*) is potentially affected by the operation of MNPS, particularly by entrainment of larvae through the cooling-water systems of the three operating units. As a result, intensive studies of the life history and population dynamics of this valuable sport and commercial species have been undertaken since 1976.

Because of heavy ice cover in the Niantic River in early 1994, the adult winter flounder survey did not begin until March 22, the latest start in 19 years, and only 4 weeks of sampling were completed. The median trawl catch-per-unit-effort (CPUE) of fish larger than 15 cm present during the spawning season was 4.5. This value was more than twice the record low CPUE of 1.9 for 1993, but nonetheless remains as the second smallest CPUE for this time-series. The Jolly stochastic model was used with mark and recapture data to estimate the absolute abundance of the adult spawning population (all winter flounder larger than 20 cm, which includes some immature fish). The most recent abundance estimate was 11,779 winter flounder for 1993, in contrast to estimates between 33 and nearly 80 thousand fish for 1984-91.

About one-third to one-half of the winter flounder found in the Niantic River during the spawning period each year are mature females. Annual female winter flounder parental stock sizes were estimated using available information on sex ratios, age, and size composition. These estimates have ranged from 7,821 (1993) to 78,629 (1982) spawning females, with corresponding total egg deposition ranging from about 6.4 to 45.6 billion.

Estimates of larval winter flounder abundance at the MNPS discharge (entrainment sampling) have been obtained since 1976, at a station in mid-Niantic Bay since 1979, and at three stations in the Niantic River since 1983. The low abundance of newly-hatched larvae in Niantic Bay compared to the Niantic River suggested that most local spawning occurred in the river. In addition, abundance indices of Stage 1 larvae in the river were significantly correlated with independent estimates of female spawner egg production. Abundance of Stage 1 and 2 larvae in the Niantic River during 1994 were among the lowest since sampling began in 1983. Abundances of Stage 3 and 4 larvae, however, were greater and above the long-term average. Annual larval abundances in the bay since 1976 appeared to reflect region-wide trends, because they were highly correlated to abundance indices for Mount Hope Bay, MA and RI.

Larval developmental stage and length were closely related. Smaller larval size-classes predominated in the river and larger size-classes were more prevalent in the bay. In Niantic Bay, growth and development were correlated with water temperature. In the river, growth appeared to be related to both water temperature (positively) and larval density (negatively). Estimated mortality of larvae in the Niantic River for 1984-94 ranged from about 82 to 98%. Mortality was consistently highest during Stage 2 of development (3- to 4-mm size-classes), which is when feeding first occurs. This stage may include a "critical period" for winter flounder as survival rates generally improve progressively for larger size-classes. Density-dependence was examined using a function comparing mortality and egg production estimates. With 1993 data excluded, a significant positive relationship was apparent, such that when egg production and larval abundance increased, mortality also increased.

Young-of-the-year winter flounder have been collected during late spring and summer at two stations in the Niantic River since 1983. Corresponding to late larval abundance, densities of newly metamorphosed young in 1994 were relatively high. Median densities for late summer were among the largest found in 12 years, with the 1994 year-class having the potential to be relatively strong.

An index of abundance was calculated for young winter flounder taken during the late fall and early winter at the trawl monitoring program stations. The 1993-94 abundance index (1993 year-class) was 7.4, the lowest recorded since 1987-88. Since 1983, when comparative data were available, this abundance index was significantly correlated with that of young fish taken in the Niantic River during summer. High indices for the 1988 and 1992 year-classes indicated that these were relatively abundant year-classes. Few juveniles were taken within the Niantic River during the adult spawning population surveys in recent years. Young-of-the-year abundance indices were not significantly correlated or were negatively correlated with those for age-3, 4, and 5 female adult spawners. Thus, none of the early life stages were considered as reliable indices of year-class strength for Niantic River winter flounder stock.

Egg production estimates from annual spawning surveys were scaled to numbers of spawning females and used as recruitment indices. These indices together with adult female spawning stock estimates and mean annual February water temperatures were used to fit a three-parameter Ricker stock-recruitment relationship (SRR). Additionally, the indirect estimate of the winter flounder theoretical rate of increase (the SRR α parameter) derived by the Connecticut Department of Environmental Protection (DEP) was used for modeling the dynamics of the winter flounder population for impact assessment purposes. The value of α, re-scaled to units of fish numbers from biomass units, was estimated as 5.42 and described the inherent potential for increase of the Niantic River winter flounder stock. The estimate of β (the second SRR parameter), which describes the annual rate of compensatory mortality as a function of stock size, showed little annual variation since 1988. The third and last parameter in the SRR described a negative relationship between winter flounder recruitment and water temperatures in February, the month when most spawning, egg incubation, and hatching occur.

The number of larvae entrained through the condenser cooling-water system at MNPS is the most direct measure of potential impact on winter flounder. Annual estimates of entrainment were related to larval densities in Niantic Bay, as well as to plant operation. The entrainment estimate for 1994 of 182.1 million was about average since three-unit operation began in 1986. Unit 1 was shut down from January 15 through May 23 because of a refueling outage. This decrease in cooling water use resulted in a calculated reduction in entrainment of about 21% (48.1 million larvae) from that expected if Unit 1 had operated during this period.

The impact of larval entrainment on the Niantic River stock depends upon the fraction of its production that is entrained each year. Empirical mass-balance calculations for 1984-94 indicated that a large number of entrained larvae came from areas of Long Island Sound other than the Niantic River. An estimated 14 to 38% of the larvae entrained at MNPS appeared to have originated from the Niantic River during these years. Percentages of the river production that were entrained annually ranged from about 5 to 22% and the estimated fraction of Niantic River winter flounder production that would have been entrained under full (100% capacity) three-unit operation ranged from 5.4 to 25.0% (geometric mean of 12.1%).

A computer simulation model (SPDM) was used for long-term assessments of MNPS impact. Input data used by the model included basic life-table parameters, the three-parameters of the SRR, February water temperature statistics, and simulation parameters specific to each model run, including a random variability component. Conditional mortality rates corresponding to postulated larval entrainment and juvenile and adult impingement at MNPS were simulated according to historical information and projections. Fishing mortality rates (F) were provided by the DEP. Initially, F was set at 0.40 and remained unchanged through the 1960s; increased gradually to 0.62 by 1988 and thereafter more rapidly to a maximum of 1.30 in 1991. Based on proposed regulatory changes, F was projected to decrease substantially through the late 1990s, dropping to 0.50 by 2001, and remaining unchanged through the rest of the simulation years. The winter flounder stock was simulated as female spawner biomass (in lbs), which is more directly related to reproductive potential than fish numbers. Annual rates of Niantic River winter flounder larval entrainment were based on actual or estimated MNPS cooling-water flow and estimated or projected entrainment as derived from mass-balance calculations. All SPDM runs were stochastic and their output consisted of 100 Monte Carlo replicates for each yearly stock projection over a 100-year period (1960-2060).

An initial stock size of 98,104 lbs was used to represent the theoretical (no fishing assumed) maximum spawning potential (MSP) of the Niantic River female spawning stock. When fishing effects were simulated, the annual projections of the initially unfished stock become the baseline time series of annual spawning biomass for Niantic River winter flounder subjected to fishing, but in the absence of any plant impact. Under the exploitation rates simulated, the stochastic mean stock size of the baseline declined to 49,476 lbs in 1970 and to its lowest point of 12,907 lbs in 1993. The latter value was about one-half of a generally accepted critical stock size, defined as 25% of MSP. Following simulated reductions in fishing, however, the stock rapidly recovered. A new series of stock size projections were then simulated by adding the effect of larval entrainment at MNPS. The lowest projected stock biomass under simultaneous fishing and effects of MNPS occurred in 1993 (10,947 lbs), whereas the greatest absolute decline relative to the baseline occurred in 2003 (a difference of 7,026 lbs). Generally, however, greater reductions in stock biomass resulted from fishing than from larval entrainment, because fishing tends to remove larger fish and reduce average weight of the remaining spawners. The simulated spawning stock returned to within 1,000 lbs of baseline levels (44,000 lbs, or 45% of MSP) about 6 years after the scheduled termination of Unit 3 operation in 2025.

The probabilities that the Niantic River female spawning stock biomass would fall below selected reference sizes (25, 30, and 40% of MSP) were determined to help assess the long-term effects of MNPS operation. A stock smaller than 25% of MSP is considered overfished, whereas one that is at 40% of MSP can maximize yield to the fisheries while remaining stable. For both baseline and MNPS-impact simulations, it was likely ($p \geq 0.95$) that the stocks were greater than 40% of MSP in 1970. At the lowest point of both stock projections in the mid-1990s, all replicates were less than 25% of MSP. Simulated reductions in fishing allowed for a rapid increase in spawner biomass in 2000. By 2010, the stocks had a high ($p \geq 0.95$) likelihood of being greater than 30% of MSP and the impacted stock had a better than even chance of being greater than 40% of MSP. This recovery, however, assumed that changes in fishing regulations would be implemented as scheduled and that they achieved expected reductions in fishing mortality. Even with reductions in fishing mortality, there still was a 34% chance that the new equilibrium stock biomass would remain smaller than 40% of MSP.

Fish Ecology Studies

Studies of fish assemblages inhabiting the area around MNPS were conducted to determine the effects of station operations. These effects have been defined as power-plant related changes in the occurrence, distribution, and abundance of fish species which could affect community structure. Fish assemblages could be adversely affected by losses due to impingement of juvenile and adult fish on the intake screens, entrainment of fish eggs and larvae through the cooling water system, or by changes in thermal regime or physical habitats.

Since 1976, trawl, seine, and ichthyoplankton monitoring programs have been conducted to determine the impact of MNPS on local fish assemblages. Over 100 different fish taxa have been collected. Six taxa (American sand lance, anchovies, grubby, silversides, tautog, and cunner) were identified that were potentially impacted by MNPS, either by entrainment of their eggs and larvae or by exposure to elevated seawater temperatures.

Abundance data were analyzed separately for two-unit (1976-1985) and three-unit (1986-1994) operational periods and for the entire 18-year data series (both periods combined) to determine if changes in abundance have occurred. For the potentially impacted fishes, declining abundance trends were found for some life stages in four (American sand lance larvae, Atlantic silverside juveniles, tautog eggs, and cunner and tautog adults) of the six taxa. Because many factors may affect the abundance of these taxa the reasons for these declines were difficult to ascertain. American sand lance larvae ranked third among fish larvae entrained and have significantly decreased in abundance in entrainment samples. Decreases in sand lance abundance were also apparent in other areas of the Northwest Atlantic Ocean. Their abundance was noted to have been inversely correlated with that of Atlantic herring and Atlantic mackerel. Given the large abundance changes of this species along the Atlantic coast, effects of MNPS operation on sand lance were difficult to quantify. The bay anchovy is typically the most abundant ichthyoplankton species collected in estuaries within its range and it was the dominant larval taxon entrained at MNPS. Egg and larval densities and entrainment estimates for 1993 were within the historic range. Similar to the sand lance, this fish also exhibits large natural abundance fluctuations. Along the coast of Connecticut, the Atlantic silverside and the inland silverside are among the most common shore-zone species. Typical of short-lived species, the catches of Atlantic silverside by trawl and seine were highly variable and annual catch indices ranged over two orders of magnitude. The catches of Atlantic and inland silversides in seine samples were all within historic ranges and were greater than the two-unit period average, except for the Atlantic silverside at the Jordan Cove station. Unlike several other potentially impacted fishes, the grubby experiences no fishing pressure. Both larval and adult grubby abundance indices were stable throughout the 18 years of monitoring. The tautog was the second-most abundant egg taxon entrained and accounted for more than 30% of the total eggs collected since 1979. During the three-unit operational period there was a significant negative trend in the densities of tautog eggs. However, larval densities were within the historic range. Trawl catches of adult tautog exhibited a highly significant negative slope for the 18-year data series. Young-of-the-year tautog accounted for a high proportion of the fish caught in the trawl since three-unit operation began. The cunner was the most abundant egg taxon entrained and accounted for more than 50% of all eggs collected since 1979. The density of cunner eggs in 1993 was within the historic range, and the 1993 egg entrainment estimate was the highest since the three-unit operational period began. The densities of larvae were low, but were within the range of previous values. Comparable to tautog, young-of-the-year cunner accounted for a higher proportion of the fish caught in the trawl since three units began operating.

Entrainment of wrasse (cunner and tautog) eggs was identified as the primary potential impact to these fishes because over 85% of the eggs entrained at MNPS were of these species. The spatial distribution of wrasse eggs was studied in 1994. Results indicated that these eggs were not concentrated near MNPS intakes, but were representative of a more homogenous distribution, including areas outside Niantic Bay. This apparent widepsread standing stock of tautog and cunner eggs in the MNPS area would minimize any impact from entrainment loss.

Lobster Studies

One of the most valuable species in the Connecticut fishing industry is the American lobster (*Homarus americanus*). Annual Connecticut landings of 0.8 to 2.7 million pounds yielded between $2.4 and 8.4 million to lobstermen employed in the fishery. Lobsters are highly exploited throughout their range and the fishery is almost completely dependent on new animals molting into legal size each year. In Long Island Sound (LIS), more than 90% of all the lobsters above the minimum legal size are removed by fishing. Over the past decade, the lobster fishery has become highly regulated in an effort to reduce fishing mortality rates and to increase larval production and subsequent recruitment. Since 1984, Connecticut lobstermen have been required to install escape vents in traps; the escape vents allow sublegal-sized lobsters to escape from traps and thereby reduce injury and mortality to this portion of the population. The minimum legal size (carapace length) of lobster was increased in Connecticut in 1989 from 81.0 mm (3 $^{3}/_{16}$ in) to 81.8 mm (3 $^{7}/_{32}$ in); minimum size was increased again in 1990 to 82.6 mm (3 $^{1}/_{4}$ in). Because of the regional economic importance of lobsters, adult lobsters have been monitored from May through October since 1978 using wire lobster traps set at three stations around MNPS; additionally, since 1984, studies have been conducted during the hatching season to estimate the number of lobster larvae entrained through cooling water systems. The objective of the lobster monitoring program is to determine if operation of MNPS has caused changes in local lobster abundance beyond those expected from natural variability and high fishing mortality rates.

The total number of lobsters caught and total catch per unit effort (CPUE) in our study area reached record levels in 1992 and remained high during 1993 and 1994. However, legal lobster catches (those individuals ≥ 82.6 mm carapace length) have significantly declined since the NUSCO study began in 1978. Legal catches were expected to improve in 1993 and 1994 after large numbers of lobsters, observed to be just below legal size in the 1992 and 1993 catches, molted to legal size. Instead, legal CPUE has continued to decline the past 2 years, in contrast with recruitment patterns observed in previous studies when strong recruit classes were followed by an increase in legal catches one to two years later. The fact that CPUE of legal-sized lobsters continued to decline despite the strong recruit classes observed since 1992 may be further evidence that the local lobster resource is currently overfished.

During 1994, the peak in number of molting lobsters was later, and growth per molt was lower than previously observed in our studies; these conditions were attributed to below normal water temperatures during spring 1994. Changes observed in the size structure, sex ratio, and proportion of berried females of local lobsters may be primarily due to increased fishing rates and to implementation of new fishery regulations. During three-unit studies, the incidence of claw loss was lower, and recapture rates and size structure of tagged lobsters caught by NUSCO and commercial lobstermen were inconsistent with results observed during two-unit studies. These changes were attributed to the implementation of the escape vent regulation in 1984. The percentage of berried females collected nearly doubled during three-unit studies and was probably related to regulations increasing the minimum legal size. The new regulations implemented to sustain the lobster resource by improving lobster survival appeared to be effective.

However, fishing effort continues to escalate and fishery managers question the long-term stability of the resource under that kind of fishing pressure.

The density of lobster larvae collected in samples of the MNPS cooling water was higher during three-unit studies due to the higher percentage of berried females. Estimates of the total number of larvae entrained at MNPS were also higher during three-unit studies due to a combination of higher larval densities and higher cooling water demand of Unit 3. The potential impact of higher larval entrainment on subsequent legal lobster abundance is difficult to assess because of the uncertainty that exists concerning larval origin, larval survival, and recruitment rates to legal size. However, the high total CPUE for lobsters in our area over the past 3 years suggests that entrainment during the initial years (1986-88) of three-unit operation has not adversely affected the local adult population.

Marine Woodborer Study

The Marine Woodborer Study report describes the local distribution of *Teredo bartschi*, a semitropical shipworm common from Texas to South Carolina, but not found farther north except in close proximity to nuclear power plants in New Jersey and Connecticut. Consistent recruitment and survival of *Teredo bartschi* has been observed in MNPS discharge waters from 1975 to 1994. Within the effluent quarry, high recruitment of *T. bartschi* was again observed in 1994, primarily in panels on the western side. This finding was consistent with the previous Exposure Panel Studies and Distribution Studies conducted since 1975. Outside the effluent quarry, *T. bartschi* remains closely associated with discharge waters, with highest recruitment at the quarry cuts (undiluted effluent). Recruitment decreased substantially with distance from the outfall; in 1994, relatively few individuals were found in panels 100 m from the quarry cuts and none were found at sites 500 m away. Sampling sites 500 m from the quarry cuts appear to be near the limit of suitable thermal conditions for *T. bartschi* recruitment; several individuals recruited onto panels there in 1993, although none were collected in 1994. *T. bartschi* has not expanded its occurrence to other areas in Long Island Sound unaffected by MNPS discharge. Although present in MNPS discharge waters for over 20 years, this species has not adapted to near-ambient or ambient temperature conditions outside the effluent quarry. Furthermore, discontinued use of woodchips after Unit 1 retubing in 1993 may result in future declines of the *T. bartschi* population within the effluent quarry.

Eelgrass

Eelgrass studies during 1994 monitored three populations in the Millstone area: Jordan Cove (JC), White Point (WP) and the Niantic River (NR). Results from 1994 indicated generally poor regional conditions for growth and reproduction of these local populations. Population parameters at WP measured in 1994 were relatively low, but still within ranges established over the previous 9 study years. The deeper water WP study site provides more protection against elevated temperatures from insolation in the summer, sediment freezing during low tides in winter, and swan grazing. Eelgrass populations at shallow water sites are more susceptible to these stress mechanisms. Overall environmental stress was apparently greatest at NR in 1994, and was related to regional and site-specific (depth-related) factors, and possibly water quality conditions in the Niantic River. *Zostera* populations were elimination at NR by September 1994. Similar localized elimination events have been observed at other sites in the Niantic River in 1986 and 1993, indicating an overall long-term population decline in the Niantic River during the study period. This population is at its lowest level since studies began in 1985, in terms of abundance and distribution, and now consists of only small isolated patches.

Shallow-water stress, as temperature extremes, were measured directly at JC, and may have contributed to population declines observed at that site. Sediment freezing likely occurred during two extreme low tide events in the winter of 1993-94. Although hydrothermal modeling studies indicate some thermal plume incursion to JC, elevated temperatures measured at JC during the summer appeared more related to insolation of the shallow sand flats in Jordan Cove than to thermal plume incursion. Large-scale decline/recovery cycles in *Zostera* populations throughout North America and Europe have been observed since the 1930s and attributed to disease and eutrophication. In particular, populations on the east coast North America, including Long Island Sound, have been declining steadily since the 1960s. It is likely that population declines at JC and other sites in the Millstone area are part of this regional trend, and not the result of MNPS operation.

Rocky Intertidal Studies

Rocky intertidal studies in 1994 characterized nearby shore communities impacted by the MNPS thermal plume, as well as those farther away and unimpacted. Conditions resulting in much of the variability among communities at sampling sites outside the influence of MNPS were related to natural factors including site orientation to prevailing wind-generated waves, the ability of available substratum (slope) to dissipate the horizontal force of those waves, and the character of that substratum (e.g., boulders, bedrock ledge, etc.). Community differences beyond those attributed to natural factors occurred within the thermal plume area at sites located on Fox Island (FE and FN), and were directly attributed to MNPS operation. Various aspects of the impact-related community changes at Fox Island were identified through separate studies which included qualitative algal sampling, estimations of intertidal organism abundance, and studies of local *Ascophyllum nodosum* populations.

Elevated temperature conditions caused by the three-unit thermal plume allowed development of a unique flora at FE. The most notable shifts in species occurrence, revealed by qualitative algal sampling, were the presence of warm water-tolerant species not typical of other sites (*Agardhiella subulata*, *Gracilaria tikvahiae*, and *Sargassum filipendula*), absence of common cold water species (*Mastocarpus stellatus*, *Dumontia contorta*, and *Polysiphonia lanosa*) and extended or reduced periods of occurrence of seasonal species with warm water or cold water affinities, respectively.

During 1994, power plant impacts on dominant species abundance patterns, caused by two-cut water circulation patterns and by three-unit operations, were observed only at FE, and were most pronounced in the low intertidal, where temperature conditions were most severe. The low intertidal community at FE, which prior to 1983 had been unimpacted and characterized by perennial populations of *Fucus*, *Chondrus*, and *Ascophyllum* and predictable seasonal peaks in barnacle and *Monostroma* abundance, has been replaced by a persistent community dominated by *Codium*, *Ulva*, *Enteromorpha*, and *Polysiphonia*. Also, populations of species observed in undisturbed transects only at FE (*Sargassum*, *Gracilaria*) continued to persist and expand during 1994.

Ascophyllum populations at three stations in the vicinity of MNPS continued to be monitored in 1993. Elevated temperatures (2-3°C above ambient) at our station nearest the discharge (FN) caused *Ascophyllum* to grow longer and more rapidly at this site, relative to stations farther away. A moderate level of growth enhancement was observed at FN during 1993-94, when compared to previous years, attributed to lessened thermal plume incursion resulting from an extended outage of Unit 3 for much of the peak growing season. As in previous years, *Ascophyllum* mortality, or loss of tagged plants and tips, at our present sampling sites was not related to proximity to the power plant but rather to degree of exposure to storm forces.

Benthic Infauna

Benthic infaunal studies during 1994 continued to monitor subtidal soft-bottom habitats in the vicinity of MNPS for power plant-related changes in sedimentary characteristics and infaunal community structure (total abundance, species number, and species composition). The top four most dominant taxa at each station in 1994 accounted for more than 60% of all individuals collected. These were: Oligochaeta, *Protodorvillea gaspeensis*, *Tellina agilis*, and Archiannelida at EF; Oligochaeta, *Tharyx* spp., *Mediomastus ambiseta*, and *Protodorvillea gaspeensis* at GN; *Aricidea catherinae*, Oligochaeta, *Mediomastus ambiseta*, and *Tellina agilis* at IN; and *Mediomastus ambiseta*, *Aricidea catherinae*, Oligochaeta, and *Tharyx* spp. at JC. In most cases, these organisms were dominant taxa in both two-unit and three-unit operational periods. Most stations were characterized by one or more clearly dominant taxa (oligochaetes at EF, GN, and JC, *Aricidea catherinae* at GN and JC, and *Tharyx* spp. at GN) during both operational periods. There has been no single dominant taxon at IN during either operational period, where mean relative abundance of any single taxon rarely exceeded 10% The control site at GN continues to exhibit stability in terms of the sedimentary environment and infaunal community structure. Some stability in these parameters has also been noted in recent years, including 1994, at sites impacted by short-term episodic disturbance events (e.g., dredging and construction activities at IN, and siltation at JC). Community recovery is ongoing, but is not complete at either site. Effluent scour at EF remains a dominant structuring factor on both the sedimentary environment and infaunal community, and is expected to continue throughout MNPS three-unit operation.

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Introduction

Reporting Requirements

This report summarizes results of ongoing environmental monitoring programs conducted by Northeast Utilities Service Company (NUSCO) in relation to the operation of the three-unit Millstone Nuclear Power Station (MNPS). MNPS can affect local marine biota in several ways: large organisms may be impinged on the traveling screens that protect the condenser cooling and service water pumps; smaller ones may be entrained through the condenser cooling-water system, which subjects them to various mechanical, thermal, and chemical effects; and marine communities in the discharge area may be subjected to thermal, chemical, and mechanical effects resulting from the outflow of the cooling water. In addition, occasional maintenance dredging is done in the vicinity of the intake structures. The basis for the studies is the National Pollutant Discharge Elimination System (NPDES) permit (CT0003263) issued by the Connecticut Department of Environmental Protection on December 14, 1992 to Northeast Nuclear Energy Company (NNECO), on whose behalf NUSCO has undertaken this work. The regulations in the permit allow the MNPS cooling water to be discharged into Long Island Sound (LIS) in accordance with Section 22a-430 of Chapter 446k of the Connecticut General Statutes and Section 301 of the Federal Clean Water Act, as amended. Paragraph 5 of the MNPS NPDES permit states that:

> *The permittee shall conduct or continue to conduct biological studies of the supplying and receiving waters, entrainment studies, and intake impingement monitoring. The studies shall include studies of intertidal and subtidal benthic communities, finfish communities and entrained plankton and shall include detailed studies of lobster populations and winter flounder populations.*

In addition, paragraph 7 of the permit requires that:

> *On or before April 30, 1993 and annually thereafter, submit for review and approval of the Commissioner a detailed report of the ongoing biological studies required by paragraph 5 and as approved under paragraph 6.*

Furthermore, a decision and order of the Connecticut Siting Council (CSC) requires that NNECO inform the Council of results of MNPS environmental impact monitoring studies and any modifications made to these studies (paragraph 6 of the proceeding entitled "Docket No. 4, Certificate of Environmental Compatibility and Public Need for an Electric Generating Facility Identified as 'Millstone Nuclear Power Station, Unit 3,' located in the Town of Waterford, Connecticut" and dated March 22, 1976). This report satisfies the requirements of the NPDES permit and of the CSC by updating and summarizing various studies conducted at MNPS that were presented most recently in NUSCO (1994).

Study Area

MNPS is situated on Millstone Point, about 8 km west-southwest of New London on the Connecticut shore of LIS (Fig. 1). The property, covering an area of about 200 ha, is bounded to the west by Niantic Bay, to the east by Jordan Cove, and to the south by Twotree Island Channel. The MNPS monitoring programs sample a study area of approximately 50 km^2 that extends from the northern portions of the Niantic River and Jordan Cove to Giants Neck, 2 km south of Twotree Island, and 2 km east of White Point. Work takes place from the shoreline into areas as deep as 20 m southwest of Twotree Island.

Strong tidal currents predominate in the vicinity of Millstone Point and influence the physical characteristics of the area. Average tidal flow through Twotree Island Channel is approximately 3,400 $m^3 \cdot sec^{-1}$ and at maximum is about 8,500 $m^3 \cdot sec^{-1}$ (NUSCO 1983). Current velocities are about 1 to 1.8 knots in the channel, slightly less (1 to 1.5 knots) near the plant and in Niantic Bay, and relatively weak in Jordan Cove and in the upper Niantic River. The currents are driven by semi-diurnal tides that have a mean and maximum range of 0.8 and 1.0 m, respectively. Thermal and salinity induced stratification may occur in regions unaffected by strong tidal currents. The greatest temperature variation has been observed in nearshore areas where water temperature can vary from -3 to 25°C; salinity varies much less and ranges from 26 to 30‰. The bottom is generally composed of fine to medium sand throughout the area, but also includes some rock outcrops and muddy sand, especially near shore. Strong winds, particularly from the southwest, can at times result in locally heavy seas (up to 1.5 m or greater) near Millstone Point. Additional information on local hydrography and meteorology can be found in NUSCO (1983).

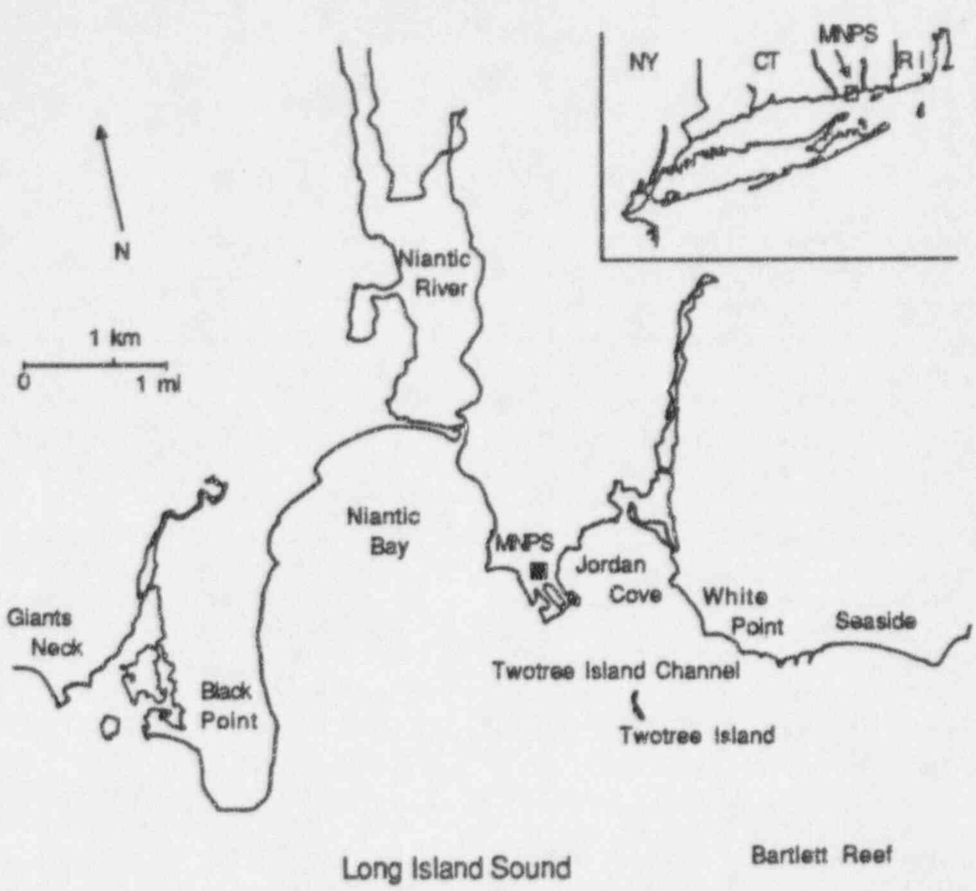

Fig. 1. The area in which biological monitoring studies are being conducted to assess the effects of the operation of MNPS.

Millstone Nuclear Power Station

The MNPS complex consists of three operating nuclear power units; a detailed description of the station was given in NUSCO (1983). Unit 1, a 660-MWe boiling water reactor, began commercial operation on November 29, 1970; Unit 2 is an 870-MWe pressurized water reactor that began commercial operation in December 1975; and Unit 3 (1150-MWe pressurized water reactor) commenced commercial operation on April 23, 1986. All three units use once-through condenser cooling water systems with rated circulating water flows of 26.5, 34.6, and 56.6 $m^3 \cdot sec^{-1}$ for Units 1 through 3, respectively. Cooling water is drawn from depths of about 1 m below mean sea level by separate shoreline intakes located on Niantic Bay (Fig. 2). The intake structures, typical of many coastal power plants, have coarse bar racks (6.4 cm on center, 5.1-cm gap) preceding vertical traveling screens to protect the plants from debris. Units 1 and 2 have always had 9.5-mm mesh screens. Unit 3 originally had 4.8-mm mesh screens, a combination of 9.5- and 4.8-mm mesh screens from early 1990 through summer 1992, and only 9.5-mm mesh screens as of August 15, 1992. Fish return systems (sluiceways) were installed at Unit 1 in December 1983 and at Unit 3 during its construction to return aquatic organisms washed off the traveling screens back to LIS. The installation and operation of sluiceways have minimized the impact of impingement at MNPS (NUSCO 1986, 1988a). A chronology of significant events associated with MNPS construction and operation, including installation of devices designed to mitigate environmental effects and unit operational shutdowns exceeding 2 weeks, are found in Table 1. Capacity factors (the electricity produced as a percentage of maximum possible production) during 1994 were 58.3% for Unit 1, 48.2% for Unit 2, and 94.0% for Unit 3.

MNPS cooling water is nominally heated in Units 1, 2, and 3 from ambient temperature to a maximum of 13.9, 12.7, and 9.5°C, respectively. Each unit has separate discharge structures that release the effluent into an abandoned granite quarry (ca. 3.5 ha surface

TABLE 1. Chronology of major construction and operation events at MNPS through 1994.

Date	Activity	Reference[a]
December 1965	Construction initiated for Unit 1	NUSCO (1973)
November 1969	Construction initiated for Unit 2 began	NUSCO (1973)
October 26, 1970	Unit 1 initial criticality; produced first thermal effluent	DNGL
November 29, 1970	Unit 1 initial phase to grid	DNGL
December 28, 1970	Unit 1 began commercial operation	DNGL
January 15, 1971 to February 22, 1971	Unit 1 shutdown	DNGL
August-December 1972	Surface boom at Unit 1	NUSCO (1978)
November 1972	Fish barrier installed at quarry cut	NUEL
September 3, 1972 to March 20, 1973	Unit 1 shutdown	DNGL
November 1972	Unit 2 coffer dam removed	NUSCO (1973)
April 18 to July 28, 1973	Unit 1 shutdown	DNGL
August-December 1973	Surface boom at Unit 1	NUSCO (1978)
July-December 1974	Surface boom at Unit 1	NUSCO (1978)
September 1 to November 5, 1974	Unit 1 shutdown	DNGL
July-October 1975	Surface boom at Unit 1	NUSCO (1978)
July 1975	Bottom boom installed at Unit 1	NUSCO (1978)
August 5, 1975	Unit 3 coffer dam construction began	NUEL
September 10 to October 20, 1975	Unit 1 shutdown	DNGL
October 7, 1975	Unit 2 produced first effluent	EDAN
November 7, 1975	Unit 2 initial criticality; produced first thermal effluent	EDAN
November 13, 1975	Unit 2 initial phase to grid	DNGL
December 1975	Unit 2 began commercial operation	NUEL
March 19, 1976	Unit 3 coffer dam construction finished	NUEL
June-October 1976	Surface boom at Unit 2	NUSCO (1978)
October 1 to December 2, 1976	Unit 1 shutdown	DNGL
December 20, 1976 to January 20, 1977	Unit 2 shutdown	DNGL
May 6 to June 25, 1977	Unit 2 shutdown	DNGL
June-October 1977	Surface boom at Unit 2	NUSCO (1978)
November 20, 1977 to May 1, 1978	Unit 2 shutdown	DNGL
March 10 to April 15, 1978	Unit 1 shutdown	DNGL
March 10 to May 21, 1979	Unit 2 shutdown	DNGL
April 28 to June 27, 1979	Unit 1 shutdown	DNGL
August 10 to 25, 1979	Unit 2 shutdown	DNGL
November 1 to December 5, 1979	Unit 2 shutdown	DNGL
May 7 to June 19, 1980	Unit 2 shutdown	DNGL
June 1 to June 18, 1980	Unit 1 shutdown	DNGL
August 15 to October 19, 1980	Unit 2 shutdown	DNGL
October 3, 1980 to June 16, 1981	Unit 1 shutdown	DNGL
January 2 to 19, 1981	Unit 2 shutdown	DNGL
December 5, 1981 to March 15, 1982	Unit 2 shutdown	DNGL
March 1981	Bottom boom removed at Unit 1	NUEL
September 10 to November 18, 1982	Unit 1 shutdown	DNGL
March 2 to 18, 1983	Unit 2 shutdown	DNGL
April-September 1983	Unit 3 coffer dam removed, intake maintenance dredging	NUEL
May 28, 1983 to January 12, 1984	Unit 2 shutdown	DNGL
December 1983	Fish return system installed at the Unit 1 intake	NUEL
August 1983	Second quarry cut opened	NUEL
April 13 to June 29, 1984	Unit 1 shutdown	DNGL
February 15 to July 4, 1985	Unit 2 shutdown	DNGL
June 1985	Intake maintenance dredging	NUEL
September 28 to November 7, 1985	Unit 2 shutdown	DNGL
October 25 to December 22, 1985	Unit 1 shutdown	DNGL
November 1985	Unit 3 produced first effluent	EDAN
February 12, 1986	Unit 3 produced first thermal effluent	EDAN
April 23, 1986	Unit 3 began commercial operation	DNGL

TABLE 1. (cont.).

July 25 to August 17, 1986	Unit 3 shutdown	DNGL
September 20 to December 18, 1986	Unit 2 shutdown	DNGL
December 1 to 15, 1986	Unit 1 shutdown	DNGL
January 30 to February 16, 1987	Unit 2 shutdown	DNGL
March 14 to April 10, 1987	Unit 3 shutdown	DNGL
June 5 to August 17, 1987	Unit 1 shutdown	DNGL
November 1, 1987 to February 17, 1988	Unit 3 shutdown	DNGL
December 31, 1987 to February 20, 1988	Unit 2 shutdown	DNGL
April 14 to May 1, 1988	Unit 3 shutdown	DNGL
May 7-22, 1988	Unit 2 shutdown	DNGL
October 23 to November 8, 1988	Unit 3 shutdown	DNGL
February 4 to April 29, 1989	Unit 2 shutdown	DNGL
April 8 to June 4, 1989	Unit 1 shutdown	DNGL
May 12 to June 12, 1989	Unit 3 shutdown	DNGL
October 21 to November 24, 1989	Unit 2 shutdown	DNGL
March 30 to April 20, 1990	Unit 3 shutdown; installation of some 9.5-mm intake screen panels	DNGL
May 8 to June 15, 1990	Unit 2 shutdown	DNGL
September 14 to November 9, 1990	Unit 2 shutdown	DNGL
February 2 to April 17, 1991	Unit 3 shutdown; installation of new fish buckets and sprayers	DNGL
April 7 to September 2, 1991	Unit 1 shutdown	DNGL
April 23 to May 11, 1991	Unit 2 shutdown	DNGL
May 26 to July 7, 1991	Unit 2 shutdown	DNGL
July 25, 1991 to February 6, 1992	Unit 3 shutdown; installation of new fish buckets and sprayers	DNGL
August 7 to September 11, 1991	Unit 2 shutdown	DNGL
October 1, 1991 to March 3, 1992	Unit 1 shutdown	MOSR
November 6 to December 27, 1991	Unit 2 shutdown	MOSR
January 28 to February 14, 1992	Unit 2 shutdown	MOSR
March 22 to April 6, 1992	Unit 1 shutdown	MOSR
May 16 to June 4, 1992	Unit 3 shutdown; installation of new fish buckets and sprayers	MOSR
May 29, 1992 to January 13, 1993	Unit 2 shutdown	MOSR
July 4 to August 15, 1992	Unit 1 shutdown	MOSR
August 15, 1992	Completed installation of new fish buckets and sprayers at Unit 3	NUEL
September 30 to November 4, 1992	Unit 3 shutdown	MOSR
July 31 to November 10, 1993	Unit 3 shutdown	MOSR
September 15 to October 10, 1993	Unit 2 shutdown	MOSR
January 17 to May 1, 1994	Unit 1 shutdown	MOSR
April 22 to June 18, 1994	Unit 2 shutdown	MOSR
July 27 to September 3, 1994	Unit 2 shutdown	MOSR
September 8-22, 1994	Unit 3 shutdown	MOSR
October 1, 1994 to spring 1995	Unit 2 shutdown	MOSR

[a] DNGL refers to the daily net generation log, NUEL to NUSCO Environmental Laboratory records, EDAN to the environmental data acquisition network, and MOSR to the monthly nuclear plant operating status report.

area, maximum depth of approximately 30 m). The thermal discharge (about 11°C warmer than ambient under typical three-unit operation) exits the quarry through two channels (cuts), whereupon the thermal effluent mixes with LIS water (Fig. 2). The cuts are equipped with fish barriers consisting of 19-mm metal grates, which serve to keep larger fish out of the quarry. The thermal plume is warmest in the immediate vicinity of the cuts and within about 1,100 m of the quarry the surface-oriented plume cools to within 2.2°C above ambient. Beyond this distance the plume is highly dynamic and varies with tidal currents (Fig. 3). Hydrothermal surveys conducted at MNPS were described in NUSCO (1988b).

Monitoring Programs

This report contains a separate section for each major monitoring program, some of which have been ongoing since 1968. These long-term studies have provided the representative data and scientific bases necessary to assess potential biological impacts as a result of MNPS construction and operation. The significance of changes found for various communities

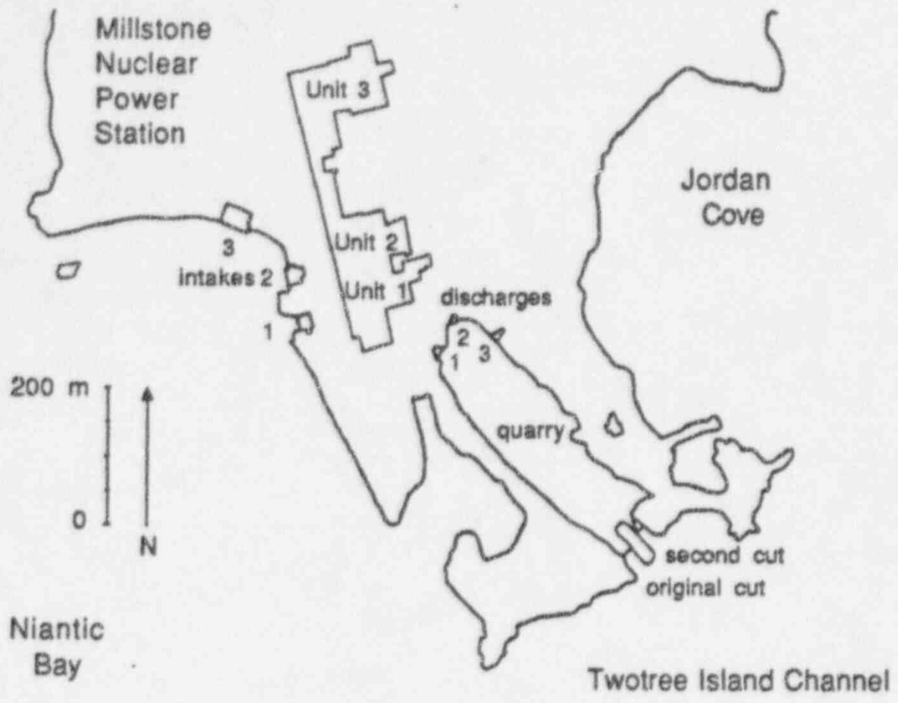

Fig. 2 The MNPS site, showing the intake and discharge of each unit, the quarry, and the two quarry discharge cuts.

and populations beyond those that were expected to occur naturally were evaluated using best available methodologies. Programs discussed below include Winter Flounder Studies, Fish Ecology Studies, Lobster Studies, Marine Woodborer Study, Eelgrass, Rocky Intertidal Studies, and Benthic Infauna. Reporting periods for each section vary and were predicated on biological considerations and processing time necessary for samples, as well as on regulatory requirements. In cases where the seasonal abundance of organisms differed from arbitrary annual reporting periods, the periods chosen were adjusted to best define the season of interest for a particular species or community.

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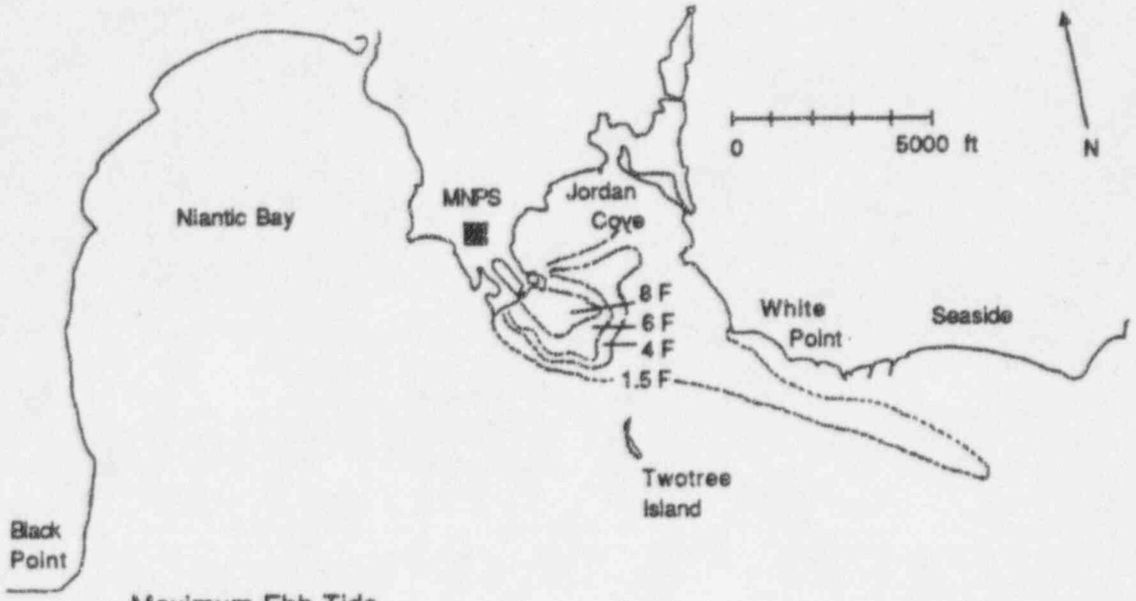
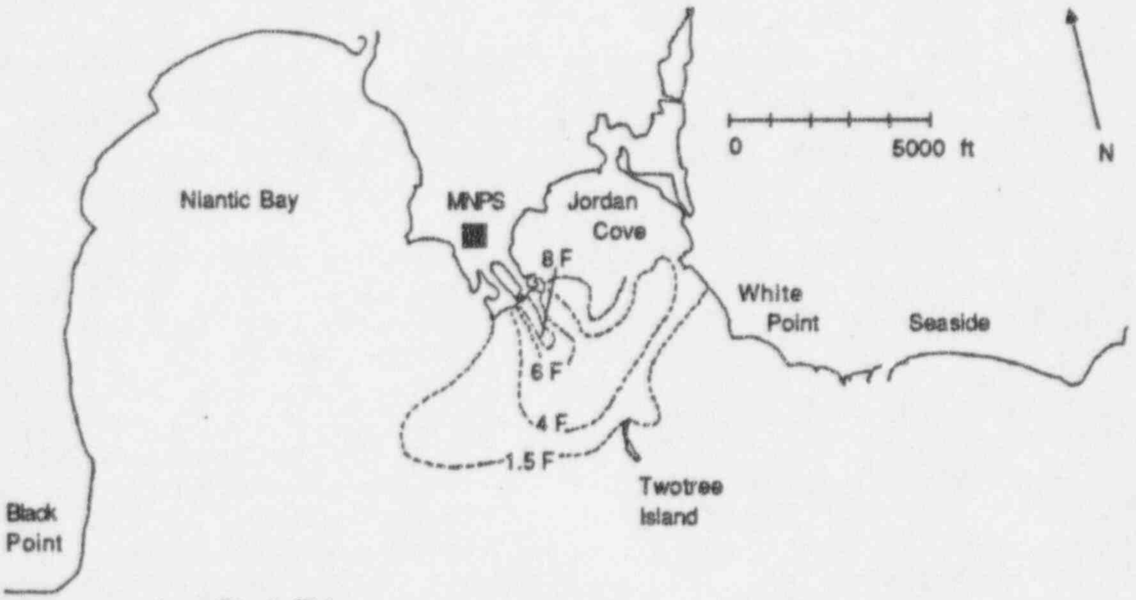
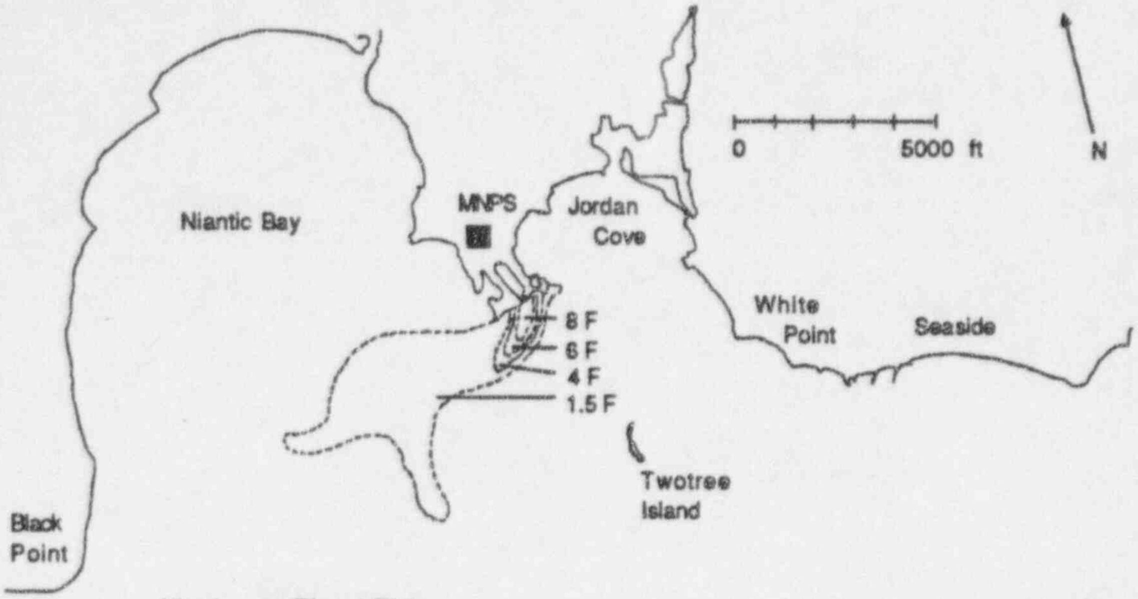


Maximum Ebb Tide

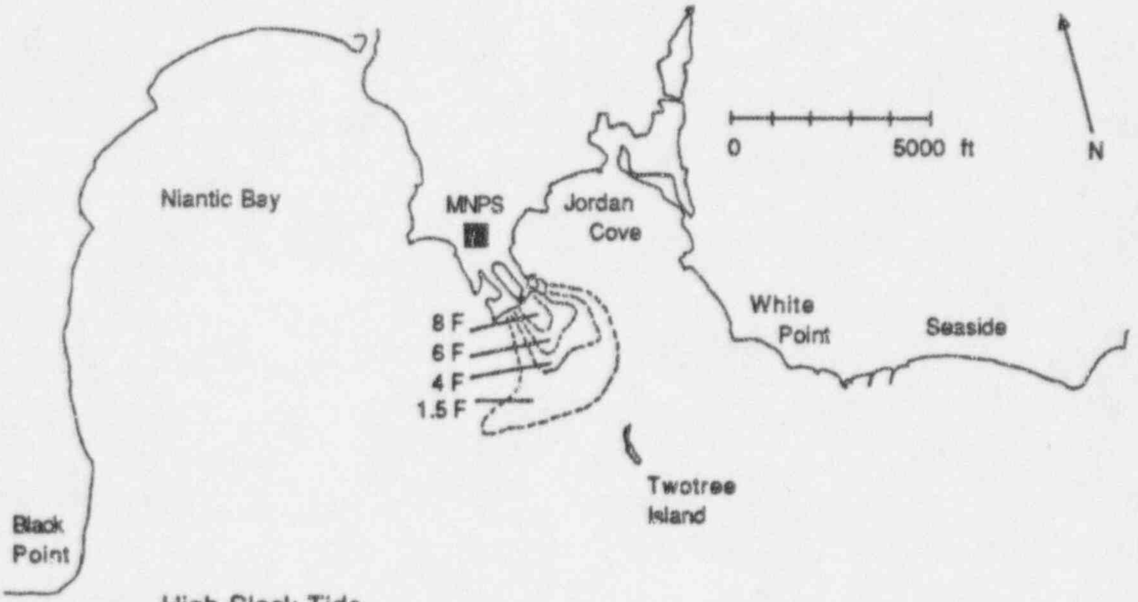


Low Slack Tide

Fig. 5. Locations of selected three-unit thermal plume isotherms (1.5°F, 4°F, 6°F, and 8°F) under various tidal conditions.



Maximum Flood Tide



High Slack Tide

Fig. 3. (continued).

Winter Flounder Studies

Winter Flounder Studies

Introduction

The winter flounder (*Pleuronectes americanus*) has been a focus of environmental impact studies by Northeast Utilities Service Company (NUSCO) at the Millstone Nuclear Power Station (MNPS) since 1973. It is an important sport and commercial fish in Connecticut (Smith et al. 1989) and an abundant member of the local demersal fish community. The winter flounder has been reported from Labrador to Georgia, but is most numerous in the central part of its range (Scott and Scott 1988), which includes Long Island Sound (LIS). Movement patterns and reproductive activity are seasonally specific and well-documented (e.g., Klein-MacPhee 1978). Most adult fish enter estuaries in late fall and early winter and spawn in upper portions of estuaries during late winter and early spring at temperatures between 1 and 10°C (peaking at 2 -5°C) and salinities of 10 to 35 ‰ (Bigelow and Schroeder 1953; Pearcy 1962; Scarlett and Allen 1992). Three years are required for oocyte maturation (Dunn and Tyler 1969; Dunn 1970; Burton and Idler 1984). In eastern LIS, females begin to mature at age 3 and 4 and males at age 2 (NUSCO 1987). Average fecundity of Niantic River females is about 561,000 eggs per fish. Eggs are demersal and hatch in about 15 days, and larval development takes about 2 months; both processes are temperature-dependent. Small larvae are planktonic and although many remain near the estuarine spawning grounds, others are carried into coastal waters by tidal currents (Smith et al. 1975; NUSCO 1989; Crawford 1990). Some of the displaced larvae are returned to the estuary on subsequent incoming tides, but many of them are swept away from the area into coastal waters, where their survival may be diminished. Larger larvae maintain some control over their position by vertical movements and may spend considerable time on the bottom. Following metamorphosis, demersal young-of-the-year winter flounder predominantly settle or move into shallow inshore waters. Yearlings (age-1 fish) become photonegative and most of them are usually found in deeper waters (Pearcy 1962; McCracken 1963). Some adult fish remain in estuaries following spawning, while others disperse offshore. By summer, most adults leave warmer shallow waters as their preferred temperature range is 12-15°C (McCracken 1963), although some remain in estuaries. These fish apparently avoid temperatures above 22.5°C by burying themselves in cooler bottom sediments (Olla et al. 1969). Other aspects of winter flounder life history have been summarized by Klein-MacPhee (1978). Because the early life history of the congeneric European plaice (*Pleuronectes platessa*) has many similarities to that of the winter flounder, relevant literature was also reviewed for this report to gain further insights into winter flounder population dynamics.

MNPS operation results in the impingement of juvenile and adult winter flounder on the traveling screens of the cooling-water intakes and the entrainment of larvae through the condenser cooling-water system. The impact of fish impingement at MNPS has been largely mitigated by the installation and operation of fish return sluiceways at MNPS Units 1 and 3 (NUSCO 1986b, 1994c). The mortality of entrained larvae potentially has greater significance as the winter flounder, unlike many marine fishes, is a product of local spawning with geographically isolated stocks associated with individual estuaries or specific coastal areas (Lobell 1939; Perlmutter 1947; Saila 1961). In particular, the population of winter flounder spawning in the nearby Niantic River has been studied in detail to assess the long-term effect of larval entrainment through the MNPS cooling-water system. The 1994 spawning season was the ninth year in which winter flounder have experienced impact from the operation of all three MNPS units. Although knowledge of annual variability is important to assess short-term impacts, most significant changes in populations affected by fisheries tend to occur on longer time scales (Cushing 1977; Steele et al. 1980). Therefore, development of a long-term assessment capability was the ultimate goal of NUSCO winter flounder studies. Presently, a combination of various sampling programs and analytical methods are used to examine current abundance of the Niantic River population for annual estimates of the spawning stock; this report section summarizes data collected during 1994 and updates results reported previously in NUSCO (1994a). A computer population simulation model, the NUSCO winter flounder stochastic population dynamics model (SPDM), is used for assessing long-term effects of MNPS operation. The SPDM can simulate the long-term effects of historical and projected rates of fishing mortality and simultaneous plant operation, resulting in annual mortalities from impingement of juveniles and adults and the entrainment of larvae through the MNPS cooling-water system. Results of SPDM simulations and a probabilistic risk assessment provide a measure of the Niantic River winter flounder population through the year 2060, well after the scheduled shutdown of MNPS Unit 3 in 2026.

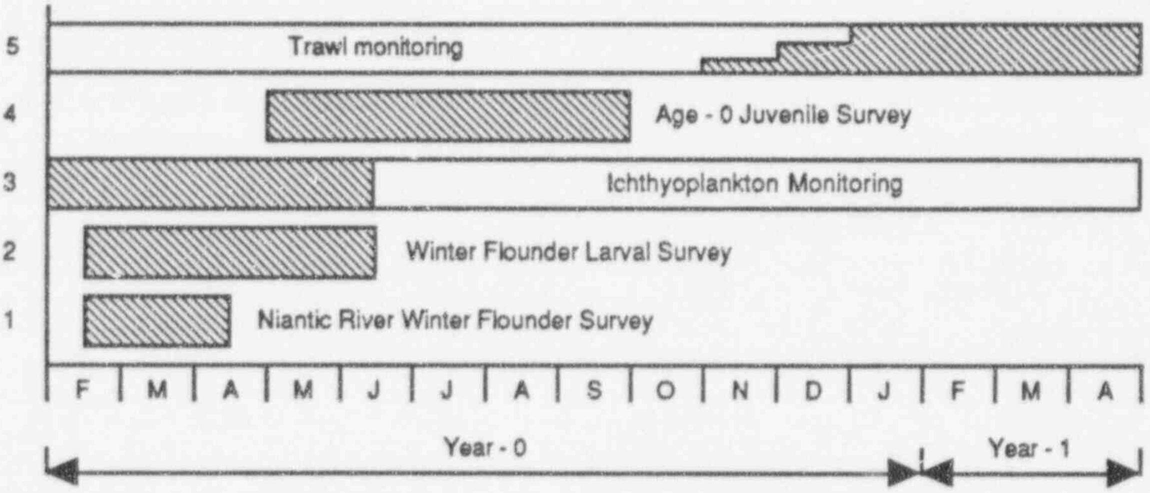
Materials and Methods

Sampling programs

Data needed to assess MNPS impact on the winter flounder come from several biological sampling programs. Some programs (e.g., Niantic River adult and larval surveys, age-0 survey) were designed to investigate specific life history stages of winter flounder. Other information comes from year-round sampling of the entire local fish community, such as the trawl monitoring program (TMP) and the entrainment ichthyoplankton monitoring program at MNPS. Additional information used in various assessments was presented in NUSCO (1987), which summarized various life history studies of the winter flounder prior to the operation of Unit 3. Ongoing sampling programs that contributed data to the Niantic River winter flounder studies are shown in Figure 1, which includes the seasonal duration of sampling and timing relative to the annual life-cycle of Niantic River winter flounder. Brief descriptions of field methodologies used in these programs are given below. Information on water temperature was obtained from continuous temperature recorders at the intakes of MNPS Units 1 and 2; daily mean temperatures were determined from available records of 15-min average temperatures. Monthly, seasonal, or annual means were calculated using daily means.

Adult winter flounder sampling

Basic sampling methodology for the adult winter flounder spawning surveys in the Niantic River has remained unchanged since 1982 (NUSCO 1987). Surveys started during mid-February through mid-March, after most ice cover disappeared from the river,



1. February-April sampling (spawning season) for adults and juveniles throughout the Niantic River.
2. February-June larval sampling at three stations in the Niantic River and one in Niantic Bay.
3. Year-round monitoring of all ichthyoplankton at the MNPS discharges.
4. May-September sampling of age-0 juveniles at two stations in the Niantic River.
5. Year-round monitoring of all benthic fishes at six stations near MNPS (juvenile data come from two stations in November, four in December, and six in January-April).

Fig. 1. Current sampling programs contributing data for computation of winter flounder abundance indices (hatched area show months from which data were used in this report).

and continued into April. Sampling usually ceased when the proportion of reproductively active females decreased to less than 10% of all females examined for 2 consecutive weeks, an indication of completion of most spawning. In each survey, the Niantic River was divided into a number of sampling areas, referred to as stations (Fig. 2). Since 1979 no samples have been taken outside of the navigational channel in the lower portion of the river because of an agreement made with the East Lyme-Waterford Shellfish Commission to protect habitat of the bay scallop (*Argopecten irradians*). Winter flounder were collected on at least 2 days of each survey week using a 9.1-m otter trawl with a 6.4-mm bar mesh codend liner. Fish caught in each tow were held in water-filled containers aboard the survey vessel before processing. Since 1983, each fish larger than 20 cm was measured to the nearest mm in total length and its gender ascertained. Before 1983, at least 200 randomly selected winter flounder were measured during each week of sampling. Those fish not measured were classified into various length and gender groupings; at minimum, all winter flounder examined were classified as smaller or larger than 15 cm. The gender and reproductive condition of larger winter flounder was determined by either observing eggs or milt, or as Smigielski (1975) suggested, by noting the presence (males) or absence (females) of ctenii on left-side caudal peduncle scales. Before release, healthy fish larger than 15 cm (1977-82) or 20 cm (1983 and after) were marked in a specific location with a number or letter made by a brass brand cooled in liquid nitrogen. Marks and brand location were varied in a manner such that the year of marking would be apparent in future collections.

Larval winter flounder sampling

Winter flounder larvae entrained through the MNPS cooling-water system were sampled at the MNPS discharges (station EN, Fig. 3) since 1976. Collections usually alternated between the discharges of Units 1 and 2, depending upon plant operation and water flow. Larvae were collected in a 1.0 x 3.6-m plankton net of 333-µm mesh deployed from a gantry system. Four General Oceanic (GO) Model 2030 flowmeters were positioned in the net mouth to account for horizontal and vertical flow variation; sample volume was determined by averaging the four volume estimates from the flowmeters. Starting in 1993, the net was deployed for 3 to 4 minutes (filtering about 200 m^3), but this varied depending upon the number of circulating pumps in operation and tidal stage. In previous years, sampling time was longer and filtered about 400 m^3. Frequency of the

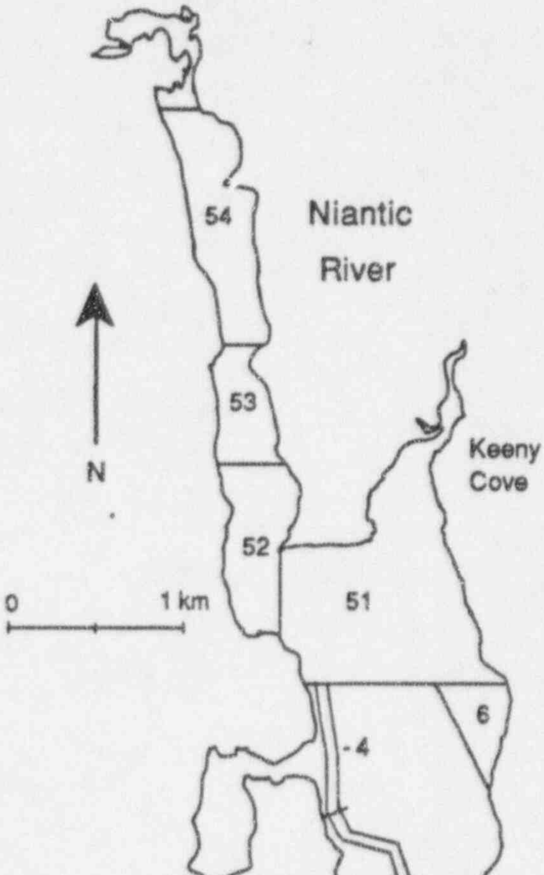


Fig. 2. Location of stations sampled in the Niantic River during 1994 for adult winter flounder from March 22 through April 14 (numbers) and age-0 winter flounder from May 25 through September 27 (letters).

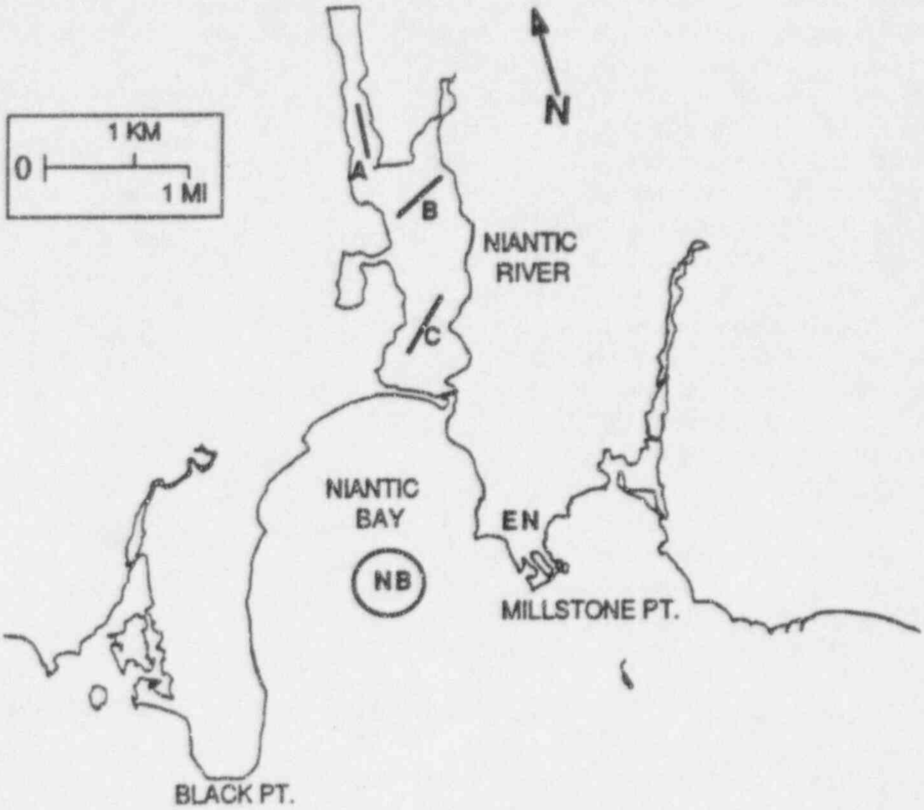


Fig. 3. Location of stations sampled for larval winter flounder during 1994.

sampling has varied since 1976 (NUSCO 1987). In 1994, samples were collected once per week during both day and night in February and June. During March through May, samples were taken on three days and nights per week. This was similar to 1993, but was one day and one night sample less per week than during March through May of 1983 through 1992. All ichthyoplankton samples, including those described below, were preserved with 10% formalin.

Winter flounder larvae have been collected in Niantic Bay at station NB since 1979 and in the Niantic River at stations A, B, and C since 1983 (Fig. 3). A 60-cm bongo plankton sampler was weighted with a 22.7-kg oceanographic depressor and fitted with 3.3-m long nets with mesh size of 202 µm during February and March and 333 µm during the remainder of the season. Volume of water filtered was determined using a single GO flowmeter mounted in the center of each bongo opening. The sampler was towed at approximately 2 knots using a stepwise oblique tow pattern, with equal sampling time at surface, mid-depth, and near bottom. The length of tow line necessary to sample the mid-water and bottom strata was determined by water depth and tow-line angle measured with an inclinometer. Nets were towed for 6 minutes (filtering about 120 m^3). One of the duplicate samples from the bongo sampler was retained for laboratory processing. The larval winter flounder sampling schedule for Niantic River and Bay was based on knowledge gained during previous years and was designed to increase data collection efficiency while minimizing sampling biases (NUSCO 1987). Larval sampling at the three Niantic River stations usually starts in early to mid-February, but due to ice conditions in 1994 sampling was not possible until March 2 and then only in the lower river at station C. Collections at stations A and B began on March 17. From then through the end of March, daytime tows were conducted within 1 hour of low slack tide. During the remainder of the season, until the disappearance of larvae at each station, tows were made at night during the second half of a flood tide. From 1983 through 1990, sampling was conducted 2 days a week. Starting in 1991, sampling was reduced to 1 day a week (NUSCO 1991a). Through 1992, station NB was sampled day and night every two weeks during February and at least once a week from March through the end of the larval winter flounder season. Beginning in 1993, station NB was sampled weekly only during the day from the start of the larval season though March and at night from April through the remainder of the larval season. Jellyfish medusae at the three river stations were removed (1-cm mesh sieve) from the samples and measured volumetrically to the nearest 100 mL.

Juvenile winter flounder sampling

Information on juvenile (age-0 and age-1) winter flounder was obtained from three sources (Fig. 1). A special sampling program specifically targeted post-larval young-of-the-year. A second source of data is the trawl monitoring program (TMP), and the third is the Niantic River adult spawning abundance surveys, during which winter flounder juveniles are collected incidentally. Data on juvenile fish abundance were available from about May of their birth year into April of the following year. Juvenile indices were referred to as age-0 when fish were collected as post-larval young in summer and during the subsequent fall and winter by the TMP. These fish became age-1 when taken during the February-April adult spawning surveys.

The abundance of post-larval age-0 winter flounder has been monitored at two stations (Fig. 2) in the Niantic River since 1983 (LR) or late 1984 (WA). Through 1992, collections were made weekly, but in 1993 and 1994 the sampling frequency was reduced to biweekly. Stations were sampled during daylight from about 2 hours before to 1 hour after high tide. Monitoring began in late May and continued through the end of September.

Young winter flounder were sampled using a 1-m beam trawl having two tickler chains and with interchangeable nets of 0.8-, 1.6-, 3.2-, and 6.4-mm mesh. In 1983, triplicate tows were made at LR using nets of increasing larger mesh as the season progressed. Beginning in 1984, two nets of successively larger mesh were used during each sampling trip; nets were deployed in a random order. A change to the next larger mesh in the four-net sequence was made when fish had grown enough to become retained by it, as use of larger meshes reduced the amount of detritus and algae collected. At each station, four replicate tows were made, two each with the two nets in use. Rarely, because of bad weather or damage to the net, only three tows were taken at a station. Tow distance was estimated by letting out a measured line attached to a lead weight as the net was hauled at approximately 25 $m \cdot min^{-1}$. The length of each tow was increased from 40 to 100 m in 20- or 40-m increments at a station as fish abundance decreased over time. However, in years when young were abundant, maximum tow length at a station was 60 or 80 m.

Catches from the TMP (see the Fish Ecology section of this report for methods) were used to follow the abundance of age-0 winter flounder during fall and winter. In addition to the TMP, juvenile winter flounder smaller than 15 cm in length (mostly age-1) were caught along with adults in the annual Niantic River spawning stock surveys. These fish were processed similarly as adults, although gender was usually not specified, and the fish were not branded. When small winter flounder were abundant, a subsample of at least 200 fish was measured each survey week; otherwise, all specimens were measured.

Indices of abundance

Data from the field sampling programs described above were used to calculate annual and seasonal indices of relative abundance. Indices, calculated using various sampling statistics, were computed for various life-stages of winter flounder, from newly hatched larvae to adult spawners and also included estimates of egg production. Specifics of each abundance index depended upon the particular stage of life, sampling effort, and suitability of the data; a detailed description of each follows. The indices enabled timely assessments to be made regarding the current status of the Niantic River winter flounder population and many of these data were used with the SPDM for long-term predictions of MNPS impact.

Relative annual abundance of adults

The relative annual abundance of winter flounder in the Niantic River during the late February-early April spawning season is described by trawl catch-per-unit-effort (CPUE). An annual CPUE was calculated by using the median catch following data standardization. Components of standardization included tow length, tow duration, weekly effort, and fish length and gender categories. Tow distance (with exceptions noted below) was fixed in 1983 because using the same tow length at all stations was expected to reduce variability in CPUE; previously, tows of variable length had been taken at all stations and catch was standardized by time of tow. A distance of 0.55 km was selected as the standard because it represented the maximum length of a tow that was formerly possible at station 1. Particularly during 1987 and 1989-91, tows one-half or two-thirds of this length were frequently taken in the upper river to avoid overloading the trawl with macroalgae and detritus. Because catch data from station 2 were used also in the TMP, tows there were made over 0.69 km, the standard for that particular sampling. Since 1990, tow distance at station 1 was reduced to 0.46 km because of the construction of a new bridge at the mouth of the river and the destruction of the old bridge.

Catches of winter flounder larger than 15 cm in tows made throughout the spawning surveys were standardized to either 15-min tows at stations 1 and 2 or 12-min tows at all other stations, as a standard tow distance was not set prior to 1983. Duration of tows varied and was usually greater in the lower river than in the upper river because of differences in tidal currents and amounts of extraneous material collected in the trawl, even though distance was similar. To lessen error in the calculation of CPUE, data from either exceptionally long or brief tows made prior to 1983 were excluded from the analyses. The minimum length of 15 cm used for CPUE calculation was smaller than the 20 cm used for mark and recapture estimates described below because of data limitations from the 1977-82 surveys. Effort was standardized within each year by replicating the median CPUE estimate in a given week as needed so that effort (number of tows) was the same for each week sampled. A 95% confidence interval (CI) was calculated for each annual median CPUE using a distribution-free method based on order statistics (Snedecor and Cochran 1967).

A second relative index of abundance used the gender and size distribution of the fish from adult spawning survey catches standardized by variable weekly and yearly effort. Adjustments to the catches were made using sampling effort to insure that each size and sex group of fish was given equal weight within each week of work, among weeks in each survey year, and to adjust for varying effort among years. Detailed methods of calculating these values were given in NUSCO (1989). To avoid confusion with the CPUE index, this measure is referred to as "annual standardized catch" throughout the remainder of this report. The annual standardized catch was the basis for the calculation of annual recruitment and egg production described below.

Absolute abundance estimates of adults

Absolute abundance estimates of winter flounder spawning in the Niantic River were obtained using mark-and-recapture methodology and the Jolly (1965) stochastic model. This model is considered among the most useful in providing abundance estimates for open populations as long as basic assumptions are approximately met (Cormack 1968; Southwood 1978; Begon 1979; Pollock et al. 1990). Annual absolute abundance estimates for Niantic River winter flounder larger than 20 cm were calculated by pooling together all fish marked and released during each annual survey and by observing the recaptures made in subsequent years. Absolute abundance estimates could not be generated for years prior to 1984 because of uncertainty in data records and ambiguity in brands used during early surveys. Estimates were made of annual population size (N) and other model parameters, including survival (ϕ), recruitment (B), and sampling intensity (p), using the computer program 'JOLLY' (Pollock et al. 1990).

Adult spawning stock size and egg production

The proportion of mature female winter flounder in each 0.5-cm length increment beginning at 20 cm was estimated from qualitative observations of reproductive condition (percent maturity by 0.5-mm size-classes) made from 1981 through the present. Pooled estimates were adjusted to give continuously increasing fractions of mature fish through 34 cm; all females this length or larger were considered to be mature. The fecundity (annual egg production per female) was estimated for each 0.5-cm size-class by using the following relationship determined for Niantic River winter flounder (NUSCO 1987):

$$\text{fecundity} = 0.0824 \cdot (\text{length in cm})^{4.506} \qquad (1)$$

This relationship was used with the annual standardized catch of mature females and their length composition to calculate egg production. Annual mean fecundity was determined by dividing the sum of all individual egg production estimates by the standardized catch of females spawning per year.

Absolute estimates of spawning females and corresponding annual egg production estimates for 1977 through 1994 were determined by assuming that the relative values represented 3.5% of the absolute values (see Absolute abundance estimates in Results and Discussion for how this fraction was determined). Annual estimates of the number of female spawners were also used in the derivation of a relationship between stock and recruitment for Niantic River winter flounder.

Development and growth, abundance, and mortality of larvae

Ichthyoplankton samples were split to at least one-half volume in the laboratory. Sample material was viewed through a dissecting microscope and winter flounder larvae were removed and counted. Up to 50 randomly selected larvae were measured to the nearest 0.1 mm in standard length (snout tip to notochord tip). The developmental stage of each measured larva was recorded using the following identification criteria:

Stage 1. The yolk-sac was present or the eyes were not pigmented (yolk-sac larvae);
Stage 2. The eyes were pigmented, no yolk-sac was present, no fin ray development, and no flexion of the notochord;
Stage 3. Fin rays were present and flexion of the notochord had started, but the left eye had not migrated to the midline;
Stage 4. The left eye had reached the midline, but juvenile characteristics were not present;
Stage 5. Transformation to the juvenile stage was complete and intense pigmentation was present near the base of the caudal fin.

Larval data analyses were based on standardized densities (number$\cdot$500m^{-3} of water sampled). A geometric mean of weekly densities was used in analyses because the data generally followed a lognormal distribution (McConnaughey and Conquest 1993) and weekly sampling frequencies varied among some stations. Because older larvae apparently remained near the bottom during the day and were not as susceptible to entrainment or the bongo sampler, data from daylight samples collected after March at stations EN and NB were excluded from abundance calculations, except for estimating entrainment at MNPS.

The distribution of larval abundance data over time is usually skewed because densities increase rapidly to a maximum and then decline slowly. A cumulative density over time from this type of distribution results in a sigmoid-shaped curve, where the time of peak abundance coincides with the inflection point. The Gompertz function (Draper and Smith 1981; Gendron 1989) was used to describe this cumulative abundance distribution because the inflection point of this function is not constrained to the mid-point of the sigmoid curve. The form of the Gompertz function used was:

$$C_t = \alpha \cdot \exp(-\exp[-\kappa \cdot \{t-p\}]) \qquad (2)$$

where C_t = cumulative density at time t
t = time in days from February 15
α = total or asymptotic cumulative density
p = inflection point scaled in days since February 15
κ = shape parameter

The time of peak abundance was estimated by the parameter p. The origin of the time scale was set to February 15, which is the approximate date when winter flounder larvae first appear in the Niantic River. Least-squares estimates, standard errors, and asymptotic 95% confidence intervals for these parameters were obtained by fitting the above equation to the cumulative abundance data using nonlinear regression methods (SAS Institute Inc. 1985). Cumulative data were obtained as the running sums of the weekly geometric means of the abundance data. The α parameter of the cumulative curve was used as an index to compare annual abundances.

A "density" function was derived algebraically by calculating the first derivative of the Gompertz function (Eq. 2) with respect to time. This density function, which directly describes the larval abundance over time (abundance curve), has the form:

$$d_t = \alpha' \cdot \kappa \cdot \exp(-\exp[-\kappa \cdot \{t-p\}] - \kappa \cdot [t-p]) \qquad (3)$$

where d_t = density at time t and all other parameters are as described for Equation 2, except for α', which was re-scaled by a factor of 7 (i.e., $\alpha' = 7\alpha$) because the cumulative densities were based on weekly geometric means and, thus, accounted for a 7-day period.

Larval mortality rates were estimated from data collected at the three Niantic River stations; data from 1983 were excluded as smaller larvae were undersampled because of net extrusion (NUSCO 1987). The abundance of 3-mm and smaller larvae was used to calculate an index of newly-hatched larvae because 3 mm was the approximate length at hatching. The decline in the frequency of larvae in progressively larger size-classes (in 1-mm groups) was attributed to both natural mortality and as a result of tidal flushing from the river. Hess et al. (1975) estimated the loss of larvae from the entire river as 4% per tidal cycle and also determined that the loss from the lower portion of the river was about 28% per tidal cycle. Thus, the weekly abundance estimates of larvae 3 mm and smaller at station C in the lower portion of the river were re-scaled by a factor of 1.93 to compensate for the 28% decline per tidal cycle (two cycles per day). The abundance of larvae in the 7-mm size-class was used to calculate an index of larval abundance just prior to metamorphosis. Because previous studies (NUSCO 1987, 1989) showed a net import of larger larvae into the Niantic River, the weekly abundance of larvae in the 7-mm size-class at station C was not adjusted for tidal flushing. To calculate each annual rate of mortality, sums were made of weekly mean abundance indices (three stations combined) of newly-hatched larvae (after adjusting for tidal flushing) and larvae in the 7-mm size-class. Survival rates from hatching through larval development were estimated as the ratio of the abundance index of the larger larvae (7-mm size-class) to that of the smaller larvae (3-mm and smaller size-classes).

The presence of density-dependent mortality was investigated by relating annual larval abundance in the 7-mm and larger size-classes from station EN to the annual egg production estimate for the Niantic River using the following relationship (Ricker 1975):

$$\log_e(L / E) = a + b \cdot E \qquad (4)$$

where L = annual larval abundance of larvae 7-mm and larger at EN as estimated by α (see Eq. 2)
E = annual estimate of egg production in the Niantic River
a = intercept
b = slope or index of mortality dependence upon annual egg abundance

Since the ratio L divided by E represents the fraction of larvae surviving from eggs to 7 mm, density-dependent mortality may be assumed when the slope (b) is significantly different from zero. This mortality is compensatory when the slope b is negative and depensatory if positive.

Regression analyses were used to examine possible relationships between variables and, at times, to make predictions. Ordinary least-squares linear regression was used when the independent variable was assumed to be measured without error (e.g., water temperature). The test of a relationship was based on the slope being significantly ($p \leq 0.05$) different from zero. Functional regression methods developed by Ricker (1973, 1984) were used in the cases where the independent variable was measured with error (e.g., abundance indices). For functional regressions, the probability that r (correlation coefficient) was significantly ($p \leq 0.05$) different from zero was the criterion used to decide whether a valid relationship existed prior to determining the slope and the 95% confidence interval for the slope.

Abundance, growth, and mortality of juveniles in summer

To analyze data and calculate CPUE, the catch of young-of-the-year winter flounder in each of the three or four replicated 1-m beam trawl tows was standardized to a 100-m tow distance before taking a mean; density was expressed as the number per 100 m^2 of bottom. For some comparisons among years, a moving average of three (1983-92) or two (1993-4) weekly density estimates was used to smooth fluctuations in abundance.

Nearly all of the age-0 winter flounder collected were measured fresh in either the field or laboratory to the nearest 0.5 mm in total length (TL). During the first few weeks of study, standard length (SL) was also measured because many of the specimens had damaged caudal fin rays and total length could not be ascertained. A relationship between the two lengths determined by a functional regression was used to convert SL to TL whenever necessary:

$$\text{TL in mm} = -0.2 + 1.212 \cdot (\text{SL in mm}) \qquad (5)$$

Growth of age-0 winter flounder at each station was examined by following weekly mean lengths throughout the sampling season. Mean lengths of young taken at the Niantic River stations LR and WA from late July through September were compared using an analysis of variance; significant differences among means were determined with Duncan's multiple-range test (SAS Institute Inc. 1985).

To calculate a total instantaneous mortality rate (Z), all young were assumed to comprise a single cohort. A catch curve was constructed such that the natural logarithm of density was plotted against time in weeks; the slope of the descending portion of the curve provided an estimate of the weekly rate for Z. Once this rate was determined, the monthly mortality rate (Z_{mo}) was calculated as $Z \cdot (30.4 / 7)$.

Abundance of juveniles during fall and winter

In fall and early winter, age-0 winter flounder gradually disperse from areas near the shoreline to deeper waters. Catch of these fish during this time period at the TMP stations (see the Fish Ecology section elsewhere in this report for methods) was also used as an index of relative abundance. Data used included November through February for inshore stations (NR and JC), December through February for nearshore Niantic Bay stations (IN and NB), and January and February at offshore stations (TT and BR). This resulted in a uniform sample size of 42 collections per season. These catches were pooled and used to calculate year-class abundance described by a Δ-mean CPUE (NUSCO 1988b). This index of abundance is the best estimator of the population mean when the data come from a distribution that contains numerous zero values and is approximately lognormal (Hennemuth et al. 1980; Pennington 1983, 1986).

The annual median CPUE of juveniles smaller than 15 cm (mostly age-1 fish) taken during the adult winter flounder spawning surveys was determined as described previously for fish larger than 15 cm. Median values were calculated for stations in the lower Niantic River navigational channel (1 and 2) as well as for all river stations combined, when sufficient data were available. For comparative purposes, an annual Δ-mean abundance index of juvenile fish of similar size was also determined using catch data from the five trawl monitoring program stations outside of the Niantic River during the period of January through April (annual sample size of 45 collections), which temporally overlapped the adult spawning surveys.

Stock and recruitment relationship

A stock-recruitment relationship (SRR) described by Ricker (1954, 1975) is the basis of the life-cycle algorithm that drives the population dynamics simulation model of Niantic River winter flounder. Application of this SRR to MNPS winter flounder stock assessment was described in detail in NUSCO (1989, 1990). The stock and recruitment data for determining the SRR were derived from the catch-at-age of female winter flounder during the Niantic River spawning survey. Because the spawning stock is made up of many year-classes, the true recruitment consists of the total reproductive contribution over the life of each individual in a given year-class (Garrod and Jones 1974; Cushing and Horwood 1977). Therefore, the index of annual parental stock size was based on derived egg production and the index of recruits or year-class size was based on calculated egg production accumulated over the life-time of the recruits. This method accounted for variations in year-class strength and in fecundity by size and age. The assumptions and methods used to age Niantic River winter flounder and to calculate a recruitment index expressed as equivalent numbers of female spawners were described in detail in NUSCO (1989, 1990) and is summarized below.

Stock and recruitment indices. Methods used to calculate the annual standardized catch index and total egg production of the parental stock were given previously. The recruitment index was determined by applying an age-length key described in NUSCO (1989) to the annual standardized catches of females partitioned into length categories. A common age-length key was used over all years because Witherell and Burnett (1993) reported that no trends were observed in mean length-at-age during 1983-91 for Massachusetts winter flounder despite a 50% reduction in biomass over that period. Aging the females allowed for the determination of their numbers by year-class present at ages 3, 4, 5, and 6^+ during successive spawning seasons. The age-6^+ group was further subdivided into the numbers of fish expected to survive to a terminal age of 15 by assuming various annual instantaneous mortality rates as fishing pressure increased from the 1970s into the 1990s. To follow each year-class from 1977 through 1990 to its terminal age (e.g., 2005 for the 1990 year-class), values of Z (= $F + M$) were used that represented estimates of current and anticipated annual instantaneous rate of fishing (F) as provided by the Connecticut Department of Environmental Protection (CT DEP). The instantaneous natural mortality rate (M) was assumed constant at 0.35 over all years. These were the same mortality rates used in the stochastic population dynamics model, discussed below. From observations made of abundance and age over the years, a large fraction of age-3 females, considerable numbers of age-4 fish, and even some age-5 females were apparently immature and not present in the Niantic River during the spawning season (NUSCO 1989). Thus, the total number of females was reduced to spawning females using length-specific proportions of mature fish estimated from annual catches in the Niantic River for fish age-3 to 5; all females age-6 and older were assumed to be mature. Because the estimates of age-3 fish were thought to be unreliable, this estimation process was only carried through the 1990 year-class (age-4 females taken in 1994). The adjusted numbers of mature fish provided an index of the fully recruited year-class expressed as the aggregated number of female spawners passing through each age-class. An implied assumption was that catches in the Niantic River were representative of the population, with the exception of immature fish that did not enter the river until fully recruited. Although this recruitment index could be used together with the annual number of female spawners to derive an SRR, this would ignore size composition differences that affected annual egg production. Therefore, the above index was adjusted for differences in fecundity among fish using the length-fecundity relationship of Niantic River winter flounder given above (Eq. 1). Finally, annual egg production was summed up over the lifetime of each year-class to determine the recruitment index as eggs and, then, converted to equivalent female spawners at the rate of one female spawner for each 561,000 eggs (i.e., the mean fecundity).

SRR parameters and biological reference points. The Ricker SRR appeared best suited for use with the Niantic River winter flounder stock because the relationship between recruitment and spawning stock indices was a dome-shaped curve with substantial decline in recruitment when the stock was larger than average (NUSCO 1989). Furthermore, this particular form of a SRR has been applied to other New England flounder stocks (Gibson 1989). The mathematical form of this SRR is:

$$R_t = \alpha \cdot P_t \cdot \exp(-\beta \cdot P_t) \qquad (6)$$

where R_t is the recruitment index for the progeny of the spawning stock P_t in year t and α and β are parameters estimated from the data. The α parameter describes the growth potential of the stock and $\log_e(\alpha)$, the slope of the SRR at the origin, is equivalent to the intrinsic natural rate of increase (Roughgarden 1979) when the stock is not exploited. The β parameter is the instantaneous rate at which recruitment declines at large stock sizes due to some form of density-dependent mortality. The natural logarithm of winter flounder recruitment was found correlated with mean water temperature during February at the intakes of MNPS, which is when most spawning and early larval development occurs (NUSCO 1988a, 1989). Therefore, the parameters α and β were estimated initially by fitting Equation 6 to the data and then re-estimated under the assumption that there was a significant temperature effect; this was accomplished by adding a temperature-effect component to Equation 6. Following Lorda and Crecco (1987) and Gibson (1987), annual mean water temperatures for a particular period were used as an explanatory variable to adjust the two-parameter SRR for temperature effects, which served to reduce recruitment variability and obtain more reliable parameter estimates for the SRR. The temperature-dependent SRR had the form:

$$R_t = \alpha \cdot P_t \cdot \exp(-\beta \cdot P_t) \cdot \exp(\phi \cdot T_{Feb}) \qquad (7)$$

where the second exponential describes the effect of February water temperature on recruitment and the new parameter ϕ represents the strength of that effect. This effect either decreases or increases the number of recruits-per-spawner produced each year because temperature was defined as the deviation (T_{Feb}) of each particular mean February temperature from a long-term (1977-90) average of February water temperatures. When the February mean water temperature is equal to the long-term average, the deviation (T_{Feb}) in Equation 7 becomes zero and the exponential term equals unity (i.e., no temperature effect). Thus, Equation 7 reduces to its initial form (Eq. 6) under average temperature conditions. Nonlinear regression methods (SAS Institute Inc. 1985) were used for estimating the parameters in the above equations.

Fishing mortality (F) is an important factor affecting the growth potential of the stock (Goodyear 1977) and, thus, is relevant for assessing other impacts. Because fishing and natural mortality of winter flounder take place concurrently through the year, the actual fraction of the stock removed by the fishery each year (i.e., the exploitation rate) is obtained as:

$$u = (F / Z)(1 - \exp[-Z]) \qquad (8)$$

Stock-recruitment theory and the interpretation of several biological reference points derived from Ricker's SRR model were discussed in detail in NUSCO (1989). The equilibrium or sustainable stock size of an exploited stock (i.e., when $F > 0$) is given by:

$$P_{E(F)} = (\log_e[\alpha] - F) / \beta \qquad (9)$$

Rearranging the terms and solving for the rate of fishing that would achieve a given equilibrium stock size results in:

$$F = \log_e(\alpha) - \beta \cdot (P_{E(F)}) \qquad (10)$$

For $F = 0$, Equation 9 becomes the equilibrium or replacement level of the unfished stock:

$$P_{rep} = (\log_e[\alpha]) / \beta \qquad (11)$$

The fishing rate for "recruitment overfishing" has been recently defined for winter flounder stocks as the rate of fishing that reduces the stock biomass to less than 25% of the maximum spawning potential (Howell et al. 1992).

Although the above equations (9-11) can be used to calculate equilibrium stock sizes and fishing rates for the winter flounder, the results are only deterministic approximations that ignore age-structured effects. Therefore, these equations are primarily useful to calculate initial values of the corresponding biological reference points. These are better estimated through simulations using the SPDM or other similar population or production models that include age structure and both natural and fishing mortality.

Assessment of MNPS operation on Niantic River winter flounder

Several well-established methods available for stock assessment are based on stock-recruitment theory (Smith 1988). These methods assume constant fishing rates and populations with stable age-structure, which result in equilibrium or steady-state stocks that replace themselves year after year. Some analytical methods are based on equilibrium equations, such as Equations 9 through 11, which have been modified to incorporate effects of mortality caused by activities other than fishing. Several problems may exist with an SRR-based approach to impact assessment at MNPS. Because stock-recruitment theory (Ricker 1954) was developed for semelparous fish (i.e., those which spawn only once in their lifetime), Equation 11 may provide unreliable estimates of equilibrium stock sizes for iteroparous fish (multi-aged spawning stocks), such as the winter flounder. Although the parameter α in Equation 9 could be adjusted for the effect of repeat spawning, this equation also assumes that no fishing mortality occurs prior to maturation. This assumption cannot be met in the case of winter flounder because many immature fish (ages-2 and 3) are vulnerable to fishing gear. Wigley and Gabriel (1991) noted that concentrations of immature winter flounder found off Rhode Island may be subjected to significant mortality from fishing. Howell and Langan (1987, 1992) found that discard mortality rates of trawl-caught fish in New England waters may be substantial. Simpson (1989) reported that about 72% of LIS winter flounder landed by the commercial fishery were between 28 and 32 cm; many of these fish would have been age-3. Additional problems are found when applying deterministic models (i.e., assuming steady-state conditions) to fish stocks whose exploitation rates are not stable, especially when such stocks increase in abundance, as in the case of the winter flounder during the late 1970s and early 1980s (Smith et al. 1989). Environmental variability also results in year-to-year variation of natural mortality rates, which further weakens the results of deterministic assessments.

An approach to stock assessment incorporating environmental variability and all types of mortality, both constant and variable, involves the computer simulation of fish populations using a simple model of population renewal with spawning stock feed-back (e.g., a functional stock-recruitment relationship). This approach has two advantages: assumptions of population equilibrium are not necessary, and much detail can be incorporated into the conditions or scenarios used to simulate changes in fish populations through time. An additional advantage is that Monte Carlo methods readily provides the stochastic (as opposed to deterministic) framework needed for probabilistic risk assessment and for testing hypotheses about the probable size of the stock at some future point. This simulation approach was applied in NUSCO (1990) to assess the impact of larval entrainment under a simple scenario. In NUSCO (1991b), the same approach used various combinations of historic and projected fishing and larval entrainment rates to assess more realistically the impact of MNPS operations on local winter flounder. In NUSCO (1992a), the impact resulting from the impingement of juvenile and adult winter flounder was also simulated. The basic steps leading to the final impact assessment using this simulation approach are: direct estimation of annual larval entrainment rates at MNPS; mass-balance calculations to estimate the fraction of Niantic River annual flounder production lost through larval entrainment at MNPS; estimation of the equivalent instantaneous mortality rates of females that were attributed to impingement; stochastic simulation of the winter flounder stock dynamics to predict stock biomass at selected levels of entrainment and fishing rates; and an analyses of simulation results leading to estimates of the probability that the stock would fall below selected reference sizes.

Estimates of larval entrainment at MNPS

The estimated number of larvae entrained in the MNPS condenser cooling water system each year is a direct measure of impact on the local winter flounder stock. Annual estimates were determined using larval densities at station EN (Fig. 3) and the volume of cooling water used by MNPS. The Gompertz density function (Eq. 3) was fitted to larval data and daily densities (number$\cdot$500m^{-3}) were calculated. Daily entrainment estimates were determined after adjusting for the daily condenser cooling-water volume and an annual estimate was calculated by summing all daily estimates during the larval season.

The reduction in entrainment as a result of the January 15 - May 23 refueling outage at Unit 1 was determined by computing entrainment estimates for these dates using the methodology given above for all three units combined and then simulating estimates using full Unit 1 flows (29.18 m$^3\cdot$sec^{-1}) for the same period. The difference between the two estimates was the reduction in entrainment attributed to the outage.

Mass-balance calculations

The number of winter flounder larvae entrained depends upon larval densities in Niantic Bay. Potential impact to the Niantic River stock from larval entrainment should be related to the number of larvae in Niantic Bay that originated from the river. Mass-balance calculations were used to investigate whether the number of winter flounder larvae entering Niantic Bay from the Niantic River could sustain the number of larvae observed in the bay during the winter flounder larval season each year (1984-94). Three potential larval inputs to Niantic Bay include eggs hatching in the bay, larvae flushed from the Niantic River, and larvae entering the bay from LIS across the boundary between Millstone Point and Black Point (Fig. 3). The few yolk-sac larvae collected annually in Niantic Bay suggested that minimal spawning and subsequent hatching occurred in the bay, which was therefore considered a negligible source of larvae. Larvae were known to be flushed from the river into the bay and this input to the bay was estimated from available data. The number of larvae entering Niantic Bay from LIS was unknown. Four ways in which larvae may leave Niantic Bay include natural mortality, entering the Niantic River during a flood tide, being entrained at MNPS, and flushing from the bay into LIS. Estimates could be made for the number of larvae lost through natural mortality, entering the Niantic River, and entrained at MNPS, but little was known about the number of larvae flushed into LIS. The numbers of larvae flushed to and from LIS were combined as the unknown (*Source or Sink*) in the mass-balance calculations. Thus, the form of the mass-balance equation was:

$$NB_{t+5} = NB_t - Ent - Mort + FromNR - ToNR \pm (Source\ or\ Sink) \qquad (12)$$

where t = time in days

NB_{t+5} = number of larvae in Niantic Bay 5 days after day t (instantaneous daily estimate)

NB_t = initial number of larvae in Niantic Bay on day t (instantaneous daily estimate)

Ent = number of larvae lost from Niantic Bay due to entrainment in the condenser cooling-water system (over a 5-day period)

$Mort$ = number of larvae lost from Niantic Bay due to natural mortality (over a 5-day period)

$FromNR$ = number of larvae flushed from the Niantic River (over a 5-day period)

$ToNR$ = number of larvae entering the Niantic River (over a 5-day period)

$Source\ or\ Sink$ = unknown number of larvae in Niantic Bay that flush to LIS or enter the bay from LIS (over a 5-day period)

Solving for the unknown *Source or Sink* term, the equation was rearranged as:

$$Source\ or\ Sink = NB_{t+5} - NB_t + Ent + Mort - FromNR + ToNR \qquad (13)$$

Because these mass-balance calculations were based on the change in the number of larvae in Niantic Bay over a 5-day period:

$$\text{5-day change} = NB_{t+5} - NB_t \qquad (14)$$

Thus:

$$\text{Source or Sink} = \text{5-day change} + Ent + Mort - FromNR + ToNR \qquad (15)$$

Daily abundance estimates were derived from the Gompertz density equation (Eq. 3) and the daily densities for Niantic Bay at two points in time (NB_t and NB_{t+5}) for each 5-day period were calculated from data collected at stations NB and EN combined. These densities, adjusted for the volume of Niantic Bay (about 50×10^6 m^3; E. Adams, Massachusetts Institute of Technology, Cambridge, MA., pers. comm.), provided an estimate of the instantaneous daily standing stock. The difference between these two estimates (NB_t and NB_{t+5}) was the term *5-day change* in Equation 15. The selection of 5 days as the period of change was arbitrary and a cursory examination of results based on 10-day periods showed that the same conclusions were reached with either 5- or 10-day periods.

Daily entrainment estimates were based on data collected at station EN and the actual daily volume of condenser cooling water used at MNPS. The daily entrainment estimates were summed over each 5-day period (*Ent*). Annual stage-specific mortality rates for 1984-89 were determined by Crecco and Howell (1990), for 1990 (V. Crecco, DEP Division of Marine Fisheries, Old Lyme, CT, pers. comm.), and for 1991 through 1994 by NUSCO staff. Mortality was partitioned among developmental stages by comparing the rates of decline of predominant size-classes of each stage. Each developmental stage was assigned a portion of the total annual larval mortality rate (Z); similar mortality rates were assumed for Stages 3 and 4. Although estimating stage-specific mortality in this manner was not precise, sensitivity analysis on the mass-balance calculations (NUSCO 1991b) indicated that larval mortality was the least sensitive parameter. These annual rates were modified to daily stage-specific mortality rates by assuming 10-day stage durations for Stages 1, 3, and 4 larvae, and 20 days for Stage 2 larvae. The proportion of each stage collected at station EN during each 5-day period was applied to the daily standing stock for Niantic Bay (NB_t) to estimate the number of larvae in each developmental stage for stage-specific mortality calculations. The daily loss due to natural mortality was summed for each 5-day period (*Mort*).

The 5-day input of larvae to Niantic Bay from the river (*FromNR*) was based on daily density estimates for station C in the river after adjusting for the rate of flushing between station C and the mouth of the river. To determine the relationship between the estimated daily density at station C and the average density of larvae leaving the river on an ebb tide, the geometric mean density of samples collected during an ebb tide for ten import-export studies conducted at the mouth of the Niantic River during 1984, 1985, and 1988 (NUSCO 1985, 1986a, 1989) were compared to the estimated daily densities at station C. The average density of larvae flushed from the Niantic River was estimated by the significant ($r = 0.969$; $p = 0.001$) functional regression equation:

$$FromNR = 9.751 + 0.473 \cdot (\text{Daily density at station C}) \qquad (16)$$

The 95% confidence interval for the slope was 0.387 - 0.579. The estimated average density, the average tidal prism of 2.7×10^6 m^3 (Kollmeyer 1972), and about 1.9 tidal prisms per day were used to estimate the daily flushing of larvae from the river into Niantic Bay. This daily input to the bay was summed for each 5-day period to calculate the term *FromNR* in the mass-balance equation.

Stepwise oblique tows were collected during 1991 in the channel south of the Niantic River railroad bridge (station RM) during a flood tide to estimate an average density to compute *ToNR* (NUSCO 1992a). In 1992 and 1993, sampling again was conducted at RM during a flood tide, but the collections were made by mooring the research vessel to the railroad bridge and taking continuous oblique tows (NUSCO 1994a). Comparison of densities from the paired stations of NB and RM showed a poor relationship. Therefore, daily densities at the two stations were estimated using the Gompertz density curve (Eq. 3). For station RM in 1992, the equation could only be adequately fit by smoothing the data using a 3-week running average prior to calculating a weekly cumulative density. The Gompertz function could not be fit to data collected at station NB during 1993. Therefore, catches from stations NB and EN were combined to calculate the weekly geometric means prior to fitting the Gompertz function and estimating daily densities for Niantic Bay. Daily density estimates for 1991-93 were combined and functional regression was used to determine the relationship between abundance at stations NB and RM. The average density of larvae flushed from Niantic Bay into the river was estimated by the significant ($r = 0.705$; $p = 0.001$) functional regression equation:

$$ToNR = 128.149 + 2.073 \cdot NB_t \qquad (17)$$

The 95% confidence interval for the slope was 1.827 - 2.351. After being adjusted for the average tidal prism and the number of tidal prisms per day, these daily estimates of the number of larvae entering the river during a flood tide were summed over each 5-day period to calculate the term *ToNR* in the mass-balance equation. Because of the large intercept in the above regression line when no larvae were present in Niantic Bay ($NB_t = 0$), the term *ToNR* was conservatively set to zero. The term *Source or Sink* in Equation 15 represents the net loss from or gain to Niantic Bay of larvae from LIS during a 5-day period that is required to balance the calculation. For a net loss of larvae (flushed to LIS), the *Source or Sink* term would be negative and for a net gain of larvae (imported from LIS), the *Source or Sink* term would be positive.

Stochastic simulation of winter flounder stock dynamics

Modeling strategy and background. The stochastic population dynamics model (SPDM) developed for the Niantic River winter flounder stock was based on the Ricker SRR fitted to the data, even though Equation 7 does not explicitly appear in the model formulation. The mechanisms underlying the Ricker form of recruitment are incorporated in the set of equations that the model uses to calculate mortality through the first year of life. Beyond that point (i.e., age-1) in the life-cycle simulation, the population model simply describes the annual reduction of each year-class through natural mortality and fishing together with aging and reproduction. This process occurs at the beginning of each model time-step of length equal to 1 year. The projection of adult fish populations over time has been implemented in many models by means of Leslie matrix equations (e.g., Hess et al. 1975; Vaughan 1981; Spaulding et al. 1983; Reed et al. 1984; Goodyear and Christensen 1984). In the SPDM, adult winter flounder were projected over time by grouping fish into distinct age-classes and by carrying out the computations needed (mostly additions and multiplications) iteratively over the age index (1 through 15) and over the number of years specified for each simulation. This approach was algebraically identical to the Leslie matrix formulation, facilitated the understanding of how the model works, and simplified the computer code when describing the fish population either as biomass (allowing for size variation within each age-class) or numbers of fish. A similar implementation of an adult fish population dynamics simulation was used by Crecco and Savoy (1987) in their model of Connecticut River American shad (*Alosa sapidissima*).

Model components. Figure 4 illustrates the components of the computer program used for the SPDM. Components depicted by solid-line boxes constitute the model presently in use and one depicted by a box with dashed lines illustrates a part of the model that was not used in the present application, but could be used in future applications. The functionality of most model components should be clear from the flow chart and no further details will be provided. Some critical components, such as the one labeled age-1 cohort and the two random input boxes are described below. A list of the actual input data used in the application of the model to the Niantic River winter flounder stock is also given.

The most critical aspects in the formulation of a stock-recruitment based population model are the specific equation and parameters used to calculate total mortality during the first year of life (i.e., from egg through age-1). The equation used for this purpose in the SPDM was derived from Ricker's equilibrium equation for Z_0 (total instantaneous mortality from egg through maturation age). This involved the extension of stock-recruitment theory, which was developed for fish that spawn only once, to iteroparous fish with multi-age spawning stocks. The form of the equation as used in the present model was:

$$Z_{0,t} = \log_e(\mathrm{FEC}) + \log_e(\mathrm{ASF}) - \log_e(\alpha) + n_t - \phi \cdot \mathrm{WT}_t - Z_{1,2} + \beta \cdot P_t \qquad (18)$$

where the subscript t denotes the time-step (each time-step represents a year) and non-subscripted terms remain constant from year to year; α, β, and ϕ are the parameters of the SR function (Eq. 7); FEC is the mean fecundity of the stock expressed as the number of female eggs produced per female spawner; ASF is a scaling factor to adjust α for the effect of a multi-age spawning stock; n_t and WT_t are independent random variates from two specified normal distributions that

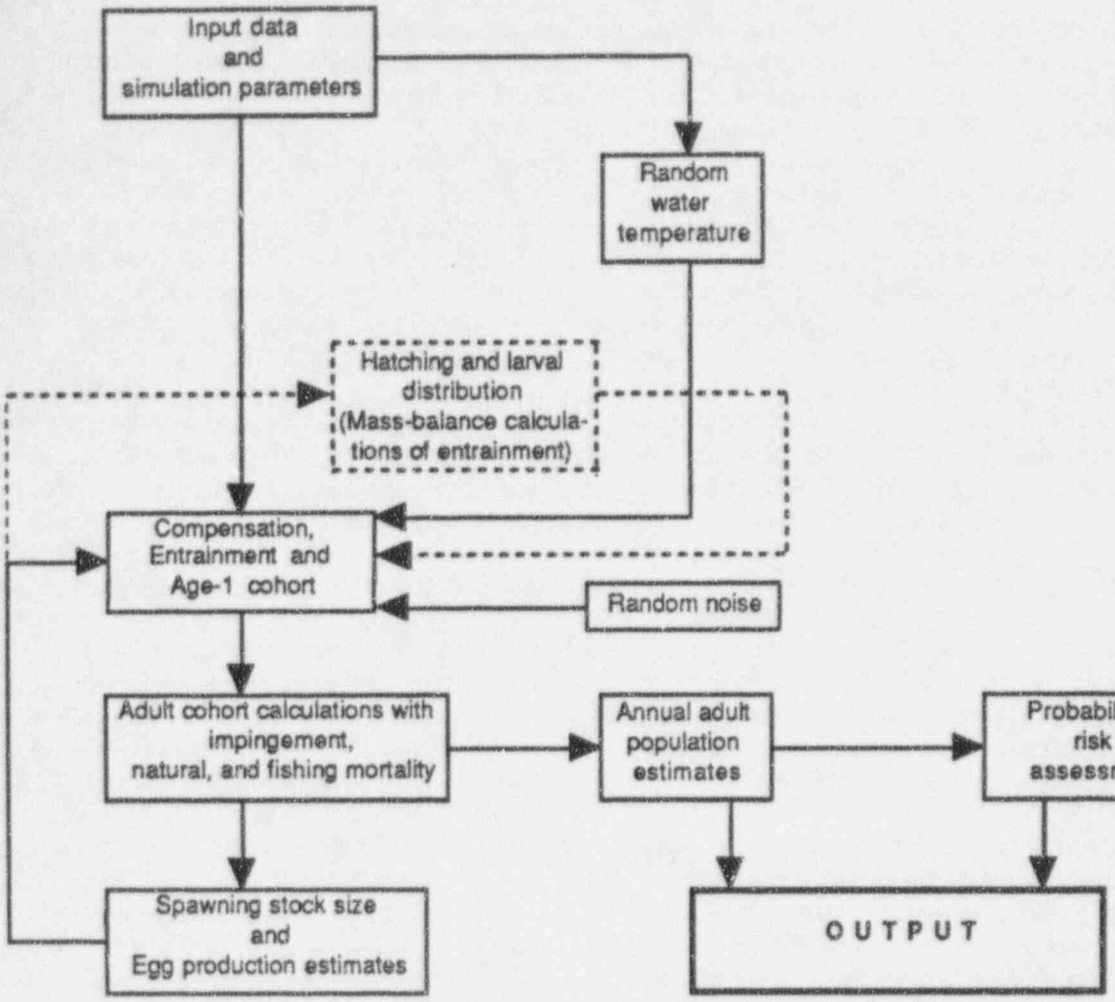


Fig. 4. Diagram of NUSCO stochastic population dynamics computer model for assessing the long-term effect of larval winter flounder entrainment at MNPS. Dashed boxes and arrows refer to components and calculations which are not an integral part of the model.

are described below; $Z_{1,2}$ is the instantaneous mortality through the immature age-classes; and the last term ($\beta \cdot P_t$) is the feed-back mechanism simulating stock-dependent compensatory mortality, which varies according to the size of the annual spawning stock P_t. The complete derivation of the above equation was given in NUSCO (1990: appendix to the winter flounder section). The scaling factor ASF is a multiplier that converts age-3 female recruits into the total spawning potential of the year-class. This spawning potential is defined as the cumulative number of mature females from the same year-class that survive to spawn year after year during the lifetime of the fish. The algebraic form of this multiplier is identical to the numerator of Equation A-4 in Christensen and Goodyear (1988).

Stochasticity in the winter flounder model (Fig. 4) has two annual components: a random term that represents uncertainties associated with the estimate of Ricker's α parameter and annual environmental variability in the form of random deviations from the long-term mean February water temperature. These two components of annual variability are incorporated into the calculation of each new year-class via the mortality from egg to age-1 (Eq. 18). The term n_t in Equation 18 (random noise) is simulated as independent random variates from a normal distribution with zero mean and variance equal to σ^2. The value of σ is estimated during the model calibration runs as the amount of variance required to generate α values within the 95% confidence interval of the estimate of α used in the model (NUSCO 1990). Similarly, the term $\phi \cdot WT_t$ in Equation 18 represents the effect of annual environmental variability of February water temperatures on larval survival. This effect becomes random when the February water temperatures are themselves simulated as independent random variates from a normal distribution with mean and variance equal to the mean and variance of February water temperatures at the MNPS intakes for 1977-90.

The stochastic simulation of fish population dynamics provides a framework for probabilistic risk assessment methodology. This type of assessment is based on Monte Carlo methods (Rubinstein 1981), where many independent random replicates of the time-series are generated so that the mean of the series and its standard error can be estimated. Monte Carlo replications can be used to derive the sample distribution function (Stuart and Ord 1987) without assuming any particular statistical distribution. This methodology was used to assess the risk of stock reduction resulting from the effects of entrainment and impingement at MNPS. The probabilities of stock reductions were empirically derived from 100 Monte Carlo replicates of winter flounder annual abundances in the time-series of impacted stocks. Briefly, the probability that a stock will be smaller than some postulated size is given by the proportion of replicates that are smaller than the reference size in a given year. Additionally, the 5th and 95th percentiles of the frequency distribution of stock sizes for specific years were calculated. These percentiles help describe the uncertainty associated with point estimates of annual stock sizes in the SPDM projections.

SPDM assumptions and limitations. Major assumptions relate to the underlying form of the SRR used and the reliability of the SRR parameter estimates. Because the SPDM incorporated the Ricker form of SRR, it was assumed that stock-dependent compensation and the postulated effect of water temperature on larval survival (Eqs. 7 and 18) applied reasonably well to the Niantic River winter flounder stock. A second assumption was that the three parameters of the SRR were correctly estimated and that α, in particular, was a reliable estimate. Although the population was not assumed to be at steady state, the average fecundity and survival rates for fish age-1 and older were assumed to remain fairly stable over the period corresponding to the time-series data used to estimate the SRR parameters. Although this last assumption can generally be met in the case of fecundity rates and adult natural mortality, fishing mortality rates are much less stable. Changes in exploitation rates from year to year should not cause estimation problems as long as the changes are not systematic (i.e., change in the same direction year after year). Because these assumptions are seldom completely met, early applications of the model (NUSCO 1990) included calibration runs to validate predictions under both deterministic and stochastic modes by comparing model results to recent series of stock abundance data. Finally, no temperature trend or large-scale environmental changes (e.g., global warming) were assumed to have occurred during the years simulated in each population projection.

Model input data. The dynamics of the Niantic River winter flounder stock were simulated using the SPDM under a credible real-time scenario running from 1960, well before operation of Unit 1, to 2060, long after the projected shutdown date for Unit 3 in 2025 (Table 1). The scenarios used power plant effects based on actual generating units in operation each year, concurrently with estimates of F that were based on historic and projected rates of commercial exploitation and sport fishing for winter flounder in Connecticut. Parameters used in the SPDM included: F, with an additional mortality equivalent of 0.01 to account for impingement (IMP) losses (NUSCO 1992a); conditional mortality rates (i.e., fraction of the annual production of winter flounder removed as a result of power plant operation) determined for larval entrainment (ENT); a schedule of changes when any of these rates was not assumed constant; and the length of the time-series in years. The combined mortality of F + IMP was used only during the simulation period (1971-2025) that corresponded to MNPS operation (Table 1).

Because the ability of a fish stock to withstand additional stress is reduced by fishing mortality (Goodyear 1980), simulations of the long-term entrainment of winter flounder larvae also included effects due to the substantial exploitation of the stock. The annual schedule of nominal fishing rates was determined from recent DEP estimates (V. Crecco, CT DEP, Old Lyme, CT, pers. comm.). These exploitation rates took into account length-limit regulations in effect from 1982-94 and from changes in regulations proposed by the DEP to reduce fishing mortality in Connecticut waters (Tables 2 and 3). Vulnerability factors for age-classes 1 through 5+ were calculated for

TABLE 1. Cooling-water requirements and dates of operation for MNPS Units 1 through 3, each with an assumed life-span of 40 years.

Unit	Cooling-water flow ($m^3 \cdot sec^{-1}$)	Fraction of MNPS total flow	Start-up date	First year of entrainment	Projected last year of operation
1	29.18	0.227	November 1970	1971	2010
2	37.62	0.292	December 1975	1976	2015
3	61.91	0.481	April 1986	1986	2025
MNPS total	128.71	1.000			

TABLE 2. Eastern Long Island Sound winter flounder length-limit and seasonal closure regulations in effect or proposed for the commercial and sport fisheries since 1982.

	Minimum length limit (in)		Minimum length limit (mm)		
Period	Commercial fishery	Sport fishery	Commercial fishery	Sport fishery	Seasonal closure
1982[a]	8	8	203	203	None
1983 (Jan-May)	8	8	203	203	None
1983 (Jun-Dec)	11	8	279	203	None
1984 (Jan-Aug)	11	8	279	203	None
1984 (Sep-Dec)	10	8	254	203	None
1985-1986	10	10	254	254	None
1987 (Jan-Aug)	10	10	254	254	Dec 1 - Mar 31 (within Niantic River)
1987 (Sep-Dec)	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
1988-1993[b]	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
≥ 1994[c]	12	11	305	279	Mar 1 - Apr 14 (in all state waters)

[a] Prior to 1982 there were no size regulations, but it was assumed that fish between 6 inches (152 mm) and 8 inches (203 mm) were subjected to about 50% of the nominal fishing mortality for each year. Fish larger than 8 inches were fully recruited to the fishery.

[b] Minimum trawl mesh codend size also increased from 4.5 to 5.0 inches.

[c] Includes an increase in the minimum trawl mesh codend size to 5.5 inches for the commercial fishery and an 8 fish creel limit for the sport fishery.

TABLE 3. Vulnerability factors[a] for eastern LIS winter flounder by age[b], adjusted for discard mortality of undersized fish vulnerable to the commercial (60% of total landings) and sport (40%) fisheries, according to fishing regulations in effect for the periods listed.

	Commercial					Sport					Total fishery				
Period	1	2	3	4	5+	1	2	3	4	5+	1	2	3	4	5+
≤ 1981	0.03	0.36	0.60	0.60	0.60	0.06	0.24	0.40	0.40	0.40	0.09	0.60	1.00	1.00	1.00
1982	0.00	0.36	0.60	0.60	0.60	0.06	0.13	0.40	0.40	0.40	0.06	0.49	1.00	1.00	1.00
1983-84	0.00	0.30	0.60	0.60	0.60	0.06	0.13	0.40	0.40	0.40	0.06	0.43	1.00	1.00	1.00
1985-87	0.00	0.30	0.60	0.60	0.60	0.06	0.06	0.40	0.40	0.40	0.06	0.36	1.00	1.00	1.00
1988-92	0.00	0.12	0.57	0.60	0.60	0.06	0.06	0.40	0.40	0.40	0.06	0.18	0.97	1.00	1.00
1993	0.00	0.04	0.42	0.56	0.60	0.06	0.06	0.40	0.40	0.40	0.06	0.10	0.82	0.96	1.00
≥ 1994	0.00	0.01	0.25	0.50	0.60	0.06	0.06	0.30	0.40	0.40	0.06	0.07	0.55	0.90	1.00

[a] These factors assume discard mortality at 50% the nominal F rate for fish caught by commercial gear and at 15% of the nominal F rate for all undersized fish caught by anglers (CT DEP estimates; P. Howell, Old Lyme, CT, pers. comm.).

[b] The notation 5+ refers to fish that are age-5 and older.

the commercial fishery (60% of the total winter flounder catch) and were based on actual or proposed changes in length limits and minimum commercial trawl fishery codend sizes; the size-at-age of female Niantic River winter flounder at mid-year (age + 0.5) determined using the von Bertalanffy growth equation

(NUSCO 1987); selection curves for 114-mm (4.5-in), 127-mm (5-in), and 140-mm (5.5-in) trawl mesh codends provided by the DEP; and a discard mortality rate of 50% for undersized fish. The sport fishery was estimated to take 40% of the total catch, having a discard mortality rate of 15%. The values of F used in the simulations were stepped up from 0.40 in the 1960s to a peak of 1.30 in 1991 (Fig. 5), which reflected the recent historical increase in fishing and the current high exploitation of winter flounder. The value of F was subsequently reduced to meet a targeted rate of 0.50 by 2001. Although the Atlantic States Marine Fisheries Commission management plan for inshore stocks of winter flounder (Howell et al. 1992) calls for a further reduction in F to about 0.43, the perhaps more realistically attainable value of 0.50 was used for all remaining years after consulting with DEP staff (V. Crecco and P. Howell, DEP Division of Marine Fisheries, Old Lyme, CT, pers. comm.). The effect of the changing fishing rates on partially vulnerable fish is seen in Figure 6. As a result of more protective regulations, the effect of commercial fishing on ages-1 and 2 has been or will be greatly diminished and many age-3 and 4 fish should be protected as well. The derivation of the equivalent mortality rate IMP was given in NUSCO (1992a) and is an additional small (0.01) component of mortality added to F during the years of MNPS operation. Other data, rates, and inputs to the SPDM are summarized on Table 4 and include the number of age-classes, age-specific rates of maturation, natural mortality, average weight and fecundity at age, the three-parameter SRR estimates, February water temperature statistics, and other specific factors for each simulation.

Conditional mortality rates for larval entrainment (ENT) from 1984 through 1994 used in SPDM simulations under actual operating conditions were estimated directly using the mass-balance calculations described above. Values of ENT determined for other years were varied stochastically. An annual value of ENT was chosen from the range of values determined

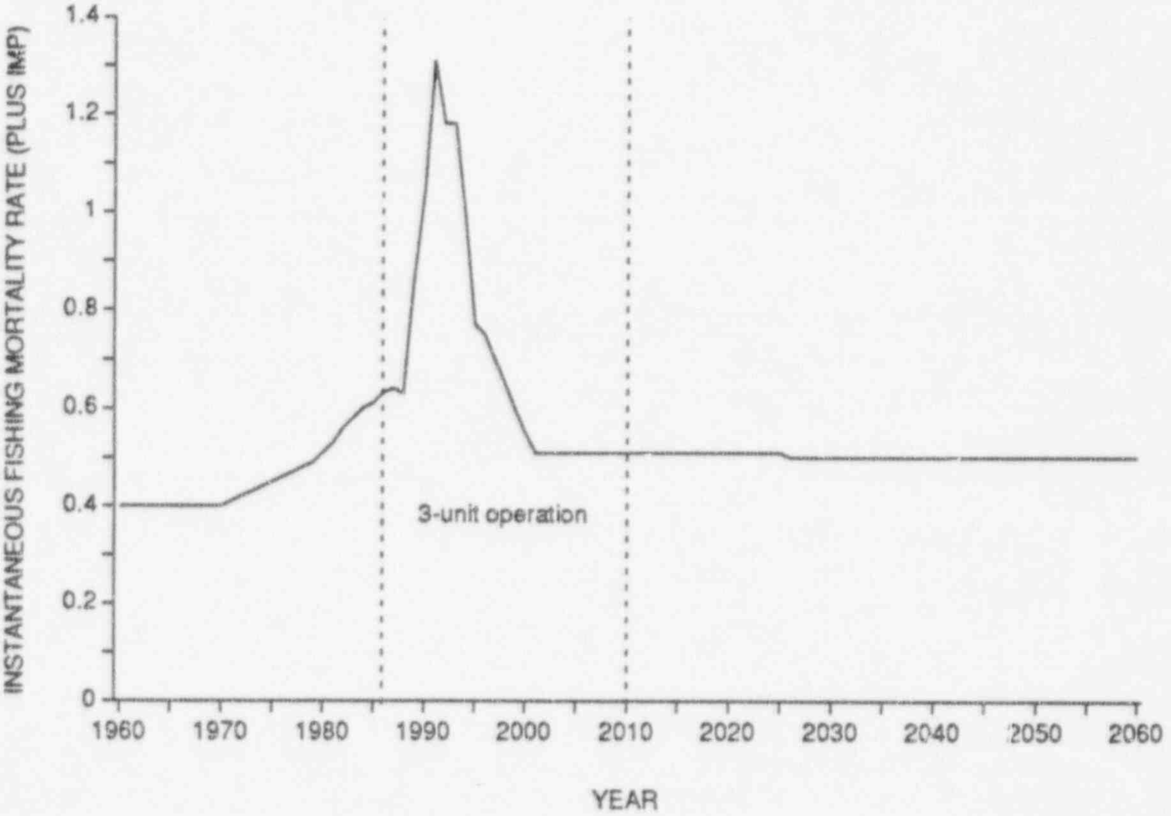


Fig. 5. Historic and projected annual mortality rate due to fishing (F), as determined in consulation with the CT DEP, plus a small (0.01) component accounting for impingement mortality (IMP) at MNPS as implemented in the SPDM simulations.

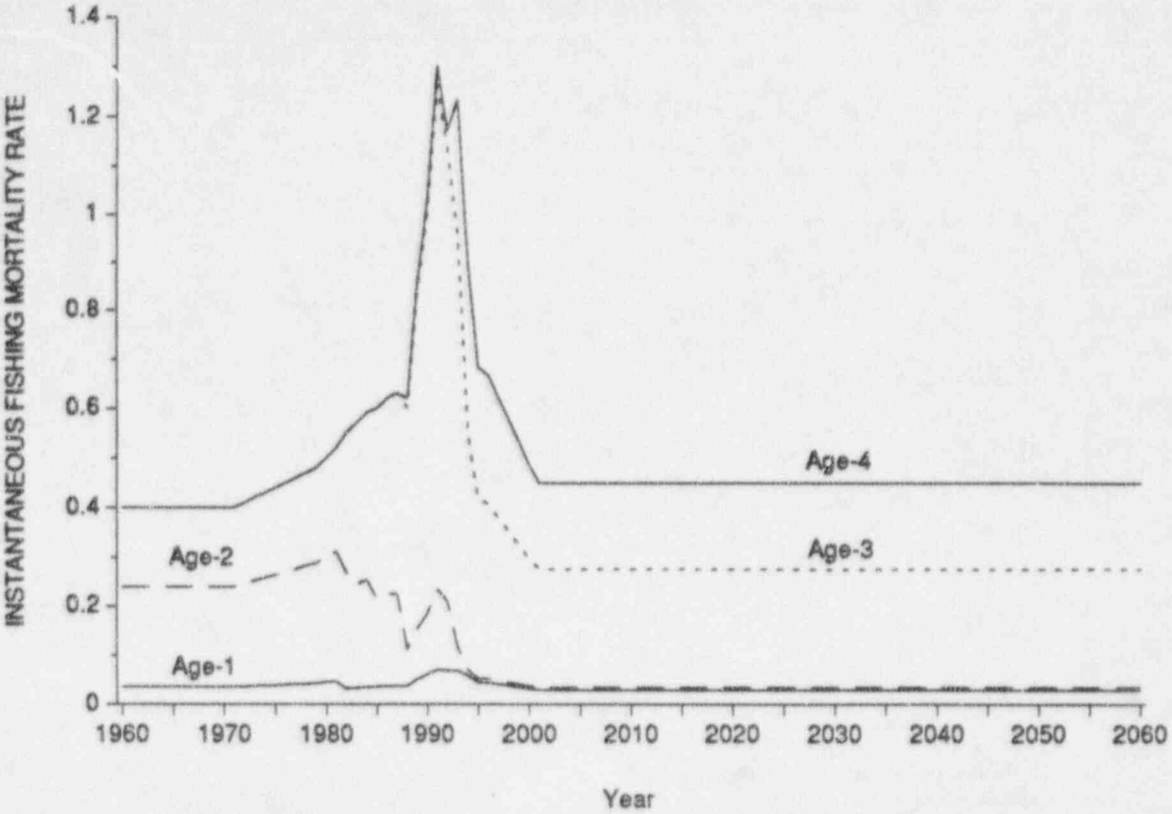


Fig. 6. Estimated reductions in F (including discard mortality) for age-1 through 4 winter flounder as a result of actual or planned regulations imposed by the CT DEP on the winter flounder commercial and sport fisheries.

from the mass-balance calculations for full MNPS three-unit operation. This was done by resampling with replacement using uniform probabilities to randomize the process. These estimates were calculated under the assumption that all three units used cooling water pumped at maximum capacity (11.1×10^6 m^3-day^{-1}). The selected value of ENT was then scaled by both the number of units in operation in a particular year (Table 1) and the fractions of cooling-water flow actually used during the annual March-May larval winter flounder season (Table 5). MNPS cooling-water use was known for 1976 through 1994 and actual flow values were used to scale the randomly selected value of ENT. Because no data were available during 1971-75 for Unit 1, flow values for these years were estimated from net electrical generation records. Estimates for 1972 and 1975, years in which this unit apparently operated near maximum capacity, were normalized to the value for 1987, the year of maximum flow for the Unit 1 time-series; other years were scaled accordingly. Since the simulation time-series extended to 2060 (including a recovery period following the end of MNPS operation), historic cooling-water flow rates calculated for 1971-94 were re-used to predict entrainment for 1995-2025 by resampling the historic flows with replacement using uniform probabilities to randomize the process. This approach assumed that the existing 24-year record of MNPS operation adequately described the operational variability expected at the station in the future. Except for those cases where randomly chosen values for a year had all three units operating near 100% capacity, annual values of ENT used in the simulations were less than the theoretical maximum under full three-unit operation.

Simulation of MNPS impact. Simulation output consisted of a time-series of annual stock sizes generated under a specified set of population parameters and conditions (including random variability) that constituted a scenario. All model runs consisted of 100 replicates of the 1960-2060 stock projection series. The final population projection resulted from averaging these identically generated 100 replicate time-series, except for the random components used to compute annual fish survival rates. It was previously concluded that 100 replicates were sufficient, given the

TABLE 4. Data, rates, and other inputs used with the Niantic River winter flounder population dynamics simulation model.

Model input	Value used or available		
Number of age-classes in population		15	
Earliest age at which all females are mature		6	
Fraction mature, mean wt (lbs), and mean fecundity by age:			
Age-1 females	0	0.011	0
Age-2 females	0	0.125	0
Age-3 females	0.08	0.554	223,735
Age-4 females	0.36	0.811	378,584
Age-5 females	0.92	1.089	568,243
Age-6 females	1.00	1.377	785,897
Age-7 females	1.00	1.645	1,004,776
Age-8 females	1.00	1.873	1,201,125
Age-9 females	1.00	2.057	1,366,951
Age-10 females	1.00	2.203	1,502,557
Age-11 females	1.00	2.304	1,598,597
Age-12 females	1.00	2.390	1,682,208
Age-13 females	1.00	2.461	1,754,800
Age-14 females	1.00	2.516	1,809,000
Age-15 females	1.00	2.552	1,845,800
Age after which annual mortality is constant		3	
Instantaneous mortality rates M and F at age-1	0.50	0[a]	
Instantaneous mortality rates M and F at age-2	0.35	0	
Instantaneous mortality rates M and F at age-3+	0.35	0	
Initial number of female spawners		67,658[b]	
Biomass of female spawners		98,104 lbs	
Mean fecundity of the stock (eggs per female spawner)		871,000[b]	
α from the three-parameter SRR for the virgin ($F = 0$) stock (numbers of fish)		5.42[c]	
β from the three-parameter SRR		2.498×10^{-5}	
ϕ from the three-parameter SRR		-0.379	
Mean February (1977-90) water temperature (°C)		2.61	
standard deviation		1.16	
minimum temperature		0.36	
maximum temperature		4.28	
Number of spawning cycles (years) to simulate		100	
Number of simulation replicates per run		100	
Fraction of age-0 group entrained at MNPS (i.e., impact)		0.00[d]	

[a] Values are entered here only when mortalities remain constant during all the spawning cycles or years simulated. Zero values direct the model to get a detailed schedule of mortalities from an auxiliary input file set up as a look-up table (see Results and Discussion).

[b] Corresponds to the unfished stock at equilibrium (see Table 32 in Results and Discussion).

[c] Indirectly calculated from life history parameters (see Stock-recruitment relationship in Results and Discussion).

[d] A zero simulates a non-impacted stock; otherwise the conditional mortality due to entrainment is used.

amount of variability present in SPDM simulations (NUSCO 1990). Thus, the Monte Carlo sample size was set to 100 and the geometric mean of the replicates was computed. All stock projections are given in units of spawning biomass (lbs) because overfishing criteria often rely on measurements of biomass and biomass assessments tend to be more conservative than those based on fish numbers. Furthermore,

TABLE 5. Annual average cooling-water flow and percent of nominal maximum flow at MNPS Units 1 through 3 during the March-May winter flounder larval entrainment season from 1971 through 1994.

	Unit 1		Unit 2		Unit 3	
Nominal flow at 100% capacity:	29.18 $m^3 \cdot sec^{-1}$		37.62 $m^3 \cdot sec^{-1}$		61.91 $m^3 \cdot sec^{-1}$	
Fraction of total MNPS flow:	0.227		0.292		0.481	
Year[a]	March-May average flow in $m^3 \cdot sec^{-1}$	% of nominal maximum	March-May average flow in $m^3 \cdot sec^{-1}$	% of nominal maximum	March-May average flow in $m^3 \cdot sec^{-1}$	% of nominal maximum
1971	-	67.41	-	-	-	-
1972	-	99.64	-	-	-	-
1973	-	33.81	-	-	-	-
1974	-	83.50	-	-	-	-
1975	-	99.64	-	-	-	-
1976	25.39	90.80	29.16	80.83	-	-
1977	27.61	98.73	24.61	68.20	-	-
1978	17.48	62.53	18.91	52.41	-	-
1979	17.18	61.44	21.48	59.53	-	-
1980	27.60	98.70	31.75	88.01	-	-
1981	1.52	5.43	33.98	94.18	-	-
1982	27.60	98.70	32.33	89.61	-	-
1983	26.79	95.83	30.90	85.63	-	-
1984	13.88	49.61	35.83	99.31	-	-
1985	27.86	99.64	16.40	45.45	-	-
1986	27.21	93.25	36.89	98.07	49.82	80.48
1987	29.01	99.40	36.99	98.32	47.12	76.12
1988	28.84	98.81	32.83	87.27	55.58	89.78
1989	13.85	47.46	24.72	65.72	51.33	82.91
1990	27.55	94.39	33.28	88.48	48.71	78.68
1991	10.79	36.98	32.29	85.83	38.65	62.44
1992	25.11	86.06	28.50	75.75	51.10	82.55
1993	27.78	95.21	33.52	89.10	58.82	95.00
1994	4.33	14.84	31.39	83.44	58.20	94.01

[a] No records of cooling-water flow were available for 1971-75. Net electrical generation records were used to estimate flow, with values for 1972 and 1975 normalized to the value for 1985 (maximum of the Unit 1 time-series); 1971, 1973, and 1974 were adjusted accordingly.

larval entrainment effects result in long-term stock reductions which can be quite different depending on whether the stock is expressed as fish numbers or as biomass. Population reproductive capacity is more accurately reflected by biomass, which takes into account the size of individual females (egg production is a function of length or weight), as well as the number of spawners.

A complete simulation of MNPS impact consisted of three model runs, which provided a set of time-series generated under the same scenario, but with different combinations of F (plus IMP) and ENT. These model runs were designed to simulate the natural variability of the theoretical unfished stock (i.e., with no fishing or plant operational effects); the reduced stock biomass when subjected to fishing mortality (i.e., the baseline time-series without MNPS effects); and the expected biomass when all three types of anthropogenic mortality (F, IMP, and ENT) occurred. The first time-series with no fishing or plant effects was the reference series against which the potential for recruitment failure was evaluated when the largest reductions of stock biomass occurred during any of the other simulations. The second time-series represented the most likely trajectory of the exploited stock without MNPS operation. The third time-series was the expected stock trajectory when the conditional mortality rates corresponding to ENT and IMP were added to the fishing mortality simulated for the baseline. This last time-series was the basis for quantitatively assessing MNPS impact on the Niantic River winter flounder population.

Results and Discussion

Seawater temperature

Monthly mean seawater temperatures recorded at the MNPS intakes were variable during 1994. Water temperatures during January through March (seasonal mean of 2.55°C) were the coldest observed since 1982 (Tables 6 and 7). The mean temperature for April (6.62°C) was consistent with the long-term average, but May was the coolest (9.96°C) since 1978. The mean for June (15.37°C), however, was slightly above average and July had the highest mean (20.30°C) in 19 years. Seasonal mean temperatures in both summer (20.13°C) and fall (12.87°C) were the second warmest of the corresponding time-series (Table 7). These extremes in water temperature likely had an important effect on spawning success, larval growth and development, settlement, and growth of young. Overall, monthly mean temperatures are most variable from January through March (monthly coefficients of variation = 25-42%; Table 6), when winter flounder spawning and early larval development occurs. Temperatures were most stable (CV = 3-5%) from late spring through early fall, when collections of winter flounder were dominated by young and other immature fish. Overall, the annual mean temperature in 1994 was 11.60°C, which was slightly warmer than the long-term average of 11.51°C.

Adult winter flounder

Relative annual abundance

Because cold winter temperature produced heavy ice cover in the Niantic River the 1994, adult winter flounder survey did not begin until March 22, which was the latest start in 19 years (Table 8). As a result, only 4 weeks of sampling were completed, even after extending the survey to mid-April to increase the number of fish marked and recaptured. By this time, no fish remained in spawning condition, as illustrated by the percentage of females 26 cm and larger that were gravid (Fig. 7). Most spawning apparently occurred earlier in the season under the ice cover. Despite the cold water temperatures in 1994, the observed fractions of gravid females during each week of sampling fell within the range of the past 5 years. Apparently, spawning was not delayed to the extent that gravid females were common after mid-March.

As a result of 4 weeks of sampling, the total of 185 tows completed in 1994 were the fewest since 118 were made in 1982 (Table 9). As in the past several years, most adults were concentrated in only a few

TABLE 6. Monthly and annual mean seawater temperature (°C) from January 1976 through December 1994 as calculated from mean daily water temperatures recorded continuously at the intakes of MNPS Units 1 and 2.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual mean
1976	3.65	3.31	4.81	7.55	10.75	15.11	18.29	19.60	18.93	15.04	9.28	4.73	10.90
1977	0.67	0.36	2.85	5.66	10.72	14.92	19.08	20.33	19.41	15.58	12.18	6.72	10.84
1978	3.01	1.09	1.67	4.85	9.10	14.24	17.68	19.82	19.24	16.14	12.47	7.74	10.64
1979	4.53	1.48	3.35	5.93	10.50	15.57	18.84	20.91	20.05	15.99	12.41	8.60	11.57
1980	5.16	2.38	2.80	6.38	10.44	14.76	18.44	20.23	20.16	16.07	10.25	5.73	11.10
1981	1.06	2.63	3.36	6.40	10.19	15.48	19.51	20.86	19.94	14.75	11.07	6.29	11.01
1982	2.20	1.56	3.04	5.41	10.06	14.16	17.98	21.10	20.01	15.95	12.47	8.97	11.13
1983	5.58	3.74	4.55	7.07	10.50	15.05	19.10	19.17	20.57	17.37	12.57	7.90	11.98
1984	4.84	4.02	3.98	6.58	10.84	15.53	18.90	20.60	19.52	16.41	13.04	9.07	11.97
1985	4.36	2.36	4.17	7.02	10.95	14.99	18.98	21.24	20.44	17.46	13.14	7.55	11.98
1986	4.62	3.38	4.11	7.25	11.32	15.99	18.83	20.62	18.80	16.53	12.43	8.19	11.89
1987	5.28	3.27	4.53	7.51	11.26	15.91	19.19	20.47	19.30	15.70	11.10	7.16	11.78
1988	2.65	2.67	4.49	7.01	10.67	14.69	18.30	20.31	18.86	14.91	11.41	7.20	11.12
1989	4.49	3.24	3.67	6.21	10.59	15.25	18.95	20.31	19.92	15.83	12.25	4.87	11.34
1990	3.60	4.28	4.96	6.84	10.73	14.93	18.65	20.80	20.23	17.74	12.47	9.12	12.08
1991	5.72	4.76	5.61	8.11	12.26	16.61	19.53	20.48	19.99	17.11	12.00	8.17	12.59
1992	5.20	3.68	4.42	6.80	10.72	15.42	18.43	19.62	19.20	15.17	11.12	7.28	11.45
1993	5.09	3.10	3.12	6.09	11.37	15.64	18.96	20.88	19.88	15.35	11.73	8.47	11.69
1994	3.15	1.59	2.81	6.62	9.96	15.37	20.30	20.78	19.27	16.27	13.21	9.15	11.60
Overall mean	3.94	2.79	3.81	6.62	10.68	15.25	18.84	20.43	19.64	16.07	11.91	7.53	11.51
CV (%)	38	42	25	12	6	4	3	3	3	5	8	18	4

TABLE 7. Seasonal[a] mean seawater temperature (°C) for 1976 through 1994 as calculated from mean daily water temperatures determined by continuous recorders at the intakes of MNPS Units 1 and 2.

Year	Winter	Spring	Summer	Fall
1976	3.94	11.14	18.94	9.69
1977	1.32	10.72	19.61	11.49
1978	1.95	9.40	18.91	12.11
1979	3.17	10.67	19.93	12.33
1980	3.47	10.53	19.61	10.69
1981	2.34	10.69	20.11	10.70
1982	2.29	9.88	19.69	12.46
1983	4.65	10.87	19.61	12.61
1984	4.29	10.99	19.68	12.84
1985	3.67	10.98	20.22	12.85
1986	4.06	11.52	19.43	12.38
1987	4.40	11.56	19.66	11.32
1988	3.28	10.79	19.16	11.17
1989	3.82	10.68	19.72	10.97
1990	4.28	10.83	19.89	13.16
1991	5.38	12.32	20.00	12.48
1992	4.45	10.98	19.08	11.19
1993	3.79	11.03	19.91	11.85
1994	2.55	10.64	20.13	12.87
Overall mean	3.53	10.85	19.64	11.85
CV (%)	29	6	2	8

[a] Winter is January through March, spring is April through June, summer is July through September, and fall is October through December.

TABLE 8. Annual Niantic River winter flounder[a] population surveys during the spawning season from 1976 through 1994.

Year	Dates sampled	Number of weeks sampled
1976	March 1 - April 13	7
1977	March 7 - April 12	6
1978	March 6 - April 25	8
1979	March 12 - April 17	6
1980	March 17 - April 15	5
1981	March 2 - April 14	7
1982	February 22 - April 6	7
1983	February 21 - April 6	7
1984	February 14 - April 4	8
1985	February 27 - April 10	7
1986	February 24 - April 8	7
1987	March 9 - April 9	5
1988	March 1 - April 5	6
1989	February 21 - April 5	7[b]
1990	February 20 - April 4	7
1991	February 13 - March 20	6
1992	February 18 - March 31	7
1993	February 16 - April 7	8[c]
1994	March 22 - April 13	4

[a] Minimum size for marking was 15 cm during 1976-82 and 20 cm thereafter.

[b] Limited sampling during week 2 because of ice formation.

[c] Almost no sampling during week 3 and limited sampling during weeks 2 and 5 because of ice and weather conditions.

areas, including the upper river arm (stations 52-54) and in station 51 (Fig. 2). The median CPUE in 1994 of winter flounder larger than 15 cm was 4.5 (Table 9; Fig. 8). This value is more than twice that of the CPUE of 1.9 for 1993, but nevertheless remains the second smallest CPUE on record. Winter flounder taken during the 1993 survey were, on average, larger than those collected during any previous surveys (NUSCO 1994a). The length-frequency distribution in 1994 was similar to that found in 1991, with the proportions of smaller fish increasing from that observed during the previous 3 years (Fig. 9). A comparison of the annual standardized catch of females from 1991 through 1994 showed a peak in catch of females at 32-34 cm (Fig. 10). Proportionately fewer larger females were found during 1994 in comparison to 1992 and 1993, except for the very largest (45^{+} cm) females, which were the most of any survey to date.

Absolute abundance estimates

Absolute abundance estimates of fish larger than 20 cm (*N*) were determined using mark and recapture data and the Jolly (1965) model. Estimates of survival (Φ), recruitment (*B*), and sampling intensity (*p*) were

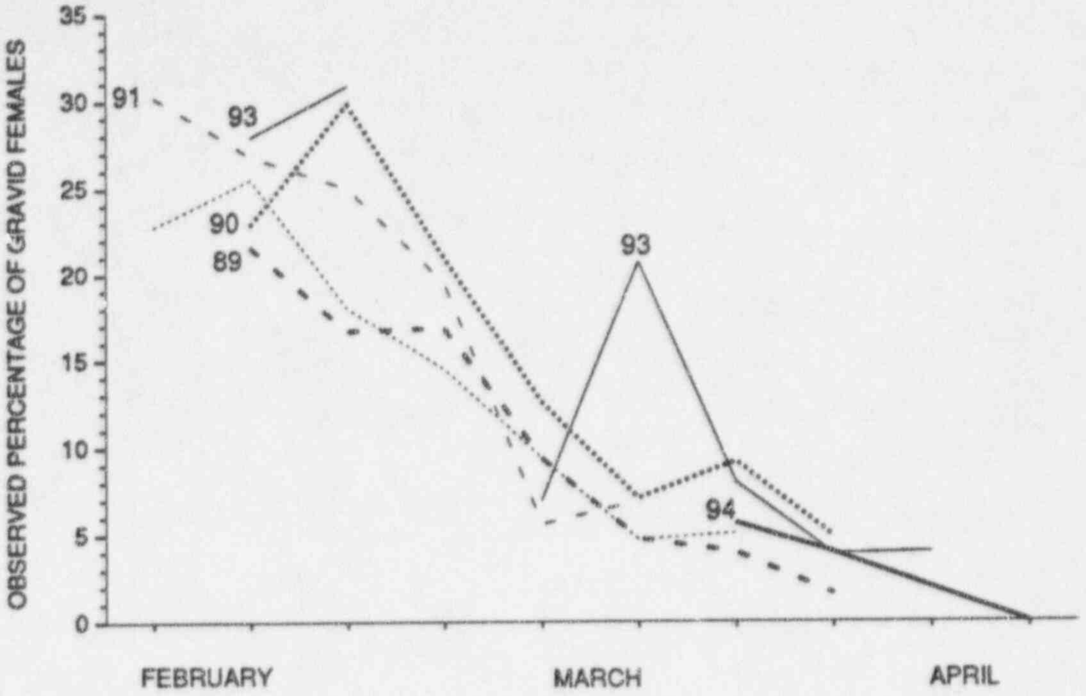


Fig. 7. Weekly percentage of Niantic River female winter flounder larger than 26 cm that were gravid during the 1989 through 1994 adult population abundance surveys. Data from weeks in 1993 during which few or no tows were taken were not included.

TABLE 9. Annual 9.1-m otter trawl adjusted median CPUE[a] of winter flounder larger than 15 cm[b] taken throughout the Niantic River during the 1976 through 1994 adult population abundance surveys.

Survey year	Weeks sampled	Tows acceptable for CPUE[c]	Adjusted number of tows used[d]	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness[e]
1976	7	143	231	37.0	34.2 - 39.6	3.01
1977	6	184	228	23.1	20.4 - 26.4	1.95
1978	6	137	159	21.0	18.8 - 27.0	1.83
1979	5	122	145	33.6	25.5 - 39.5	1.52
1980	5	112	145	36.0	30.0 - 43.2	1.68
1981	7	182	231	51.6	45.6 - 56.4	3.50
1982	5	118	150	42.6	42.6 - 46.0	1.14
1983	7	232	238	30.2	26.2 - 31.8	0.85
1984	7	245	287	16.8	15.8 - 18.0	1.17
1985	7	267	280	14.8	14.2 - 15.4	1.33
1986	7	310	336	10.2	9.7 - 11.1	1.47
1987	5	233	270	14.8	14.1 - 16.2	1.46
1988	6	293	312	16.8	15.7 - 17.5	0.50
1989	6	277	318	12.2	11.1 - 13.3	1.08
1990	7	320	343	9.6	8.7 - 10.3	3.04
1991	6	302	330	12.3	11.1 - 13.4	2.62
1992	7	380	406	6.2	5.6 - 6.6	1.29
1993	7[f]	288	392	1.9	1.7 - 2.6	1.92
1994	4	185	212	4.5	3.8 - 5.0	2.68

[a] Catch per standardized tow (see Materials and Methods).
[b] Mostly age-2 and older fish.
[c] Only tows of standard time or distance were considered.
[d] Effort equalized among weeks.
[e] Zero for symmetrically distributed data.
[f] Because of low effort, data from the third week of sampling not used for the computation of CPUE.

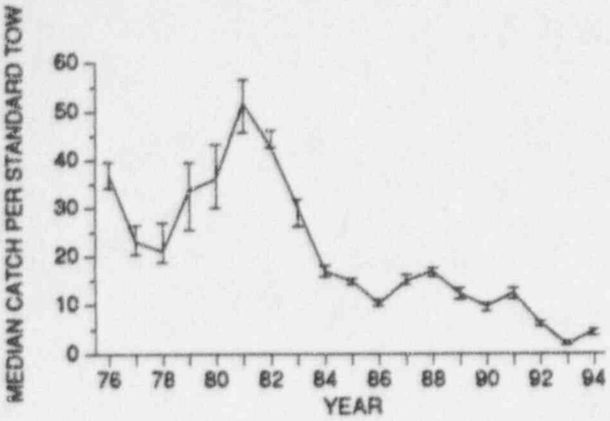


Fig. 8. Annual median CPUE and 95% confidence interval of Niantic River winter flounder larger than 15 cm from 1976 through 1994.

also generated by this model. Because of the continued depression in the Niantic River winter flounder population and limited sampling this year, only 1,033 fish 20 cm and larger were marked with a freeze brand, which was a slight increase from 1993 (Table 10). Seventy-six previously marked fish were recaptured in 1994, which was the fewest of this time-series. Nearly equal numbers of fish that were marked in 1992 (n = 25) and 1993 (27) were observed.

Addition of recapture data from 1994 resulted in an increase of about 2,500 fish for the 1992 estimate reported in NUSCO (1994a), from 12,178 to 14,632 (Table 11). The initial abundance estimate for 1993 was 11,779 winter flounder; this value will be subject to change as additional marked fish are recaptured in future surveys. The standard errors of N given in Table 11 are correlated with N because of the particular form of Jolly's variance formula. Therefore, the 95% confidence intervals computed are generally considered unreliable as a measure of sampling error, except at very high sampling intensities (Manly 1971; Roff 1973; Pollock et al. 1990).

Sampling intensity (p), or the probability that a fish will be captured, was a relatively high estimate of 0.160 in 1992, but decreased to 0.083 in 1993. Sampling intensities of about 0.10 are recommended to obtain reliable and precise estimates of population size and survival rates with the Jolly model (Bishop and Sheppard 1973; Nichols et al. 1981). Hightower and Gilbert (1984) found that low sampling effort may give acceptable estimates if population size is relatively large and the number of marked animals is also relatively high. However, Gilbert (1973) and Carothers (1973) reported that N was underestimated and had low accuracy when sampling intensities were low (5-9%), regardless of population size or number of fish marked. Loss of marks because tags were not observed, or from mortality of fish handled, also requires increased sampling effort. Other sampling errors, model assumptions, and biases inherent in the Jolly model that could have affected these estimates were discussed in NUSCO (1989) and Pollock et al. (1990). Nevertheless, even though the Jolly estimates are subject to considerable error, CPUE and Jolly abundance estimates were significantly (Pearson correlation coefficient = 0.89; $p < 0.001$) correlated (Fig. 11). Thus, based on a median CPUE of 4.5, absolute abundance of winter flounder in 1994 would have been approximately 15,000 fish. By extrapolation, abundance in 1981 could have exceeded 200,000 winter flounder.

Estimates of survival (Φ) have varied considerably from year to year (0.180-0.853; Table 11). Estimated recruitment (B) was particularly low in both 1992 (3,473) and 1993 (4,127). These two estimated population parameters are considered to be less reliable than those of abundance when using the Jolly model (Bishop and Sheppard 1973; Arnason and Mills 1981; Hightower and Gilbert 1984). As for other parameter estimates based on only 1 year of recapture information, those for Φ and B may change considerably with the addition of data from the next annual survey. The low estimates of B for 1992 and 1993, however, appeared to accurately reflect weak recruitment of winter flounder noted in recent years.

Because of a reasonable correspondence between median trawl CPUE and Jolly abundance estimates, the annual standardized catches of all fish larger than 20 cm for 1984-93 were compared to total abundance estimates from the Jolly model. As in previous reports, the relative numbers of females and eggs produced each year were assumed to represent, conservatively, about 3.5% of the absolute values (range of 2.7 - 7.0%) and a multiplier of 28.571 (the ratio of 100 to 3.5) was used to scale abundance indices to absolute numbers of winter flounder spawning in the Niantic River that are given below. This adjustment also assumed that the ratios of annual standardized catch to absolute abundance for 1977 through 1983 would have been similar to those for 1984-93, had estimates of absolute abundance been available for the earlier period.

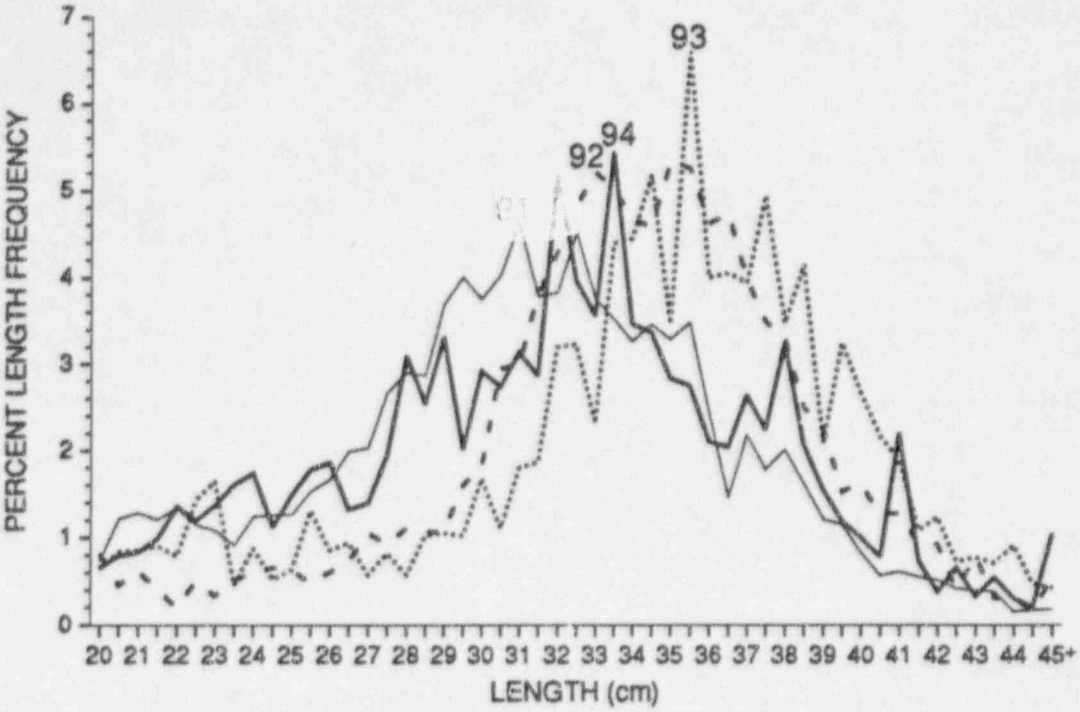


Fig. 9. Comparison of percent length-frequency distributions of all winter flounder 20 cm and larger taken in the Niantic River during the spawning season from 1991 through 1994.

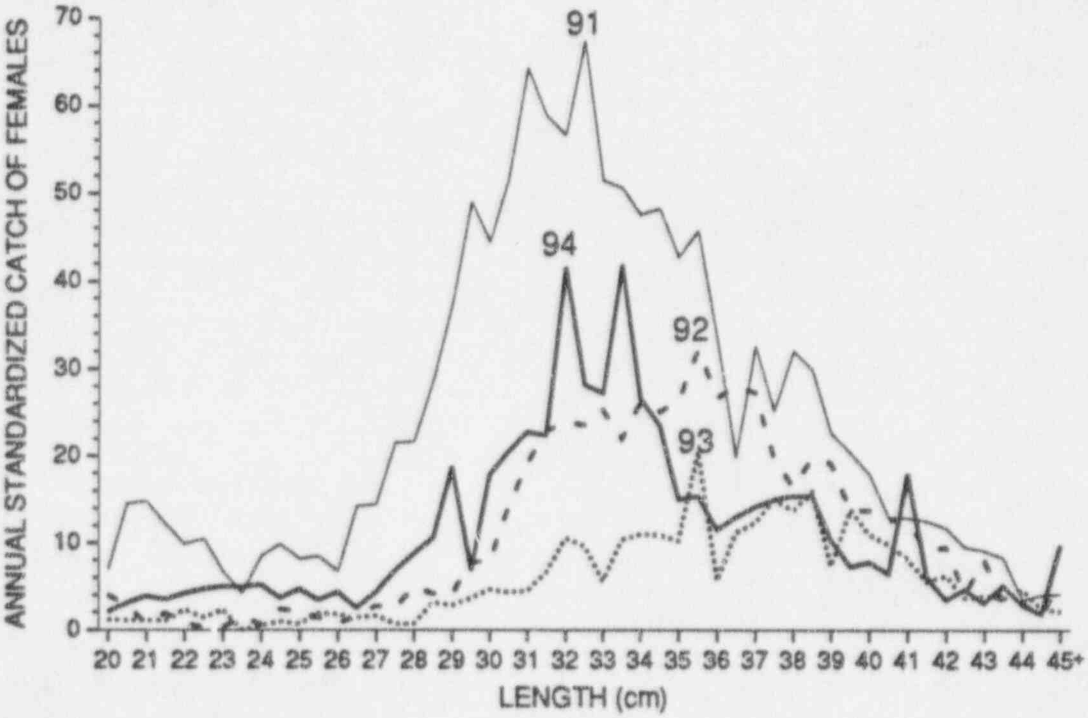


Fig. 10. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season from 1991 through 1994.

TABLE 10. Mark and recapture data from 1983 through 1994 used for estimating abundance of winter flounder larger than 20 cm in the Niantic River during the spawning season.

Survey year	Total number observed	Total not previously marked	Number marked and released	Total number recaptured	Number of fish marked in a given year recaptured during subsequent annual surveys:										
					83	84	85	86	87	88	89	90	91	92	93
1983	5,615	5,615	5,615	0	-										
1984	4,103	3,973	4,083	130	130										
1985	3,491	3,350	3,407	141	47	94									
1986	3,031	2,887	3,010	144	23	45	76								
1987	2,578	2,463	2,573	115	2	13	27	73							
1988	4,333	4,106	4,309	227	7	22	31	63	104						
1989	2,821	2,589	2,752	232	2	11	9	33	32	145					
1990	2,297	2,135	2,275	162	1	7	4	15	14	38	83				
1991	4,333	4,067	4,324	266	1	5	4	12	27	33	54	130			
1992	2,346	2,119	2,336	227	0	0	1	2	3	21	20	53	127		
1993	984	830	972	154	0	0	0	1	0	4	4	15	21	109	
1994	1,035	959	1,033	76	0	0	0	1	0	0	4	5	14	25	27

TABLE 11. Estimated abundance[a] of winter flounder larger than 20 cm taken during the spawning season in the Niantic River from 1984 through 1993 as determined by the Jolly (1965) mark and recapture model.

Year	Abundance estimate (*N*)	Standard error of *N*	95% CI for *N*	Probability of survival (Φ)	Standard error of Φ	95% CI for Φ
1983				0.328	0.040	0.251 - 0.405
1984	57,706	8,370	41,300 - 74,112	0.558	0.065	0.430 - 0.686
1985	79,607	10,851	58,338 - 100,876	0.360	0.041	0.279 - 0.440
1986	49,057	6,194	36,917 - 61,197	0.654	0.068	0.522 - 0.786
1987	75,909	9,783	56,733 - 95,084	0.598	0.063	0.476 - 0.721
1988	66,946	7,278	52,681 - 81,211	0.453	0.048	0.360 - 0.546
1989	41,777	4,744	32,479 - 51,075	0.394	0.042	0.312 - 0.476
1990	33,270	3,832	25,759 - 40,781	0.853	0.100	0.657- 1.048
1991	62,032	7,516	47,301 - 76,763	0.180	0.024	0.133 - 0.227
1992	14,632	1,966	10,779 - 18,486	0.525	0.118	0.294 - 0.755
1993	11,799	2,706	6,495 - 17,104			
Mean	49,274	2,191	44,979 - 53,568	0.490	0.016	0.459 - 0.521

Year	Sampling intensity (*p*)	Stcndard error of *p*	95% CI for *p*	Annual recruitment (*B*)	Standard error of *B*	95% CI for *B*
1984	0.071	0.0103	0.050 - 0.091	47,428	9,083	29,626 - 65,231
1985	0.044	0.0060	0.032 - 0.055	20,454	5,200	10,262 - 30,647
1986	0.061	0.0078	0.046 - 0.077	43,850	8,499	27,191 - 60,509
1987	0.034	0.0044	0.025 - 0.042	21,555	6,404	9,002 - 34,108
1988	0.065	0.0071	0.051 - 0.078	11,471	3,669	4,280 - 18,661
1989	0.067	0.0077	0.052 - 0.082	16,832	3,108	10,740 - 22,943
1990	0.069	0.0080	0.053 - 0.084	33,688	5,595	22,721 - 44,654
1991	0.070	0.0085	0.053 - 0.086	3,473	1,319	888 - 6,059
1992	0.160	0.0216	0.117 - 0.202	4,127	1,423	1,335 - 6,917
1993	0.083	0.0191	0.046 - 0.120			
Mean	0.072	0.0036	0.065 - 0.079	22,542	1,036	20,511 - 24,573

[a] Estimates may vary from those reported in NUSCO (1994a) because of mark and recapture data added from the 1994 adult winter flounder population survey.

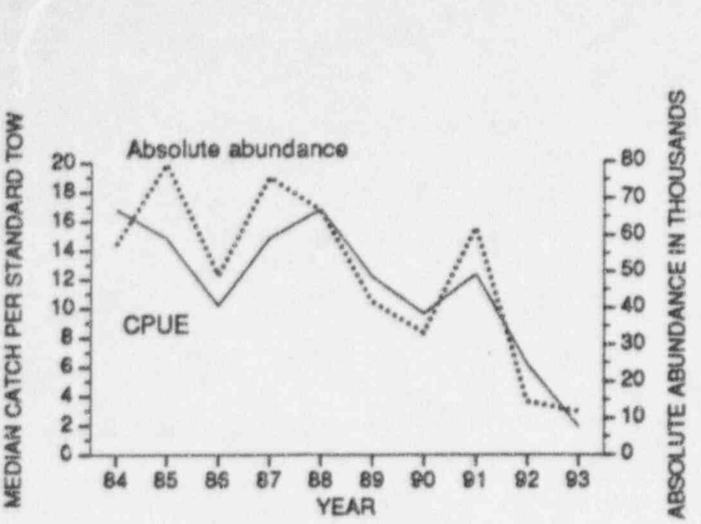


Fig. 11. Comparison between the estimates of absolute abundance in thousands of winter flounder larger than 20 cm in the Niantic River during the spawning season (dashed line) and the corresponding median CPUE (solid line) from 1984 through 1993.

Spawning stock size and egg production

The size of the Niantic River winter flounder female spawning stock is used in various assessments of MNPS impact. The annual standardized catch of female spawners (an index of spawning stock size) and the production of eggs were determined from available data on sex ratios, sexual maturity, and fish length-frequencies. The sex ratio of winter flounder larger than 20 cm during the 1994 spawning season in the Niantic River was 1.70 females for each male (Table 12), the largest ratio found since 1980. The 18-year average was 1.35. Ratios of 1.50 to 2.33 in favor of females have been reported by Saila (1962a, 1962b) and by Howe and Coates (1975) for other winterflounder populations in southern New England. Witherell and Burnett (1993) also reported greater proportions of female winter flounder in Massachusetts waters, particularly in older age-classes. They believed that males likely have a higher natural mortality rate, based on evidence of earlier ages of senescence reported for males by Burton and Idler (1984).

The rate of spawning was determined by observing weekly changes in the percentage of gravid females larger than 26 cm, the size at which about half of all observed females were mature (NUSCO 1988a). This is comparable to L_{50} estimates of size-at-maturity of 28.3 and 27.6 cm reported for Massachusetts waters by Witherell and Burnett (1993) and O'Brien et al. (1993), respectively. In recent years, spawning in the Niantic River was mostly completed by late March or early April as relatively few gravid females were found afterwards (Fig. 7). During most years, ice in the upper river prevented the start of field work in January or early February, so approximately two-thirds of the females examined during late February and early March had spawned before sampling began. Spawning was likely correlated with water temperature as in relatively cold years (e.g., 1977 and 1978) proportionately fewer females spawned during the earlier portion of the survey, compared to warmer years (e.g., 1989 and 1992) when more fish were spent at the beginning of sampling. However, this could not be ascertained in 1994, which was a very cold year, as ice cover prevented sampling until late March. By then, fractions of egg-bearing females had decreased to about 5% or less and were not dissimilar to proportions observed in 1989-93.

During each year, the proportion of females considered to be mature in each 0.5-cm length increment was used with the annual standardized catch of females to obtain relative annual abundance indices for female spawners. Mature females comprised approximately one-third to one-half of each yearly total, with relative numbers of female spawners ranging from a low of 274 in 1993 to 2,752 in 1982 (Table 13). Varying sex ratios and differences in percent maturity due to changes in length-frequency distributions somewhat

TABLE 12. Female to male sex ratios of winter flounder taken during the spawning season in the Niantic River from 1977 through 1994.

Year	All fish captured	Measured fish > 20 cm
1977	1.03	1.26
1978	2.23	1.95
1979	1.37	1.21
1980	2.66	2.03
1981	1.42	1.61
1982	1.16	1.70
1983	1.52	1.52
1984	1.07	1.07
1985	1.37	1.37
1986	0.92	0.92
1987	0.78	0.78
1988	1.50	1.50
1989	1.32	1.32
1990	1.24	1.24
1991	1.22	1.22
1992	1.26	1.26
1993	1.47	1.47
1994	1.70	1.70
Geometric mean	1.35	1.36

TABLE 13. Relative and absolute annual standardized catch of female winter flounder spawners and corresponding egg production in the Niantic River from 1977 through 1994.

Survey year	Relative index of spawning females[a]	% mature females[b]	Average fecundity[c]	Relative index of total egg production[d]	Total female stock size[e]	Total egg production (X 10^9)[e]
1977	884	36	446,336	394.6	25,260	11.274
1978	1,412	51	508,096	717.5	40,349	20.501
1979	1,120	37	478,108	535.3	31,989	15.294
1980	903	34	469,976	424.3	25,793	12.122
1981	2,669	44	518,275	1,383.1	76,248	39.517
1982	2,752	49	580,227	1,596.8	78,629	45.622
1983	1,869	46	578,845	1,082.0	53,406	30.914
1984	871	40	575,822	501.6	24,886	14.330
1985	928	43	609,215	565.2	26,510	16.150
1986	655	42	667,065	436.7	18,704	12.477
1987	852	39	624,085	531.6	24,339	15.190
1988	1,279	53	677,910	866.9	36,539	24.770
1989	984	52	728,042	716.2	28,108	20.464
1990	579	42	639,541	370.4	16,546	10.582
1991	1,061	47	603,132	639.6	30,300	18.275
1992	534	52	732,317	391.1	15,260	11.175
1993	274	54	816,885	223.6	7,821	6.389
1994	508	55	650,130	330.2	14,513	9.435

[a] Based on proportion of the relative annual standardized catches of winter flounder that were mature females.

[b] As a proportion of all winter flounder 20 cm or larger.

[c] Total egg production divided by the number of spawning females.

[d] A relative index for year-to-year comparisons and not an absolute estimate of production.

[e] Calculated on the assumption that the relative annual standardized catches were approximately 3.5% of absolute values.

affected average fecundity, which was low during the late 1970s when smaller fish were more abundant, but increased during recent years because of increasing proportions of older and larger fish. The relative index of total egg production reflected female stock abundance and length distribution and was greatest from 1981 through 1983 because of peak population abundance and moderate average fecundity. The average fecundity estimate for 1994 decreased from 1992 and 1993 as comparatively more smaller fish were present (Fig. 10).

Absolute estimates of spawning females and associated egg production were generated by multiplying corresponding relative numbers by 28.571 (see Absolute abundance estimates, above). Female stock size was between approximately 7,821 and 78,629 fish, while estimates of annual egg production ranged from about 6.4 to 45.6 billion (Table 13). The total number of female spawners was used as an estimate of parental stock size for the SRR, which will be discussed below. Egg production was greatest in the early 1980s, but estimates were also relatively high in 1988 and 1989 as proportionally older and larger females dominated a moderately-sized reproductive stock. Egg production decreased to about 10.6 billion in 1990 because of a decline in female abundance and in average size, increased to 18.3 billion in 1991 as the number of spawners increased, decreased once again to 11.2 billion in 1992, and finally fell to a series low of 6.4 billion in 1993. Egg production then increased by 48% to 9.4 billion in 1994, reflecting the increase in spawner abundance.

Comparatively little is known about the egg stage of winter flounder. Buckley et al. (1991) noted that female size and time of spawning affected various reproductive parameters, including egg size, fecundity, and viability. Embryos produced earlier in the season appeared to have a survival advantage, particularly over those from smaller fish late in the season. Egg deposition apparently takes place on gravel bars, algal mats, eelgrass beds, and near freshwater springs in Rhode Island salt ponds (Crawford 1990). Viable hatch is greatest at 3°C in salinities of 15 to 35‰ and decreases with increasing temperature (Rogers 1976). Based on estimated egg production and abundance of Stage 1 larvae, egg mortality may be considerable. DeBlois and Leggett (1991) found that the amphipod *Calliopius laeviusculus* preyed heavily upon demersal capelin (*Mallotus villosus*) eggs, removing up to 39% of the production. They suggested that invertebrate predation on demersal fish eggs may be an important regulatory mechanism for population size in marine fishes having demersal eggs. Morrison et al. (1991) reported high mortality of demersal Atlantic herring (*Clupea harengus*) eggs in the Firth of Clyde, Scotland because of heavy deposition of organic matter resulting from a bloom of the diatom *Skeletonema costatum*. The decomposing material caused a depletion of oxygen and egg death due to anoxia. *Skeletonema costatum* was one of the most abundant of the phytoplankton collected at MNPS during entrainment sampling from 1977 through 1980 (NUSCO 198'). However, highest densities occurred in summer, after the winter flounder spawning season.

Larval winter flounder

Abundance and distribution

The α parameter of the Gompertz function (Eq. 2) was used as an index for temporal (year to year) and spatial (Niantic River and Bay) abundances of winter flounder larvae. Based on the 95% confidence interval around the α parameter estimate, larval abundance during 1994 in the river (stations A, B, and C combined) and the bay (stations EN and NB combined) was greater than in 1993, but within the range for the 12-year series (Table 14). In general, annual abundances in the bay varied less than in the river. In 1985, 1988, and 1989, larval abundance in the river was much greater than in the bay. No consistent relationship was found between the annual abundances between the two areas (Spearman's rank-order correlation coefficient r = 0.441; p = 0.152) during the 12-year period. This lack of a relationship has two possible causes. First, if many of the larvae in the bay came from the river, then highly variable annual larval mortality rates occurred prior to the period when larvae were flushed from the river to the bay. Secondly, the Niantic River may not be the only source of larvae entering the bay and this possibility was discussed in detail in NUSCO (1992a, 1992b, 1993, 1994a) and will be addressed again later in this section. Larval abundance in the bay appeared to reflect regional-wide trends as annual abundance (α parameter) at EN since 1976 was highly correlated (Spearman's rank-order correlation coefficient r = 0.651; p = 0.009) with annual abundance indices in Mount Hope Bay, MA and RI (Marine Research, Inc. 1992; M. Scherer, Marine Research, Inc, Falmouth, MA., pers comm.). However, no relationship was found between the abundances in the Niantic River (1984-94) and Mount Hope Bay (Spearman's rank-order correlation coefficient r = 0.018; p = 0.958).

Annual spatial abundances of the first four larval developmental stages were based on cumulative weekly geometric means (Figs. 12 and 13). The abundance distribution of Stage 5 was not examined because so few were collected. Cumulative density data (the running sum of the weekly geometric means) was used to compare abundances as a surrogate for the α parameter from the Gompertz function (Eq. 2) because in some instances this function could not be fitted. This usually occurred when a developmental stage was rarely collected at a station (e.g., Stage 1 at stations EN and NB or Stage 4 at station A). Cumulative weekly geometric means and the corresponding α parameters were found to be highly correlated

TABLE 14. Larval winter flounder abundances and 95% confidence intervals for the Niantic River and Bay as estimated by the α parameter from the Gompertz function.

Year	Niantic River	Niantic Bay
1983	1,863 (1,798 - 1,929)	3,730 (3,670 - 3,791)
1984	5,018 (4,884 - 5,152)	2,200 (2,088 - 2,311)
1985	11,924 (11,773 - 12,075)	1,801 (1,717 - 1,886)
1986	1,798 (1,726 - 1,871)	1,035 (979 - 1,091)
1987	5,381 (5,172 - 5,589)	1,301 (1,240 - 1,363)
1988	24,004 (23,644 - 24,364)	1,784 (1,708 - 1,861)
1989	18,586 (17,965 - 19,207)	1,751 (1,696 - 1,806)
1990	5,544 (5,378 - 5,709)	1,532 (1,474 - 1,589)
1991	4,083 (3,973 - 4,193)	1,444 (1,388 -1,500)
1992	10,646 (10,184 - 11,108)	4,415 (4,214 - 4,617)
1993	1,513 (1,470 - 1,557)	459 (391 - 526)
1994	5,685 (5,564 - 5,805)	2,378 (2,269 - 2,486)

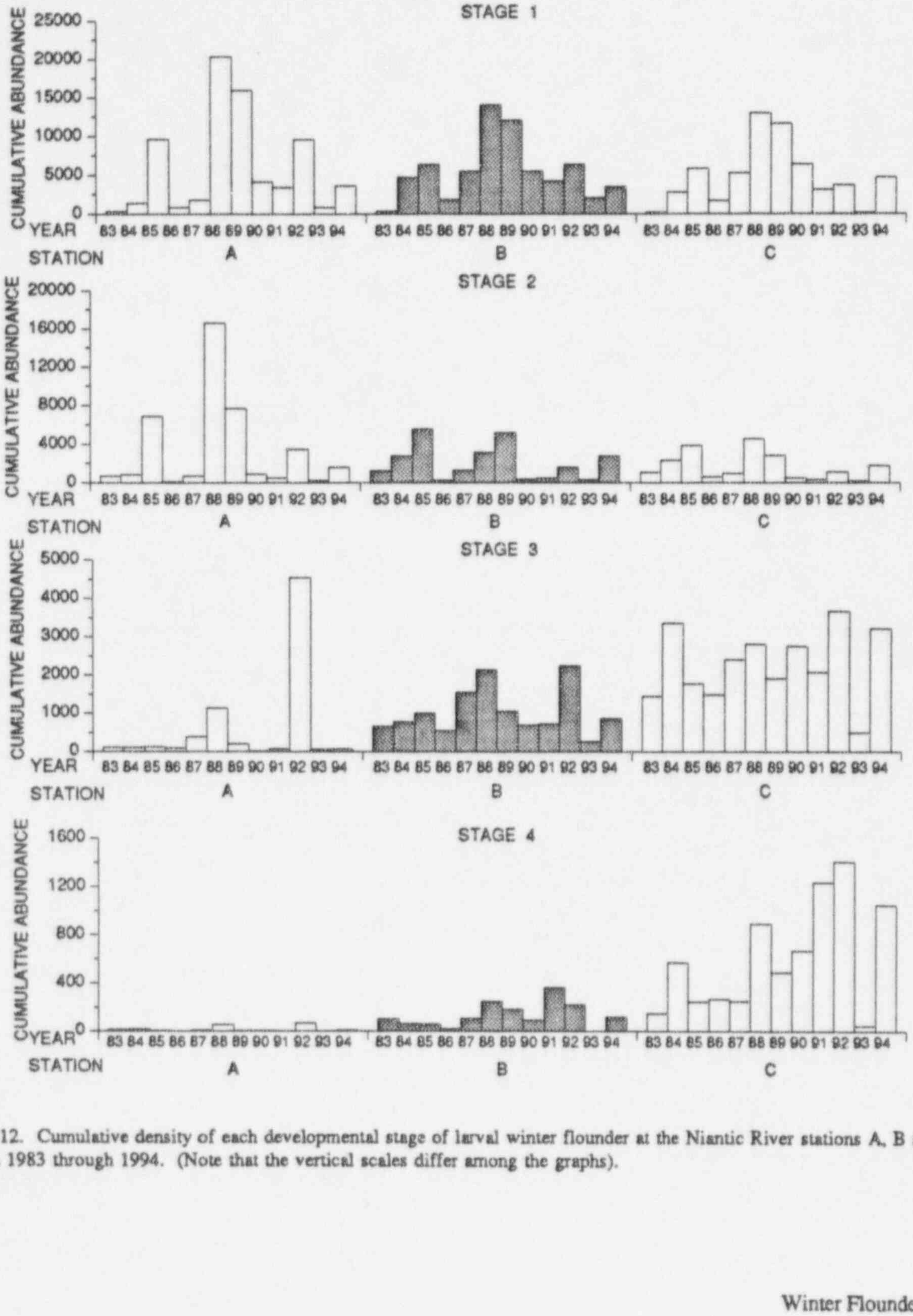


Fig. 12. Cumulative density of each developmental stage of larval winter flounder at the Niantic River stations A, B and C from 1983 through 1994. (Note that the vertical scales differ among the graphs).

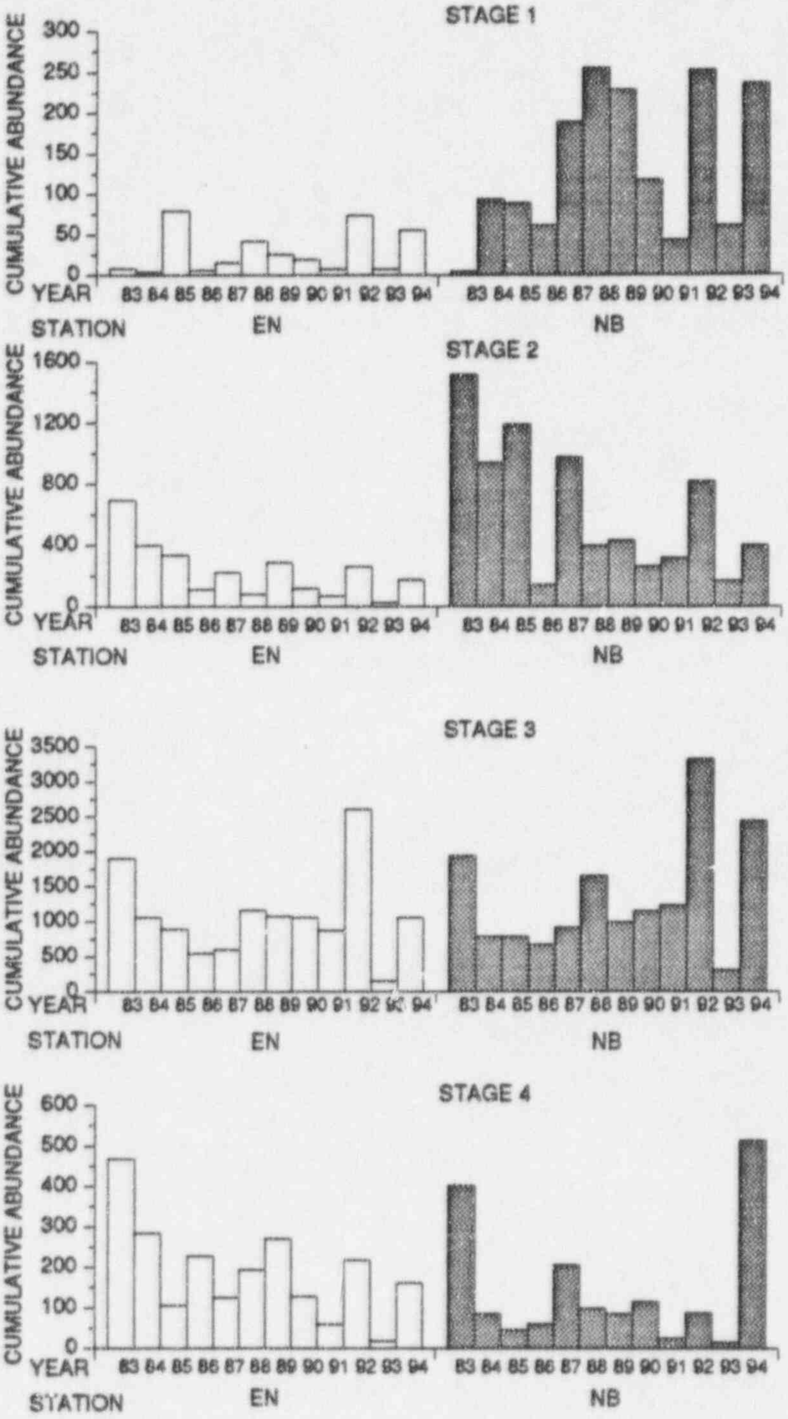


Fig. 13. Cumulative density of each developmental stage of larval winter flounder at the Niantic Bay stations EN and NB from 1983 through 1994. (Note that the vertical scales differ among the graphs).

(Spearman's rank-order correlation coefficient $r = 0.999$; $p < 0.001$) in a previous study (NUSCO 1989), indicating that cumulative weekly geometric means could be used as an alternative index of larval abundance.

Stage 1 abundance during 1994 in the river was one of the lowest during the 12-year period of sampling at all three stations, but showed an increase compared to 1993 (Fig. 12). A comparison of annual Stage 1 abundance among years showed a similar relative ranking at the three stations, with 1988 and 1989 ranked the highest and 1983, 1986, and 1993 the lowest. Except for a slightly greater abundance at station A in some years, annual abundances at the three river stations have been similar. This indicated a somewhat homogeneous distribution of Stage 1 larvae throughout the river. Because winter flounder eggs are demersal and adhesive and the duration of Stage 1 is short (about 10 days), the homogenous distribution suggested that spawning was not restricted to a specific area of the river or, conversely, that the river is well-mixed. Low abundance in 1983 was attributed, in part, to undersampling because of net extrusion (NUSCO 1987). However, this was rectified in 1984 when a net with smaller mesh (202 µm) was used during the early portion of the larval season. Abundance of Stage 1 larvae at the two Niantic Bay stations was low in comparison to the river, indicating that little, if any, spawning occurred in the bay, even though 1994 was among the highest (Fig. 13). Abundances at station NB were consistently greater than at EN, possibly because NB was located closer to the river mouth, the likely source of Stage 1 larvae, or because undersampling occurred at EN as a result of extrusion through the 333-µm mesh net used there. At station NB, ranks of annual abundance indices were similar to those of the river stations and this suggested that most Stage 1 larvae collected in the bay probably originated from the Niantic River. Significant ($p \leq 0.05$) positive correlations were found among Stage 1 annual abundances at all stations, except between stations EN and NB (Table 15).

Stage 2 abundance in 1994 at the three river stations was also among the lowest in the 12-year time-series. In general, annual ranks of Stage 2 abundance at the three river stations were similar to those of Stage 1 (Fig. 12). This implied a similar annual rate of larval loss (mortality and flushing) during larval development from Stage 1 to 2. Annual abundances at stations B and C were almost identical. Stage 2 larvae occurred predominantly in the river, but were more prevalent in the bay compared to Stage 1 (Fig. 13). As with Stage 1, Stage 2 abundance at station NB was greater than at EN, suggesting that either station NB was closer to the source of newly hatched larvae or that smaller Stage 2 larvae were extruded through the 333-µm mesh net used at EN. There was a significant ($p \leq 0.05$) positive correlation among all river stations and between stations EN and NB (Table 15).

Generally the later developmental stages (3 and 4) of winter flounder larvae were not homogeneously distributed within the Niantic River. The abundance decline at the upper river stations (A and B) as development progressed likely represented a gradual flushing to the lower portion of the river and into the bay. Stage 3 larvae were usually most abundant at stations B and C with their abundance at the two bay stations (EN and NB) increasing to levels similar to or greater than at stations A and B (Figs. 12 and 13). In comparison to other years, Stage 3 larvae were relatively numerous at stations C and NB in 1994, but were about average in abundance at stations B and EN. Annual abundance patterns were similar at the two bay stations and were significantly ($p \leq 0.05$) correlated (Table 15). Similar to Stage 3, Stage 4 larvae were generally more abundant at station C and the two Niantic Bay stations in comparison to stations A and B. In 1994, Stage 4 larvae were relatively numerous at station C and NB and were about average in abundance at the remaining stations. The high abundance at these two stations this year was indicative of good recruitment to the demersal young-of-the-year developmental stage, which is discussed below.

Annual abundance of newly hatched winter flounder larvae should relate to adult reproductive capacity (egg production) and the fraction of eggs that hatch. To examine this relationship, the annual egg production estimates (Table 13) were compared to the annual abundance of Stage 1 larvae. The index of Stage 1 larval abundance was the α parameter from the Gompertz function (Eq. 2) for the Niantic River (stations A, B, and C combined). A functional regression indicated a strong positive relationship ($r = 0.788$; $p = 0.004$) between egg production and Stage 1 abundance (Fig. 14). The abundance of newly hatched larvae was directly related to the adult reproductive capacity under the assumption that egg hatchability was similar among years. The consistency of this relationship implied good precision in the sampling of Stage 1 larvae and, additionally, that egg production estimates were a reasonable index of annual reproductive capacity.

TABLE 15. Matrix of Spearman's rank-order correlations among stations for the annual cumulative abundance of each developmental stage of larval winter flounder from 1983 through 1994.

Stage	Station	B	C	EN	NB
1	A	0.9301[a] 0.0001 **	0.9091 0.0001 **	0.7496 0.0050 **	0.7413 0.0058 **
	B		0.8811 0.0002 **	0.6235 0.0303*	0.7063 0.0102 *
	C			0.6445 0.0303 *	0.7133 0.0092 **
	EN				0.5786 0.0622 NS
2	A	0.8601 0.0003 **	0.8392 0.0006 **	0.3077 0.3306 NS	0.3497 0.2652 NS
	B		0.9231 0.0001 **	0.5315 0.0754 NS	0.6084 0.0358 *
	C			0.5245 0.0800 NS	0.5175 0.0849 NS
	EN				0.8182 0.0021 **
3	A	0.8532 0.0004 **	0.3846 0.2170 NS	0.4825 0.1121 NS	0.2727 0.3911 NS
	B		0.6573 0.0202 *	0.5035 0.0952 NS	0.3846 0.2170 NS
	C			0.4685 0.1245 NS	0.7622 0.0040 **
	EN				0.7063 0.0102 *
4	A	0.3803 0.2227 NS	0.3240 0.3043 NS	0.5071 0.0925 NS	0.5679 0.0541 NS
	B		0.7413 0.0058 **	0.0629 0.8459 NS	0.2907 0.3593 NS
	C			0.0420 0.8970 NS	0.1856 0.5635 NS
	EN				0.4168 0.1777 NS

[a] The two statistics shown in each correlation matrix element are:
correlation coefficient (r), and
probability of a larger r (NS - not significant [$p > 0.05$], * - significant at $p \le 0.05$, ** - significant at $p \le 0.01$).

Dates of peak abundance, estimated from the inflection point p of the Gompertz function (Eq. 2), were used to compare the times of occurrence in the river (station A, B, and C combined) and bay (stations EN and NB combined) for each developmental stage (Table 16). Dates of peak abundance of Stage 1 larvae could not be estimated for bay stations because this larval stage was rarely collected there and, similarly, for

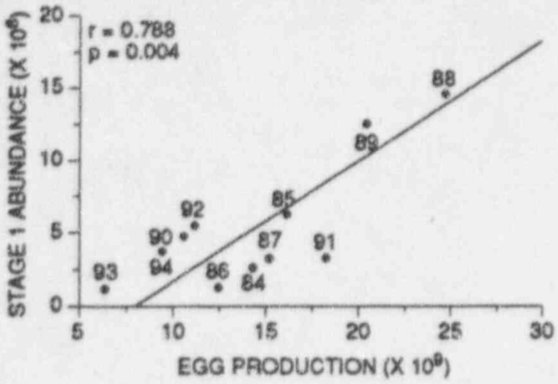


Fig. 14. The relationship (functional regression) between annual Stage 1 abundance in the Niantic River, estimated from the α parameter of the Gompertz function, and egg production from 1984 through 1994.

Stage 4 in the river during 1993 because of low abundance. In 1994, the dates of peak abundance of Stage 1 larvae in the river were much later than during the previous 11-year period. Based on water temperatures of 2 to 3°C during February (Table 6) and egg incubation times reported by Buckley (1982), peak spawning generally occurred in early to mid-February but may have occurred later in 1994. Buckley et al. (1990) reported that egg developmental time was inversely related to water temperature during oocyte maturation and egg incubation. Colder 1994 winter water temperatures could have delayed hatching. Comparison between the 1983-94 February water temperatures (Table 6) and the annual dates of Stage 1 peak abundance in the river showed a significant negative relationship (Spearman's rank-order correlation coefficient r = -0.599; p = 0.040). Later peaks

TABLE 16. Estimated dates of peak abundance of larval winter flounder for each development stage in the Niantic River and Bay and the number of days ocorresponding to the 95% confidence interval.

Year	Stage 1	Stage 2	Stage 3	Stage 4
		Niantic River		
1983	March 5 (3)	March 15 (2)	April 18 (1)	May 2 (4)
1984	March 7 (5)	March 9 (5)	April 24 (5)	May 19 (10)
1985	March 11 (1)	March 16 (2)	April 25 (3)	May16 (7)
1986	February 26 (1)	March 11 (5)	April 20 (3)	May 12 (10)
1987	March 10 (2)	March 17 (3)	April 20 (2)	May 9 (4)
1988	February 29 (1)	March 9 (1)	April 7 (4)	May 1 (5)
1989	March 8 (6)	March 12 (5)	April 14 (3)	May 11 (9)
1990	February 17 (3)	February 18 (5)	April 21 (2)	May 9 (14)
1991	February 27 (3)	March 14 (11)	April 13 (5)	April 29 (3)
1992	March 16 (4)	April [illegible]	April 16 (2)	May 2 (2)
1993	March 9 (2)	March 14 (8)	April 11 (7)	-[a]
1994	March 22 (4)	March 31 (5)	April 24 (1)	May 10 (3)
		Niantic Bay		
1983	-	April 7 (2)	April 23 (1)	May 10 (4)
1984	-	April 8 (2)	May 4 (3)	May 25 (8)
1985	-	April 1 (4)	April 29 (6)	May 18 (3)
1986	-	April 5 (30	April 28 (3)	May 11 (2)
1987	-	April 6 (6)	April 28 (2)	May 16 (4)
1988	-	March 24 (3)	April 22 (2)	May 9 (5)
1989	-	April 13 (1)	April 23 (2)	May 17 (3)
1990	-	April 3 (8)	April 23 (2)	May 7 (5)
1991	-	March 28 (5)	April 11 (3)	April 29 (4)
1992	-	April 15 (4)	April 30 (2)	May 7 (4)
1993	-	April 3 (44)	May 6 (8)	May 23 (11)
1994	-	April 14 (2)	May 2 (2)	May 20 (3)

[a] Due to low abundance during the 1993 sampling, the Gompertz function could not be fitted to the data.

were also evident of Stages 2 and 3 larvae in the river and bay, but were similar to some other years. This may have been related to the effect of water temperature on egg and larval developmental rates, as the 1994 seasonal winter temperature was about 1°C lower than the long-term average (Table 7). By Stage 4 of development, the dates of peak abundance in 1994 in the river and bay were about average compared to the previous years, suggesting accelerated developmental rate that compensated for the later peaks in previous stages. This may have been due to near-average water temperatures during spring in 1994 (Table 7) and the positive relationship between water temperature and larval developmental rate that is discussed below.

Development and growth

The length-frequency distribution of each larval stage has remained consistent since developmental stage determination began in 1983 (NUSCO 1987, 1988a, 1989, 1990, 1991b, 1992a, 1993, 1994a). Stage-specific length-frequency distributions by 0.5-mm size-classes in 1994 showed some separation in predominant size-classes by developmental stage, which was particularly evident in the later stages (Fig. 15). Stage 1 larvae were primarily (76%) in the 2.5 to 3.0-mm size-classes, 86% of Stage 2 were 2.5 to 3.5 mm, 80% of Stage 3 were 4.5 to 7.0 mm, and 89% of Stage 4 were 6.5 to 8.5 mm. These consistent results from year to year indicated that developmental stage and length of larval winter flounder were closely related. These data agreed with laboratory studies on larval winter flounder, which showed that there were positive correlations between growth and developmental rates (Chambers and Leggett 1987; Chambers et al. 1988). This relationship was the basis for the estimation of developmental stage from length-frequency data.

The length-frequency distributions of larvae (all stages combined) collected in the Niantic River (stations A, B, and C combined) were different from those obtained for Niantic Bay (stations EN and NB combined) in 1994 (Fig. 16). Differences in size-class distribution between the two areas were consistent with previous findings (NUSCO 1987, 1988a, 1989, 1990, 1991b, 1992a, 1993, 1994a) and the pattern seen in spatial distribution by developmental stage, where Stage 1 and 2 larvae were more abundant in the river than in the bay (see Figs. 12 and 13). Smaller size-classes predominated in the river, which had about

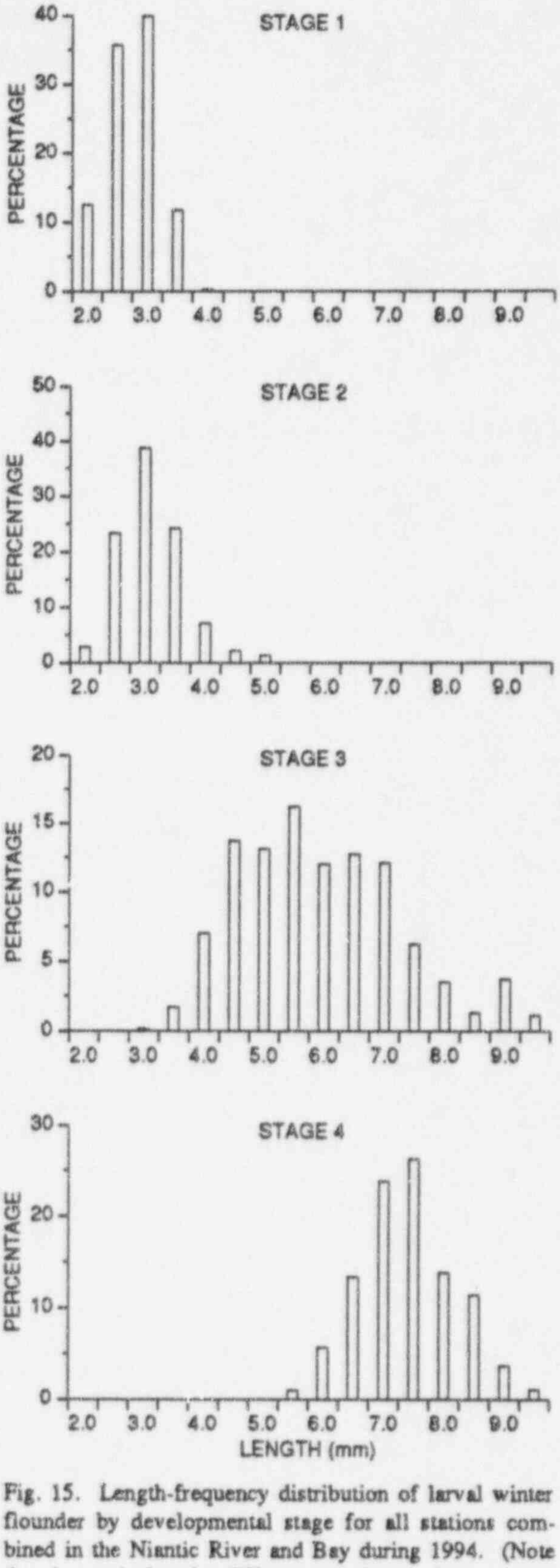


Fig. 15. Length-frequency distribution of larval winter flounder by developmental stage for all stations combined in the Niantic River and Bay during 1994. (Note that the vertical scales differ among the graphs).

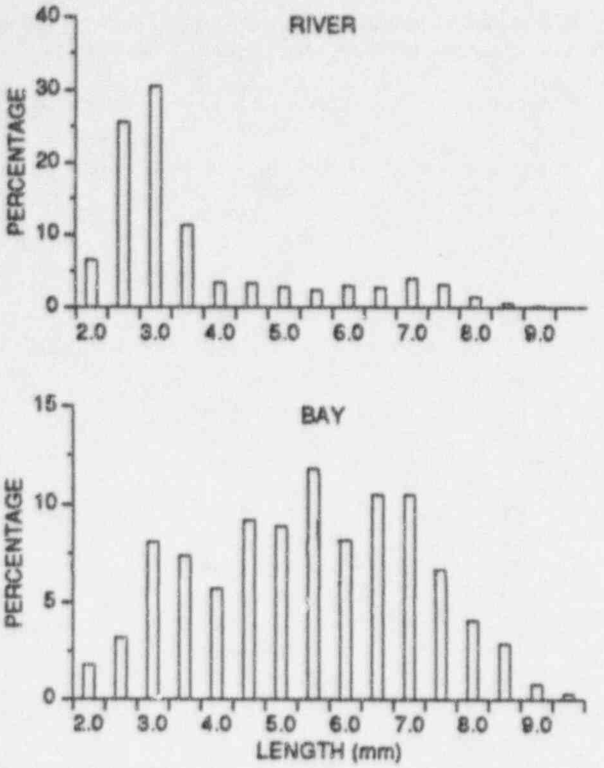


Fig. 16. Length-frequency distribution of larval winter flounder in the Niantic River and Bay during 1994. (Note that the vertical scales differ between the graphs).

74% of the larvae in the 3.5-mm and smaller size-classes. By contrast, more than 74% of the larvae in the bay during 1994 were in the 4.5-mm and larger size-classes. A slight increase in frequency of larger size-classes in the river has been apparent in some previous years in the river (NUSCO 1987, 1988a, 1989, 1991b, 1992a, 1993, 1994a), suggesting that some older larvae were imported to the river. Import of larger size-classes was apparent in the length-frequency distribution at a station in the river mouth sampled in 1991-93 during maximum flood current (NUSCO 1994a).

Length-frequency data from entrainment collections taken from 1976 through 1994 (station EN) were used to estimate larval winter flounder growth rates in Niantic Bay. Weekly mean lengths during a season formed a sigmoid-shaped curve (NUSCO 1988a). The linear portion of the sigmoid curve usually occurred during the middle of the larval season and growth rates were estimated by fitting a linear model to individual larval length measurements over time during that particular period. This model adequately described growth and all slopes (growth rate as $\mathrm{mm \cdot day^{-1}}$) were significantly ($p \leq 0.001$) different from zero (Table 17). In addition, most intercepts of the linear regression were about 3, the approximate size of winter flounder larvae at hatching. Annual growth rates for station EN were variable and ranged from 0.048 to 0.100 $\mathrm{mm \cdot day^{-1}}$, with 1994 about average. To validate this estimation technique, growth rates were estimated from length data collected at station NB from 1979 through 1989 (NUSCO 1990); annual growth rates were highly correlated ($r = 0.89$; $p \leq 0.001$) with those from station EN.

In laboratory studies, water temperature affected the growth rate of winter flounder larvae (Laurence 1975; NUSCO 1988a). To examine the effect of temperature on estimated annual growth rates, mean water temperatures in Niantic Bay, determined using data collected from continuous recorders in the intakes of Units 1 and 2, were calculated for a 40-day period starting at the beginning of the week when the first larval length measurements were used to estimate the annual growth rate (Table 17). Starting points varied from February 28 (1993) to April 3 (1977). A positive exponential relationship was found between growth rate and water temperature (Fig. 17). A similar exponential relationship of temperature to growth was reported for larval plaice by Hovenkamp and Witte (1991). If temperature affects growth rate, then the length of a larva at a specific time during the season should be related to water temperatures to which it has been exposed. Therefore, the mean length of larvae collected at station EN during the period of April 1-15 for each year was compared to the mean March water temperatures (Fig. 18). Again, there was a positive relationship with larger mean lengths associated with warmer March temperatures. The small mean length in 1994 was associated with one of coldest average March water temperatures during the 19-year period.

As concluded previously from comparisons of annual length-frequency distribution and developmental stages, growth and larval development were found to be closely related. If water temperature affects growth rates, then it should also affect larval developmental time. The timing of peak larval abundance should therefore be related to the rates of recruitment and loss (including mortality and juvenile metamorphosis), which, in turn, would be affected by larval development. Annual dates of peak abundance of larval winter flounder collected at EN were negatively correlated to the mean water temperature in March and April (Fig. 19). Earlier dates of peak

TABLE 17. Annual larval winter flounder growth rates in Niantic Bay as estimated from a linear regression fitted to length data collected at station EN. The 95% confidence intervals and mean water temperatures during the first 40 days of the time period are also given.

Year	Time period included[a]	Growth rate ($mm \cdot day^{-1}$)	95% confidence interval	Mean water temperature (°C)[b]
1976	March 21 - May 2	0.100	0.098 - 0.102	7.0
1977	April 3 - June 5	0.076	0.073 - 0.079	6.7
1978	March 26 - June 11	0.055	0.052 - 0.056	4.8
1979	March 25 - June 10	0.058	0.056 - 0.060	5.9
1980	March 23 - June 8	0.060	0.058 - 0.062	5.9
1981	April 5 - May-31	0.064	0.061 - 0.067	7.3
1982	March 28 - May 30	0.063	0.060 - 0.066	5.8
1983	March 6 - May 22	0.056	0.054 - 0.058	5.2
1984	March 25 - May 13	0.069	0.066 - 0.072	6.4
1985	March 17 - June 2	0.059	0.057 - 0.061	6.0
1986	March 30 - May 11	0.094	0.087 - 0.101	7.6
1987	March 22 - May 17	0.079	0.075 - 0.083	7.0
1988	March 27 - May 8	0.088	0.083 - 0.093	7.1
1989	March 26 - May 7	0.069	0.060 - 0.078	7.0
1990	March 4 - May 13	0.071	0.066 - 0.076	5.3
1991	March 10 - April 21	0.059	0.048 - 0.070	4.7
1992	March 15 - May 3	0.064	0.059 - 0.069	5.5
1993	February 28 - May 16	0.048	0.040 - 0.056	3.3
1994	March 27 - June 12	0.076	0.070 - 0.082	6.5

[a] Time period of the weekly mean lengths used to estimate growth rate.

[b] Mean during a 40-day period starting at the beginning of the week that the first weekly mean length was used in estimating growth rate.

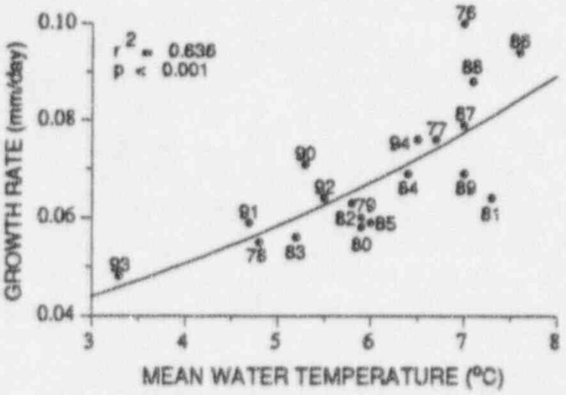


Fig. 17. The exponential relationship between mean water temperature T (°C) and the estimated growth rate G (mm per day) of winter flounder larvae at station EN from 1976 through 1994 ($G = 0.029 \cdot e^{0.142T}$).

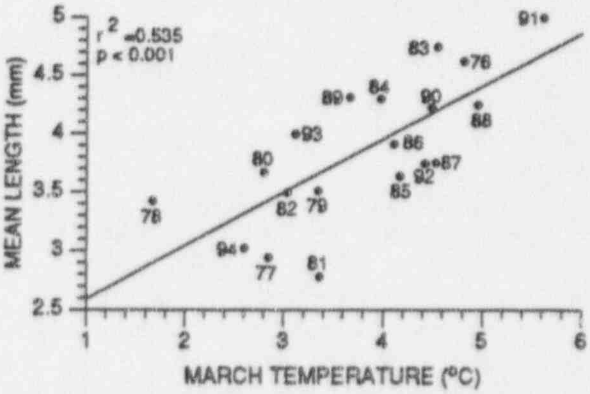


Fig. 18. The relationship between annual mean March water temperature (°C) and the mean length of winter flounder larvae during April 1-15 at station EN for 1976 through 1994.

abundance were associated with warmer mean water temperatures. This agreed with the results of Laurence (1975), who found that winter flounder larvae metamorphosed 31 days earlier at 8°C than at 5°C. Annual dates of peak abundance varied by 41 days during the 17-year period, possibly because of a 3.6°C difference in the March-April water temperature between the earliest (April 13, 1991) and the latest (May 23, 1978) dates of peak abundance. Although the March water temperatures in 1994 were below the 1976-94 mean (Table 6), the near average temperature in April apparently resulted in a date of peak abundance in the mid range for the 19-year time-series. Despite the wide range in annual growth rates, a consistent relationship was found between length-frequency distribution and stage of development (Fig. 15). This

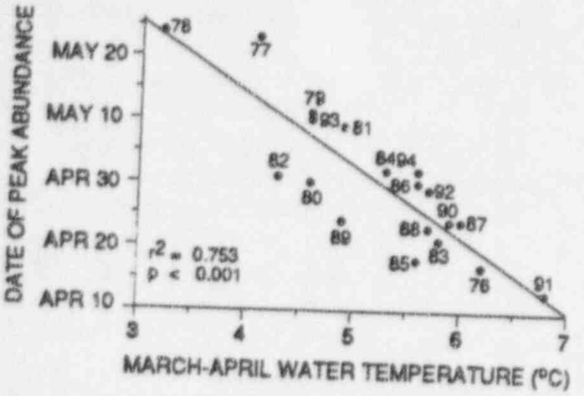


Fig. 19. The relationship between March-April mean water temperature (°C) and the annual date of peak abundance (estimated from the Gompertz function) of winter flounder larvae at station EN from 1976 through 1994.

was consistent with laboratory observations of larval winter flounder as Chambers et al. (1988) found that, at metamorphosis, age was more variable than length and larval age and length were independent.

Growth rates were also estimated for Niantic River larvae using length data from station C with the methods given above. Station C was selected for this analysis because all developmental stages were collected there in abundance (Fig. 12). Estimated growth rates for larvae from the river were generally greater than for larvae from the bay and the rate for 1994 for the river was among the lowest during the 12-year period (Table 18). Again, a linear model provided a good fit and slopes (growth rates as mm·day^{-1}) were highly significantly ($p \leq 0.001$) different from zero. Growth of larvae in the river was similar to laboratory growth rates of 0.104 and 0.101 mm·day^{-1} at mean water temperatures of 6.9 and 7.5°C, respectively (NUSCO 1988a). An annual mean water temperature was determined from surface and bottom temperatures measured at the time of sample collection during a 6-week period starting the same week from which the first length measurements were used in the growth rate calculation. Previously, there was no apparent relationship between growth rates in the river and water temperature, but a negative relationship was found between growth and the abundance of Stage 2 larvae, suggesting density-dependent growth (NUSCO 1990, 1991b, 1992a, 1993). The annual index of Stage 2 larval abundance was the α parameter (Eq. 2) for all three river stations combined (Table 18). The abundance of Stage 2 larvae was examined because during this developmental stage larvae begin to feed. With the addition of 1993 and 1994 data, this density-dependent growth relationship was no longer significant ($p = 0.421$) when tested with a functional regression. Because there was a strong relationship between growth and water temperature in the bay, both Stage 2 abundance and water temperature were used as independent variables in a multiple regression model to examine growth rates. Prior to conducting multiple regression analysis it was determined that the

TABLE 18. Annual larval winter flounder growth rates in the Niantic River as estimated from a linear regression fit to length data collected at station C. The 95% confidence intervals for the growth rate, mean water temperatures during the first 6 weeks of the time period, and the annual abundance indices of Stage 2 larvae in the river are also given.

Year	Time period included[a]	Growth rate (mm·day^{-1})	95% confidence interval	Mean water temperature (°C)[b]	Stage 2 abundance index[c]
1983	March 20 - May 1	0.100	0.096 - 0.104	6.1	749
1984	March 25 - May 6	0.100	0.094 - 0.105	6.4	1,501
1985	March 31 - May 26	0.084	0.080 - 0.088	7.7	4,676
1986	March 23 - May 4	0.109	0.103 - 0.115	8.0	176
1987	March 22 - May 10	0.099	0.095 - 0.103	7.2	829
1988	March 20 - May 21	0.099	0.094 - 0.104	6.8	4,469
1989	March 26 - May 21	0.087	0.082 - 0.092	7.4	3,976
1990	March 25 - May 13	0.106	0.099 - 0.113	7.5	365
1991	March 10 - April 28	0.123	0.114 - 0.132	6.9	252
1992	March 15 - May 17	0.088	0.083 - 0.093	5.7	1,367
1993	March 7 - May 16	0.070	0.065 - 0.075	4.1	133
1994	March 20 - May 29	0.072	0.068 - 0.076	4.7	1,248

[a] Time period of the weekly mean lengths used to estimate growth rate.
[b] Mean during a 6-week period starting the week of the first weekly mean length used in estimating growth rate.
[c] α parameter from the Gompertz function for Stage 2 larvae in the Niantic River (three stations combined).

two independent variables were not correlated (Spearman's rank-order correlation coefficient $r = 0.126$; $p = 0.697$). The multiple regression was significant ($p = 0.010$; $r^2 = 0.643$), with the coefficient for temperature being positive and for Stage 2 abundance negative. These results suggest that winter flounder larval growth in the Niantic River may be a function of both water temperature and larval density. A laboratory growth study of larval winter flounder held at 8°C showed a decrease in growth rate as prey densities decreased (Laurence 1977). This study, along with the apparent density-dependent growth in the Niantic River, suggest that as the number of feeding larvae increased, the numbers of available prey declined to levels less than optimum for larval growth.

Slight declines in growth rate caused by less than optimal food, unfavorable temperatures, disease, or pollution leads to longer developmental times, during which high rates of mortality have a profound effect on recruitment (Houde 1987). Food availability and water temperature appeared to be the two most important factors controlling larval growth (Buckley 1982). Although Laurence (1975) demonstrated that the metabolic demands of larval winter flounder increased at higher temperatures, the growth rate also increased if sufficient food resources were available, and other laboratory studies (Laurence 1977; Buckley 1980) showed that larval winter flounder growth rates depend upon prey availability. In summary, growth and development of larvae in Niantic Bay correlated with water temperature, but in the Niantic River growth appeared to be an interaction of water temperature and density-dependency.

Mortality

Based on length-frequency distributions in the river during 1994 (Fig. 16) and previous years, most winter flounder larval mortality occurred between the 3.0- to 4.0-mm size-classes. In 1994 there was about an 89% decline in frequency of occurrence between these two size-classes, which included yolk-sac (Stage 1) and first-feeding Stage 2 larvae. This initial large decline was followed by smaller decreases to the 5.5-mm size-class, indicating a reduction in the mortality rate. Pearcy (1962) reported a greater mortality for young winter flounder larvae (20.7%·day^{-1}) compared to older individuals (9.1%·day^{-1}). In a laboratory study on winter flounder larvae, Chambers et al. (1988) reported that larval mortality was concentrated during the first 2 weeks after hatching. Based on the estimated growth rate in the river for 1984 of 0.072 mm·day^{-1} (Table 18), it would require about 14 days to grow from 3 to 4 mm. The above 89% decline between these size-classes would be equivalent to a mortality of about 15%·day^{-1}, similar to that reported by Pearcy for young winter flounder larvae in the Mystic River. Laurence (1977) found that winter flounder larvae had a low energy conversion efficiency at first feeding (i.e., Stage 2) compared to later developmental stages, and that it was probably a "critical period" in larval development. Hjorleifsson (1989) showed that the ratio between RNA and DNA, an index of condition and growth rate, was lowest at the time of first feeding of winter flounder (about 4 mm) and that these ratios were affected by food availability. The "critical period" concept, hypothesized by Hjort (1926), was discussed by May (1974) for marine fishes. In many cases, the strength of a year-class is thought to be determined by the availability of sufficient food after completion of yolk absorption.

Predation may be an important cause of larval winter flounder mortality. The escape response of larval winter flounder to predators was studied by Williams and Brown (1992). They found that escape response increased with increasing larval size, but remained slower than that of other larval fishes examined. Larval winter flounder may be vulnerable to both fish and invertebrate predators. Although susceptible to attacks by planktivorous fishes, the occurrence and abundance of fishes that could potentially prey on larval winter flounder is low, particularly during the early portion of the larval winter flounder season. Most predation is likely by invertebrate contact predators, including carnivorous copepods and amphipods, cnidarians, and ctenophores.

There are numerous accounts of jellyfish preying upon and affecting the abundance of fish larvae. Several species of hydromedusae and the scyphomedusan *Aurelia aurita* prey upon Atlantic herring larvae (Arai and Hay 1982; Moller 1984), and laboratory studies with Atlantic cod (*Gadus morhua*), plaice, and Atlantic herring have shown that the capture success by *A. aurita* increased with medusal size (Bailey and Batty 1984). Evidence of a causal predator-prey relationship on larvae of plaice and European flounder (*Platichthys flesus*) by *A. aurita* and the ctenophore *Pleurobrachia pileus* was reported by van der Veer (1985). However, predation by these species was believed to only terminate the plaice larval season and did not ultimately affect year-class strength (van der Veer 1985; van der Veer et al. 1990). Pearcy (1962) stated that *Sarsia tubulosa* medusae were important predators of larval winter flounder in the Mystic River, CT, and had greatest impact on younger, less mobile larvae. Crawford and Carey (1985) reported large numbers of the moon jelly (*A. aurata*) in Point Judith Pond, RI and believed that they were a significant predator of larval winter flounder.

A possible predator of winter flounder larvae in the Niantic River was medusae of the lion's mane jellyfish (*Cyanea* sp.), which was prevalent in the upper portion of the river at station A. Marshall and Hicks (1962) also reported that jellyfish were abundant in the upper river. A laboratory study showed that winter flounder larvae contacting the tentacles of the lion's mane jellyfish were stunned and ultimately died, even if not consumed by the medusa (NUSCO 1988a). During 6 of the 12 years (1983, 1984, 1986, 1989, 1990, and 1994) that larvae were sampled at station A, weekly mean larval abundance was negatively correlated ($p \leq 0.05$, Spearman's rank-order correlation coefficient range of –0.736 to –0.927) with weekly mean jellyfish volume during the period when both medusae and larvae were collected. In 1994, jellyfish abundance was below average compared to previous years (Fig. 20) and during some weeks abundance was below the 95% confidence interval for the 1983-93 period. Although during some years there appeared to be a relationship between the temporal decline of winter flounder larval abundance at station A in the Niantic River with the occurrence of lion's mane jellyfish medusae, for other years there was no relationship between annual larval abundance at station A and annual mean jellyfish volume. The decline in larval abundance at station A may also be attributed to a gradual flushing of larvae out of the upper portion of the river and, thus, a definitive predator-prey relationship cannot be stated.

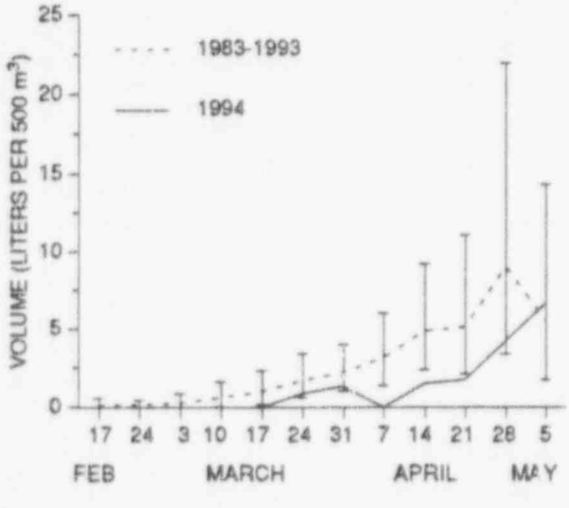


Fig. 20. Comparison of *Cyanea* sp. weekly mean volumes collected in 1983-93 (with 95% confidence intervals) to weekly volumes in 1994. Collections were made at station A in the Niantic River.

The possibility of density-dependent mortality of winter flounder larvae was examined using a function (Eq. 4) provided by Ricker (1975) that requires estimates of annual spawning stock size and larval recruitment. The annual egg production estimate in the Niantic River (Table 13) was used as a measure of spawning stock size. The α parameter from the Gompertz function fit to the abundance of 7-mm and larger larvae collected from 1976 through 1994 at station EN was selected as a measure of larval recruitment, even though many of these larvae may be from other sources (see Mass-balance calculations below). Larvae in the 7-mm and larger size-classes were used because they would soon metamorphose into juveniles. A larval recruitment index was calculated by taking the logarithm of the ratio of the α parameter for 7-mm and larger larvae to the egg production estimates. This value was plotted against egg production estimates and the slope determined with functional regression (Fig. 21). Although there was some scatter around the relationship with the 1993 data point the most aberrant, a significant ($r = -0.605$; $p = 0.008$) negative relationship was found, indicating that compensatory mortality occurred during the

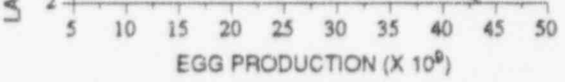


Fig. 21. The relationship (functional regression) between the annual winter flounder egg production in the Niantic River and the larval recruitment index (logarithm of the ratio of the annual abundance of 7 mm and larger larvae to the egg production) at station EN from 1976 through 1994.

winter flounder larval period. Because there was increasing evidence that many of the winter flounder larvae collected at station EN did not originate from the Niantic River, this compensatory relationship suggested that annual egg production estimates for the Niantic River were consistent with regional trends in winter flounder egg production.

Because the egg production estimate was used in calculating the larval recruitment index above, a possibility existed of introducing correlation between the independent (egg production) and dependent (recruitment index) variables. Therefore, another approach for detecting the presence of density-dependent larval mortality for the Niantic River stock was used, where annual larval mortality rates from the river were compared to estimates of river spawning stock size (i.e., egg production). Total larval mortality in the river for 1984-94 ranged from 82.4 to 97.9%, with a mean instantaneous rate (Z) of 2.72 (Table 19). To determine if density-dependent mortality could be identified in the larval stage, the values of Z were compared to egg production estimates using functional regression. With 1933 data included, there was no significant (p = 0.138) relationship between mortality and egg production estimates. This was because in 1993 the lowest annual larval abundance index in the Niantic River occurred when the mortality rate was among the highest (Tables 14 and 19). Therefore, when 1993 data were excluded from the analysis, a significant (r = 0.658; p = 0.038) positive relationship was apparent, such that when egg production and larval abundance increased, larval mortality also increased (Fig. 22). This indicated that density-dependent larval mortality may have occurred in the Niantic River, except for the anomolous 1993 year-class.

TABLE 19. Estimated larval winter flounder total mortality from hatching to the 7-mm size-class.

	Abundance index			
Year	Newly hatched	7-mm size-class	Mortality (%)	Instantaneous mortality rate
1984	6,500	654	89.9	2.30
1985	13,773	452	96.7	3.42
1986	2,483	438	82.4	1.73
1987	6,480	474	92.7	2.62
1988	24,561	678	97.2	3.59
1989	19,192	394	97.9	3.88
1990	7,915	653	91.7	2.49
1991	3,992	560	86.5	2.00
1992	8,020	609	92.4	2.58
1993	1,874	88	95.3	3.06
1994	7,270	761	89.5	2.26
				mean = 2.72

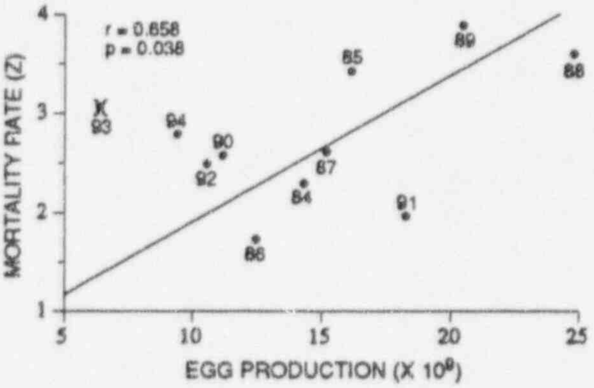


Fig. 22. The relationship (functional regression) between the annual winter flounder egg production in the Niantic River and the larval recruitment index (logarithm of the ratio of the annual abundance of 7 mm and larger larvae to the egg production) at station EN from 1976 through 1994 with 1993 data excluded.

Juvenile winter flounder

Age-0 juveniles (summer)

Abundance. Although beam trawls are much more efficient than small otter trawls for collecting juvenile flatfish (Kuipers et al. 1992), the densities of young-of-the-year winter flounder reported below should be regarded as minimum estimates because of collection inefficiencies. For example, using a beam trawl Berghahn (1986) caught more young plaice at night in comparison to samples taken during the day and Rogers and Lockwood (1989) showed that replacing tickler chains normally used with even heavier, spiked chains nearly doubled catches. Efficiency of the NUSCO 1-m beam trawl was discussed in NUSCO (1987, 1990). Large mats of the alga *Enteromorpha clathrata*, which hampered sampling efficiency in the Niantic River during 1993, were not present this year.

Despite only average production of early stage larvae in 1994, the abundance of Stage 4 larvae was relatively high (Figs. 12 and 13). Following their metamorphosis, numbers of young in the Niantic River were also high in 1994, with densities of more than 1 fish·m^{-2} found during the first 2 months of sampling at stations LR and WA (Fig. 23). Initial recruitment and peak abundance of young was greater

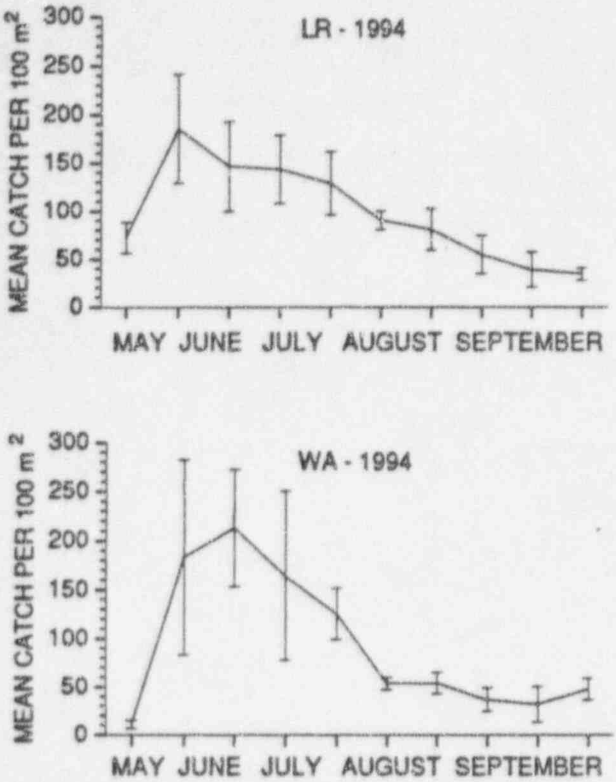


Fig. 23. Weekly mean CPUE (±2 standard errors) of age–0 winter flounder taken in the Niantic River during 1994. (Note that the vertical scales differ among the graphs).

at WA than at LR, but the decrease throughout summer was greater at WA. Similar to observations made in several other years, a small increase in abundance occurred at WA during the last day of sampling in September, perhaps the result of additional fish entering shallow water once water temperatures began to decrease from late summer peaks. Saucerman and Deegan (1991) also found that young winter flounder responded to warm water temperatures during late August in Waquoit Bay, MA by moving into deeper water and returning to the shallows after those areas became cooler.

Abundance indices for early summer at both LR ($128.8 \cdot 100m^{-2}$) and WA (126.3) were only exceeded by the density of $156.6 \cdot 100m^{-2}$ for LR recorded in 1990 (Table 20). Abundance indices for the second half of the summer at LR (62.9) and WA (49.2) were greater than any of those found during 12 years of sampling in the river. A plot of moving average densities for each station illustrated the relative strength of winter flounder year-classes since 1984 (Figs. 24 and 25). In comparison to 1993, during which few winter flounder were produced, the 1994 year-class ranks with 1988 as one of the most abundant of the past decade.

Growth. Increases in mean length over time were used to express growth of age-0 winter flounder. A consistent, relatively rapid increase in biweekly mean lengths was observed from May through late June (Fig. 26), which was also typical during other years sampled. Smaller increases in mean lengths occurred through the end of sampling in September. Fast growth after settlement followed by a rapid decline in growth rate was also reported for young winter flounder in New Jersey bays by Sogard and Able (1992), who reported nearly imperceptible growth by the time young reached 50 mm in length. Growth of age-0 winter flounder in the Niantic River was less variable than abundance as the weekly means had relatively small confidence intervals. It is likely that growth compensation occurs in winter flounder where size-at-age, which may diverge in larval stages, converges during the early juvenile phase and progressive declines are seen in size-at-age differences (Bertram et al. 1993).

Mean length of young at LR during late summer (July through September) of 1994 was 40 mm, in contrast to a mean of 61 mm in 1993 (Table 21). The mean of 39 mm at WA was the lowest observed at either river station during any year. The relatively large mean lengths in association with low abundance in some years (e.g., 1983, 1984, 1989, and 1993) and small mean lengths in 1994 when fish were very abundant may be indicative of density-dependent growth. However, this has not been consistent for all years of study (e.g., 1988), as other environmental factors also influenced growth of young winter flounder. Growth was likely affected by water temperature and even though faster growth occurs in warm waters, optimal growth temperatures can be exceeded (Sogard and Able 1992). Bergman et al. (1988) and van der Veer et al. (1990) noted that growth of young plaice in northwestern Europe was not food-limited, but was related to prevailing water temperatures and the length of the growing season in different nursery areas. Furthermore, fish grew more rapidly on the warmer nursery grounds in embayments than did fish settling on beaches in the cooler North Sea. Water temperatures were particularly warm during 1994, especially in July (Table 6); this probably affected growth by increasing respiratory and other metabolic demands. Bergman et al. (1988) and Zijlstra et al. (1982) re-examined reports by Steele and

TABLE 20. Seasonal 1-m beam trawl median CPUE (number·$100m^{-2}$) of age-0[a] winter flounder at two stations in the lower Niantic River (LR and WA) from 1983 through 1994.

Survey year[a]	Station	Season[b]	Tows used for CPUE	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness[c]
1983	LR	Early	30	32.7	20.0 - 50.7	2.29
	LR	Late	27	10.0	8.0 - 13.3	0.49
1984	LR	Early	40	18.8	16.7 - 25.0	0.63
	LR	Late	36	6.3	3.8 - 7.5	0.58
	WA	Late	32	11.3	8.0 - 17.5	0.94
1985	LR	Early	40	13.3	10.0 - 16.3	0.91
	LR	Late	32	7.0	6.0 - 8.0	0.97
	WA	Early	40	15.0	10.0 - 20.0	0.81
	WA	Late	32	9.0	8.0 - 10.0	0.70
1986	LR	Early	39	33.8	23.3 - 40.0	0.33
	LR	Late	36	13.8	12.5 - 17.5	0.80
	WA	Early	40	21.7	12.5 - 26.7	1.49
	WA	Late	36	18.1	15.0 - 20.0	2.03
1987	LR	Early	40	59.2	53.3 - 73.3	-0.12
	LR	Late	36	17.9	12.5 - 26.7	0.70
	WA	Early	40	28.3	21.7 - 38.3	0.27
	WA	Late	36	10.6	6.0 - 13.8	0.83
1988	LR	Early	40	61.3	52.5 - 72.5	0.37
	LR	Late	36	60.0	50.0 - 70.0	1.17
	WA	Early	40	40.0	32.5 - 51.7	0.13
	WA	Late	36	38.3	33.3 - 51.7	0.22
1989	LR	Early	40	17.5	11.7 - 21.7	0.09
	LR	Late	36	8.8	7.0 - 11.3	0.84
	WA	Early	40	10.0	8.3 - 13.8	1.16
	WA	Late	34	5.5	4.0 - 10.0	0.66
1990	LR	Early	40	156.3	137.5 - 187.5	1.05
	LR	Late	36	20.0	15.0 - 52.5	1.10
	WA	Early	40	68.8	50.0 - 95.0	0.62
	WA	Late	36	13.5	10.0 - 19.0	1.20
1991	LR	Early	44	77.5	51.7 - 90.0	0.96
	LR	Late	36	21.7	18.3 - 28.3	0.75
	WA	Early	44	37.9	30.0 - 43.3	1.34
	WA	Late	36	25.8	21.3 - 31.7	1.27
1992	LR	Early	40	90.0	57.5 - 122.5	1.16
	LR	Late	36	28.1	23.8 - 33.3	0.51
	WA	Early	40	74.6	56.7 - 82.5	1.35
	WA	Late	36	30.0	27.5 - 32.5	0.23
1993	LR	Early	20	10.6	7.0 - 15.0	0.68
	LR	Late	20	5.0	3.0 - 7.0	1.15
	WA	Early	20	5.0	3.8 - 7.5	2.57
	WA	Late	20	5.5	4.0 - 10.0	0.77

TABLE 20. (continued).

Survey year[a]	Station	Season[b]	Tows used for CPUE	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness[c]
1994	LR	Early	20	128.8	125.5 - 172.5	0.38
	LR	Late	20	62.9	38.3 - 75.0	0.26
	WA	Early	20	126.3	92.5 - 192.5	0.31
	WA	Late	20	49.2	35.0 - 55.0	-0.79

[a] For age-0 fish, the year-class is the same as the survey year.

[b] Early season corresponds to late May through July and late to August through September.

[c] Zero for symmetrically distributed data.

Edwards (1970), Lockwood (1972), and Rauck and Zijlstra (1978) of density-dependent growth of age-0 plaice in Britain. They concluded that increases in length corresponded to maximum growth expected from prevailing water temperatures and that growth was not density-dependent. Similarly, Pihl and van der Veer (1992) determined that growth of young plaice in Swedish bays appeared to be affected by ambient water temperatures and was not food-limited. However, Berghahn (1987) and Karakiri et al. (1989) suggested that food limitation and not water temperature could have been responsible for growth differences of plaice observed among different years within the German Wadden Sea.

Other factors found to affect growth of young winter flounder include physical location and specific habitat (Sogard 1990; Sogard and Able 1992). Benthic food production and its availability also may differ among areas within the Niantic River and likely changes from year to year. Karakiri et al. (1989) reported differences in the size of young plaice of similar age between Wadden Sea estuarine nursery grounds (larger fish) and coastal waters off Germany (smaller fish). They suggested that the differences were due to lower water temperature, food limitation, or wave action in the waters outside of the Wadden Sea. Al-Hossaini et al. (1989) reported greater growth for cohorts of plaice that settled relatively early in Wales, but these fish also had higher mortality. Conversely, growth was slower for late-settling cohorts, but survival was higher. Similarly, growth of young Niantic River winter flounder was affected by many factors, with temperature and abundance likely the major determinants.

Mortality. Catch curves constructed from weekly abundance data by year and station were used to obtain estimates of monthly instantaneous mortality rate (Z_{mo}); this method assumed that young comprised a single-age cohort throughout the season. With some exceptions, the catch curves generally fit the data well with relatively high r^2 values (Table 22). No estimates could be determined for LR and WA during the high abundance year of 1988 as slopes of these catch curves were not significantly different from zero and for WA in 1986 and 1993 because of considerable variation in weekly abundance during those years. The Z_{mo} estimate for station LR in 1994 was 0.476 (equivalent to a survival of 62.1%). As expected from the relatively greater decline in abundance of young at WA than at LR (Fig. 23), Z_{mo} for fish at that station was larger (0.538; S = 58.4%). Long-term averages of Z_{mo} at LR and WA were 0.618 and 0.550, respectively. Mortality estimates for Niantic River winter flounder were usually greater than the equivalent Z_{mo} value of 0.371 reported by Pearcy (1962) for the Mystic River, CT estuary, but were similar to various estimates (0.563 - 0.693) made for young plaice in British coastal embayments (Lockwood 1980; Poxton et al. 1982; Poxton and Nasir 1985; Al-Hossaini et al. 1989).

During 1988-92, when both areas were sampled, mortality of young was much greater at two stations sampled in Niantic Bay than in the Niantic River (NUSCO 1994a). Except for a station just outside the mouth of the Niantic River in 1988, no young were found in Niantic Bay following mid-summer. Even in 1988, however, densities at the Niantic Bay station in late summer were only 10 to 15% of those in the river. Because of the apparent lack of production of young in Niantic Bay, no further sampling was proposed at this site beginning in 1994.

NUSCO (1994a) concluded that high natural mortality of young winter flounder in Niantic Bay was the probable reason for declines in density following larval metamorphosis and settlement to the bottom and not off-station emigration. Thus, it is likely that

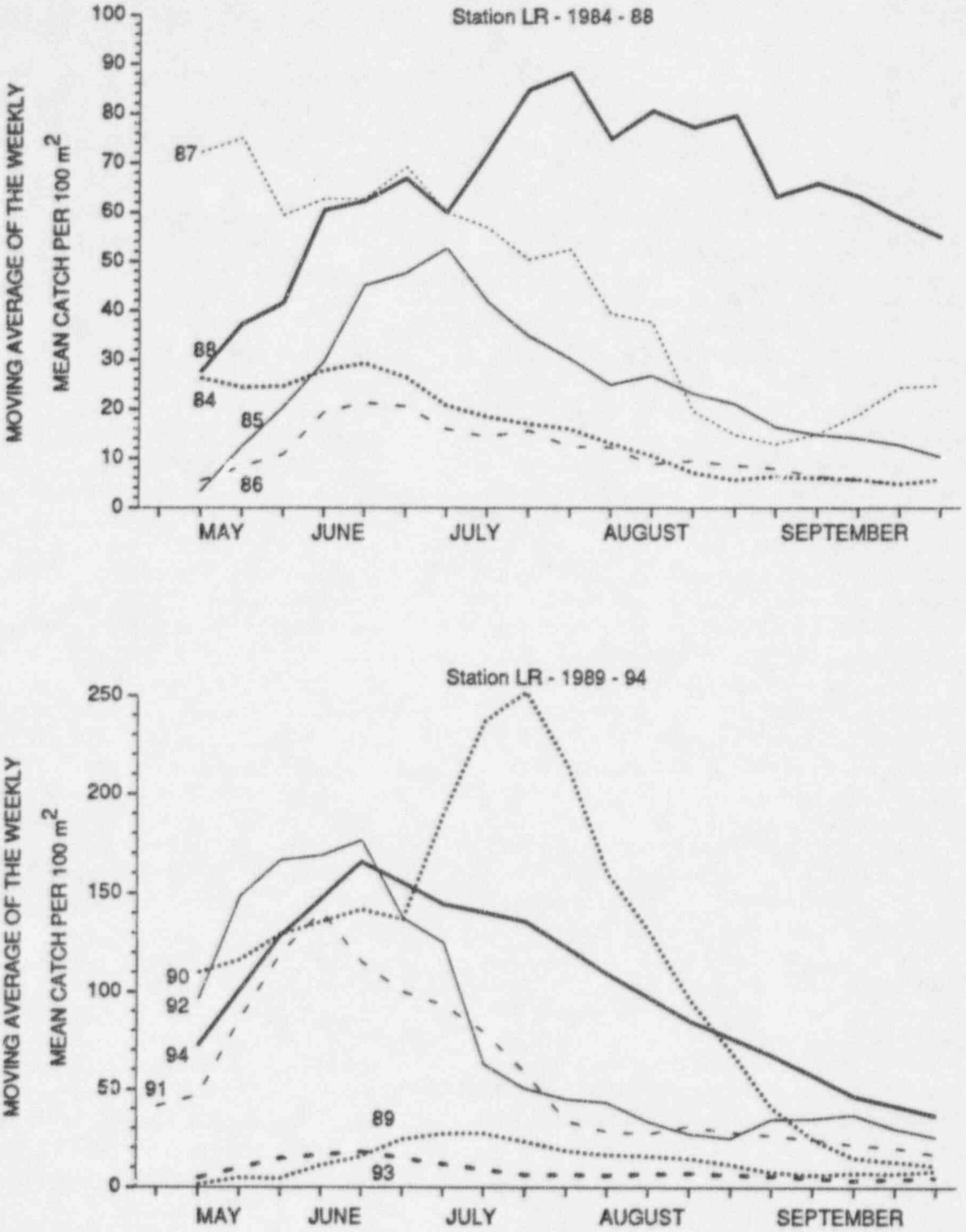


Fig. 24. Moving average of weekly mean CPUE of age-0 winter flounder taken at station LR in the Niantic River from 1984 through 1988 and 1989 through 1994. (Note that the vertical scales differ between the graphs).

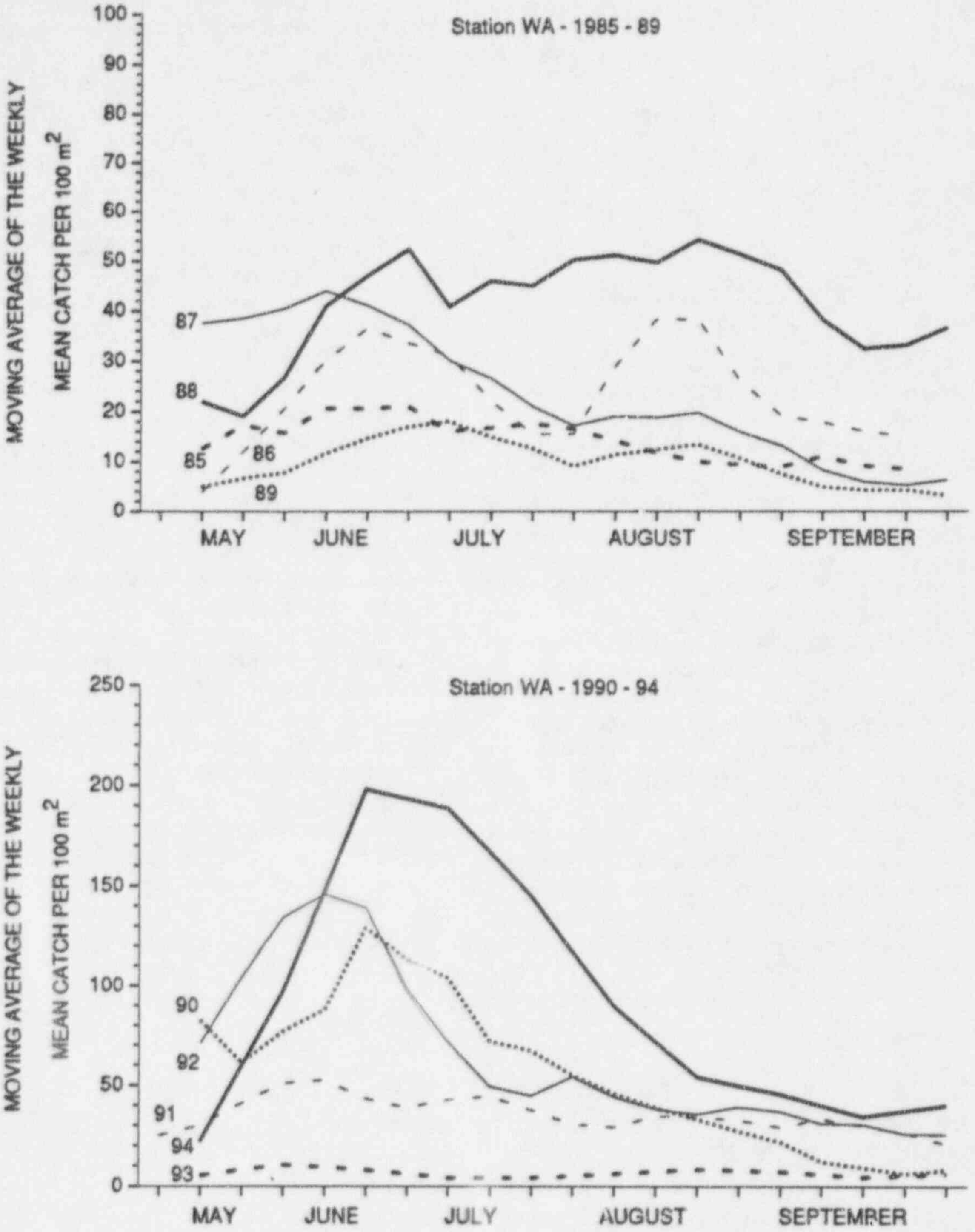


Fig. 25. Moving average of weekly mean CPUE of age-0 winter flounder taken at station WA in the Niantic River from 1985 through 1989 and 1990 through 1994. (Note that the vertical scales differ between the graphs).

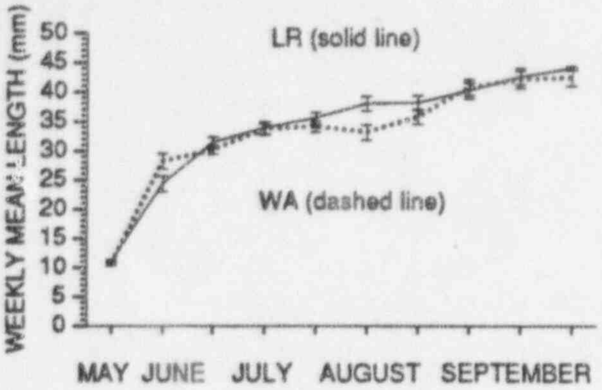


Fig. 26. Weekly mean length (±2 standard errors) of age-0 winter flounder taken in the Niantic River during 1994.

few older juveniles are produced in the bay compared to the Niantic River. The cause of high mortality shortly before or after settlement has not been investigated. Of all life stages of marine fishes, least is known about larval and early juvenile stages, yet this is likely where relative year-class strengths are determined (Sissenwine 1984; Bailey and Houde 1989). Predation by caridean shrimp (*Crangon* spp.) has been suggested as the cause of high mortality after metamorphosis for both winter flounder (Witting and Able 1993) and plaice (Lockwood 1980; van der Veer and Bergman 1987; Pihl 1990; van der Veer et al. 1990; Pihl and van der Veer 1992). Van der Veer et al. (1990) speculated that, in general, predation by crustaceans on young may be a common regulatory process for flatfishes. Witting and Able (1993) found that the size of age-0 winter flounder significantly affected their probability of predation by sevenspine bay shrimp (*Crangon septemspinosa*), with greatest risk found for smallest fish. Young apparently outgrew predation by shrimp when they reached 17 mm in length, which meant that fish would have to double in size after settlement before attaining a size refuge from shrimp predation. Therefore, the duration of time spent in a vulnerable size range, which is related to growth rate, affects the vulnerability of young winter flounder to predation by shrimp and other organisms. Variation in growth, which can depend upon specific location of settling, specific habitat within a location, or temperature (Sogard 1990; Sogard and Able 1992) may have significant implications for young winter flounder survival after metamorphosis.

Recruitment of many fishes is greatly affected by density-dependent processes occurring during the first year of life following the larval stage (Bannister et al. 1974; Cushing 1974; Sissenwine 1984; Anderson 1988; Houde 1989; Myers and Cadigan 1993a, 1993b). Bannister et al. (1974), Lockwood (1980), and van der Veer (1986) all reported density-dependent mortality for young plaice, although examination of their findings indicated that greatest rates of mortality occurred only when extremely large year-classes of plaice were produced (three to more than five times larger than average). NUSCO (1994a) noted that when densities were about 1.5 to $2 \cdot m^{-2}$ during early summer in 1990 and 1992 (Figs. 24 and 25), higher mortality was observed than occurred in other years. Survival rates were highest in 1988, despite abundance that was well above average, as peak densities

TABLE 21. Comparison of the mean lengths of age-0 winter flounder taken at stations LR and WA in the Niantic River during late July through September of 1983 through 1994.

Mean length[a] in mm for station and year:

66	61	59	58	57	56	55	51	51	51	50	48	47	46	45	44	43	43	42	42	42	40	39
LR	LR	LR	LR	LR	LR	WA	WA	LR	WA	WA	LR	WA	LR	LR	WA	LR	WA	WA	WA	WA	LR	WA
83	93	84	89	85	91	91	93	88	88	89	92	87	86	87	92	90	84	85	90	86	94	94

Difference between the late seasonal mean at LR as compared to that for WA:

Year	84	85	86	87	88	89	90	91	92	93	94
Difference in mm	16	15	4	-2	0	8	1	1	4	10	2

[a] Means joined by underlining are not significantly ($p \leq 0.05$) different from each other, as determined by analysis of variance and Duncan's multiple-range test.

TABLE 22. Monthly instantaneous total mortality rate (Z) estimates as determined from catch curves for age-0 winter flounder taken at two stations (LR and WA) in the Niantic River from 1984 through 1994.

Year	Station	n[a]	slope[b]	Standard error	r^2	Station	n[a]	slope[b]	Standard error	r^2
1984	LR	16	-0.129 **	0.017	0.80	WA	-	-	-	-
1985		15	-0.118 **	0.015	0.82		16	-0.084 **	0.023	0.51
1986		15	-0.127 **	0.012	0.89		-[c]	-[c]	-	-
1987		15	-0.108 **	0.021	0.67		16	-0.139 **	0.016	0.84
1988		19	NS	-	-		19	NS	-	-
1989		12	-0.154 **	0.022	0.84		13	-0.145 **	0.028	0.71
1990		13	-0.322 **	0.028	0.92		15	-0.235 **	0.028	0.84
1991		18	-0.140 **	0.016	0.82		18	-0.049 **	0.011	0.54
1992		18	-0.129 **	0.019	0.74		16	-0.112**	0.009	0.91
1993		9	-0.087 *	0.028	0.57		10	NS	-	-
1994		9	-0.110 **	0.008	0.96		9	-0.124 **	0.020	0.84

Year	Station	Mortality (Z_{mo})	Survival (S_{mo})	Station	Mortality (Z_{mo})	Survival (S_{mo})
1984	LR	0.560	57.1%	WA	-	-
1985		0.512	59.9%		0.363	69.9%
1986		0.552	57.6%		-[c]	-
1987		0.469	62.6%		0.604	54.7%
1988		-	-		-	-
1989		0.669	51.2%		0.630	53.3%
1990		1.398	24.7%		1.021	36.0%
1991		0.608	54.4%		0.213	80.8%
1992		0.560	57.1%		0.486	61.5%
1993		0.377	68.6%		-	-
1994		0.476	62.1%		0.538	58.4%
	Mean	0.618	53.9%	Mean	0.550	57.7%
	SD	0.286		SD	0.253	
	CV	46%		CV	46%	

[a] Weekly sampling during 1984-92 and biweekly sampling during 1993-94. WA was not sampled in 1984.

[b] Probability level that the slope of the catch curve differs from zero is shown:
NS - not significant ($p > 0.05$), * - significant at $p \leq 0.05$, ** - significant at $p \leq 0.01$.

[c] Although having a significant slope, the catch curve for 1986 at station WA did not provide a reliable estimate of Z because of considerable variation in weekly abundance.

that year remained below $1 \cdot m^{-2}$. However, although peak densities at LR during 1994 approached that of 1992 ($>1.5 \cdot m^{-2}$), the mortality rate during July through September was apparently less this year than in either 1992 or 1990. At WA, the 1994 abundance peak was the highest of all years sampled and densities reached about $2 \cdot m^{-2}$. Although a sharp decline occurred in July and early August, the abundance curve flattened out in late August and September, perhaps implying a lessening of mortality rate. Thus, the relationship between density and mortality rate for young winter flounder may be subject to considerable variability (i.e., regulatory mechanism not well-established). As mortality rate was apparently not excessive during the summer of 1994, the number of young remaining was relatively substantial and this year-class has the potential of being relatively strong.

Age-0 juveniles (late fall and early winter)

As water temperatures decrease in fall, young winter flounder disperse from shallow waters near the shoreline to deeper waters and become available for sampling by the otter trawl used in the year-round trawl monitoring program (TMP). Young are first regularly captured by trawl at the two shallower inshore stations (NR and JC) adjacent to inshore nursery grounds in November, the near-shore Niantic Bay stations (IN and NB) in December, and at the deeper-water stations in LIS (TT and BR) in January.

A Δ-mean (NUSCO 1988b) index of relative abundance was developed for these age-0 fish using TMP catch data, beginning with the months given above and continuing through the end of February. This November-February period is a transition following the 1-m beam trawl sampling of young in summer and preceding the catch of this cohort of fish as age-1 juveniles during the intensive adult winter flounder survey that takes place in the Niantic River, usually from late February through early April. Based on the availability of data for this report, the most recent Δ-mean CPUE is for the 1993 year-class. The Δ-mean CPUE of 7.4 is the lowest recorded since 1987-88 (Table 23). The strength of the 1988 and 1992 year-classes of young are evident from the corresponding Δ-means of 29.6 and 31.1, respectively.

Since 1983, when data were first available from beam trawl sampling, the fall-early winter Δ-means were compared to a 1-m beam trawl median CPUE from late summer using data from both stations LR and WA in the Niantic River (Fig. 27). These abundance indices track each other closely and are significantly correlated (Spearman's rank-order correlation coefficient r = 0.84; p = 0.002). However, no obvious relationship (r = 0.05; p = 0.89) was found between the TMP Δ-mean CPUE and median CPUE of winter flounder smaller than 15 cm taken in the Niantic River during the subsequent (late February-early April) adult winter flounder survey (Fig. 28). For the 1985 and earlier year-classes, more juvenile winter flounder were taken in the river than at the six TMP stations (five of which are outside of the Niantic River). But since the 1988 year-class was produced, consistently more fish have been taken by the TMP, although the TMP Δ-mean CPUE for the 1993 year-

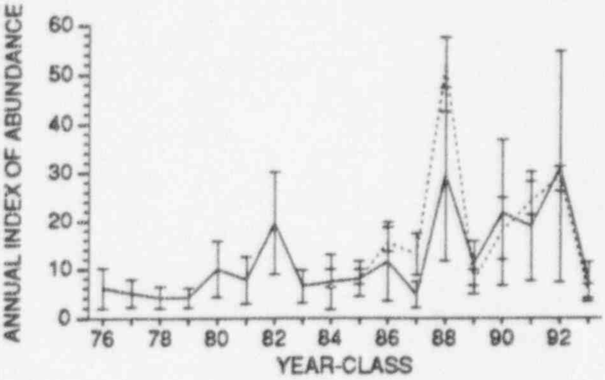


Fig. 27. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the 1983-93 late summer Niantic River (stations LR and WA combined) age-0 1-m beam trawl median CPUE (dashed line) for the 1976-93 year-classes.

TABLE 23. The late fall-early winter seasonal[a] Δ-mean CPUE[b] of age-0[c] winter flounder taken at the six trawl monitoring stations in the vicinity of MNPS from 1976-77 through 1993-94.

Survey year[c]	Number of samples	Non-zero observations	Δ-mean[b]	95% confidence interval
1976-77	42	36	6.1	2.0 - 10.3
1977-78	42	38	5.1	2.3 - 7.9
1978-79	42	36	4.2	2.0 - 6.4
1979-80	42	38	4.2	2.2 - 6.2
1980-81	42	39	10.1	4.3 - 15.9
1981-82	42	39	7.7	2.9 - 12.5
1982-83	42	37	19.6	9.0 - 30.3
1983-84	42	39	6.6	3.2 - 10.0
1984-85	42	35	7.4	1.7 - 13.1
1985-86	42	39	8.1	4.4 - 11.7
1986-87	42	39	11.7	3.4 - 19.9
1987-88	42	41	4.8	2.1 - 7.5
1988-89	42	41	29.6	11.8 - 47.3
1989-90	42	42	11.3	6.7 - 15.9
1990-91	42	40	21.7	6.7 - 36.8
1991-92	42	41	19.0	7.6 - 30.3
1992-93	42	39	31.1	7.4 - 54.8
1993-94	42	38	7.4	3.4 - 11.4

[a] Data restricted to November-February for NR and JC, December-February for IN and NB, and January-February for TT and BR.

[b] Catch per standardized tow of 0.69 km (see Materials and Methods of Fish Ecology section).

[c] For age-0 fish, the year-class is the same as the first year given.

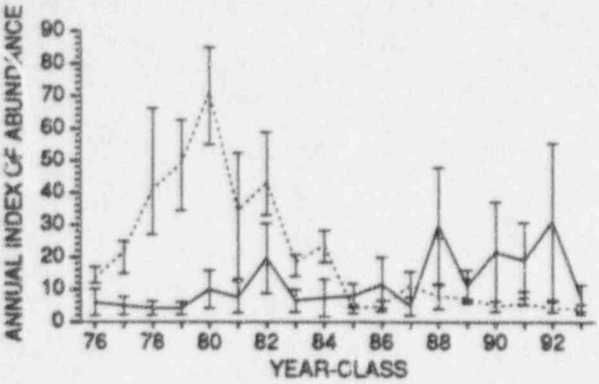


Fig. 28. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the Niantic River (stations 1 and 2) spawning survey median CPUE (dashed line) of winter flounder smaller than 15 cm for the 1976-93 year-classes.

class declined relative to 1992.

Relationships among abundance indices of juvenile winter flounder may have been obscured by differences in sampling gear and variations in fish behavior. Major biases in abundance estimation can arise from size selectivity of the gear, spatial distribution of individuals in relation to the gear, and behavior of fish in the vicinity of the gear (Parrish 1963). Mean lengths of age-0 winter flounder taken by otter trawl in fall were usually about 15 to 25 mm larger than those taken during the immediately preceding months by 1-m beam trawl. This size difference was greater than would have been expected from growth alone and suggests that CPUE indices were biased because smaller individuals were excluded from the catch. Differences in mean length by year (Table 21) may also have differentially biased the trawl sampling. The fixed locations of otter trawl sampling stations in relation to the habitat available to juveniles also may have affected catches. Movements of small juveniles were probably influenced by factors such as water temperature and tide. Moreover, their availability to sampling gear in fall and winter appeared to have varied from week to week and year to year. Relatively large confidence intervals around the Δ-mean CPUE values were probably a consequence of this variation. In contrast, variation was less in data collected during summer by the relatively efficient 1-m beam trawl. Furthermore, sampling in summer occurred weekly or biweekly during the same tidal stage and in areas known to be preferred habitat of young winter flounder. Finally, most likely a mixture of juveniles from a number of sources occurs throughout LIS during the winter. This would have influenced measures of abundance because of potential variable contributions from different stocks.

Age-1 juveniles (late winter)

Small winter flounder are incidentally captured each year during the February-April adult winter flounder surveys in the Niantic River. An annual median CPUE was calculated for winter flounder smaller than 15 cm, which included mostly age-1 fish spawned during the previous year; adjustments made to the catch data for the calculation of CPUE were similar to those previously discussed for adult fish. In some annual comparisons, data were restricted to stations 1 and 2 in the navigational channel (Fig. 2) because the distribution of small winter flounder generally varied more than for adult fish and, moreover, no tows were made in the upper river from 1977 through 1980.

The 1994 median CPUE for age-1 juveniles taken in the navigational channel of the lower Niantic River was 4.1, which similar to the CPUE of 4.3 for 1993 (Table 24). When tows from throughout the river were considered in the calculation, the median CPUE was 1.8, marking the fifth consecutive year that relatively low values were recorded for this time-series (Table 25). Behavior of juvenile winter flounder largely influences their availability to sampling and apparently varies from year to year as a result of changing environmental conditions. Distribution and, therefore, relative abundance of small winter flounder differs between Niantic Bay and River from year to year (NUSCO 1993). Data from the TMP this spring, however, are not yet available for determining abundance of winter flounder in Niantic Bay during the 1994 adult spawning season. A Δ-mean CPUE was computed for winter flounder smaller than 15 cm taken by the TMP from January through April at stations outside of the Niantic River. This time span overlapped the spawning period and also served to increase sample size. Values of this index were then compared to the CPUE median of fish found within the river during the spawning season (Fig. 29). Generally, the catch of age-1 winter flounder in the winter and early spring fluctuated less outside than inside the Niantic River. As the number of small fish in the river declined to low levels in recent years, abundance of age-1 winter flounder in Niantic Bay remained relatively constant. The CPUE of fish found in the bay was greater than that of fish taken in the

TABLE 24. Annual 9.1-m otter trawl adjusted median CPUE[a] of winter flounder smaller than 15 cm[b] taken in the navigational channel of the lower Niantic River during the 1976 through 1994 adult population abundance surveys.

Survey year[b]	Weeks sampled	Tows acceptable for CPUE[c]	Adjusted number of tows used[d]	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness[e]
1976	7	98	154	20.0	19.0 - 20.0	2.77
1977	6	166	229	13.5	12.0 - 17.0	1.50
1978	6	129	156	21.6	15.0 - 25.0	1.59
1979	5	107	136	41.0	27.0 - 66.3	2.82
1980	5	110	145	49.3	34.5 - 62.6	1.30
1981	7	93	140	71.1	55.0 - 84.8	0.79
1982	5	50	70	34.4	13.2 - 52.5	1.46
1983	7	77	77	43.0	33.0 - 58.8	0.55
1984	7	72	77	18.5	14.2 - 20.8	2.23
1985	7	82	84	23.6	18.4 - 28.2	1.27
1986	7	72	118	4.1	2.7 - 5.3	1.57
1987	5	41	50	5.0	4.3 - 6.7	2.08
1988	6	49	54	11.2	7.7 - 15.7	1.38
1989	7	50	54	7.9	4.0 - 11.9	1.19
1990	7	65	91	7.4	5.8 - 13.3	2.06
1991	6	45	60	4.9	3.3 - 6.5	2.55
1992	7	31	49	5.6	5.2 - 9.4	2.10
1993	7[f]	36	48	4.3	3.1 - 6.3	2.02
1994	4	22	24	4.1	2.5 - 5.5	1.07

[a] Catch per standardized tow (see Materials and Methods).

[b] Mostly age-1 fish, so predominant age-class was produced 1 year before the survey year.

[c] Only tows of standard time or distance made in the navigational channel of the lower river were considered.

[d] Effort equalized among weeks.

[e] Zero for symmetrically distributed data.

[f] Because of low effort, data from the third week of sampling not used for the computation of CPUE.

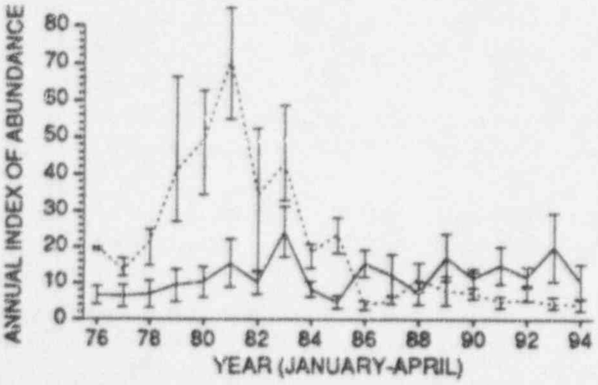


Fig. 29. Comparison between the annual January-April Δ-mean CPUE (solid line) for all trawl monitoring program stations except NR and the Niantic River (stations 1 and 2) spawning survey median CPUE (dashed line) for winter flounder smaller than 15 cm from 1976 through 1994.

river from 1986 through 1994, except for 1988. A small CPUE value from the Niantic River may not necessarily represent a continued decline in abundance. Even a relatively small increase in catch for the much larger geographical area of Niantic Bay and nearby LIS could account for a low abundance index in the river as fish dispersed from a relatively confined area to more open waters. As a result of the differential distribution and abundance of age-1 juveniles, perhaps as a result of variable environmental conditions influencing their behavior and availability to sampling, their abundance indices remain generally unreliable predictors of future adult population size.

Comparisons among life-stages of winter flounder year-classes

Abundance indices for various life-stages of the 1976 through 1994 year-classes of winter flounder discussed throughout this report are summarized in Table 26. Coefficients of variation (CV) were used to

TABLE 25. Comparison of annual 9.1-m otter trawl adjusted median CPUE[a] of winter flounder smaller than 15 cm[b] taken in the navigational channel of the lower Niantic River with those caught throughout the entire sampling area of the river during the 1976 through 1994 adult population abundance surveys.

	Navigational channel only:			Entire area of river sampled:		
Survey year[b]	Adjusted number of tows used[c]	Median CPUE estimate	Coefficient of skewness[d]	Adjusted number of tows used[c]	Median CPUE estimate	Coefficient of skewness[d]
1976	154	20.0	2.77	231	14.4	2.84
1977	229	13.5	1.50	Insufficient tows made in upper river		
1978	156	21.6	1.59	Insufficient tows made in upper river		
1979	136	41.0	2.82	Insufficient tows made in upper river		
1980	145	49.3	1.30	Insufficient tows made in upper river		
1981	140	71.1	0.79	182	14.0	1.64
1982	70	34.4	1.46	118	8.7	2.40
1983	77	43.0	0.55	238	11.5	1.80
1984	77	18.5	2.23	287	6.4	4.08
1985	84	23.6	1.27	280	13.3	2.36
1986	118	4.1	1.57	336	4.0	1.47
1987	50	5.0	2.08	270	3.2	2.46
1988	54	11.2	1.38	312	3.7	3.03
1989	54	7.9	1.19	318	6.1	1.64
1990	91	7.4	2.06	320	2.0	5.00
1991	60	4.9	2.55	330	1.4	5.41
1992	49	5.6	2.10	406	2.0	4.58
1993	48	4.3	2.02	392	1.9	3.08
1994	24	4.1	1.07	212	1.8	1.74

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[a] Catch per standardized tow (see Materials and Methods).

[b] Mostly age-1 fish, so predominant age-class was produced 1 year before the survey year.

[c] Effort equalized among weeks.

[d] Zero for symmetrically distributed data.

TABLE 26. Comparison of indices of abundance for various life-stages of winter flounder for the 1976 through the 1994 year-classes.

	Adult indices		Larval indices				Juvenile indices				
			Niantic River stations (Feb-Jun)								
Year-class	Female spawners (Feb-Apr)	Annual egg production	Stage 1 (3 mm)	Stage 2 (3.5 mm)	Stage 3 (6 mm)	Stage 4 (7.5 mm)	MNPS (EN) (≥7 mm)	Lower river (May-Jul)	Lower river (Aug-Sep)	River/bay Δ-mean (Nov-Feb)	Age-1 CPUE (Feb-Apr)
76	-	-	-	-	-	-	854	-	-	6.1	13.5
77	884	394.6	-	-	-	-	567	-	-	5.1	21.6
78	1,412	717.5	-	-	-	-	754	-	-	4.2	41.0
79	1,120	535.3	-	-	-	-	641	-	-	4.2	49.3
80	903	424.3	-	-	-	-	845	-	-	10.1	71.1
81	2,669	1,383.1	-	-	-	-	561	-	-	7.7	4.4
82	2,752	1,596.8	-	-	-	-	610	-	-	19.6	43.0
83	1,869	1,082.0	-	749	408	56	1,215	32.7	10.0	6.6	18.5
84	871	501.6	2,601	1,501	573	67	917	18.8	6.3	7.4	23.6
85	928	565.2	6,260	4,676	584	35	312	13.3	7.0	8.1	4.1
86	655	436.7	1,279	176	301	24	510	33.8	13.8	11.7	5.0
87	852	531.6	3,218	829	1,036	48	315	59.2	17.9	4.8	11.2
88	1,279	866.9	14,491	4,469	1,531	210	419	61.3	60.0	29.6	7.9
89	984	716.2	12,463	3,976	589	73	327	17.5	8.8	11.3	7.4
90	579	370.4	4,728	355	258	57	508	156.3	20.0	21.7	4.9
91	1,061	639.6	3,248	252	343	112	439	77.5	21.7	19.0	5.6
92	534	391.1	5,476	1,367	2,339	195	1,003	90.0	28.1	31.1	4.3
93	274	223.6	1,187	133	111	6[a]	130	10.6	5.0	7.4	4.1
94	508	330.2	3,692	1,248	429	90	834	128.8	62.9	-	-

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[a] An approximation based on cumulative geometric weekly means. Gompertz function could not be fit to the data as larvae were only collected during 2 weeks of sampling.

examine annual variability in these abundance indices (Table 27). The CVs of Niantic River adult and juvenile winter flounder abundance indices increased slightly over those given in NUSCO (1994a). Variability was lowest (CV = 61%) for the number of females spawning in the Niantic River and of associated egg production (57%). For the first three adult female age-classes, variability decreased from age-3 (98%) through age-5 (69%). This likely reflected variation in recruitment of year-classes as well as the varying numbers of immature ages-3 and 4 fish present in the river each year. Miller et al. (1991) noted that interannual variability of many flatfishes appeared to decrease with age. Small decreases (–1 to –5%) were noted for CVs of larval abundance indices from those reported last year. Variability among larval stages was greatest in Stage 2 larvae (104%). This was expected because much of the compensatory mortality is believed to occur during this stage of development. Annual variability in larval abundance decreased for the subsequent developmental stages, 3 (90%) and 4 (78%). The CV became relatively higher (91%) in age-0 young in late summer, but decreased again to 71% during fall and early winter after young left shallow inshore waters. Another increase in CV to 95% of age-1 juveniles in the Niantic River during the adult surveys was probably related to the previously discussed annual differences in distribution related to their behavior as well as from actual variation in year-class strength. Rothschild and DiNardo (1987) reported a median CV for recruitment indices of various marine fishes of 70%, although various flatfishes had CV values mostly less than 75%. The CV for abundance of European flounder decreased from 172% (n = 9) in the larval stage to 99% (n = 8) in newly settled young to 80% (n = 8, 12) for both young in September and at age-1 (van der Veer et al. 1991). As summarized by van der Veer (1986), the highest CV for yearly abundance estimates of different life stages of plaice in The Netherlands occurred during larval development in late winter (n = 4, CV = 95%) and at first settlement of pelagic juveniles in spring following larval metamorphosis and settling (9, 62%). Less variation was found in post-larval young during mid-summer (9, 30%) and age-2 recruits (9, 35%). He attributed the decline in variation of abundance of older juveniles to a density-dependent regulatory mechanism that operated during and shortly after larval settlement. Van der Veer (1986), van der Veer and Bergman (1987), and Bergman et al. (1988) noted that recruitment variability in plaice in The Netherlands was stabilized between years as a result of density-dependent regulatory processes (i.e., shrimp predation) on newly metamorphosed fish. The CVs for year-class strength of plaice in Swedish bays varied to a greater degree (67-118%), which was thought related to temperature effects during the larval stage and more variable crustacean predation on newly metamorphosed young in northern waters than in The Netherlands (Pihl 1990; Pihl and van der Veer 1992). Population regulation in flatfishes may be coarsely determined in earliest life history by variabile survival of eggs and larvae and then fine-tuned by density-dependent mortality of newly metamorphosed juveniles (van der Veer and Bergman 1987).

TABLE 27. Coefficients of variation (CV) for annual abundance indices[a] of various life stages of Niantic River winter flounder.

Life stage	Abundance index used	Number of observations	CV
Female spawners	Annual standardized catch	18	61%
Age-3 females	Annual standardized catch	16	98%
Age-4 females	Annual standardized catch	15	76%
Age-5 females	Annual standardized catch	14	69%
Eggs	Egg production index	18	57%
Stage 1 larvae	α parameter of Gompertz function	11	81%
Stage 2 larvae	α parameter of Gompertz function	12	104%
Stage 3 larvae	α parameter of Gompertz function	12	90%
Stage 4 larvae	α parameter of Gompertz function	12	78%
Age-0 young	Median CPUE at station LR (May-July)	12	81%
Age-0 young	Median CPUE at station LR (August-Sept)	12	91%
Age-0 young	Fall-winter Δ-mean at trawl stations	18	71%
Age-1 juveniles	Median CPUE of fish < 15 cm in Niantic River	18	95%

[a] Indices used correspond to those given on Table 26, except for age-3 through age-5 females.

Relationships among abundance indices of winter flounder for the same year-class are of interest for impact assessment. Knowledge of the earliest possible measure of relative year-class strength is desirable because it enables predictions of future recruitment to the adult stock, thus providing an early warning of decreases in stock abundance. If indices for all life-stages are assumed to be accurately and precisely measured each year, they should be correlated after applying appropriate time lags, except when processes such as density-dependent mortality or size-selective fishing result in a lack of colinearity between two consecutive life-stages. Female spawners and egg production were highly correlated, which was expected because calculation of the latter included female spawner abundance as part of the methodology of estimation (Table 28). Significant or near-significant correlations were found among most larval stage abundance indices. Niantic River Stage 4 larval abundance was also significantly correlated with age-0 juveniles during summer and late fall-early winter (Table 29). Age-0 juvenile abundance indices during summer and late fall-early winter were also correlated (Fig. 27). The abundance of larvae 7 mm and larger collected at MNPS (station EN) was not significantly correlated with any of the adult, larval, or juvenile abundance indices, except for age-1 juveniles taken in the Niantic River during the adult winter flounder surveys. However, the relationship between these two abundance indices was not clear and may be an artifact of the long time-series (Fig. 30). Also, the age-1 index was also not correlated with any of the other larval or juvenile indices.

If catch indices were assumed to be representative of annual relative abundances, Niantic River winter flounder were found to be fully recruited only at about age-5 (NUSCO 1990). Thus, age-3 or age-4 fish probably should not be used as an index of year-class strength because it is likely that only a fraction of these fish occur on the spawning grounds each year.

TABLE 28. Matrix of Spearman's rank-order correlations among various winter flounder spawning stock and larval abundance indices. Except for larvae 7 mm and larger taken at the MNPS intakes, all other indices refer to adults or larvae collected in the Niantic River.

Index[a]	Adult egg production	Stage 1 larvae	Stage 2 larvae	Stage 3 larvae	Stage 4 larvae	Larvae at MNPS intakes (≥7 mm)
Female spawners	0.9484[b] 0.0001 ** 18	0.5182 0.1025 NS 11	0.3776 0.2262 NS 12	0.2797 0.3786 NS 12	0.2797 0.3786 NS 12	0.1331 0.5985 NS 18
Adult egg production		0.5818 0.0604 NS 11	0.4336 0.1591 NS 12	0.4056 0.1908 NS 12	0.2797 0.3786 NS 12	0.0134 0.9579 NS 18
Stage 1 larvae			0.7909 0.0037 ** 11	0.6364 0.0353 * 11	0.6273 0.0388 * 11	-0.0273 0.9366 NS 11
Stage 2 larvae				0.7902 0.0022 ** 12	0.4406 0.1517 NS 12	-0.0420 0.8970 NS 12
Stage 3 larvae					0.5594 0.0586 NS 12	0.0629 0.8459 NS 12
Stage 4 larvae						0.3776 0.2262 NS 12

[a] Indices used correspond to those given on Table 26.

[b] The three statistics shown in each correlation matrix element are:
correlation coefficient (*r*),
probability of a larger *r* (NS - not significant [$p > 0.05$], * - significant at $p \leq 0.05$, ** - significant at $p \leq 0.01$), and
number of annual observations (sample size).

TABLE 29. Matrix of Spearman's rank-order correlations among various larval and juvenile winter flounder abundance indices.

Index[a]	Niantic River Stage 4 larvae	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles
MNPS intake larvae ($\geq$7 mm)	0.3776[b] 0.2262 NS 12	0.4615 0.1309 NS 12	0.3357 0.2861 NS 12	-0.1415 0.5754 NS 18	0.5349 0.0222 * 18
Niantic River Stage 4 larvae		0.5874 0.0446 * 12	0.7133 0.0092 ** 12	0.6287 0.0383 * 11	0.3189 0.3391 NS 11
Lower river early age-0 juveniles			0.8881 0.0001 ** 12	0.6333 0.0365 * 11	0.0775 0.8209 NS 11
Lower river late age-0 juveniles				0.6970 0.0171 * 11	0.0638 0.8522 NS 11
Fall-early winter river-bay age-0 juveniles					-0.4357 0.0707 NS 18

[a] Indices used correspond to those given on Table 26.

[b] The three statistics shown in each correlation matrix element are:
correlation coefficient (r),
probability of a larger r (NS - not significant [$p > 0.05$], * - significant at $p \leq 0.05$, ** - significant at $p \leq 0.01$), and
number of annual observations (sample size).

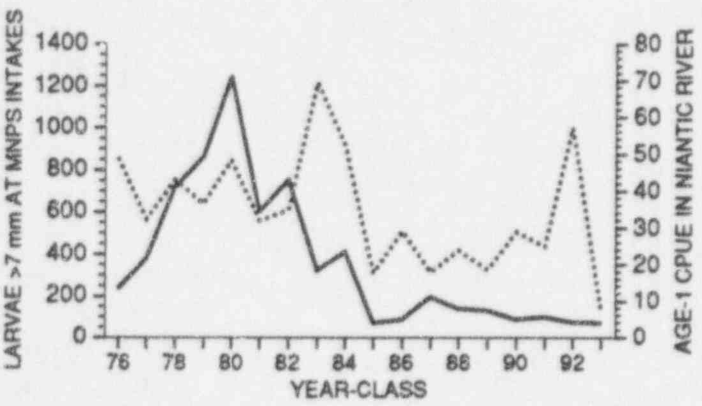

Fig. 30. Comparison between the α index of abundance of larval winter flounder 7 mm and larger taken at the MNPS intakes (dashed line) and the Niantic River (stations 1 and 2) spawning survey median CPUE (solid line) for the 1976-93 year-classes.

Furthermore, this fraction may vary from year to year. Several significant correlations were found between the abundance of female spawners and those for 7-mm and larger larvae at MNPS and also with some juvenile winter flounder indices (Table 30). The CPUE of age-1 fish taken in the river during the adult spawning surveys was significantly correlated with both age-3 and 4 female spawners. However, there was no correlation found with age-5 females. Significant negative correlations were found between both age-3 and age-4 females and the age-0 fall-winter Δ-mean CPUE, but the weakest of the negative correlations (not significant) was found between age-5 spawners and the fall-early winter juveniles, which should have been one of the most reliable as females are fully recruited by this age. If negative correlations persist in future years, they could be interpreted as an indication of unknown processes operating after winter flounder become age-1 that result in fewer adults being recruited in spite of larger numbers of juveniles. Possibilities include variable discard mortality of juveniles in the commercial fishery; high rates of fishing; and non-random fishing effort, which may occur in overfished stocks. Meanwhile, none of these life-stage indices can presently be used as a reliable measure of year-class strength.

TABLE 30. Matrix of Spearman's rank-order correlations among various winter flounder larval and female spawner abundance indices.

Index[a]	MNPS intake larvae (≥7 mm)	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles
Age-3 female spawners[b]	0.4088[c] 0.1159 NS 16	-0.2000 0.6059 NS 9	0.0167 0.9661 NS 9	-0.6107 0.0120 * 16	0.7382 0.0011 ** 16
Age-4 female spawners[b]	0.4357 0.1045 NS 15	-0.0952 0.8225 NS 8	-0.3095 0.4556 NS 8	-0.8954 0.0001 ** 15	0.5679 0.0272 * 15
Age-5 female spawners[b]	0.6659 0.0093 ** 14	-0.0357 0.9394 NS 7	-0.2143 0.6645 NS 7	-0.4841 0.0795 NS 14	0.2923 0.3105 NS 14

[a] Early life history indices used correspond to those given on Table 26.

[b] Determined by applying an age-length key (NUSCO 1[illegible]9) to the length distribution of annual standardized female abundances.

[c] The three statistics shown in each correlation matrix element are:
correlation coefficient (*r*),
probability of a larger *r* (NS - not significant [$p > 0.05$], * - significant at $p \leq 0.05$, ** - significant at $p \leq 0.01$), and
number of annual observations (sample size).

Stock-recruitment relationship (SRR)

Sampling-based estimates. Egg production estimates from annual spawning surveys were used to determine recruitment because the abundance of other early life-stages have not been reliably correlated with adult spawners. Both recruitment and the parental spawning stock indices were scaled to absolute population size as described previously (see Absolute abundance estimates, above). The resulting annual values were used with the Ricker SRR model as estimates of adult female spawning stock and potential female recruitment (Table 31). The addition of new catch data from the 1994 adult winter flounder survey resulted in some differences between present estimates of spawners and recruits reported in NUSCO (1994a). A two-parameter SRR model (Eq. 6) was initially fitted to the spawner and recruit data. The stock growth potential parameter α (scaled as numbers of fish) was estimated as 2.057, with a standard error of 0.699 (34% of the parameter value).

Two-parameter model estimates were used as initial values for fitting the three-parameter SRR model with temperature effects (Eq. 7). The three-parameter SRR explained 63% of the variability associated with the recruitment index. Relationships resulting from fits of both Ricker models are shown in the central portion of Figure 31 as follows: the unadjusted SRR (two-parameter model; Eq. 7) is shown as the broken-line curve and the three-parameter model (SRR adjusted for T_{Feb}) is represented by the solid-line. The outermost two dashed lines describe low recruitment in the warmest year (1990; $T_{Feb} = +1.67$) and high recruitment in the coldest year (1977; $T_{Feb} = -2.25$). For the three-parameter model, α was estimated at 2.071, with a standard error of 0.428, which is 21% of the parameter value (Table 32). Estimates of the α have ranged from 1.977 through 2.646 and those for ϕ from -0.412 through -0.259 (NUSCO 1990, 1991b, 1992a, 1993, 1994a). Variation in estimates of α could be caused by increasing fishing mortality rates on winter flounder in addition to the inherent instability of parameter estimates fitted to small data sets. In particular, the influence of the 1988-90 data points on the estimate of α were illustrative of higher recent exploitation (Fig. 31). Because of the relatively high abundance of juvenile winter flounder from the 1988 year-class, numbers of females were expected to increase in 1992 and 1993 and form the bulk of the spawning population. However, these recruitment indices were much below expected values, likely the result of high fishing mortality rates in recent years.

The estimate of Ricker's β parameter, which describes the annual rate of compensatory mortality as a function of the stock size is important in SPDM simulations. The parameter estimate of 2.498×10^{-5} has remained fairly constant (range of $2.140 - 2.523 \times 10^{-5}$; NUSCO 1990, 1991b, 1992a, 1993, 1994a).

TABLE 31. Annual Niantic River winter flounder stock-recruitment data based on indices of egg production for the 1977 through the 1990 year-classes with mean February water temperature and deviations (T_{Feb}) from the mean.

Year-class	Index of female spawners (P)[a]	Index of female recruits (R)[a]	R/P ratio	Mean February water temperature (°C)	Deviation from mean February water temperature (T_{Feb})
1977	20,097	65,125	3.24	0.36	-2.25
1978	36,544	45,750	1.25	1.09	-1.52
1979	27,262	36,758	1.35	1.48	-1.13
1980	21,608	28,861	1.34	2.38	-0.23
1981	70,441	28,076	0.40	2.63	0.02
1982	81,324	32,709	0.40	1.56	-1.05
1983	55,104	35,881	0.65	3.74	1.13
1984	25,544	27,195	1.06	4.02	1.41
1985	28,789	25,727	0.89	2.36	-0.25
1986	22,241	24,208	1.09	3.38	0.77
1987	27,076	21,077	0.78	3.28	0.67
1988	44,153	16,107	0.36	2.67	0.06
1989	36,478	10,271	0.28	3.24	0.63
1990	18,863	10,231	0.54	4.28	1.67
Mean	36,823	29,141	0.79	2.61	
CV	53%	49%		44%	

[a] Scaled number of female spawners and recruits from expected egg production; scaling factors used were 561,000 eggs per females and a multiplier of 28.571 to convert relative abundance to an absolute population size. Indices of female spawners and recruits differ from those reported in NUSCO (1994a) because of data added from the 1994 adult winter flounder population survey.

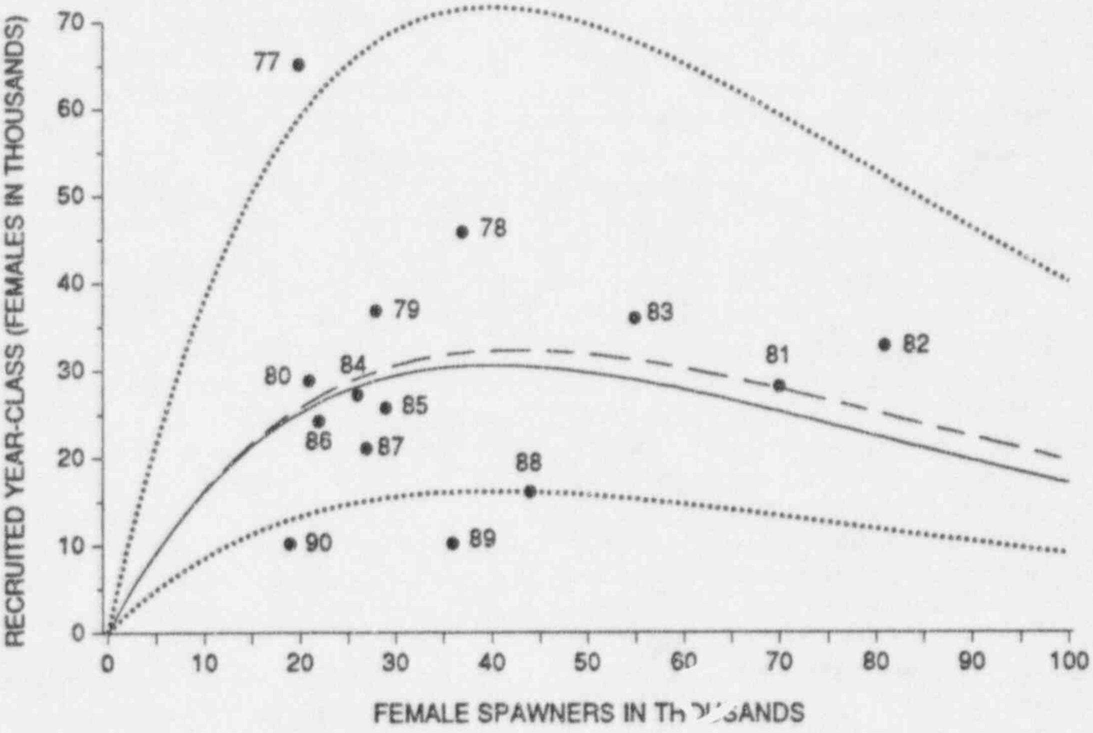

Fig. 31. The Ricker SRRs for Niantic River winter flounder (see text for explanation of the four curves plotted). Calculated recruitment indices for the 1977 through the 1990 year-classes are shown.

TABLE 32. Parameters of the Ricker stock-recruitment model fitted to data for Niantic River female winter flounder spawners from 1977 through 1990 and some derived points of reference.

Model parameters and reference points			
Model parameters (determined from numbers of fish):	Estimated value	Standard error	t^a
α_0 (compensatory reserve for unfished stock)	5.42	-	-
α (current compensatory reserve)	2.071[b]	0.428	4.84 **
β (stock-dependent compensatory rate)	2.498×10^{-5}	4.78×10^{-6}	5.22 **
ϕ (environmental [temperature] effect)	-0.379	0.077	4.89 **
Derived points of reference:		Numbers of fish	Biomass (lbs)
Unfished stock equilibrium size (P_{rep}; called maximum spawning potential by Howell et al. 1992)		67,658[c]	98,104
Present (through 1990) equilibrium size ($P_{E(F)}$)		29,145[c]	29,145
F for $P_{E(F)}$ = 29,145 female spawners		0.96	-
Estimate of critical stock size (25% of maximum spawning potential)		-	24,526

[a] t-statistic for parameter estimate $\neq 0$ with d.f. = n-3 = 11 (** - significant at $p \leq 0.01$).

[b] Includes the effects of recent exploitation rates.

[c] Average weight of female spawner for unfished stock is 1.45 lbs and for current exploited stock is 1 lb.

The parameter ϕ, which was an estimate of the effect of February temperature deviations (T_{Feb}) from the 1977-90 mean of 2.61°C, was -0.379. The values for ϕ are negative and although the relationship between winter flounder recruitment and February temperatures remain unknown, February coincides with most spawning, egg incubation, and hatching. These processes, as well as larval growth, are all temperature-dependent. Buckley et al. (1990) noted that the winter flounder reproductive process appears to have been optimized for cold winter temperatures followed by gradual spring warming. Adult acclimation temperatures and egg and larval incubation temperature affected larval size and biochemical composition. Cold winters and warm springs produced the largest larvae in the best condition at first feeding. This favored good survival and may partly explain the observed correlation between cold years and strong year-classes of winter flounder. Townsend and Cammen (1988) noted that the metabolic rates of pelagic consumers are more sensitive to lower temperature than rates of photosynthesis by phytoplankton, which bloom more in response to the amount of solar radiation received. Therefore, an earlier bloom in a cold year has the possibility of lasting longer before being grazed down by zooplankton. This allows for a greater contribution of organic matter to the benthos than in other years, benefiting juvenile demersal fishes that metamorphose just after the spring bloom of phytoplankton and have to outgrow various predators. As noted previously, the effect of temperature on potential prey or predators of larvae and newly metamorphosed juveniles, such as the sevenspine bay shrimp, may be an additional means for control of population abundance. Strong year-classes of plaice were also associated with cold winters, likely because the predatory brown shrimp (*Crangon crangon*) suffers high mortality or migrates out of plaice nursery areas (Zijlstra and Witte 1985; van der Veer 1986; Pihl 1990; Pihl and van der Veer 1992).

In addition to the above SRR parameters directly estimated from stock-recruitment data (Fig. 31), Table 32 includes four derived biological reference points. Ricker's stock-at-replacement, or P_{rep} (Eq. 11), was estimated at 67,658 female spawners and is the unfished equilibrium spawning stock size, also known as the maximum spawning potential (MSP). This reference point, expressed in units of biomass as 98,104 lbs, is the basis for the critical stock size (25% of MSP) below which the stock is considered overfished (Howell et al. 1992). The present equilibrium size $P_{E(F)}$ (Eq. 9) of 29,145 spawners refers to the sustainable or equilibrium size to which the stock could converge if present (through 1990) exploitation and other conditions remained unchanged. The calculated (Eq. 10) value of F that would achieve equilibrium stock size was 0.96, which is much higher than the DEP estimates of F for those years. This difference can mostly be due to the lack of age structure in Ricker's model, which causes fishing mortality to be concentrated in a single year for each year-class; winter flounder year-classes are exploited during many years. As mentioned previously in the Materials and Methods section, these reference points derived from fishery data are only deterministic approximations useful for comparative purposes across stocks and, in this study, to compare to the corresponding and more realistic values derived through simulation using SPDM.

Estimation of α for SPDM simulations. The above stock-recruitment-based estimates of α for the Niantic River winter flounder provided an underestimate of the true slope at the origin for this stock. The method of calculating annual recruitment included the effects of fishing on winter flounder age-2 and older as well as the entrainment of larvae at MNPS. Therefore, these direct estimates of α correspond to a compensatory reserve diminished by existing larval entrainment and exploitation rates. The concept of compensatory reserve in fishing stocks and the effect of exploitation on the shape of the reproduction curve when the recruitment index is based on the exploited stock was discussed by Goodyear (1977: Fig. 1). Thus, if larval entrainment and fishing rates increase, the field estimates of recruitment will be smaller and so will the estimates of α (i.e., the "remaining" compensatory reserve). To assess impacts appropriately, the inherent potential of a stock to increase in the absence of fishing and plant effects must be determined. Crecco and Howell (1990) investigated the possibility of using indirect methods to estimate the true α parameter (i.e., for the unfished stock when $F = 0$). They used four indirect methods (Cushing 1971; Cushing and Harris 1973; Longhurst 1983; Hoenig et al. 1987; Boudreau and Dickie 1989) based on different life history parameters. Because these methods do not depend upon direct estimates of recruitment, they avoid biases caused by changing fishing rates and provide independent means of validating SRR-based estimates. The present study used a Ricker SRR α parameter estimate derived from the value of 3.74 in biomass units reported by Crecco and Howell (1990: Table 2). This value was re-scaled for numbers of fish on the basis of the following relationship:

$$\alpha_{\text{biomass}} = \alpha \,/\, (\text{mean weight per mature female fish}) \qquad (19)$$

where the mean weight was calculated for a population at equilibrium and one for which only natural mortality was assumed to have occurred (i.e., the unfished population). A mean weight of 1.45 lbs per female spawner for the Niantic River unfished winter flounder stock was calculated (Table 33) using population data previously reported (NUSCO 1990). Using this mean weight, the re-scaled α parameter for this study was obtained as:

$$\alpha_0 = \alpha_{\text{biomass}} \cdot (\text{mean weight}) = 3.74 \cdot (1.45 \text{ lbs}) = 5.42 \qquad (20)$$

This parameter describes the inherent potential of a stock for increase because the natural logarithm of α is the slope of the SRR at the origin for the unfished stock (Ricker 1954) and that slope, in turn, corresponds to the intrinsic rate of natural increase of the population (Roughgarden 1979). Since the slope of the SRR at the origin decreases with increasing exploitation rates, it is useful to think of α as the "remaining growth potential" or "growth reserve" of the stock. Consequently, the large difference between the derived value of α (5.42) and direct regression estimates of α reflects respective differences in growth reserves between unfished and highly exploited stocks of winter flounder. Using an unfished stock as a starting point for simulations also has other advantages, depending upon the particular scenario selected. For example, simulation in this report includes initially moderate fishing rates that are much lower than those affecting the data on which the regression estimate of α was based. The data-based estimates of the other two SRR parameters (β and ϕ), which do not depend upon fishing and entrainment rates, were used in the population simulations as given in Table 32.

MNPS impact assessment

Larval entrainment

Estimates of larval entrainment at MNPS. The number of winter flounder larvae entrained in the condenser cooling water of MNPS is the most direct measure of potential impact on the Niantic River winter flounder stock. Annual totals of entrained larvae were related to larval densities in Niantic Bay and plant operations (i..e, cooling-water volume). Nearly all winter flounder larvae collected at station EN were taken from February through June, with most (> 90%) found in April and May. The entrainment estimate for 1994 (182.1 million) was about average since three-unit operations began in 1986 (Table 34). From January 15 through May 23

TABLE 33. Biomass calculations for the Niantic River winter flounder female spawning stock at equilibrium based on an instantaneous natural mortality rate of M = 0.35 and an instantaneous fishing mortality rate of F = 0 (virgin stock).

Age	Female population size	Fraction mature	Number of mature females	Weight of mature females (lbs per fish)	Eggs per mature female	Spawning stock biomass (lbs)	Egg production (millions)
2	1,000.00	0.00	0.00	-	-	-	0.000
3	704.69	0.08	56.38	0.554	223,735	31.23	12.613
4	496.59	0.36	178.77	0.811	378,584	144.98	67.680
5	349.94	0.92	321.94	1.088	568,243	350.27	182.942
6	246.60	1.00	246.60	1.377	785,897	339.56	193.800
7	173.77	1.00	173.77	1.645	1,004,776	285.86	174.604
8	122.46	1.00	122.46	1.873	1,201,125	229.36	147.086
9	86.29	1.00	86.29	2.057	1,366,951	177.51	117.959
10	60.81	1.00	60.81	2.203	1,502,557	133.97	91.371
11	42.85	1.00	42.85	2.304	1,598,597	98.73	68.503
12	30.20	1.00	30.20	2.390	1,682,208	72.17	50.798
13	21.28	1.00	21.28	2.461	1,754,800	52.37	37.342
14	15.00	1.00	15.00	2.516	1,809,000	37.73	27.127
15	10.57	1.00	10.57	2.552	1,845,800	26.97	19.505
Total	3,361.05		1,366.91			1,980.71	1,191.329

Mean weight per mature female fish = (1,980 lbs + 1,367 mature females) = 1.45 lbs (37.6 cm fish)
Mean fecundity (virgin stock) = 871,548 eggs per female spawner

during the winter flounder larval season Unit 1 was not operating due to a refueling outage (Fig. 32). The decrease in cooling water usage resulted in a calculated reduction in larval entrainment of about 21% (48.1 million larvae) from that expected if Unit 1 had operated during this period. This estimated entrainment reduction was higher than the 14% reported in NUSCO (1994b), which was based on a long-term

TABLE 34. Annual abundance index (α parameter of the Gompertz function) with 95% confidence interval of winter flounder larvae in entrainment samples and total annual entrainment estimates during the larval season of occurrence, and the volume of seawater entrained at MNPS each year from 1976 through 1994 during an 136-day period from February 15 through June 30.

Year	α parameter	Standard error	95% confidence interval	Number entrained ($X 10^6$)	Seawater volume entrained ($m^3 X 10^6$)
1976	1,656	32	1,588 - 1,724	107.6	662.8
1977	751	47	650 - 852	31.2	585.6
1978	1,947	352	1,186 - 2,706[a]	87.4	490.9
1979	1,296	81	1,121 - 1,470	47.7	474.1
1980	2,553	37	2,475 - 2,632	175.7	633.3
1981	1,163	23	1,113 - 1,213	47.7	455.2
1982	2,259	36	2,184 - 2,334	170.4	674.1
1983	2,966	21	2,921 - 3,012	219.3	648.0
1984	1,840	47	1,741 - 1,939	88.1	573.8
1985	1,585	48	1,483 - 1,686	83.3	528.1
1986	903	31	837 - 968	130.6	1,353.4
1987	1,194	23	1,145 - 1,242	172.0	1,323.6
1988	1,404	42	1,315 - 1,493	193.3	1,381.7
1989	1,677	13	1,650 - 1,704	175.0	1,045.9
1990	1,073	25	1,021 - 1,125	138.8	1,302.7
1991	1,149	18	1,110 - 1,189	121.3	934.4
1992	3,974	76	3,812 - 4,136	513.9	1,199.3
1993	328	23	280 - 377	45.1	1,412.3
1994	1,709	38	1,626 - 1,790	182.1	1,174.6

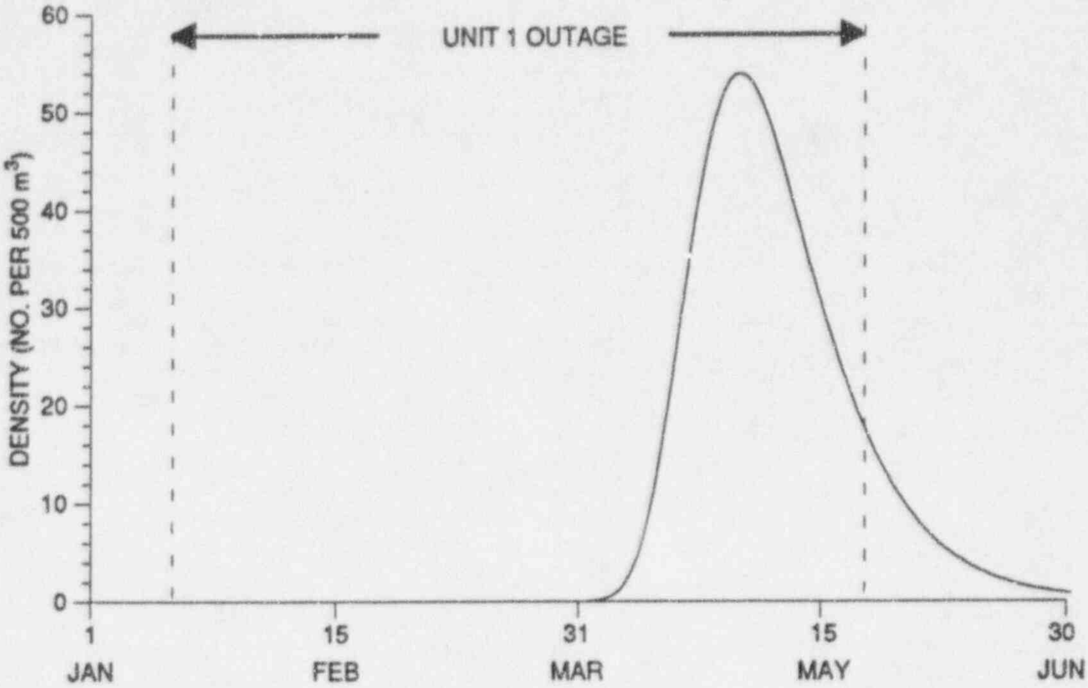

Fig. 32. Abundance curve for entrained winter flounder larvae in relation to the period of Unit 1 refueling outage during 1994.

(1976-91) average annual abundance distribution of entrained larval winter flounder.

As in previous years, Stage 3 larvae predominated in entrainment collections. In 1994, the percentages of each developmental stage entrained were 4% for Stage 1, 17% for Stage 2, 66% for Stage 3, and 13% for Stage 4. These proportions were similar to findings in previous years. Percentages for 1983-93 combined were 3% for Stage 1, 22% for Stage 2, 64% for Stage 3, and 11% for Stage 4 of development.

Effect of entrainment on a year-class. To determine the effect of winter flounder entrainment on a year-class, the relationship between entrainment estimates and various indices of juvenile abundance were examined. Annual entrainment estimates were significantly correlated with two abundance indices of juvenile winter flounder (Table 35). These were for the age-0 fish taken in late summer at station LR and during late fall-early winter at the TMP stations. The abundance of young at LR in early summer, which immediately follows the larval entrainment season, was not significantly correlated with estimated entrainment. Although significant, the form of the relationships between the entrainment estimates and these two age-0 abundance indices was not obvious (Fig. 33). Densities of winter flounder larvae 7 mm and larger taken at EN (an index of abundance rather than one of estimated impact) were not significantly correlated with the age-0 abundance indices (Table 29). Also, a first-year survival rate index (CPUE of age-1 fish collected in the Niantic River during the spawning surveys divided by the 7-mm larval index) was not significantly correlated with entrainment estimates. Furthermore, the significant correlation coefficients found were positive, implying no apparent entrainment effect. In general, even negative correlations between annual entrainment and abundance of early life history stages do not necessarily imply an entrainment impact unless positive correlations can be found between those early life history stages and mature female fish.

Mass-balance calculations. The magnitude of the impact of entrainment on the Niantic River winter flounder stock depends upon how many of the entrained larvae originated from this stock. Hydrodynamic modeling (NUSCO 1976) and current drogue studies (NUSCO 1992b) showed that much of the condenser cooling-water used by MNPS enters Niantic

TABLE 35. Spearman's rank-order correlations between the annual estimates of larval winter flounder entrainment at MNPS and the abundance indices of several post-entrainment early life history stages.

Index[a]	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles	Apparent larval survival rate
Annual estimate of entrainment	0.5245[b] 0.0800 NS 12	0.6573 0.0202 * 12	0.5473 0.0187 * 18	-0.0796 0.7537 NS 18	-0.3356 0.1734 NS 18

[a] Indices used correspond to those given on Table 26, except for the apparent survival rate, which is the age-1 index divided by the index of 7 mm and larger larvae at EN in Niantic Bay.

[b] The three statistics shown in each correlation matrix element are:
correlation coefficient (*r*),
probability of a larger *r* (NS - not significant [$p > 0.05$], * - significant at $p \leq 0.05$), and
number of annual observations (sample size).

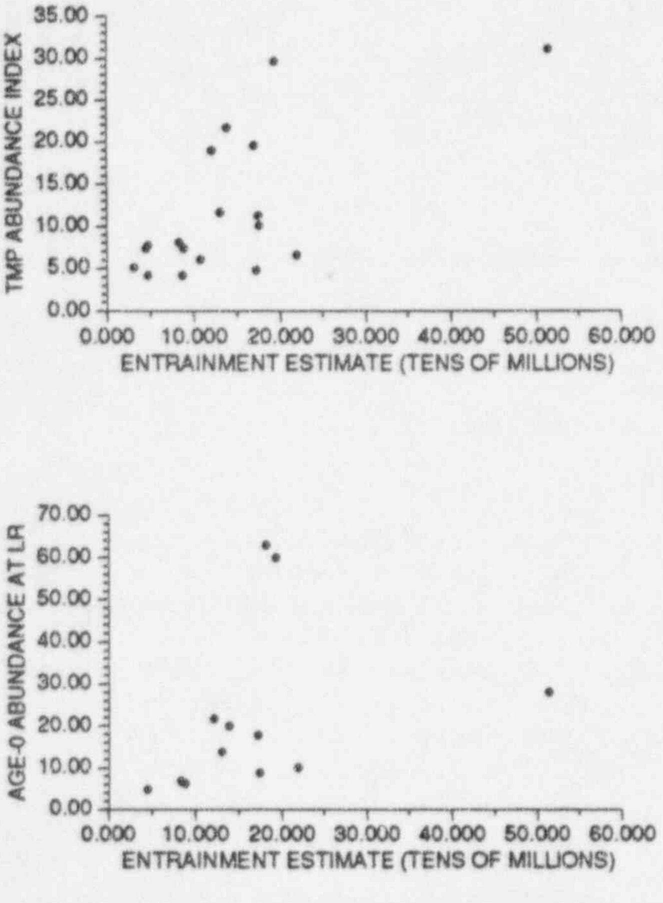

Fig. 33. Relationship between the annual entrainment estimate of winter flounder larvae at MNPS and the late fall-early winter seasonal Δ-mean CPUE of age-0 winter flounder from all trawl monitoring program stations (TMP) for the 1976-93 year-classes, and between the entrainment estimate and the median CPUE of age-0 winter flounder taken at station LR in the Niantic River during late summer for the 1983-93 year-classes.

Bay from LIS. Other stocks are known to spawn both to the east and west of the bay and results from tidal studies also indicated that a large number of winter flounder larvae enter Niantic Bay from LIS (NUSCO 1992a, 1992b). To determine if the number of winter flounder larvae leaving the Niantic River could support the number of larvae observed in the bay each year, mass-balance calculations were made for 1984 through 1994; nine of these years (1986-94) occurred during three-unit operation. The results of each 5-day period in 1994 are provided as an example of these calculations (Table 36). Results for other years were given in NUSCO (1993, 1994a).

During the 1994 larval season, the sign of the term *5-day change* shifted from positive to negative when the estimated number of larvae in Niantic Bay started to decline during a 5-day period beginning on April 21 (Table 36). Also, in early April the sign of the *Source or Sink* term changed from negative to positive. A negative *Source/Sink* term indicated a net loss of larvae from Niantic Bay during the first part of the larval season. During the 5-day period starting on about April 1, the *Source or Sink* term became positive, an indication that larvae from other sources (i.e., LIS) were required to support the change in larval abundance and balance the equation. Timing of this change in the *Source or Sink* term in 1994 was similar to previous years (NUSCO 1993), except for an earlier date (February 25) in 1993 (NUSCO 1994a). During peak entrainment (April and May), fewer larvae were entrained than were imported from LIS, indicating that sources other than the Niantic River provided larvae found in Niantic Bay.

During each 5-day period the proportion of entrained larvae from the Niantic River was estimated from the

TABLE 36. Results of mass-balance calculations for each 5-day period in 1994.

Start of 5-day period	*5-day change* (X 10^6)	Number entrained (*Ent*) (X 10^6)	Loss due to mortality (*Mort*) (X 10^6)	Number from the Niantic River (*FromNR*) (X 10^6)	Number to the Niantic River (*ToNR*) (X 10^6)	*Source or Sink* (X 10^6)
2-15	0.0[a]	0.0	0.0	0.6	0.0	-0.6
2-20	0.0	0.0	0.0	0.9	0.0	-0.9
2-25	0.0	0.0	0.0	2.1	0.0	-2.1
3-02	0.0	0.0	0.0	4.7	0.0	-4.7
3-07	0.0	0.0	0.0	9.3	0.0	-9.3
3-12	0.1	0.0	0.0	15.6	6.7	-8.7
3-17	1.1	0.0	0.1	22.3	7.1	-14.0
3-22	4.3	0.0	0.8	28.2	9.4	-13.6
3-27	10.1	0.0	2.8	32.2	16.5	-2.8
4-01	15.0	0.1	4.3	34.0	29.8	15.1
4-06	15.3	1.0	11.2	33.8	46.7	40.3
4-11	10.7	5.1	15.3	31.9	61.6	60.8
4-16	4.0	15.2	14.9	29.0	70.3	75.4
4-21	-2.1	25.3	10.3	25.6	71.8	79.7
4-26	-6.2	28.3	8.9	22.1	67.6	76.5
5-01	-8.0	25.5	5.1	18.7	59.9	63.7
5-06	-8.3	19.3	4.1	15.6	51.0	50.5
5-11	-7.6	14.8	2.8	12.9	42.3	39.5
5-16	-6.5	12.6	2.1	10.6	34.3	32.2
5-21	-5.2	11.1	1.4	8.6	28.1	26.7
5-26	-4.1	8.2	1.3	7.0	22.9	21.2
5-31	-3.2	5.6	0.9	5.7	18.8	16.5
6-05	-2.4	3.9	0.6	4.6	15.7	13.2
6-10	-1.8	2.6	0.4	3.8	13.4	10.8
6-15	-1.4	1.6	0.3	3.1	11.6	9.1
6-20	-1.0	1.2	0.2	2.6	10.3	8.2
6-25	-0.7	0.8	0.2	2.1	9.3	7.4

[a] Due to rounding, any zero value represents less than 50,000 larvae.

ratio of larvae entering the bay from the river (*FromNR*) to the total input from both sources (*FromNR + Source or Sink*). This proportion was applied to the total number entrained in that 5-day period to estimate the number entrained from the Niantic River. During 5-day periods when there was a net loss (negative *Source or Sink* term) or when the proportion from the river was greater than one, all larvae entrained were assumed to have originated from the Niantic River. This estimate was conservative, because the results of a dye study and larval dispersal modelling (Dimou and Adams 1989) showed that only about 20% of the water discharged from the Niantic River passed through MNPS during full three-unit operation. Estimates of annual total entrainment and the annual number entrained from the Niantic River were determined by summing over all 5-day periods. Based on mass-balance calculations for data collected in 1984-94, about 14 to 38% of winter flounder larvae entrained by MNPS originated from the Niantic River (Table 37). For 1994, the estimated number of larvae entrained that originated from the river was about average compared to previous years.

The potential impact of larval entrainment on the population depends upon the age of each larva at the time it is entrained. Older individuals have a greater probability to contribute to year-class strength than younger ones. Therefore, the estimated number of each developmental stage entrained during each 5-day period was based on the proportion of each stage collected at station EN. By applying the proportion of entrainment attributed to the Niantic River (*FromNR / [FromNR + Source or Sink]*), the number of larvae in each stage was allocated to each of the two sources (Niantic River or other) for every 5-day period. The annual number of each larval stage entrained from each source was estimated by summing over all 5-day periods (Table 38). Most of the Stage 3 larvae (the

TABLE 37. Estimates of total number of larval winter flounder entrained, number of larvae entrained from the Niantic River, and the percentage of total entrainment attributed to the Niantic River for 1984-94.

Year	Total entrainment ($X 10^6$)	Niantic River larval entrainment ($X 10^6$)[a]	% entrainment attributed to the Niantic River
1984	88.1	33.1	37.6
1985	83.3	28.8	34.6
1986	130.6	28.9	22.1
1987	172.0	42.8	24.9
1988	193.3	40.8	21.1
1989	175.0	34.5	19.7
1990	138.8	39.7	28.6
1991	121.3	36.3	29.9
1992	513.9	82.5	16.1
1993	45.1	6.2	13.7
1994	182.1	44.8	24.6

[a] A slight increase than previously reported (NUSCO 1994a) due to a correction in computer coding in estimating the term *ToNR*.

TABLE 38 Estimated number of winter flounder larvae entrained at MNPS by developmental stage from the Niantic River and other sources, based on mass-balance calculations for 1984 through 1994.

Year	Source	Stage 1 ($X 10^6$)	Stage 2 ($X 10^6$)	Stage 3 ($X 10^6$)	Stage 4 ($X 10^6$)
1984	Niantic River	0.2	15.4	14.4	3.2
	Other	0.1	25.2	25.9	3.7
1985	Niantic River	3.5	17.9	7.1	0.4
	Other	0.8	11.1	35.9	6.7
1986	Niantic River	0.7	7.7	15.9	4.5
	Other	1.5	25.6	63.1	11.4
1987	Niantic River	0.8	15.6	24.5	1.9
	Other	0.6	31.5	89.1	7.9
1988	Niantic River	4.1	9.8	25.3	1.6
	Other	1.2	8.1	119.4	23.9
1989	Niantic River	2.9	11.5	19.7	0.5
	Other	4.3	42.4	85.0	8.8
1990	Niantic River	1.0	6.4	28.5	3.8
	Other	0.9	12.8	76.1	9.4
1991	Niantic River	0.3	3.7	27.5	4.9
	Other	0.7	9.2	68.5	6.7
1992	Niantic River	5.8	10.4	57.2	9.0
	Other	31.4	56.5	308.8	34.6
1993	Niantic River	0.3	1.2	4.1	0.5
	Other	1.3	5.4	24.2	8.1
1994	Niantic River	1.7	9.8	27.4	5.9
	Other	3.9	28.5	87.0	17.9

predominant stage entrained) originated from sources other than the Niantic River. As mentioned previously, some larger larvae from other areas may have entered the Niantic River during a flood tide and caused the increased frequency noted in larger size-classes (Fig. 16). Results from a special bay-wide sampling during 1991 (NUSCO 1992a) showed that in April and May, when about 75% of Stage 3 larvae were entrained, more larvae entered Niantic Bay from LIS east of Millstone Point and passed by the MNPS intakes during a flood tide than were flushed out of the bay to LIS during an ebb tide. Therefore, greater densities of Stage 3 larvae were expected at station EN during a flood than an ebb tide. This was confirmed in NUSCO (1993), where significantly ($p \leq 0.05$) greater Stage 3 densities found in April and May from 1983 through 1992 at station EN were from collections made during flood tides as compared to ebb tides.

Estimated production loss from the Niantic River stock. Estimates of larvae entrained by stage from the river were compared to annual abundance estimates for each larval stage in the Niantic River to determine the percentage of production loss from the Niantic River stock (Table 39). Estimates of Niantic River Stage 1 larvae entrained were calculated from daily abundance estimates (Eq. 3) at station C, following an evaluation presented in NUSCO (1993). This study indicated that entrainment sampling may underestimate Stage 1 larval abundance because of net extrusion. Based on dye studies (Dimou and Adams 1989), 20% of the Niantic River discharge passes through MNPS during full three-unit operation. Therefore, for full three-unit operation, 20% of the daily density of Stage 1 larvae at station C were used as an estimate of Stage 1 larval entrainment from the Niantic River. During periods of reduced plant operation, estimates were proportionally reduced based on daily water volume use. Entrainment estimates for Niantic River Stages 2, 3, and 4 larvae were from the results of mass-balance calculations, which used entrainment sampling densities. The estimated percentage of the Niantic River winter flounder production entrained annually since 1984 ranged from 4.7 to 22.0% and had a geometric mean of 9.2%. In 1984-93, production losses were slightly higher than reported in NUSCO (1994a) due to a computer coding error, which was corrected and the estimates recomputed. Based on several special studies (NUSCO 1992a, 1992b) and the empirical mass-balance calculations, a large number of larvae entrained at MNPS likely came from areas other than the Niantic River.

The above mass-balance calculations were based on actual daily condenser cooling-water volumes. To determine the production loss for projected full (100% capacity) three-unit operations, these calculations were recomputed based on a maximum daily condenser cooling-water volume of 11.1 million $m^3 \cdot day^{-1}$ (Table 39). To increase the time-series, three-unit operation was simulated to include 1984 and 1985, prior to Unit 3 start-up. Estimated annual percentages of the Niantic River winter flounder production that would have been entrained since 1984 under simulated three-unit operation ranged from 5.4 to 25.0% (geometric mean = 12.1%). These estimated annual reductions in year-class strength were used in impact assessment simulations with the SPDM as described below.

Stochastic simulation of the Niantic River winter flounder stock

Model simulation of MNPS impact. The initial input data used to run the SPDM were described in the Materials and Methods section (Tables 1-5; Figs. 4-6). Simulations were made from 1960, a decade before Unit 1 went online, until 2060, 35 years after Unit 3 is scheduled to be retired. The model accessed a secondary input file, which included fishing (plus impingement mortality) rates and the larval entrainment losses (i.e., % Niantic River annual larval production loss) assumed for each year of the simulation (Table 40). The combined mortality of fishing (F) and impingement (IMP) was used only during the years 1971-2025, corresponding to actual or expected MNPS operation. Rates of larval entrainment (ENT) during 1971-94 were based on known rates of MNPS cooling-water flow and calculated entrainment of Niantic River winter flounder larvae as derived from the mass-balance calculations discussed above. Entrainment rates from 1995 through 2025 were estimated by randomly selecting values of ENT from the historic time-series for full three-unit operation (Table 39) and of cooling-water flow for each unit (Table 5), which also depended upon a unit retirement schedule (Table 1). ENT was then adjusted to account for historically varying flows by unit during the larval winter flounder season. In this process, it was assumed that MNPS would operate during a larval winter flounder season in the future as it had in the past. Calculated annual values of larval production loss used for simulations in this report are given in Table 40.

TABLE 39. Estimated abundance of winter flounder larvae in the Niantic River and the number and percentage of the production entrained from the Niantic River by developmental stage for 1984-94. Numbers of larvae from the Niantic River were based on the most recent mass-balance calculations.

			Actual MNPS operating conditions:		Projected full[d] MNPS three-unit operating conditions:	
Year	Stage of development	Niantic River abundance[a] (X 10^6)	Entrainment from the Niantic River[b] (X 10^6)	% of the production[c]	Entrainment from the Niantic River (X 10^6)	% of the production[c]
1984	Stage 1	2864	10.3	0.4	22.6	0.8
	Stage 2	685	15.4	2.2	32.7	4.8
	Stage 3	337	14.4	4.3	34.9	10.4
	Stage 4	235	3.2	1.4	9.0	3.8
	Total		43.3	**8.2**	99.2	**19.7**
1985	Stage 1	3228	15.6	0.5	44.2	1.4
	Stage 2	773	17.9	2.3	44.7	5.8
	Stage 3	380	7.1	1.9	15.2	4.0
	Stage 4	265	0.4	0.2	1.0	0.4
	Total		41.0	**4.8**	105.1	**11.5**
1986	Stage 1	2494	11.6	0.5	14.4	0.6
	Stage 2	700	7.7	1.1	8.5	1.2
	Stage 3	366	15.9	4.3	15.7	4.3
	Stage 4	255	4.5	1.8	5.1	2.0
	Total		39.7	7.7	43.7	**8.1**
1987	Stage 1	3036	34.4	1.1	39.8	1.3
	Stage 2	853	15.6	1.8	18.1	2.1
	Stage 3	445	24.5	5.5	25.4	5.7
	Stage 4	311	1.9	0.6	2.0	0.6
	Total			**9.1**	85.3	**9.8**
1988	Stage 1	4951	83.7	1.7	92.1	1.9
	Stage 2	741	9.8	1.3	10.3	1.4
	Stage 3	267	25.3	9.5	27.1	10.1
	Stage 4	192	1.6	0.8	1.7	0.9
	Total		120.4	**13.3**	131.2	**14.3**
1989	Stage 1	4091	66.5	1.6	84.3	2.1
	Stage 2	570	11.5	2.0	14.3	2.5
	Stage 3	188	19.7	10.5	24.1	12.8
	Stage 4	126	0.5	0.4	0.7	0.6
	Total		98.2	**14.5**	123.4	**17.9**
1990	Stage 1	2115	33.2	1.6	36.7	1.7
	Stage 2	869	6.4	0.7	7.6	0.9
	Stage 3	239	28.5	11.9	32.4	13.6
	Stage 4	206	3.8	1.8	4.3	2.1
	Total		71.9	**16.1**	81.0	**18.2**
1991	Stage 1	3653	8.0	0.2	13.0	0.4
	Stage 2	2549	3.7	0.1	5.2	0.2
	Stage 3	775	27.5	3.5	36.2	4.7
	Stage 4	628	4.9	0.8	6.4	1.0
	Total		44.1	**4.7**	60.8	**6.2**

TABLE 39. (cont.).

			Actual MNPS operating conditions:		Projected full[d] MNPS three-unit operating conditions:	
Year	Stage of development	Niantic River abundance[a] (X 10^6)	Entrainment from the Niantic River[b] (X 10^6)	% of the production[c]	Entrainment from the Niantic River (X 10^6)	% of the production
1992	Stage 1	2234	23.0	1.0	28.6	1.3
	Stage 2	936	10.4	1.1	11.8	1.3
	Stage 3	344	57.2	16.6	64.4	18.7
	Stage 4	276	9.0	3.3	10.5	3.8
	Total		99.6	**22.0**	115.3	**25.0**
1993	Stage 1	1277	11.7	0.9	13.3	1.0
	Stage 2	660	1.2	0.2	1.3	0.2
	Stage 3	119	4.1	3.4	4.2	3.5
	Stage 4	83	0.5	0.6	0.5	0.6
	Total		17.5	**5.1**	19.3	**5.4**
1994	Stage 1	1886	27.4	1.5	36.1	1.9
	Stage 2	1035	9.8	0.9	12.5	1.2
	Stage 3	519	27.4	5.3	35.7	6.9
	Stage 4	451	5.9	1.3	7.0	1.6
	Total		70.5	**9.0**	91.3	**11.5**
Geometric mean				**9.2**		**12.1**

[a] Abundance estimates for 1984-89 were from Crecco and Howell (1990) and those for 1990-94 were calculated by NUSCO staff.

[b] Entrainment estimates attributed to the Niantic River are higher than those in Table 37 due to adjustments made for Stage 1 entrainment.

[c] A slight increase than previously reported (NUSCO 1994a) due to a correction in computer coding in estimating the term *ToNR*.

[d] Although only MNPS Units 1 and 2 operated in 1984 and 1985, the projected values assume full three-unit operation for all years.

Expected changes in the values of F over time were determined after consultation with DEP Marine Fisheries (V. Crecco and P. Howell, CT DEP, Old Lyme, CT, pers. comm.) and reflect recent changes in regulations to considerably reduce F (Table 2). Nominal fishing mortality rates were initially set at F = 0.40, remained unchanged through the 1960s, increased gradually to 0.62 in 1988, and thereafter increased more rapidly to a maximum rate of 1.30 in 1991 (Table 40; Fig. 5). Note that the rates in Table 40 included an additional mortality of F = 0.01 that accounted for fish impingement during the years of MNPS operation (NUSCO 1992a). After 1991, and as a result of implemented or proposed regulatory changes to the fisheries, F was projected to decrease substantially through the late 1990s, reach a low of 0.50 in 2001, and remain unchanged throughout the rest of the simulation time period.

Simulation results. Three stochastic time-series of female spawning stock sizes were generated by SPDM simulation runs: a theoretical unfished stock, whose size was dependent only upon the dynamics of winter flounder reproduction and environmental variability; a baseline stock, whose size was affected by rates of fishing in addition to the above; and an impacted stock, which further added the effects of MNPS entrainment and impingement to those of fishing and natural variation. Because the baseline stock projections include fishing but no power plant effects, this time-series was used as the reference against which the impacted stock projections were compared. Therefore, the baseline needs to be a fair representation of past and projected trends of Niantic River winter flounder abundance. The unfished stock size used initially in the simulations was 98,104 lbs (value of P_{rep}; equivalent to 67,658 female spawners), which was based on the age and size structure of female winter flounder (Tables 5, 32, and 33). The initial stock size represents the maximum spawning potential (MSP) for the unfished Niantic River female spawning stock. The geometric mean estimate of MSP from the SPDM was 98,046 lbs, remarkably similar to the deterministic estimate of P_{rep} used to initiate the simulations. The stochastic mean size of the exploited stock by 1970 (under the starting nominal fishing rate of F = 0.40) was quickly

TABLE 40. Schedule of conditional entrainment (ENT values), fishing (F) mortalities with adjustments for impingement (IMP), and fishing discard mortalites as implemented in the 1994 SPDM simulations.

Time step	Simulation year	% of year-class reduction based on calculated or simulated levels of ENT[a]	Nominal F (plus IMP)[b]	Fractional fishing discard F for : Age-1	Age-2	Age-3	Age-4
0	1960	0.0	0.40	0.0360	0.2400	0.4000	0.4000
1	1961	0.0	0.40	0.0360	0.2400	0.4000	0.4000
2	1962	0.0	0.40	0.0360	0.2400	0.4000	0.4000
3	1963	0.0	0.40	0.0360	0.2400	0.4000	0.4000
4	1964	0.0	0.40	0.0360	0.2400	0.4000	0.4000
5	1965	0.0	0.40	0.0360	0.2400	0.4000	0.4000
6	1966	0.0	0.40	0.0360	0.2400	0.4000	0.4000
7	1967	0.0	0.40	0.0360	0.2400	0.4000	0.4000
8	1968	0.0	0.40	0.0360	0.2400	0.4000	0.4000
9	1969	0.0	0.40	0.0360	0.2400	0.4000	0.4000
10	1970	0.0	0.40	0.0360	0.2400	0.4000	0.4000
11	1971	0.1530 X ENT = 1.239	0.41	0.0390	0.2440	0.4040	0.4040
12	1972	0.2262 X ENT = 1.402	0.42	0.0399	0.2500	0.4140	0.4140
13	1973	0.0767 X ENT = 0.882	0.43	0.0408	0.2560	0.4240	0.4240
14	1974	0.1895 X ENT = 1.175	0.44	0.0417	0.2620	0.4340	0.4340
15	1975	0.2262 X ENT = 4.456	0.45	0.0426	0.2680	0.4440	0.4440
16	1976	0.4421 X ENT = 11.053	0.46	0.0435	0.2740	0.4540	0.4540
17	1977	0.4232 X ENT = 4.147	0.47	0.0444	0.2800	0.4640	0.4640
18	1978	0.3018 X ENT = 3.471	0.48	0.0453	0.2860	0.4740	0.4740
19	1979	0.3133 X ENT = 3.603	0.49	0.0462	0.2920	0.4840	0.4840
20	1980	0.4810 X ENT = 5.532	0.51	0.0480	0.3040	0.5040	0.5040
21	1981	0.2873 X ENT = 1.781	0.53	0.0498	0.3160	0.5240	0.5240
22	1982	0.4857 X ENT = 6.946	0.56	0.0360	0.2735	0.5540	0.5540
23	1983	0.4675 X ENT = 8.509	0.58	0.0372	0.2491	0.5740	0.5740
24	1984	8.2	0.60	0.0384	0.2577	0.5940	0.5940
25	1985	4.8	0.61	0.0390	0.2200	0.6040	0.6040
26	1986	7.7	0.63	0.0402	0.2272	0.6240	0.6240
27	1987	9.1	0.64	0.0408	0.2308	0.6340	0.6340
28	1988	13.3	0.63	0.0402	0.1156	0.6054	0.6240
29	1989	14.5	0.86	0.0540	0.1570	0.8285	0.8540
30	1990	16.1	1.04	0.0648	0.1894	1.0031	1.0340
31	1991	4.7	1.31	0.0810	0.2380	1.2650	1.3040
32	1992	22.0	1.18	0.0732	0.2146	1.1389	1.1740
33	1993	5.1	1.18	0.0732	0.1210	0.9634	1.1272
34	1994	9.0	1.00	0.0624	0.0733	0.5485	0.8950
35	1995	U1, U2, U3 flow X ENT = 13.622	0.77	0.0486	0.0572	0.4220	0.6880
36	1996	U1, U2, U3 flow X ENT = 4.532	0.75	0.0474	0.0558	0.4110	0.6700
37	1997	U1, U2, U3 flow X ENT = 3.110	0.70	0.0444	0.0523	0.3835	0.6250
38	1998	U1, U2, U3 flow X ENT = 15.961	0.65	0.0414	0.0488	0.3560	0.5800
39	1999	U1, U2, U3 flow X ENT = 14.992	0.60	0.0384	0.0453	0.3285	0.5350
40	2000	U1, U2, U3 flow X ENT = 8.016	0.55	0.0354	0.0418	0.3010	0.4900
41	2001	U1, U2, U3 flow X ENT = 6.707	0.51	0.0330	0.0390	0.2790	0.4540
42	2002	U1, U2, U3 flow X ENT = 4.450	0.51	0.0330	0.0390	0.2790	0.4540
43	2003	U1, U2, U3 flow X ENT = 12.370	0.51	0.0330	0.0390	0.2790	0.4540
44	2004	U1, U2, U3 flow X ENT = 11.757	0.51	0.0330	0.0390	0.2790	0.4540
45	2005	U1, U2, U3 flow X ENT = 19.371	0.51	0.0330	0.0390	0.2790	0.4540
46	2006	U1, U2, U3 flow X ENT = 8.673	0.51	0.0330	0.0390	0.2790	0.4540
47	2007	U1, U2, U3 flow X ENT = 6.934	0.51	0.0330	0.0390	0.2790	0.4540
48	2008	U1, U2, U3 flow X ENT = 8.506	0.51	0.0330	0.0390	0.2790	0.4540
49	2009	U1, U2, U3 flow X ENT = 10.477	0.51	0.0330	0.0390	0.2790	0.4540
50	2010	U1, U2, U3 flow X ENT = 16.898	0.51	0.0330	0.0390	0.2790	0.4540
51	2011	U2, U3 flow X ENT = 7.648	0.51	0.0330	0.0390	0.2790	0.4540
52	2012	U2, U3 flow X ENT = 9.464	0.51	0.0330	0.0390	0.2790	0.4540

TABLE 40. (continued).

Time step	Simulation year	% of year-class reduction based on calculated or simulated levels of ENT[a]	Nominal F (plus IMP)[b]		Fractional fishing discard F for :		
				Age-1	Age-2	Age-3	Age-4
53	2013	U2, U3 flow X ENT = 11.098	0.51	0.0330	0.0390	0.2790	0.4540
54	2014	U2, U3 flow X ENT = 6.605	0.51	0.0330	0.0390	0.2790	0.4540
55	2015	U2, U3 flow X ENT = 12.372	0.51	0.0330	0.0390	0.2790	0.4540
56	2016	U3 flow X ENT = 3.669	0.51	0.0330	0.0390	0.2790	0.4540
57	2017	U3 flow X ENT = 3.702	0.51	0.0330	0.0390	0.2790	0.4540
58	2018	U3 flow X ENT = 5.534	0.51	0.0330	0.0390	0.2790	0.4540
59	2019	U3 flow X ENT = 8.180	0.51	0.0330	0.0390	0.2790	0.4540
60	2020	U3 flow X ENT = 3.499	0.51	0.0330	0.0390	0.2790	0.4540
61	2021	U3 flow X ENT = 11.325	0.51	0.0330	0.0390	0.2790	0.4540
62	2022	U3 flow X ENT = 2.833	0.51	0.0330	0.0390	0.2790	0.4540
63	2023	U3 flow X ENT = 4.451	0.51	0.0330	0.0390	0.2790	0.4540
64	2024	U3 flow X ENT = 7.142	0.51	0.0330	0.0390	0.2790	0.4540
65	2025	U3 flow X ENT = 5.534	0.51	0.0330	0.0390	0.2790	0.4540
66	2026	0.0	0.50	0.0300	0.0350	0.2750	0.4500
67	2027	0.0	0.50	0.0300	0.0350	0.2750	0.4500
68	2028	0.0	0.50	0.0300	0.0350	0.2750	0.4500
69	2029	0.0	0.50	0.0300	0.0350	0.2750	0.4500
70	2030	0.0	0.50	0.0300	0.0350	0.2750	0.4500
71	2031	0.0	0.50	0.0300	0.0350	0.2750	0.4500
72	2032	0.0	0.50	0.0300	0.0350	0.2750	0.4500
73	2033	0.0	0.50	0.0300	0.0350	0.2750	0.4500
74	2034	0.0	0.50	0.0300	0.0350	0.2750	0.4500
75	2035	0.0	0.50	0.0300	0.0350	0.2750	0.4500
76	2036	0.0	0.50	0.0300	0.0350	0.2750	0.4500
77	2037	0.0	0.50	0.0300	0.0350	0.2750	0.4500
78	2038	0.0	0.50	0.0300	0.0350	0.2750	0.4500
79	2039	0.0	0.50	0.0300	0.0350	0.2750	0.4500
80	2040	0.0	0.50	0.0300	0.0350	0.2750	0.4500
81	2041	0.0	0.50	0.0300	0.0350	0.2750	0.4500
82	2042	0.0	0.50	0.0300	0.0350	0.2750	0.4500
83	2043	0.0	0.50	0.0300	0.0350	02750	0.4500
84	2044	0.0	0.50	0.0300	0.0350	0.2750	0.4500
85	2045	0.0	0.50	0.0300	0.0350	0.2750	0.4500
86	2046	0.0	0.50	0.0300	0.0350	0.2750	0.4500
87	2047	0.0	0.50	0.0300	0.0350	0.2750	0.4500
88	2048	0.0	0.50	0.0300	0.0350	0.2750	0.4500
89	2049	0.0	0.50	0.0300	0.0350	0.2750	0.4500
90	2050	0.0	0.50	0.0300	0.0350	0.2750	0.4500
91	2051	0.0	0.50	0.0300	0.0350	0.2750	0.4500
92	2052	0.0	0.50	0.0300	0.0350	0.2750	0.4500
93	2053	0.0	0.50	0.0300	0.0350	0.2750	0.4500
94	2054	0.0	0.50	0.0300	0.0350	0.2750	0.4500
95	2055	0.0	0.50	0.0300	0.0350	0.2750	0.4500
96	2056	0.0	0.50	0.0300	0.0350	0.2750	0.4500
97	2057	0.0	0.50	0.0300	0.0350	0.2750	0.4500
98	2058	0.0	0.50	0.0300	0.0350	0.2750	0.4500
99	2059	0.0	0.50	0.0300	0.0350	0.2750	0.4500
100	2060	0.0	0.50	0.0300	0.0350	0.2750	0.4500

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[a] ENT values for 1984-94 were estimates made under actual MNPS operating conditions as shown on Table 39. For 1971-83 and 1995-2025, ENT values were randomly selected from projected rates determined from mass-balance calculations for full three-unit operation during 1984-94 (Table 39). To adjust the chosen values of ENT, actual MNPS flow values were used for 1971-83 and randomly selected values from Table 5 were used for 1995-2025. The values of ENT given above were used in the SPDM simulations for this report.

[b] F values were obtained from the DEP (P. Howell and V. Creeco, CT DEP, Old Lyme, CT, pers. comm.). Impingement mortality was implemented as an equivalent instantaneous mortality rate (0.01) held constant throughout the MNPS operational period (1971-2025).

reduced to 49,476 lbs. The simulated baseline (the solid line in Figs. 34A and 35) responded as expected to the high rates of fishing through 1991 and the stock steadily declined to its lowest point of 12,907 lbs in 1993 and rose only slightly to 13,704 lbs in 1994. The estimated biomass for 1993 was only about half of the critical stock size (defined as a stock biomass equal to 25% of the MSP) of 24,526 lbs, shown as the dashed line in Figures 34A and B; this reference stock size will be discussed in greater detail below. Allowing for natural variation in the simulation, maximum replicate values of stock sizes for 1992-94 were below 25% of the MSP and the minimum value in 1993 was as small as 8.3% of MSP. The simulation illustrated that the baseline population could fall below the critical stock size at any time from 1984 through 1999. However, if reductions in F work as planned the stock should recover rapidly following its lowest point in 1993.

To determine the effect of MNPS on the Niantic River female spawning stock, the baseline time-series was compared to the impacted time-series, which is shown in Figure 34B and as the dashed line in Figure 35. The impacted series corresponds to projections of the same initial stock as the baseline, but with additional annual losses due to MNPS operation (i.e., ENT + IMP). In this impacted population projection, the stock did not respond to larval losses due to entrainment until 1974 (the fourth year of Unit 1 operation), when biomass began to decline below baseline levels (Fig. 35). The lowest projected stock biomass (10,947 lbs) was reached in 1993, whereas the greatest absolute decline relative to the baseline occurred in 2003 (a difference of 7,026 lbs), when the effects of reductions in F since 1994 were propagated through the spawning population. From this point on, biomass of the impacted stock generally paralleled that of the baseline and began to approach it as MNPS units went offline. The impacted stock moved to within 1,000 lbs of the baseline in 2031 (6 years after the end of Unit 3 operation in 2025) and became virtually identical to it by 2033. These projections depended upon the rates used, but actual winter flounder abundance could depart considerably from predictions if annual rates and simulated conditions are not matched by real conditions.

The different nature of stock reductions caused directly by fishing and impingement and those resulting from larval losses through entrainment at MNPS is related to the age structure of the spawning stock. Fishing reduces biomass of the stock at a greater rate than it reduces the number of spawners because it tends to select for larger fish and, thus, reduces the average weight of the spawners remaining in the stock. However, the most important difference between fishing (with an added component accounting for impingement) and larval entrainment is that the former process removes fish from each year-class every year for as long as any fish remain, while the latter causes a reduction only once in the lifetime of each generation and, then, very early in the life history of a species. The relative effects of stock reductions due to fishing and MNPS impact can be assessed by comparing the unfished stock projection line to those for the fished stock with and without plant effects (Fig. 36). Most biomass reductions were due to fishing. However, as fishing mortality was reduced and stock biomass increased, absolute reduction in population size due to MNPS impact became larger until units began to cease operation.

Stock sizes projected for each simulation scenario at seven selected points in time are given in Table 41; losses relative to the theoretical unfished stock for each particular year are shown as percentages. Stock sizes representing the fifth and ninety-fifth percentiles for the 100 Monte Carlo replicates generated for each year are also given. The theoretical unfished stock in each of the 7 years shown varied little and averaged about 104,000 lbs for each year. Prior to MNPS operation in 1970, the baseline and the impacted stocks were identical (49,476 lbs) and made up about 47% of the unfished stock. By 1990, winter flounder spawning stocks under full MNPS three-unit operation declined nearly 50% relative to 1970. However, this was mostly the result of increased fishing as the impacted stock was only about 2,100 lbs less than the baseline. As noted previously, smallest stock sizes were found in 1993 as a result of high rates of exploitation during the early 1990s. In 1994, both the baseline and impacted stocks increased by 6% from 1993, but biomass remained only 13.1% and 11.1%, respectively, of the unfished stock. By 2000, the baseline stock had responded more rapidly to decreased fishing than the impacted stock and represented 39.4% of the unfished population as opposed to 32.8% for the impacted stock. The gap between the two began to narrow in 2010 and 2020 and both were nearly the same by 2030, when the impacted stock was considered to be fully recovered. Finally, the time-series were indistinguishable after 2035.

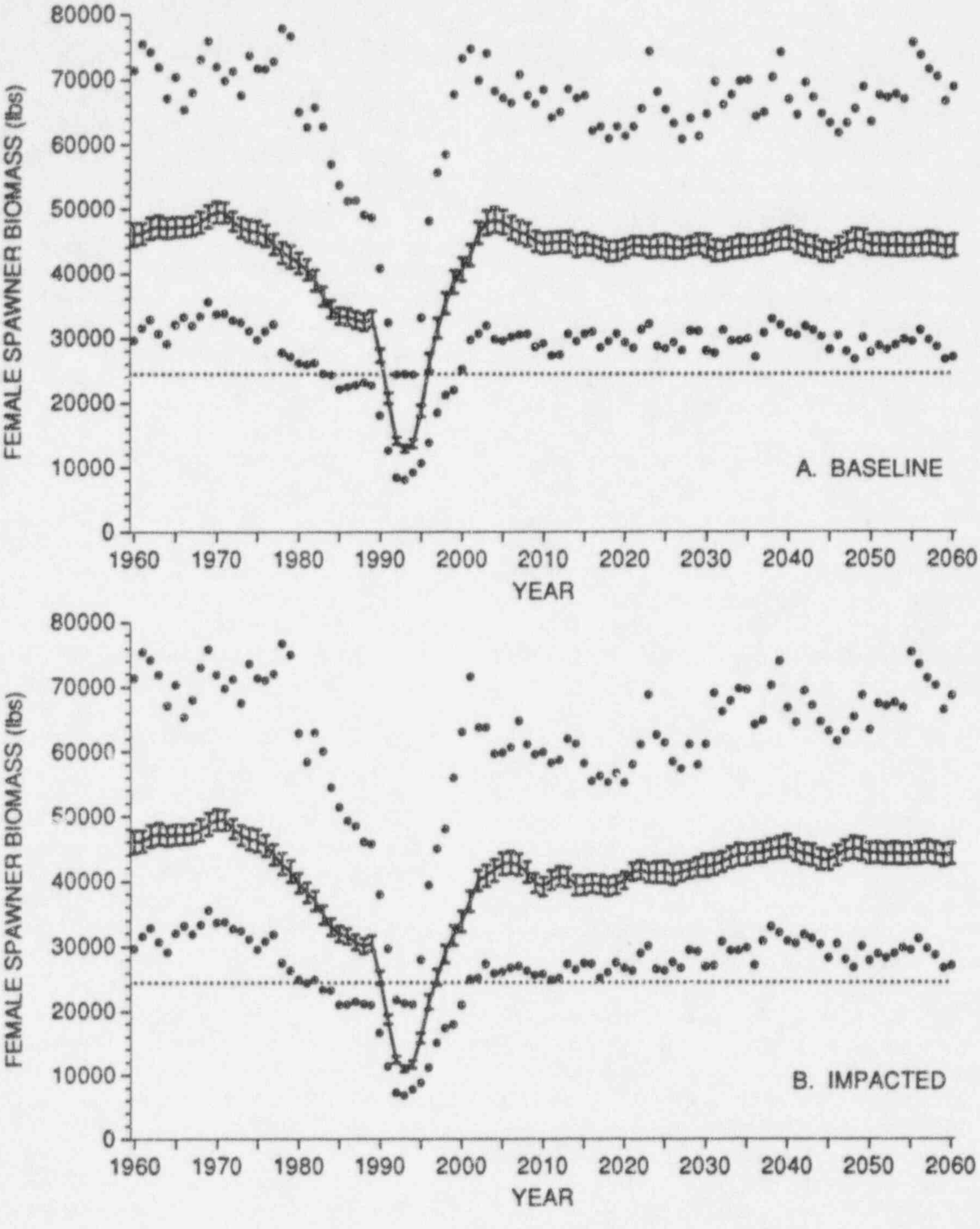

Fig. 34. Stochastic variability associated with the projected Niantic River female winter flounder stock expressed as biomass (lbs) for: A. The baseline stock with simulated sport and commercial fishing rates, but no effects from MNPS operation, and B. The impacted stock with both fishing effects and MNPS impact. The solid lines are the geometric means and 95% confidence interval (100 Monte Carlo replications) of each stock size trajectory and are equal to the baseline and impacted stocks illustrated on Figure 35. The symbols above and below the line correspond to the largest and smallest stocks among the 100 replicates generated for each year. The dashed line represents the critical stock size (here, 24,526 lbs), defined as when stock biomass has been reduced to 25% of the maximum spawning potential (Howell et al. 1992).

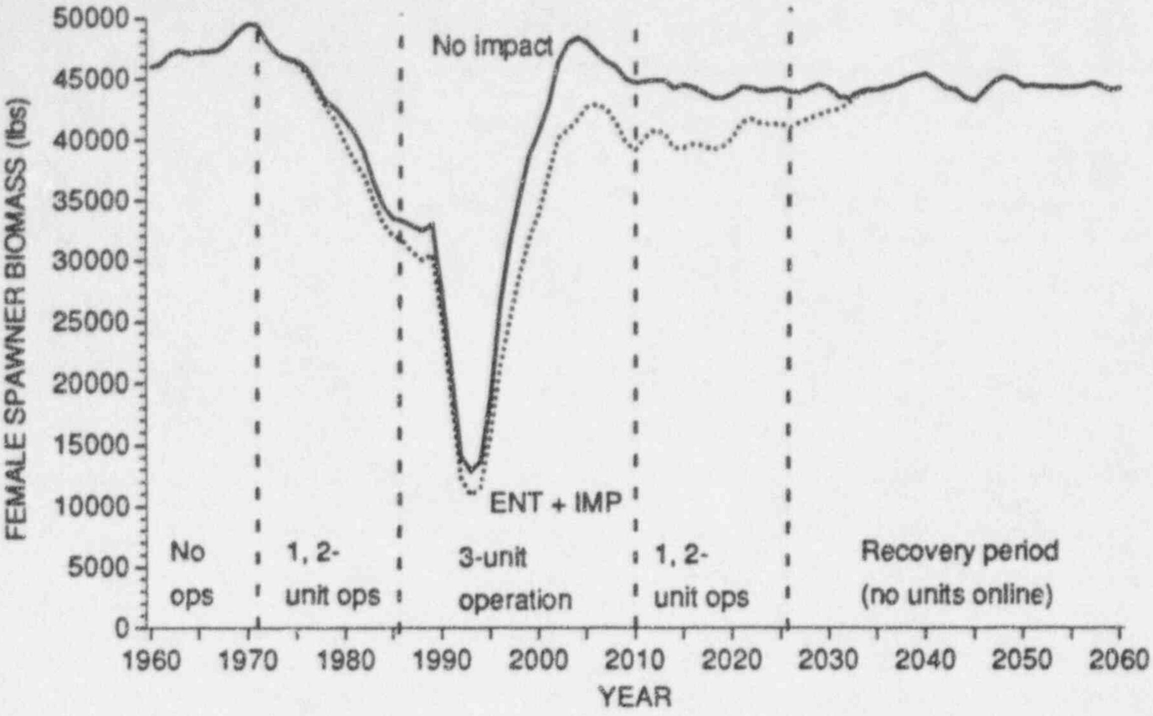

Fig. 35. Results of the SPDM simulation showing the combined effects of fishing and calculated larval entrainment rate and fish impingement rates (dashed line labeled ENT + IMP) on the biomass of the Niantic River female winter flounder spawning stock. Entrainment rates changed annually according to the number of MNPS units in operation and fishing rates were also variable (see text and Table 40 for details). The solid line labeled "No impact" is the baseline with fishing effects only. All stock sizes are averages of 100 Monte Carlo replicates.

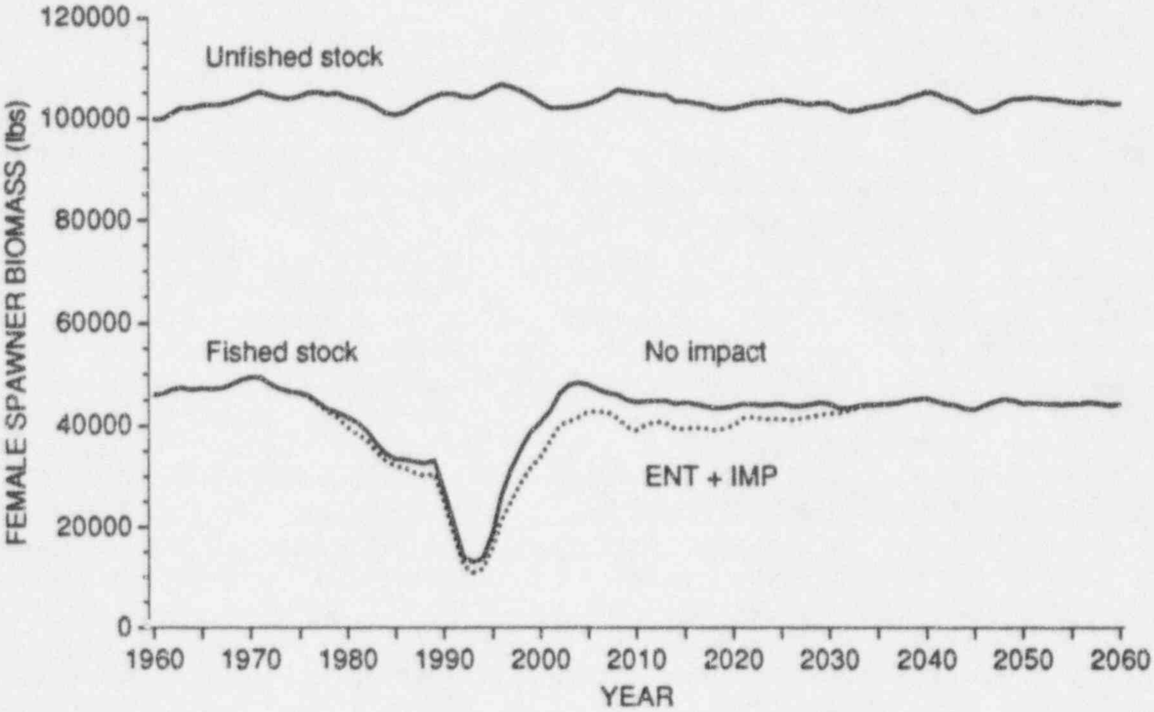

Fig. 36. Illustration of the effects of fishing (solid line labeled "No impact") and MNPS operation under calculated larval entrainment and impingement rates (dashed line labeled "ENT + IMP") relative to the theoretical (SRR-based estimate) unfished stock expressed as female spawner biomass in lbs. All stock sizes are averages of 100 Monte Carlo replicates.

TABLE 41. Expected biomass in pounds of female winter flounder spawners at seven selected points in time during SPDM simulations of the Niantic River population (see Figures 34 and 35). Expected mean stock sizes are geometric means of 100 Monte Carlo replicates and the fifth and ninety-fifth percentiles of stock sizes for the 100 replicates of each year are given.

Type of population simulated	1970	1980	1990	1994	2000	2010	2020	2030	2040
Theoretical unfished stock[a]									
Geometric mean	104,536	104,237	104,839	104,832	103,093	104,940	101,757	102,698	104,965
5th percentile	85,740	83,017	86,337	84,670	81,700	85,664	83,008	84,804	85,393
95th percentile	123,550	125,320	127,149	127,086	129,077	129,264	118,877	121,864	127,151
Baseline[b]									
Geometric mean	49,476	41,587	27,383	13,704	40,568	44,604	43,706	44,134	45,274
5th percentile	38,572	30,634	19,999	9,879	27,264	33,364	33,160	32,687	33,379
95th percentile	63,070	53,912	36,332	19,858	55,852	59,369	54,809	59,899	61,959
% of the theoretical unfished stock	47.3%	39.9%	26.1%	13.1%	39.4%	42.7%	43.0%	43.0%	43.1%
Impact (ENT + IMP)[c]									
Geometric mean	49,476	39,737	25,253	11,633	33,849	39,131	40,009	42,277	45,274
5th percentile	38,572	29,158	18,267	8,334	22,062	29,125	30,213	31,776	33,384
95th percentile	63,070	51,648	33,802	16,940	46,856	51,957	49,409	56,765	61,916
% of the theoretical unfished stock	47.3%	38.1%	24.1%	11.1%	32.8%	37.3%	39.3%	41.2%	43.1%

[a] No fishing or MNPS effects.

[b] Fishing effects, but no MNPS impact.

[c] Combined effects of entrainment and impingement (ENT + IMP) at MNPS in addition to fishing.

Probabilistic assessment of MNPS effects. The stochastic variability associated with stock projections for the baseline and impacted stocks (Fig. 34) formed the basis for probabilistic analyses. These analyses took into account not only the mean stock biomass predicted for each year, but also the empirical frequency distribution of 100 replicate predictions that were both smaller and larger than the mean. To assess effects of MNPS operation, the probabilities that the Niantic River female winter flounder spawning stock would fall below selected reference sizes were determined directly from the frequency distribution of 100 replicates of each annual stock size. The reference sizes were percentages (25, 30, and 40%) of the biomass of spawning females for the unfished stock (i.e., the MSP) as suggested in the Atlantic States Marine Fisheries Commission management plan for inshore stocks of winter flounder (Howell et al. 1992). In eastern LIS, values of F ranging from 0.37 to 0.68 would be necessary to achieve maximum yield, depending upon various combinations of length (10, 11, or 12 inches) and trawl codend mesh (3.5, 4.5, 5.0, or 5.5 inches) restrictions imposed on the commercial fishery. A stock that has been reduced to less than 25% of the MSP is considered overfished and its continued maintenance is questionable. Furthermore, spawner abundance may decline to even lower levels. Alternatively, fishing rates that preserve 40% of MSP allow for the preservation of the stock and maximize yield to the fisheries. This level may be viewed as conservative because the simulations suggested that even under moderate exploitation in the 1960s and 1970s, a number of simulated projections were below 40% of MSP.

In 1970, both the baseline and impacted stocks were likely ($p \geq 0.95$) larger than 40% of MSP (Table 42). However, by 1980 both stocks had increased probabilities (0.36, 0.45) of falling below 40% of MSP. In 1990, the stocks were almost certainly less than 40% of MSP and likely ($p \geq 0.68, 0.79$) less than 30% of MSP. The impacted stock also had a probability of 0.40 of falling below 25% of MSP. At the

TABLE 42. Probabilities of Niantic River female spawning stock biomass falling below three selected reference sizes at seven selected points in time. Reference sizes are expressed as a percentage of the maximum spawning potential (MSP) of 98,104 lbs for the theoretical unfished stock (F = 0). Probabilities were based on the empirical probability distribution function corresponding to 100 Monte Carlo replications.

Type of population simulated	Reference stock size[a]	1970	1980	1990	1994	2000	2010	2020	2030	2040
Baseline[b]	25% of MSP	0.00	0.00	0.26	1.00	0.00	0.00	0.00	0.00	0.00
Impacted[c]	25% of MSP	0.00	0.00	0.40	1.00	0.08	0.00	0.00	0.00	0.00
Baseline	30% of MSP	0.00	0.02	0.68	1.00	0.09	0.01	0.01	0.01	0.00
Impacted	30% of MSP	0.00	0.07	0.79	1.00	0.25	0.05	0.00	0.02	0.00
Baseline	40% of MSP	0.05	0.36	0.99	1.00	0.41	0.20	0.27	0.28	0.21
Impacted	40% of MSP	0.05	0.45	1.00	1.00	0.76	0.51	0.45	0.34	0.21

[a] Corresponds to reference stock sizes given in Howell et al. (1992) of 25%, 30%, and 40% of the MSP (24,526, 29,431, and 39,242 lbs, respectively).

[b] Fishing effects, but no MNPS impact.

[c] Combined effects of entrainment and impingement (ENT + IMP) at MNPS in addition to fishing.

lowest points of both stock projections in the mid-1990s, all replicates were below 25% of MSP. Reductions in fishing rates in the late 1990s allowed for a rapid increase in spawning biomass above this critical level to more optimal stock sizes by 2000. Spawning biomass would then be likely greater than 25% of MSP and the impacted stock had a one in four chance of being greater than 30% of MSP. By 2010 and 2020, the stocks had high (≥ 0.95) likelihood of being above 30% of MSP and the impacted stock had a better than even probability of being greater than 40% of MSP. For a winter flounder stock to reach a more desirable size, which according to Howell et al. (1992) is greater than 40% of MSP, fishing mortality would have to be further reduced. The Niantic River stock stabilized at a biomass of about 44,000 lbs following the shutdown of MNPS in 2025 and probabilities that this stock would be smaller than 40% of MSP were about 20 to 30% annually.

Conclusions

Since sampling began in 1976, adult winter flounder abundance indices reached their lowest point in 1993. Although the catch of spawners in the Niantic River increased in 1994, abundance remained the second lowest on record. Densities of early stage larvae found in Niantic River and Bay were relatively low. However, the densities of late stage larvae in the lower Niantic River and in Niantic Bay were above-average and young collected during summer in the Niantic River were among the most abundant since 1983. Thus, the 1994 year-class appears to be relatively strong. A number of environmental and biological factors interact to affect winter flounder growth and survival within a particular year. It appears that the unusually cold winter, low spawning stock, warm early spring, and cool late spring contributed positively to the success of the 1994 year-class. Newly implemented fishing regulations are designed to protect these fish as they grow and age, insuring an increase in the numbers of spawners recruited in subsequent years.

The long-term assessment of MNPS operation and its effects on Niantic River winter flounder was made using the NUSCO stochastic population dynamics model, which took into account fishing mortality as well as plant operation. Effects of MNPS operation were primarily from larval entrainment, which from 1984 through 1994 reduced Niantic River larval production by 4.7 to 22.0% each year. Model simulations indicated that fishing alone reduced female spawner biomass from about 100,000 lbs for the theoretical unfished stock to 45-50,000 lbs in the 1960s and 1970s. Following those years, large increases in fishing mortality reduced the baseline (fishing effects only) stock biomass rapidly to about 12,900 lbs in 1993 and to 13,700 lbs in 1994. MNPS impact further reduced stock sizes in the mid-1990s by an additional 1,000 lbs each year.

Present stock biomass is considerably below the critical stock size, defined by the Atlantic States Marine Fisheries Commission (ASMFC) as 25% of the maximum spawning potential (MSP; i.e., equilibrium spawner biomass in the absence of fishing). MSP was determined to be 24,526 lbs for the Niantic River population. When substantial reductions in fishing mortality, which should occur in the near future as a result of new regulations, were incorporated into the model, the simulated stock recovered quickly. By 2000, the baseline and MNPS-impacted stocks had high probability of being larger than 25% of MSP and by 2010 of being greater than 30% of MSP. As stock abundance increased, however, the effect of MNPS operation became relatively larger in terms of absolute losses of stock biomass. After the cessation of MNPS operation in 2025, the two biomass time-series became identical within a few years. The stock was then considered to be fully recovered and stabilized at a biomass of about 44,000 lbs, or 45% of MSP. This stock recovery, however, assumed that planned changes in fishing regulations had been implemented as scheduled and that they achieved the expected reductions in fishing mortality. Even with reductions in fishing mortality there still was a 34% chance that this new equilibrium stock biomass would remain smaller than 40% of MSP, a targeted goal of the ASMFC.

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Fish Ecology Studies

Fish Ecology Studies

Introduction

Millions of dollars accrue annually by both commercial and sport fishing in Connecticut (Sampson 19[illegible], Blake and Smith 1984). Clearly fish are important members of the estuarine community in eastern Long Island Sound (LIS). The objective of the fish ecology monitoring programs at Millstone Nuclear Power Station (MNPS) is to determine whether operation of the three generating units affects the local fish communities. These effects have been defined as power-plant related changes in the occurrence, distribution and abundance of fish species, which would affect community structure. Annual abundance estimates have been calculated for various life stages of fish in the vicinity of MNPS to help assess the effects of station operation. Direct MNPS impacts (entrainment, impingement, and thermal changes) affecting fish populations are easily measured, but the quantification of long-term effects for these impacts is more difficult. Numbers of fish eggs and larvae entrained through MNPS cooling-water system have been reliably estimated. How this loss actually affects local populations is influenced by many mechanisms, such as compensatory mortality, density-dependent growth, fecundity of individual species, population age structure, and life history strategies. Impingement of juvenile and adult fish at the MNPS intakes can also be measured but, as in the case of eggs and larvae, the implications of removal are more difficult to ascertain. In addition to impingement, fish populations are also affected by natural and fishing mortality. Changes in the thermal regime of local waters are well-documented. If water temperatures exceed tolerance levels, fish move from the area thus changing local populations, especially if the area vacated is a major spawning or nursery ground. Trawl, seine and ichthyoplankton monitoring programs have successfully measured impacts from MNPS on local fish populations. These programs provide a basis for identifying which taxa could be potentially impacted, as well as long-term abundance trends used to assess changes in local populations. Life history and population characteristics of potentially impacted species are reported and evaluated to determine potential detrimental MNPS impacts. Data from June 1993 through May 1994 are summarized and compared to historical data from June 1976 through May 1993 of trawl, seine, and larval entrainment collections and from June 1979 through May 1993 for entrained eggs and larvae collected in Niantic Bay.

Materials and Methods

This report year comprises the 12-month period from June of a given year through May of the following year. Because of occasional overlap in the occurrence of a species during the May-June transitional period, species-specific analyses are based on actual periods of occurrence instead of being constrained to the May 31 endpoint. When the season of occurrence of a species crossed a calendar year, the year was reported as "1993-94", but when the species occurred only within a calendar year, the year was reported as "1994". Materials and methods of the 1993-94 reporting period are essentially the same as those used in previous years (NUSCO 1990, 1991, 1992, 1993, 1994).

Ichthyoplankton Program

Ichthyoplankton (fish eggs and larvae) entrained through the MNPS cooling-water system were collected both day and night three times each week from June through August, one day and one night per week in September 1993, and one day per week from October through January 1994. In February 1994, samples were collected both day and night once per week and in March through May 1994, three times per week. Generally, samples were collected each week at only one of the three plant discharges (station EN, Fig. 1), with the site of collection alternating weekly between Units 1 and 2. To collect samples from the discharge water, a 1.0 x 3.6-m conical plankton net with 333-µm mesh was deployed with the aid of a gantry system. Four General Oceanic flowmeters (Model 2030) were mounted in the mouth of this net and positioned to account for horizontal and vertical flow variations. Sample volume (about 200 m^3, except during periods of high plankton or detritus concentrations) was determined by an average of the volume estimates from the four flowmeters.

Plankton samples were separated using a NOAA-Bourne splitter (Botelho and Donnelly 1978);

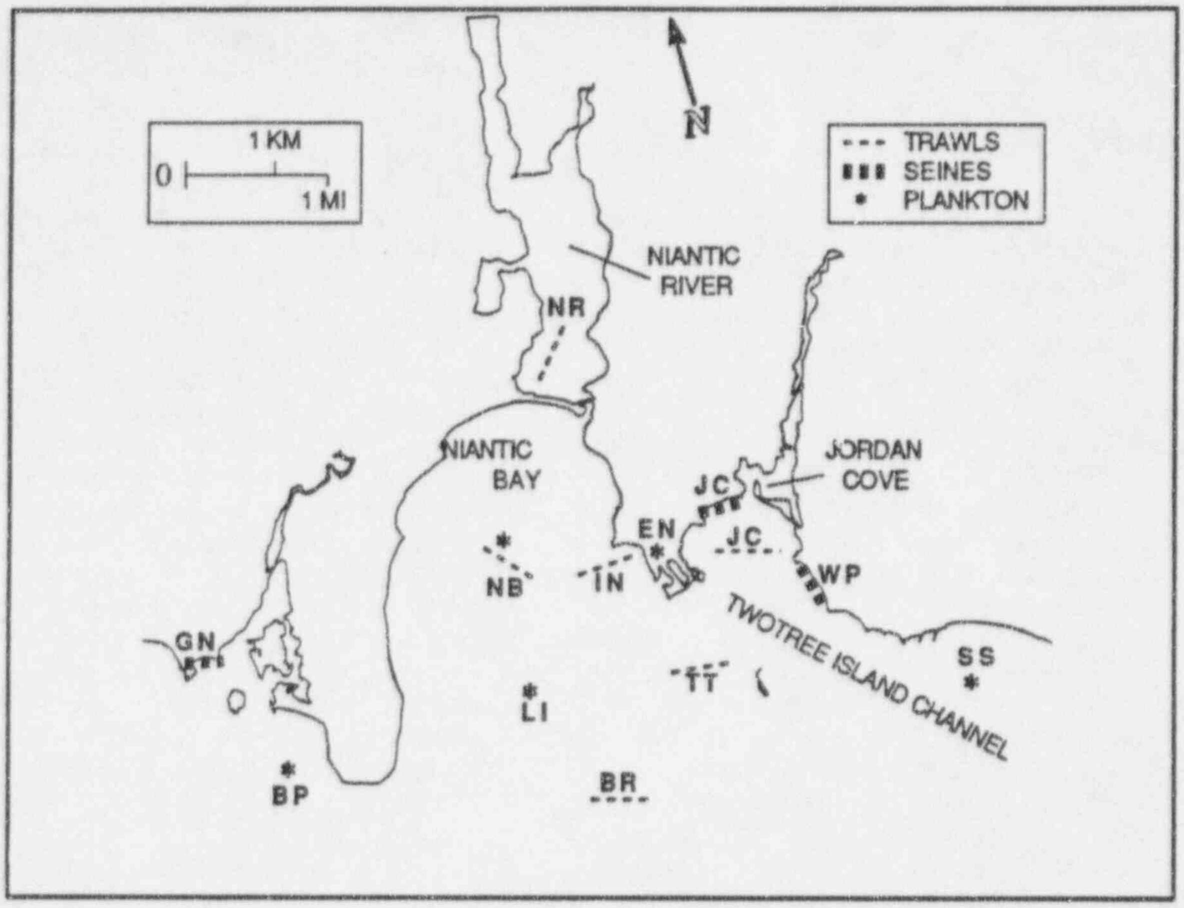

Fig. 1. Location of trawl, seine, and ichthyoplankton sampling stations.

ichthyoplankton were removed from the samples with the aid of dissecting microscopes. Successive splits were completely sorted until at least 50 larvae (and 50 eggs for samples processed for eggs) were found, or until one-half of the sample was examined. Samples examined for larvae included all EN samples collected from January through May and July through December. In the June samples, only two (one day and one night) EN samples per week were examined. All EN samples collected in April through September were examined for fish eggs. Larvae were identified to the lowest practical taxon. Wrasse (tautog, *Tautoga onitis* and cunner, *Tautogolabrus adspersus*) eggs were distinguished from a weekly composite sample of their eggs using the criterion of bimodality of egg diameters (Williams 1967). Ichthyoplankton densities are reported as number of organisms per 500 m^3.

Wrasse Egg Distribution Study

In addition to ichthyoplankton monitoring, the spatial distribution of tautog and cunner eggs in the MNPS area was examined in 1994. Five stations were sampled (Fig. 1) and three of them (BP, LI, and SS) were new collection locations. The water depth at all offshore stations ranged from about 6 to 10 m. Station BP was sampled during an ebb tide and station SS during a flood tide, so that collection densities would represent eggs potentially imported into Niantic Bay from the west and east, respectively. The remaining three stations (EN, NB, and LI) were sampled during both tidal stages. Station EN was sampled using the previously described entrainment gantry system and the other four stations with a bongo frame fitted with 333-µm mesh nets using a stepwise oblique tow pattern for a 6-minute duration.

Samples were collected during 0500 to 1100 hours. This time period was selected because 24-h studies conducted in 1993 (NUSCO 1994) showed that tautog and cunner egg densities remain relatively stable at this time. Stations EN and NB were sampled at approximately at the same time. The collection sequence of stations sampled with bongo nets was LI, NB, and SS during a flood tide and LI, NB, and BP during an ebb tide. These sequences facilitated paired comparisons of stations BP, LI, and SS with EN using the Wilcoxon's signed-ranks test (Sokal and Rohlf 1969). By sampling NB second in the sequence (with EN sampled nearly simultaneously), the sampling interval between EN and the other three stations was minimized. Sampling dates were June 23 and 24 during a flood tide and June 29 and 30 during an ebb tide. These time periods occurred during peak densities of tautog and cunner eggs. On each sampling date, three sequences of samples were taken (LI, NB, EN, BP during an ebb and LI, NB, EN, SS during a flood), with the first sequence starting about one hour before maximum tidal current, the second starting near maximum current, and the third immediately after the second was completed. A minimum of 100 eggs (if present) per sample was measured to distinguish between tautog and cunner eggs.

Trawl Program

Triplicate bottom tows were made using a 9.1-m otter trawl with a 0.6-cm codend liner. Demersal fishes were collected biweekly throughout the year at six stations: Niantic River (NR), Jordan Cove (JC), Twotree Island Channel (TT), Bartlett Reef (BR), Intake (IN) and Niantic Bay (NB) (Fig. 1). Occasionally, only two replicate tows were taken at a station because of damage to gear or severe weather. A standard tow was 0.69 km and this distance was measured using onboard radar. When the trawl net became loaded with macroalgae and detritus, tow distances were shortened and catches standardized to 0.69 km. Catch was expressed as the number of fish per standardized tow (CPUE). Up to 50 randomly chosen individuals of certain selected species per station were measured (total length) to the nearest mm.

Seine Program

Shore-zone fish were sampled using a 9.1 x 1.2-m knotless nylon seine net of 0.6-cm mesh. Triplicate shore-zone hauls were made parallel to the shoreline at White Point (WP), Jordan Cove (JC), and Giants Neck (GN) biweekly from May through November in 1993 (Fig. 1). Beginning May 1994, sampling was reduced to one station, JC. A standard haul distance was 30 m. Collections were made during a period 2 hours before to 1 hour after high tide; generally all three stations were sampled the same day. Fish in each haul were identified to the lowest possible taxon, counted, and the total length of up to 50 randomly selected individuals of each species from each replicate were measured to the nearest mm. Catch was expressed as number of fish per haul.

Data Analyses

Abundance Estimates

Occurrence, distribution, and abundance of selected potentially impacted fish, as well as observed spatial and temporal fluctuations, were analyzed to assess the possibility of plant-related impacts. Indices of fish abundance were selected on the basis of underlying distributional assumptions; failure of the data to conform to these assumptions may reduce the precision of the estimates or, worse, provide biased results. Thus, the Δ-mean was used as an index of abundance.

The Δ-mean was selected to describe annual abundance trends because it is the best estimator of the mean of a population that approximately follows the lognormal distribution and contains many zeros (Hennemuth et al. 1980; Pennington 1983, 1986). Calculation of this index and its variance estimate was described in detail in NUSCO (1988). The Δ-mean was used as an index of abundance for juvenile and adult fish collected in the trawl and seine programs, and for larvae and fish eggs collected at EN. The Δ-mean indices of ichthyoplankton species were weighted by the largest number of samples collected in a week to standardize data across weeks and years. For species that occurred seasonally, the data used for calculating the Δ-mean were restricted to their period of occurrence to reduce the number of zero values in the distribution tails. Two-unit operational period Δ-means were calculated from the beginning of two-unit operation (1976) to the beginning of three-unit operation (1986). A non-parametric, distribution-free, Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether the direction and rate of change of a time-series of annual Δ-means represented a significant trend. Sen's nonparametric estimator of the slope (Sen 1968) was used to describe linear trends whenever significant. This approach to trend analysis was suggested by Gilbert (1989) as particularly suitable for the analysis of environmental monitoring data because no distributional assumptions are required.

Entrainment Estimates

Entrainment estimates of dominant ichthyoplankton were calculated from daily density estimates at EN. These estimates were determined from a Gompertz function fitted to the entrainment data. The distribution of egg and larval abundance over time is usually skewed because densities increase rapidly to a maximum and then decline slowly. The cumulative density over time from this type of distribution resembles a sigmoid-shaped curve, for which the inflection point occurs at the time of peak abundance. The Gompertz function (Draper and Smith 1981) was used to describe the cumulative egg and larval abundance distribution. Thus, the inflection point was not constrained to be the mid-point of the sigmoid curve as is the case in the frequently used logistic and probit curves. The form of the Gompertz function used (Gendron 1989) was:

$$C_t = \alpha \cdot \exp(-\exp[-k \cdot \{t-p\}]) \qquad (1)$$

where:

C_t = cumulative density at time t

t = time in days from the date when the eggs or larvae generally first occur

α = total or asymptotic cumulative density

p = inflection point in days since first date of occurrence

k = shape parameter

The origin of the time scale was set to the date when the eggs or larvae generally first appeared in the waters off MNPS. Least-squares estimates, standard errors, and asymptotic 95% confidence intervals of these parameters were obtained by fitting the above equation to the cumulative abundance data using nonlinear regression methods (SAS Institute Inc. 1985). The cumulative data were obtained as the running sums of the weekly geometric means of the abundance data per unit volume.

A "density" function was derived algebraically by calculating the first derivative of the Gompertz function (Eq. 1) with respect to time. This density function, which directly describes the larval abundance over time (abundance curve), has the form:

$$d_t = \alpha' \cdot k \cdot \exp(-\exp[-k \cdot \{t-p\}] - k \cdot [t-p]) \qquad (2)$$

where α' equals $7 \cdot \alpha$ because the cumulative densities were based on weekly (7-day period) geometric means, d_t is density on day t and all the other parameters are as described in Equation 1.

Daily entrainment was estimated by multiplying these daily densities d_t by the daily volume of cooling water that passed through MNPS. Annual entrainment estimates were determined by summing all daily estimates during the period of occurrence.

Results and Discussion

From June 1976 through May 1994, over one hundred species of egg, larval, juvenile, and adult fishes were collected in the MNPS monitoring programs (Appendix 1). The most common fishes were winter flounder (*Pleuronectes americanus*), anchovies (*Anchoa mitchilli* and *A. hepsetus*), silversides (*Menidia menidia* and *M. beryllina*), grubby (*Myoxocephalus aenaeus*), American sand lance (*Ammodytes americanus*), skates (*Raja erinacea, R. ocellata* and *R. eglanteria*), scup (*Stenotomus chrysops*), windowpane (*Scophthalmus aquosus*), tautog, and cunner.

Sixty-one taxa were represented in ichthyoplankton samples; of these, three egg and fifteen larval taxa were found in sufficient numbers to calculate Δ-mean densities (Table 1). All 1993-94 egg and larval densities at EN were within historic ranges. However, densities of anchovy larvae at EN were the third lowest recorded with only anchovy larvae densities in 1987-88 and 1992-93 lower.

Over the past 18 years, 103 fish taxa were caught in trawls and 50 taxa were captured in seines (Appendix I). Six taxa in trawl samples (winter flounder, scup, windowpane, skates, silversides, and anchovies) and one taxon in seine samples (silversides) accounted for over 80% of the catch in each monitoring program (Appendices II, III, IV, and V). Winter flounder continued to dominate the trawl catches, accounting for 38% of the catch from 1976-77 through 1993-94. Although the 1993-94 catch was the second lowest in the 18 year series, winter flounder accounted for 41% of the catch this year. (Appendix II). Historically, scup comprised 29% of the catch, but only accounted

TABLE 1. The Δ-mean[a] density (no.·500 m^{-3}) of the most abundant fish eggs and larvae collected at EN for each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

Taxon	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94
EGGS[a]																		
T. adspersus				5,870	8,223	5,171	5,501	7,068	5,719	7,484	2,969	5,002	5,395	6,904	4,998	6,954	4,416	5,436
T. onitis				1,364	2,842	2,647	2,244	2,114	2,157	3,237	2,756	3,011	2,269	2,887	2,060	1,878	1,449	1,596
Anchoa spp.				1,447	1,245	1,080	765	2,257	4,880	145	910	89	38	54	127	476	107	542
LARVAE[b]																		
Anchoa spp.	1,152	931	483	2,168	2,430	5,768	816	1,421	302	1,102	1,244	126	359	619	1,122	799	178	203
P. americanus	106	143	114	285	129	233	297	210	180	87	109	116	203	106	99	388	21	142
A. americanus	94	318	119	111	136	21	27	18	9	3	13	41	31	24	7	18	28	42
M. aenaeus	41	38	36	38	107	72	68	50	68	34	29	95	63	30	24	58	34	48
B. tyrannus	5	4	4	0	3	1	11	23	2	41	3	2	6	72	18	97	41	9
T. adspersus	29	58	1	13	58	78	31	49	4	12	4	5	9	14	68	209	8	10
P. gunnellus	13	13	16	13	58	27	13	14	14	22	4	26	9	6	3	15	8	28
T. onitis	37	36	1	11	46	83	44	33	3	15	3	7	17	15	33	99	13	6
E. cimbrius	2	8	6	8	6	1	6	13	5	8	8	12	45	31	37	98	5	18
U. subbifurcata	5	9	14	14	16	17	6	4	60	7	9	23	41	51	34	28	2	18
Liparis spp.	27	30	10	16	22	5	13	8	36	1	4	42	18	12	3	23	14	12
S. fuscus	4	7	4	9	8	13	7	9	9	5	4	6	7	5	3	5	3	6
S. aquosus	10	11	1	5	5	5	2	13	3	1	4	3	5	3	4	12	2	2
P. triacanthus	14	3	1	2	11	17	9	9	1	2	3	0	9	5	29	10	2	2
Gobiidae	6	3	1	0	1	0	0	1	4	3	3	2	4	8	7	12	2	5

[a] Data seasonally restricted to May 22-July 23 for *T. adspersus*, May 23-August 25 for *T. onitis* and June 15-August 5 for *Anchoa* spp.

[b] Data seasonally restricted to July-September for *Anchoa* spp., March-June for *P. americanus*, December-May for *A. americanus*, February-May for *M. aenaeus*, January-May for *P. gunnellus*, July-December for *B. tyrannus*, to June-August for *T. adspersus*, June-August for *T. onitis*, March-May for *Liparis* spp., April-September for *S. fuscus*, April-June for *U. subbifurcata*, April-July for *E. cimbrius*, May-October for *S. aquosus*, June-September for *P. triacanthus*, and June-November for Gobiidae.

for 5% during 1993-94. The low catch of scup in 1993-94 resulted in a low percent contribution, thus increasing the relative contributions of the other dominant species caught by trawl. In 1993-94, windowpane and skates accounted for 14% and 15% of the total, respectively, which was well above the 18-year (1976-77 through 1993-94) average of 7% each. Historically, silversides accounted for 4% of the trawl catch, but this year only 3%. Anchovies accounted for less than 1% of the catch in 1993-94, which was an historic low.

The Δ-mean catch for scup was also at an historic low and winter flounder had the second lowest, with only 1977-78 lower (Table 2). The Δ-mean catch of other taxa were all within the ranges of previous annual Δ-means. During the three-unit operational period, the Δ-mean trawl catch of winter flounder exhibited a significant ($p = 0.013$) negative slope (–0.5); no significant trends were found for other species collected by the trawl monitoring program.

Silversides dominated seine catches accounting for 81% of the catch from 1976-77 through 1993-94 and 78% of the catch in 1993-94 (Appendix IV).

Cunner, tautog, and anchovies accounted for 88% of all eggs entrained from June 1976 through May 1994 (Table 3). Anchovies, winter flounder, American sand lance, and grubby accounted for 80% of the entrained larvae during the same period (Table 3). More cunner eggs were entrained in 1993 than any other year (Table 4). Except for cunner eggs, entrainment estimates of eggs and larvae were within historic ranges (Tables 4 and 5).

Additional data analyses were completed for six taxa that were identified with potential for impact, based on their prevalence in entrainment samples or their susceptibility to thermal impacts. Accordingly, the Δ-mean densities (no.·500 m^{-3}) for eggs and larvae, Δ-mean catches (no.·0.69 km^{-1}) for trawl, and (no.·30 m^{-1}) seine monitoring programs were calculated for abundant life stages of American sand lance, anchovies, silversides, grubby, cunner and tautog. The winter flounder is discussed in a separate section (see Winter Flounder Studies) and is not included among these fishes.

TABLE 2. The Δ-mean* catch (no.·0.69 km^{-1}) of the most abundant fish collected by trawl for each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

Taxon	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94
P. americanus	16.6	13.5	16.7	26.8	32.6	24.1	41.8	27.7	29.5	22.0	19.8	19.3	26.2	18.2	19.1	17.1	17.3	15.7
S. chrysops	10.6	19.8	13.3	18.5	17.0	20.4	27.5	26.6	22.3	13.6	30.6	21.7	18.0	14.5	120.9	212.0	63.4	3.9
Anchoa spp.	11.1	3.3	39.3	0.1	0.1	4.0	0.2	0.4	0.7	113.8	57.3	1.6	3.1	15.9	0.0	0.6	0.6	0.1
S. aquosus	2.9	2.4	1.8	2.9	3.5	2.9	6.7	5.0	4.4	4.7	3.8	4.0	5.1	5.7	3.5	1.8	2.9	4.8
Raja spp.	1.4	1.2	0.8	0.8	2.0	1.4	6.1	5.3	3.1	8.5	4.5	4.6	6.3	5.3	6.3	6.4	3.9	5.3
Menidia spp.	16.2	9.7	2.8	6.2	6.5	1.8	1.5	2.1	0.5	1.9	17.8	2.3	3.4	1.9	1.8	5.9	13.4	2.0

* Data seasonally restricted to June-October for *S. chrysops*, August-October for *Anchoa* spp., October-February for *Menidia* spp., and the remaining taxa year-round (June-May).

TABLE 3. Taxonomic composition of ichthyoplankton collected at EN (as a percentage of the total) from June 1976 through May 1994 for larvae and April 1979 through September 1993 for eggs.

Taxa	Larvae	Eggs
Anchoa spp.	54.6	6.6
Pleuronectes americanus	12.8	
Ammodytes spp.	7.9	
Myoxocephalus aenaeus	4.5	
Brevoortia tyrannus	3.7	
Tautogolabrus adspersus	2.4	52.8
Pholis gunnellus	2.2	
Tautoga onitis	2.1	28.6
Enchelyopus cimbrius	1.7	
Ulvaria subbifurcata	1.3	
Liparis spp.	1.2	
Syngnathus fuscus	1.0	
Scophthalmus aquosus	0.8	
Peprilus triacanthus	0.8	
Gobiidae	0.6	

American sand lance

American sand lance were caught primarily as larvae in the winter and spring and were seldom caught by trawl or seine (Table 5). Larval catches were variable and annual entrainment estimates have ranged from 5 to 190 million larvae. Their abundance varied over two orders of magnitude during the past 18 years (Table 6). Larval densities during two- and three- unit operational periods were compared to assess abundance trends and annual variation (Fig. 2). Because sand lance larvae were so abundant from 1976-77 through 1980-81, larval densities in the three-unit period have been lower than during the two-unit period. Larval densities decreased significantly ($p < 0.05$; Mann-Kendall test) during the two-unit period (slope = –19). The 1993-94 densities of sand lance larvae were the highest of the three-unit period.

Declines in sand lance abundance were also apparent in other areas of the Northwest Atlantic Ocean. Larval densities in LIS over a 32-yr period (1951-83) were highest in 1965-66 and 1978-79. Density figures in the latter years corresponding to a peak observed throughout the entire range of American sand lance (Monteleone et al. 1987). Nizinski et al. (1990) also reported a peak in sand lance abundance throughout the Northwest Atlantic in 1981, with numbers declining since then. Sand lance abundance was noted to be inversely correlated with that of Atlantic herring and Atlantic mackerel (Sherman et al. 1981; Nizinski et al. 1990). Sand lance likely increased in abundance, replacing their herring and mackerel competitors, which had been reduced by overfishing in the 1970s (Sherman et al. 1981). In more recent years, Atlantic mackerel, which prey heavily upon sand lance (Monteleone et al. 1987), have become very abundant as sand lance abundance decreased. Given the large abundance changes of this species along the Atlantic coast, effects of MNPS operation on sand lance are difficult to ascertain.

Anchovies

The bay anchovy is the most common fish along the Atlantic coast and the most abundant ichthyoplankton species within its range (Leak and Houde 1987). Larval anchovies dominated plankton collections and anchovy eggs ranked third in abundance. The entrainment estimates for eggs and larvae for 1993 fell within historic ranges (Tables 4 and 5). The 1993 anchovy egg density was within the range of densities since 1987 (Table 7). All egg densities during three-unit operational period were below the two-unit average, because after 1984 (before the three-unit period) densities declined an order of magnitude and remained low (Fig. 3). Larval densities in 1993 were the third lowest on record (only the 1987 and 1992 densities were lower, Fig. 3; Table 7).

TABLE 4. Estimated number of cunner, tautog, and anchovy eggs entrained each year at MNPS and the volume of cooling water on which the entrainment estimates were based (two-unit operational period: 1976-85; three-unit operational period: 1986-93).

	Cunner		Tautog		Anchovy	
Year	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)
1979	1,534	728	705	728	215	711
1980	2,302	806	1,273	806	91	795
1981	1,736	816	1,735	816	172	799
1982	2,726	853	1,486	853	234	843
1983	2,631	798	1,180	798	618	786
1984	2,031	827	1,369	827	652	812
1985	2,802	831	1,784	831	20	825
1986	2,932	1,870	3,907	1,870	517	1,846
1987	4,533	1,784	3,740	1,784	37	1,752
1988	4,386	1,953	2,813	1,953	16	1,920
1989	3,885	1,643	3,094	1,643	5	1,611
1990	3,651	1,823	2,185	1,823	28	1,795
1991	4,758	1,265	1,589	1,265	147	1,247
1992[b]	2,754	1,565	1,390	1,565	17	1,537
1993	5,750	1,748	2,168	1,748	237	1,728

[a] Volume was determined from the condenser cooling water flow at MNPS during the season of occurrence for each taxa.

[b] Corrections have been made to the 1992 estimates due to an error in calculating volume of cooling water during August, and September.

TABLE 5. Estimated number of anchovy, winter flounder, American sand lance and grubby larvae entrained each year at MNPS and the volume of cooling water on which the entrainment estimates were based (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

	Anchovy		Winter Flounder		American sand lance		Grubby	
Year	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)	No. entrained ($x10^6$)	Volume (m^3)[a] ($x 10^6$)	No. entrained ($x10^6$)	Volume (m^3)[a] ($x10^6$)
1976	419	616	108	663	20	839	13	625
1977	424	570	31	586	84	983	32	653
1978	173	657	87	491	190	808	11	446
1979	887	552	48	474	154	941	21	534
1980	918	505	176	633	124	1,090	34	702
1981	1,784	633	48	455	90	713	43	414
1982	464	550	170	674	32	1,065	49	629
1983	623	482	219	648	41	1,127	57	704
1984	169	602	88	574	20	981	41	643
1985	712	601	83	528	10	1,031	37	582
1986	1,328	1,259	131	1,353	5	1,734	56	1,286
1987	124	1,161	172	1,324	48	2,186	55	1,370
1988	396	1,338	193	1,382	126	2,036	124	1,273
1989	546	1,201	174	1,046	55	1,927	72	1,110
1990	1,025	1,272	139	1,303	61	2,242	49	1,335
1991	478	786	121	934	7	1,330	34	1,024
1992	174	1,018	514	1,199	32	1,672	76	1,132
1993	220	1,098	45	1,412	50	2,261	54	1,374
1994	-[b]	-	182	1,175	77	2,091	58	1,118

[a] Volume was determined from the condenser cooling water flow at MNPS during the season of occurrence for each taxa.

[b] Not calculated because larvae occur after end of report period (May 1994).

TABLE 6. The Δ-mean[a] density (no.·500 m^{-3}) and 95% confidence interval for American sand lance larvae collected at EN during each report year from June 1976 through May 1994 (two-unit operational period: 1976-77 through 1984-85; three-unit operational period: 1985-86 through 1993-94).

Year	EN
1976-77	94 ± 17
1977-78	318 ± 117
1978-79	119 ± 25
1979-80	111 ± 26
1980-81	136 ± 32
1981-82	21 ± 4
1982-83	27 ± 8
1983-84	18 ± 4
1984-85	9 ± 2
1985-86	3 ± 1
1986-87	13 ± 4
1987-88	41 ± 13
1988-89	31 ± 13
1989-90	24 ± 7
1990-91	7 ± 2
1991-92	18 ± 6
1992-93	28 ± 10
1993-94	43 ± 13

[a] Data seasonally restricted to December - May.

All three-unit Δ-mean densities were below the two-unit Δ-mean average (Fig. 3), but there was no significant difference between densities ($p > 0.05$; Mann-Kendall test).

Juvenile anchovies resulting from the summer spawn are typically captured by trawl sampling from August through October, predominantly in Niantic Bay. Even though anchovies rank fifth among fish caught by trawl, Δ-mean CPUE could not be calculated because catches of anchovies were highly variable. Of all the anchovies caught during the past 18 years, 70% were collected in only two years, 1985-86 and 1986-77 (Appendix II). Anchovies mature within a few months of hatching and live only 1 or 2 years; such short-lived species usually exhibit large oscillations in abundance.

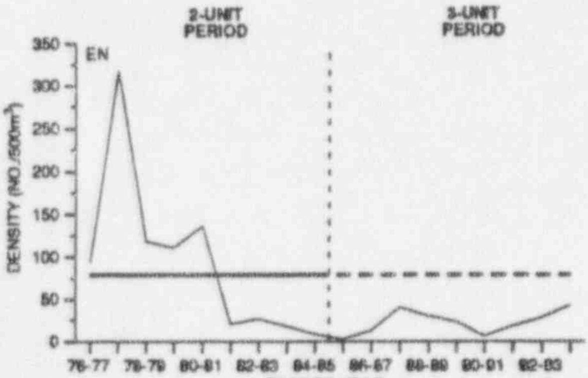

Fig. 2. The annual (——) Δ-mean densities (no.·500 m^{-3}) of American sand lance larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— - -) that is extended over the three-unit operational period as a reference level.

TABLE 7. The Δ-mean[a] density (no.·500 m^{-3}) and 95% confidence interval for anchovy eggs and larvae collected at EN and larvae collected at NB during each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

	EGGS	LARVAE
Year	EN	EN
1976		1,152 ± 419
1977		931 ± 408
1978		483 ± 206
1979	1,447 ± 336	2,168 ± 908
1980	1,245 ± 597	2,430 ± 1,249
1981	1,080 ± 264	5,768 ± 3,326
1982	765 ± 228	816 ± 240
1983	2,257 ± 1,076	1,421 ± 530
1984	4,880 ± 3,680	302 ± 165
1985	145 ± 75	1,102 ± 453
1986	910 ± 547	1,244 ± 893
1987	89 ± 46	126 ± 69
1988	37 ± 33	359 ± 216
1989	54 ± 47	619 ± 416
1990	127 ± 117	1,122 ± 853
1991	476 ± 526	799 ± 801
1992	107 ± 112	178 ± 80
1993	542 ± 400	203 ± 103

[a] Data seasonally restricted to June 15 - August 5 for eggs and July - September for larvae.

Silversides

Along the Connecticut coast, the Atlantic silverside and the inland silverside are the most common shore-zone species. The Atlantic silverside is the most dominant. Essentially, all the silversides caught by trawl were the Atlantic silverside; less than 0.1% caught in the past 18 years were the inland silverside. Historically, more than 80% of the silversides collected by seine were the Atlantic silverside, although this proportion has varied from year to year. Both species are so abundant in the shore-zone that they can be analyzed separately. Trawl and seine catches were highly variable and annual catch indices ranged over two orders of magnitude, which is typical of short lived species such as these species. The Δ-mean trawl CPUE for Atlantic silverside at all stations was

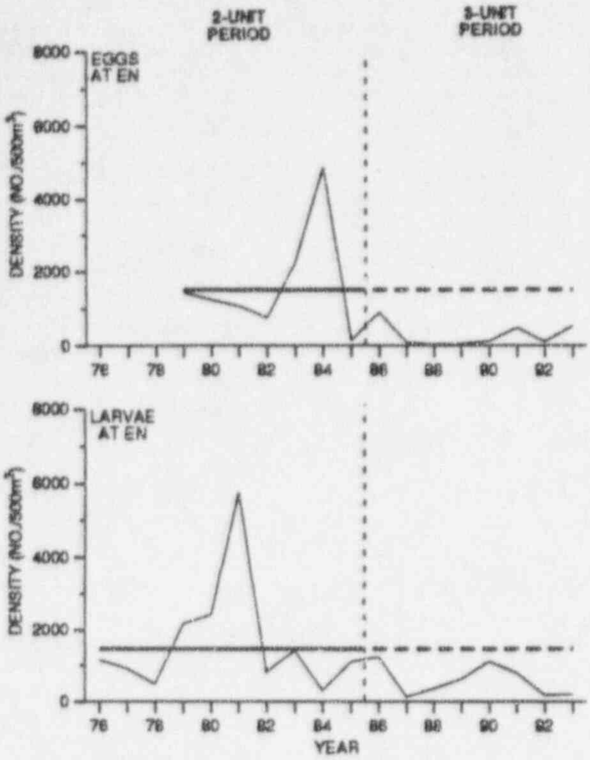

Fig. 3. The annual (——) Δ-mean densities (no.·500 m^{-3}) of anchovy eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level.

TABLE 8. The Δ-mean[a] catch (no.·0.69 km^{-1}) and 95% confidence interval for Atlantic silverside collected by trawl at selected stations during each report year from June 1976 through May 1994 (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1993-94).

Report Year	IN	JC	NB	NR
1976-77	15 ± 16	13 ± 20	6 ± 8	77 ± 283
1977-78	29 ± 92	6 ± 612	18 ± 25	10 ± 21
1978-79	60 ± 105	9 ± 8	8 ± 7	2 ± 1
1979-80	42 ± 276	6 ± 17	0.7 ± 2	4 ± 6
1980-81	8 ± 17	4 ± 5	19 ± 42	3 ± 4
1981-82	6 ± 9	1 ± 0	5 ± 6	6 ± 8
1982-83	2 ± 4	1 ± 2	1 ± 12	12 ± 5
1983-84	2 ± 4	4 ± 1	4 ± 1	1 ± 6
1984-85	2 ± 6	5 ± 11	1 ± 1	1 ± 1
1985-86	7 ± 8	6 ± 8	2 ± 1	3 ± 6
1986-87	5 ± 3	8 ± 7	4 ± 3	110 ± 222
1987-88	3 ± 5	2 ± 2	3 ± 4	15 ± 27
1988-89	2 ± 1	1 ± 0	1 ± 0	25 ± 14
1989-90	1 ± 1	2 ± 2	1 ± 1	12 ± 20
1990-91	1 ± 0	1 ± 0	1 ± 0	17 ± 11
1991-92	[illegible] ± 10	2 ± 1	5 ± 3	19 ± 7
1992-93	115 ± 156	7 ± 2	24 ± 19	24 ± 15
1993-94	5 ± [illegible]	3 ± 2	1 ± 2	3 ± 2

[a] Data seasonally restricted to November - February at IN, NB, and NR, and October - January at JC.

within the historic range (Table 8). In 1993-94, trawl catches at all stations were below the two-unit average (Fig. 4). This was the first year since three-unit operation that the annual Δ-mean catches at NR was below the two-unit average. Although trawl catches were low in 1993-94 there were no significant difference between the two-unit and three-unit CPUE ($p > 0.05$; Mann-Kendall test).

The Δ-mean catches of Atlantic and inland silversides caught by seine in 1993 were all within historic ranges (Table 9; Fig. 5). Catches of Atlantic silverside in 1993 were below the two-unit average at JC and WP and above at GN; inland silversides were above the two-unit average at WP and GN and below at JC. Historically, Atlantic silversides were more abundant than inland silversides. However, during 1990 and 1991 inland silversides were more abundant than Atlantic silversides at JC (Fig. 6). In 1992 and 1993, Atlantic silversides again dominated the catch at JC, but catches of both species remained within historic values. The catch of Atlantic silversides at JC decreased significantly ($p < 0.01$; Mann-Kendall test) during the three-unit period.

To determine if a change in length-frequency distributions occurred after Unit 3 became operational, the length frequencies (expressed as percentages) were examined for the periods before and after three-unit operation and for the 1993-94 study period. The length-frequency distribution for silversides collected by seine and trawl remained similar during these two operational periods (Fig. 7).

Grubby

The grubby is the fourth most abundant larval fish entrained, accounting for 4.5% of all larvae collected at EN from June 1976 through May 1994 (Table 3). Entrainment estimates ranged from 11 million in 1978 to 124 million in 1988 (Table 5). An estimated 58 million larvae were entrained in 1994, which was within the range of previous estimates. The Δ-mean larval density in 1994 was within the range of historic data (Table 10). Three-unit operational annual Δ-mean larval densities fluctuated around the two-unit average and the 1994 Δ-mean was just below the two-unit average (Fig. 8); thus there were no significant ($p > 0.05$; Mann-Kendall test) data trends. The grubby was the seventh-most abundant fish taken by trawl, accounting for more than 2% of the catch at all

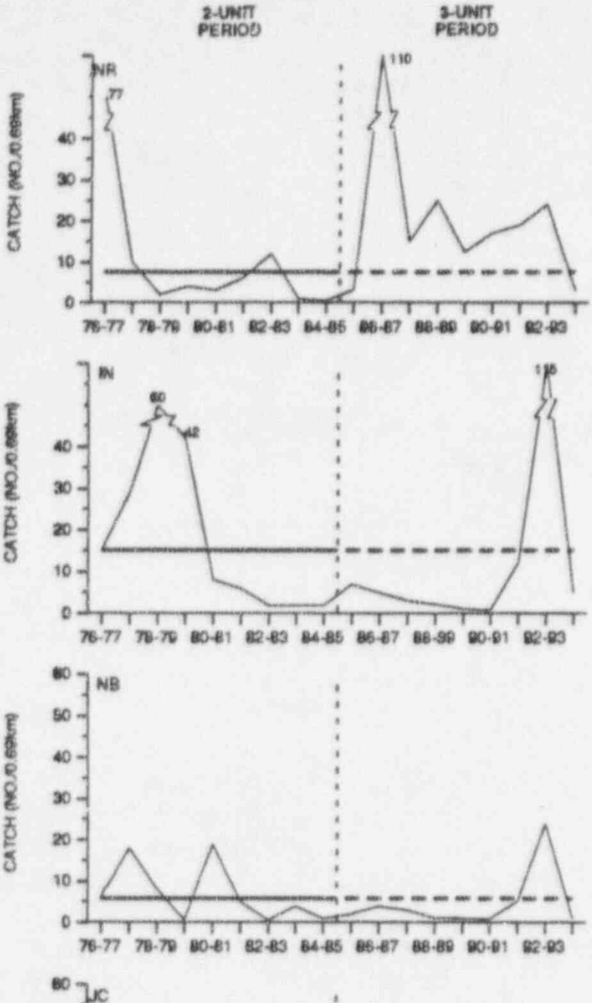

Fig. 4. The annual (——) Δ-mean densities (no.·0.69 km^{-1}) of Atlantic silverside taken by trawl at NR, IN, NB, and JC during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level.

stations over the past 18 years. The largest two peaks in abundance at NR appeared to follow peaks observed in larval densities by 1-2 years. In 1993-94, catches at IN were above the two-unit Δ-mean average, while NR and JC (Fig. 9) were below, but all were within the range of historic values (Table 11). Similar to the larval abundance indices, the three-unit operational annual Δ-mean indices of grubby taken by trawl fluctuated around the two-unit average; thus there was no trend ($p > 0.05$; Mann-Kendall test) in abundance (Fig. 9). The normalized (each period equals 100%) trawl length-frequency distributions of grubby were similar before and after three-unit operation (Fig. 10) although the 1993-94 distribution had greater frequencies of smaller fish.

TABLE 9. The Δ-mean[a] catch (no.·30 m^{-1}) and 95% confidence interval for Atlantic silverside and inland silverside collected by seine during each report year from June 1981 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

Atlantic silverside

Year	JC	GN	WP
1981	152 ± 251	83 ± 78	32 ± 49
1982	114 ± 162	46 ± 109	157 ± 526
1983	397 ± 598	35 ± 49	109 ± 153
1984	29 ± 24	18 ± 11	3 ± 1
1985	19 ± 12	54 ± 45	5 ± 4
1986	172 ± 385	58 ± 46	16 ± 9
1987	109 ± 90	50 ± 27	66 ± 68
1988	96 ± 108	36 ± 34	36 ± 23
1989	70 ± 93	36 ± 34	33 ± 22
1990	83 ± 80	70 ± 44	65 ± 52
1991	38 ± 11	61 ± 25	72 ± 34
1992	78 ± 55	68 ± 30	101 ± 70
1993	60 ± 73	39 ± 36	17 ± 12

Inland silverside

Year	JC	GN	WP
1981	3 ± 3	1 ± 1	1 ± 3
1982	6 ± 16	1 ± 2	9 ± 44
1983	88 ± 243	3 ± 5	1 ± 3
1984	3 ± 2	1 ± 1	0 ± 0
1985	4 ± 8	0 ± 0	0 ± 0
1986	14 ± 21	2 ± 2	1 ± 5
1987	3 ± 2	1 ± 1	0 ± 0
1988	27 ± 54	1 ± 1	1 ± 1
1989	14 ± 16	1 ± 2	6 ± 28
1990	133 ± 234	14 ± 24	43 ± 148
1991	74 ± 37	10 ± 9	9 ± 7
1992	43 ± 27	3 ± 1	1 ± <1
1993	5 ± 5	1 ± 2	2 ± 3

[a] Data seasonally restricted to June - November at all stations.

Tautog

The tautog is an important recreational fish resource in coastal waters from Massachusetts to New Jersey. In Connecticut, commercial landings have increased

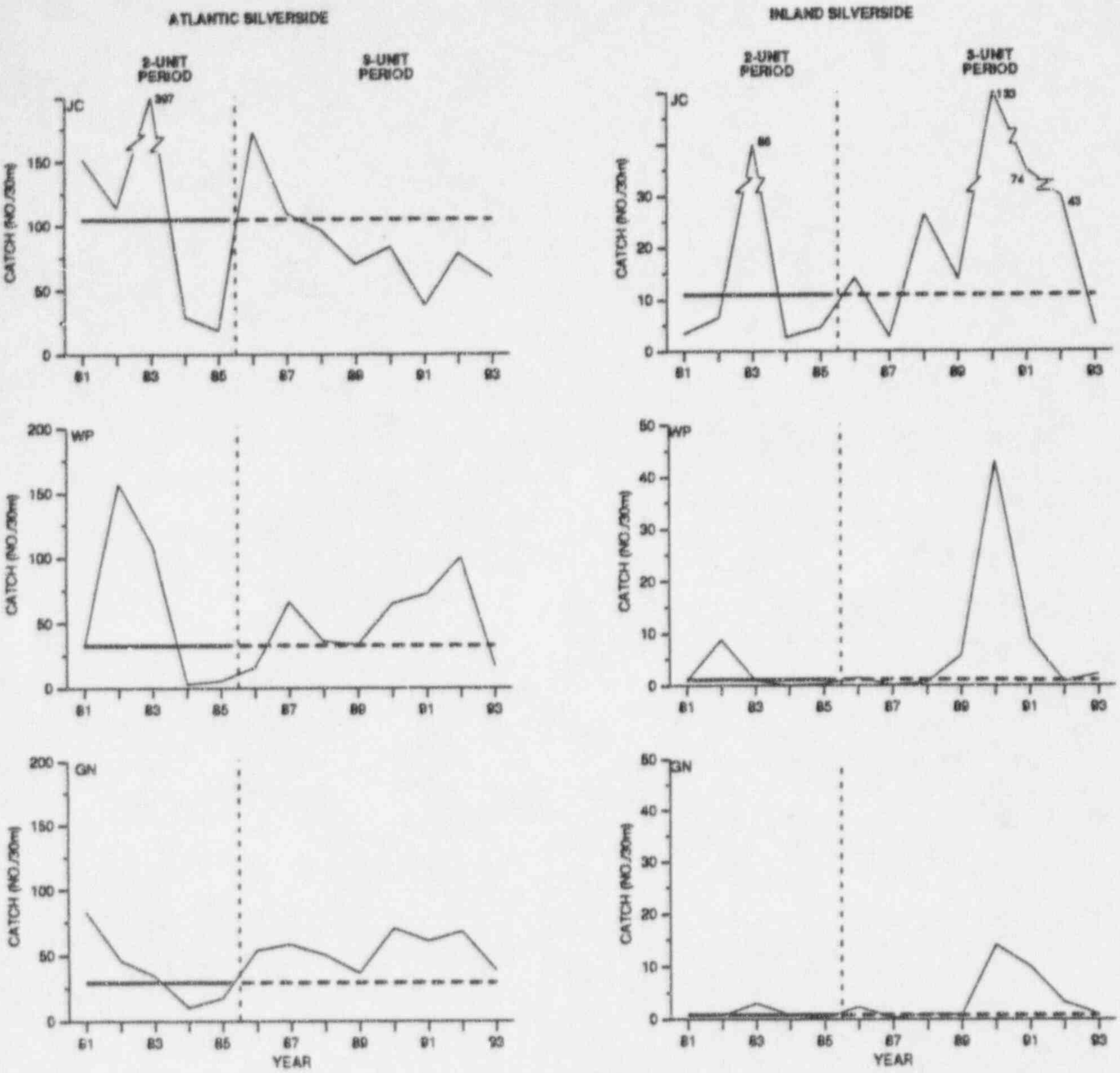

Fig. 5. The annual (——) Δ-mean densities (no.·30 m^{-1}) for Atlantic silverside and inland silverside taken by seine at JC, WP, and GN during two-unit and three-unit operational periods. The two-unit operational period (1981-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level.

significantly during the past decade. Tautog reside in rocky nearshore areas in the spring, summer, and fall (Bigelow and Schroeder 1953; Wheatland 1956; Cooper 1965); juveniles also dwell among macroalgae (Tracy 1910; Briggs and O'Conner 1971). Adults move into deeper water in the winter and remain dormant, while juveniles overwinter in a torpid state near shore (Cooper 1965; Olla et al. 1974).

The tautog is the second most abundant egg taxon entrained and accounted for 29% of fish eggs collected since 1976 (Table 3). At MNPS, tautog eggs are found each year during mid to late June. Eggs are pelagic and hatch in 42-45 hours at 22°C (Williams 1967). Entrainment ranged from 705 million in 1979 to 3.9 billion in 1986 and was estimated at 2.2 billion in 1993 (Table 4). The 1993 Δ-mean density of eggs at EN was below the two-unit operational Δ-mean (Table 12; Fig. 11). During the three-unit operational

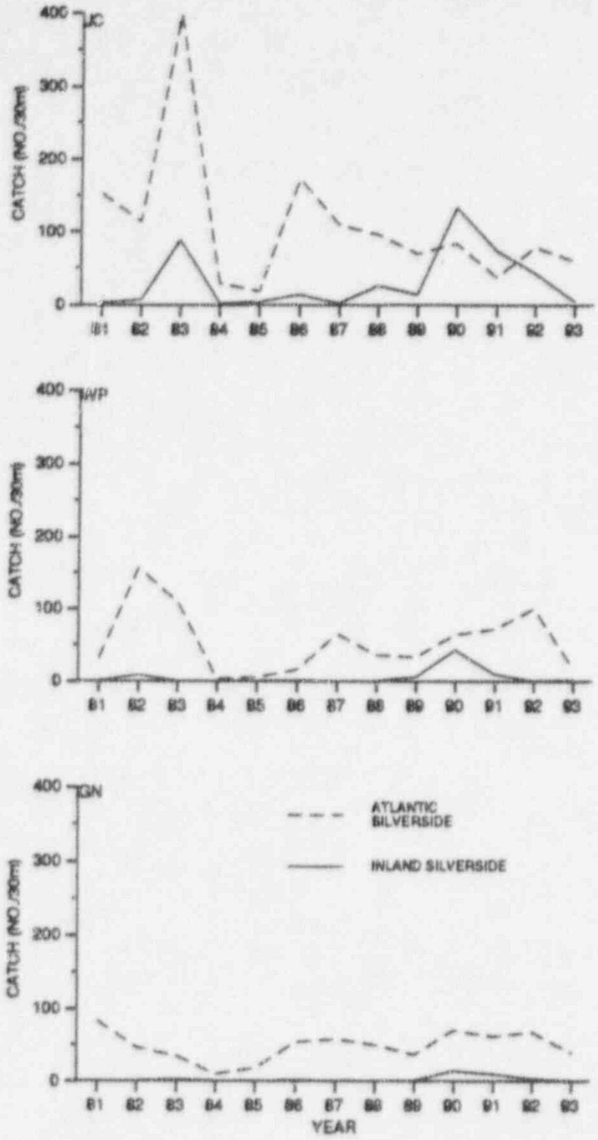

Fig. 6. The annual Δ-mean catch (no.·30 m^{-1}) for the Atlantic silverside and inland silverside taken by seine at JC, WP, and GN from 1981 through 1993.

period there was a significant ($p < 0.05$; Mann-Kendall test) negative trend (−103) in the densities of tautog eggs.

Although tautog eggs are the dominant fish eggs in the plankton samples, tautog larvae accounted for only 2% of all fish larvae collected (Table 3). No correlation was seen between the abundance of eggs and larvae within a year (Fig. 11). Except for 1990 and 1991, larval densities at EN were low since 1984. These low densities were similar to those observed for cunner larvae; however, there was no significant negative trend ($p > 0.05$; Mann-Kendall test).

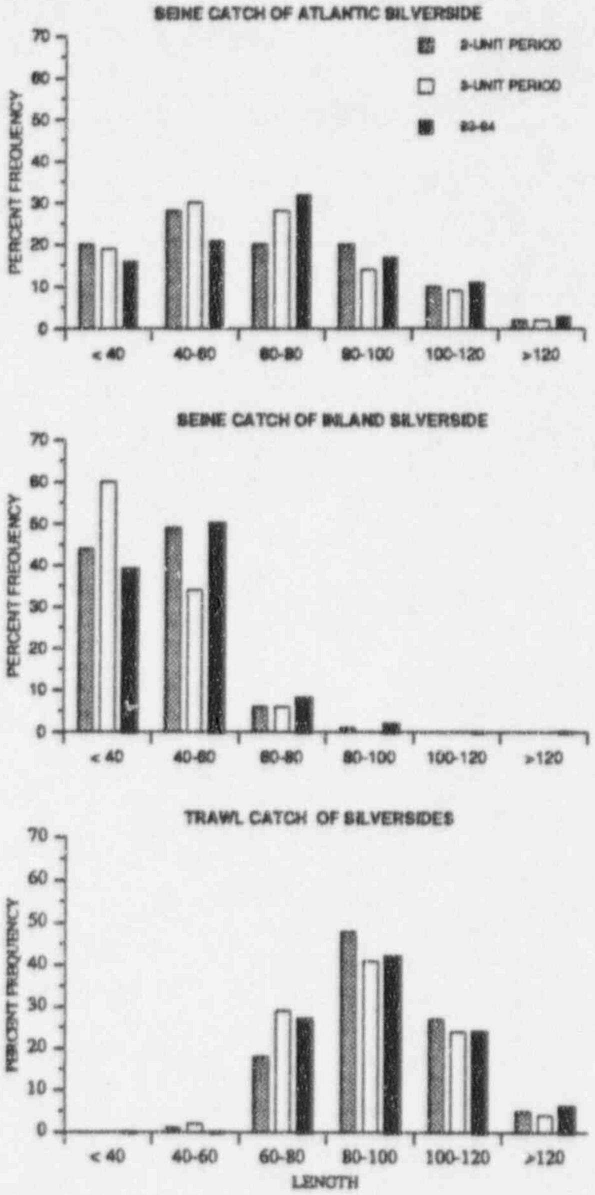

Fig. 7. Length-frequency distribution by 20-mm intervals for the Atlantic silverside and inland silverside taken by seine and trawl during two-unit (1976-85) and three-unit (1986-93) operational periods and the 1993-94 report year.

Historically, tautog catches in the MNPS trawl program have been low. The 1993-94 total catch was among the lowest recorded in the 18-year data series (Appendix II). Because tautog catches were low and the data contained too many zeroes, annual Δ-mean catches could not be calculated. As an alternative, the annual sums of all the catches at each station were used as index of annual abundance (Appendix II).

TABLE 10. The Δ-mean[a] density (no.·500 m^{-3}) and 95% confidence interval for grubby larvae collected at EN during each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

Year	EN
1977	41 ± 9
1978	38 ± 9
1979	36 ± 7
1980	38 ± 7
1981	107 ± 27
1982	72 ± 13
1983	68 ± 19
1984	50 ± 15
1985	68 ± 23
1986	34 ± 10
1987	29 ± 7
1988	95 ± 35
1989	63 ± 18
1990	30 ± 8
1991	24 ± 6
1992	58 ± 17
1993	34 ± 9
1994	48 ± 16

[a] Data seasonally restricted to February - May.

There was a highly significant ($p < 0.001$; Mann-Kendall test) negative slope (-9.2) for the 18-year trawl data series. The CT DEP began conducting a random trawl survey of LIS in 1984. The DEP trawl catch index of tautog also declined from 1984 to 1992. This catch index indicates that the decline in trawl catch near MNPS may be a part of a general decline throughout LIS.

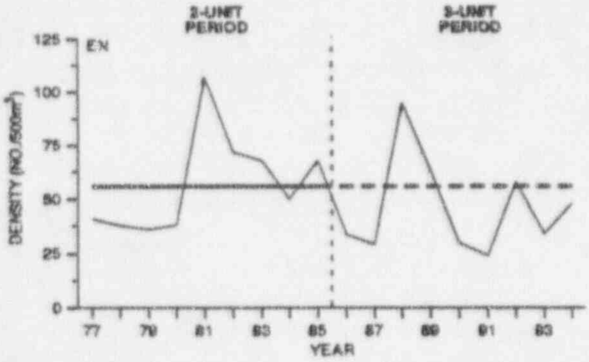

Fig. 8. The annual (——) Δ-mean densities (no.·500 m^{-3}) of grubby larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— - -) that is extended over the three-unit operational period as a reference level.

TABLE 11. The Δ-mean[a] catch (no.·0.69 km^{-1}) and 95% confidence interval for grubby collected by trawl at selected stations during each report year from June 1976 through May 1994 (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1993-94).

Report year	NR	JC	IN
1976-77	0.9 ± 0.3	0.6 ± 0.2	0.6 ± 0.1
1977-78	0.5 ± 0.1	2.2 ± 0.5	1.1 ± 0.2
1978-79	1.2 ± 0.2	2.0 ± 0.6	0.7 ± 0.2
1979-80	3.3 ± 0.9	0.7 ± 0.1	0.9 ± 0.2
1980-81	3.8 ± 1.1	1.1 ± 0.2	2.1 ± 0.6
1981-82	7.5 ± 2.5	1.0 ± 0.2	2.3 ± 0.6
1982-83	11.7 ± 2.7	1.4 ± 0.2	2.2 ± 0.5
1983-84	4.1 ± 0.8	1.7 ± 0.3	1.7 ± 0.3
1984-85	5.9 ± 1.2	1.6 ± 0.3	0.9 ± 0.2
1985-86	2.3 ± 0.5	1.4 ± 0.3	0.7 ± 0.1
1986-87	7.2 ± 2.3	1.1 ± 0.2	0.9 ± 0.2
1987-88	3.7 ± 1.2	1.2 ± 0.2	1.1 ± 0.2
1988-89	10.5 ± 2.3	1.0 ± 0.1	1.4 ± 0.3
1989-90	3.6 ± 2.0	0.4 ± 0.1	1.0 ± 0.3
1990-91	8.0 ± 2.0	0.4 ± 0.1	0.8 ± 0.2
1991-92	3.4 ± 0.5	0.5 ± 0.1	1.0 ± 0.2
1992-93	6.2 ± 2.0	1.4 ± 0.3	1.9 ± 0.3
1993-94	2.2 ± 3.0	0.7 ± 0.5	1.9 ± 3.8

[a] Data seasonally restricted to December - June at IN, and year-round at JC and NR (June - May).

Length-frequency distributions of tautog before and after three-unit operation were calculated. Ages were assigned to length categories based on recent age-length work in LIS reported by Simpson (1989). Young-of-the-year and age-2 tautog accounted for a high proportion of the fish caught after three-unit operation began (Fig. 12). In 1993-94, the proportion of age-2 fish also increased. If egg losses due to entrainment affected recruitment, then juvenile abundance should decrease and the relative abundance of older fish should rise. The percentage of juvenile fish increased during the three-unit operational period, and, therefore, changes in the relative proportion of juveniles and adults was probably unrelated to entrainment losses.

Current values of F (fishing mortality) for tautog were estimated to be about one-third of F_{max} in LIS (Simpson 1989), and, at this rate, the losses resulting from egg and larval entrainment when converted to equivalent-adults would have to amout to more than twice the current annual commercial catches in order to become critical.

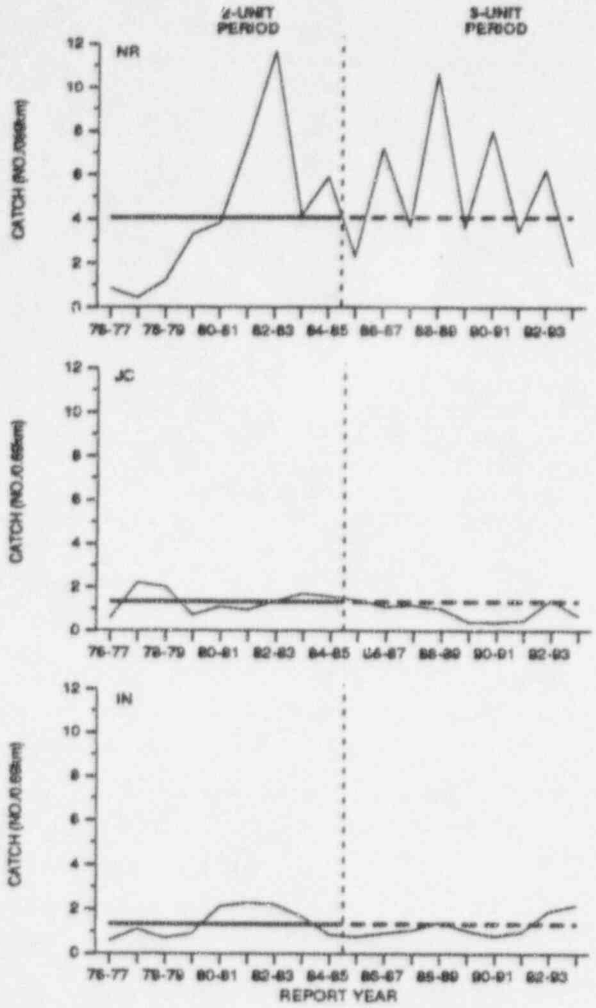

Fig. 9. The annual (——) Δ-mean densities (no.·0.69 km^{-1}) of grubby taken by trawl at NR, JC, and IN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— - -) that is extended over the three-unit operational period as a reference level.

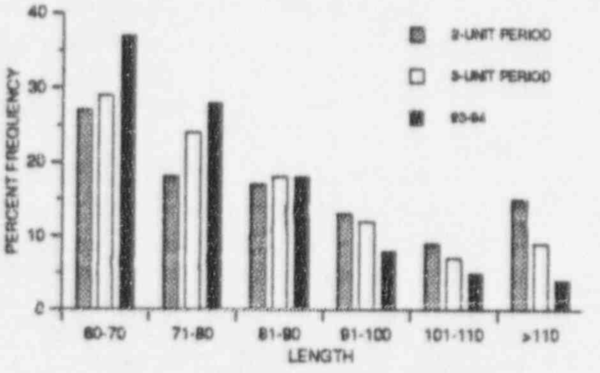

Fig. 10. Length-frequency distribution by 10-mm intervals of grubby taken by trawl during two-unit (1976-85) and three-unit (1986-93) operational periods and the 1993-94 report year.

TABLE 12. The Δ-mean[a] density (no.·500 m^{-3}) and 95% confidence interval for tautog eggs and larvae collected at EN during each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

	EGGS	LARVAE
Year	EN	EN
1976		37 ± 16
1977		36 ± 17
1978		1 ± 1
1979	1,364 ± 231	11 ± 5
1980	2,842 ± 623	46 ± 18
1981	2,647 ± 434	83 ± 36
1982	2,244 ± 434	44 ± 21
1983	2,114 ± 472	33 ± 21
1984	2,157 ± 440	3 ± 2
1985	3,237 ± 1,073	15 ± 12
1986	2,756 ± 794	3 ± 2
1987	3,011 ± 823	7 ± 3
1988	2,269 ± 600	17 ± 10
1989	2,887 ± 1,000	15 ± 7
1990	2,060 ± 933	33 ± 28
1991	1,878 ± 765	99 ± 51
1992	1,449 ± 589	13 ± 4
1993	1,596 ± 567	6 ± 3

[a] Data seasonally restricted to May 23 - August 20 for eggs and June - August for larvae.

Cunner

The cunner inhabits rocky coastal habitats from northern Newfoundland to the mouth of the Chesapeake Bay (Bigelow and Schroeder 1953; Serchuk 1972; Olla et al. 1974, 1979; Dew 1976). Individuals maintain highly localized home ranges and mature in 1 to 2 years (Dew 1976). In cold weather (water temperatures below 8°C), they become torpid (Dew 1976; Olla et al. 1979). Cunner eggs are pelagic and hatch in 2 to 6 days, depending upon water temperature (Williams 1967; Dew 1976).

In the MNPS area, cunner eggs and larvae were found primarily from June through August, and juveniles and adults were caught at all six trawl stations, mostly from spring through fall. Eggs are abundant at EN from May through July and larvae occur from mid-June to mid-July.

The Δ-mean density of cunner eggs in 1993 was within the historic range (Table 13). The 1993 egg entrainment estimate was the highest since the three-unit operational period began (Table 4). The Δ-mean densities of larvae were low, but were within the range of previous values (Table 13). The 1993 Δ-mean density for larvae was below the two-unit operational Δ-mean (Fig. 13). Most (6 of 8 years) of the annual

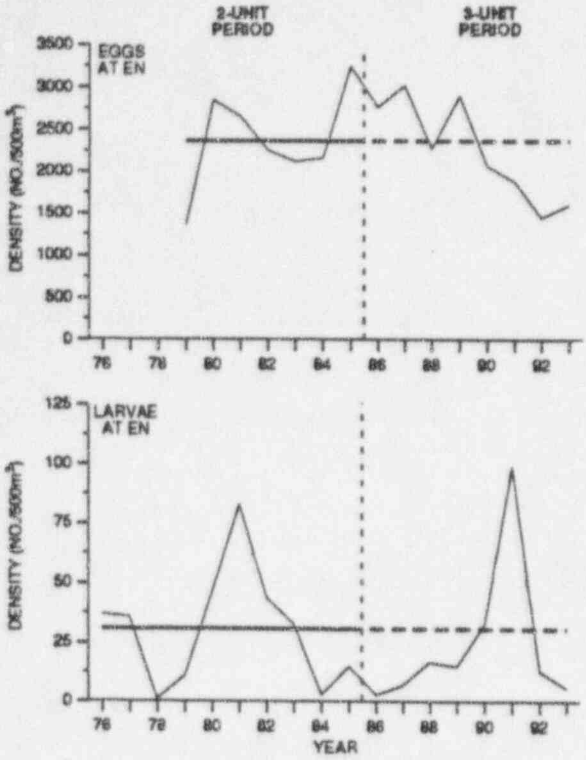

Fig. 11. The annual (——) Δ-mean densities (no.·500 m^{-3}) of tautog eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level. (Note that different vertical scales were used for eggs and larvae.)

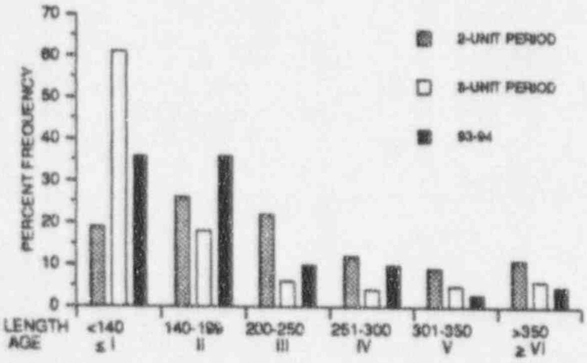

Fig. 12. Frequency distribution by length (mm) and age (determined from age-length key of Simpson 1989) for tautog taken by trawl during two-unit (1976-85) and three-unit (1986-93) operational periods and the 1993-94 report year.

TABLE 13. The Δ-mean[a] density (no.·500 m^{-3}) and 95% confidence interval for cunner eggs and larvae collected at EN and larvae collected at NB during each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

	EGGS	LARVAE
Year	EN	EN
1976		29 ± 14
1977		58 ± 28
1978		1 ± 0
1979	5,870 ± 1,301	13 ± 5
1980	8,223 ± 1,645	58 ± 19
1981	5,171 ± 882	78 ± 36
1982	5,501 ± 1,377	31 ± 14
1983	7,068 ± 2,679	49 ± 26
1984	5,719 ± 1,246	4 ± 2
1985	7,484 ± 2,659	12 ± 10
1986	2,969 ± 1,082	5 ± 1
1987	5,002 ± 1,644	5 ± 3
1988	5,395 ± 1,756	9 ± 4
1989	6,904 ± 3,077	14 ± 12
1990	4,998 ± 2,250	68 ± 61
1991	6,954 ± 3,228	209 ± 157
1992	4,416 ± 2,238	8 ± 4
1993	5,436 ± 2,364	10 ± 6

[a] Data seasonally restricted to May 22 - July 23 for eggs, and June - August for larvae.

three-unit operational Δ-mean indices were below the two-unit average.

The trawl catch of cunner at IN and JC has been declining since 1980 (Fig. 14) and this low abundance continued during 1993-94 at these two stations (Fig. 14; Table 14). Cunner trawl catches at IN and JC trended significantly ($p < 0.05$; Mann-Kendall test) downward (–3 at IN, –14 at JC) during the two-unit operational period. However, abundance was low throughout the three-unit period. Thus, no trends in abundance were found ($p > 0.05$, Mann-Kendall test). The Δ-mean catch at NB was above the two-unit operational Δ-mean in 1992-93, but was below again this year (Fig. 14).

To determine an age-frequency distribution, ages were assigned based on an age-length key provided by Serchuk (1972). A normalized frequency distribution was calculated for both the two- and three-unit periods and the 1993-94 report year. The distributions for these three periods appeared quite different, as over 50% of the cunner caught during three-unit operation were young-of-the-year (Fig. 15). Although the abun-

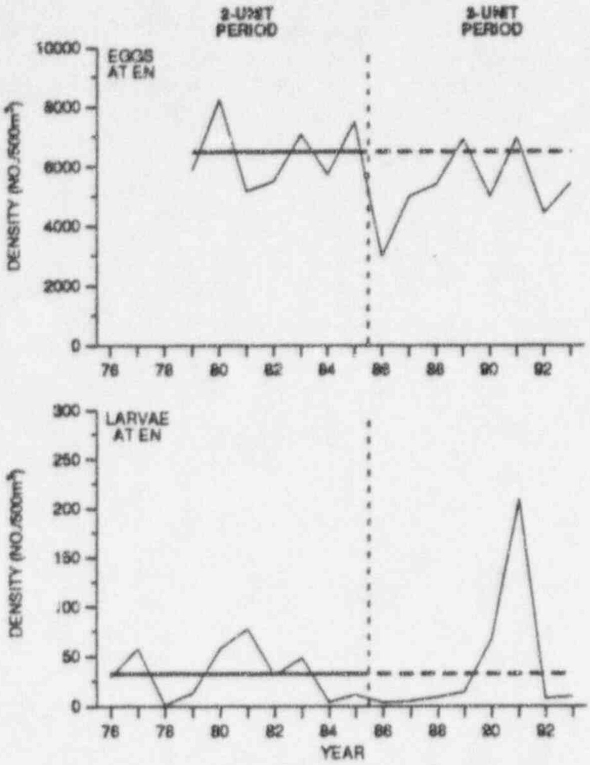

Fig. 13. The annual (——) Δ-mean densities (no.·500 m^{-3}) of cunner eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level. (Note that different vertical scales were used for eggs and larvae.)

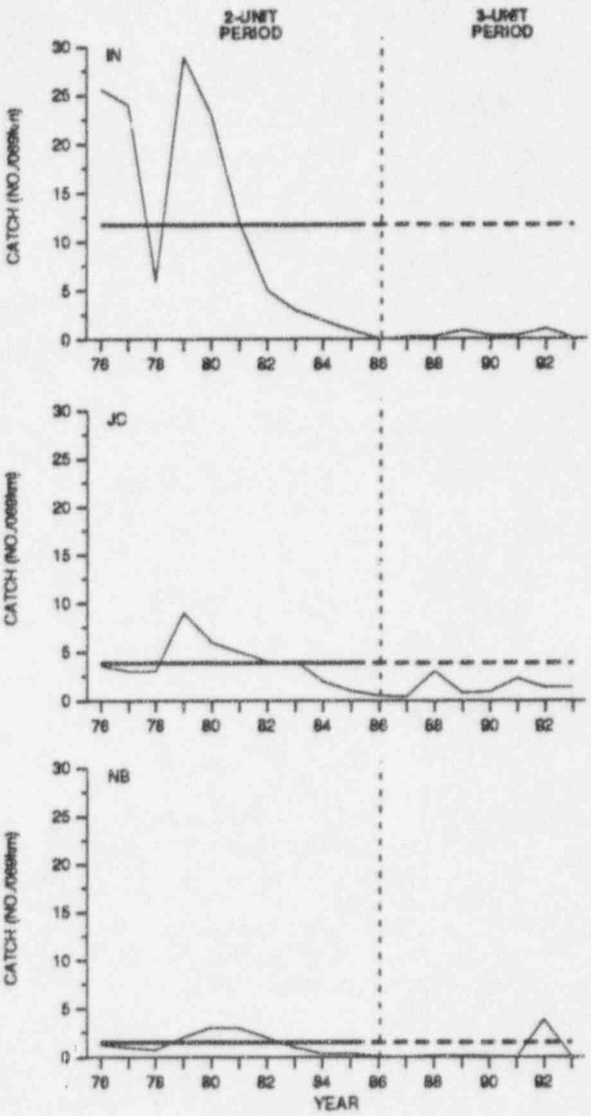

Fig. 14. The annual (——) Δ-mean densities (no.·0.69 km^{-1}) of cunner taken by trawl at IN, JC, and NB during two-unit and three-unit operational periods. The two-unit operational period (1976-1985) Δ-mean density is represented by the flat line (—— – –) that is extended over the three-unit operational period as a reference level.

dance of adult cunner in the MNPS area has decreased over the past decade, the declining trend was regional and was observed in LIS (Smith et al. 1989), Cape Cod Bay (MRI 1994), and Mount Hope Bay (Lawton et al. 1994).

Wrasse Egg Distribution Study

Fish eggs entrained through the MNPS cooling-water system are at risk of suffering high mortality because they are exposed to elevated water temperatures, mechanical stresses, and intermittent chlorination. Mortality caused by entrainment could affect local fish stocks because early life mortality rates influence adult abundance (Cushing and Harris 1973; Cushing 1974; DeAngelis et al. 1977). Over 85% of the eggs entrained at MNPS were wrasse (tautog or cunner) eggs. Thus, studies were conducted in 1990, 1991, and 1993 to determine entrainment mortality of these eggs. Initial results indicated that natural hatching rates and entrainment survival were high. However, subsequent studies indicated that entrainment mortality may be over 95%. This year the spatial distribution of tautog and cunner eggs entrained by MNPS was examined to determine if eggs were mostly from a concentrated distribution located in immediate vicinity of the MNPS intakes or were representative of a more homogenous distribution, including areas outside of Niantic Bay. Station EN

TABLE 14. The Δ-mean[a] catch (no.·0.69 km^{-1}) and 95% confidence interval for cunner collected by trawl at selected stations during each report year from June 1976 through May 1994 (two-unit operational period: 1976-85; three-unit operational period: 1986-94).

Year	IN	JC	NB
1976	26.0 ± 19.0	4.0 ± 2.0	1.0 ± 0.7
1977	24.0 ± 23.0	3.0 ± 1.0	1.0 ± 0.6
1978	6.0 ± 3.7	3.0 ± 1.4	0.7 ± 0.3
1979	29.0 ± 23.0	9.0 ± 5.0	2.0 ± 1.0
1980	23.0 ± 16.0	6.0 ± 2.0	3.0 ± 1.2
1981	12.0 ± 10.0	5.0 ± 2.2	3.0 ± 0.9
1982	5.0 ± 3.0	4.0 ± 2.0	2.0 ± 0.9
1983	3.0 ± 1.3	4.0 ± 2.0	1.0 ± 0.6
1984	2.0 ± 1.0	2.0 ± 1.0	0.4 ± 0.2
1985	1.0 ± 0.6	1.0 ± 0.5	0.4 ± 0.7
1986	0.1 ± 0.2	0.5 ± 0.4	0.1 ± 0.1
1987	0.2 ± 0.2	0.4 ± 0.2	0.0 ± 0.0
1988	0.3 ± 0.1	3.0 ± 3.4	0.2 ± 0.1
1989	0.9 ± 0.4	0.8 ± 0.4	0.2 ± 0.2
1990	0.4 ± 0.1	0.9 ± 0.2	0.1 ± 0.1
1991	0.4 ± 0.1	2.3 ± 0.7	0.0 ± 0.0
1992	1.0 ± 0.7	1.4 ± 0.5	3.8 ± 1.0
1992	0.1 ± 1.1	1.4 ± 0.7	<0.1 ± 0.1

[a] Data seasonally restricted to May - August at IN, May-September at JC, and April - November at NB.

was sampled in conjunction with four other stations, so that collection densities represented eggs potentially imported into Niantic Bay from the west or east.

Results indicated that the density of eggs at each station were similar and had overlapping 95% confidence intervals for the geometric mean densities at each station (Fig. 16). The lack of localized egg concentrations was confirmed by the results of paired comparisons between station EN and the other stations (NB, LI, SS, and BP) when tested with the Wilcoxon's signed-ranks test (Sokal and Rohlf 1969). Although the number of paired comparisons was rather low (12 pairs for NB and LI and 6 for SS and BP), no significant ($p \leq 0.05$) differences were detected between station EN and the other four stations. These data suggested that eggs were not concentrated near MNPS and entrainment densities of tautog and cunner eggs were representative of a more homogenous distribution, including areas outside of Niantic Bay. Therefore, potential entrainment losses are minimized due to an apparently large standing stock of tautog and cunner eggs in the Millstone area and, possibly, over larger areas within LIS.

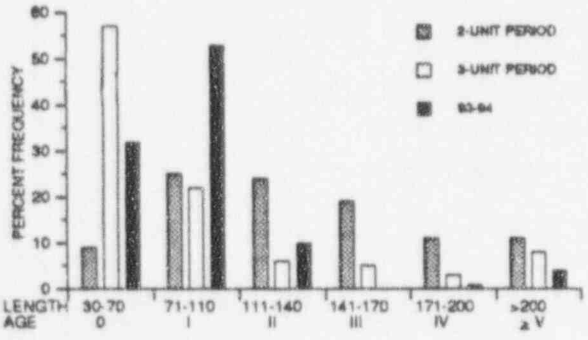

Fig. 15. Frequency distribution by length (mm) and age (determined from age-length key of Serchuk 1972) for cunner taken by trawl during two-unit (1976-85) and three-unit (1986-93) operational periods and the 1993-94 report year.

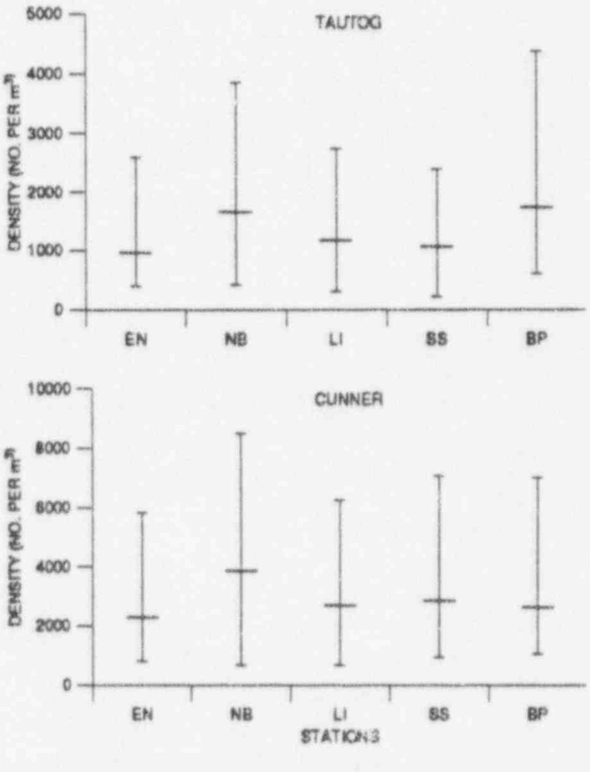

Fig. 16. Comparison of spatial distribution of tautog and cunner eggs in the Millstone area, based on geometric mean densities with 95% confidence intervals for each station during a special study conducted in 1994.

Conclusions

Results of the 1993-94 fish ecology studies and data analyses were similar to results from previous years. In 1993-94, negative abundance trends were found for some life stages in four of the six species examined. These abundance declines do not necessarily suggest or implicate MNPS operation as a causative factor. Rather, overall changes in fish populations this year may be related to a number of factors, including fishing mortality, density-dependent growth, age structure of the population, compensatory mortality, or physical parameters such as air and water temperature and precipitation.

American sand lance larvae, Atlantic silverside juveniles, and cunner and tautog adults exhibited negative trends in abundance. American sand lance inhabit the study area for short periods of time, exhibit large year to year fluctuations, and are probably more affected by events elsewhere. Cunner and tautog adults have declined in trawl catches concurrently with a shift in age structure to juveniles that accounted for a high proportion of the catch. The abundance of tautog and cunner adults in the MNPS area has been decreasing over the past decade. Entrainment of cunner and tautog eggs was identified as the primary potential impact on these fish because more than 85% of the eggs entrained at MNPS were of these two species. The decrease in abundance, however, reflected declining trends in LIS (Smith et al. 1989), Cape Cod Bay (Lawton et al. 1994) and Mount Hope Bay (MRI 1993). The shift in age structure from older to younger fish could not be attributed to entrainment, but may be indicative of increased fishing pressure on older fish. Entrainment losses alone would likely cause a shift in the age distribution towards older fish because of the loss of new recruits, but increased fishing pressure would shift the age structure from old to young. Furthermore, spatial distribution studies conducted in 1994 suggested that these eggs were not concentrated near MNPS, but were representative of a more homogenous distribution, including areas outside of Niantic Bay. This apparently large standing stock of tautog and cunner eggs in the MNPS area minimizes the relative impact due to entrainment mortality.

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APPENDIX I. List of fishes collected in the Fish Ecology sampling programs.

Scientific name	Common name	Trawl	Seine	Ichthyoplankton
Acipenser oxyrhynchus	Atlantic sturgeon	•		
Alosa aestivalis	blueback herring	•	•	
Alosa mediocris	hickory shad	•		
Alosa pseudoharengus	alewife	•	•	•
Alosa sapidissima	American shad	•	•	
Alosa spp.	river herring	•	•	•
Aluterus schoepfi	orange filefish	•		
Ammodytes americanus	American sand lance	•	•	•
Anchoa hepsetus	striped anchovy			•
Anchoa mitchilli	bay anchovy	•	•	•
Anguilla rostrata	American eel	•	•	•
Apeltes quadracus	fourspine stickleback	•	•	•
Bairdiella chrysoura	silver perch	•		
Bothus ocellatus	eyed flounder	•		
Brevoortia tyrannus	Atlantic menhaden	•	•	•
Brosme brosme	cusk	•		
Caranx crysos	blue runner	•	•	
Caranx hippos	crevalle jack	•	•	
Centropristis striata	black sea bass	•		•
Chaetondon ocellatus	spotfin butterflyfish	•		•
Clupeidae	herrings	•		•
Clupea harengus	Atlantic herring	•	•	•
Conger oceanicus	conger eel	•		•
Cyclopterus lumpus	lumpfish	•		•
Cynoscion regalis	weakfish	•	•	•
Cyprinodon variegatus	sheepshead minnow		•	•
Dactylopterus volitans	flying gurnard	•		
Dasyatis centroura	roughtail stingray	•		
Decapterus macarellus	mackerel scad	•		
Decapterus punctatus	round scad	•		
Enchelyopus cimbrius	fourbeard rockling	•		•
Etropus microstomus	smallmouth flounder	•		•
Eucinostomus lefroyi	mottled mojarra		•	
Fistularia tabacaria	bluespotted cometfish	•		
Fundulus diaphanus	banded killifish		•	
Fundulus heteroclitus	mummichog	•	•	
Fundulus luciae	spotfin killifish		•	
Fundulus majalis	striped killifish		•	
Gadidae	codfishes	•		•
Gadus morhua	Atlantic cod	•		•
Gasterosteus aculeatus	threespine stickleback	•	•	•
Gasterosteus wheatlandi	blackspotted stickleback	•	•	•
Godiidae	gobies	•		•
Gobiosoma ginsburgi	seaboard goby	•		
Hemitripterus americanus	sea raven	•		•
Hippocampus erectus	lined seahorse	•		•
Labridae	wrasses			•
Lactophrys spp.	boxfish	•		
Leiostomus xanthurus	spot	•		
Liparis spp.	seasnail	•		•
Lophius americanus	goosefish	•		•
Lucania parva	rainwater killifish	•	•	
Macrozoarces americanus	ocean pout	•		
Melanogrammus aeglefinus	haddock	•		
Menticirrhus saxatilis	northern kingfish	•	•	•
Menidia beryllina	inland silverside	•	•	
Menidia menidia	Atlantic silverside	•	•	•
Merluccius bilinearis	silver hake	•	•	•
Microgadus tomcod	Atlantic tomcod	•		•
Monacanthus hispidus	planehead filefish	•		

APPENDIX I. (continued).

Scientific name	Common name	Trawl	Seine	Ichthyoplankton
Monocanthus spp.	filefish	•		
Morone americana	white perch	•		•
Morone saxatilis	striped bass	•	•	
Mugil cephalus	striped mullet	•	•	•
Mugil curema	white mullet		•	
Mullus auratus	red goatfish	•		
Mustelis canis	smooth dogfish	•		
Myliobatis freminvillei	bullnose ray	•		
Myoxocephalus aenaeus	grubby	•	•	•
Myoxocephalus octodecemspinosus	longhorn sculpin	•		•
Myoxocephalus spp.	sculpin	•		
Ophidiidae	cusk-eels	•		
Ophidion marginatum	striped cusk-eel	•	•	•
Ophidion welshi	crested cusk-eel	•		
Opsanus tau	oyster toadfish	•		
Osmerus mordax	rainbow smelt	•	•	•
Paralichthys dentatus	summer flounder	•		•
Paralichthys oblongus	fourspot flounder	•		•
Peprilus triacanthus	butterfish	•	•	•
Petromyzon marinus	sea lamprey	•		
Pholis gunnellus	rock gunnel	•	•	•
Pleuronectes americanus	winter flounder	•	•	•
Pleuronectes ferrugineus	yellowtail flounder	•		•
Pollachius virens	pollock	•		•
Pomatomus saltatrix	bluefish	•	•	
Priacanthus arenatus	bigeye	•		
Priacanthus cruentatus	glasseye snapper	•		
Pristigenys alta	short bigeye	•		
Prionotus carolinus	northern searobin	•	•	•
Prionotus evolans	striped searobin	•	•	•
Pungitius pungitius	ninespine stickleback	•	•	•
Raja eglanteria	clearnose skate	•		
Raja erinacea	little skate	•		
Raja ocellata	winter skate	•		
Salmo trutta	brown trout	•		
Sciaenidae	drums			•
Scophthalmus aquosus	windowpane	•	•	•
Scomber scombrus	Atlantic mackerel	•		•
Scyliorhinus retifer	chain dogfish	•		
Selar crumenopthalmus	bigeye scad	•		
Selene setapinnis	Atlantic moonfish	•		
Selene vomer	lookdown	•	•	
Synodus foetens	inshore lizardfish	•		
Sphyraena borealis	northern sennet	•		
Sphoeroides maculatus	northern puffer	•	•	•
Squalus acanthias	spiny dogfish	•		
Stenotomus chrysops	scup	•		•
Strongylura marina	Atlantic needlefish		•	
Syngnathus fuscus	northern pipefish	•	•	•
Tautogolabrus adspersus	cunner	•	•	•
Tautoga onitis	tautog	•	•	•
Trachinotus falcatus	permit	•	•	
Trachurus lathami	rough scad	•		
Trachinocephalus myops	snakefish	•		
Trinectes maculatus	hogchoker	•		
Ulvaria subbifurcata	radiated shanny	•		•
Upeneus parvus	dwarf goatfish	•		
Urophycis chuss	red hake	•	•	
Urophycis tenuis	white hake	•	•	
Urophycis spp.	hake	•	•	•

APPENDIX II. Total number of samples collected and number of fish caught by trawl for each report year (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1993-94).

Year	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94
Number of samples	468	468	468	468	468	467	474	480	468	468	468	465	468	468	468	468	468	468
Taxon [a]																		
P. americanus	7,415	6,045	7,236	11,442	13,296	10,749	19,201	12,560	13,260	9,849	9,321	8,877	13,440	8,690	9,378	8,511	9,828	6,856
S. chrysops	1,918	4,040	2,556	4,094	3,844	3,403	4,896	5,268	4,206	2,640	5,205	3,632	3,294	2,869	10,497	25,287	9,710	890
S. aquosus	1,480	1,296	875	1,508	2,016	1,518	3,517	2,475	2,199	2,483	1,655	1,966	2,399	2,735	1,656	876	1,519	2,389
Raja spp.	661	579	362	402	954	696	2,797	2,493	1,583	3,801	2,207	2,183	2,864	2,437	2,858	2,872	1,892	2,461
Anchoa spp.	979	580	2,226	16	109	578	38	109	157	10,003	8,038	292	496	1,241	31	1,557	80	11
Menidia spp.	2,152	1,647	1,463	1,340	882	501	518	583	322	519	3,438	698	982	485	474	1,346	3,567	428
M. aenaeus	266	636	297	342	632	870	996	672	477	341	727	434	989	615	640	451	857	520
Gadidae	112	326	230	211	3,296	1,424	476	481	562	630	168	593	88	84	121	106	207	444
Prionotus spp.	338	322	138	313	405	661	1,059	422	371	395	436	159	356	1,277	363	435	327	173
T. adspersus	838	875	400	1,399	940	840	611	362	248	119	147	61	205	109	103	141	401	76
P. triacanthus	37	44	407	174	44	69	182	244	19	135	132	111	1,831	179	1,878	426	1,302	264
P. dentatus	286	141	92	75	122	240	250	269	1,937	281	653	617	360	80	393	403	634	465
Urophycis spp.	99	87	103	69	163	313	615	286	251	272	286	164	174	141	335	91	964	231
M. bilinearis	425	163	69	134	558	220	382	147	100	175	197	118	73	321	124	179	337	45
E. microstomus	43	7	0	3	31	91	94	56	85	218	640	190	359	62	494	394	694	217
G. aculeatus	30	12	47	77	206	103	63	218	1,102	116	354	405	94	10	15	447	172	133
P. gunnellus	85	106	99	65	251	273	302	145	127	151	186	203	407	189	155	126	152	224
S. fuscus	43	54	49	88	151	264	232	202	254	196	207	275	321	85	154	134	175	325
T. onitis	229	283	263	270	146	228	239	140	119	134	215	87	162	85	185	111	131	111
O. mordax	111	286	90	5	123	63	89	26	227	391	257	249	152	26	48	35	334	9
H. americanus	34	48	39	148	278	410	557	377	125	41	45	11	3	7	12	38	1	1
A. quadracus	10	6	24	27	194	765	76	11	112	130	107	52	31	11	18	100	69	90
B. tyrannus	1	14	11	1	1	1	0	1	0	34	10	4	1	1,320	5	205	64	21
C. striata	33	9	3	4	10	63	23	38	30	80	412	16	53	69	130	94	60	10
P. oblongus	31	7	21	11	51	32	138	34	81	66	72	28	123	155	92	28	55	56
M. octodecemspinosus	11	10	97	40	30	145	172	51	20	13	12	5	12	18	56	222	9	3
A. pseudoharengus	11	272	13	17	4	15	5	26	4	16	208	1	4	3	14	41	35	21
O. tau	98	21	7	18	31	35	25	23	24	32	56	51	[illegible]	30	55	17	5	10
A. americanus	5	59	128	36	117	14	19	11	19	6	11	29	1	1	1	1	2	1
A. rostrata	19	16	8	5	10	37	29	24	22	34	28	22	20	5	15	8	2	4
C. lumpus	19	11	28	58	11	0	14	1	29	1	1	44	6	1	7	6	21	9
Liparis spp.	9	27	10	10	18	33	15	16	11	3	18	8	12	22	2	3	22	6
S. maculatus	16	10	1	0	9	14	16	15	7	7	3	1	9	14	26	50	26	4
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	4	11	26	26	34	52	30
C. regalis	9	21	4	2	2	45	7	0	1	5	36	5	14	9	6	1	5	3
C. harengus	1	9	13	0	0	1	0	2	9	63	10	2	1	2	1	10	19	21
A. sapidissima	33	6	1	5	40	12	0	29	0	0	1	1	9	5	3	7	1	0
H. erectus	0	0	0	0	0	0	0	1	4	7	20	12	6	4	17	67	4	0
Clupeidae	2	1	0	0	0	0	0	0	0	110	0	0	0	0	0	0	0	0
Gobiidae	3	0	0	0	4	0	0	3	9	7	2	5	10	2	23	17	1	25
M. canis	2	5	45	11	1	5	4	6	0	2	2	1	2	2	4	2	0	0
M. hispidus	3	6	6	4	0	0	8	1	8	9	2	2	2	11	9	5	1	11
M. americanus	5	7	9	2	2	2	2	2	3	1	0	6	2	2	1	10	2	0
S. setapinnis	0	0	0	0	0	0	0	0	1	0	0	0	30	0	0	0	1	0
D. volitans	3	0	0	0	0	1	3	1	0	1	3	4	1	2	4	3	3	2
A. schoepfi	0	2	2	1	1	0	0	1	1	2	2	0	3	4	6	1	1	2
O. marginatum	0	0	0	0	0	0	0	0	1	2	4	4	4	0	4	3	2	5
P. saltatrix	1	1	0	2	1	2	3	3	0	0	0	2	1	0	1	11	0	0
Men. saxatilis	0	1	0	1	0	3	1	0	0	0	4	2	1	0	3	0	1	5
L. xanthurus	5	6	0	0	0	0	2	0	0	3	1	0	5	0	0	0	0	0
A. aestivalis	3	11	8	12	4	1	1	17	5	2	4	2	2	0	0	10	4	0
P. ferrugineus	7	5	5	2	3	15	6	0	4	0	0	23	0	3	0	3	2	4
M. americana	8	17	3	5	8	2	1	0	0	0	0	0	5	11	1	3	3	5

APPENDIX II (continued).

Year	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94
Taxon [a]																		
F. tabacaria	2	3	0	0	3	0	1	0	8	1	2	0	1	3	22	9	5	4
C. oceanicus	1	0	0	0	1	0	0	0	2	0	1	1	1	3	0	1	1	6
S. vomer	1	2	0	0	0	0	0	0	0	0	1	1	11	1	0	0	0	1
U. subbifurcata	0	2	0	0	1	1	0	0	0	1	1	1	4	0	1	2	3	0
L. americanus	2	0	0	0	1	0	1	1	0	0	1	4	0	0	2	3	0	1
M. auratus	0	0	1	0	0	0	2	0	0	0	1	0	4	0	0	0	1	6
P. pungitius	0	0	0	0	1	2	0	0	5	1	5	0	0	0	0	0	0	0
Gasterosteidae	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fundulus spp.	0	0	0	0	0	5	2	0	0	2	1	0	1	0	0	0	0	1
S. borealis	0	0	0	0	0	0	0	1	1	2	0	1	6	0	0	0	0	0
P. cruentatus	0	0	0	0	0	1	0	2	3	1	0	0	4	5	3	0	0	0
G. wheatlandi	0	0	0	0	0	1	1	1	0	1	2	0	1	0	0	1	1	0
S. foetens	0	1	4	0	0	3	1	0	0	0	0	0	0	1	1	0	0	0
C. ocellatus	0	0	0	0	1	0	0	1	0	0	1	0	1	0	3	0	0	0
E. cimbrius	0	0	0	0	0	1	0	0	0	0	1	1	1	0	0	1	2	0
Mor. saxatilis	0	0	2	1	0	1	1	0	0	1	0	0	0	4	1	0	0	2
P. arenatus	0	0	0	0	0	0	0	0	2	1	0	0	1	1	3	1	0	0
T. lathami	0	0	0	4	0	0	0	0	0	0	0	0	4	0	0	0	0	0
T. maculatus	3	1	0	0	0	0	0	0	0	1	2	1	0	0	0	0	0	0
Lactophrys spp.	0	0	0	0	0	0	0	0	3	0	0	0	0	0	3	3	0	1
S. acanthias	0	0	0	0	0	0	1	0	1	0	0	0	0	1	0	4	0	1
C. crysos	0	0	0	0	1	0	1	0	1	2	0	0	0	0	0	0	0	0
M. cephalus	0	0	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0
P. alta	0	0	0	0	0	1	0	0	2	1	1	0	0	1	0	0	0	0
C. hippos	0	0	0	0	0	0	1	0	0	1	0	0	0	2	0	0	0	0
S. scombrus	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	1	0	0
U. parvus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
D. punctatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0
L. parva	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0
A. mediocris	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
C. variegatus	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0
D. macarellus	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
M. aeglefinus	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0
Monacanthus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
A. maculatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
A. oxyrhynchus	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
B. brosme	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
B. chrysoura	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
B. ocellatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
D. centroura	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
M. freminvillei	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Myoxocephalus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Ophidiidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
P. marinus	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
S. crumenopthalmus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
S. marina	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
S. retifer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
S. trutta	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
T. falcatus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
T. myops	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Total	17,941	18,147	17,497	22,469	29,010	24,773	37,699	27,860	28,169	33,546	35,568	21,681	29,524	23,478	30,471	44,957	33,768	16,665

[a] Fish identified to the lowest practical taxon.

APPENDIX III. Total number of samples collected and number of fish caught by trawl at each station (June 1976 - May 1994).

Station	JC	NR	NB	TT	BR	IN	TOTAL
Number of samples:	1,406	1,406	1,406	1,406	1,406	1,406	8,436
Taxon [a]							
P. americanus	15,903	67,974	22,620	26,916	22,508	30,033	185,954
S. chrysops	9,047	566	34,994	15,844	10,306	27,492	98,249
S. aquosus	1,849	3,607	3,159	4,485	14,713	6,749	34,562
Raja spp.	2,105	41	5,344	9,113	12,479	5,020	34,102
Anchoa spp.	3,343	1,177	18,226	339	25	3,431	26,541
Menidia spp.	4,620	6,431	2,210	1,736	486	5,862	21,345
M. aenaeus	1,486	5,182	611	655	918	1,910	10,762
Gadidae	2,415	850	2,862	1,080	290	2,062	9,559
Prionotus spp.	131	1,025	707	1,356	3,734	997	7,950
T. adspersus	1,956	395	632	285	533	4,074	7,875
P. triacanthus	73	13	1,047	1,959	2,955	1,431	7,478
P. dentatus	1,065	1,930	1,057	1,850	314	1,082	7,298
Urophycis spp.	567	87	487	435	2,445	623	4,644
M. bilinearis	200	10	538	681	1,579	759	3,767
E. microstomus	272	36	745	351	1,682	592	3,678
G. aculeatus	2,187	1,369	18	12	6	12	3,604
P. gunnellus	1,655	559	330	277	48	377	3,246
S. fuscus	921	1,640	171	126	137	214	3,209
T. onitis	795	826	313	216	277	708	3,135
O. mordax	1,422	302	245	174	101	277	2,521
H. americanus	444	82	405	298	541	405	2,175
A. quadracus	268	1,560	1	1	1	2	1,833
B. tyrannus	536	1,073	40	6	7	32	1,694
C. striata	96	212	75	53	76	625	1,137
P. oblongus	0	12	73	14	952	30	1,081
M. octodecemspinosus	3	0	20	53	834	16	926
A. pseudoharengus	10	67	58	30	277	268	710
O. tau	9	574	0	0	0	13	596
A. americanus	20	96	6	29	301	9	461
A. rostrata	43	234	0	20	3	8	308
C. lumpus	168	10	21	8	3	57	267
Liparis spp.	26	12	38	40	102	27	245
S. maculatus	19	103	17	10	18	61	228
Alosa spp.	7	47	18	24	77	10	183
C. regalis	22	1	30	11	77	34	175
C. harengus	70	4	44	14	31	6	169
A. sapidissima	8	17	56	12	38	22	153
H. erectus	53	63	9	3	2	12	142
Clupeidae	0	1	0	1	0	111	113
Gobiidae	4	103	2	0	0	2	111
M. canis	10	1	40	3	36	4	94
M. hispidus	21	2	13	12	25	17	90
A. aestivalis	1	29	17	11	14	14	86
P. ferrugineus	0	2	0	8	72	0	82
M. americana	10	25	6	2	6	23	72
F. tabacaria	40	13	1	1	0	9	64
M. americanus	0	0	0	1	55	2	58
S. setapinnis	16	0	8	2	0	6	32
D. volitans	2	15	0	0	0	14	31
A. schoepfi	10	0	4	2	6	7	29
O. marginatum	5	6	2	5	10	1	29
P. saltatrix	3	5	6	8	5	1	28
L. xanthurus	4	0	8	0	4	6	22
Men. saxatilis	1	3	4	5	1	8	22

APPENDIX III (continued).

Station Taxon [a]	JC	NR	NB	TT	BR	IN	TOTAL
P. cruentatus	2	0	2	5	2	8	19
S. vomer	1	1	15	0	0	1	18
C. oceanicus	1	4	3	3	7	0	18
U. subbifurcata	3	0	1	1	11	1	17
L. americanus	1	0	2	2	9	2	16
M. auratus	1	0	0	2	1	11	15
P. pungitius	10	3	0	0	0	1	14
Fundulus spp.	1	12	0	0	0	0	13
Gasterosteidae	2	11	0	0	0	0	13
Mor. saxatilis	0	13	0	0	0	0	13
S. borealis	5	6	0	0	0	0	11
S. foetens	0	3	0	2	5	1	11
Lactophrys spp.	7	2	0	0	0	1	10
G. wheatlandi	7	2	0	0	0	0	9
P. arenatus	0	1	1	0	2	5	9
S. acanthias	0	0	1	0	7	0	8
T. lathami	4	0	3	0	0	1	8
T. maculatus	5	2	0	0	0	1	8
C. ocellatus	2	2	1	0	0	2	7
E. cimbrius	2	0	0	0	3	2	7
P. alta	3	0	0	1	1	1	6
C. crysos	0	0	2	0	1	2	5
C. hippos	0	0	0	0	0	4	4
S. scombrus	0	0	1	1	1	1	4
M. cephalus	0	2	1	0	0	0	3
D. punctatus	0	0	0	0	3	0	3
L. parva	0	0	0	0	3	0	3
A. mediocris	1	0	0	0	1	0	2
C. variegatus	0	1	0	1	0	0	2
D. macarellus	1	0	1	0	0	0	2
M. aeglefinus	1	0	0	1	0	0	2
Monacanthus spp.	1	1	0	0	0	0	2
Myoxocephalus spp.	0	1	0	0	0	0	1
A. maculatus	1	0	0	0	0	0	1
A. oxyrhynchus	0	0	1	0	0	0	1
B. brosme	0	0	0	1	0	0	1
B. chrysoura	0	0	0	0	1	0	1
B. ocellatus	0	1	0	0	0	0	1
D. centroura	1	0	0	0	0	0	1
M. freminvillei	0	0	1	0	0	0	1
Ophidiidae	0	0	0	0	1	0	1
P. marinus	0	0	0	1	0	0	1
S. crumenopthalmus	0	0	0	1	0	0	1
S. marina	0	0	0	0	1	0	1
S. retifer	0	1	0	0	0	0	1
S. trutta	0	1	0	0	0	0	1
T. falcatus	0	0	0	0	0	1	1
T. myops	1	0	0	0	0	0	1
U. parvus	1	0	0	0	0	0	1
Total	54,007	98,462	97,374	65,610	79,170	95,613	490,236

[a] Fish identified to the lowest practical taxon.

APPENDIX IV. Total number of samples collected and number of fish caught by seine for each report year (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1992-94).

Year	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93	93-94
Number of samples	66	72	72	72	72	72	98	120	174	156	156	156	156	180	180	180	135	123
Taxon [a]																		
Menidia spp.	40,619	18,194	1,335	1,062	7,996	3,186	5,413	9,807	1,538	1,375	5,441	8,542	6,107	5,044	11,191	8,596	9,503	4,209
Fundulus spp.	1,695	1,199	815	659	952	613	915	1,081	1,463	906	111	432	3,142	831	859	1,224	513	936
A. quadracus	464	603	258	266	49	94	89	1,827	167	106	297	98	152	302	123	1,078	55	37
B. tyrannus	0	0	17	0	4	0	7	1	0	8	6	6	3		521	2,652	2	49
C. variegatus	48	673	39	30	10	352	146	50	29	28	2	2	21	3	30	1,170	76	4
A. americanus	6	520	16	51	10	318	82	30	21	0	7	1	4	0	47	156	27	59
P. saltatrix	1	0	1	6	0	2	135	4	19	35	12	12	5	6	825	14	0	18
P. pungitius	5	1	28	2	5	2	10	321	8	11	8	4	30	24	3	6	0	1
S. fuscus	9	3	9	108	6	8	21	12	35	30	33	19	74	11	17	40	11	9
G. aculeatus	9	154	27	5	3	2	5	53	6	6	19	15	38	8	0	3	0	14
P. americanus	4	6	4	1	6	5	2	3	17	40	18	17	16	48	9	10	2	1
M. cephalus	0	4	3	23	41	1	4	4	1	0	38	4	46	0	0	1	1	1
G. wheatlandi	0	0	0	0	0	8	6	6	19	12	9	22	9	8	0	1	0	5
A. pseudoharengus	0	0	0	0	0	0	0	1	93	0	0	4	0	6	0	0	0	0
Gadidae	2	0	9	2	20	16	11	8	11	11	8	0	2	2	0	0	1	1
M. curema	0	0	0	0	0	0	0	1	9	0	0	0	43	3	22	1	0	0
L. parva	1	2	0	0	0	0	0	2	0	1	0	16	14	2	1	32	7	0
C. harengus	0	0	0	0	0	0	2	0	0	0	30	0	6	1	0	0	0	1
A. rostrata	10	5	12	3	2	0	1	1	0	0	3	0	0	0	0	0	0	0
T. falcatus	0	0	1	0	3	0	0	0	0	0	0	0	22	7	0	0	0	0
M. aenaeus	3	2	1	2	0	0	3	1	3	3	3	2	4	0	0	0	3	2
O. mordax	0	0	0	0	0	0	0	0	0	2	0	0	18	0	0	0	0	0
Anchoa spp.	0	0	0	0	2	0	7	2	1	0	0	0	0	0	4	0	0	1
T. onitis	0	0	0	0	0	0	4	0	0	0	0	0	2	0	7	1	1	0
A. aestivalis	2	6	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	1
Gasterosteus spp.	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0	1	0
C. hippos	0	0	1	0	0	1	0	0	0	1	0	0	4	1	3	0	0	1
S. maculatus	0	0	0	1	0	0	1	0	0	3	3	0	1	0	1	0	0	0
T. adspersus	0	0	2	0	0	0	3	0	1	0	0	0	0	0	1	0	0	0
Men. saxatilis	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. aquosus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0	0
S. marina	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1	0
A. sapidissima	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
P. gunnellus	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
P. triacanthus	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0
S. vomer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0
C. regalis	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Clupeidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Prionotus spp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Urophycis spp.	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Total	42,881	21,372	2,579	2,221	9,109	4,609	6,868	13,216	3,444	2,582	6,061	9,196	9,763	6,311	13,667	14,986	10,205	5,351

[a] Fish identified to the lowest practical taxon.

APPENDIX V. Total number of samples collected and number of fish caught by seine at each station (June 1976 - May 1994).

Year	JC	GN	WP	TOTAL
Number of samples	738	779	786	2,303
Taxon [a]				
Menidia spp.	101,941	25,458	21,953	149,352
Fundulus spp.	13,713	2,577	2,076	18,366
A. quadracus	6,028	19	22	6,069
B. tyrannus	762	13	2,525	3,300
C. variegatus	1,831	849	34	2,714
A. americanus	5	218	1,133	1,356
P. saltatrix	955	53	87	1,095
P. pungitius	357	103	10	470
S. fuscus	96	67	292	455
G. aculeatus	277	30	60	367
P. americanus	43	12	155	210
M. cephalus	99	43	29	171
G. wheatlandi	64	27	40	131
A. pseudoharengus	8	96	0	104
Gadidae	66	32	6	104
M. curema	66	14	2	82
L. parva	66	7	5	78
Anchoa spp.	16	2	24	42
C. harengus	39	0	0	39
A. rostrata	31	2	4	37
T. falcatus	30	3	0	33
M. aenaeus	8	13	9	30
O. mordax	18	0	2	20
T. onitis	12	2	1	15
A. aestivalis	3	6	3	12
C. hippos	10	0	2	12
Gasterosteus spp.	0	1	11	12
S. maculatus	0	2	8	10
T. adspersus	6	1	0	7
S. aquosus	0	0	3	3
A. sapidissima	0	0	2	2
Men. saxatilis	1	0	1	2
P. gunnellus	0	0	2	2
P. triacanthus	0	1	1	2
S. marina	2	0	0	2
C. regalis	1	0	0	1
Clupeidae	0	1	0	1
Prionotus spp.	0	1	0	1
S. setapinnis	0	1	0	1
S. vomer	1	0	0	1
Urophycis spp.	0	1	0	1
Total	126,555	29,655	28,502	184,712

[a] Fish identified to the lowest practical taxon.

Lobster Studies

Lobster Studies

Introduction

Benthic habitats in the vicinity of Millstone Nuclear Power Station (MNPS) support an active local fishery for the American lobster (*Homarus americanus*). As one of the most valuable species in the Connecticut fishing industry (Blake and Smith 1984), the Long Island Sound (LIS) lobster population is subject to intense fishing pressure. Annual Connecticut landings of 0.8 to 2.7 million pounds from 1978 to 1994 yielded between $2.4 and 8.4 million to lobstermen employed in the fishery (Smith et al. 1989; Connecticut Department of Environmental Protection CT DEP, Marine Fishery Statistics). Nearly 30% of the total Connecticut landings during 1994 were made in New London county, which includes the Millstone Point area. Lobsters are highly exploited throughout their range and further increases in fishing effort could result in overfishing, and thereby impact recruitment and lead to a decline in abundance of coastal populations (Anthony and Caddy 1980). Subsequently, because fishing effort has steadily increased over the last decade, lobster fisheries have become more highly regulated, primarily through the implementation of increased minimum size limits. New federal regulations have been proposed by the Atlantic States Marine Fisheries Commission (ASMFC) in an amendment to the American Lobster-Fishery Management Plan (FMP), which incorporates a mandatory $3^1/_4$ in (82.6 mm) minimum carapace length, prohibitions on possession of berried females and landing of lobster meat (parts of tails or claws), and mandatory installation of escape vents. The amended FMP will be implemented March 1995 in lobster producing States from Maine to Delaware. The federal FMP cautioned that further amendments may be necessary to effectively manage the lobster stocks and preserve the resource. In Connecticut waters, the $3^1/_4$ in (82.6 mm) minimum legal size has been enforced since 1990 and installation of escape vents in traps was mandatory beginning in 1984. The prohibitions on possession of berried females and lobster parts have long been enforced in the LIS lobster fishery.

High exploitation rates may also increase susceptibility of the local lobster population to impacts associated with coastal power generation. The potential impacts of power plant operations on the local populations of lobsters include entrainment of larval lobsters through the cooling water systems, impingement of juveniles and adults on the intake traveling screens, and effects of the heated discharge. Entrainment and impingement contribute additional mortality to the local lobster population and may thereby alter recruitment patterns. Therefore, lobster population studies have been and continue to be part of ecological monitoring programs associated with coastal power generating stations (Dean and Ewart 1978; LILCO 198[illegible]; BECO 1994; NAESCO 1994).

Studies of the lobster population in the vicinity of Millstone Point have been ongoing since 1978 to assess potential impacts associated with the construction and operation of MNPS. The objectives of this study are to evaluate year-to-year, seasonal, and among station changes in catch-per-unit-effort, as well as population characteristics such as size frequency, growth rate, sex ratios, female size at sexual maturity, characteristics of egg-bearing females, and lobster movements. Additionally, since 1984, studies have been conducted during the hatching season to estimate the number of lobster larvae entrained through cooling water systems. Impacts associated with plant operations on the local lobster population were assessed by comparing results of the 1994 study to other 3-unit operational study years (1986-1993) and to data collected during 2-unit operations (1978-1985). Results from the 2-unit period were also compared to combined 3-unit operational data (1986-1994) to assess impacts associated with the addition of Millstone Unit 3. When appropriate, results of our lobster study were compared to other studies conducted in LIS and elsewhere.

Materials and Methods

Full descriptions of methods used to conduct lobster population studies are in NUSCO (1982, 1987a). To summarize, four pot-trawls, each consisting of five double-entry wire pots (76 x 51 x 30 cm; 2.5 cm^2 mesh) equally spaced along a 50-75 m line buoyed at both ends, were used to collect

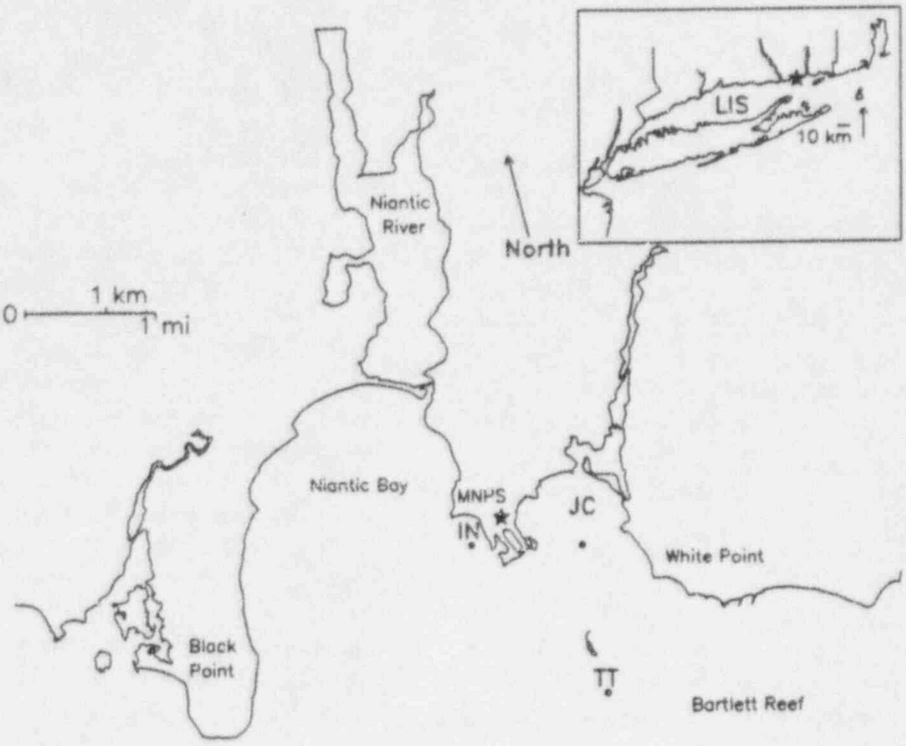


Fig. 1. Location of the Millstone Nuclear Power Station (MNPS), and the three lobster sampling stations (●), JC=Jordan Cove, IN=Intake, TT=Twotree.

lobsters from May through October. Pot-trawls were set near rocky outcrops at three stations (Fig. 1). Pots in Jordan Cove (average depth 6 m) were set around High Rock 500 m east of the Millstone discharge. The Intake station (average depth 5 m) was 600 m west of the discharge near the power plant intake structures, and the Twotree station (average depth 12 m) was located south of Millstone Point, about 1600 m offshore near Twotree Island. Beginning in 1984, pots were individually numbered to determine the variability in catch among pots, and to provide more accurate values for catch-per-pot than an average catch-per-pot based on a total of 20 pots per sampling location. Pots were hauled on Monday, Wednesday, and Friday of each week, weather permitting; on holiday weeks pots were checked on the first and last workdays. On each sampling trip, surface and bottom water temperatures and salinities were recorded at each station. Lobsters larger than 55 mm carapace length were banded to restrain chelipeds, brought to the laboratory, and kept separated by station in labeled tanks supplied with a continuous flow of seawater. Pots were rebaited with flounder carcasses after lobsters were removed and reset in the same area. On Fridays, lobsters caught that week were examined and the following data recorded: sex, presence of eggs (berried), carapace length (CL), crusher claw position, missing claws, and molt stage (Aiken 1973). Lobsters were tagged with a serially numbered international orange sphyrion tag (Scarratt and Elson 1965; Scarratt 1970), and released at the site of capture. Recaptured tagged lobsters, severely injured or newly molted (soft) lobsters, and those smaller than 55 mm CL were released untagged after recording the above data.

Beginning in 1981, size at which females became sexually mature was estimated by measuring (to the nearest millimeter) the maximum outside width of the second abdominal segment of all females. Female size at sexual maturity was estimated by the carapace length corresponding to the inflection point of the sigmoid or "S" curve obtained by plotting the ratio of abdominal width to carapace length against carapace length (Skud and Perkins 1969; Krouse 1973).

Lobster larvae have been sampled from 1984 to

1994 during the period of their occurrence (May through July) at the discharges of Units 1, 2 and 3. Samples were collected with a 1.0 x 6.0 m conical plankton net of 1.0 mm mesh. Cooling water volume sampled was estimated from the average readings of four General Oceanic flowmeters located in the mouth of the net; about 4000 m^3 of cooling water were filtered in each sample by deploying the net for 45-60 minutes. Day and night samples were collected four days a week from 1984 to 1993, and three days a week in 1994. Each sample was placed in a 1.0 mm mesh sieve and kept in tanks supplied with a continuous flow of seawater. Shortly after collection, samples were sorted in a white enamel pan; larvae were examined for movement and classified as live or dead. Lobster larvae were also classified by stage according to the criteria established by Herrick (1911). Larvae abundance in entrainment samples was standardized as the number of larvae per unit-volume. The seasonal (May through July) mean density was calculated as the mean of the assumed "delta" distribution, referred to as Δ-mean (Pennington 1983; NUSCO 1988a). To estimate total number of larvae entrained, the Δ-mean density was scaled by the total volume of water pumped through all units during the sampling period.

Impingement studies were conducted at Unit 1 and 2 intakes from 1975 through 1987; results summarized in NUSCO (1987a) included estimates of total number of lobsters impinged, as well as mean size, sex ratio, proportion of culls, and survival probabilities for impinged lobsters. Impacts on the local lobster population associated with impingement of lobsters at Units 1 and 3 were mitigated by installing fish return systems in the intakes, which return impinged organisms to LIS (NUSCO 1986, 1987b). Subsequently, NUSCO and the CT DEP agreed to discontinue impingement monitoring (NUSCO 1988b).

Catch-per-unit-effort (CPUE, i.e., the number of lobsters caught per pothaul) was used to describe the annual abundance of lobsters in the MNPS area. Because these CPUE data are ratios, which are not additive and have an asymmetric distribution about the arithmetic mean, the geometric mean was the statistic chosen to analyze trends in CPUE. The geometric mean is better suited for constructing asymmetric confidence intervals for skewed data (Snedecor and Cochran 1967; McConnaughey and Conquest 1993). Annual geometric mean CPUEs were calculated for all lobster sizes. The annual abundance (CPUE) of legal-size lobsters was estimated by using the Δ-mean. The Δ-mean was a more appropriate statistic for describing the CPUE of legal-size lobster, since a large number of zero observations were present in the data (i.e., many pots contained no legal-size lobsters). Both geometric means of all lobsters and Δ-means of legal-size lobsters were used to compare annual variation in CPUE. In the following "Results and Discussion" section, the geometric mean abundance of all lobsters is called "mean total CPUE" while the Δ-mean abundance of legal-size lobsters is referred to as "mean legal CPUE". The distribution-free, Mann-Kendall test (Hollander and Wolfe 1973) was used to determine presence of significant trends in the time series of annual CPUE data, and of several other selected population characteristics. Trend slopes, when significant, were calculated using Sen's estimator of the slope (Sen 1968; Gilbert 1987).

The influence of water temperature on lobster molting was examined by estimating the time when lobster molts peaked each year and correlating annual molt peaks with water temperature. Time of molting peaks was estimated by the inflection points of the Gompertz growth function fitted to data reflecting the cumulative percentage of molting lobsters at weekly intervals during the molting season. This growth function has the form:

$$C_t = 100e^{-e^{-k(t-p)}}$$

where C_t = cumulative percentage of molting lobsters at time t,
t = time in weeks,
p = inflection point scaled in weeks from May 1st, and
k = shape parameter.

The first derivative of the Gompertz function with respect to time yields a "molt frequency distribution" function which describes the distribution of annual molts. The times of annual molting peaks were then correlated with mean bottom water temperature during May to investigate a possible relationship between water temperature and molting.

Results and Discussion

Temperature and Salinity

Monthly mean surface and bottom water temperatures during 2-unit, 3-unit and 1994 studies are presented in Table 1. The mean surface and bottom water temperatures during May 1994 were among the coldest observed in our studies. Mean bottom water temperatures in May 1994 ranged between 8.2 (Twotree) and 8.7°C (Intake), lower than the means recorded during 2-unit (8.9-9.3°C) and 3-unit studies (9.3-10.0°C; Table 1). With the exception of Jordan Cove, surface water temperatures during May 1994 were also among the lowest observed since the study began, 0.5 to 1.5 °C cooler than operational period means. Conversely, water temperatures in July 1994 were the warmest observed in our studies; surface and bottom water temperatures at Intake and Twotree were about 1°C warmer in 1994 than July means for 2-unit and 3-unit studies (Table 1).

Surface water temperatures during the period of 3-unit operation (1986-1994) ranged between 9.7 and 22.1°C, and were warmer than the 2-unit temperatures of 9.4-21.2°C. The highest temperatures were recorded at Jordan Cove during both 2-unit (10.2-21.2°C) and 3-unit studies (12.1-22.1°C). In general, water temperatures at the nearshore sites (Jordan Cove and Intake, 4-6 m depth) were similar and consistently warmer than at the offshore Twotree site (12 m depth). Our long-term field data for water temperatures at Jordan Cove are consistent with results of hydrothermal studies, which indicated that a 2.2°C isotherm resulting from 3-unit operation could extend into the Jordan Cove area. In addition, a 0.8°C isotherm extends 600 m from the discharge to a depth of 3 to 5 m (NUSCO 1988c), and may reach some of the pot-trawls deployed at the Jordan Cove and Intake sites.

Salinities were similar during 2-unit and 3-unit operation; mean surface and bottom water salinities ranged between 29.4 and 31.6$^{\circ}/_{oo}$. Due to the spring freshwater runoff, salinities were generally lower at all stations in May and June.

TABLE 1. Mean monthly surface and bottom water temperatures measured at each station during 2-unit (1979-85), 3-unit (1986-94) and 1994 studies.

	Surface			Bottom		
	2-Unit	3-Unit	1994	2-Unit	3-Unit	1994
			Jordan Cove			
MAY	10.2	12.1	10.5	9.2	9.7	8.5
JUN	15.1	17.0	15.4	13.9	14.4	13.8
JUL	19.5	20.8	20.7	18.0	18.2	18.9
AUG	21.2	22.1	21.8	19.9	20.0	19.9
SEP	20.3	21.3	20.9	19.2	19.1	18.4
OCT	16.8	17.3	17.9	16.0	16.0	15.8
			Intake			
MAY	10.1	10.9	9.6	9.3	10.0	8.7
JUN	14.9	15.7	15.1	14.1	14.6	14.2
JUL	19.2	19.5	20.4	18.3	18.6	19.7
AUG	20.7	21.1	21.1	20.1	20.2	20.1
SEP	19.8	20.0	19.2	19.4	19.4	18.7
OCT	16.1	16.5	16.4	15.9	16.1	15.9
			Twotree			
MAY	9.4	9.7	8.2	8.9	9.3	8.2
JUN	14.2	14.6	14.1	13.7	14.0	13.6
JUL	18.3	18.5	19.4	17.7	17.8	18.7
AUG	20.0	20.1	19.9	19.6	19.7	19.6
SEP	19.3	19.2	18.5	19.1	18.9	18.1
OCT	16.0	15.8	15.7	15.9	15.8	15.7

Abundance and Catch-per-Unit-Effort

Annual catch statistics of lobsters caught in wire traps are presented in Table 2. The total number of lobsters caught during 1994 was 9,849, which was within the range of previous 3-unit studies (7,106-11,438), but larger than the range of values reported in 2-unit studies (6,376-9,109) when 20 traps were used at each station (Table 2). Lower catches from 1978 to 1981 of 2-unit studies (1,824-3,259) occurred when only 10 wire pots were used at each station. The geometric mean total CPUE for 1994 of 2.199 lobsters/pot was within the range of previous 3-unit years (1.531-2.457), but higher than the range of 0.904 to 2.006 reported in 2-unit years. Total lobster catches were generally higher during 3-unit studies (1986-94 mean CPUE=1.849) than during 2-unit studies (1978-85 mean CPUE=1.364). A significant increasing trend (slope=0.051, p=0.01) was observed for the overall time series (1978-1994) of total CPUE.

TABLE 2. Catch statistics of lobsters caught in wire[a] pots from 1978 to 1994.

	Total number lobster caught	Number pots hauled	Geometric mean total CPUE	95% C.I.	Delta mean legal CPUE[b]			95% C.I. ≥ 81.0 mm
					≥ 81.0	≥ 81.8	≥ 82.6	
1978	1824	1026	1.600	1.454 - 1.761	0.173	0.118	0.096	0.144 - 0.202
1979	3259	2051	1.404	1.302 - 1.513	0.128	0.101	0.079	0.107 - 0.148
1980	2856	2116	1.103	0.997 - 1.221	0.109	0.076	0.063	0.092 - 0.126
1981	2236	2187	0.904	0.839 - 0.974	0.098	0.079	0.069	0.083 - 0.113
1982	9109	4340	2.006	1.925 - 2.089	0.165	0.126	0.106	0.144 - 0.186
1983	6376	4285	1.331	1.250 - 1.418	0.148	0.109	0.093	0.128 - 0.168
1984	7587	4550	1.607	1.540 - 1.677	0.159	0.120	0.104	0.140 - 0.179
1985	7014	4467	1.352	1.252 - 1.460	0.105	0.080	0.068	0.090 - 0.120
1986	7211	4243	1.585	1.501 - 1.673	0.086	0.060	0.049	0.074 - 0.097
1987	7280	4233	1.633	1.562 - 1.707	0.079	0.054	0.046	0.070 - 0.089
1988	8871	4367	1.929	1.846 - 2.015	0.079	0.052	0.047	0.068 - 0.091
1989	7950	4314	1.729	1.645 - 1.817	0.112	0.065	0.053	0.097 - 0.126
1990	7106	4350	1.531	1.455 - 1.610	0.161	0.102	0.076	0.143 - 0.179
1991	7597	4404	1.542	1.437 - 1.654	0.183	0.117	0.091	0.159 - 0.206
1992	11438	4427	2.457	2.352 - 2.565	0.208	0.114	0.085	0.186 - 0.229
1993	10195	4194	2.301	2.198 - 2.408	0.197	0.111	0.080	0.175 - 0.220
1994	9849	4256	2.199	2.104 - 2.298	0.200	0.108	0.071	0.178 - 0.223
2-Unit 78-85	40261	25022	1.364	1.337 - 1.403	0.134	0.100	0.085	0.127 - 0.141
3-Unit 86-94	77497	38788	1.849	1.815 - 1.884	0.144	0.087	0.066	0.138 - 0.151

[a] 10 wire pots used at each station from August through October 1978, and from May through October 1979-81; 20 wire pots used at each station from May through October 1982-94.

[b] The minimum legal-size from 1978 to 1988 was 81.0 mm ($3\,{}^{3}/_{16}$ in), minimum legal-size was increased in 1989 to 81.8 mm ($3\,{}^{7}/_{32}$ in), and in 1990 to 82.6 mm ($3\,{}^{1}/_{4}$ in).

The 1994 Δ-mean legal CPUE (0.071) was below the range of values reported in other 3-unit study years when the legal size was 82.6 mm (0.076-0.091) and 81.0 mm (0.079-0.086), but higher than the 1989 mean (0.065) when legal size was 81.8 mm. Legal CPUE during 1994 and in other 3-unit years were lower when compared to legal CPUE reported in 2-unit studies (1978-85 Δ-mean range=0.098-0.173). Legal catches in our traps have exhibited a significant declining trend since 1978 (slope=-0.005, p=0.001) and was most likely due to increases in minimum legal size, which began in 1988, and to an escalation in fishing effort, which has nearly doubled since 1979 (NMFS 1993). The magnitude of annual legal catches is highly dependent on the number of lobsters just below (one molt from) legal size. Each year, more than 90% of the legal-size lobsters caught in our area had recently molted from the sublegal size class. Over the past three years, total CPUEs were the highest observed in our 17 years of lobster studies; the majority of the total catch is composed of sublegal-sized lobsters. The fact that catches of sublegal-sized lobsters has increased in recent years is important in maintaining a steady supply of recruits for the legal size class. The higher total CPUEs during the past three years is likely related to the increase in minimum legal size, and, as intended by the regulation, could lead to higher legal catches in the future as sublegal-sized lobsters molt into the legal size class.

During 1994, total and legal CPUE were highest at Twotree (2.502 and 0.082, respectively); since 1978, this site has yielded the highest catches and largest lobsters (Fig. 2). Lobster catches during 1994 were intermediate at Jordan Cove (total CPUE=2.267, legal CPUE=0.075) and lowest at Intake (total CPUE=1.874, legal CPUE=0.058). Total CPUE at each site was generally higher during 3-unit studies than during 2-unit studies (Fig. 2). Since 1978, total CPUE has significantly increased at Jordan Cove (slope=0.048 p=0.039) and Twotree (slope=0.095, p=0.001); no trends in total CPUE were evident at Intake. In comparison

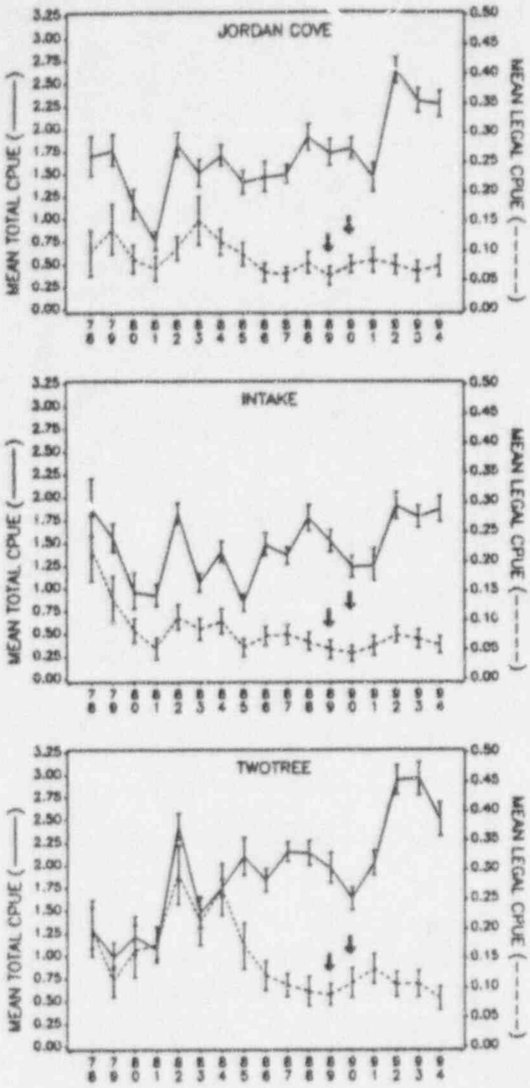


Fig. 2. Mean total CPUE (geometric mean ± 95% C.I.) and mean legal CPUE (Δ-mean ± 95% C.I.) of lobsters caught at each station from 1978 to 1994 (arrows indicate increases in minimum legal size from 81.0 mm to 81.8 mm in 1989 and 82.6 mm in 1990).

to the increasing trend in total CPUE, legal CPUE at each site has significantly declined since 1978 (Jordan Cove slope=-0.002, p=0.021; Intake slope=-0.004, p=0.011; Twotree slope=0.007, p=0.007). These declines were more likely related to increased fishing effort and increases in the minimum legal size than to power plant operations.

TABLE 3. Monthly mean total CPUE and Δ-mean legal CPUE during 2-unit (1979-85) and 3-unit studies (1986-94) and during 1994.

	Total CPUE			Legal CPUE[a]		
	2-Unit	3-Unit	1994	2-Unit	3-Unit	1994
MAY	1.468	2.138	2.710	0.112	0.065	0.032
JUN	1.656	2.303	2.447	0.147	0.112	0.055
JUL	1.838	2.077	2.247	0.204	0.114	0.151
AUG	1.428	1.883	2.451	0.137	0.083	0.100
SEP	1.102	1.454	1.832	0.113	0.052	0.041
OCT	0.943	1.367	1.608	0.093	0.045	0.054

[a] The minimum legal-size during 2-unit studies was 81.0 mm, minimum legal size was increased during 3-unit studies to 81.8 mm in 1989 and to 82.6 mm in 1990.

Seasonal lobster abundance patterns varied over the study period; during 2- and 3-unit studies, total CPUEs were highest in July (1.838) and June (2.303), respectively (Table 3). In 1994, total CPUE was highest in May (2.710). Total CPUE was consistently lowest in October during both operational periods and 1994 (Table 3). Legal CPUE typically peaked in July following the spring molt (2-unit=0.204, 3-unit=0.114, 1994=0.151; Table 3). Catchability of lobsters is directly influenced by water temperature; when water temperature rises above 10°C, lobster activity (e.g., feeding, movement, and molting) increases (McLeese and Wilder 1958; Dow 1966, 1969, 1976; Flowers and Saila 1972; NUSCO 1994).

Lobster catches were shown to be influenced by the presence of other species in traps during previous studies (NUSCO 1987a, 1994). Incidental catches of other species affect lobster catches by occupying space in the trap, consuming the bait, and blocking trap entrances. During 1994, lobster CPUE was negatively influenced by catches of spider crabs at Intake and Twotree (Table 4). No species affected lobster catch at Jordan Cove during 1994. Spider crab catches continued to be high during 1994 (n=13,803) and have influenced lobster catches in all but one year since 1984. Throughout the North Atlantic Ocean, researchers have demonstrated competition, niche segregation and interactions between lobsters and crabs in field and laboratory studies (Richards et al. 1983; Richards and Cobb 1987; Hudon and Lamarche 1989; Miller 1989). In addition to spider crabs, lobster catches at Twotree during 1994 were affected by the catches of tautog (*Tautoga onitis*), a known predator of lobsters (Bigelow and Shroeder 1953; Auster 1985; Cobb et al. 1986).

TABLE 4. Total number of lobsters and incidental catch of other species caught in traps.

	Range 1984-1993	1994
Lobster	7014-11438	9849
Rock, Jonah crab	79-2033*	294
Spider crab	1344-13086*	13803*
Hermit crab	286-721*	217
Blue crab	21-148	69
Winter flounder	8-45*	13
Summer flounder	4-60*	12
Skates	14-54	27
Oyster toadfish	8-76	11
Scup	27-288	45
Cunner	67-207	48
Tautog	39-250	89*
Sea raven	0-20	0
Whelks	27-178*	87

(*) Covariance analysis identified these catches as significant factors affecting lobster CPUE (p<0.05).

On several occasions we have observed large tautog in traps along with signs of predation activity (damaged and dead lobsters, or only parts of lobsters).

Population Characteristics

Size Frequency

Carapace length statistics for lobsters caught in wire traps from 1978 to 1994 are presented in Table 5. Mean carapace length (CL) of 70.3 mm during 1994 was within the range of previous 3-unit CLs (69.5-70.8 mm) but below the range of CL means reported in 2-unit studies (70.7-71.8 mm; Table 5). The overall mean CL of lobsters collected during 3-unit studies was smaller (70.1 mm) than during 2-unit studies (71.3 mm). Legal-size lobsters (≥ 82.6 mm) comprised 3.1% of the total catch during 1994, which was the smallest

TABLE 5. Summary of lobster carapace length statistics in wire pot catches from May through October, 1978-1994.

	N[a]	Carapace length (mm)		Percentage of legal sizes[b]		
		Range	Mean±95% CI	≥ 81.0	≥ 81.8	≥ 82.6
1978	1508	53-111	71.4 ± 0.33	7.5	5.9	4.8
1979	2846	44-100	71.2 ± 0.26	7.6	6.6	5.1
1980	2531	40-96	70.7 ± 0.27	6.4	5.0	4.1
1981	1983	43-96	71.0 ± 0.33	8.8	7.6	6.6
1982	7835	45-103	70.8 ± 0.15	6.7	5.7	4.7
1983	5432	40-121	71.7 ± 0.19	9.1	7.4	6.3
1984	6156	45-107	71.8 ± 0.18	8.7	7.3	6.4
1985	5723	38-101	71.3 ± 0.17	5.9	5.1	4.3
1986	5961	36-107	70.1 ± 0.17	4.4	3.6	3.0
1987	5924	36-99	70.2 ± 0.17	3.9	3.2	2.7
1988	7145	21-97	69.5 ± 0.16	3.2	2.6	2.3
1989	6715	34-107	69.9 ± 0.17	4.5	3.5	2.9
1990	6040	36-102	70.2 ± 0.20	7.9	5.9	4.5
1991	6449	31-101	70.2 ± 0.20	8.5	6.5	5.0
1992	9594	20-103	70.1 ± 0.15	6.4	4.3	3.3
1993	8487	30-102	70.8 ± 0.15	6.7	4.6	3.3
1994	7841	34-100	70.3 ± 0.17	7.1	4.7	3.1
2-Unit 78-85	34014	38-121	71.3 ± 0.07	7.5	6.3	5.3
3-Unit 86-94	64156	20-107	70.1 ± 0.06	5.9	4.3	3.3

[a] Recaptures not included.

[b] The minimum legal size from 1978 to 1988 was 81.0 mm (3 $^{3}/_{16}$ in), minimum legal size was increased in 1989 to 81.8 mm (3 $^{7}/_{32}$ in), and in 1990, to 82.6 mm (3 $^{1}/_{4}$ in).

TABLE 6. Summary of lobster carapace length statistics in wire pot catches at each station from May through October, during 2-Unit (1978-1985), 3-Unit (1986-1993) and 1994 studies.

	Mean carapace length (mm)[a]	Percentage of legals (≥82.6 mm)
JORDAN COVE		
2-Unit range	69.8 - 71.1	2.5 - 5.9
3-Unit range	69.0 - 70.2	2.1 - 4.8
1994 mean	69.9	3.4
INTAKE		
2-Unit range	69.2 - 71.8	2.9 - 5.7
3-Unit range	68.9 - 70.2	1.8 - 3.6
1994 mean	69.5	2.8
TWOTREE		
2-Unit range	71.3 - 73.7	4.4 - 10.4
3-Unit range	70.0 - 71.9	2.6 - 6.2
1994 mean	71.1	3.1

[a] Recaptures not included.

percentage reported in our studies (2-unit range 5.9-9.1%; previous 3-unit range 3.2-5.0%). Since 1978, the percentages of legal lobsters in our catch have significantly declined (slope = -0.310, $p < 0.01$).

When the three stations were compared, the largest lobsters were caught at Twotree during 1994 (mean CL = 71.1 mm) followed by Jordan Cove (69.9 mm) and Intake (69.5 mm; Table 6). The 1994 mean sizes at each site were within the ranges of previous 3-unit studies and at Jordan Cove and Intake, they were within the ranges of 2-unit studies (Table 6). However, the mean CL at Twotree during 1994 was smaller than the range of annual means reported in 2-unit studies (71.3-73.7 mm). Twotree catches have typically yielded the highest percentage of legal-sized lobsters; however, during 1994, the percentage of legals was highest at Jordan Cove (3.4%) followed by Twotree (3.1%) and Intake (2.8%; Table 6). The 1994 percentages at each site were within the range of 3-unit percentages. The percentage of legal lobsters at Intake and Twotree during 1994 were below the percentages reported during 2-unit studies; percent legal size at Jordan Cove during 1994 was within the range of 2-unit studies. Trends in percentages of legal-sized lobsters at each station have exhibited significant declines since 1978. The declines in percent legal size were due, in part, to the increases in the minimum legal size in 1988 and 1990, but was probably more related to increased fishing effort, which has more than doubled since 1978 (Blake 1991; NMFS 1993).

Sex Ratios

The sex ratio of lobsters collected during 1994 was 0.79 females per male, compared to a range of 0.71-0.88 in prior years of 3-unit operation and 0.79-0.97 during 2-unit operation (Table 7). More females (1.24 per male) were caught at Twotree in 1994 than at the nearshore Jordan Cove and Intake sites (0.54 and 0.67 females per male, respectively). Sex ratios at Twotree and Intake during 1994 were within the range of previous 2- and 3-unit studies; at Jordan Cove, the 1994 sex ratio of 0.54 was within the range of 3-unit studies but below the range of 2-unit studies (0.60-0.79). The overall ratio of females to males was higher during 2-unit studies (0.86 females per male) than during 3-unit studies (0.80 females per male). At Jordan Cove and Intake, sex ratios during 2-unit studies were 0.67 and 0.72 females per male, respectively, higher than the ratios during 3-unit

TABLE 7. Female to male sex ratios[a] of lobsters caught in wire pots from May through October, 1978-1994.

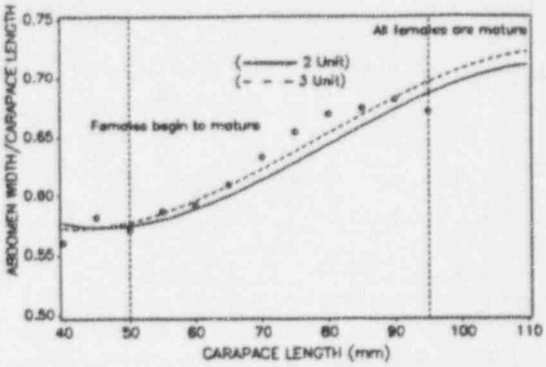
	Jordan Cove	Intake	Twotree	All Stations
1978	0.79	0.97	1.02	0.92
1979	0.68	0.83	1.15	0.82
1980	0.66	0.90	1.15	0.88
1981	0.70	0.71	1.19	0.86
1982	0.62	0.66	1.09	0.79
1983	0.72	0.67	1.25	0.87
1984	0.60	0.71	1.22	0.82
1985	0.70	0.67	1.38	0.97
1986	0.65	0.73	1.26	0.87
1987	0.71	0.63	1.24	0.88
1988	0.68	0.72	1.15	0.85
1989	0.64	0.65	1.08	0.79
1990	0.60	0.65	0.90	0.71
1991	0.51	0.57	1.13	0.74
1992	0.43	0.47	1.45	0.73
1993	0.47	0.59	1.59	0.84
1994	0.54	0.67	1.24	0.79
2-Unit 78-85	0.67	0.72	1.21	0.86
3-Unit 86-94	0.56	0.62	1.24	0.80

[a] Recaptures not included.

studies (0.56 and 0.62, respectively). At Twotree, the female-to-male ratios were similar during 2-unit (1.21) and 3-unit studies (1.24). Since 1975, Twotree has consistently exhibited proportionally more females than have the inshore stations, and except for 1990, the sex ratio at Twotree has consistently reflected more females than males (Keser et al. 1983). Female-to-male ratios of lobsters caught by commercial lobstermen in LIS ranged between 1.06 and 1.81 (Smith 1977) and more recently, Blake (1994) reported sublegal lobster sex ratios of 2.09 and 2.71 females per male in the western and eastern LIS commercial fishery, respectively. Several factors have been suggested to cause a predominance of females in the commercial fishery: differences in female lobster behavior related to molting and reproduction, fishery regulations designed to protect egg bearing females, and the fact that mature females molt less frequently than males (Ennis 1980). The overall sex ratio of lobsters in the MNPS area is close to the 1:1 sex ratio reported by other researchers for predominantly sublegal (< 81.0 mm CL) populations of lobsters (Herrick 1911; Templeman 1936; Ennis 1971, 1974; Stewart 1972; Krouse 1973; Thomas 1973; Cooper et al. 1975; Briggs and Mushacke 1980).

Reproduction

The sexual maturity of female lobsters has been investigated in our studies since 1981. The most obvious indication that females are mature is the presence of external eggs (berried). Another method, first described by Templeman (1935), is based on abdominal width measurements of females, which markedly increases during maturation. Abdominal width to carapace length ratio plotted against CL provides an index of female size at sexual maturity (Skud and Perkins 1969; Krouse 1973). Mean ratios of abdominal width to carapace length were calculated for each 5 mm CL increment and plotted against the carapace length of lobsters collected during 2-unit (1981-85) and 3-unit (1986-94) operations and for 1994 alone (Fig. 3). During 1994, females began to mature at about 55 mm CL, and all females were mature above 90 mm CL. The close correspondence between the 2- and 3-unit curves in Figure 3 indicates that female size at sexual maturity was similar during both operational periods. Size distribution of berried females provides further evidence of the unusually small size at which females become sexually mature in our study area. The smallest berried female collected in our area was 60 mm CL and 50% were below 76 and 78 mm CL during 3- and 2-unit studies, respectively (Fig. 4). Our results agree with other studies conducted in LIS; Briggs and Mushacke (1979), using the same morphometric technique, found that females in western LIS begin to mature at 60 mm CL and most are mature at about 80 mm CL. Blake (1994) estimated female size at sexual maturity using cement gland staging (Aiken and Waddy 1982) and found that half of the females in LIS reach functional maturity (will extrude eggs) at about 73 mm CL. In contrast to



2-unit: $y=1.28-(3.13*10^{-2})x+(4.40*10^{-4})x^2-(1.85*10^{-6})x^3$, $r^2=.30$
3-unit: $y=1.08-(2.59*10^{-2})x+(4.05*10^{-4})x^2-(1.84*10^{-6})x^3$, $r^2=.32$
1994: $y=1.65-(5.32*10^{-2})x+(8.36*10^{-4})x^2-(4.05*10^{-6})x^3$, $r^2=.35$

Fig. 3. Morphometric relationship between the abdominal width to carapace length ratio (y) and the carapace length (x) of female lobsters during 2-unit (——) and 3-unit studies (- - -) and during 1994 (o o o).

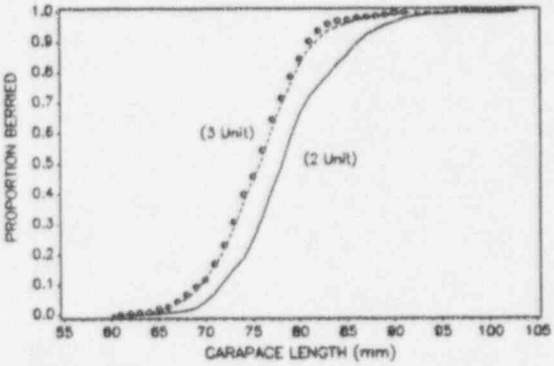


Fig. 4. Proportion of berried females in each 1 mm carapace length interval during 2-unit (——), 3-unit (- - -) and 1994 studies (o o o).

TABLE 8. Percentage of berried females caught at each station and annual carapace length statistics from 1978-94.

	Percentage of berried females				Carapace Length (mm)			Percentage of sublegal sizes [b]		
	All stations	Jordan Cove	Intake	Twotree	N[a]	Range	Mean ± 95% C.I.	<81.0	<81.8	<82.6
1978	3.4	1.4	2.6	5.3	58	74 - 88	80.1 ± 1.04	**73**	78	78
1979	3.1	1.9	2.7	7.2	70	64 - 93	80.5 ± 1.28	**59**	64	70
1980	3.3	3.5	1.8	5.6	71	66 - 93	79.1 ± 1.27	**70**	73	79
1981	4.2	1.6	2.7	7.1	82	69 - 97	81.2 ± 1.35	**55**	59	62
1982	3.1	0.8	0.9	6.1	108	64 - 99	80.0 ± 1.08	**60**	66	70
1983	4.7	2.1	3.2	8.5	123	66 - 103	80.5 ± 1.04	**63**	65	67
1984	6.2	3.6	3.5	10.6	173	62 - 95	79.1 ± 0.87	**69**	75	76
1985	6.2	3.5	4.5	8.5	171	63 - 94	77.0 ± 0.81	**82**	85	86
1986	4.8	3.0	2.3	8.0	135	65 - 94	78.0 ± 0.95	**77**	80	83
1987	5.7	3.2	1.9	9.6	158	62 - 90	76.5 ± 0.67	**92**	92	93
1988	3.8	2.4	1.9	6.4	124	63 - 90	76.9 ± 0.82	**89**	90	90
1989	5.4	2.8	3.3	8.2	161	65 - 98	77.3 ± 0.78	82	**85**	88
1990	6.6	2.7	4.0	11.2	165	65 - 102	78.1 ± 0.82	75	81	**87**
1991	8.2	3.2	1.5	13.5	226	62 - 96	78.0 ± 0.75	71	78	**82**
1992	12.1	3.4	1.7	19.3	491	60 - 93	75.3 ± 0.44	89	92	**94**
1993	12.2	3.1	2.7	19.4	476	62 - 90	75.6 ± 0.43	88	91	**93**
1994	10.8	6.1	4.7	16.9	372	61 - 91	75.9 ± 0.52	85	90	**93**
2-Unit 78-85	4.3	2.0	2.2	7.1	856	62 - 103	79.4 ± 0.39	68	72	74
3-Unit 86-94	8.1	3.3	2.7	13.3	2308	60 - 102	76.4 ± 0.21	84	88	89

[a] Recaptures not included.

[b] The minimum legal size from 1978 to 1988 was 81.0 mm (3 $^{3}/_{16}$ in), minimum legal size was increased in 1989 to 81.8 mm (3 $^{7}/_{32}$ in), and in 1990 to 82.6 mm (3 $^{1}/_{4}$ in).

the LIS population of females, Gulf of Maine females seldom become sexually mature at less than 81 mm CL, and only a small percentage are mature between 81 and 90 mm CL (Krouse 1973; Krouse et al. 1993). Earlier maturation of females in LIS is attributed to the warmer LIS water temperatures (Smith 1977; Aiken and Waddy 1980). Sexual maturity of males was not investigated in our study because other researchers documented that the size at which males become mature varies only slightly throughout the range of lobsters. In western LIS, males are mature (i.e., produce mature spermatozoa) at 40 to 44 mm CL, and over half are mature at 50 to 54 mm CL (Briggs and Mushacke 1979); in northern waters (Maine), males also begin to mature at relatively small sizes (50% mature at 44 mm CL; Krouse 1973).

The percentage of females that were berried during 1994 was 10.8%, which was within the range of percentages reported in other 3-unit studies (3.8-12.2%), but higher than the range reported in 2-unit studies (3.1-6.2%; Table 8). When the three stations were compared, the percentage of berried females was highest at Twotree (16.9%); this percentage was within the range of previous 3-unit years (6.4-19.4%) and higher than 2-unit years (5.3-10.6%; Table 8). Percentages at Jordan Cove (6.1%) and Intake (4.7%) during 1994 were the highest reported since the study began (0.8-3.6% and 0.9-4.5%, respectively). A consistent pattern of more berried females at Twotree than at the nearshore Jordan Cove and Intake sites has been noted since 1975 (Keser et al. 1983). The overall percentage of berried females during 3-unit studies (8.1%) was higher than during 2-unit studies (4.3%). The mean carapace length of 75.9 mm for berried females collected during 1994 was within the range of average sizes reported in previous 3-unit studies (75.3-78.1 mm), but below the range reported in 2-unit studies (77.0-81.2 mm; Table 8). Berried females were smaller (76.4 mm) during 3-unit studies than during 2-unit studies (79.4 mm), due to the larger proportion of sublegal-sized berried females collected since 1986. Only 7% of the berried females were above the minimum legal size of 82.6 mm during 1994 compared to a range between 18% and 45% prior to 1986 (Table 8). High rates of fishing remove most females shortly after they reach legal size or after berried females release eggs. The apparent stability of the LIS lobster population, despite current high exploitation rates, may be due to the fact that females become mature and bear eggs at sizes well below the legal size. The regulation to increase the minimum legal size appears to be effective, as the percentage of berried females in our catch has increased in recent years. The increase in minimum legal size should improve egg production and subsequent recruitment by allowing more females to spawn before reaching legal size. However, if fishing effort continues to escalate, higher yields (CPUE) due to increased egg production may not materialize.

Molting and Growth

Lobster growth is a function of size and weight increment per molt and molt frequency, with temperature as the most important factor regulating these processes (Aiken 1980). During 1994, the majority of molting lobsters were caught from late spring (end of May) to early summer (middle of June). In several of the previous study years, a second peak in the catch of molting lobsters was observed in autumn (Keser et al. 1983). Smaller lobsters (60-70 mm CL) may have a higher probability of molting twice in a year than do larger lobsters (Blake and Landers, unpublished data). Frequency and timing of lobster molts were examined using the Gompertz growth function fitted to cumulative percent-molt data for 2- and 3-unit studies (Fig. 5). The inflection points of the growth curves were used to estimate annual dates of peak molting. Annual molting peaks were significantly ($p<0.05$) correlated with mean May bottom water temperatures and indicated that molting occurred earlier when May water temperatures were warmer than average. Conversely, peaks occurred later when May water temperatures were colder than average. Peak molt during 2-unit studies occurred on 27 June, which was one week later than the peak molt observed during 3-unit studies (20 June; Fig. 5); the corresponding average bottom water temperatures during May of the two study periods were 9.2°C (2-unit) and 9.7°C (3-unit). The latest peak in number of molting lobsters for these studies occurred in 1994 (12 July) when average May water temperatures were the coldest recorded to date (8.4°C; Fig. 6). The earliest molting peak occurred in 1991 (12 June) when average bottom water temperature during May was 10.8°C (Fig. 6). The influence of varying water temperature on the molt cycle has been well documented by Aiken and Waddy (1980), and Templeman (1936) found that molting was delayed a week or more when summer water temperatures in the Canadian Maritimes dropped 1°C.

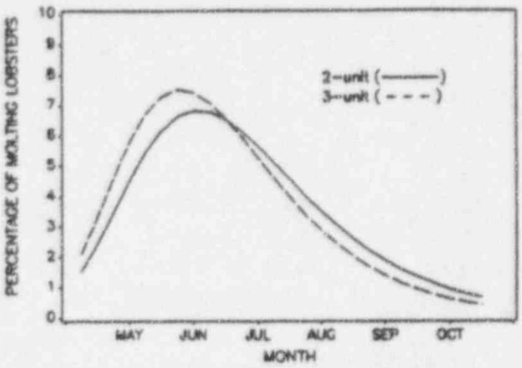


Fig. 5. Molt frequency curves estimated from the Gompertz function of lobsters caught during 2-unit studies (1978-85; —) and 3-unit studies (1986-94; - - -).

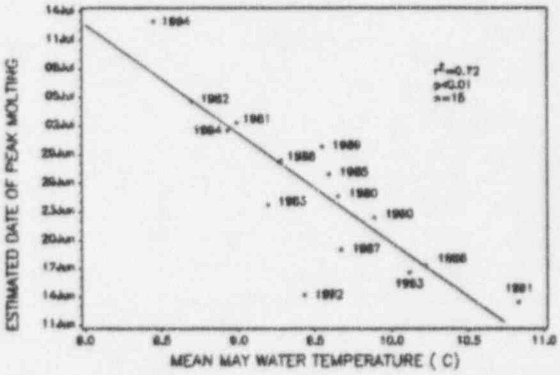
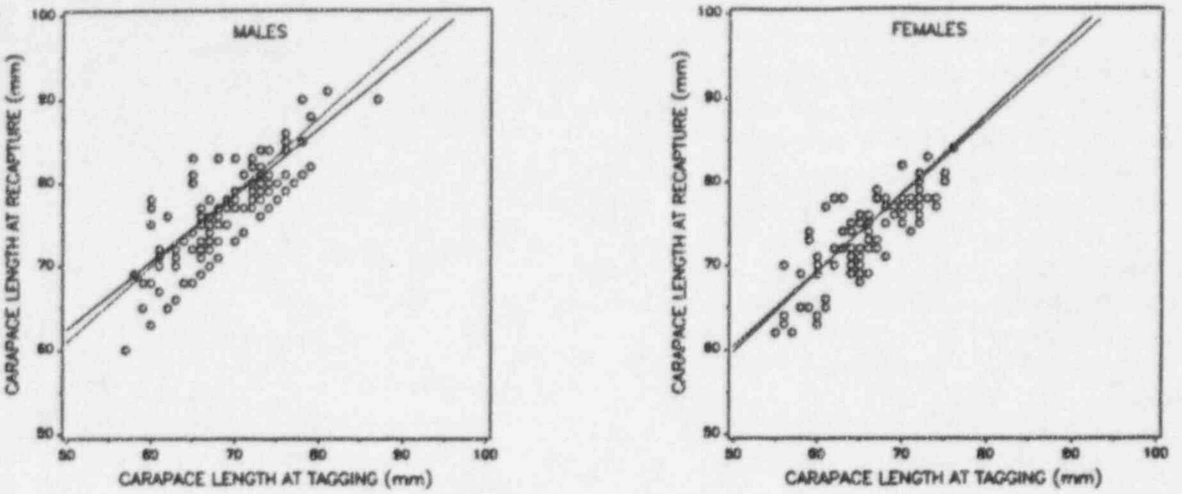


Fig. 6. Relationship between the date of peak molting (t from the Gompertz function) and annual mean bottom water temperature during May.

Growth per molt was determined for 260 lobsters that had molted between tagging and recapture during 1994. Simple linear regressions best describe growth for the size range of lobsters caught in our studies (Wilder 1953; Kurata 1962; Mauchline 1976) and were used to compare growth for males and females caught during 1994 and in 2-unit and 3-unit studies (Fig. 7). Growth



	MALES				FEMALES			
	N	Growth model	R^2	Increment	N	Growth model	R^2	Increment
2-Unit	380	$y=22.168+0.805(x)$	0.70	8.9 mm, 13.3%	587	$y=12.678+0.942(x)$	0.79	8.7 mm, 13.0%
3-Unit	728	$y=16.193+0.895(x)$	0.76	9.1 mm, 13.7%	913	$y=15.325+0.901(x)$	0.73	8.9 mm, 13.7%
1994	162	$y=20.374+0.806(x)$	0.64	7.4 mm, 11.4%	98	$y=22.522+0.772(x)$	0.65	7.0 mm, 10.3%

y=size at recapture, x=size at tagging (mm).

Fig. 7. Linear regressions, parameter estimates and average growth increments (mm and %) of carapace lengths at tagging and recapture times for male and female lobsters caught during 2-unit studies (1978-85; ——), 3-unit studies (1986-93; - - -), and during 1994 (O).

TABLE 9. Summary of lobster growth (in mm and as a percentage) at each station in wire pot catches for the period May through October from 1979 to 1994.

	Jordan Cove			Intake			Twotree		
	N	Growth (mm)	Percentage	N	Growth (mm)	Percentage	N	Growth (mm)	Percentage
1979	33	7.3	10.6	22	8.8	12.8	21	11.1	16.3
1980	38	8.6	12.7	21	8.7	12.5	33	10.1	14.8
1981	29	7.9	11.8	24	8.9	13.1	40	10.3	15.4
1982	48	9.0	13.3	55	7.8	12.0	96	9.1	13.2
1983	40	8.8	13.4	23	9.5	14.2	71	9.6	14.5
1984	85	9.0	13.8	44	7.8	12.3	79	8.8	12.9
1985	63	8.4	12.8	25	8.8	13.7	77	8.0	11.7
1986	61	9.1	13.5	39	7.5	11.6	69	8.6	12.9
1987	71	7.9	12.0	41	8.6	12.8	67	8.9	13.2
1988	93	8.5	12.8	58	9.5	15.2	104	9.6	14.7
1989	82	9.3	14.3	72	9.5	14.4	80	9.2	14.1
1990	93	9.1	13.9	51	9.2	14.2	58	10.2	15.5
1991	57	8.4	12.6	65	8.9	13.4	65	9.8	14.7
1992	107	8.9	13.8	48	8.8	13.2	81	9.4	14.6
1993	68	8.5	13.0	35	8.4	12.7	76	8.6	13.2
1994	84	7.3	10.9	63	7.0	10.7	113	7.2	10.6
2-Unit 79-85	336	8.6	12.9	214	8.5	12.8	417	9.2	13.6
3-Unit 86-94	716	8.6	13.0	472	8.7	13.2	713	8.9	13.6

increments averaged 7.4 mm (11.4%) and 7.0 mm (10.3%) during 1994 for males and females, respectively, and were smaller than the average growth increments during 2-unit (8.9 mm, 13.3% for males; 8.7 mm, 13.0% for females) and previous 3-unit studies (9.1 mm, 13.7% for males; 8.9 mm, 13.7% for females). The smaller growth increments observed during 1994 may be related to the below average water temperatures during the spring. The smaller incremental growth of females was related to their reproductive cycle; energy that could be used for carapace growth is diverted to widening of the abdomen and development of ovaries. Below normal spring water temperatures during 1994 influenced growth at each station. Growth increments at Jordan Cove (7.3 mm), Intake (7.0 mm), and Twotree (7.2 mm) were the lowest observed since the study began (1978-1993 ranges: Jordan Cove 7.3-9.3 mm, Intake 7.5-9.5 mm, Twotree 8.0-11.1 mm; Table 9). Lobster growth appears to be unaffected by plant operation; growth per molt at Jordan Cove was 8.6 mm during both 2- and 3-unit studies and varied only slightly at Intake and Twotree during the two study periods (by 0.2 and 0.3 mm, respectively; Table 9). In addition, lobster growth determined from our tag and recapture studies during 2- and 3-unit operation was within the range of growth values reported throughout LIS, where growth has averaged between 11.6% and 15.8% for males and between 12.0% and 15.4% for females (Stewart 1972; Briggs and Mushacke 1984; Blake 1994).

Culls

The percentage of lobsters missing one or both claws (culls) was 9.8% of the total catch during 1994, which was the smallest percentage reported since the study began (previous range 10.0-15.5%; Table 10). Percentage of culls was lowest at Twotree (6.9%) and highest at Jordan Cove (11.8%); the 1994 values at these two sites were within the range of previous 2- and 3-unit studies (Table 10). At Intake, the percentage of culls was 11.5% during 1994, which was the lowest percentage observed at this site since 1978 (12.6-17.8%; Table 10). Claw-loss was lower during combined 3-unit studies (10.8%) than during the 2-unit study period (12.1%), likely due to the implementation of the escape vent regulation in 1984. This regulation requires that pots contain a $1^3/_4$ by 6 inch opening to allow escape of sublegal-sized lobsters, and thereby reduces injury and mortality associated with overcrowded pots (Landers and Blake 1985). The benefits of incorporating escape vents in lobster traps has been well documented throughout New England (Krouse and Thomas 1975; Fair and Estrella 1976; Krouse 1978; Pecci et al. 1978; Fogarty and Borden 1980; Krouse et al. 1993).

TABLE 10. Percentage of culls (lobsters missing one or both claws) caught in wire pots from 1978-1994.

	Jordan Cove	Intake	Twotree	All Stations
1978	21.5	14.7	9.8	15.5
1979	17.3	17.8	8.8	15.5
1980	13.5	16.4	10.4	13.4
1981	13.4	16.7	7.1	12.1
1982	13.9	14.1	7.0	11.3
1983	14.6	15.3	8.2	12.4
1984	11.5	15.1	6.6	10.8
1985	15.1	13.9	7.2	11.1
1986	10.9	14.7	6.8	10.6
1987	11.9	14.7	6.2	10.3
1988	13.7	13.6	6.5	11.1
1989	14.4	14.3	8.8	12.2
1990	12.3	16.2	8.1	11.9
1991	14.5	14.0	8.2	11.8
1992	11.4	12.9	6.9	10.0
1993	11.2	12.6	7.7	10.1
1994	11.8	11.5	6.9	9.8
2-Unit 78-85	14.4	15.2	7.6	12.1
3-Unit 86-94	12.4	13.7	7.3	10.8

Tagging Program

Of the 7,533 lobsters tagged during 1994, 26.2% were recaptured in NUSCO traps, which represented the highest percentage recaptured in both 2- and previous 3-unit studies (14.4-25.2%; Table 11). While the percentage of recaptures was relatively high in NUSCO traps, the percentage recaptured in commercial traps during 1994 was only 9.3%, which was the smallest percentage reported since 1978 (previous range 17.1-47.6%). More lobsters were recaptured in NUSCO traps during the period of 3-unit operation (21.5%) than during 2-unit operation (18.9%). In contrast, fewer lobsters were recaptured by commercial lobstermen during 3-unit (18.1%) than during 2-

TABLE 11. Lobster tag and recapture statistics for NUSCO pots (May-Oct.) and commercial pots (Jan.-Dec.) from 1978 to 1994.

		NUSCO				Commercial			
	Number tagged	Number recaptured	Percentage recaptured	Percentage legal[a]	Mean CL(mm)	Number recaptured	Percentage recaptured	Percentage legal[a]	Mean CL(mm)
1978	2768	498	18.0	16.7	75.5	884	31.9	43.6	81.1
1979	3732	722	19.4	11.5	75.1	1778	47.6	27.2	77.6
1980	3634	522	14.4	18.8	75.7	1363	37.5	27.5	76.4
1981	4246	707	16.7	12.0	74.8	1484	35.0	25.9	76.3
1982	7575	1282	16.9	10.4	73.2	2519	33.2	23.0	75.5
1983	5160	932	18.1	11.3	73.6	2266	43.9	27.6	76.9
1984	5992	1431	23.9	8.4	73.0	1290	21.5	34.3	78.8
1985	5609	1216	21.7	7.7	73.2	1185	21.1	29.3	78.3
1986	5740	1194	20.8	4.7	72.3	1177	20.4	27.5	78.2
1987	5680	1356	23.9	5.5	72.8	1160	20.4	25.3	78.9
1988	6837	1725	25.2	4.3	72.0	1383	20.2	26.7	78.0
1989	6438	1233	19.2	4.4 (9.3)	72.9	1183	18.4	20.7 (24.8)	78.2
1990	5741	1066	18.6	5.5 (12.7)	73.3	1007	17.5	26.5 (32.8)	79.3
1991	6136	1109	18.1	7.4 (13.9)	73.4	1228	20.0	33.9 (41.5)	80.8
1992	9126	1842	20.2	3.9 (9.3)	72.4	1560	17.1	23.4 (28.5)	79.5
1993	8177	1708	20.9	3.6 (8.8)	73.4	1741	21.3	27.0 (47.4)	79.4
1994	7533	1974	26.2	3.1 (9.3)	73.4	701	9.3	19.1 (27.5)	77.7
2-Unit 78-85	38716	7310	18.9	11.0	73.9	12769	33.0	27.5	77.1
3-Unit 86-94	61408	13207	21.5	3.7 (8.4)	72.8	11140	18.1	24.5 (32.1)	79.0

[a] The minimum legal size from 1978 to 1988 was 81.0 mm (3 $^{3}/_{16}$ in), minimum legal size was increased in 1989 to 81.8 mm (3 $^{7}/_{32}$ in), and 1990 82.6 mm (3 $^{1}/_{4}$ in). Parenthetical values for percentage legal represent lobsters ≥ 81.0 mm carapace length.

unit studies (33.0%). The shift in percentages recaptured in NUSCO and commercial traps during 2- and 3-unit operations appears due to the escape vent regulation implemented in 1984 and not to plant operation. Installation of escape vents, coupled with the fact that most of our tagged lobsters are sublegal, resulted in fewer tagged lobsters retained in commercial traps. NUSCO traps do not have escape vents and have retained greater numbers of tagged sublegal lobsters since implementation of the escape vent regulation in 1984. Average size of lobsters recaptured in NUSCO during 1994 (73.4 mm) was within the range of previous 2-unit (73.0-75.7 mm) and 3-unit studies (72.0-73.4 mm; Table 11). Lobsters recaptured in commercial traps have consistently been larger than those recaptured in NUSCO traps; during 1994, the mean CL of lobsters recaptured in the commercial traps was 77.7 mm, 4.3 mm larger than the mean CL in NUSCO traps. The overall mean CL in commercial traps was smaller during 2-unit (77.1 mm) than during 3-unit studies (79.0 mm); this difference was due to the implementation of the escape vent regulation. Before escape vents were required (1978-83), commercial lobstermen recaptured many of the sublegal-sized tagged lobsters. Since the regulation was enforced, many sublegals escaped from the vented commercial pots, but were still retained in unvented NUSCO pots. In eastern LIS, Landers and Blake (1985) noted a substantial reduction in the number of sublegal-sized lobsters retained in vented pots, without a corresponding decrease in the catch of legal-sized lobsters. In Maine waters, Krouse et al. (1993) examined lobster catches in traps equipped with a variety of escape vent sizes (1 $^{3}/_{4}$, 1 $^{13}/_{16}$, 1 $^{7}/_{8}$ by 5 $^{3}/_{4}$ in). They found that 1 $^{7}/_{8}$ x 5 $^{3}/_{4}$ in vents retained fewer sublegals than did traps with smaller escape vents, and that the overall catch of legals was comparable for the 1 $^{3}/_{4}$ and 1 $^{7}/_{8}$ in vented traps.

The percentages of legal-sized lobsters (≥ 82.6 mm) recaptured in NUSCO (3.1%) and commercial (19.1%) traps during 1994 were the lowest reported since the study began (previous range 3.6-18.8% and 20.7-43.6%, respectively; Table 11). The overall percentage of legal-sized lobsters recaptured in NUSCO traps during 3-unit studies (3.7%) was substantially lower than the percentage during 2-unit studies (11.0%). Declines were also noted in the percentage of legal-sized lobsters recaptured by commercial lobstermen between the two operational periods (24.5% in 3-

unit vs. 27.5% in 2-unit studies). Declines in percentage of legal-sized recaptures are attributed to the increase in minimum legal size beginning in 1989 and to an increase in fishing effort, which has more than doubled since 1978.

Movement

Mark and recapture techniques were also used to follow lobster movements in the MNPS area. Tag return information from NUSCO sampling and commercial lobstermen were used to assess the extent of lobster migrations. Since 1978, the majority of lobsters recaptured in NUSCO traps were caught at the station where they were released (Intake 88%, Jordan Cove and Twotree 96%; Fig. 8); of the exchanges that did occur, most were between the nearshore Intake and Jordan Cove sites. The pattern of small-scale lobster movements was also observed in the percentage of commercial returns made within 5 km of MNPS during 1994 (99%); similar high percentages were reported in other 2- and 3-unit studies (94 and 96%, respectively). During 1994, lobsters traveled an average of 2.17 km before they were caught by commercial lobstermen, similar to the average distances traveled in 2-unit (2.36 km) and 3-unit studies (2.82 km). Another tagging study conducted in eastern LIS by Stewart (1972) demonstrated a strong homing behavior of the nearshore lobster population. Because lobsters are territorial and nocturnal, individuals have a limited home range; they leave their burrows at night and return to the same shelters before dawn. Similarly, our tagging studies indicate a predominance of localized movement which is typical of nearshore lobsters in coastal waters elsewhere in eastern North America (Templeman 1940; Wilder and Murray 1958; Wilder 1963; Cooper 1970; Stewart 1972; Cooper et al. 1975; Fogarty et al. 1980; Krouse 1980, 1981; Campbell 1982; Ennis 1984).

Most of the lobsters recaptured more than 5 km from the study area moved to the east or southeast, with only a few (21) of the lobsters migrating to western LIS (Fig. 9). Since 1978, over 800 lobsters were recaptured in The Race, a deep water channel 10.5 km from MNPS between LIS and Block Island Sound, which suggests a migration route for lobsters exiting LIS. Once out of the sound, lobsters moved easterly and were caught along the Rhode Island coast, Buzzards Bay (MA), and along the south shore of Cape Cod (Fig. 10). Some lobsters traveled southeast to offshore waters and were reported caught on the

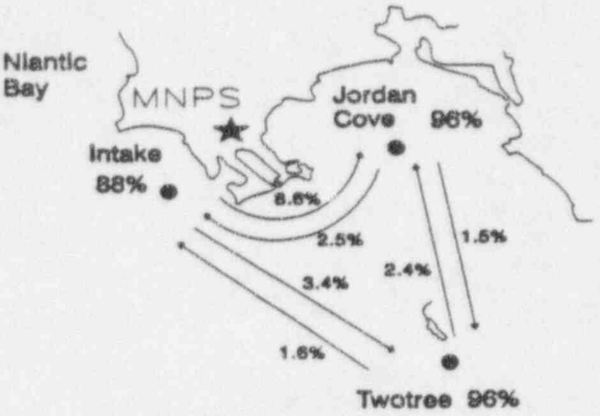


Fig. 8. Within station recapture and lobster movements between the three lobster sampling locations (1978-1994).

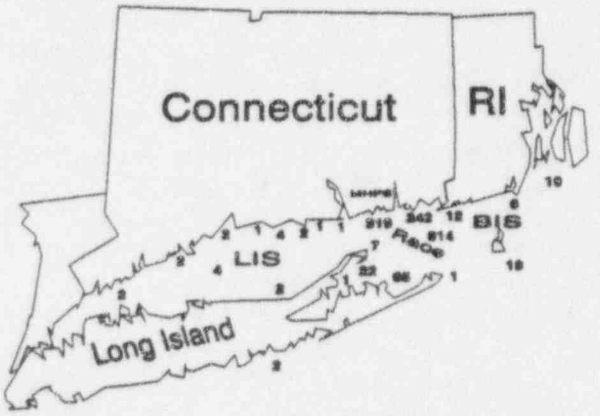


Fig. 9. Recapture location and number of lobsters caught by commercial lobstermen in the vicinity of Long Island and Block Island Sounds (1978-1994).

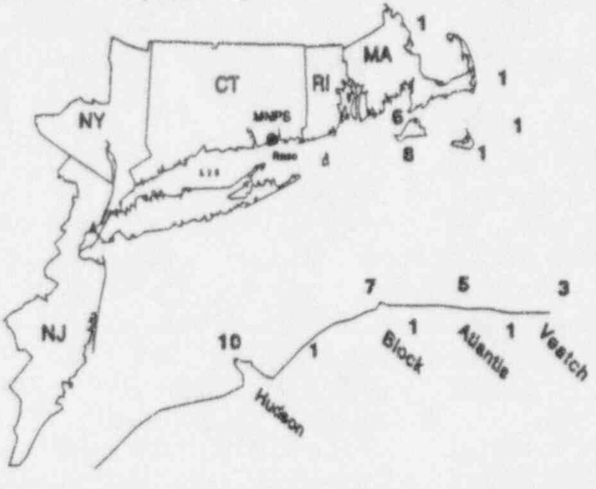


Fig. 10. Number of tag returns at locations >50 km from MNPS (1978-1994).

edge of the continental shelf in deep water submarine canyons (Block n=7, Hudson n=10, Atlantis n=5, Veatch n=3). Similar offshore migrations were demonstrated by other researchers working in waters from Canada to southern New England (Saila and Flowers 1968; Uzmann et al. 1977; Cooper and Uzmann 1980; Campbell and Stasko 1985, 1986).

Entrainment

Lobster larvae were found in MNPS cooling water samples from 25 May to 4 August 1994, which was the latest commencement and termination of the period of occurrence for larvae in our 11 year entrainment study. The delayed 1994 larval season was most likely due to below normal spring water temperature, which probably slowed hatching and development of larvae. A total of 257 lobster larvae was collected during 1994, which was within the range of values for 3-unit studies (157-625), but higher than the number collected in 2-unit studies (102 and 143). Lobster larvae pass through four pelagic stages before settling to the bottom as juveniles. During 1994, stage composition of larvae collected in entrainment samples was 40% Stage I, 3% Stage II and III combined, and 57% Stage IV (Fig. 11). Stage I larvae predominated in many of the previous collections ranging 38% and 90% of the four larval stages collected during 2- and 3-unit studies (Fig. 11). Stage II and III larvae were rarely collected in our entrainment samples, and with the exception of the 1988 and 1992 collections, these larval stages have accounted for only 5% of the total larvae collected since 1984. Stage IV larvae comprised between 4 and 52% of the four larval stages collected in previous studies (Fig. 11). Larval lobster distribution and dispersal in coastal waters have been associated with surface water circulation patterns (Fogarty 1983) which develop visible sea surface "fronts" often seen on the surface waters as "slick" lines. These lines delineate zones where wind-induced surface currents converge and downwell, and have been reported to contain high densities of planktonic organisms including lobster larvae (Cobb et al. 1983; Blake 1988). This mechanism of larval transport would explain the patchy distribution of lobster larvae observed in our entrainment studies. Other researchers working in southern New

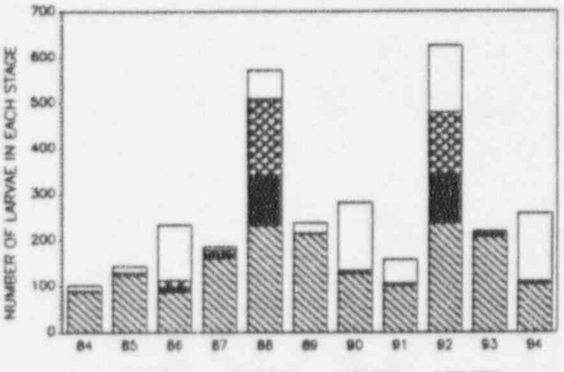


Fig. 11 Annual number of lobster larvae (Stage I-IV) collected in samples of the MNPS discharges from 1984 to 1994.

TABLE 12. Delta mean density (number per 1000 m^3 ± 5% C.I.) of lobster larvae collected in day and night entrainment samples from 1984 to 1994.

Year	Time of day	Delta mean density[a]	95% C.I.
1984	Day	0.158	0.061-0.256
	Night	0.737	0.138-1.336
1985	Day	0.390	-0.041-0.820
	Night	0.620	0.2[illegible]-0.951
1986	Day	0.324	0.063-0.585
	Night	1.399[b]	0.556-2.242
1987	Day	0.791	0.040-1.542
	Night	0.667	0.205-1.129
1988	Day	0.727	-0.199-1.653
	Night	0.688	0.271-1.106
1989	Day	0.158	0.087-0.229
	Night	1.403[b]	0.537-2.269
1990	Day	0.341	0.101-0.581
	Night	1.167[b]	0.569-1.765
1991	Day	0.287	0.131-0.442
	Night	0.756[b]	0.502-1.010
1992	Day	1.299	0.043-2.555
	Night	1.369	0.530-2.209
1993	Day	0.963	-0.207-2.132
	Night	1.168	-0.097-2.433
1994	Day	0.268	0.085-0.452
	Night	1.505[b]	0.706-2.303

[a] Number per 1000 m^3.

[b] Significant difference between day and night densities based on 2-sample t-tests ($p<0.05$).

TABLE 13. Annual delta mean density (number per 1000 m^3) of lobster larvae in entrainment samples during their season of occurrence and annual entrainment estimates with 95% C.I. for MNPS from 1984 to 1994.

Year	Time period included	Number larvae	Mean density[a]	95% C.I.	Cooling Vol. ($m^3 \times 10^6$)	Estimate	95% C.I.
1984	21May-10Jul	102	0.409	0.184-0.635	189.4	77,458	34,847-120,259
1985	15May-16Jul	142	0.504	0.258-0.749	255.1	128,550	65,806-191,040
1986[b]	14May-14Jul	232	0.857	0.418-1.297	666.2	566,619	278,457-864,017
1987	18May-30Jun	184	0.943	0.274-1.613	423.8	399,608	116,111-683,529
1988	16May-1Aug	571	0.717	0.296-1.137	837.6	600,573	247,935-952,372
1989	22May-28Jul	237	0.701	0.358-1.044	562.8	394,518	201,480-587,556
1990	14May-30Jul	280	0.748	0.436-1.060	779.1	582,738	339,671-825,805
1991	7May-22Jul	157	0.525	0.365-0.685	564.1	296,173	205,910-386,435
1992	19May-14Jul	625	1.334	0.652-2.016	461.2	615,285	300,724-929,846
1993	24May-25Jun	218	1.081	0.273-1.889	360.6	389,767	98,433-681,101
1994	25May-4Aug	257	0.908	0.445-1.371	745.2	676,639	331,613-1,021,665

[a] Mean densities are based on the delta-mean (NUSCO 1988b and Pennington 1983).

[b] Unit 3 began commercial operation.

England waters have found similar high variability in both the numbers and stages of larvae collected (Bibb et al. 1983; Fogarty 1983; Lux et al. 1983; Blake 1984, 1988).

Entrainment samples collected at night during 1994 contained significantly higher densities of lobster larvae (1.505 per 1000 m^3) than samples collected during the day (0.268 per 1000 m^3; Table 12). Significantly higher night lobster larvae densities than day densities were observed in four previous 3-unit study years (1986, 1989, 1990, 1991; Table 12). The factors causing differences in the day and night lobster larvae entrainment densities are unclear and the rarity of night sampling by other researchers in New England provides little information on the diel behavior of the larval phases. Early laboratory studies demonstrated positive phototaxis of Stage I larvae (Templeman 1937, 1939). In contrast, field surveys conducted on Browns Bank, southwest of Nova Scotia, indicated that most Stage I larvae were collected at depths between 15 and 30 m during the day and rarely found below 10 m at night (Harding et al. 1987). More recent laboratory studies by Boudreau et al. (1991) indicated that thermal gradients were the primary factors influencing vertical migration of lobster larvae, with all four stages seeking the warm water above the thermocline regardless of time of day.

The Δ-mean density of lobster larvae for all entrainment samples collected in 1994 was 0.908 per 1000 m^3, which was within the range of densities reported in previous 3-unit studies (0.525-1.334) but higher than the densities reported in 2-unit studies (0.409 and 0.504; Table 13). An estimated 676,634 lobster larvae were entrained through the MNPS cooling water system in 1994, which was the highest estimate reported since our entrainment studies began (1984-93 estimates ranged from 77,458 to 615,285; Table 13). Since Unit 3 began commercial operation in 1986, entrainment estimates have been substantially higher, because the cooling water demand of Unit 3 alone is approximately the volume required by Units 1 and 2 combined. The higher entrainment estimate for 1994 was due to a combination of the long period of occurrence of larvae (72 days), relatively high density, and high cooling water flow at the three MNPS Units during June and July, particularly at Unit 3, which operated at full power throughout the summer. Similarly, in 1988 and 1990 when the larval period was long (78 days) and cooling water flows were high, entrainment estimates were correspondingly high.

Evaluating the effect of entrainment on lobster recruitment in the MNPS area is difficult because of the unreliability of survival estimates for larvae and post-larvae (Phillips and Sastry 1980; Caddy and Campbell 1986; Cobb 1986; Blake 1991). Estimates of survival during the larval phase are wide ranging, from < 1% in Canadian waters (Scarratt 1964, 1973; Harding et al. 1982) to > 50% in LIS (Lund and Stewart 1970; Blake 1991). In addition, little is known on the source of larvae entrained through the MNPS cooling water systems. Most of the entrained Stage I larvae probably originate from the berried females in the MNPS area, because Stage I larvae are only in the water column for 3 to 5 days before molting to the next stage. The source of Stage IV larvae entrained at MNPS is probably not from the local population because these larvae are between 4 and 6 weeks old and water currents would carry them to other areas of LIS. Stage IV larvae were also reported to exhibit directional swimming behavior and moved tens of kilometers from the origin of hatching (Cobb et al. 1989; Rooney and Cobb 1991; Katz et al. 1994). Lund and Stewart (1970) indicated that the large number of berried females found in western LIS (27%; Smith 1977) may be responsible for recruitment of Stage IV larvae in middle and eastern LIS.

Conclusions

The status of the LIS lobster fishery has been characterized as highly exploited. Since 1978, fishing effort in Connecticut waters has more than doubled. The fishery is almost completely dependent on new animals molting into legal size; each year nearly all the lobsters above the minimum legal size are removed by fishing. The total number of lobsters caught and total CPUE in our study area reached record levels in 1992 and remained high during 1993 and 1994. However, legal lobster catches have significantly declined since the NUSCO study began in 1978. Legal catches were expected to improve in 1993 and 1994 after large numbers of lobsters, observed to be just below legal size in the 1992 catches, molted to legal size. Instead, legal CPUE continued to decline during the past two years in contrast with recruitment patterns of previous study years when strong recruit classes were followed by increased legal catches one to two year later. The fact that legal catches did not improve during the past two years may be further evidence that the local lobster resource is currently overfished.

During 1994, the peak in number of molting lobsters was later, and growth per molt was lower than previously observed in our studies; these conditions were attributed to below normal water temperatures during spring 1994. Changes in the size structure, sex ratio and proportion of berried females of local lobsters may be primarily due to increased fishing rates and to implementation of fishery regulations in 1984 (escape vents) and 1988 (increased minimum size). The lower incidence of claw loss, and changes in recapture rates and size structure of tagged lobsters caught in NUSCO and commercial traps during 3-unit studies were attributed to the use of escape vents. The percentage of berried females collected nearly doubled during 3-unit studies and was probably related to the increase in the minimum legal size. Both of these regulations were implemented to improve lobster survival and appear effective. However, fishing effort (number of fishermen and traps, and frequency of trap hauls) continues to increase and fishery managers question the stability of the resource under that kind of fishing pressure.

Lobster larvae densities were higher during 3-unit studies due to the higher percentage of berried females. Estimated numbers of larvae entrained through the MNPS cooling water systems were also higher during 3-unit operation, due to the combination of higher densities and the higher cooling water demand of Unit 3. Higher larval entrainment may affect subsequent legal lobster abundance, but quantification of this impact is difficult given the uncertainty of larval origin, larval survival, and recruitment rates to legal size. Since lobsters require 4-5 years of growth before they are vulnerable to capture, and an additional 2 years of growth to reach legal size, a decline in local lobster abundance caused by larval entrainment would not be apparent for several years. The high total CPUE for lobsters in our study area in the past three years suggests that entrainment during the initial years of 3-unit operation (1986-88; 7-9 years ago) has not decreased the local adult population.

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Marine Woodborer Study

Marine Woodborer Study

Introduction

The Marine Woodborer Study focuses on a single species, the shipworm *Teredo bartschi*, which has maintained a resident population in the warm water discharge of the Millstone Nuclear Power Station (MNPS) since 1975. *T. bartschi* is a semitropical species common from Texas to South Carolina (Turner 1966) and first demonstrated an ability to colonize warm water discharges in temperate climates during the early 1970s at the Oyster Creek Nuclear Generating Station (OCNGS) in New Jersey (Turner 1973; Hoagland 1983). Hoagland (1983) reported that distribution and abundance of the Oyster Creek population followed the pattern of power plant operation. An extended outage of OCNGS in the mid-1980s resulted in eradication of this shipworm population from the Oyster Creek and Forked River areas, with no reoccurrence through 1993 (Hillman and Belmore 1994).

At MNPS, temperature conditions of the cooling water discharges permitted *Teredo bartschi* to maintain a resident population over the last 20 years. This population was believed to have the potential of adapting to the cold water temperatures common to the winter months of LIS, thereby extending its distribution in the northeast (Hoagland 1981, 1983). The objective of the present study is to determine abundance and distribution of *T. bartschi* at sites both within and outside the thermal influences of discharge waters. This report covers the second year of sampling and data analysis of the *T. bartschi* study. Study design is based on over 20 years of monitoring data collected during the former Exposure Panel Studies (1968-1992).

Materials and Methods

Wood panels were submerged approximately 1 m from the water surface at seven sites: Effluent (EF), Effluent West (EW), Quarry Cuts (QC), Effluent Buoy (EB), High Rock (HR), Jordan Cove (JC), and White Point (WP) (Fig. 1). The EF site is on the east side of the MNPS effluent quarry, EW is on the west side of the quarry and panels at QC are located in the quarry cuts, where discharge waters from three Units enter LIS; panels at these three sites are exposed to undiluted effluent. EB is approximately 100 m outside the quarry cuts, where surface panels are directly in the path of the discharge plume during ebb tides, to ambient tidal waters during flood tides. HR and JC are approximately 500 m outside the quarry cuts, but within the discharge mixing zone (2-4 °C isotherms; NUSCO 1988) during ebb tides; HR is approximately 100 m closer than JC to the channel and the ebb tide path of discharge waters. Large rock outcroppings at the HR site cause eddies and alter water circulation patterns, which result in decreased effluent mixing and increased water temperatures that could enhance recruitment of *T. bartschi*. WP is the sample site most distant from MNPS (approximately 1700 m from the quarry cuts), and is considered unimpacted by the MNPS discharge.

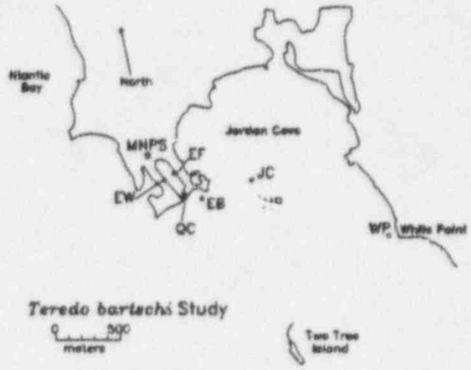

Fig. 1. Location of woodborer sampling sites in the vicinity of Millstone Nuclear Power Station: EF=Effluent (east side), EW=Effluent (west side), QC=Quarry Cuts, EB=Effluent Buoy, HR=High Rock, JC=Jordan Cove, WP=White Point.

At HR, JC and WP, two sets of three panels were deployed 1 m below the water surface by means of floats (double lobster pot buoys) anchored with a weighted (80 lbs.) wire lobster pot (Fig. 2); at EB, a similar array was attached 1 m below the mooring buoy used to mark the Effluent station of the Benthic Infauna program. The buoyed panels at EF on the east side of the effluent quarry were attached to a pulley-and-line

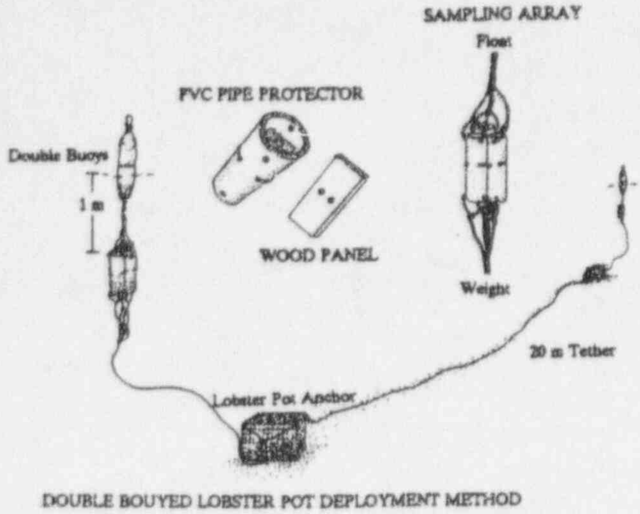

Fig. 2. The double buoyed lobster pot deployment method used to place wood panels 1 m below the surface at EF, HR, JC and WP in the Woodborer Study at MNPS.

system to facilitate retrieval, while those on the west side were attached to a stainless steel rack as used in previous studies (NUSCO 1993). At QC, panels were attached to a buoy and stiff-arm system to ensure that the panels remained submerged at the proper depth in the eddy currents. Panel placement in the water column (near the surface versus near the bottom) has been shown to affect shipworm recruitment beyond the quarry cuts (NUSCO 1991, 1992, 1993). Surface deployment tended to reduce settlement of *Teredo navalis*, the native shipworm, and enhance *T. bartschi* settlement during the 1990-1992 Discharge Study (NUSCO 1993).

Each panel (clear white pine 25.4 x 8.9 x 1.9 cm) was secured in a separate section of PVC pipe (35 cm in length x 10.2 cm diameter), except at EW where old methodology (excluding plexiglass backers) was used (NUSCO 1993). Two sets of panels per site were deployed in May and collected in November (redundant sampling was used to minimize data loss). Three panels at each site were processed by scraping fouling organisms and debris from all surfaces, X-raying (250 kV, 5 mA, for 45 s) each panel, and removing all or at least 70 shipworms per panel. Radiographs were used to subjectively estimate wood-loss, to locate shipworms within the panels, and to estimate total shipworm abundance. Shipworms were identified after removal from panels to determine percentage composition of *T. bartschi* and *T. navalis*. Shipworms smaller than 5 mm in tube length were classified only as juvenile teredinids and, although included as a component of shipworm abundance, were not included in the 70 shipworm subsample used for identifications.

Results and Discussion

Teredo bartschi continued to populate wooden panels in the Millstone Quarry and at sites outside the Quarry directly exposed to MNPS discharge waters. Recruitment of this shipworm at QC in 1994 (133/panel) was higher than that recorded in 1993 (16/panel) and was also higher at EB (1.3 vs. 0.7/panel; Table 1). *T. bartschi* did not recruit on panels at HR in 1994, while some recruitment was observed in 1993 (1/panel). The newly established (1993) EF site failed to recruit *T. bartschi* in 1993 and 1994, suggesting that water circulation or the placement of panels at this site may be inappropriate for recruitment. In 1994, panels were placed on the west side of the Millstone Quarry (EW) at the same location and using the sampling method described in NUSCO (1992). This sampling site was reestablished to evaluate the absence of *T. bartschi* from EF panels as observed in 1993. Densities of this shipworm at EW in 1994 (200/panel) were similar to those reported in panels from 1990-1992 (NUSCO 1992, 1993), indicating that panels at EF do not effectively reflect *T. bartschi* recruitment and abundance patterns in the effluent quarry. This situation was suspected in 1993 when *T. bartschi* was absent at EF, but present at QC, EB and HR. *T. bartschi* has not been collected at JC or WP during either 1993 or 1994.

Surface panels beyond the quarry cuts have been monitored since 1990 and during this period *T. bartschi* was collected at distances of 100 m (EB), 300 m and 500 m (HR) from QC. This shipworm was collected most consistently at EB, with densities ranging from <1 shipworm/panel in 1993 to a high of 78 shipworms/panel in 1990. Absence of *T. bartschi* in panels at JC and WP indicates that this immigrant species has not adapted to local ambient conditions. Although untreated oak pilings at White's Dock provide an attractive food resource for this shipworm, cool (<22 °C) summer temperatures combined with cold winter conditions appear unsuitable for recruitment and survival of *T. bartschi*.

Table 1. Average density (Ave.) and its standard error (SE) for shipworm abundance data collected from surface panels in the vicinity of MNPS from May to November during 1993 and 1994.

SPECIES	EF		EW		QC		EB		HR		JC		WP	
	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE
T. bartschi														
1993	0.0	0.0	-	-	15.7	8.1	0.7	0.7	1.0	0.6	0.0	0.0	0.0	0.0
1994	0.0	0.0	200.0	0.0	133.3	88.2	1.3	1.3	0.0	0.0	0.0	0.0	0.0	0.0
T. navalis														
1993	34.7	2.9	-	-	26.7	1.9	68.3	6.7	38.3	3.4	14.7	2.3	64.0	13.7
1994	30.7	0.9	9.0	0.6	15.7	3.5	34.0	6.4	38.3	2.6	16.7	0.3	30.3	1.8
Juveniles														
1993	0.0	0.0	-	-	0.3	0.3	3.0	1.5	2.0	0.0	2.0	0.58	5.0	1.5
1994	0.7	0.7	266.7	33.3	116.7	92.8	1.7	1.2	1.7	1.2	0.3	0.3	0.7	0.7
% Wood-loss														
1993	28.3	1.7	-	-	27.3	8.2	55.0	5.0	45.0	8.7	6.3	0.9	45.0	2.9
1994	26.7	1.7	76.3	[illegible]	58.3	16.9	35.0	5.8	28.3	1.7	8.3	0.9	10.7	0.7

Although not a major focus of this study, abundance of the native shipworm, *Teredo navalis* was also determined. In 1994, *T. navalis* was collected at all sites, with average densities ranging from 9/panel at EW to 38/panel at HR (Table 1). At most sites, these densities (and associated wood-loss) were lower than those in 1993. At QC, wood-loss in 1994 (58%) was higher than in 1993 (27%) despite lower densities of *T. navalis* (16/panel vs. 27/panel); however, this wood-loss was associated with high *T. bartschi* abundance (133/panel in 1994 vs. 16/panel in 1993). Highest wood-loss in 1994 occurred at EW (76%), also related to high *T. bartschi* density (200/panel).

Conclusions

Consistent recruitment and survival of *Teredo bartschi* remains in MNPS discharge waters. Results from over 20 years of study indicates that this population has not adapted to near-ambient or ambient temperature conditions at sites >500 m from the discharge. Panels at the EF site did not effectively monitor the presence of this shipworm in the Millstone Quarry. Under the current sampling design, *T. bartschi* recruitment on the western side of the quarry (EW) and in the western cut (replicates 1 and 2 at QC) has been similar to that observed during the previous Exposure Panel Studies and Distribution Studies. Sampling site HR (500 m from the quarry cuts) appears to be near the limit of suitable thermal conditions for *T. bartschi* recruitment; several individuals recruited onto panels there in 1993, although none were collected in 1994. In summary, *T. bartschi* occurs in MNPS water by virtue of higher water temperatures than those in Long Island Sound. Although a population has persisted in the MNPS Quarry since 1975, there is no evidence to suggest that *T. bartschi* has expanded its occurrence to other areas in Long Island Sound unaffected by MNPS discharge.

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Eelgrass

Eelgrass

Introduction

Many shallow subtidal areas in the vicinity of Millstone Nuclear Power Station support extensive populations of eelgrass (*Zostera marina* L.). The ability of this species to adapt to wide ranges of temperature, salinity and water depth (Osterhout 1917; Setchell 1929; Uphof 1941; Burkholder and Doheny 1968; Dillon 1971; Thayer et al. 1984) permits populations to successfully colonize many estuaries and lagoons of temperate and warm boreal coasts in the Atlantic and Pacific Oceans (Setchell 1935). The importance of this species to coastal ecosystems, now widely recognized and described in more detail in previous reports (e.g., NUSCO 1994), was brought to the forefront after the disappearance of most eastern North American and European populations attributed to 'Wasting Disease' (Tutin 1942; Rasmussen 1973, 1977). Following the destruction of *Zostera* populations, increased wave scour and changes in current patterns resulted in shoreline erosion. Concurrently, declines were observed in abundance of many animal species, including commercially important fishes and lobster (Stauffer 1937; Dexter 1947; Milne and Milne 1951; Orth 1973, 1977; Rasmussen 1973, 1977; Thayer et al. 1975; Zieman 1982).

Because of the ecological importance of eelgrass and the prediction that the 3-unit thermal plume would extend to the nearest population of the species in Jordan Cove (ENDECO 1977), the present study was initiated in 1985 to monitor this population and others nearby. Throughout the range of eelgrass, researchers have demonstrated that temperature changes can affect populations by reducing growth rate, lowering resistance to disease, and reducing the production and germination of seeds (Burkholder and Doheny 1968; Phillips 1974, 1980; Orth and Moore 1983). Objectives of the present study are to identify temporal patterns of eelgrass distribution, abundance and reproduction, and to determine the extent to which these patterns may be affected by natural variability or by MNPS operation.

Materials and Methods

Three eelgrass study sites in the vicinity of MNPS were sampled during 1994 (White Point-WP, Jordan Cove-JC, Niantic River-NR) (Fig. 1). The WP and JC stations are located 1.6 km and 0.5 km east of the power plants discharge, respectively, and are within the area potentially influenced by the 3-unit thermal plume (ENDECO 1977; NUSCO 1988). The NR site, located about 3 km from Millstone Point, is a control station in an area unaffected by power plant operation (Fig. 1). Water depths (at mean low water) were 2.5 m at WP, 1.5 m at NR and 1.1 m at JC.

The WP and JC sites have been sampled since 1985. The NR site has been relocated several times since 1985, due to shifts in eelgrass abundance patterns in the Niantic River. The original sampling site (#1, also designated 'old' in this and previous reports), located midway between Camp Weicker and the navigation channel (Fig. 1), was sampled throughout 1985 and in June 1986. A substantial population decline at site #1 was noted in July 1986, so another NR sampling site was established (#2) 50 m to the south, nearer the navigation channel. Site #2 was sampled for the remainder of the 1986 season; however, by September 1986, the eelgrass population at this site had also disappeared. In June 1987, a new NR sampling site was established at the nearest viable population, located in the lower river (#3). A slower, but steady, decline of the eelgrass population at site #3 has been documented since 1987 (NUSCO 1993), and by June 1993, no eelgrass shoots were observed at this site. In contrast, an extensive recovery of the eelgrass population at the old NR site (#1) was noted in 1993, and NR samples were again taken at this station during the 1993 and 1994 sampling periods (June-September). Monthly surveys and sediment sampling of NR #3 have continued after population elimination; however, no eelgrass recolonization has been observed at this site through the 1994 sampling year.

Samples were collected monthly from June through September 1994, the period of maximum standing stock and plant density. At each station,

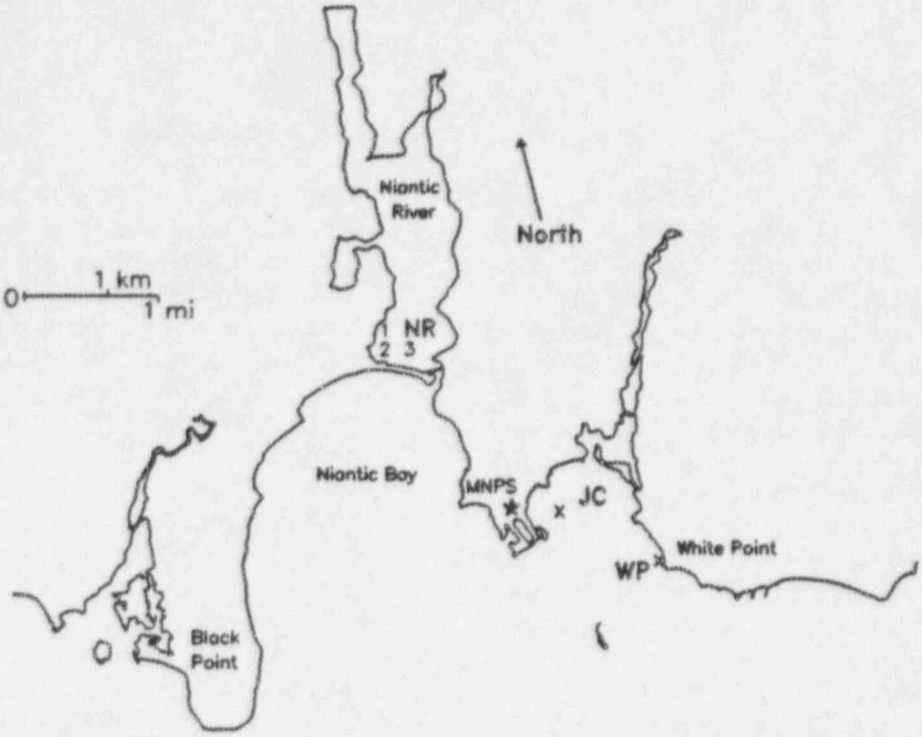

Fig. 1. Map of the Millstone Point area showing the location of eelgrass sampling stations, JC=Jordan Cove (x), NR=Niantic River (1=sampled 1985-June 1986 and 1993-1994, 2=sampled July 1986-September 1986, 3=sampled 1987-1992, was sampled in 1993-1994, but no eelgrass was present), WP=White Point (x).

16 samples were collected by SCUBA divers from randomly placed quadrats (25x25 cm, 0.0625 m^2) within a 10 m radius of the station marker. The upright shoots from plants within each quadrat were harvested, placed in a 0.333 mm mesh bag, and taken to the laboratory for processing. A 3.5 cm diameter x 5 cm deep core was taken together with eelgrass samples for analysis of sedimentary characteristics at each station. Temperature in Jordan Cove was measured by submerging an encased thermistor-recorder. Continuous temperature measurements have been recorded in Jordan Cove since 1991. Temperatures reported in this report cover the period from June 1 through September 30, when additional temperature increases above ambient from the 3-unit operation of MNPS could be most detrimental to eelgrass in Jordan Cove.

All shoots collected were counted in the laboratory and the longest blade of each shoot (up to 20 plants per sample) was measured to the nearest centimeter. The number of reproductive shoots in each sample was recorded and used to estimate the percentage of reproductive shoots in the population. Shoots were rinsed in freshwater to remove invertebrates and at the same time, all epiphytes were removed. Epiphytes on eelgrass shoots were minimal throughout the study, their weights were not recorded. Eelgrass standing stock was estimated as the weight of the shoots taken from each quadrat. From 1985 to 1987, shoots were weighed, then dried in an oven at 80 °C to a constant weight. Dry weights from 1988 to 1993 were estimated from the wet-weight/dry-weight relationship and presented in this report as grams dry weight/m^2 (g/m^2).

Nonparametric methods were used to examine trends in the time series of eelgrass shoot density and standing stock. The distribution-free, Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether the time-series of mean monthly standing stock biomass or shoot density exhibited significant trends. The slope of the trend, when significant, was estimated by Sen's estimator of the slope (Sen 1968). Eelgrass shoot length was not statistically analyzed, because growth occurs at the base of the shoot (from a basal meristem) and tips continuously erode, and

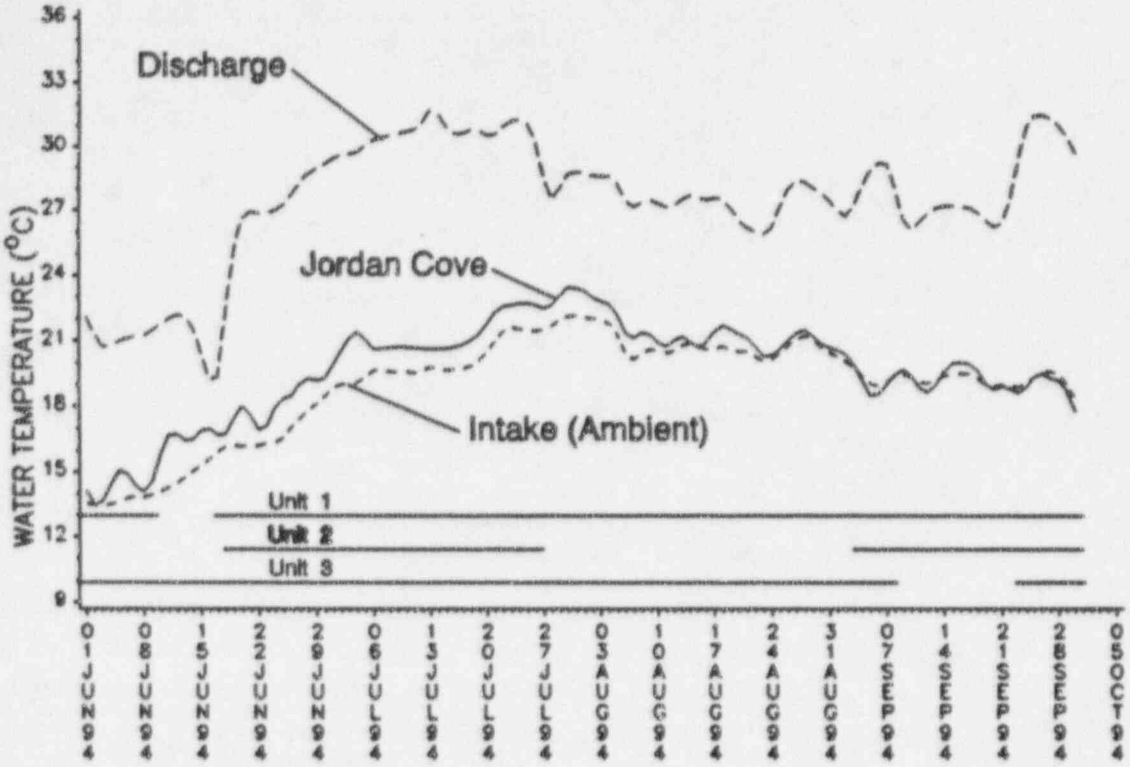

Fig. 2. Daily average water temperature at the Jordan Cove eelgrass station and at the MNPS intake and discharge, June thhrough September 1994. Units 1, 2, and 3 operations are also indicated for the same period.

because leaf turnover rate is highest during the summer (Roman and Able 1988).

Mean sediment grain size and silt/clay content were determined using the dry sieving method (Folk 1974). Sediment samples were heated to 500 °C for 24 h to determine organic content, estimated as the difference between dry-weight and ash-weight. Both silt/clay and organic content were recorded as a percentage of the total sediment sample weight. Additional sediment samples collected at the NR site #3 were analyzed to evaluate whether any sediment changes noted could be associated with the decline of eelgrass at this site.

Results

Temperature

Daily average water temperatures during 1994 (June through September) at JC and the MNPS intake and discharge are shown in Figure 2. Temperatures range from 13.3 to 23.8°C at JC, from 13.3 to 22.2°C at the intakes and from 16.9 to 32.4°C at the effluent discharge. The maximum difference between JC and intake (ambient) daily average water temperatures occurred on 4 July, when average water temperature at JC was 2.9°C warmer than at the intakes.

Sediments

Sediments supporting local eelgrass populations have been characterized since 1985 through monthly (June-September) measurements of mean grain size, silt/clay and organic content (Fig. 3). Overall, sediments at stations nearest MNPS (JC and WP) have been more stable over time than those at control stations in the Niantic River. Sediments collected during 1994 were coarser at JC (monthly mean grain size ranging from 0.20 to 0.25 mm), compared to those at WP (0.11-0.12 mm). Sediments at JC contained less silt/clay during 1994 (2.2-7.9%) than those at WP (8.1-15.3%). Similarly, sediment organic content was generally lower at JC (0.9-2.0%) than at WP (1.7-2.7%) in 1994. All sediment parameters measured in 1994

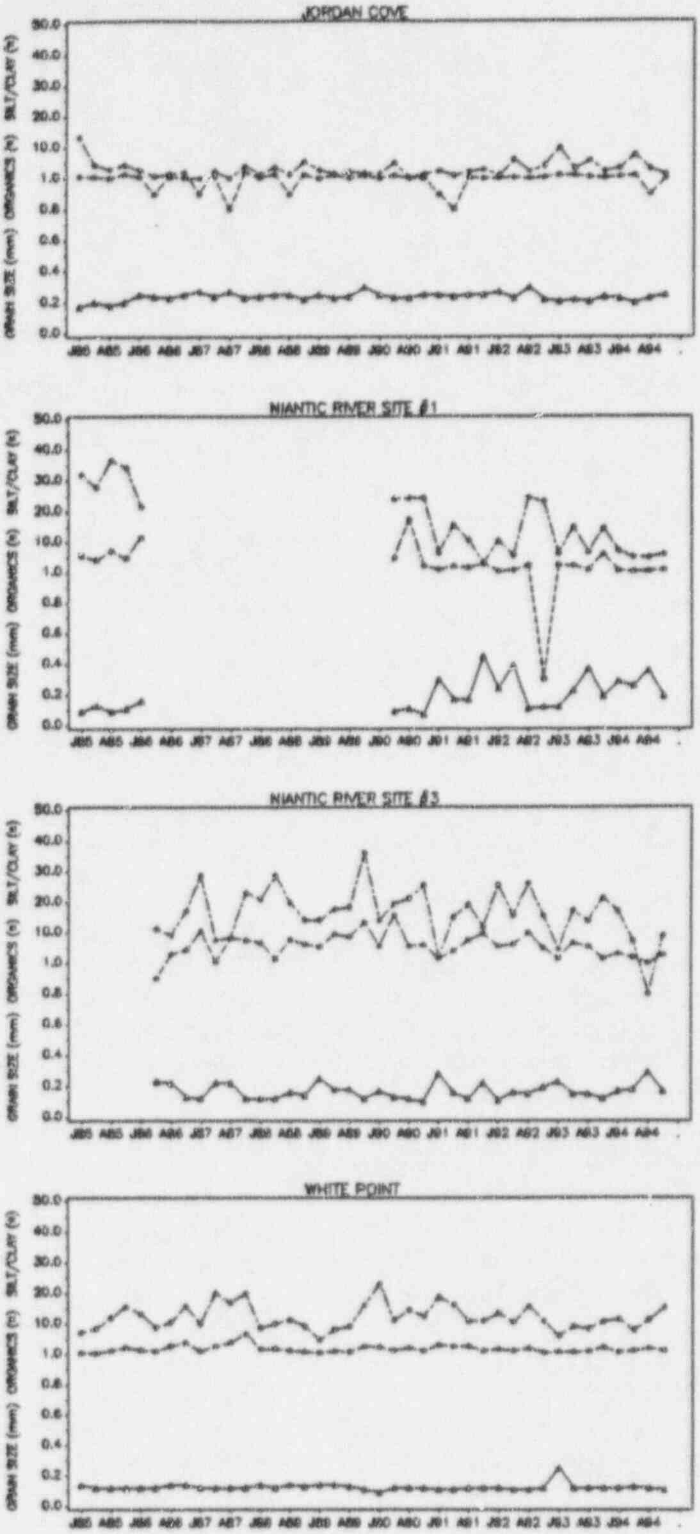

Fig. 3. Mean grain size (▲), organic content (★) and silt/clay content (○) of sediments at Millstone eelgrass stations, Jordan Cove, Niantic River (Site #1 and Site #3) and White Point sampled during the period June-September from 1985 through 1994.

TABLE 1. Annual and monthly average shoot density (no./m^2), length (cm) and dry weight standing stock (gm/m^2) for eelgrass sampled near MNPS during the June to September.

	ANNUAL MEANS										1994			
	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	Jun	Jul	Aug	Sept
Shoot Density														
Jordan Cove	572	713	542	468	411	338	603	630	484	282	230	384	251	261
Niantic River	413	72	294	307	240	225	249	233	385	132	182	340	5	-
White Point	286	218	227	161	335	185	242	204	310	141	181	152	108	122
Shoot Length														
Jordan Cove	57	57	77	75	74	38	48	53	54	35	29	35	39	35
Niantic River	50	39	81	88	94	73	51	48	58	28	24	30	24	-
White Point	107	116	126	86	110	106	87	72	107	95	111	108	78	69
Standing Stock														
Jordan Cove	243	276	258	238	202	105	169	210	160	60	44	63	70	43
Niantic River	156	32	184	181	183	143	81	79	125	18	23	47	1	-
White Point	265	260	201	90	236	180	148	110	275	100	147	110	71	72

- No plants were collected

at JC and WP were within the ranges for previous years.

Historically, sediment characteristics have been more variable at sites within the Niantic River (NR #1 and NR #3; Fig. 3). Ranges of monthly mean grain size at NR #1 and NR #3 were similar to each other, but considerably wider (0.19-0.36 mm and 0.17-0.29 mm, respectively) than ranges at WP and JC during 1994. Monthly silt/clay content in 1994 ranged from 5.4 to 7.5% at NR #1, and from 0.8 to 17.1% at NR #3. Sediment organic content ranges for 1994 at NR sites (1.3-1.7% at #1 and 1.0-3.8% at #3) were comparable to those at JC and WP. With the exception of low silt/clay content at NR #3 in August, all 1994 sediment characteristic measurements were within the historical ranges for each NR site.

Shoot Density

Relationships among sampling stations for eelgrass shoot density were consistent with those reported in most previous years. Annual mean shoot density was typically highest at JC (282 shoots/m^2 in 1994; Table 1), while densities at NR and WP were lower and more similar to each other (132 and 141 shoots/m^2, respectively) in 1994. Although spatial relationships were consistent during 1994, shoot densities were low area-wide compared to previous years. In fact, 1994 means were the lowest recorded over the study period at JC and WP, and the second lowest recorded at NR (next to 1986). Monthly shoot densities were highest in June at WP (181 shoots/m^2), and in July at JC and NR (384 and 340 shoots/m^2, respectively; Table 1). Moderate fluctuation in monthly shoot densities was observed at JC and WP when compared to the extreme fluctuation at NR. Following the density peak in July at NR, density declined dramatically to 5 shoots/m^2 by August and no plants were collected during September sampling.

Ten year time-series of monthly shoot densities at each station are presented in Figure 4. Trend analysis applied to this data indicated a significant decreasing trend at JC (slope=-4.917 shoots/m^2/sample period; $p<0.05$). No significant trend was detected at WP. Trend analysis was not performed on NR data due to insufficient data for any one site resulting from sampling station relocation following localized population elimination events.

Shoot Length

Yearly (1985-1994) and 1994 monthly (June-September) average shoot lengths are presented in Table 1. Shoots during 1994 were longest at WP (95 cm), shortest at NR #1 (28 cm) and intermediate at JC (35 cm). The 1994 average

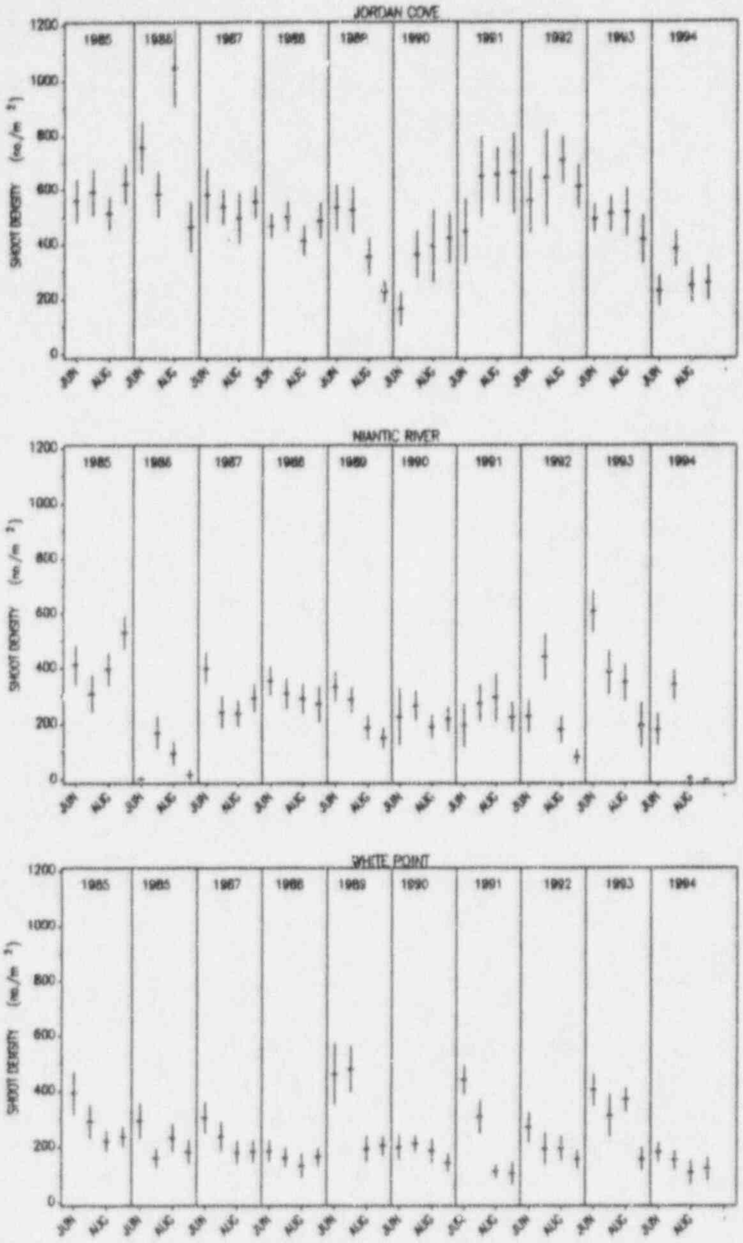

Fig. 4. Mean number of eelgrass shoots per m^2 ± 95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1994.

shoot length values at JC and NR #1 stations were the lowest values recorded at these stations since 1985 (previous ranges of 38-75 cm and 39-94 cm, respectively). The 1994 shoot length average at WP (95 cm) was within the overall range of previous annual means (72-126 cm).

Monthly shoot lengths in 1994 at WP declined from June through September (111 to 69 cm; Fig. 5); shoot lengths at JC increased slightly from 29 cm in June to 39 cm in August, then declined to 35 cm in September. Monthly shoot lengths at NR #1 increased slightly from June (24 cm) to July (30 cm) then decreased in August (24 cm). No plants were present in the NR #1 sampling area at the time of September 1994 collections.

Standing Stock

Eelgrass standing stock during 1994 was highest at WP (100 g/m^2), followed by JC (60 g/m^2) and

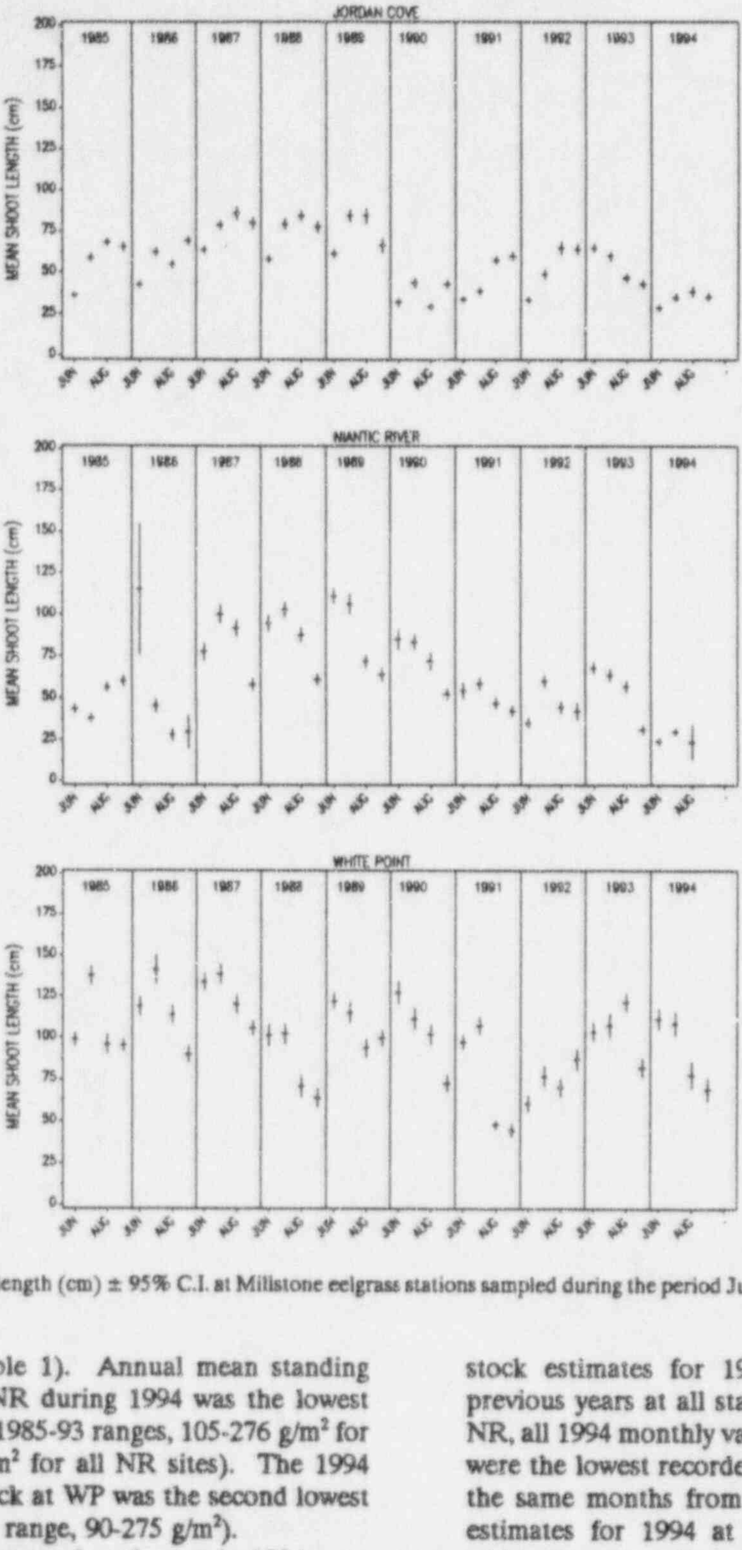

Fig. 5. Mean shoot length (cm) ± 95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1994.

NR (18 g/m^2; Table 1). Annual mean standing stock at JC and NR during 1994 was the lowest recorded to date (1985-93 ranges, 105-276 g/m^2 for JC and 32-183 g/m^2 for all NR sites). The 1994 mean standing stock at WP was the second lowest reported (1985-93 range, 90-275 g/m^2).

Monthly standing stock estimates in 1994 were highest in June at WP (147 g/m^2) and highest in July at JC and NR #1, (83 g/m^2 and 47 g/m^2, respectively) (Table 1). Monthly mean standing stock estimates for 1994 were low relative to previous years at all stations (Fig. 6). At JC and NR, all 1994 monthly values except for June at NR were the lowest recorded to date when comparing the same months from previous years. Monthly estimates for 1994 at WP, although low, were within the ranges for previous years.

Trend analysis, performed on monthly estimates over the ten year time-series, indicated that standing stocks had significantly declined since

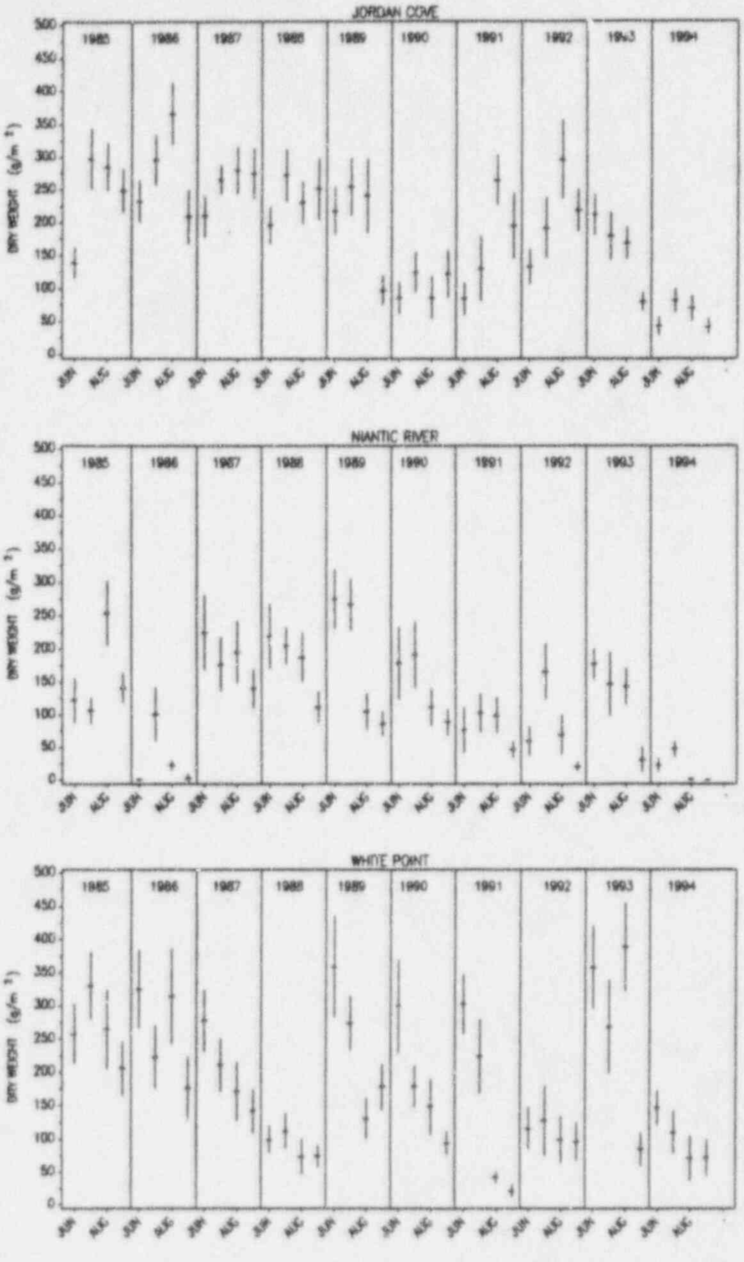

Fig. 6. Mean dry weight (grams per m²) ± 95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1994.

1985 at JC (slope=-3.180 g/m²/sample period, $p<0.001$) and WP (slope -3.782 g/m²/sample period, $p<0.007$). Trend analysis was not performed on the NR standing stock data due to frequent station relocation.

Reproductive Shoots

Lowest annual percentages of reproductive shoots were observed at JC and NR during 1994 (0.2 and 0.6% respectively; Table 2). At WP, 7.6% of the shoots collected in 1994 were reproductive, which was within the range of previous annual percentages (0.4-10.4%). Monthly percentage of reproductive eelgrass shoots in 1994 was highest in June at WP (16.0%); the monthly percentage of reproductive shoots was also highest in June at NR and JC, but accounted for only 0.4% and 1.6% of the shoots, respectively. No reproductive shoots were collected at any site after July 1994.

TABLE 2. Number of reproductive shoots, total number of shoots and percentage of reproductive shoots at eelgrass sampling stations from June 1985 through September 1994.

	Annual		June			July			August			September		
	#[a]	%	#	Total[b]	%	#	Total	%	#	Total	%	#	Total	%
							Jordan Cove							
1985	44	1.9	10	561	1.8	23	591	3.9	11	514	2.1	0	622	0.0
1986	70	2.5	23	756	3.0	21	585	3.6	13	1046	1.2	13	464	2.8
1987	72	3.3	18	581	3.1	24	537	4.5	19	496	3.8	11	555	2.0
1988	58	3.1	20	469	4.3	11	502	2.2	2	415	0.5	25	487	5.1
1989	30	1.8	16	534	3.0	12	526	2.3	2	356	0.6	0	228	0.0
1990	9	0.7	2	167	1.2	7	365	1.9	0	395	0.0	0	424	0.0
1991	24	1.0	14	448	3.1	10	647	1.5	0	654	0.0	0	662	0.0
1992	17	0.7	9	558	1.6	8	643	1.2	0	708	0.0	0	611	0.0
1993	93	4.8	56	493	11.4	36	510	7.1	1	516	0.2	0	417	0.0
1994	2	0.2	1	230	0.4	1	383	<0.1	0	251	0.0	0	261	0.0
							Niantic River							
1985	53	3.2	33	414	8.0	19	308	6.2	1	398	0.3	0	532	0.0
1986	15	5.3	1	3	33.3	14	170	8.2	0	95	0.0	0	18	0.0
1987	21	1.8	4	401	1.0	11	242	4.5	6	239	2.5	0	294	0.0
1988	44	3.6	19	356	5.3	17	309	5.5	0	290	0.0	8	273	2.9
1989	68	7.1	36	333	10.8	21	288	7.3	11	187	5.9	0	150	0.0
1990	53	5.9	19	225	8.4	32	266	12.0	2	189	1.1	0	218	0.0
1991	12	1.2	5	197	2.5	7	276	2.5	0	296	0.0	0	227	0.0
1992	5	0.5	1	229	0.4	4	442	1.0	0	181	0.0	0	81	0.0
1993	134	8.7	94	607	15.4	38	387	9.8	2	350	0.6	0	196	0.0
1994	3	0.6	3	182	1.6	0	340	0.0	0	5	0.0	0	0	0.0
							White Point							
1985	27	2.4	8	394	2.0	17	290	5.9	2	222	0.9	0	238	0.0
1986	79	9.1	51	293	17.4	14	161	8.7	6	234	2.6	8	182	4.4
1987	53	5.8	20	305	6.6	12	238	5.0	13	180	7.2	8	184	4.3
1988	30	4.7	3	186	1.6	13	161	8.1	5	133	3.8	9	164	5.5
1989	63	4.7	31	461	6.7	32	480	6.7	0	194	0.0	0	204	0.0
1990	77	10.4	47	199	23.6	25	212	11.8	5	186	2.7	0	144	0.0
1991	28	2.9	16	441	3.6	12	308	3.9	0	112	0.0	0	105	0.0
1992	4	0.4	1	270	0.4	3	194	1.5	0	195	0.0	0	155	0.0
1993	48	3.9	20	403	5.0	17	313	5.4	11	368	3.0	0	156	0.0
1994	43	7.6	29	181	16.0	14	152	9.2	0	108	0.0	0	122	0.0

[a] Total number of reproductive shoots.

[b] Total number of shoots (vegetative + reproductive).

Discussion

Results of the 1994 eelgrass monitoring program indicated a general area-wide decline in local populations. Specifically, study parameters used as indices for overall population health and productivity (shoot density, standing stock and number of reproductive shoots) were low at all stations during 1994 when compared to previous years. The extent of such declines varied from station to station, with no clear relation between degree of population decline and site proximity to the MNPS thermal plume.

The eelgrass population at WP exhibited the least severe declines in 1994, relative to populations at other study sites. While several indices at WP were low in 1994, most were still within the historical ranges of previous study years. Only standing stock at WP exhibited a significant decreasing trend over the ten-year time-series. This analysis was strongly affected by the 1994 data, since no such trend was detected last year (NUSCO 1994). No trend was evident in WP shoot density data, and shoot lengths in 1994 were consistent with those observed previously. Both numbers and percentages of reproductive shoots at WP were well within their respective historical ranges. The sedimentary environment at WP remained stable over the study period, including 1994.

The Niantic River sites showed the most dramatic decline, and historically the most variability. Rapid decline and local elimination of the current study population at NR #1 occurred in 1994, a scenario which has become typical for populations in the Niantic River since 1985. In fact, similar population demise was observed at the same site in 1986, at NR #2 in 1987, and a more gradual loss of plants at NR #3 was noted from 1987 to 1993. While localized eelgrass recolonization has been reported in the Niantic River previously (at NR #1 from 1989-1993; NUSCO 1994), little reestablishment of populations elsewhere in the river, including study sites NR #2 and NR #3, has been observed over the study period. In a recent survey, only small patches of eelgrass were observed along the main navigation channel in the lower river and near the entrance to Smith Cove farther upriver. These findings indicate a general population decline in the Niantic River. Eelgrass biology and population occurrence in the North Atlantic shows unpredictable behavior, and wide-spread population decline. Specifically, long-term eelgrass decline/recovery cycles have been reported in the Niantic River since the 1930s (Marshall 1994). Loss of eelgrass has been attributed to a variety of causes, ranging from natural, e.g., 'wasting disease' (den Hartog 1987), severe storms (Patriquin 1975), or uprooting by swans (Marshall 1994) to human activities, e.g. eutrophication (Bulthuis 1983; Orth and Moore 1983; Cambridge and McComb 1984; Neverauskas 1985; Burkholder 1993), land reclamation, or changes in near-shore land use (Kemp et al. 1983). The disappearance of eelgrass from the Niantic River in the late 1980s was attributed to a decline in water quality and the presence of *Labyrinthula* (Short 1988), and it is likely that these factors, perhaps further exacerbated by swan grazing, are largely responsible for more recent population losses at NR #3 and NR #1. It is reasonable to expect small transient populations to reappear in the river in the future through seed production and dispersal from nearby healthy populations. Regardless of the nature of environmental conditions in the Niantic River, none of the factors affecting eelgrass there appears related to the operation of MNPS, as these population sites are well beyond any influence of the power plant.

Power plant impacts, if they were to occur, would most likely be evident in eelgrass populations near the MNPS thermal discharge, such as the population in Jordan Cove. With modeling predictions indicating that the JC study population may be exposed the MNPS 3-unit thermal plume (ENDECO 1977; NUSCO 1988), population characteristics at this site have been closely monitored since 1985. Analysis of the current ten-year time-series showed declining trends in two important population parameters: shoot density and standing stock biomass. Levels of both shoot density and standing stock were relatively high in early study years (e.g., 1986), and with some fluctuations, have experienced long-term decreases as well as more pronounced declines after 1992. Also indicative of population stress at JC is the lack of reproductive shoots during 1994. Associating the general decline of the Jordan Cove eelgrass population with a power plant impact is difficult, however, due to the high natural variability of environmental conditions at this site.

Jordan Cove is shallow, with large sand flats that are exposed to summer heating, and, during extremely low tides, freezing in winter. Furthermore, similar to the Niantic River, shallow water in Jordan Cove allows for grazing of eelgrass by swans. Therefore, Jordan Cove is more susceptible to natural environmental stress than is the deeper water site, WP.

The importance of temperature in regulating eelgrass growth and development was first stressed by Setchell (1929). It was later shown that eelgrass is sensitive to small temperature variations (Thayer et al. 1984). Eelgrass does not produce seeds at temperatures above 15-20 °C (Burkholder and Doheny 1968; Orth and Moore 1983). Higher water temperatures, e.g., from heated effluents of power plants, could eliminate eelgrass from nearby areas (Phillips 1974; Thayer et al. 1984). Studies of another seagrass, *Thalassia*, in Florida (Roessler and Zieman 1969; Wood et al. 1969; Zieman 1970; Roessler 1971) and of a salt marsh grass (*Spartina alterniflora*) in Maine (Keser et al. 1978), showed a significant decline in abundance of these plants in the vicinity of power plant effluents. Elevated water temperatures increased respiration beyond levels that could be supported by plant photosynthesis. Recognizing that eelgrass meadows are among the most productive of marine systems (Mann 1973; McRoy and McMillan 1977; Zieman and Wetzel 1980) and act to stabilize sediments (Wood et al. 1969; Zieman 1972; Orth 1977), a decline in eelgrass abundance at Jordan Cove could also effect the movement of sediments and species abundance of associated infaunal communities.

The possibility of thermal plume impacts at JC was also investigated with data from a continuous temperature recorder deployed on the sediment surface within the JC study population. Some elevation in JC water temperatures was observed during the summer 1994, compared to ambient (at the MNPS intakes) temperatures (Fig. 2). This disparity from ambient temperatures, however, appears to be more related to insolation of the shallow water sand flats than to incursion of the thermal plume. Maximum temperature differences (2-3°C) coincided with the period of maximum daily solar irradiance (June-July), a relationship consistent with previous years (cf. NUSCO 1992, 1993 and 1994). Regardless of the cause, these increased temperatures could stress this population for reasons described above. Temperature data from the winter of 1993-94 also suggest stressful conditions for the JC eelgrass population. On two occasions, extreme low tides left the JC study area exposed to colder than normal air temperatures (with recorder registering minimum temperatures of ca. -8°C at the sediment surface), and may have resulted in sediment freezing and rhizome damage.

Conclusions

Eelgrass studies during 1994 indicated generally poor regional conditions for growth and reproduction of local populations of *Zostera marina*. Populations declines appeared related to site-specific stress mechanisms in combination with area-wide ecological factors. The eelgrass population at WP appeared to be most robust in the MNPS area. Population parameters at WP measured in 1994 were within ranges established over the previous nine study years. The deeper water WP study site (2.5 m depth) provides more protection against elevated temperatures from insolation in the summer, sediment freezing during low tides in winter, and swan grazing. Eelgrass populations at shallow water sites (NR #1 and JC; 1.1-1.5 m depth) are more susceptible to these stress mechanisms. Overall environmental stress was apparently greatest at NR #1 in 1994, and was related to regional and site-specific (depth-related) factors, and possibly water quality conditions in the Niantic River. *Zostera* populations were eliminated at NR #1 by September 1994. Similar localized elimination events have been observed at other sites in the Niantic River during the study period.

Shallow-water stress, as temperature extremes, were measured directly at JC, and were possibly responsible for population declines observed at that site. Sediment freezing may have occurred during two extreme low tide events in the winter of 1993-94. Elevated temperatures measured at JC during the summer appeared more related to insolation of the shallow sand flats in Jordan Cove than to thermal plume incursion. Low-level thermal plume incursion to JC is still a possibility, however, based on hydrothermal modeling studies, and could hinder recovery of this already stressed population from its most extreme decline to date.

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Rocky Intertidal Studies

Rocky Intertidal Studies

Introduction

A substantial portion of the shoreline in the vicinity of Millstone Nuclear Power Station (MNPS) is composed of rock ledge and boulders supporting a rich and diverse community of attached biota. Shore communities such as these are often in close proximity to coastal power plant heated effluents, and are therefore particularly vulnerable to thermal impacts. To effectively document overall ecological impacts associated with coastal power generation, rocky shore community studies have been, and continue to be, an important aspect of biological monitoring programs associated with nuclear power plants along the New England coastline (Vadas et al. 1976, 1978; Wilce et al. 1978; NAESCO 1994; NUSCO 1994).

Rocky intertidal studies are part of an extensive environmental monitoring program whose primary objective is to determine whether differences that exist among communities at several sites in the Millstone Point area can be attributed to construction and operation of MNPS, in particular since Unit 3 began operation in 1986. To achieve this objective, studies were designed and implemented to identify attached plant and animal species found on nearby rocky shores, to describe temporal and spatial patterns of occurrence and abundance of these species, and to identify physical and biological factors that induce variability in local rocky intertidal communities. This research includes qualitative algal sampling, abundance (percentage cover) measurements of intertidal organisms, and growth and mortality studies of *Ascophyllum nodosum*. The following report discusses results of sampling and analysis in the 1993-94 study year, and compares these results to those of 2-unit operational studies (March 1979-February 1986), and 3-unit operational studies to date (March 1986-September 1994).

Materials and Methods

Qualitative Sampling

Qualitative algal collections were made monthly at nine rocky intertidal stations (Fig. 1). These stations are, in order of most to least exposed to prevailing winds and storm forces: Bay Point (BP), Fox Island-Exposed (FE), Millstone Point (MP), Twotree Island (TT), White Point (WP), Seaside-Exposed (SE), Seaside-Sheltered (SS), Giants Neck (GN), and Fox Island-Sheltered (FS). The MP and TT stations were added in September 1981; all other stations have been sampled since March 1979. A year of qualitative sampling is from March to the following February, i.e., the latest year of qualitative algal data (1993) comprises collections from March 1993 to February 1994. The 1985 sample year (3/85 - 2/86) terminated the 2-unit operational period; the 1986 sample year (3/86 - 2/87) was the first in the 3-unit operational period.

The FE station, approximately 100 m east of the MNPS discharges, is directly exposed to the 3-unit thermal plume (during part of the tidal cycle); FS, WP, TT, and MP are between 300 and 1700 m from the discharges, and potentially impacted by the plume. Stations at BP, GN, SE, and SS are unaffected by MNPS operation.

Qualitative collections were used to characterize the attached flora at each site during each sampling period. At each station, specimens of observed algal species were collected at low tide from intertidal and shallow subtidal zones, including tidepools. Algal samples were identified fresh or after short-term freezing. Voucher specimens were made using various methods: in saturated NaCl brine, as dried herbarium mounts, or as microscope slide preparations.

The qualitative species list includes all attached, macroscopic algal species. Excluded from these studies are diverse diatom taxa, blue-green algae and some crustose, endophytic or endozooic algal species. These elements of the microbiota are present, but too difficult to consistently collect, and for many species to identify as components of

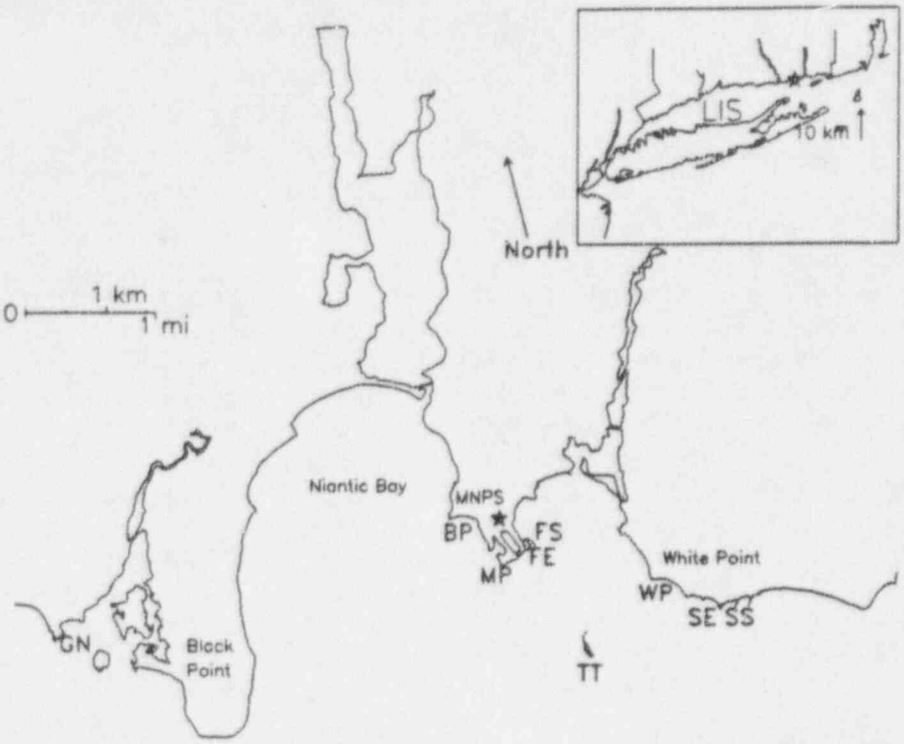


Fig. 1. Location of the MNPS rocky intertidal sampling sites: GN = Giants Neck, BP = Bay Point, MP = Millstone Point, FE = Fox Island-Exposed, FS = Fox Island-Sheltered, TT = Twotree Island, WP = White Point, SE = Seaside-Exposed, SS = Seaside-Sheltered.

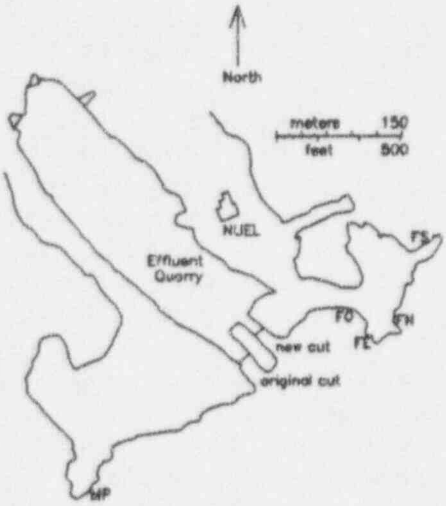


Fig. 2. Detail map of the MNPS vicinity: FO = original experimental *Ascophyllum* site (1979-84), FN = new experimental *Ascophyllum* site (1985-present), MP, FE, FS as in Fig. 1.

the flora used in a monitoring program. However, we included taxa that are, or may be, conspecific or subspecific forms, or alternate life history stages of erect macroalgae. For simplicity, we refer to each of these entities as a species throughout this report. Except where noted, nomenclature follows that of South and Tittley (1986).

Abundance Measurement

The abundance of rocky intertidal organisms was expressed as percentage of substratum cover. At each qualitative collection station except TT (because of insufficient exposed bedrock), five permanent strip transects were established perpendicular to the water-line, 0.5 m wide and extending from Mean High Water to Mean Low Water levels. Each transect was subdivided into 0.5 m x 0.5 m quadrats and was sampled six times per year, in odd numbered months (or a total of 46 times in the Unit 3 operational period to date). These transects are considered 'undisturbed', as they experienced no experimental manipulations. The total number of quadrats in each transect depended on the slope of the transect. The percentage of substratum cover of all organisms and remaining free space in each quadrat was subjectively determined. Understory organisms, species that were partially or totally obscured by the canopy layer, were assigned a percentage value that approximately corresponded to their actual substratum coverage. Each quadrat was assigned to a zone based on its tidal height: Zone 1 (high intertidal), Zone 2 (mid intertidal), or Zone 3 (low intertidal).

Ascophyllum nodosum Studies

Growth and mortality of tagged individuals of the perennial brown alga, *Ascophyllum nodosum*, were studied at two reference stations (GN, 6.5 km west of the discharge and WP, 1.5 km east of the discharge, Fig. 1) and an experimental station (FN, ca. 150 m from the quarry discharges, northeast of the Fox Island-Exposed sampling site, Fig. 2). *Ascophyllum* populations at GN and WP have been monitored since 1979, and those at FN since 1985. *Ascophyllum* was also monitored at FO, ca. 75 m east of the original Millstone quarry cut, from 1979 to 1984. The FO *Ascophyllum* population was eliminated in the summer of 1984 by exposure to elevated temperatures from the thermal plume discharged through two quarry cuts (NUSCO 1985).

Ascophyllum plants were measured monthly, after onset of new vesicle formation, from April to the following April. At each station, fifty plants were marked at their bases with a numbered plastic tag, and five apices on each plant were marked with colored cable ties. Linear growth was determined by measurements made from the top of the most recently formed vesicle to the apex of the developing axis, or apices if branching had occurred. Monthly measurement of tagged plants began in June; in April and May, vesicles were not yet sufficiently large to be tagged, and five tips were measured on each of 50 randomly chosen plants. Tags lost to plant breakage were not replaced, and the pattern of loss was used as a measure of mortality. Loss of the entire plant was assumed when both the base tag and tip tags were missing. Tip survival was based on the number of remaining tip tags.

Data Analysis

Analysis of qualitative algal collections includes a calculation of a frequency of occurrence index, based on the percentage of collections in which each species was found out of all possible collections (e.g., at a station, in a month, during 2-unit or 3-unit operation). This index was used to calculate similarities among collections, using the Bray-Curtis formula (Clifford and Stephenson 1975):

$$S_{jk} = \frac{\sum_{i=1}^{n} 2\min(X_{ij}, X_{ik})}{\sum_{i=1}^{n} (X_{ij} + X_{ik})}$$

where S_{jk} is the similarity index between collections j and k; X_{ij} is the frequency of occurrence index for species i in collection j; X_{ik} is the index in collection k; and n is the number of species in common. A flexible-sorting ($\alpha = -0.25$), clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967).

Quantitative analyses included determination of abundance of intertidal organisms as percentage of substratum covered by each taxon. Unoccupied substrata were classed as free space. Cover values of selected species were plotted against time. Similarities of communities among stations and between operational periods were calculated using the Bray-Curtis coefficient formula cited above, substituting untransformed percentages for frequency of occurrence indices. The same clustering algorithm was used to form station/period groupings.

Data from Fox Island - Exposed (FE) were also analyzed separately to determine relationships among qualitative algal collections and quantitative percent-coverage values. Similarity indices were calculated between each possible pair of yearly collections at FE; these annual comparisons permitted better resolution of the community changes that have occurred at this site.

A Gompertz growth curve was fitted to *Ascophyllum* length data using non-linear regression methods (Draper and Smith 1981). The Gompertz function form used (Gendron 1989) has three parameters, related by the formula:

$$L_t = \alpha e^{-e^{-k(t-t_0)}}$$

where L_t is the predicted length at time t, α is the asymptotic length (limit of total increase for the growing season), k is the rate of decrease of specific growth (shape parameter), and t_0 denotes the time at which the inflection point occurs (time when length is increasing most rapidly). Growth curve parameters were compared among stations and between periods using 2-sample t-tests ($P=0.05$) based on the asymptotic standard errors of the parameter estimates. Growth data representing the latest growing season (1993-1994) were plotted for all stations together and for each station separately, with summaries of 2-unit (1979-1986) and 3-unit (1986-1994) operational data. Because the FN station was established in 1985, 2-unit operational data from this site included only the 1985-86 growing season.

Results and Discussion

Qualitative Algal Studies

Water temperature is an important factor in determining distribution of many benthic marine macroalgal species. In fact, biogeographical regions for macroalgal species distribution in the world oceans are primarily delimited by water temperature isotherms (Lüning 1990). Therefore, because some macroalgal communities experience altered temperature regimes resulting from the MNPS thermal effluent discharge, shifts in spatial and temporal species occurrence patterns are a likely impact. The current qualitative algal sampling program is used to monitor these patterns by applying various floristic analyses to data compiled from periodic algal collections.

Results of qualitative sampling studies conducted since 1979 are presented as percent frequency of species occurrence during 3-unit and 2-unit operational periods by month (Table 1) and by station (Table 2). The 1993 total of species collected and identified was 121, which was typical of previous years and within the range of annual totals for 2-unit (101-131) and 3-unit (118-126) periods (NUSCO 1993). During 1993, no new species were added to the overall total of 161 species collected since 1979.

Although they are not newly found at our collection sites, two species have been renamed, based on recent taxonomic research. The red alga previously reported as *Phyllophora truncata* is now called *Coccotylus truncatus* (Wynne and Heine 1992). Another red alga, previously reported as *Antithamnion nipponicum* is now called *Antithamnion pectinatum* (Athanasiadis and Tittley 1994).

One power plant thermal impact on the local macroalgal community is seen as shifts in seasonal occurrence patterns of annual species typical of either warm-water or cold-water periods of the year. Therefore, it is necessary to first identify components of the flora which exhibit this type of natural seasonality, and then use this information as a baseline from which power plant induced changes can be assessed. In the Millstone area, a suite of species typical of cold-water period (January-June) collections includes *Dumontia contorta, Polysiphonia urceolata, Spongonema tomentosum, Desmarestia viridis, Chorda tomentosa, Ulothrix flacca, Urospora penicilliformis, Monostroma pulchrum, M. grevillei* and *Spongomorpha arcta* (Table 1). The floristic grouping characteristic of warm-water (July-December) collections includes *Champia parvula, Lomentaria baileyana, Callithamnion roseum, Ceramium diaphanum, Grinnellia americanum, Dasya baillouviana, Giffordia mitchelliae, Enteromorpha clathrata, Bryopsis plumosa* and *B. hypnoides*.

Thermal impact-related differences or shifts in the natural occurrence patterns described above (decreased occurrence of cold-water species resulting from an abbreviated season, or increased occurrence over an extended season for species with warm-water affinities) could be detected by comparing operational period frequencies (2-unit vs. 3-unit) at stations potentially exposed to the thermal plume. The only station where such shifts have been and continue to be evident is the NUSCO site nearest the discharge, FE. For example, two cold-water red algae, *Dumontia contorta* and *Polysiphonia urceolata*, were common components of the winter/spring flora at FE during 2-unit operation, occurring in 24% and 26%, respectively, of collections there prior to Unit 3 start-up, but neither has been collected at FE during 3-unit operation (Table 2). Other cold-water species occurred occasionally at FE during

TABLE 1. Qualitative algal collections (Mar. 1979-Feb. 1994) by month, during 2-unit (3/79-2/86) and 3-unit (3/86-2/94) operating periods. Values represent number of times found, as a percentage of possible times found. A dash before a species indicates that is was included in collections made in the latest report year. Taxa enclosed in quotes are, or may be, conspecific or subspecific forms, or alternate life-history stages; see text for additional explanation.

	2-Unit Operation												3-Unit Operation											
Rhodophyta	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Stylonema alsidii	5	2	2	2	0	4	9	12	21	33	7	5	3	5	1	1	0	1	3	8	10	10	3	4
-Erythrotrichopeltis ciliaris	32	18	18	12	16	11	11	18	32	49	32	25	21	21	11	15	11	6	8	18	29	42	22	25
-Erythrotrichia carnea	4	4	0	0	2	2	0	0	9	4	4	2	1	5	4	10	1	0	4	4	13	14	6	7
Erythrocladia subintegra	0	0	2	0	0	2	0	2	2	5	5	5	1	0	0	1	0	0	0	3	1	3	3	0
Erythropeltis discigera	7	4	2	0	4	0	2	2	7	5	5	5	1	3	4	1	1	0	1	0	4	8	7	4
-Bangia atropurpurea	65	79	77	86	26	11	4	7	18	21	35	54	58	73	96	76	38	17	6	6	31	31	39	49
-Porphyra leucosticta	46	68	61	65	44	26	12	9	9	19	18	25	71	78	88	82	72	31	10	7	7	32	29	46
-Porphyra umbilicalis	46	53	77	77	93	72	58	40	25	14	23	28	54	57	71	90	85	58	43	35	29	36	44	60
-Porphyra linearis	0	0	0	0	0	0	0	0	0	0	0	2	3	2	3	0	1	0	0	0	0	0	0	3
Porphyropsis coccinea	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
Audouinella purpurea	4	4	0	4	2	2	2	4	2	0	5	2	0	0	0	0	1	0	0	0	0	0	0	0
-Audouinella secundata	35	53	37	35	35	40	25	23	21	37	18	19	26	30	25	31	22	15	15	10	10	13	8	4
Audouinella daviesii	9	0	2	7	4	4	5	2	2	2	7	0	0	0	3	1	3	0	4	3	0	4	1	0
-Audouinella saviana	16	5	14	12	9	11	5	5	19	18	7	11	6	5	8	21	15	7	3	6	17	15	15	8
Audouinella sp.	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
Audouinella dasyae	0	0	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	1
-Gelidium pusillum	7	9	4	9	7	9	11	9	9	14	12	12	28	22	22	22	21	18	22	24	24	28	35	33
-Nemalion helminthoides	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	11	3	0	0	0	0
-Bonnemaisonia hamifera	9	9	18	18	26	53	33	9	0	0	5	5	1	6	6	14	28	44	38	11	3	1	0	4
'Trailliella intricata'	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
-Agardhiella subulata	5	5	5	4	11	14	12	12	18	11	12	12	13	13	10	4	4	10	13	13	14	14	17	18
-Polyides rotundus	5	11	0	7	11	4	11	12	14	14	19	12	6	3	3	1	3	6	7	7	4	0	3	4
-Cystoclonium purpureum	81	77	68	74	79	79	46	23	12	47	56	68	69	68	57	58	67	68	26	13	18	22	44	50
-Gracilaria tikvahiae	0	0	0	0	0	0	0	2	2	0	0	0	10	6	4	3	1	0	0	4	6	4	8	7
-Ahnfeltia plicata	47	51	49	49	42	44	37	37	37	46	47	56	26	22	18	14	18	24	18	17	14	14	18	26
-Phyllophora pseudoceranoides	23	14	16	16	11	11	12	12	4	19	16	16	13	11	1	3	6	7	4	4	6	7	6	8
Phyllophora traillii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
-Coccotylus truncatus	12	18	7	5	9	12	12	9	11	11	16	19	6	6	14	7	7	8	3	4	3	3	6	3
-Chondrus crispus	96	96	98	98	98	98	98	98	96	96	96	96	99	98	97	97	97	97	99	99	99	97	99	99
-Mastocarpus stellatus	74	56	53	47	56	58	60	56	58	65	77	65	65	70	64	68	67	69	65	65	67	61	68	68
Petrocelis middendorfii	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	1	0	0	0	0	0
Rhodophysema georgii	0	2	7	2	9	2	2	4	4	0	0	0	0	0	3	0	6	3	0	0	0	0	0	0
-Corallina officinalis	60	61	58	56	51	51	61	63	60	68	58	67	71	76	58	63	64	69	60	68	61	63	65	76
-Dumontia contorta	46	65	81	82	81	47	5	2	2	0	2	7	33	49	68	76	74	32	8	1	0	0	0	8
Gloisiphonia capillaris	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	3	1	0	0	0	0	0	0
-Choreocolax polysiphoniae	9	12	12	9	9	2	4	4	2	5	2	9	13	11	10	8	6	7	8	3	6	10	4	6
Hildenbrandia rubra	4	2	2	2	0	0	2	0	2	5	0	2	1	0	0	0	0	4	1	0	3	1	0	1
-Palmaria palmata	32	33	49	44	39	46	33	28	21	12	28	30	21	27	22	19	22	26	32	24	15	14	10	18
-Champia parvula	35	21	11	7	4	4	35	65	74	79	65	46	15	6	8	4	4	7	50	75	81	67	49	39
-Lomentaria baileyana	4	0	0	0	0	0	5	30	49	28	7	2	0	0	0	0	0	1	6	21	28	13	3	0
-Lomentaria clavellosa	11	5	9	16	7	2	2	4	7	4	7	4	6	5	7	8	6	0	1	1	0	0	1	1
-Lomentaria orcadensis	2	2	4	0	0	0	0	0	2	5	0	0	1	2	0	0	0	0	0	0	1	0	0	3
-Antithamnion cruciatum	47	25	5	18	7	16	46	63	70	74	74	67	21	16	8	10	15	21	43	49	26	29	28	15
-Antithamnion pectinatum	0	0	0	0	0	0	0	0	0	0	0	0	76	71	39	25	32	39	53	65	78	71	82	83
Callithamnion corymbosum	0	0	0	0	0	2	0	7	9	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0
-Callithamnion roseum	7	2	2	0	0	0	9	18	35	18	14	5	0	0	3	1	3	4	6	18	21	8	8	4
-Callithamnion tetragonum	65	46	23	33	21	11	25	26	44	46	72	53	11	17	15	15	8	3	4	7	8	11	10	14
-Callithamnion byssoides	0	0	0	0	0	0	4	4	2	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0
-'Callithamnion baileyi'	0	0	0	0	0	0	0	0	0	5	7	2	40	29	19	13	8	13	26	25	36	43	56	39
-Ceramium deslongchampii	4	4	0	2	0	2	0	4	4	4	4	9	3	2	0	1	3	0	0	0	1	3	6	1
-Ceramium diaphanum	7	0	0	2	0	0	25	68	49	51	11	12	0	0	0	0	1	6	35	58	56	33	13	1
-Ceramium rubrum	88	88	74	81	89	91	95	84	95	93	91	88	90	92	76	81	83	90	92	92	90	86	88	89
Ceramium fastigiatum	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Spermothamnion repens	54	33	28	26	18	28	28	35	40	72	74	70	47	29	26	22	24	44	38	51	38	43	43	53
-Spyridia filamentosa	0	0	0	0	0	2	2	4	12	4	2	2	0	0	1	0	0	0	0	4	10	10	6	3
Scagelia pylaisaei	5	2	2	4	5	0	0	2	2	4	4	4	0	0	4	3	3	1	1	3	0	0	0	1
-Griffithsia globulifera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	3	1	0

TABLE 1. (cont.)

Rhodophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Grinnellia americanum	4	0	0	2	2	0	4	9	2	11	12	7	4	0	1	0	0	3	7	8	15	11	15	6
-Phycodrys rubens	2	4	5	12	7	7	4	5	5	7	7	0	0	0	1	7	6	7	3	1	1	3	1	4
-Dasya baillouviana	7	2	0	0	0	0	7	39	30	25	23	11	7	2	1	1	0	3	13	36	32	39	31	28
-Chondria sedifolia	0	0	0	0	0	0	0	4	5	5	0	0	0	0	0	0	0	0	4	1	4	1	0	0
-Chondria baileyana	2	0	2	0	2	0	0	4	16	11	0	5	1	0	0	0	0	0	4	7	13	6	3	1
-Chondria tenuissima	0	0	0	2	0	0	4	2	4	0	0	0	0	0	0	0	0	0	3	4	4	1	0	0
Chondria dasyphylia	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Polysiphonia denudata	0	0	0	0	0	0	5	0	4	9	5	7	3	2	1	0	4	3	1	4	0	1	0	3
-Polysiphonia harveyi	70	39	25	23	19	53	82	95	91	65	68	63	15	8	6	7	10	28	32	35	33	19	25	15
-Polysiphonia lanosa	82	74	68	60	70	63	61	60	65	72	65	74	74	70	71	65	69	76	67	65	68	67	74	75
-Polysiphonia nigra	5	9	7	11	18	16	2	4	2	4	4	7	1	5	8	8	18	13	0	0	0	3	1	3
-Polysiphonia nigrescens	19	14	18	32	19	28	23	16	18	25	23	21	14	22	6	14	17	24	18	14	18	11	14	18
-Polysiphonia urceolata	19	16	30	54	60	32	18	4	4	2	2	4	13	21	21	22	35	24	4	4	3	3	1	10
-Polysiphonia elongata	0	0	0	0	0	2	0	2	0	0	0	4	1	0	0	0	0	1	0	0	0	1	0	0
-Polysiphonia fibrillosa	5	2	0	0	0	0	2	0	0	2	0	12	3	0	0	0	0	0	0	0	1	4	0	3
-Polysiphonia flexicaulis	0	0	0	0	0	0	0	0	0	0	0	2	11	2	0	0	0	0	0	4	1	7	3	4
-Polysiphonia novae-angliae	70	61	42	30	39	44	68	74	74	88	86	84	93	87	57	40	38	75	92	97	99	97	99	99
Rhodomeia confervoides	7	9	30	19	9	4	0	2	2	0	2	2	1	3	3	6	3	0	0	0	0	0	0	0

Phaeophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Ectocarpus fasciculatus	7	18	12	26	33	35	25	16	28	32	19	9	1	3	14	18	32	36	15	14	24	21	28	11
-Ectocarpus siliculosis	19	32	47	53	60	70	60	39	28	26	23	14	17	32	35	50	44	44	33	32	21	19	28	17
-Ectocarpus sp.	5	12	9	7	0	5	7	4	7	2	5	4	3	0	3	1	0	1	1	1	0	0	0	1
-Giffordia granulosa	5	2	4	4	4	7	2	0	5	2	5	4	1	3	1	4	8	14	3	3	1	1	8	4
-Giffordia mitchelliae	4	7	2	5	14	19	19	32	32	28	14	7	8	8	6	6	14	11	13	31	39	43	21	11
-Pilayella littoralis	21	18	25	35	51	32	12	16	14	21	12	18	11	16	21	26	36	35	7	10	3	21	8	17
-Spongonema tomentosum	18	28	42	30	19	0	2	4	0	7	2	2	4	22	40	24	13	3	1	4	0	6	0	4
Entonema aecidioides	0	0	0	4	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Acinetospora sp.	0	0	0	4	2	0	0	2	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0
Feldmannia sp.	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0
-Ralfsia verrucosa	42	56	37	39	46	47	58	65	68	67	54	46	68	65	53	60	40	44	67	74	76	63	72	65
-Elachista fucicola	42	51	61	72	86	82	70	81	77	49	35	26	57	59	71	81	76	88	89	81	69	61	63	54
Halothrix lumbricalis	0	2	0	0	5	4	2	0	2	2	0	0	0	0	3	3	6	1	1	0	0	0	0	0
-Leathesia difformis	0	2	0	9	19	26	26	11	0	0	0	0	0	0	0	15	36	50	39	4	0	0	0	0
-Chordaria flagelliformis	0	0	0	5	19	49	37	30	19	5	2	4	0	0	1	3	14	26	25	15	8	6	1	0
Sphaerotrichia divaricata	0	0	0	2	5	11	2	0	0	0	0	0	0	0	1	0	3	0	0	0	0	0	0	0
Cladosiphon zosterae	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Eudesme virescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
-Pogotrichum filiforme	0	14	5	2	5	0	0	0	2	0	0	0	0	3	4	3	0	0	0	0	0	0	0	0
-Desmotrichum undulatum	0	2	14	7	4	2	0	0	0	4	0	4	0	8	6	7	6	4	1	0	0	0	0	0
-Phaeosaccion collinsii	0	4	2	0	0	0	0	0	0	0	0	0	0	2	4	0	0	0	0	0	0	1	0	0
-Punctaria latifolia	2	9	16	12	4	9	0	0	0	5	4	4	3	5	10	4	7	6	1	1	0	0	3	6
-Punctaria plantaginea	2	4	4	2	5	7	7	5	4	0	2	0	0	3	0	3	6	3	3	1	1	1	3	3
-Petalonia fascia	70	84	68	86	84	84	72	9	5	12	42	63	69	90	86	97	90	81	47	7	1	6	40	57
-Scytosiphon lomentaria	46	79	93	95	93	95	86	18	5	9	21	32	28	70	92	99	93	90	64	8	0	1	8	18
Delamarea attenuata	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Desmarestia aculeata	7	2	9	16	9	12	2	7	4	7	11	4	7	8	4	11	13	7	17	4	6	8	6	4
-Desmarestia viridis	2	4	30	44	49	39	2	0	0	0	2	0	0	5	22	49	57	33	0	1	0	0	0	0
-Chorda filum	0	0	0	2	9	19	16	0	0	0	0	0	0	0	0	1	7	25	17	7	0	0	0	0
-Chorda tomentosa	0	0	5	18	28	2	9	0	0	0	0	0	0	5	15	29	19	7	0	0	0	0	0	0
Laminaria digitata	2	0	0	0	4	0	0	2	0	0	0	4	0	2	1	4	0	0	0	0	1	1	0	0
-Laminaria longicruris	9	12	9	14	14	11	12	11	12	18	12	7	11	13	15	15	18	17	18	25	17	22	15	11
-Laminaria saccharina	53	37	53	63	82	77	82	75	60	58	49	58	54	38	57	71	88	96	79	75	64	65	57	51
-Sphacelaria cirrosa	30	16	9	12	16	16	19	21	21	32	37	39	38	27	21	19	28	25	26	22	35	39	38	43
Sphacelaria rigidula	0	0	0	0	0	0	0	0	2	0	0	0	0	2	0	0	1	0	0	1	0	1	3	0
-Ascophyllum nodosum	96	96	98	98	98	98	98	98	96	96	96	96	96	95	96	96	96	96	96	96	96	96	96	96
-Fucus distichus s edentatus	7	11	18	21	16	4	2	2	0	2	5	4	6	8	11	14	3	6	0	4	6	1	0	3
-Fucus distichus s evanescens	14	12	21	19	23	7	2	5	0	4	9	0	3	8	11	13	6	3	4	4	1	3	1	1
-Fucus spiralis	2	2	2	9	5	11	7	7	14	7	7	4	11	5	6	10	11	14	11	10	15	11	8	10
-Fucus vesiculosus	96	96	98	98	98	100	100	100	96	96	96	96	100	100	100	100	100	100	100	100	100	100	100	100
-Sargassum filipendula	0	0	0	0	0	0	0	2	0	0	0	0	8	8	7	7	7	7	7	8	8	8	8	8

TABLE 1. (cont.)

	2-Unit Operation												3-Unit Operation											
Chlorophyta	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Ulothrix flacca	53	61	74	70	47	7	5	0	2	9	23	32	44	70	71	67	26	7	6	1	7	7	17	26
-Urospora penicilliformis	61	72	74	70	30	7	2	4	2	7	28	44	60	57	65	47	24	7	8	8	7	14	31	50
-Urospora wormskjoldii	9	7	9	16	2	7	4	0	0	0	0	2	8	6	19	8	10	7	8	6	4	8	7	17
'Urospora collabens'	7	19	9	2	5	2	2	0	0	0	0	4	1	2	1	0	3	0	1	0	0	0	0	0
Acrochaete viridis	2	2	0	2	0	0	0	0	0	2	2	2	0	0	0	0	1	1	1	0	0	0	0	0
-Monostroma grevillei	25	58	54	60	51	12	2	2	2	0	7	5	6	51	54	75	49	1	0	0	0	0	0	3
-Monostroma pulchrum	19	44	88	91	86	16	4	0	0	4	2	2	1	24	78	79	74	4	0	0	0	0	0	0
-Monostroma oxysperma	2	0	0	2	2	0	0	0	0	0	0	0	0	2	4	7	6	1	0	0	0	0	0	0
-Spongomorpha arcta	7	18	39	51	54	32	5	0	0	0	4	4	4	17	24	29	47	24	4	0	0	0	0	1
-Spongomorpha aeruginosa	4	2	7	14	18	16	5	4	4	0	0	0	1	0	1	22	35	28	3	0	0	1	0	0
'Codiolum gregarium'	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Capsosiphon fulvescens	0	2	0	5	4	4	2	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Capsosiphon groenlandicum	2	12	12	14	11	7	2	0	0	0	2	11	3	2	1	1	1	1	0	0	1	0	0	1
-Blidingia minima	58	44	44	53	70	65	47	67	58	49	47	53	64	78	63	76	79	82	75	81	79	64	72	65
Blidingia marginata	9	4	0	0	2	0	4	0	0	4	2	2	0	0	0	0	1	1	3	0	0	0	0	0
-Enteromorpha clathrata	4	5	2	12	18	19	37	46	37	28	5	4	3	0	3	4	7	8	24	33	36	15	3	1
-Enteromorpha flexuosa	33	33	25	28	37	40	40	32	46	60	53	35	53	51	61	43	49	36	44	53	71	60	65	65
-Enteromorpha intestinalis	25	26	35	44	49	47	46	51	35	25	21	16	15	17	22	38	29	39	46	35	26	28	10	14
-Enteromorpha linza	51	30	32	56	67	63	65	60	60	70	63	49	50	49	56	76	78	76	74	69	79	74	68	60
-Enteromorpha prolifera	42	39	33	28	35	37	25	33	32	47	44	47	13	14	6	15	19	18	8	14	13	24	15	21
-Enteromorpha torta	2	0	0	0	4	7	5	7	2	5	2	0	0	0	0	0	1	4	3	1	0	0	0	0
-Enteromorpha ralfsii	0	0	0	0	0	11	7	4	5	2	2	0	0	2	1	0	1	6	6	6	3	3	0	0
Percursaria percursa	2	0	0	5	4	9	4	2	2	2	0	0	0	0	1	0	0	0	0	0	1	1	0	0
-Ulva lactuca	96	88	77	84	89	98	93	91	96	98	96	96	94	94	76	83	89	97	96	99	100	96	96	94
-Prasiola stipitata	19	21	23	23	23	23	28	25	25	25	19	30	38	35	26	29	32	33	35	31	32	29	31	32
-Chaetomorpha linum	79	58	40	37	56	89	91	95	96	95	84	74	54	33	17	21	46	71	90	86	86	78	72	54
Chaetomorpha melagonium	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
-Chaetomorpha aerea	30	25	26	21	19	26	37	28	42	30	37	30	47	46	28	40	49	42	51	51	40	46	47	46
-Cladophora albida	0	0	2	2	11	9	16	11	12	4	4	2	0	0	0	3	0	3	1	6	4	1	0	0
-'Cladophora flexuosa'	14	2	4	7	14	25	37	26	18	28	12	11	10	2	1	4	22	54	75	39	40	31	13	10
'Cladophora glaucescens'	0	0	0	0	2	2	2	2	0	0	0	0	0	0	0	1	0	0	0	1	1	0	0	0
Cladophora laetevirens	0	0	0	2	2	4	4	0	4	2	0	0	0	0	0	0	0	0	1	0	0	0	0	0
-'Cladophora refracta'	9	7	2	2	5	32	33	26	28	37	18	16	3	5	4	1	4	11	3	3	3	3	1	7
-Cladophora sericea	12	5	5	25	53	42	35	37	23	21	18	14	10	2	14	22	39	31	39	36	28	25	17	28
'Cladophora crystallina'	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	4	3	0	0	0
-Cladophora hutchinsiae	2	4	4	2	9	9	7	12	12	14	11	2	3	3	1	3	4	11	8	7	6	4	4	1
-Cladophora rupestris	0	2	0	2	5	2	9	7	0	2	2	0	1	2	1	0	8	10	7	6	3	1	3	0
-Cladophora ruchingeri	0	0	0	0	0	0	0	2	0	0	0	0	0	0	1	0	6	8	14	24	13	14	10	0
-Rhizoclonium riparium	9	30	25	16	18	18	30	30	28	23	12	16	7	0	8	4	10	26	26	25	13	14	3	7
'Rhizoclonium kerneri'	2	0	4	0	0	4	0	0	0	0	0	2	0	0	1	0	1	1	3	1	0	0	0	0
-'Rhizoclonium tortuosum'	0	0	0	0	0	5	0	0	0	0	2	0	0	0	0	0	0	1	0	0	0	0	0	0
-Bryopsis plumosa	7	0	0	0	4	2	12	16	11	12	7	7	7	3	0	0	0	3	11	6	11	3	6	7
-Bryopsis hypnoides	0	0	0	0	2	5	9	2	5	7	2	4	3	0	1	0	7	15	21	19	6	10	11	6
Derbesia marina	7	5	4	9	0	0	2	4	4	9	12	11	0	0	0	0	1	0	1	1	1	4	0	1
-Codium fragile	89	79	68	75	75	82	81	84	88	95	89	86	93	92	76	75	83	90	96	96	96	92	86	90

TABLE 2. Qualitative algal collections (Mar. 1979-Feb. 1994) by station, during 2-unit (3/79-2/86) and 3-unit (3/86-2/94) operating periods. Values represent number of times found, as a percentage of possible times found. Taxa enclosed in quotes are, or may be, conspecific or subspecific forms, or alternate life-history stages; see text for additional explanation. The last three columns represent 2-unit, 3-unit, and overall study summaries ('T'=present, <1%).

	2-Unit Operation									3-Unit Operation									Summaries		
Rhodophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot
Stylonema alsidii	14	11	6	2	12	11	10	1	6	12	3	0	1	11	2	8	0	0	8	4	6
Erythrotrichopeltis ciliaris	32	20	19	23	26	20	27	11	23	37	15	7	11	43	17	26	3	13	23	19	21
Erythrotrichia carnea	4	2	0	2	1	4	4	4	0	11	1	5	2	16	9	4	1	2	2	6	4
Erythrocladia subintegra	1	4	4	2	4	0	0	1	2	0	0	2	0	7	0	0	0	0	2	1	1
Erythropeltis discigera	4	6	6	0	5	2	1	2	5	0	6	2	2	8	0	2	3	3	4	3	3
Bangia atropurpurea	48	45	54	35	42	30	49	38	25	49	52	48	40	42	34	48	43	28	40	43	42
Porphyra leucosticta	42	27	56	44	32	23	37	31	24	49	42	61	46	51	48	39	41	33	33	46	40
Porphyra umbilicalis	64	42	50	54	54	42	52	64	33	39	51	72	51	74	48	51	75	38	50	55	53
Porphyra linearis	0	0	0	0	0	0	1	0	0	0	0	4	1	0	0	1	2	0	T	1	1
Porphyropsis coccinea	0	0	0	0	0	2	0	0	0	0	0	1	0	0	1	0	0	0	T	T	T
Audouinella purpurea	2	0	0	0	14	0	1	0	1	0	0	1	0	0	0	0	0	0	2	T	1
Audouinella secundata	39	49	44	23	24	24	30	27	25	21	28	24	7	19	9	9	21	16	31	17	24
Audouinella daviesii	5	4	2	6	2	2	4	4	4	2	2	1	0	2	1	2	2	2	4	2	2
Audouinella saviana	17	12	19	15	12	13	8	2	6	18	8	4	6	17	12	19	2	8	11	11	11
Audouinella sp.	1	1	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	T	T	T
Audouinella dasyae	0	2	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0	0	T	T	T
Gelidium pusillum	7	0	0	4	1	63	1	0	0	75	0	1	0	33	100	12	3	1	9	25	18
Nemalion helminthoides	0	0	0	0	0	0	0	0	0	0	6	4	2	0	0	0	0	0	0	1	1
Bonnemaisonia hamifera	1	20	2	33	0	1	20	29	33	1	3	2	34	0	1	16	23	38	15	13	14
'Trailliella intricata'	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	T	T	T
Agardhiella subulata	5	7	8	2	24	8	17	4	12	0	2	2	0	81	8	5	2	4	10	12	11
Polyides rotundus	4	8	8	8	5	8	18	2	26	1	0	4	8	6	0	4	1	9	10	4	7
Cystoclonium purpureum	58	50	67	65	58	58	71	49	62	61	25	53	69	6	35	56	55	59	59	47	52
Gracilaria tikvahiae	0	1	0	0	1	0	0	0	0	0	0	2	1	35	1	1	0	0	T	4	3
Ahnfeltia plicata	20	39	58	92	73	19	52	24	55	8	7	38	41	1	6	34	16	20	45	19	31
Phyllophora pseudoceranoides	8	7	10	29	10	8	40	7	11	5	1	5	11	3	1	18	3	8	14	6	10
Phyllophora traillii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	T	T
Coccotylus truncatus	11	11	13	10	11	11	21	5	13	3	2	4	17	1	1	14	3	6	12	6	8
Chondrus crispus	100	100	100	100	79	100	100	100	100	100	100	100	100	82	100	100	100	100	97	98	98
Mastocarpus stellatus	25	61	90	100	17	21	65	96	98	27	40	96	100	0	43	100	96	96	60	66	64
Petrocelis middendorfii	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	T	T	T
Rhodophysema georgii	0	1	0	0	0	2	4	1	12	0	1	0	1	0	1	0	1	4	2	1	2
Corallina officinalis	2	100	100	31	95	70	82	32	27	1	98	99	29	95	100	95	59	19	60	66	63
Dumontia contorta	39	19	40	48	26	45	33	32	39	39	9	31	51	0	41	25	20	45	35	29	32
Gloisiphonia capillaris	0	0	2	0	1	0	0	0	0	0	0	0	0	1	0	2	0	0	T	T	T
Choreocolax polysiphoniae	14	21	4	2	2	0	5	0	6	12	25	13	3	0	0	2	0	13	6	7	7
Hildenbrandia rubra	1	0	0	15	0	1	1	0	1	0	0	0	7	0	0	1	0	1	2	1	1
Palmaria palmata	36	39	17	69	8	8	44	25	58	15	14	9	62	2	3	22	17	43	33	21	26
Champia parvula	37	35	15	33	31	38	61	26	46	31	27	29	35	39	35	44	24	42	37	34	35
Lomentaria baileyana	21	7	2	6	17	13	17	2	2	12	0	0	1	13	15	9	1	3	10	6	8
Lomentaria clavellosa	5	5	0	10	2	5	10	7	12	1	0	4	3	0	0	11	1	7	6	3	4
Lomentaria orcadensis	1	0	0	8	0	0	2	0	1	0	0	0	1	0	0	2	0	2	1	1	1
Antithamnion cruciatum	44	52	33	40	33	40	62	30	43	19	24	26	25	23	21	32	17	24	43	24	32
Antithamnion pectinatum	0	0	0	0	0	0	0	0	0	40	68	84	65	68	55	61	47	45	0	59	33
Callithamnion corymbosum	0	7	0	2	2	1	5	1	1	0	0	0	0	0	0	0	0	0	2	0	1
Callithamnion roseum	10	6	10	4	18	11	10	4	8	3	2	4	6	29	4	5	1	2	9	6	8
Callithamnion tetragonum	38	37	52	63	50	25	45	23	31	2	21	21	16	1	8	15	0	8	39	10	23
Callithamnion byssoides	1	5	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	1	T	T
'Callithamnion baileyi'	2	2	2	2	0	0	1	0	1	28	49	46	40	2	22	37	16	19	1	29	17
Ceramium deslongchampii	13	1	0	4	0	4	1	1	0	2	4	1	1	0	0	3	2	1	3	2	2
Ceramium diaphanum	18	29	2	33	8	13	26	19	19	8	21	4	28	4	7	29	25	25	19	17	18
Ceramium rubrum	88	95	88	92	87	77	94	87	86	86	99	77	91	78	87	97	85	86	88	87	88
Ceramium fastigiatum	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Spermothamnion repens	46	58	40	46	21	27	68	26	48	47	52	24	31	9	28	63	42	47	42	38	40
Spyridia filamentosa	12	1	0	0	0	4	1	0	0	16	3	1	1	0	1	2	0	1	2	3	3
Scagelia pylaisaei	1	4	6	2	0	1	6	1	4	0	2	0	1	1	1	1	4	2	3	1	2
Griffithsia globulifera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	2	0	1	T

TABLE 2. (cont.)

	2-Unit Operation									3-Unit Operation									Summaries		
Rhodophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot
Grinnellia americanum	1	1	2	2	1	1	13	2	12	0	0	0	3	23	3	8	3	13	4	6	5
Phycodrys rubens	0	0	2	8	1	2	18	6	11	0	1	0	8	0	0	9	1	6	5	3	4
Dasya baillouviana	8	14	2	10	11	21	18	4	13	23	12	11	7	47	15	21	3	6	12	16	14
Chondria sedifolia	5	1	0	2	0	1	0	0	1	4	0	0	0	0	3	1	0	0	1	1	1
Chondria baileyana	5	5	13	4	0	4	4	0	1	9	2	3	1	1	7	1	0	1	3	3	3
Chondria tenuissima	5	0	0	0	1	1	0	0	0	2	1	0	0	1	5	0	0	0	1	1	1
Chondria dasyphylla	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Polysiphonia denudata	0	4	0	2	5	2	6	1	1	0	4	1	1	6	0	2	1	1	2	2	2
Polysiphonia harveyi	58	63	50	50	70	54	60	51	57	9	19	26	18	36	20	18	13	17	58	20	37
Polysiphonia lanosa	86	86	100	44	46	27	81	50	94	81	99	100	37	0	38	98	80	98	68	70	69
Polysiphonia nigra	1	17	0	2	4	7	12	6	11	9	4	1	2	5	1	15	5	2	7	5	6
Polysiphonia nigrescens	19	23	4	15	15	20	55	10	20	26	21	3	4	12	4	49	6	15	21	16	18
Polysiphonia urceolata	42	20	23	27	24	5	29	4	13	26	19	23	8	0	0	34	5	3	20	13	16
Polysiphonia elongata	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	1	0	2	1	T	T
Polysiphonia fibrillosa	5	1	0	2	4	0	2	1	1	0	2	1	1	1	0	0	3	0	2	1	1
Polysiphonia flexicaulis	0	0	2	0	0	0	0	0	0	2	7	1	2	1	0	5	5	0	T	3	2
Polysiphonia novae-angliae	60	62	73	58	70	60	68	57	64	80	84	81	77	99	77	86	71	74	63	81	73
Rhodomela confervoides	14	8	2	10	5	2	6	5	10	1	2	0	2	0	0	1	2	3	7	1	4

	2-Unit Operation									3-Unit Operation									Summaries		
Phaeophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot
Ectocarpus fasciculatus	17	20	48	33	27	10	20	20	15	17	28	12	40	4	6	22	19	16	22	18	20
Ectocarpus siliculosis	49	38	25	50	44	37	45	35	29	48	29	26	36	24	23	44	29	18	39	31	35
Ectocarpus sp.	2	6	17	13	2	4	2	4	8	2	2	2	1	0	0	0	1	1	6	1	3
Giffordia granulosa	2	4	8	6	5	2	2	1	4	4	2	2	12	9	1	7	2	0	4	4	4
Giffordia mitcheliiae	18	13	6	8	32	15	25	6	6	22	19	12	16	55	11	21	2	1	15	18	17
Pilayella littoralis	65	7	6	21	6	58	20	7	6	40	4	2	24	4	54	18	4	7	23	18	20
Spongonema tomentosum	17	20	10	17	12	5	11	13	11	13	7	9	12	12	6	13	12	6	13	10	11
Entonema aecidioides	0	2	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	T	0	T
Acinetospora sp.	0	1	0	0	0	2	0	1	0	0	0	0	0	0	1	2	0	0	1	T	T
Feldmannia sp.	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Ralfsia verrucosa	76	58	42	15	61	60	68	27	42	72	81	65	67	11	73	80	60	52	52	62	58
Elachista fucicola	70	62	69	58	61	45	61	73	54	77	78	68	73	60	61	75	81	64	61	71	66
Halothrix lumbricalis	1	4	0	0	0	0	4	0	2	2	3	2	3	0	0	0	0	0	1	1	1
Leathesia difformis	12	1	27	4	14	1	12	1	4	25	9	26	14	4	11	17	2	1	8	12	10
Chordaria flagelliformis	15	23	27	15	5	2	27	12	7	4	31	17	11	2	0	8	2	1	14	8	11
Sphaerotrichia divaricata	0	1	2	4	0	0	4	4	1	0	1	1	1	0	0	0	0	0	2	T	1
Cladosiphon zosterae	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Eudesme virescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	T	T
Pogotrichum filiforme	4	1	4	4	0	2	5	1	1	0	1	0	2	1	1	1	0	1	2	1	1
Desmotrichum undulatum	7	6	0	0	0	2	5	0	4	4	3	0	7	5	1	1	0	1	3	3	3
Phaeosaccion collinsii	1	0	0	0	0	1	1	0	0	0	0	1	1	0	0	1	1	1	T	1	1
Punctaria latifolia	5	5	2	10	2	11	5	5	4	2	4	1	5	4	5	4	4	3	5	4	4
Punctaria plantaginea	5	4	4	0	0	8	7	0	1	9	2	0	1	0	6	1	0	0	3	2	3
Petalonia fascia	54	74	63	46	54	56	70	55	38	60	67	59	65	45	58	61	49	36	57	56	56
Scytosiphon lomentaria	64	74	48	48	49	64	60	49	40	52	58	53	47	38	56	52	41	31	56	47	51
Delamarea attenuata	0	0	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Desmarestia aculeata	2	1	4	13	5	10	17	5	11	11	1	2	22	0	0	15	9	11	7	8	8
Desmarestia viridis	12	17	17	25	8	11	17	8	19	13	7	13	26	6	11	19	13	19	14	14	14
Chorda filum	2	1	0	13	1	1	11	4	4	1	2	2	14	0	0	12	5	7	4	5	4
Chorda tomentosa	0	4	2	23	2	4	6	4	8	3	4	2	27	0	1	5	6	7	5	6	6
Laminaria digitata	0	0	0	13	0	0	0	0	0	0	0	0	6	0	0	0	0	1	1	1	1
Laminaria longicruris	11	4	13	42	1	1	19	14	14	6	0	8	63	0	6	24	19	21	12	16	14
Laminaria saccharina	62	69	63	96	60	50	61	62	54	69	79	72	88	39	52	74	74	53	62	67	65
Sphacelaria cirrosa	49	25	10	6	49	27	18	0	4	67	23	7	0	75	56	28	9	4	22	30	27
Sphacelaria rigidula	0	0	0	0	0	0	1	0	0	2	0	1	0	0	2	0	1	0	T	1	T
Ascophyllum nodosum	100	100	100	100	79	100	100	100	100	100	100	100	100	62	100	100	100	100	97	96	96
Fucus distichus s edentatus	6	7	13	23	11	6	2	7	1	2	5	8	17	1	0	3	5	3	7	5	6
Fucus distichus s evanescens	11	12	2	23	6	6	8	10	12	2	11	7	15	0	0	2	3	3	10	5	7
Fucus spiralis	5	33	8	0	0	1	4	2	1	7	47	31	3	0	0	0	0	3	6	10	8
Fucus vesiculosus	100	100	100	100	82	100	100	100	100	100	100	100	100	100	100	100	100	100	98	100	99
Sargassum filipendula	0	1	0	0	0	0	0	0	0	0	0	0	0	69	0	0	0	0	T	8	4

TABLE 2. (cont.)

	2-Unit Operation									3-Unit Operation									Summaries		
Chlorophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot
Ulothrix flacca	37	37	27	17	32	29	45	31	24	38	31	29	33	16	28	32	26	25	32	29	30
Urospora penicilliformis	38	42	40	33	31	21	43	31	24	35	44	36	27	29	32	32	31	16	33	31	32
Urospora wormskjoldii	5	8	6	0	7	4	7	2	0	12	7	8	3	23	8	14	3	3	5	9	7
'Urospora collabens'	6	6	6	4	4	1	2	2	6	0	3	0	0	2	0	1	0	0	4	1	2
Acrochaete viridis	0	2	6	0	0	1	0	0	0	1	1	0	0	0	0	1	0	0	1	T	1
Monostroma grevillei	26	23	19	19	23	32	23	15	25	21	21	19	26	1	19	19	27	22	23	20	21
Monostroma pulchrum	32	33	25	29	24	23	33	33	31	21	20	23	31	4	23	21	26	25	30	22	25
Monostroma oxysperma	0	1	0	0	0	0	0	0	2	0	1	1	1	0	0	1	2	8	T	2	1
Spongomorpha arcta	29	27	31	25	18	4	18	10	7	20	12	31	18	5	6	13	6	2	18	13	15
Spongomorpha aeruginosa	10	5	15	0	6	6	8	5	1	9	16	18	6	1	2	6	5	5	6	8	7
'Codiolum gregarium'	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	T	0	T
Capsosiphon fulvescens	1	1	0	0	0	2	6	0	2	0	0	0	0	0	0	0	0	0	2	0	1
Capsosiphon groenlandicum	6	20	8	2	2	2	5	2	5	0	1	0	2	1	2	1	1	1	6	1	3
Blidingia minima	61	55	71	71	70	8	64	43	62	66	67	87	84	82	51	77	61	82	55	73	65
Blidingia marginata	1	2	0	2	4	1	4	1	2	3	1	0	0	0	0	0	0	0	2	T	1
Enteromorpha clathrata	27	4	10	4	25	33	33	1	14	16	13	5	0	14	19	21	3	14	18	12	14
Enteromorpha flexuosa	38	40	35	27	63	37	54	12	33	63	52	57	44	84	43	61	35	49	38	54	47
Enteromorpha intestinalis	52	39	40	19	32	27	55	15	30	36	35	27	16	19	21	39	16	32	35	27	30
Enteromorpha linza	50	74	75	40	71	35	67	48	42	71	86	87	58	84	60	74	52	37	55	68	62
Enteromorpha prolifera	42	37	35	40	35	35	54	15	40	21	12	11	9	14	16	21	11	21	37	15	25
Enteromorpha torta	5	1	2	0	1	6	6	0	2	2	0	0	1	1	1	2	0	0	3	1	2
Enteromorpha ralfsii	4	1	0	0	4	8	4	0	0	4	0	1	2	1	6	2	2	1	2	2	2
Percursaria percursa	4	1	0	0	1	[illegible]	4	0	1	0	1	0	0	0	0	2	0	0	2	T	1
Ulva lactuca	96	90	96	94	96	89	98	86	86	94	97	95	91	91	92	96	94	88	92	93	93
Prasiola stipitata	51	1	4	90	1	1	0	75	8	64	0	0	93	1	0	1	79	48	24	32	28
Chaetomorpha linum	56	81	88	79	69	68	75	79	85	60	60	58	72	14	53	65	77	76	75	59	66
Chaetomorpha melagonium	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	T	T	T
Chaetomorpha aerea	23	26	29	2	74	40	45	2	10	55	37	67	8	72	53	62	6	40	29	44	38
Cladophora albida	5	4	4	0	6	8	14	2	6	2	1	1	0	2	4	3	0	0	6	2	3
'Cladophora flexuosa'	8	26	31	17	13	11	19	17	12	20	32	28	17	18	27	38	22	25	16	25	21
'Cladophora glaucescens'	0	0	2	0	0	0	4	0	0	0	0	0	0	0	2	1	0	0	1	T	T
Cladophora laetevirens	0	0	2	0	0	1	2	1	5	0	0	0	0	0	0	0	0	1	1	T	1
'Cladophora refracta'	8	37	23	8	18	12	24	17	12	1	5	5	2	5	5	5	5	1	18	4	10
Cladophora sericea	24	20	13	2	40	32	40	11	20	34	19	18	8	28	46	35	5	25	24	24	24
'Cladophora crystallina'	0	0	0	0	0	0	0	2	0	1	0	0	0	1	1	2	0	0	T	1	T
Cladophora hutchinsiae	5	1	10	2	13	7	12	2	11	8	5	8	1	4	5	3	2	4	7	5	6
Cladophora rupestris	2	2	2	4	2	4	4	0	2	1	2	3	2	9	4	4	2	3	2	4	3
Cladophora ruchingeri	0	0	0	0	1	0	0	0	0	3	4	6	1	19	18	14	1	1	T	7	4
Rhizoclonium riparium	29	19	8	6	19	49	29	10	10	29	8	12	3	4	19	20	9	3	21	12	16
'Rhizoclonium kerneri'	2	1	0	0	0	1	0	1	1	3	0	0	0	0	1	1	1	0	1	1	1
'Rhizoclonium tortuosum'	1	0	0	0	2	0	1	0	0	1	0	0	0	0	0	0	0	0	1	T	T
Bryopsis plumosa	7	5	0	6	19	4	5	6	4	2	2	2	5	22	2	3	0	3	6	5	5
Bryopsis hypnoides	2	4	4	0	4	6	1	1	4	11	5	2	7	19	8	12	4	6	3	8	6
Derbesia marina	6	2	0	2	25	7	0	0	2	1	0	0	0	4	0	1	0	2	5	1	3
Codium fragile	81	85	88	100	100	87	81	64	69	92	82	87	87	100	97	97	76	81	83	89	86

the 3-unit period, but much less frequently than during the 2-unit period. These include *Ulothrix flacca, Monostroma grevillei, M. pulchrum* and *Spongomorpha arcta*. By contrast, a number of warm-water seasonal species have become more common at FE during 3-unit operation. Among these species are *Callithamnion roseum, Grinnellia americanum, Dasya baillouviana, Giffordia mitchelliae*, and *Bryopsis hypnoides*.

Distribution of perennial species at FE has also changed during 3-unit operation. This study has documented the establishment of populations of species with geographical distributions which extend into warm temperate and tropical regions, and are therefore tolerant of 3-unit temperature regimes at FE (e.g., *Gracilaria tikvahiae, Agardhiella subulata* and *Sargassum filipendula*; Taylor 1957; Luning 1990). Similarly, some species near the southern limit of their normal geographical ranges, such as *Mastocarpus stellatus* and *Polysiphonia lanosa*, experienced population elimination at FE during 3-unit operation.

Some other changes to the overall flora occurred during the 3-unit period that are unrelated to power plant operation. Increased occurrence of *Gelidium pusillum* has been observed at FE and several sites farther from the discharge, including a pronounced increase (from 7% to 75%) at our control site GN. The area-wide introduction of an exotic species *Antithamnion pectinatum* (*A. nipponicum* in previous reports), also not considered a power plant impact, occurred during 3-unit operation as well.

Relationships among composite collections made during 2-unit and 3-unit periods at each station were also examined using cluster analysis techniques and illustrated by the resulting clustering dendrogram (Figure 3a). In short, the addition of 1993 data has resulted in no substantive change in major groupings, relative to recent previous years (e.g., NUSCO 1992, 1994). Excluding the FE 3-unit collection, primary groupings of all other collections are distinguished by floras which develop on differing substrata. Collection areas at sites comprising Group I are composed of bedrock ledge, with subgroups separated into exposed sites (WP, MP, BP and FE 2-unit; Subgroup Ia) and sheltered sites (GN and FS; Subgroup Ib). Collection substrata at Group II sites (TT, SS and SE) are primarily large boulders and relatively unstable cobble with few horizontal surfaces. The strong dissimilarity of the FE 3-unit collection (Group III) to all other collections is due to the unique floristic assemblage that has developed at this site in response to exposure to the 3-unit thermal plume.

A similar analysis was applied to annual collections at FE (Fig. 3b) to illustrate yearly changes to the algal community brought about by important power plant operational events. Group I represents collections made during 2-unit 1-cut operation, when the unimpacted flora at FE was similar to that observed at other exposed sites (see Fig. 3a and NUSCO 1987). Temperature conditions were altered substantially due to operational changes occurring in 1984 (2-unit, 2-cut operation) and 1986 (Unit 3 start-up), and these events were reflected in the characteristic disturbed or early successional stage flora collected at FE from 1984 to 1987 (Group II). Elevated temperature conditions persisted, but were more consistent in subsequent years comprising collections in Group III. These conditions allowed for more long-term development of the unique flora now observed at FE, characterized by shifts in temporal and spatial species occurrence patterns described above. Similar floristic shifts have been observed by other researchers studying attached algae near thermal effluents (Vadas et al. 1976; Wilce et al. 1978; Schneider 1981).

Abundance Measurement

Power plant impact may also change certain patterns of organismal distribution and community dominance hierarchy on nearby rocky shores. These changes may not be evident when using descriptors of qualitative community characteristics, presented in the previous section, which effectively illustrate impacts such as temporal and spatial elimination or introduction of species. Therefore, quantification of species abundance and distribution patterns, through determination of percent substratum coverage, complements the qualitative studies and has become an integral part of the overall rocky intertidal monitoring program.

The objective of these quantitative studies was to sample organism abundance over an area sufficiently large as to accurately describe large-scale patterns of abundance at each sampling site, to relate these patterns to site-specific physical and

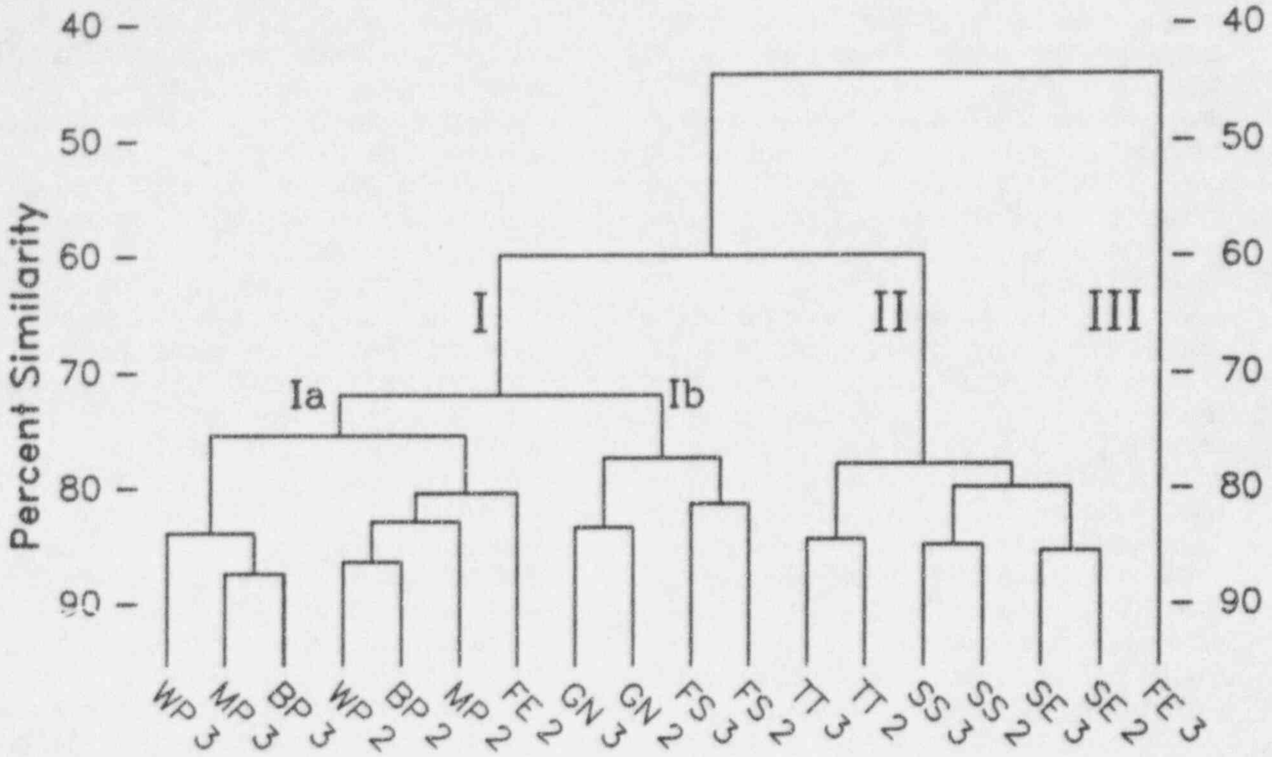


Fig. 3a. Clustering dendrogram of percent similarity of qualitative algal collections, by station and operational period (2-unit - 3-unit).

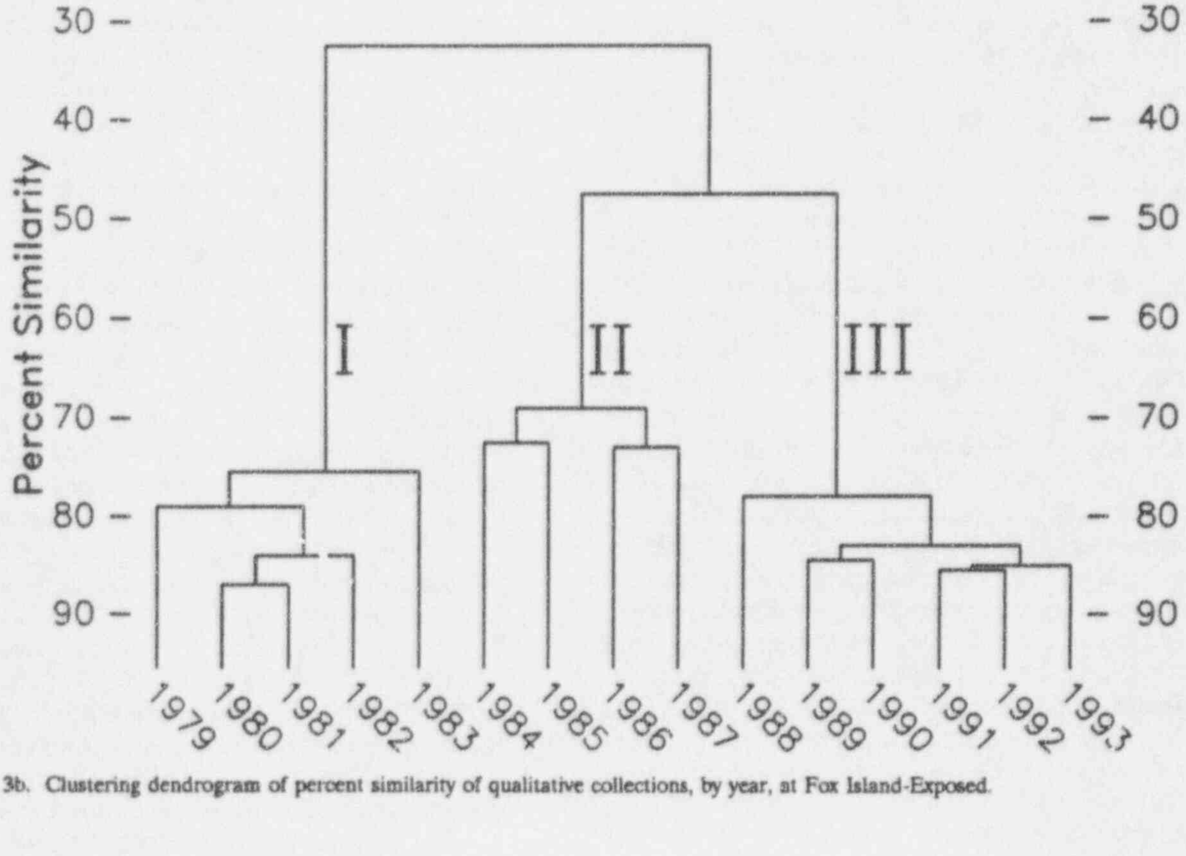


Fig. 3b. Clustering dendrogram of percent similarity of qualitative collections, by year, at Fox Island-Exposed.

biological controlling mechanisms and, of primary concern, to determine if any of these mechanisms result from, or are influenced by, the operation of MNPS. Subsections describing abundance patterns of important intertidal organisms, i.e., barnacles, *Fucus*, *Chondrus* and common epiphytes, along with analyses of overall community structure, are included below.

Barnacles

Intertidal barnacles (primarily *Semibalanus balanoides*) are the dominant sessile invertebrates on local rocky shores and are most abundant in the mid intertidal (Zone 2). Barnacle abundance also varies seasonally through an annual cycle of reproduction and settlement in early spring, rapid growth and increases in surface cover in summer. Barnacle cover decreases through autumn and winter due to competition for space, predation and physical disturbance (Connell 1961; Menge 1976; Bertness 1989; NUSCO 1993).

In the Millstone area, this annual cycle of barnacle abundance was apparent at all sites, in all three intertidal zones in 1993-94 (Fig. 4). Maximum barnacle cover in the high intertidal (Zone 1) during 1993-94 ranged from 6% (FS and GN) to 47% (MP). Minimum coverage in Zone 1 ranged from 1% at FS and GN to 9% at MP. In the mid intertidal (Zone 2), maximum barnacle cover was lowest at FE (21%) and highest at MP (87%); minimum cover was lowest at FE (2%) and highest at GN (35%). Low intertidal (Zone 3) maxima during 1993-94 ranged from <1% (FE) to 40% (FS). The annual minimum in Zone 3 was lowest at FE (0%), and highest at FS (13%).

Barnacle abundance patterns at NUSCO study sites (excluding FE) have been remarkably consistent over the study period, including the present sampling year, and reflect the temporal stability of environmental conditions at these sites. Spatial variability in these conditions is considerable, however; natural site-specific factors modify annual barnacle cycles and patterns of zonation. Degree of site exposure to wind and waves and slope of available substratum appear to be the most important controlling mechanisms, and their effects on local barnacle population dynamics are described in detail in NUSCO (1993).

The effect of periodic thermal plume incursion is superimposed on these natural controlling mechanisms at FE, and has resulted in significant impacts to the shore biota there, including barnacles. Due to the influence of tides, these impacts are most notable in the low intertidal (Zone 3). Zone 3 barnacles are exposed to elevated discharge temperatures for 9-10 hours each tidal cycle during 3-unit operation, whereas barnacles in Zones 1 and 2 experience a tidally-induced refuge from maximum 3-unit thermal plume incursion, as they are exposed to air during most of this time. These conditions have directly and indirectly modified the pattern of barnacle abundance in Zone 3 at FE. Elevated temperatures directly impacted low intertidal barnacles by causing complete population mortality in late summer every year since Unit 3 start-up, including the present study year. Reduced early summer maxima, or the lack of any substantial recruitment in Zone 3, particularly since 1989, is not considered a direct thermal effect of 3-unit conditions, but rather a shift in community dominance patterns. The development and persistence of an extensive low intertidal *Codium fragile* population at FE competitively excludes barnacles through preemption of habitat space (Underwood and Denley 1984; NUSCO 1993).

Fucus

The brown alga *Fucus vesiculosus* is a codominant in the mid intertidal zone with barnacles, and is locally the most common intertidal species of macroalgae, occurring in all three intertidal zones. Other species of *Fucus* included in our abundance estimates are found occasionally at our study sites, but contribute relatively little in terms of percent substratum coverage. These include *F. distichus* subsp. *edentatus*, *F. distichus* subsp. *evanescens* (both occur mostly subtidally) and *F. spiralis*, which occurs in the high intertidal.

Temporal and zonal *Fucus* abundance patterns in the Millstone area, presented in Figure 5, are similar to those reported elsewhere in New England (Lubchenco 1980, 1983; Topinka et al. 1981). At most NUSCO study sites, *Fucus* abundance typically peaks annually in late summer or autumn, reflecting high recruitment and growth rates prior to and during this period. Abundance maximum during 1993-94 in Zone 1 was highest at FE (27%), with the low maxima of 2% or less at

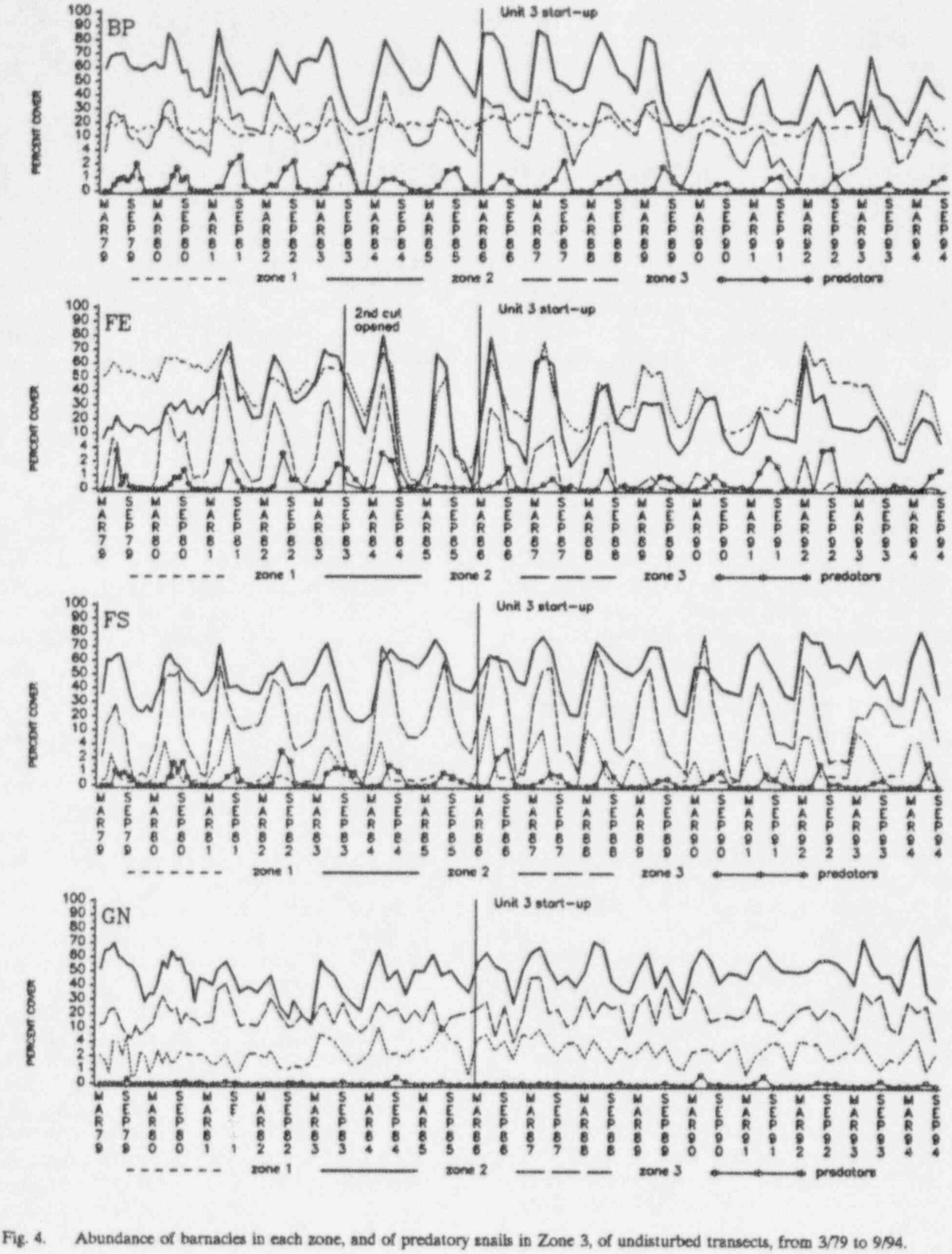


Fig. 4. Abundance of barnacles in each zone, and of predatory snails in Zone 3, of undisturbed transects, from 3/79 to 9/94.

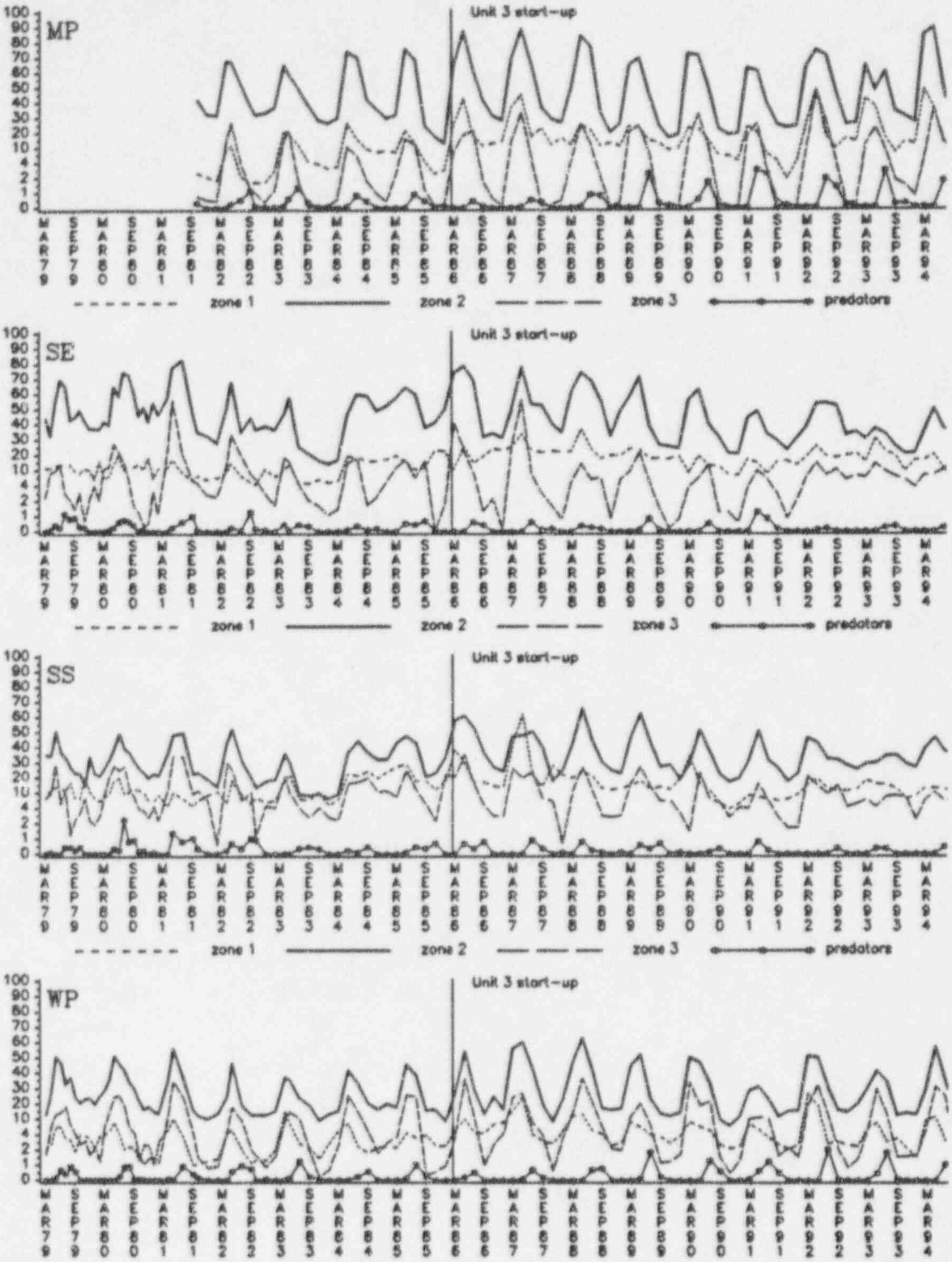

MP
SE
SS
WP
Unit 3 start-up
100
90
80
70
60
50
40
30
20
10
4
2
1
0
MAR79
SEP79
MAR80
SEP80
MAR81
SEP81
MAR82
SEP82
MAR83
SEP83
MAR84
SEP84
MAR85
SEP85
MAR86
SEP86
MAR87
SEP87
MAR88
SEP88
MAR89
SEP89
MAR90
SEP90
MAR91
SEP91
MAR92
SEP92
MAR93
SEP93
MAR94
zone 1
zone 2
zone 3
predators

Fig. 4. (cont.)

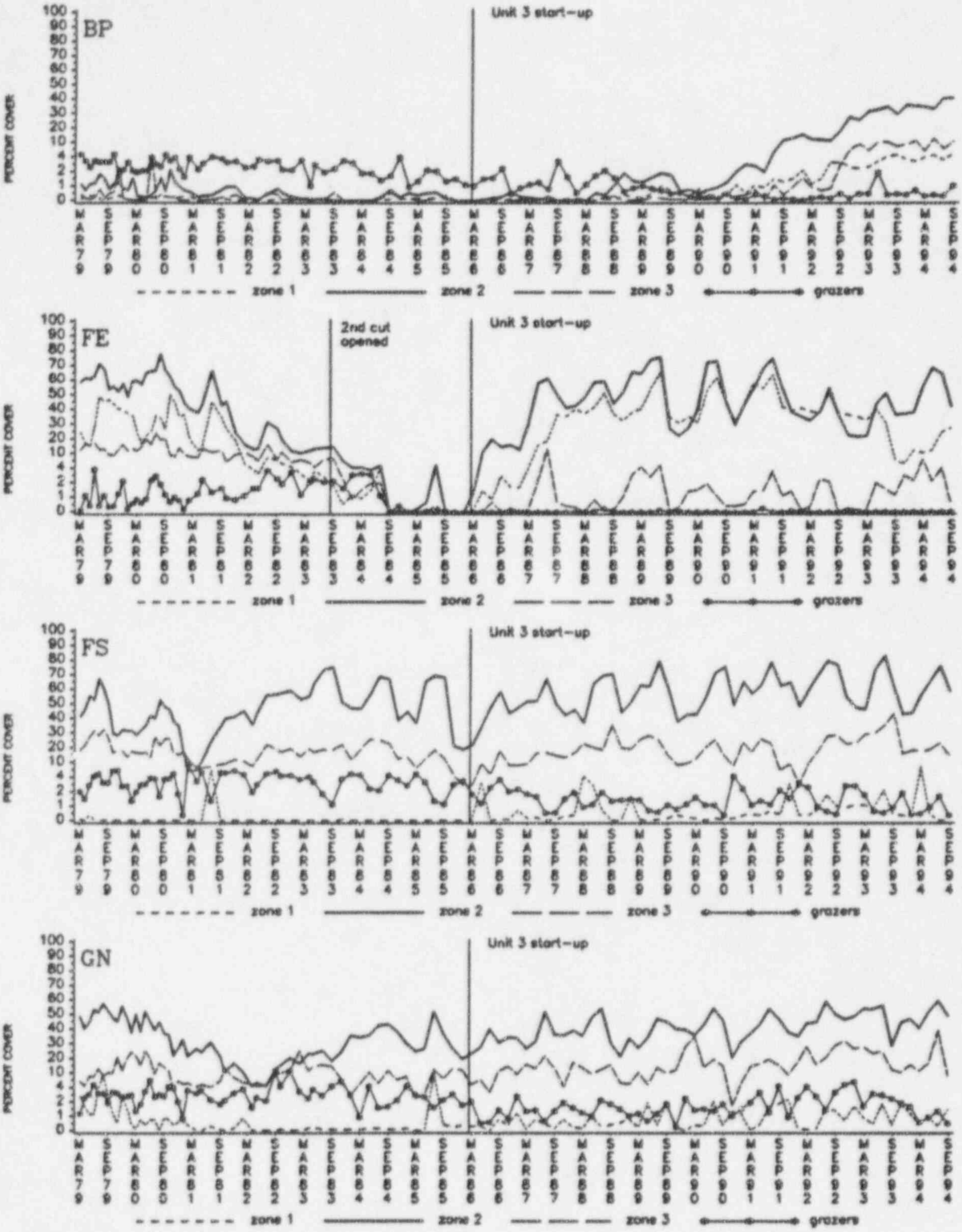


Fig. 5. Abundance of *Fucus* in each zone, and of grazing snails in Zone 3, of undisturbed transects, from 3/79 to 9/94.

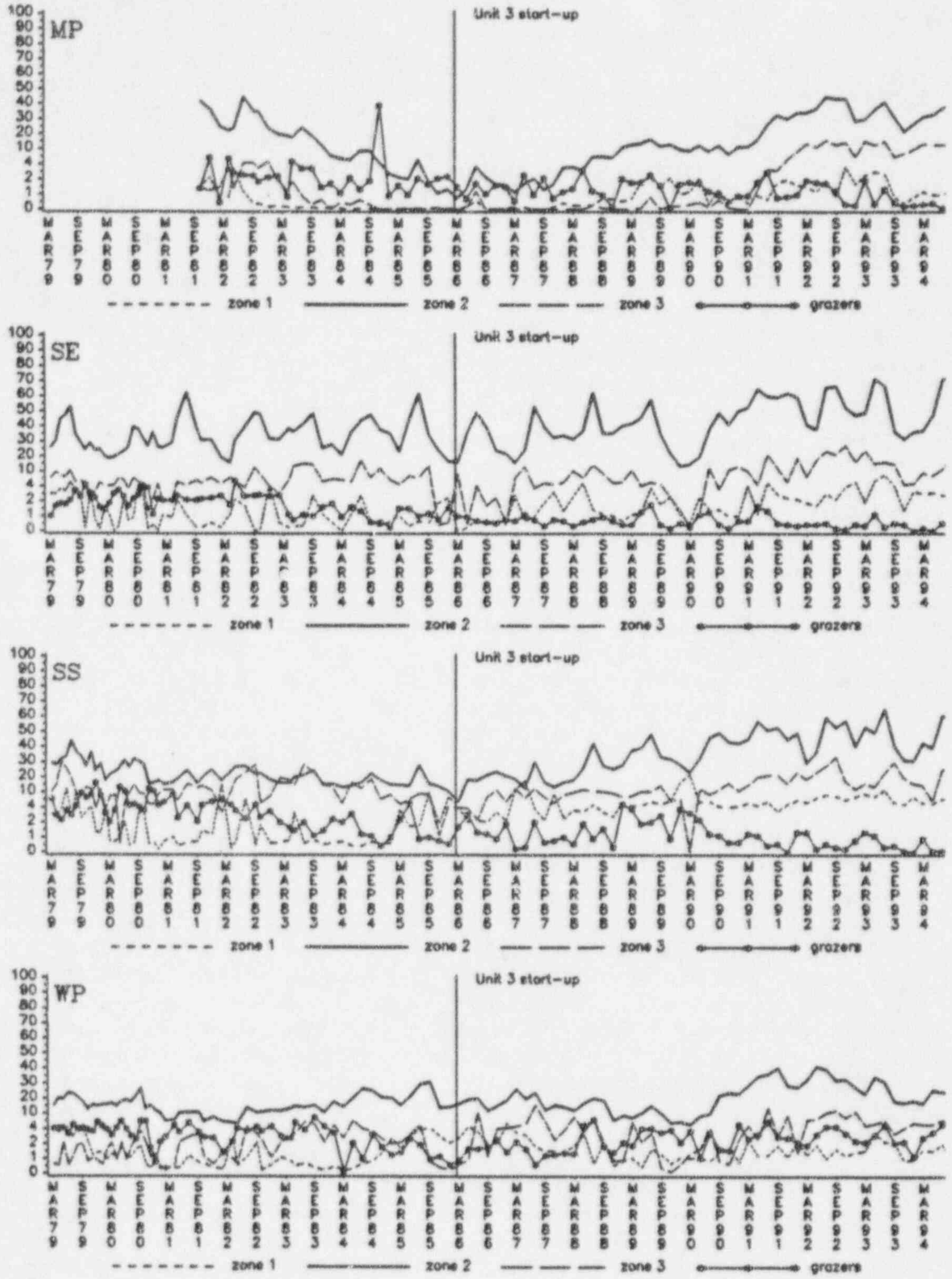

MP
SE
SS
WP
Unit 3 start-up
100
90
80
70
60
50
40
30
20
10
4
2
1
0
MAR79
SEP79
MAR80
SEP80
MAR81
SEP81
MAR82
SEP82
MAR83
SEP83
MAR84
SEP84
MAR85
SEP85
MAR86
SEP86
MAR87
SEP87
MAR88
SEP88
MAR89
SEP89
MAR90
SEP90
MAR91
SEP91
MAR92
SEP92
MAR93
SEP93
MAR94
zone 1
zone 2
zone 3
grazers

Fig. 5. (cont.)

GN, MP and WP (Fig. 5). Highest Zone 2 cover during 1993-94 occurred at FS (75%); maximum cover was lowest at WP (27%). In Zone 3, maximum *Fucus* cover during 1993-94 reached a high of 40% at GN; the low peak abundance values occurred at FE and WP (8%).

Spatial variability in *Fucus* abundance patterns exhibited by local populations is a reflection of, and can be explained by, environmental conditions unique to each site. In general, *Fucus* is most abundant on moderately exposed shores, common environments of most of our study sites. *Fucus* abundance is limited at highly exposed sites by physical stress from wave shock, while at sheltered sites these species are often outcompeted for space by another fucoid, *Ascophyllum nodosum* (Schonbeck and Norton 1978, 1980; Keser and Larson 1984). Vertical distribution patterns of intertidal *Fucus* are also controlled by the degree of wave exposure, as well as slope of available substratum. More detailed description of the role these natural characteristics play in determining *Fucus* zonation patterns at each study site is provided in previous reports (NUSCO 1992, 1993).

In addition to these natural site-specific characteristics, physical stress in the form of heat from the MNPS discharge is an important mechanism controlling *Fucus* abundance in Zone 3 at FE. Elevated temperatures during periods of thermal plume incursion have resulted in virtual elimination of *Fucus* in Zone 3 each year since the opening of the second quarry cut in 1983 and throughout 3-unit operation (Fig. 5). Thermal stress was most severe at FE in Zone 3, because organisms there were submerged and exposed to elevated temperatures for much of the tidal cycle. After Unit 3 went on line, thermal stress at mid and upper intertidal levels was substantially reduced due to increased discharge velocity, and *Fucus* populations in Zones 1 and 2 returned to abundance levels similar to those observed from 1979 to 1983.

Several long-term fluctuations in *Fucus* abundance have been observed at other study sites, located farther from the discharge than FE, which we cannot directly relate to either natural or impact-related factors. Previous reports (e.g., NUSCO 1992, 1993, 1994) have described a protracted decline/recovery cycle at MP that continues to the present report year (Fig. 5). Because of the proximity of this site to the MNPS discharge (ca. 250 m to the east), the possibility of a power plant impact exists. Indeed, physical data, in the form of water temperatures measured by continuous recorders deployed at MP, indicate some thermal plume incursion (water temperatures 2-3°C above ambient during slack tides with one to three units operating). However, direct evidence linking the pattern of *Fucus* abundance at MP to power plant impact is absent, since the present *Fucus* population at FE recovered relatively rapidly after Unit 3 start-up and currently endures much greater temperature extremes. Elevated temperatures may have indirectly impacted *Fucus* recruitment at MP, by enhancement of earlier seasonal migration and higher feeding rates of grazers (mostly *Littorina littorea*; cf. Newell et al. 1971; NUSCO 1993). In any case, *Fucus* now appears to be well established at MP under 3-unit operating conditions with a luxuriant cover in Zones 2 and 3 observed during 1993-94. Similar establishment of a substantial *Fucus* population at BP has been noted in recent years, and may be part of an even longer abundance cycle than that noted at MP, and beginning before 1979; little *Fucus* was observed at this site prior to 1991 (Fig. 5). Some less dramatic increasing trends have been observed at other study sites in recent years (e.g., SE, SS and WP), perhaps indicating an area-wide trend unrelated to power plant operation.

Chondrus and common epiphytes

Another dominant intertidal species in the Millstone area is the perennial red alga, *Chondrus crispus*. Local populations of this species form extensive low intertidal turfs on all but the most sheltered rocky shores, and can effectively exclude other species, including *Fucus*. Several seasonally abundant algal taxa coexist with *Chondrus* in Zone 3 (e.g., *Monostroma* spp. and *Polysiphonia* spp.) by attaching to the upright portion of the *Chondrus* plant itself (as epiphytes) instead of competing directly for primary space. Owing to the importance of these algal populations to low intertidal community structure, and the susceptibility of low intertidal populations to power plant impacts (as mentioned in previous sections), documentation of abundance patterns of *Chondrus* and its associated epiphytes is therefore critical to our ecological monitoring program.

Perennial populations of *Chondrus* have

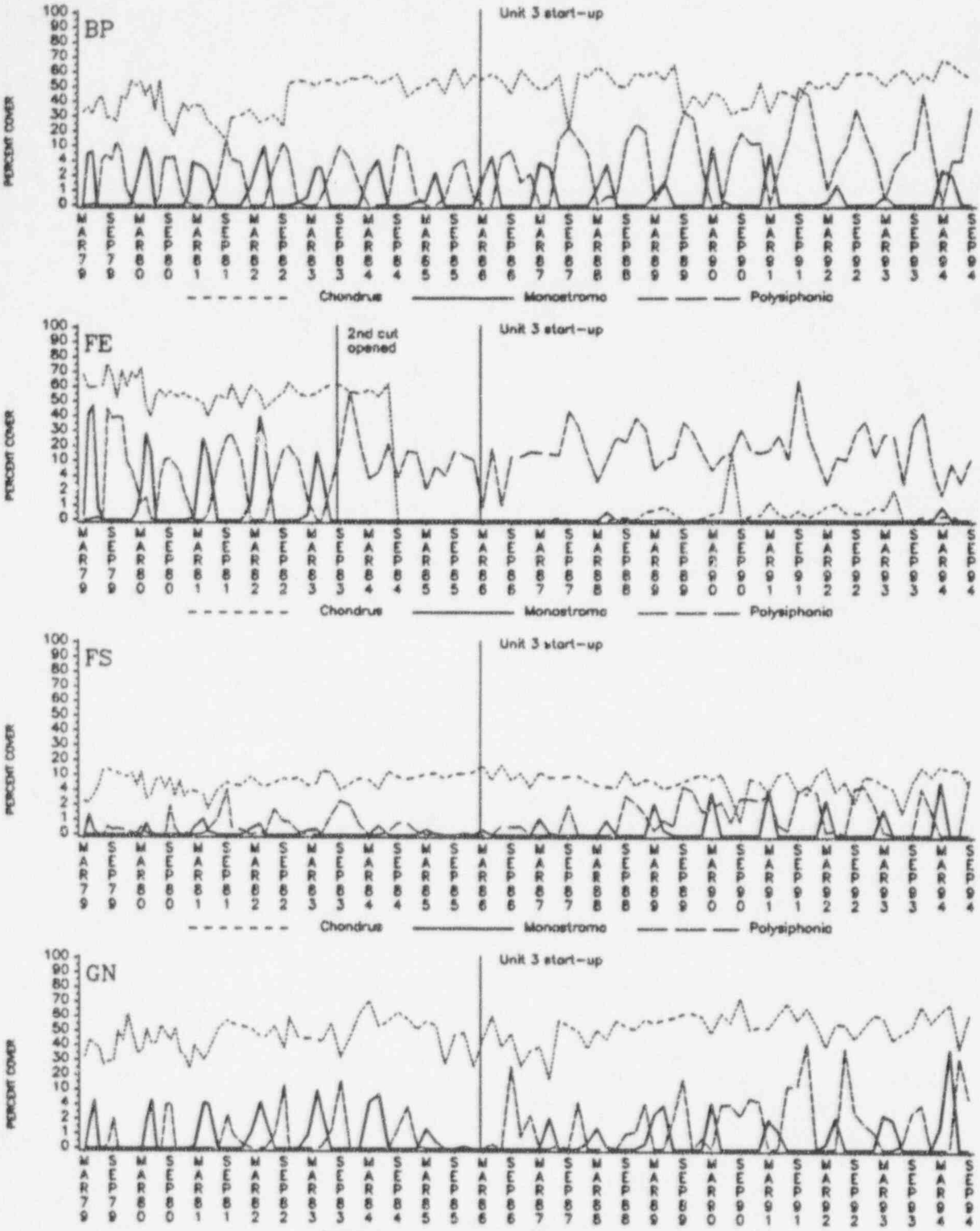
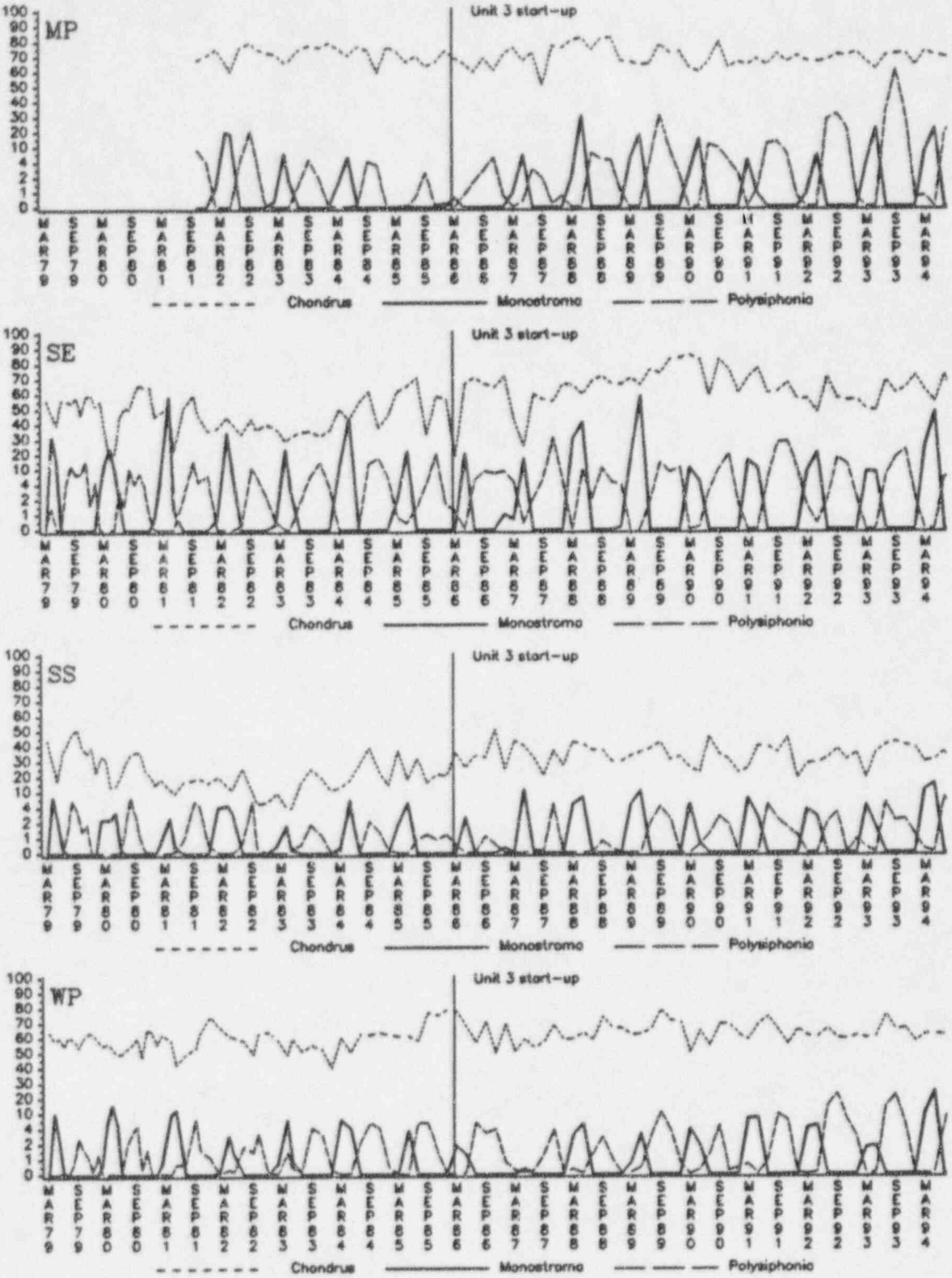


Fig. 6. Abundance of *Chondrus* and major epiphytes in Zone 3 of undisturbed transects, from 3/79 to 9/94.


MP
SE
SS
WP
Unit 3 start-up
100
90
80
70
60
50
40
30
20
10
4
2
1
0
MAR79
SEP79
MAR80
SEP80
MAR81
SEP81
MAR82
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MAR90
SEP90
MAR91
SEP91
MAR92
SEP92
MAR93
SEP93
MAR94
Chondrus
Monostroma
Polysiphonia

Fig. 6. (cont.)

maintained relatively stable abundance levels at 7 of the 8 study sites (all but FE) throughout the study period (Fig. 6). Abundance maxima at these sites during 1993-94 ranged from 16% at FS to approximately 65-70% at MP, SE and WP. In general, *Chondrus* abundances observed during 1993-94 at all sites except FE were similar to those observed throughout our studies.

Chondrus is now a relatively minor component of the low intertidal community at FE, with abundance estimates never exceeding 1% during 1993-94. Low *Chondrus* abundance has been typical for this site since 1984; estimates for that period have ranged from 0% to 14%, but have generally been <2%. Prior to 1984, abundance estimates were much higher (40-75%) and comparable to those at other exposed sites. This extensive population was eliminated in 1984 by elevated water temperatures from the 2-cut 2-unit discharge (NUSCO 1987). Since that time, only a few scattered plants have been observed in upper Zone 3 study quadrats during cooler months, and are typically eliminated each summer by elevated water temperatures from the 2-cut 3 unit discharge. This consistent scenario indicates that any successful reestablishment of the *Chondrus* population at FE is unlikely under current operating conditions. The low intertidal community at FE is now composed primarily of an extensive *Codium fragile* population, persistent populations of *Sargassum filipendula* and ephemeral algae including *Ulva lactuca*, *Enteromorpha* spp. and *Polysiphonia* spp. and a seasonally heavy set of the blue mussel, *Mytilus edulis*.

The presence of both warm-water and cold-water seasonal epiphytes in the local low intertidal zone provides the opportunity to document potential temporal shifts in abundance of these species in response to altered temperature regimes, such as those at FE. *Polysiphonia* spp., particularly *P. novae-angliae* and *P. harveyi*, are common warm-water epiphytes on *Chondrus*, *Ascophyllum* and *Codium*, and also grow attached to rock. The annual abundance cycle of these species is characterized by a late summer peak, with cover declining to near 0% by winter (Fig. 6.). Peak abundances during 1993-94 varied considerably from station to station, being lowest at FS and SS (8%) and highest at BP and FE (approximately 45%). Within-station variability from year to year was much less at all stations, except at FE, throughout the study period. Elevated temperature regimes at FE since the opening of the second quarry cut (1983) have produced favorable conditions for these species by extending the season of occurrence and increasing the levels of peak abundance. These temperature regimes at FE have also allowed *Polysiphonia* spp. to persist through cold water months, when these species are typically absent from other sites, including FE prior to 1983.

The annual abundance cycle of *Monostroma* spp. (*M. grevillei* and *M. pulchrum*) is nearly opposite to that described for *Polysiphonia* spp, i.e., peak abundance is observed during cold water months (late winter/early spring) and virtual absence is noted during warm-water months (July-December; Table 1, Fig. 6). For the first time since 1988, and only the second time since 1983, *Monostroma* spp. occurred at every NUSCO study site in 1994, including our impacted site, FE. Peak abundance levels occurred in the period March-May and ranged from 1% (FE) to 47% (SE). In general, *Monostroma* abundance levels during 1994 were high relative to previous years at each site. This area-wide trend was likely the result of colder than normal spring temperature conditions (see Lobster section of this report), which extended the period of suitable growing conditions for *Monostroma*. Colder spring temperatures even created an abbreviated period of these conditions at FE, which rarely occurred in previous years since 1983.

Community Analysis

Abundances of selected rocky intertidal taxa, represented as percent substratum coverage in undisturbed transects, were presented in previous sections. These species, and over one hundred others, comprise the rocky shore communities in the vicinity of Millstone Station. To permit comparisons among such communities, multivariate analyses are commonly used. For instance, a Bray-Curtis Similarity Index, using abundances of all taxa in Zones 1 and 2, was calculated for all sites over 2-unit and 3-unit operating periods, and for Fox Island-Exposed, for each year. The resulting similarity matrices, with multiple pair-wise comparisons, were illustrated as dendrograms, using a clustering algorithm (Figs. 7a and 7b).

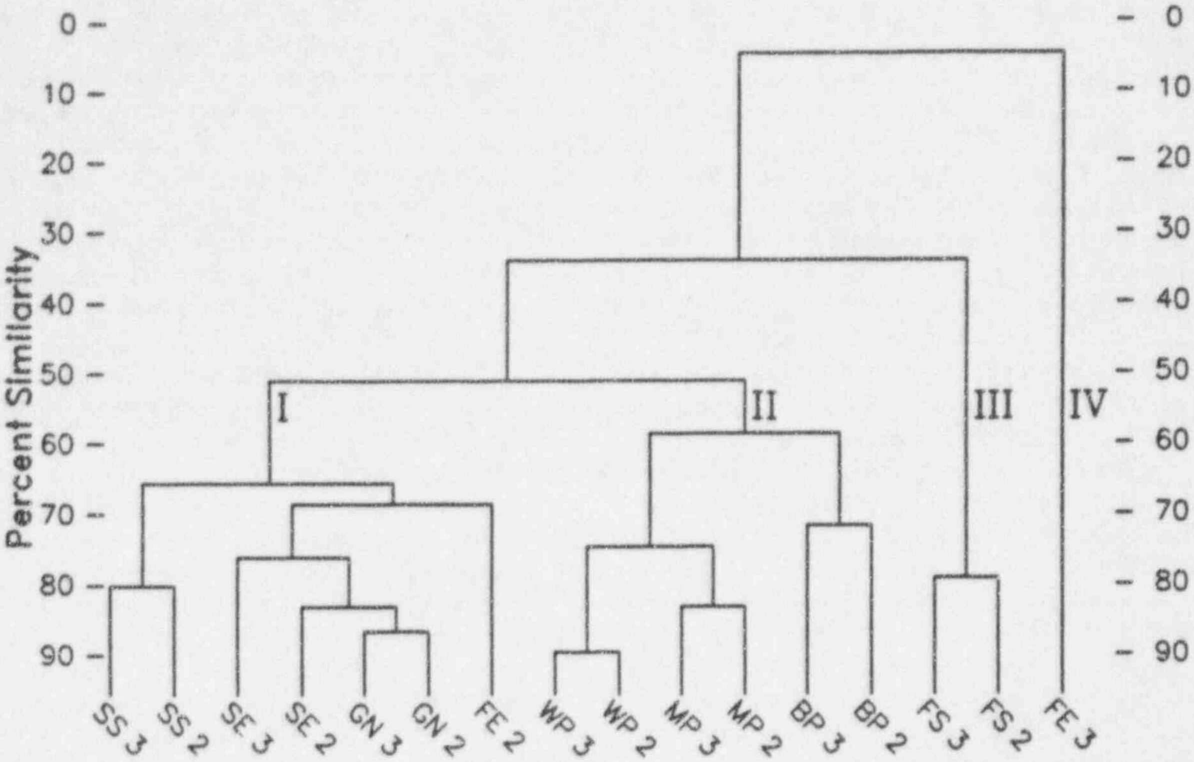

Fig. 7a. Clustering dendrogram of percent similarity of undisturbed communities, by station and operational period (2-unit - 3-unit).

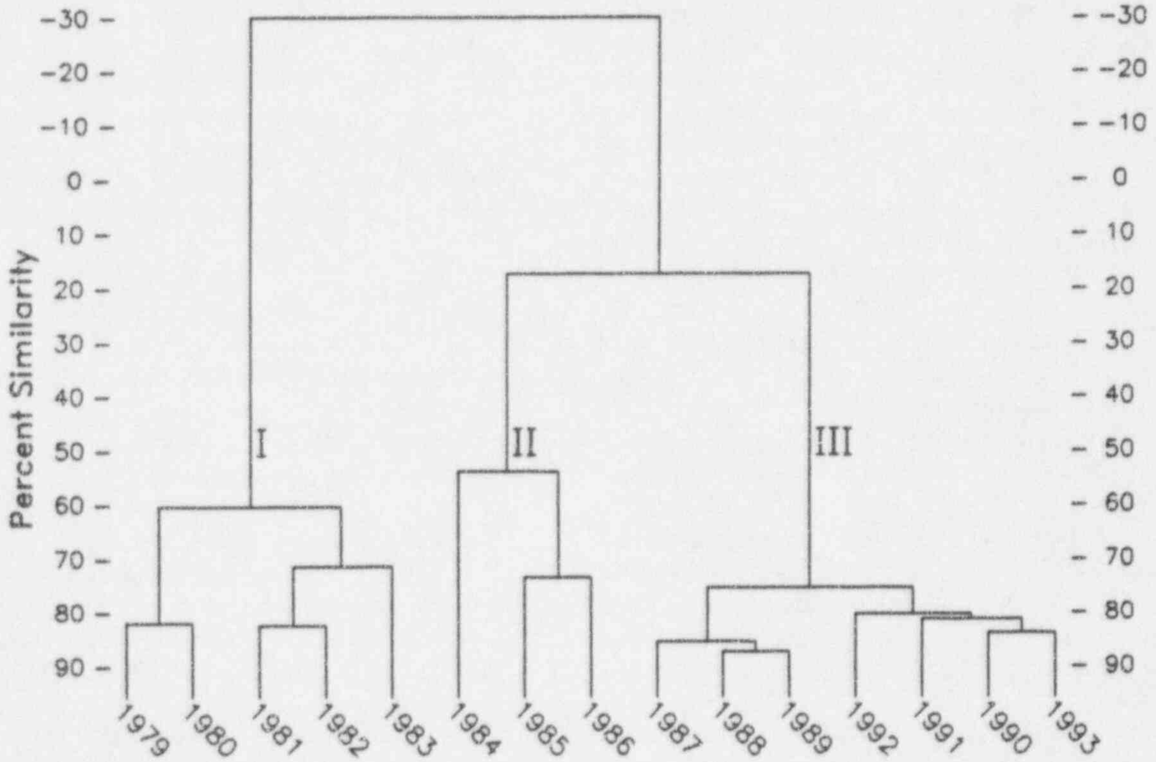

Fig. 7b. Clustering dendrogram of percent similarity of undisturbed communities, by year, at Fox Island-Exposed.

Among the station/operating period comparisons (Fig. 7a), similarities were generally highest between operational periods at the same station, indicating a relatively consistent species composition throughout the study period. A clear exception to this generalization occurred at Fox Island-Exposed, where the 3-unit collections (Group IV) were greatly dissimilar to those made during 2-unit operation, and to all other station/period collections. Differences among station groupings resulted from differing proportions of cover by intertidal taxa (Table 3a). For instance, Group I (SS, SE, GN and FE2) was distinguishable from Group II (WP, MP and BP) at approximately 50% similarity, at least partially on the basis of higher abundance of *Fucus* and less *Chondrus*. Group III (FS) was less similar (33% similarity), with higher percentages of *Semibalanus*, *Fucus* and available free space (rock), and very little *Chondrus*. Again, collections from Fox Island-Exposed during 3-unit operation (Group IV) were most dissimilar (6% similarity), owing to reduced abundance of *Semibalanus*, free space, *Chondrus* and *Littorina littorea* (Table 3a), and high abundance of *Codium* and ephemeral algae (*Polysiphonia novae-angliae*, *Enteromorpha* spp., *Ulva lactuca*).

The current FE shore community is distinctly different from that found previously at this site, and from those shore communities at other stations. However, the changes that occurred at FE are not directly attributable to start-up of Unit 3 in 1986. Rather, they result from the altered thermal regime produced by the opening of the second quarry cut in 1983 (NUSCO 1984, 1987, 1994). To illustrate the effects that construction and operational events have had on the FE rocky shore community, a clustering dendrogram was generated using annual collections (Fig. 7b). Three groups are evident at the 50% similarity level; Group I (1979-1983) represents an unimpacted community, before changes associated with the opening of the second cut were observed. Group II (1984-1986) consists of annual collections made during the period of maximal thermal incursion (2-cut, 2-unit operation), and Group III (1987-1993) represents years of community development at FE under 3-unit operating conditions. As with the station/period comparisons, differences among groups resulted from proportional changes in species composition (Table 3b). For example, Group I collections were typical of other moderately exposed sites, with extensive populations of *Fucus* and *Chondrus*. In the years following the opening of the second cut (Group II), *Fucus* and *Chondrus* were reduced or eliminated, and *Codium* became the dominant macroalga; *Polysiphonia* and *Enteromorpha* spp. were also abundant. Following Unit 3 start-up (Group III), *Codium* and ephemeral algae remained dominant, especially in the low intertidal quadrats, preventing the re-establishment of *Chondrus*. However, *Fucus* populations in the mid intertidal recovered to 1979-1983 levels (cf. previous *Fucus* section). Other significant events during this period were the appearance, persistence and expansion of populations of the perennials *Sargassum filipendula* and *Gracilaria tikvahiae*. The high level of similarity among annual collections in recent years at FE (>75% from 1987 to 1993, >80% from 1990 to 1993; Fig. 7b) indicates a consistent species composition, and a relatively stable community under 3-unit operating conditions.

Table 3a. Average percent substratum coverage of taxa with mean overall (all stations, both operating periods) abundance >1%, in groupings determined by community analysis; group numbers correspond to those in Fig. 7a.

taxon	Group I	Group II	Group III	Group IV
Semibalanus balanoides	25.83	24.56	34.68	11.18
Fucus vesiculosus	21.73	7.34	34.05	21.90
rock	17.78	19.70	31.79	5.18
Chondrus crispus	24.48	31.80	4.16	0.38
Codium fragile	1.13	0.23	1.25	37.30
Polysiphonia novae-angliae	1.55	2.12	0.34	11.34
Mytilus edulis	1.88	4.58	0.63	8.10
Enteromorpha linza	0.53	1.91	0.27	8.90
Ascophyllum nodosum	3.37	4.61	3.62	0.01
Ulva lactuca	2.23	1.93	0.64	5.87
sand	4.61	0.02	3.45	0.16
Enteromorpha flexuosa	0.98	0.57	0.56	5.34
Littorina littorea	2.04	2.51	2.48	0.01

Table 3b. Average percent substratum coverage of taxa with mean overall (Fox Island - Exposed only, all years) abundance >1%, in groupings determined by community analysis; group numbers correspond to those in Fig. 7b.

taxon	Group I	Group II	Group III
Codium fragile	0.95	24.98	38.20
Fucus vesiculosus	24.33	2.76	24.13
Semibalanus balanoides	18.94	15.29	10.27
rock	16.91	16.04	3.81
Chondrus crispus	30.28	4.88	0.44
Polysiphonia novae-angliae	5.90	6.03	12.07
Enteromorpha flexuosa	0.92	14.54	4.87
Enteromorpha linza	1.23	6.31	8.60
Ulva lactuca	3.09	3.00	6.09
Mytilus edulis	0.34	2.55	8.18
diatoms	0.11	1.41	2.28
Ralfsia verrucosa	2.77	1.01	0.00
Enteromorpha clathrata	0.00	3.74	0.02
Elachista fucicola	2.04	0.08	1.17
dead barnacles	1.43	1.00	0.84

Ascophyllum nodosum Studies

Ascophyllum nodosum studies during 1993-94 included monitoring of growth and mortality trends for three local populations of this common perennial brown alga. Previous Millstone studies have identified *Ascophyllum* as a key species within the overall ecological monitoring program, and studies elsewhere document the value of this species as a sensitive indicator of local environmental conditions. An extensive review of phenological, ecological and applied monitoring studies of *Ascophyllum* is presented in NUSCO (1993). Growth and mortality results from the most recent sampling year (1993-94) are presented below and compared with results from overall 2-unit and 3-unit operational periods.

Growth

Ascophyllum growth, described by the α parameter in the Gompertz growth model fitted to the data, is presented in Figure 8. *Ascophyllum* growth during 1993-94 (Fig. 8a) was significantly higher ($P<0.05$) at FN (115.5 mm) than growth at both GN (92.1 mm) and WP (73.7 mm). Growth differences between GN and WP were also significant. Inflection points for each growth curve, which identify the time of maximum growth rate, were similar in 1993, occurring slightly earlier at FN (19 July) than at GN and WP (21 July for both sites). Annual growth at GN during 1993-94 was not significantly different than growth over either 3-unit (96.4 mm) or 2-unit (90.1 mm) operational periods (Fig. 8b). However, the difference between growth estimates during 3-unit and 2-unit operation at GN was significant. The difference between operational period estimates at WP was also significant (Fig. 8c), with the 2-unit estimate (90.2 mm) higher than the 3-unit estimate (86.8 mm). Growth during both operational periods at WP was significantly higher than growth during 1993-94. At FN, growth during 1993-94 was significantly higher than during the 1985-86 2-unit year (90.5 mm), but not significantly different from the 3-unit operational mean (118.5 mm; Fig. 8d). The 3-unit mean was also significantly higher than growth during the 2-unit year.

Power plant operation continued to impact the *Ascophyllum* population at FN during 1993-94. Periodic thermal plume incursion at this site during 3-unit operation (water temperatures elevated 3-4°C for 3-4 hours each tidal cycle) creates conditions which enhance *Ascophyllum* growth. Specifically, these temperature conditions increased growth by: 1) extending the period of "normal" or "ambient" peak growing conditions for local *Ascophyllum* populations (18-21°C; Kanwisher 1966; Chock and Mathieson 1979); 2) more closely synchronizing these periods of optimal growing temperatures with the period of maximum daily solar irradiance (June); and 3) elevating temperatures in late summer above normal maxima but below stress levels (22-25°C), increasing plant respiration and growth rates without exceeding photosynthate production (Brinkhuis et al. 1976; Stromgren 1977, 1981; Vadas et al. 1978). Similar

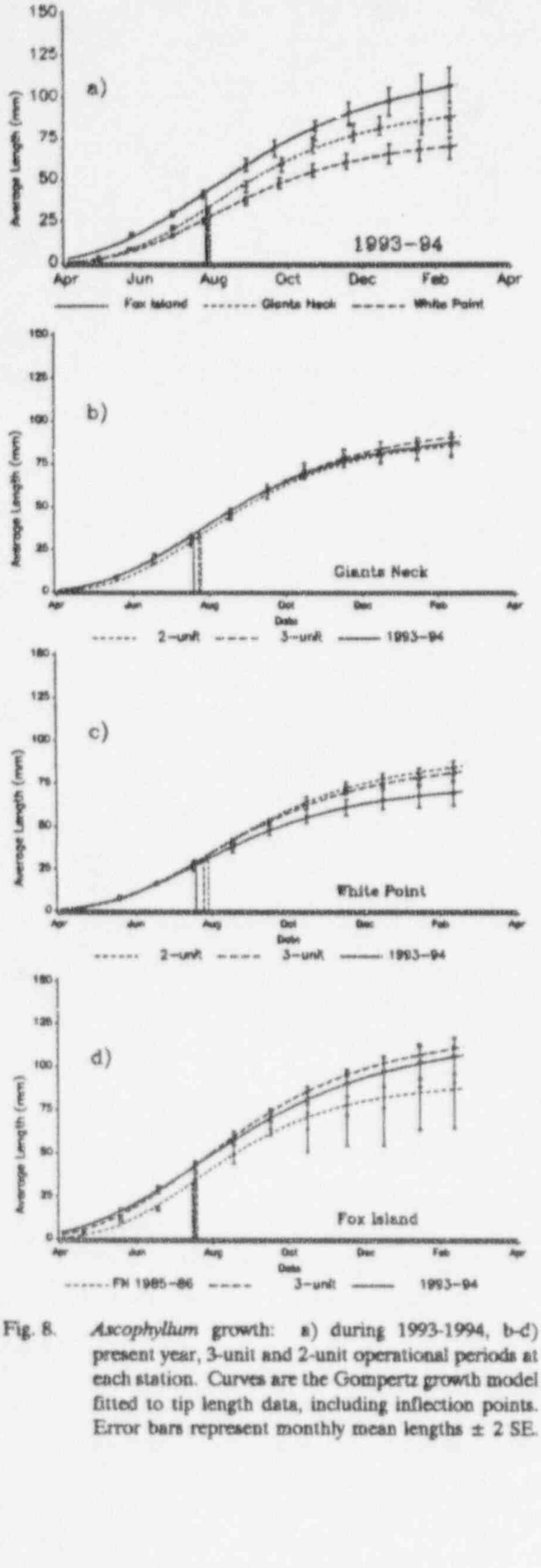

Fig. 8. *Ascophyllum* growth: a) during 1993-1994, b-d) present year, 3-unit and 2-unit operational periods at each station. Curves are the Gompertz growth model fitted to tip length data, including inflection points. Error bars represent monthly mean lengths ± 2 SE.

conditions have been implicated when *Ascophyllum* growth enhancement was reported near other coastal power plants (Vadas et al. 1976, 1978; Wilce et al. 1978). Consequently, higher growth rates and significantly longer apical tips have been consistently observed at FN, when compared to the two reference site populations.

Another relatively consistent spatial relationship between reference sites became apparent during 3-unit studies. During most 3-unit years, including the present study year (1993-94), higher growth occurred at GN, the control site well beyond any power plant influence (NUSCO 1992, 1994). Several conclusions can be drawn from this relationship. First, site-specific natural environmental conditions are important factors in determining *Ascophyllum* growth, and have been apparently better at GN, particularly during 3-unit study years. And second, although predictions that the 1.5°F thermal plume isotherm extends to WP during ebb tide (NUSCO 1988), lower growth there relative to that at GN indicates that the WP population remains unimpacted by MNPS operation.

Year to year variability in *Ascophyllum* growth was least at reference sites and reflects the relative temporal stability of site-specific natural environmental conditions (Fig. 8b and c). Higher variability in annual growth noted at FN results from population exposure to a fluctuating thermal load from the power plant discharge created by scheduled and unscheduled unit outages. As discussed in previous reports (e.g., NUSCO 1992, 1994), annual growth at FN is highest when periods of 3-unit operation (maximum thermal load) are longer or more frequent during the peak growing season (May-November), compared to years when one or more unit outages occurs during that time (NUSCO 1992, 1993). The degree of growth enhancement at FN during 1993-94 was intermediate compared to previous years and likely due to the extended refueling outage of Unit 3 from August through October 1993.

Mortality

Indicators of environmental stress to local *Ascophyllum* populations include patterns of frond base tag loss (plant loss; Fig. 9) and apical tag loss (tip loss; Fig. 10) resulting from plant breakage or mortality. Plant loss at GN during 1993-94 (56%)

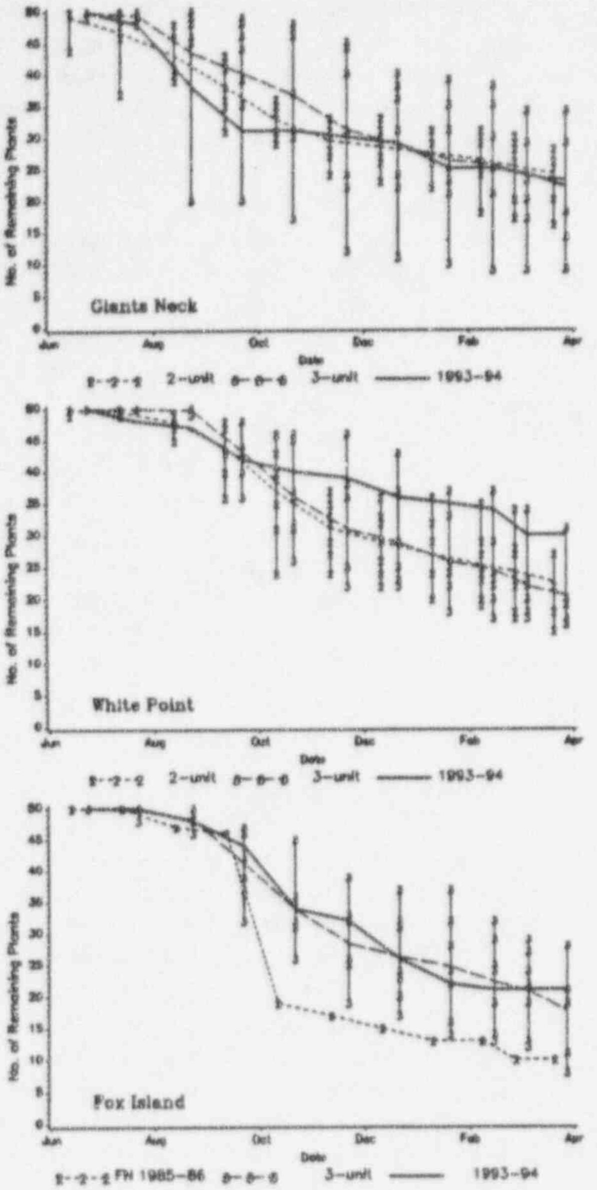

Fig. 9. *Ascophyllum* mortality, as number of remaining tagged plants, at each station.

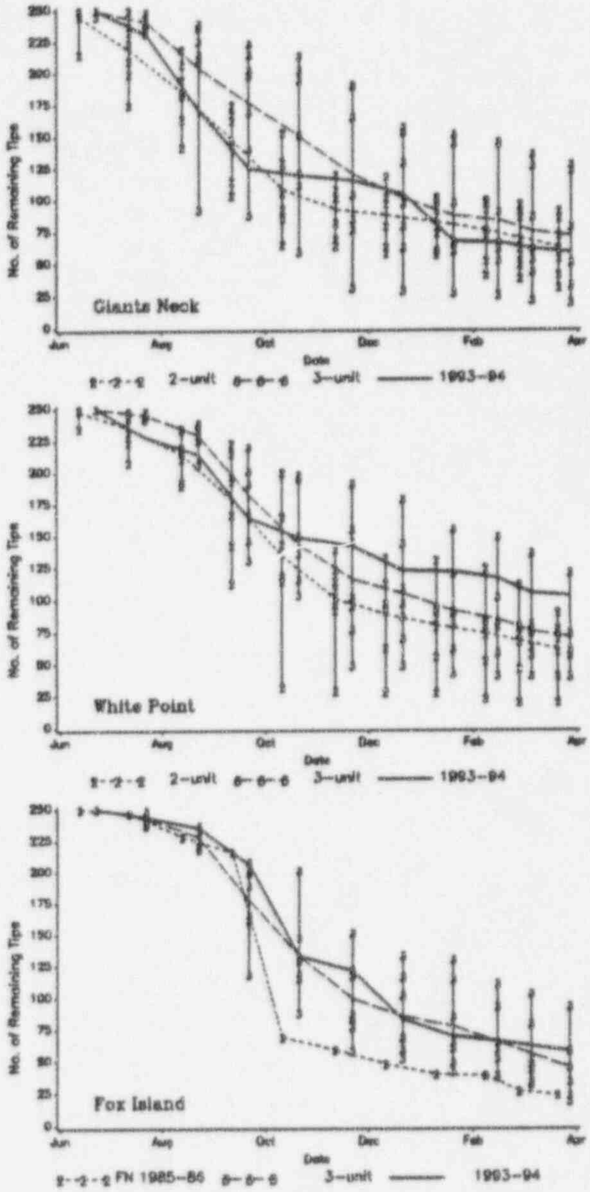

Fig. 10. *Ascophyllum* mortality, as number of remaining tagged tips, at each station.

was slightly higher than both 2-unit (52%) and 3-unit (54%) operational means. At WP, plant loss during 1993-94 (40%) was low relative to operational period means of 55% and 59% for 2-unit and 3-unit periods, respectively. Similarly, 1993-94 plant loss at FN (58%) was lower than the 1985-86 2-unit year (80%) and 3-unit (65%) means. Temporal relationships for tip loss at each station in terms of tip loss were similar to those described above for plant loss. Tip loss at GN during 1993-94 was 77%, which was higher than the 2-unit and 3-unit means of 75% and 72%, respectively. Conversely, 1993-94 tip loss at WP (59%) was low compared to the 2-unit (75%) and 3-unit (72%) means. Low degree of tip loss for 1993-94 was also observed at FN (77%), relative to the operational period means of 90% (1985-86 2-unit year) and 82% (3-unit years).

Three-unit *Ascophyllum* studies have revealed no evidence of impact-related mortality to local populations. Somewhat higher mortality rates have been observed at FN, our sampling site nearest the discharge, than at reference sites. However, these higher mortality rates do not appear to be related to proximity to the discharge, but rather to the exposed orientation of this site, in contrast to the more sheltered reference sites. Furthermore, no consistent relationship between level of plant operation (discharge thermal load) and mortality at FN, similar to that described for growth, has been apparent during 3-unit years. An area-wide seasonal pattern of mortality has been observed throughout our studies which further implicates wave-induced stress as a major cause of mortality. During both 2-unit and 3-unit operational periods, mortality rates were highest during the months of August through November, when strong storms and high energy waves were frequent. Many studies elsewhere point to stress from wave action as the most important factor and report a strong relationship between mortality and degree of site exposure to prevailing winds and storms (Baardseth 1955, 1970; Jones and Demetropoulos 1968; Vadas et al. 1976, 1978; Wilce et al. 1978; Cousens 1982, 1986; Vadas and Wright 1986).

The status of recovery of the *Ascophyllum* population at FO, our original experimental site, following power plant-induced population elimination in 1984 has not changed from that reported in recent years (NUSCO 1993, 1994). Some individual plants have settled, grown and persisted at FO during 3-unit operation; however, no significant recovery has occurred to date. Environmental conditions at FO created by 3-unit operation, although less stressful than those during 2-unit 2-cut operation, may be outside the extremely limited range of conditions required for successful widespread recruitment.

Conclusions

Differences among rocky intertidal communities monitored during NUSCO studies were primarily attributed to variation in site-specific physical environmental conditions. At sites not affected by the MNPS thermal plume, variability in the level of wave-induced disturbance was the direct or underlying cause of the most notable among-site differences in the occurrence and distribution of local species. Susceptibility of local rocky shore communities to wave-induced disturbance was determined through site orientation in relation to prevailing wind-generated waves and ability of available substratum (slope) to dissipate the horizontal force of those waves.

Physical factors were also responsible for community differences at sites in the thermal plume area, such as those observed in the Fox Island area (FE and FN). The ecological effects of elevated temperature regimes on population and community parameters were detected at these sites, and directly attributed to the operation of MNPS. For example, qualitative studies continued to document shifts in occurrence of the algal flora at FE during 1993-94, which included presence or extended season of occurrence for species with warm-water affinity and absence or abbreviated season for species with cold-water affinity.

Impact-related shifts in species abundance were observed during 1993-94 only at FE, and were most pronounced in the low intertidal, where elevated temperature conditions were most severe. The low intertidal community at FE, which prior to 1983 was unimpacted and characterized by populations of *Chondrus* and *Fucus* and seasonally abundant ephemeral algae, has been replaced by a persistent community dominated by *Codium, Ulva, Enteromorpha* and *Polysiphonia*. Also, populations of species observed in undisturbed transects only at FE (*Sargassum, Gracilaria*) continued to persist and expand during 1994.

More subtle thermal incursion (elevated temperatures 2-3°C above ambient) impacted the *Ascophyllum* population nearest the discharge (FN) by causing plants to grow longer and more rapidly at that site during 1993-94 relative to growth of *Ascophyllum* at more distant stations, a pattern consistent with most 3-unit years. The degree of growth enhancement at FN during 1993-94 was intermediate, compared to previous years at that site, likely due to reduced overall thermal plume incursion resulting from an extended outage of Unit 3 for much of the peak growing season. *Ascophyllum* plant and tip mortality were associated primarily with exposure to storm forces, rather than with proximity to the discharge.

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Benthic Infauna

Benthic Infauna

Introduction

Most of the subtidal benthic habitat in the vicinity of Millstone Nuclear Power Station (MNPS) consists of soft sediment (sand and mud), which supports an abundant and diverse community of invertebrates. Infaunal communities are an important component of coastal ecosystems for several reasons. Infauna are a source of food for numerous invertebrate and vertebrate species, including demersal fishes (Richards 1963; Moeller et al. 1985; Watzin 1986; Horn and Gibson 1988; Commito and Boncavage 1989; Franz and Tanacredi 1992; Commito et al. 1995). Additionally, sediment reworking resulting from the burrowing and tube-building activities of infauna promotes nutrient recycling from the sediments to the water column (Goldhaber et al. 1977; Aller 1978; Gaston and Nasci 1988).

The close association of benthic communities with the sediments, where most pollutants ultimately accumulate, also make them an effective integrator of short- and long-term environmental conditions (Diaz and Schaffner 1990; Warwick et al. 1990). Because many studies have documented changes in benthic communities following disturbance (Boesch and Rosenburg 1982; Young and Young 1982; Gaston and Nasci 1988; Regnault et al. 1988; Rees and Eleftheriou 1989; Warwick et al. 1990; NAESCO 1994), a framework of baseline studies exists to aid in evaluating impacts of human activities on marine benthic systems.

Environmental variability characteristic of coastal systems (Holland 1985; Nichols 1985; Holland et al. 1987; Warwick 1988; Rees and Eleftheriou 1989), together with a lack of complete understanding of how physical and biological factors combine to impose structure on, and control the functions of, benthic communities (Diaz and Schaffner 1990). Thus long-term monitoring studies are necessary to assess the impacts of human activities on marine environments (Thrush et al. 1994). Such studies are the principal means of characterizing species composition and fluctuations in abundance which might occur in response to acute or chronic climatic conditions (Boesch et al. 1976; Flint 1985; Jordan and Sutton 1985), to variations in biological factors such as competition and predation (e.g., Levinton and Stewart 1982; Woodin 1982; Kneib 1988), or to human activities.

Accordingly, benthic infauna in soft-bottom subtidal habitats in the vicinity of MNPS have been monitored since 1973. This monitoring program was designed to measure infaunal species composition and abundance, to identify spatial and temporal patterns in community structure and abundance, and to assess whether observed changes might have been the result of construction and operation of MNPS. To date, Millstone studies have identified impacts to infaunal communities that were attributed to Unit 3 intake construction (NUSCO 1987) and to 3-unit operations (NUSCO 1988a), as well as regional shifts in species composition and abundance that apparently were the result of natural events. This report presents data collected during the 1994 sampling year, and compares them to results of monitoring local infaunal communities during 2-unit (1979-85) and 3-unit (1986-1994) operational periods at MNPS.

Materials and Methods

Subtidal infaunal communities in the vicinity of MNPS were sampled quarterly (September, December, March and June) from 1979 through 1994 at four stations (Fig. 1). The Giants Neck station (GN), located 6 km west of MNPS, is outside the area potentially affected by power plant operations. This station was used to identify possible region-wide shifts in infaunal community structure and composition which occur independently of power plant operations. The intake station (IN) is located 100 m seaward of MNPS Unit 2 and Unit 3 intake structures, and is exposed to scour produced by inflow of cooling water and the effects of periodic dredging. The effluent station (EF), located approximately 100 m offshore from the station discharge into Long Island Sound, is exposed to increased water temperatures, scour, and to chemical or heavy metal additions to the cooling water discharge. The Jordan Cove station (JC) is located 500 m east of MNPS and is considered potentially impacted by 3-unit operations. The area encompassing this station can experience increases in water temperatures of 0.8 to 2.2°C above ambient during certain tidal stages (principally ebb tide) due to the

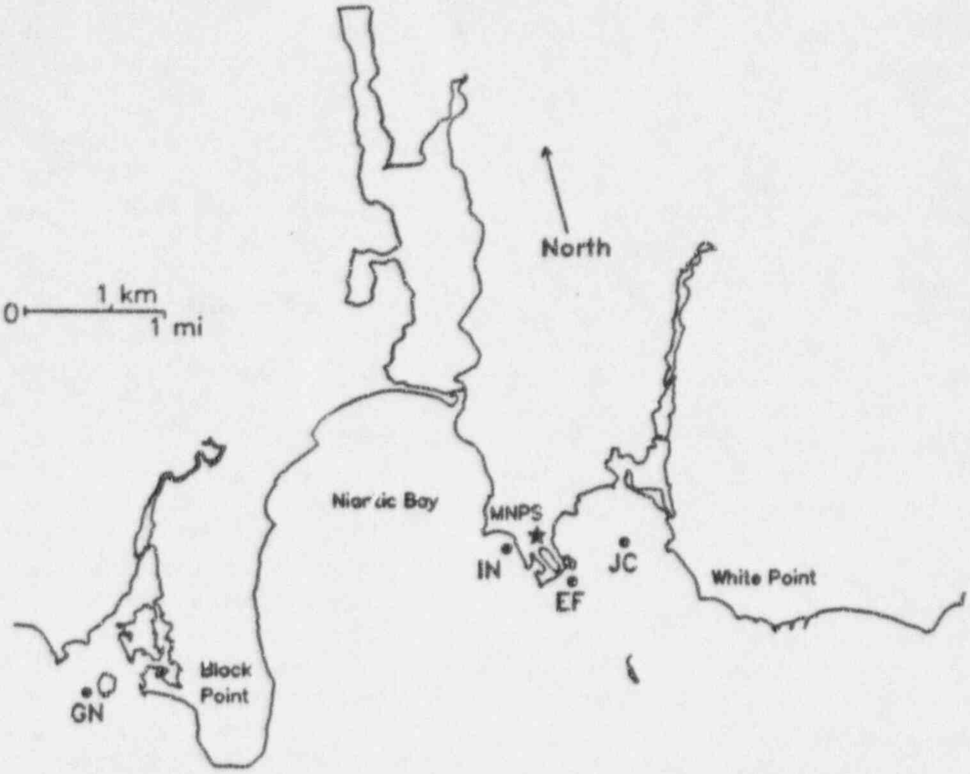


Fig. 1. Map of the Millstone Point area showing the location of the infaunal sand stations (EF=Effluent, GN=Giants Neck, IN=Intake, JC=Jordan Cove).

3-unit thermal discharge of MNPS (NUSCO 1988b). At each station, ten replicate samples (0.0079 m^2 each) were collected by SCUBA divers using a hand-held coring device 10 cm in diameter x 5 cm deep. Each sample was placed in a 0.333 mm mesh Nitex bag and was brought to the surface. Samples were then taken to the laboratory, where they were fixed with 10% buffered formalin and after a minimum of 48 h, organisms were floated from the sediments onto a 0.5 mm mesh sieve and preserved in a 70% ethyl alcohol/Rose Bengal solution. Samples were examined using dissecting microscopes (10x) and organisms were sorted into major groups (annelids, arthropods, molluscs, and others) for later identification to the lowest practical taxon and counted. Oligochaetes and rhynchocoels were each treated in aggregate because of the difficulties associated with identifying these organisms. Organisms that were too small to be quantitatively sampled by our methods (meiofauna; e.g., nematodes, ostracods, copepods, and foraminifera) were not sorted. Grain size and the silt/clay fraction were determined from a 3.5 cm diameter x 5 cm core, taken at the time of infaunal sampling. Sediment samples were analyzed using the dry sieving method described by Folk (1974).

Data Analyses

This report summarizes results of the macrofaunal sampling program conducted from September 1979 to June 1994. The period extending from September 1979 through June 1985 is considered the 2-unit operational period, while September 1985 through June 1994 is referred to as the 3-unit period. A sampling year encompasses quarterly collections made from September through June of the following calendar year.

Sediments

Sediment sieve fraction weights were used to construct cumulative curves for 2-unit (1980-1985) and 3-unit (1986-1994) operational periods by pooling quarterly weights from each sieve used for grain size analysis within each operational period, with years serving as replicates. Shifts in sedimentary environments over the 2-unit and 3-unit operational periods were then quantitatively assessed using the Gompertz function. This function has a sigmoid shape and can describe cumulative data (e.g., growth data) that are not necessarily symmetrical about the midpoint of their range (Draper and Smith 1981). This feature provides the flexibility to fit cumulative data with or without an inflection point (s-shaped versus parabolic) within the observational range. The form of the Gompertz function used was:

$$C_{\phi} = 100\exp(-\beta e^{-\kappa\phi})$$

where C_{ϕ} is the cumulative sediment weight at a given particle size, and β and κ are the location and shape parameters, respectively (Draper and Smith 1981). This function was fitted to data separately for 2-unit and 3-unit operational periods using non-linear regression methods. Two-sample t-tests were used to test for differences between the β and κ parameters of curves based on data collected during each operational period.

Multiple Regression Analyses

Multiple regression techniques were used to minimize the unexplained temporal variation in community abundance, in number of species and in the abundance of numerically dominant taxa. Several explanatory variables (described below) were used to remove variation that was attributable to fluctuations in sediments, reproductive or recruitment cycles, or climatic conditions. This technique was used to improve the sensitivity of analyses later performed to identify and compare long-term trends in data from the 2-unit and 3-unit operational periods. Analyses were based on average quarterly abundance data after $\ln(x+1)$ transformation and on species number collected from September 1979 through June 1994. Explanatory variables used in the regression analyses were as follows:

Precipitation - Daily precipitation records compiled by the U.S. Weather Bureau at the Groton Filtration Plant, Groton, CT were obtained from June 1976 through June 1994. Values to the nearest 0.01 inch were used as "rain" data.

Water and Air Temperatures - Ambient water temperatures (at the intake structures) and air temperatures (recorded at the 33 foot level of the Millstone meteorological tower) were obtained from the Northeast Utilities Environmental Data Acquisition Network (EDAN). Daily averages of 15-minute values were calculated for the period June 1976 through June 1994.

Wind Speed and Direction - Wind speed and direction (at the 33 foot level of the Millstone meteorological tower) were extracted from the EDAN database for each 15-min interval from June 1976 through June 1994. These values were used to calculate a wind index, which was the wind speed weighted by wind direction. A navigational chart of the sampling area was used to calculate site-specific wind directional weighting coefficients. The directional weight ranged from 0, when wind could not influence the station, to a maximum of 1, when wind-induced waves could directly affect the area. The wind index was then computed by multiplying the directional weight by the wind speed. Because the effect of wind was assumed to be cumulative, daily averages were derived using only wind index values greater than 0 (that is, when the wind was from a direction which could produce wind effects).

Climatic Extremes (Deviations) - Additional explanatory variables were created to represent unusual climatic conditions which occurred during the sampling period. High or low deviations (i.e., extremes) were derived for wind, rain, water and air temperature data and calculated as the difference between the quarterly mean or daily value and the 17-year mean (1977-1994) for that quarter. Deviations based on quarterly means reflect the effects of longer-term extremes (e.g., an unusually cold winter), while those based on daily values tend to remove the effects of shorter-term episodic events (e.g., storms). Daily deviations were averaged and also summed in each sampling quarter to assess cumulative effects.

Sedimentary Parameters - Sediment mean grain size and silt/clay content were obtained as part of the monitoring studies and these quarterly values were used directly as explanatory variables in the multiple regression models.

Reproductive - Recruitment Component - Infaunal organisms in the Millstone area exhibit annual peaks in abundance, often reflecting the seasonal nature of reproduction and recruitment cycles or periods of favorable climatic conditions. Spectral analyses of quarterly data showed annual cycles in community abundance and number of species. To account for this periodicity, harmonic terms (L[illegible]da and Saila 1986) with a period of 1 year were also included as explanatory variables in the regression models.

In all, 32 variables were initially used during model selection steps. These variables included two sedimentary parameters, two seasonal/reproductive components and seven climatic variables, each of which had four values representing daily and quarterly high and low extremes.

Response Modeling and Trend Analysis

Quarterly abundance and species number data from September 1979 through June 1985 (2-unit period) and September 1985 through June 1994 (3-unit period) were detrended separately using a linear regression model. A step-wise multiple regression was then conducted with the residuals of quarterly data (i.e., the variability or "noise" about the linear trend) over the entire sampling period to identify explanatory factors and combinations of factors whose regression coefficients were significantly different from zero ($p \leq 0.05$). This probability level was chosen to guard against fitting more parameters than could be reliably estimated, given the sample size. The model that minimized the mean-square-error and maximized the R^2 was selected to "clean" the original data, and produce a time-series free from variation attributable to concomitant physical factors or known biotic processes such as reproductive/recruitment cycles. Straight lines were then refitted to the variance-reduced time-series corresponding to the 2-unit and 3-unit periods. The nonparametric (i.e., distribution-free) Mann-Kendall test (Hollander and Wolfe 1973) was next used to determine whether these 2-unit and 3-unit series exhibited significant trends and Sen's nonparametric estimator of the slope (Sen 1968) was used to test for trend differences. These two tests were suggested by Gilbert (1989) as particularly well suited for analyzing environmental monitoring data because no distributional assumptions are required, and because relatively short time series ($n < 10$) are acceptable. In this report, plots of the original quarterly data, adjusted data and a graphical representation of the linear trend are provided for community abundance, numbers of species and for selected taxa.

Community Analyses

Abundances of the top ten numerically dominant taxa collected at each station were used to construct cumulative abundance curves (k-dominance curves) for 2-unit and 3-unit operational periods. Comparison of k-dominance curves has been suggested as a means of assessing shifts in the structure of macrofaunal communities (Warwick 1986; Warwick et al. 1987). Curves were constructed by plotting percentages of cumulative abundance (ordinate) versus the natural logarithm of a taxon's rank (abscissa). To assess possible shifts in infaunal community structure between 2-unit and 3-unit operational periods, the same Gompertz function used for sediment data analysis was fitted to cumulative abundance data by substituting species abundance for sediment weight, and species rank for particle size in the equation. Two-sample t-tests were used to compare parameters of curves representing data collected during the 2-unit and 3-unit operational periods.

Comparisons of annual collections at each station were made by calculating the Bray-Curtis similarity index between each pair of years, using the formula (Clifford and Stephenson 1975):

$$S_{jk} = \frac{\sum_{i=1}^{n} 2\min(X_{ij}, X_{ik})}{\sum_{i=1}^{n} (X_{ij} + X_{ik})}$$

where S_{jk} is the similarity index between year j and year k; X_{ij} is the log transformed (ln+1) abundance of species i in year j; X_{ik} is the abundance in year k; and n is the number of species in common, that met the criterion of accounting for at least 0.5% of the individuals collected. A flexible-sorting ($\beta = -0.25$), clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967).

Results

Sedimentary Environment

Sediments at infaunal sampling stations in the vicinity of Millstone during 1994 contained fine to coarse sands (Fig. 2). Sediments were generally finest at JC (quarterly mean grain size range 0.14

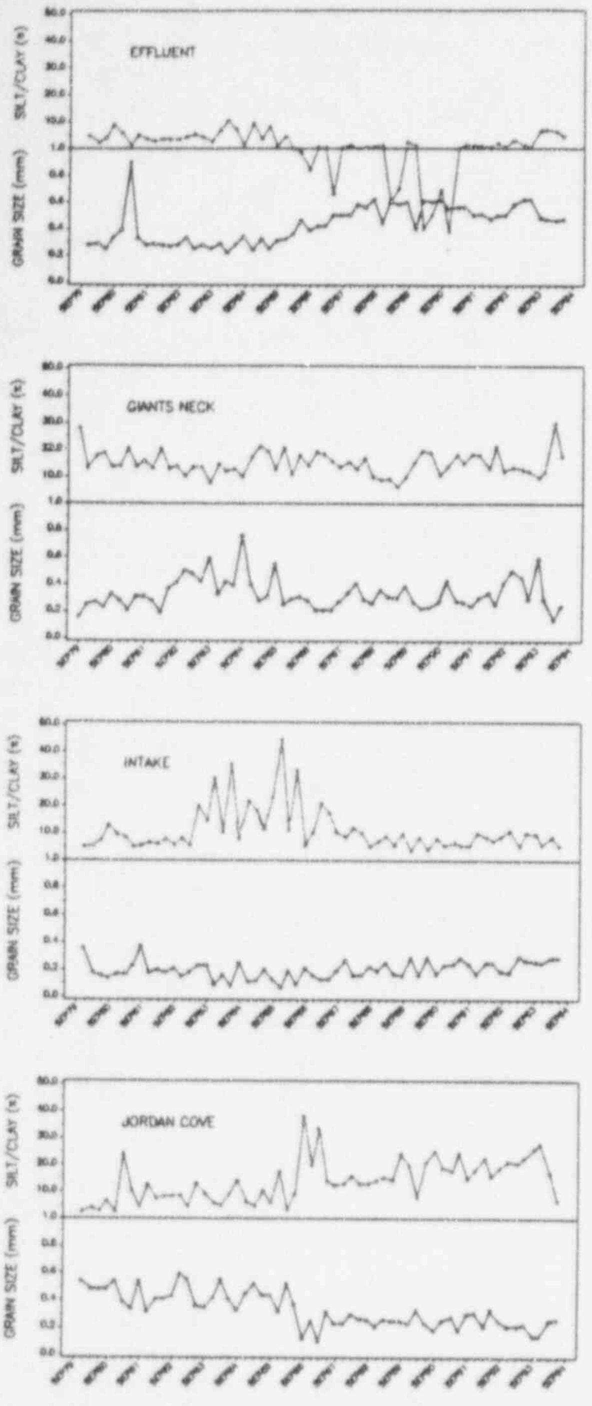


Fig. 2. Quarterly mean grain size (mm) and silt/clay content (%) of sediments at Millstone subtidal stations from September 1979 to June 1994.

to 0.26 mm) and coarsest at EF (0.46-0.48 mm). Quarterly mean grain size ranges for IN and GN were 0.25-0.29 mm and 0.14-0.59 mm, respectively. Silt/clay content estimates in sediments collected in 1994 were more variable and generally higher at JC (quarterly range 7.0-27.9%) and GN (9.8-29.9%), than at EF (5.4-7.2%) and IN (5.8-9.6%). Sediment mean grain size and silt/clay contents observed during 1994 were, in most cases, within the ranges of these two sediment parameters during both 2-unit and 3-unit operational periods with the following exceptions; silt/clay contents at EF in 1994 were the highest recorded during the 3-unit period, but within the 2-unit range. Conversely, silt/clay content in June 1994 at JC was the lowest recorded at this site since September 1986. March 1994 values at GN were outside previous ranges for both silt/clay (higher) and mean grain size (lower).

Cumulative curves based on sediment sieve fraction weights (Fig. 3) were used to characterize subtidal environments, and allowed statistical comparison of sediments collected at each station during 2-unit (1979-85) and 3-unit (1986-94) operational periods. Based on t-tests of Gompertz parameters derived from model curves, significant differences between 2-unit and 3-unit periods were noted at all stations except GN. The shift at EF reflected the declining silt/clay fraction and the increasing grain size since Unit 3 began operation (Fig. 2). Conversely, silt/clay content increased and average grain size decreased over the same time period at JC. An operational period difference in sediments at IN (coarser during 3-unit operation) resulted from the addition of data from the especially coarse sediments collected in 1994; no between-period difference was detected for IN sediments in recent previous years (e.g., NUSCO 1993, 1994).

General Community Composition

Mean numbers of species and of individuals in major invertebrate groups collected during 1994, and during 2-unit and 3-unit operating periods, are presented in Table 1. The annual mean numbers of species at infaunal stations in 1994 ranged from 108 (GN) to 122 (EF). The 1994 means at GN and JC were lower than means for the 2-unit period (125 and 118 species, respectively) and the 3-unit period (119 species at both stations). The

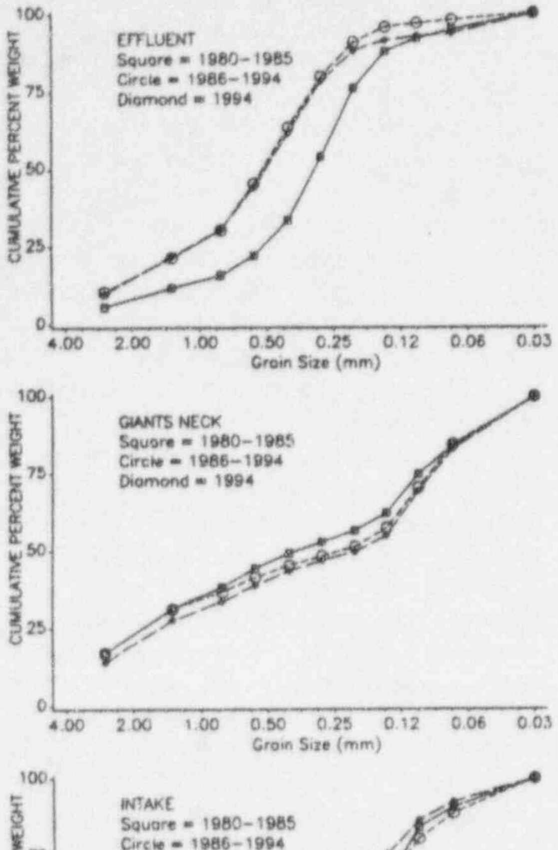


Fig. 3. Cumulative curves based on fractional weights of sediments collected during the 2-unit (1980-1985) and 3-unit (1986-1994) operational periods, and during 1994, at Millstone subtidal stations.

1994 mean number of species at IN (112) was higher than both operational means: 87 (2-unit) and 102 (3-unit). For both operational periods, mean number of species has been highest at EF and lowest at IN in relation to operational means at other stations.

The largest number of individual organisms collected in 1994 was at JC (14,167), with smallest numbers collected at EF (5,149) and IN (5,255). A total of 9,132 organisms was collected at GN in 1994. Spatial relationships among sampling sites over both operational periods for numbers of individuals were similar to those in 1994: highest at JC during 2-unit (12,110) and 3-unit periods (12,576) and lowest at IN (2,405 and 5,594, respectively). Numbers of individuals collected in 1994 were lower than operational period means at EF and GN, higher than operational means at JC, and between operational means at IN.

Polychaetes were the dominant group in terms of numbers of species at all stations during 1994, ranging from 56-64 species. Polychaetes were also the most abundant taxon in the 2-unit and 3-unit operating periods (ranges 43-67 and 53-63, respectively). Except at EF during the 3-unit period and during 1994, polychaetes also dominated in terms of numbers of individuals (nearly 50% of total individuals). Mollusc and arthropod species were not as abundant as polychaetes; 1994 ranges for numbers of species were 21-27 for both groups. Numbers of molluscs and arthropods reported for GN, IN and JC in 1994 were generally consistent with operational period means. At EF during 3-unit operation, including 1994, oligochaetes were dominant. At GN and JC, oligochaetes were second in abundance (after polychaetes) during 1994 and over both operational periods, followed by either arthropods and molluscs. Oligochaetes were also ranked second in 1994 at IN, but were ranked third and fourth during 2-unit and 3-unit periods, respectively. Arthropods and molluscs ranked third or fourth in abundance at IN during 1994 and both operational periods. Rhynchocoels and "Others" contributed little to total abundance during either operational period.

Some notable differences in general community composition between operational periods were evident on the basis of species number. The polychaete-dominated community at EF, present during 2-unit operation, was replaced during the 3-unit period by one dominated by oligochaetes.

TABLE 1. Annual mean number of species (S), number of individuals (N) for each major taxon collected in 1994, during 2-unit (1980-1985) and 3-unit (1986-1994) operational years at Millstone subtidal stations.

	1994		2-Unit Period (1980-1985)				3-Unit Period (1986-1994)			
	(S)	(N)	MEAN (S)	CV[a]	MEAN (N)	CV	MEAN (S)	CV	MEAN (N)	CV
Effluent										
Polychaeta	64	1853	67	2.7	4675	17.7	62	2.5	2499	11.6
Oligochaeta	-	2081	-	-	2885	13.9	-	-	4043	12.0
Mollusca	27	672	29	4.8	497	29.3	27	3.7	494	16.8
Arthropoda	27	405	39	4.5	723	21.9	29	4.7	415	8.4
Rhynchocoela	-	84	-	-	138	23.2	-	-	171	27.6
'Others'	4	54	4	20.4	11	48.7	4	7.5	136	37.5
Total	122	5149	139		8930		122		7758	
Giants Neck										
Polychaeta	56	5612	67	4.4	6683	12.9	60	4.6	6960	8.0
Oligochaeta	-	2837	-	-	1932	12.6	-	-	2318	5.4
Mollusca	25	303	20	9.9	260	20.7	25	5.1	285	8.7
Arthropoda	23	313	35	4.6	624	5.8	31	8.4	970	22.3
Rhynchocoela	-	50	-	-	62	20.4	-	-	72	18.7
'Others'	4	17	3	26.4	8	43.2	3	10.9	17	25.5
Total	108	9132	125		9569		119		10622	
Intake										
Polychaeta	62	3389	43	3.3	1110	9.3	53	4.0	3043	19.0
Oligochaeta	-	1023	-	-	253	16.2	-	-	466	23.1
Mollusca	21	531	18	10.8	199	27.0	20	4.1	506	12.1
Arthropoda	27	284	25	8.2	829	47.4	27	4.4	1552	52.6
Rhynchocoela	-	26	-	-	15	26.4	-	-	25	17.4
'Others'	2	2	1	68.3	1	74.2	1	39.5	2	27.4
Total	112	5255	87		2405		102		5594	
Jordan Cove										
Polychaeta	62	10083	64	4.7	6513	23.2	63	2.4	8131	11.6
Oligochaeta	-	2307	-	-	4124	24.2	-	-	2662	6.8
Mollusca	26	1060	24	12.8	446	24.6	27	3.2	755	8.4
Arthropoda	21	610	27	6.2	641	55.4	26	5.9	925	32.5
Rhynchocoela	-	104	-	-	79	12.3	-	-	92	12.1
'Others'	2	3	3	33.1	4	28.1	3	7.3	11	33.2
Total	111	14167	118		12110		119		12576	

[a] C.V. of the mean estimate = (Standard Error/Mean) x 100

Abundances of polychaetes and oligochaetes were more similar during recent years (e.g., NUSCO 1993, 1994), including 1994. Also, the number of arthropod species at EF was lower during 3-unit operation than during the 2-unit period. The opposite trend was observed at JC; i.e., more polychaetes and arthropods, and fewer oligochaetes were collected, on average, in the 3-unit period than in the 2-unit period. At IN during 3-unit operation numbers in all taxonomic groups increased. At GN, there was little change in numbers between operational periods.

Community Abundance

Ranges of average quarterly abundance (individuals per core) at infaunal stations during 1994 were 110-144 at EF, 205-273 at GN, 84-154 at IN, and 302-372 at JC (Fig. 4). At each station, 1994 densities were within the range for their respective 15-year time series. In general, infaunal abundance at all stations during the 3-unit period (1986-1994) was similar to that observed during the 2-unit operational years. Overall, seasonal and annual fluctuations were lowest at EF and highest at IN; however, no consistent seasonal periodicity was evident in community abundance at any station during the 15-year period. Analyses of long-term trends in community abundance indicated a significant ($p>0.01$) increasing trend at EF during 2-unit operation, resulting primarily from peak abundances recorded near the end of that period (1984). During the 3-unit operating period, a significant trend (increase) was detected only at JC.

Numbers of Species

The mean number of species (per core) collected during 1994 ranged from 21-24 at EF, 20-24 at GN, 18-22 at IN, and 24-29 at JC (Fig. 5). These means were within the range of quarterly means observed at each station over the previous 15-year period. A significant increasing trend in quarterly species number was evident at EF and GN during 2-unit operation, and resulted from high species richness evident at most stations during the period 1984-87. During 3-unit operation, there were no significant trends in species richness evident at EF, GN, or JC. However, at IN, there was a significant increasing trend in quarterly numbers of species detected for the 3-unit operating period.

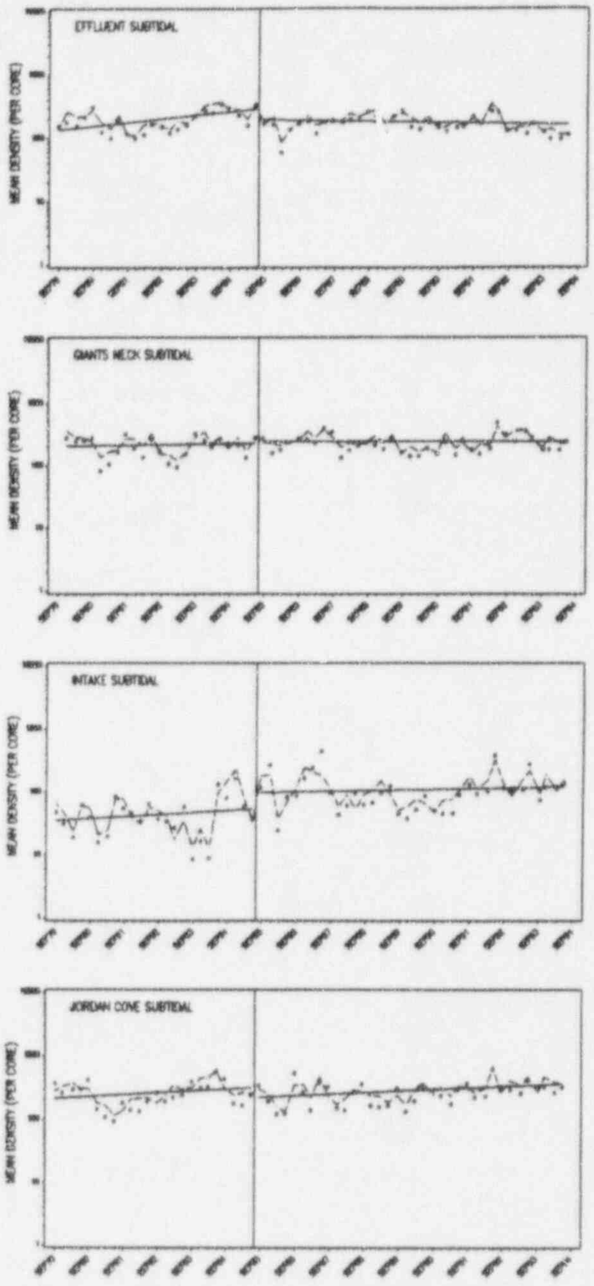


Fig. 4. Quarterly abundance data (dots), and variance-reduced data (dashed line), and linear-trends for subtidal communities before and after Unit 3 operation at MNPS.

Community Dominance

Dominant taxa identified in infaunal samples during 1994 included representatives of the class Oligochaeta, the polychaete species *Aricidea catherinae*, *Mediomastus ambiseta*, *Tharyx* spp.,

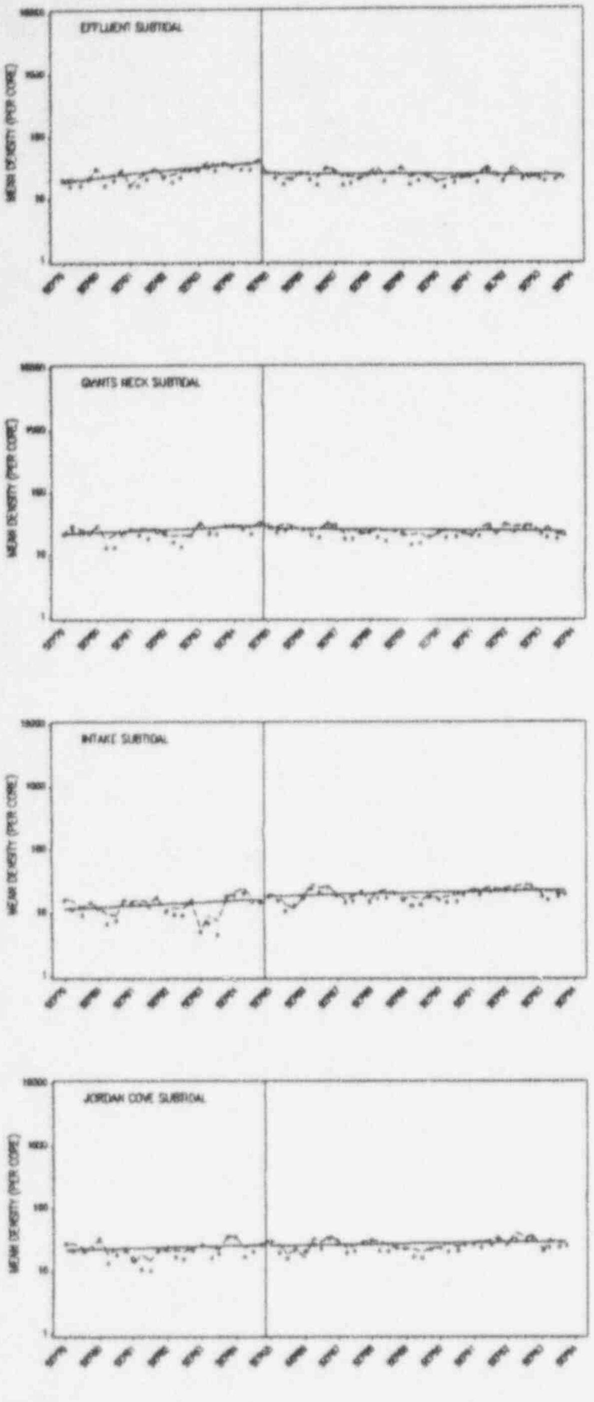


Fig. 5. Quarterly number of species data (dots), and variance-reduced data (dashed line), and linear-trends for subtidal communities before and after Unit 3 operation at MNPS.

Protodorvillea gaspeensis, *Prionospio steenstrupi*, *Exogone hebes*, representatives of the polychaete order Archiannelida, and the mollusc *Tellina agilis*. The top four ranked taxa at each station in 1994 accounted for more than 60% of all individuals, and were: Oligochaeta, *Protodorvillea gaspeensis*, *Tellina agilis*, Archiannelida, at EF; Oligochaeta, *Tharyx* spp., *Mediomastus ambiseta*, *Protodorvillea gaspeensis* at GN; *Aricidea catherinae*, Oligochaeta, *Mediomastus ambiseta*, *Tellina agilis* at IN; and *Mediomastus ambiseta*, *Aricidea catherinae*, Oligochaeta, *Tharyx* spp. at JC (Table 2). In most cases, these organisms were dominant taxa in both 2-unit and 3-unit operational periods.

Year to year shifts in dominant taxa at infaunal monitoring stations have been a common occurrence over the study period, with 1994 being no exception. For example, there was a large increase in relative abundance of the opportunistic polychaete *Mediomastus ambiseta* at GN, IN and JC in 1994, accounting for 18.7%, 10.3% and 30.0% of the individuals collected, respectively. Operational means for both 2-unit and 3-unit periods were on the order of 4-10% at these stations. Another dominant polychaete, *Aricidea catherinae*, exhibited notable shifts in abundance during 1994 relative to operational periods. However, these shifts were less consistent among stations than those mentioned previously for *M. ambiseta*. Increases in *A. catherinae* relative abundance were observed at IN and JC during 1994, with the most substantial increase at IN (24.6% of the total individuals collected during 1994, compared to approximately 7% for both operational periods). Conversely, *A. catherinae* contributed only 2.8% to total abundance at GN in 1994, compared to 19.6% and 13.1% for 2-unit and 3-unit periods, respectively. At EF, *A. catherinae* decreased from 4.1% during 2-unit operation to 1.6% during 3-unit operation, and was not among the dominants in 1994.

Similar spatial variability in relative abundance shifts was exhibited by other historical dominants in 1994, including oligochaetes, species of the polychaete genus *Tharyx* and the mollusc *Tellina agilis*. Oligochaete abundances in 1994 were higher than both operational period means at GN (31.4% vs. 20.8% (2-unit) and 23.1% (3-unit)) and IN (19.5% vs. 11.1 (2-unit) and 8.9% (3-unit)). Oligochaete relative abundance at JC in 1994 (16.3%) declined compared to operational period means, particularly the 2-unit period (40.8%). At EF, 1994 relative oligochaete abundance (40.4%) was higher than the 2-unit mean (32.9%) but lower than the 3-unit mean (51.4%). *Tharyx* spp.

TABLE 2. Mean relative abundance[a] (%) and coefficient of variability (CV[b]) for each of the ten most abundant taxa collected at the Millstone subtidal stations during 1994, 2-Unit operational years (1980-1985) and 3-Unit operational years (1986-1994).

	1994	2-Unit Period (1980-1985)		3-Unit Period (1986-1994)	
	%	MEAN %	CV	MEAN %	CV
Effluent					
Oligochaeta	40.4	32.9	4.4	51.4	1.5
Protodorvillea gaspeensis	8.1	4.6	13.3	8.1	3.6
Tellina agilis	7.8	2.9	17.9	3.0	21.6
Archiannelida	5.9	1.6	25.2	1.9	32.6
Prionospio steenstrupi	3.4	1.5	31.4	2.1	28.1
Microphthalmus aberrans	2.5	1.5	23.7	1.9	15.4
Spiophanes bombyx	2.3	1.5	25.3	1.8	25.7
Ampelisca vadorum	2.2	1.5	29.1	1.6	28.1
Parapionosyllis longicirrata	1.7	1.6	18.8	2.1	12.9
Pagurus acadianus	1.6	1.6	26.2	2.1	12.0
Rhynchocoela	-	2.4	12.2	3.0	14.3
Tharyx spp.	-	9.0	16.9	2.7	19.8
Polycirrus eximius	-	10.8	10.2	3.0	25.3
Aricidea catherinae	-	4.1	20.6	1.6	26.7
Ampharete americana	-	-	-	1.6	21.6
Scoletema tenuis	-	-	-	1.9	14.7
Clymenella mucosa	-	-	-	1.4	35.2
Giants Neck					
Oligochaeta	31.4	20.8	3.9	23.1	2.3
Tharyx spp.	25.0	20.2	2.2	19.3	4.4
Mediomastus ambiseta	18.7	4.8	27.5	5.1	22.9
Protodorvillea gaspeensis	3.0	3.8	5.5	3.8	8.3
Aricidea catherinae	2.8	19.6	5.0	13.1	8.0
Exogone dispar	2.8	2.7	19.6	3.2	6.4
Prionospio steenstrupi	1.6	2.0	16.1	3.8	25.7
Tellina agilis	1.3	1.5	25.8	1.6	32.1
Scoletema tenuis	0.8	3.1	14.1	3.3	9.1
Ampelisca vadorum	0.6	1.2	18.5	2.6	23.3
Capitella spp.	-	1.8	19.4	1.8	11.2
Ampharete americana	-	1.2	25.3	1.5	25.5
Pagurus acadianus	-	1.3	34.3	1.5	22.7
Polydora quadrilobata	-	-	-	1.7	28.0
Mitrella lunata	-	-	-	1.4	31.0

[a] Based on log-transformed data

[b] C.V. of the mean estimate = (Standard Error/Mean) X (100)

- = Not among the dominant taxa

TABLE 2, continued.

	1994	2-Unit Period (1980-1985)		3-Unit Period (1986-1994)	
	%	MEAN %	CV	MEAN %	CV
Intake					
Aricidea catherinae	24.6	6.7	18.2	7.0	16.2
Oligochaeta	19.5	11.1	9.4	8.9	10.4
Mediomastus ambiseta	10.3	4.0	29.6	4.5	23.4
Tellina agilis	6.7	4.3	15.9	4.2	15.8
Exogone hebes	5.6	3.9	23.1	5.4	18.2
Protodorvillea gaspeensis	4.4	1.9	23.4	2.8	19.6
Tharyx spp.	3.8	3.9	21.6	4.3	11.8
Microphthalmus aberrans	2.8	1.4	37.0	2.4	20.9
Nucula proxima	1.7	2.6	26.5	4.1	19.4
Prionospio steenstrupi	1.7	2.3	28.3	4.1	27.4
Capitella spp.	-	3.9	24.9	3.8	20.9
Ampelisca vadorum	-	2.4	37.1	3.0	20.3
Ampelisca verrilli	-	4.7	27.5	2.8	23.9
Spiophanes bombyx	-	2.8	24.9	2.2	15.4
Gammarus lawrencianus	-	2.1	42.1	1.5	49.4
Leptocheirus pinguis	-	-	-	2.7	39.5
Sabellaria vulgaris	-	-	-	1.3	57.6
Maldanidae	-	-	-	1.4	19.4
Owenia fusiformis	-	-	-	2.3	28.0
Jordan Cove					
Mediomastus ambiseta	30.0	7.2	26.3	9.6	15.3
Aricidea catherinae	21.6	14.8	6.1	16.6	7.3
Oligochaeta	16.3	40.8	3.9	22.3	3.7
Tharyx spp.	4.0	4.4	6.7	4.2	4.6
Tellina agilis	3.6	2.2	23.6	2.1	17.0
Prionospio steenstrupi	2.8	1.4	24.8	4.9	16.8
Scoletoma tenuis	2.7	4.7	13.1	5.6	6.4
Ampelisca vadorum	2.5	1.1	30.1	1.4	40.1
Nucula proxima	2.1	1.3	13.1	2.8	15.2
Exogone hebes	1.6	1.5	29.9	1.8	12.3
Leptocheirus pinguis	-	1.5	34.5	3.6	28.7
Capitella spp.	-	2.2	17.9	2.1	20.0
Polycirrus eximius	-	4.3	14.2	4.5	13.7
Microphthalmus aberrans	-	1.5	22.6	2.7	18.4
Cossura longocirrata	-	-	-	1.5	22.8

increased in relative abundance at GN during 1994 (25.0% compared to means of 20.2% and 19.3% for 2-unit and 3-unit periods, respectively). Abundance levels of *Tharyx* spp. in 1994 were more consistent with operational period means at IN and JC. *Tharyx* spp. relative abundance at EF decreased from 9.0% during 2-unit operation to 2.7% during 3-unit operation; this taxon was not among dominants in 1994. More subtle changes (2-5% increase) in abundance of the mollusc *Tellina agilis*, relative to operational period means, were observed at EF and IN in 1994, while 1994 abundance levels at GN and JC were comparable to their respective operational means.

Other organisms, noted historically for more site-specific dominance, exhibited notable shifts in relative abundance in 1994. Some examples include the polychaete *Polycirrus eximius*, which was among the dominant taxa at EF and JC during both operational periods, but was not among the dominants at either site in 1994. Similarly, two species of the amphipod genus *Ampelisca* (*A. vadorum* and *A. verrilli*), most typical of IN collections during both operational periods, were not dominants at that site in 1994.

Much less common than annual fluctuations in relative abundance were differences between operational periods. In fact, relative abundances of dominant taxa between operational periods have been consistent, with few exceptions. For example, oligochaetes continued to be the most common taxon overall during 3-unit operation (accounting for 8.9 to 51.4% of individuals), as they had been during 2-unit operation (11.1 to 40.8%); however, as discussed in the previous section, oligochaetes were more abundant at EF and less abundant at JC during 3-unit operation. Most stations were characterized by one or more clearly dominant taxa (oligochaetes at EF, GN and JC, *Aricidea catherinae* at GN and JC and *Tharyx* spp. at GN) during both operational periods. At IN, however, mean relative abundance of any single taxon rarely exceeded 10% during either operational period.

Dominant Taxa

Eight infaunal taxa have been identified as being affected or potentially affected by construction and operation of MNPS. Trends in the abundance of these taxa were examined using the same techniques as those applied to overall community abundance and numbers of species time-series. For a review of the general ecology of these dominant taxa, refer to NUSCO (1992).

Oligochaetes - Oligochaetes were ranked first in overall abundance during both 2-unit and 3-unit operating periods, accounting for 11-41% and 9-51% of total individuals, respectively (Table 2). Oligochaete abundances during 1994 were generally highest at GN (48-88/core) and EF (38-75/core), lowest at IN (21-26/core) and intermediate at JC (24-67/core). These densities were within the ranges of densities for previous study years (Fig. 6a-d).

Trend analysis of operational period oligochaete abundance time-series revealed an increase at EF, and a decline at GN during 2-unit operation (Fig. 6a and b). Trend analysis for the 3-unit period indicated a significant increase in oligochaete abundance at IN (Fig. 6c), and a significant decrease at EF. At GN and JC, oligochaete abundance has remained at a consistent level during the 3-unit operating period (Fig. 6b and d).

Aricidea catherinae - *A. catherinae* was among the dominant taxa at all stations during 2-unit operation, and at all stations, except EF, during the 3-unit operating period (Table 2). Quarterly average densities during 1994 were lowest at EF and GN; 1-3/core and 4-9/core, respectively. Highest quarterly abundance estimates occurred at JC (45-131/core), with intermediate abundances at IN (3-44/core). All 1994 densities at GN were lower than any recorded previously during the study period for that station (Fig. 6f), and December, March and June densities at IN were the highest observed at that site (Fig. 6g). Densities at EF and JC in 1994 were within the range of previous study years (Fig. 6e and h).

The average quarterly abundance of *Aricidea catherinae* exhibited several trends over both 2-unit and 3-unit operating periods (Fig. 6e-h). Relationships among operational period trends at EF and IN were similar; at both sites, abundance of *A. catherinae* significantly declined during 2-unit operation, and significantly increased after Unit 3 start-up. Abundance of *A. catherinae* also increased at JC during 3-unit operation. Conversely, abundances have significantly declined at GN during 3-unit operation, largely due to low densities in 1994.

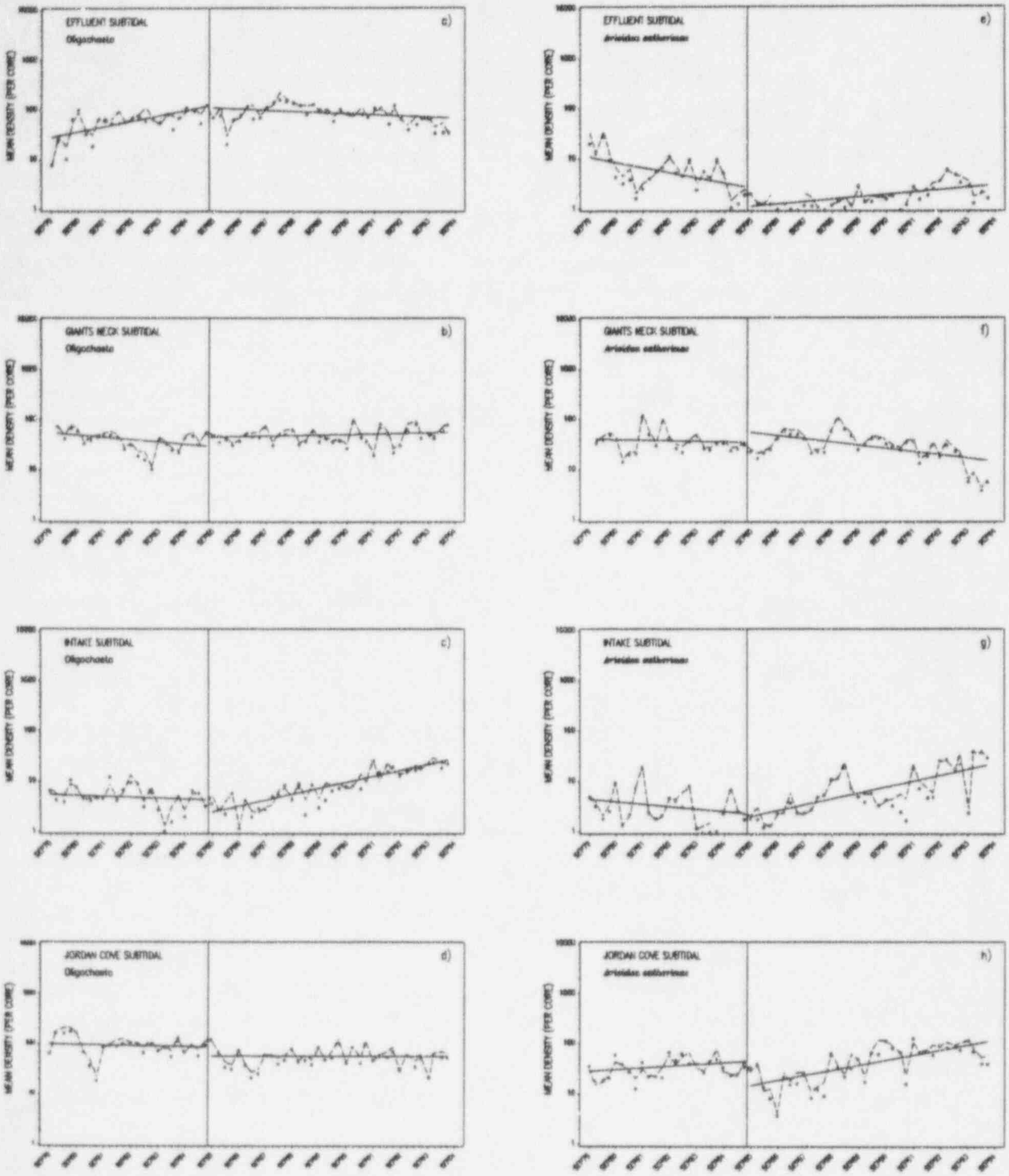


Fig. 6. Quarterly abundance data (dots), and variance-reduced data (dashed line), and linear-trends for selected dominant organisms comprising Millstone subtidal communities before and after Unit 3 operation at MNPS.

Mediomastus ambiseta - *M. ambiseta* exhibited a pulse in abundance at three of the four sampling sites in 1994 (Fig. 6i-k) similar to the area-wide pulse observed during the period 1984-87 at all stations (NUSCO 1992). Highest densities were recorded at JC in 1994 (77-131/core), followed by GN (27-52/core) and IN (7-17/core). None of the abundance trends at these stations during 2-unit operation were significant; however, in spite of increases in 1994, a significant decreasing trend, noted after 1993 at GN (NUSCO 1994), was still apparent with the addition of the most recent data. No similar pulse in *M. ambiseta* abundance was observed at EF, as this species was not among the dominant taxa collected there in 1994 (Table 2).

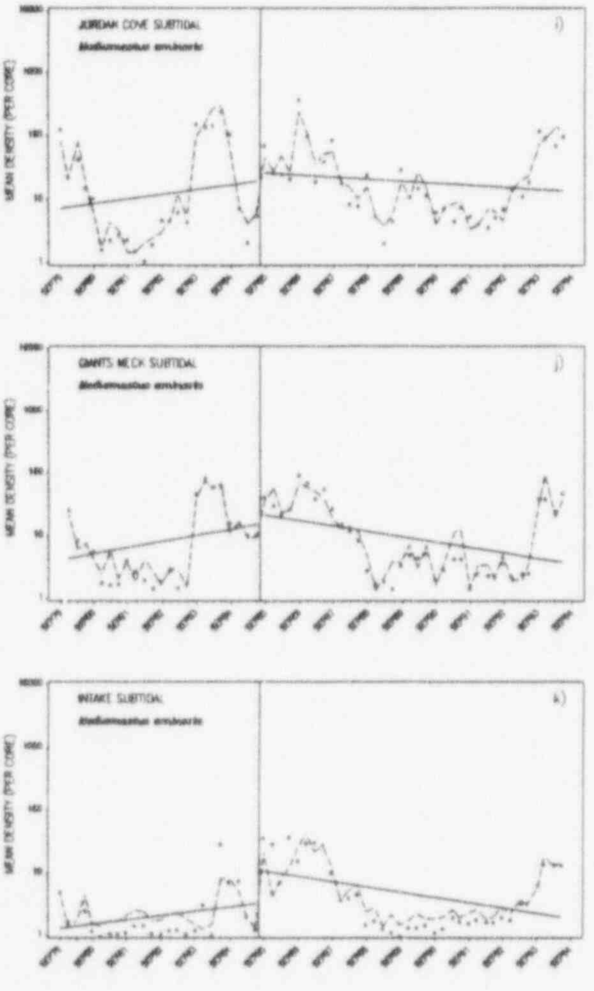
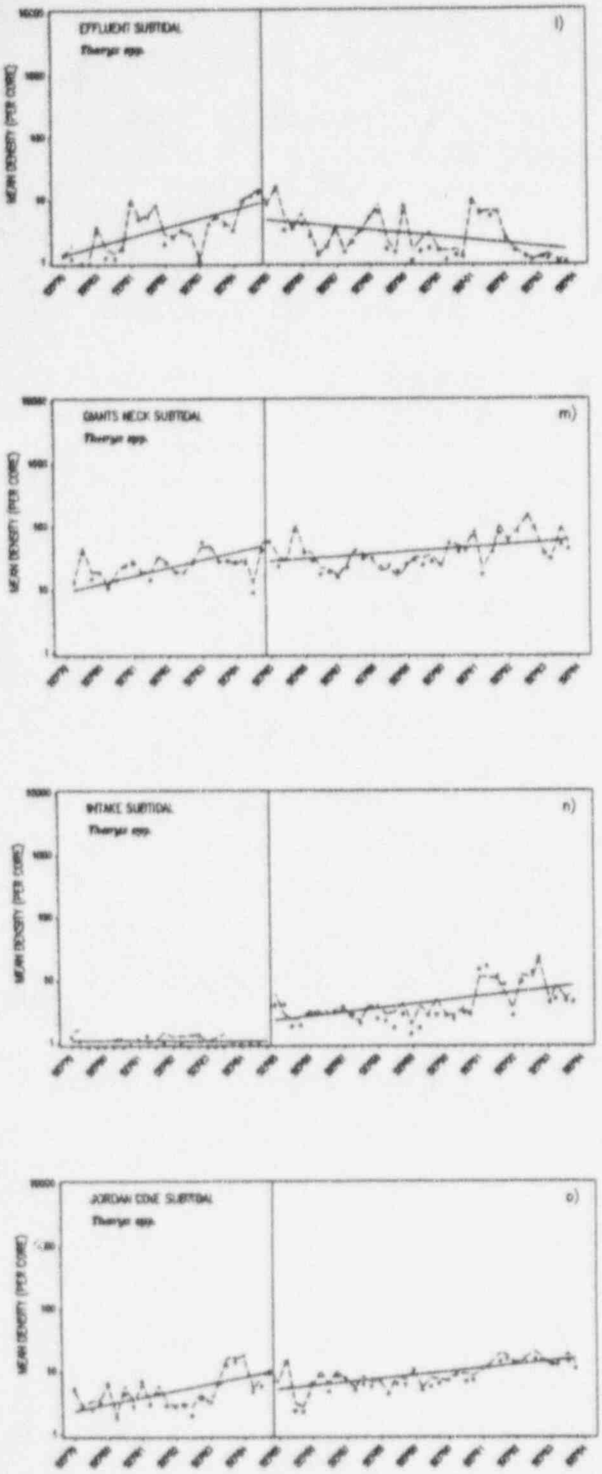


Fig. 6, continued.

***Tharyx* spp.** - *Tharyx* spp. were among the dominant taxa at most stations during 2-unit and 3-unit operating periods. *Tharyx* spp. exhibited high variability among sampling stations, with 1994 quarterly densities (Fig. 6l-o) lowest at EF (approximately 1/core), highest at GN (34-59/core) and intermediate at IN and JC (5-6/core and 13-16/core, respectively). At the GN reference station, *Tharyx* spp. densities have been consistently high, ranking second and during both operating periods and 1994 (Table 2). At IN and JC, *Tharyx* spp. ranking was also consistent, but at a lower level over the entire study period, including 1994. *Tharyx* spp. was not among the dominant taxa at EF in 1994, while ranking third and sixth during 2-unit and 3-unit operating periods, respectively.

Results of trend analysis on *Tharyx* spp. abundance during 2-unit operation indicated a significant increasing trend at GN and JC. Trend analysis of 3-unit operating data indicated significant trends in *Tharyx* spp. abundance at all sites, with a decrease detected at EF, and increases at the remaining sites.

Polycirrus eximius - *P. eximius* has been among the dominant taxa for 2-unit and 3-unit periods at two stations: EF (ranking second and third, respectively) and JC (ranking sixth in both periods; Table 2). This species was not among the dominant taxa collected in 1994, with quarterly mean densities of approximately 1/core for each sampling period at EF and a range of quarterly mean densities of 2-4/core at JC (Fig. 6p-q). These densities were among the lowest recorded at EF, and were also relatively low at JC. Trend analysis indicated significant increasing densities at both stations during the 2-unit operational period. However, significant increasing trends in *P. eximius* abundance at both sites during 3-unit operation noted in previous reports (e.g., NUSCO 1993, 1994) were no longer apparent with the addition of 1994 data. Historically, *P. eximius* has exhibited both seasonal periodicity and regional long-term cycles at all stations except IN (NUSCO 1993).



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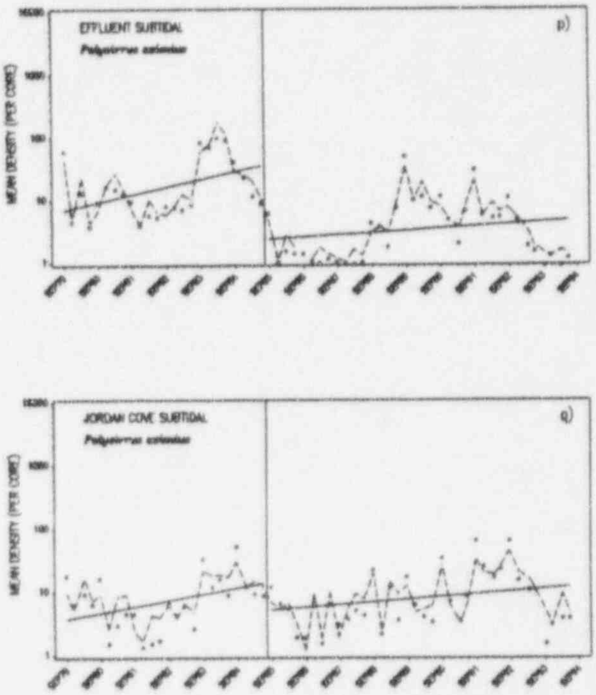
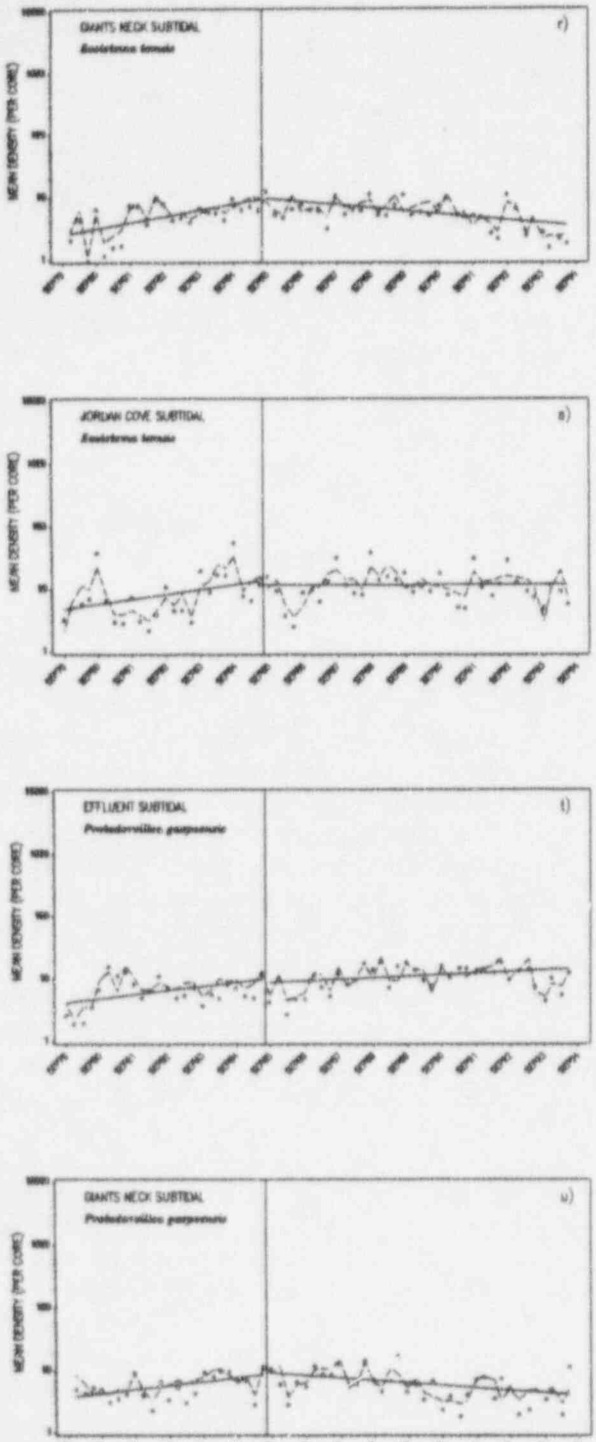


Fig. 6, continued.

Scoletema tenuis - *S. tenuis* was a dominant component of infaunal communities during the 2-unit and 3-unit operating period at GN (ranking sixth and seventh, respectively) and JC (ranking fourth for both periods; Table 2). Average densities during 1994 were 2-3/core and 6-13/core at GN and JC, respectively (Fig. 6r-s). These densities were within the range of density values from previous years. At both GN and JC, an increasing trend was observed during 2-unit operation, likely due to low abundances of *S. tenuis* at the beginning of the time series at both sites. Abundances continued to decline in 1994 at GN, with the significant decreasing trend for the 3-unit period, first noted last year (NUSCO 1994), still apparent. Abundances at JC have been more consistent during 3-unit operation, with no significant trend detected over that period.

Protodorvillea gaspeensis - *P. gaspeensis* was among the dominant infaunal organisms at EF, ranking fourth and second in the 2-unit and 3-unit operating periods, respectively (Table 2). *P. gaspeensis* was also a dominant at GN, ranking fifth during each operational period. *P. gaspeensis* rankings during 1994 were similar to those observed over both operational periods at EF and GN (second and fourth, respectively). Similarly, quarterly values were within the range of previously reported values, averaging 8-16/core at EF and 2-13/core at GN (Fig. 6t-u). Significant increasing trends occurred at EF and GN during the 2-unit operating period. Abundance of *P. gaspeensis* significantly decreased at GN during the 3-unit period; no trend was detected at EF during 3-unit operation. Until 1994, densities of this species were steadily increasing at EF, resulting in a significant trend (NUSCO 1993, 1994).

Nucula proxima - This small bivalve was a minor, but consistent, component of the infaunal communities at IN and JC over both operational periods, typically ranking sixth or lower (Table 2). *N. proxima* was among the top ten dominants at IN and JC during 1994; quarterly mean density ranges were 1-6/core and 5-12/core, respectively (Fig. 6v-w). Quarterly densities in 1994 were within the range of those recorded in previous years. No significant trends were observed at either IN or JC during 2-unit operation. During 3-unit operation, *N. proxima* exhibited a decreasing trend at IN, and conversely, an increasing trend at JC.



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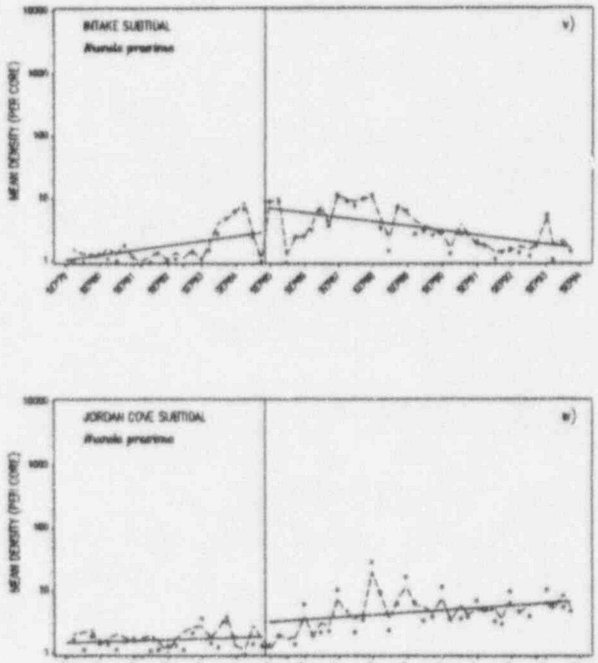

Fig. 6, continued.

Cumulative Abundance Curves

Cumulative abundance curves (Fig. 7) were used to characterize and compare infaunal communities at each station sampled over the 2-unit and 3-unit operational periods. At EF and JC, the location parameter (i.e., percent contribution of the top ranked organism) was significantly different between the two operating periods, reflecting changes in the overall contribution of the dominant taxon (oligochaetes), and the overall shape of the curves were significantly different between the 2-unit and 3-unit operating periods. There was no significant difference in the shape parameter between the 2-unit and 3-unit periods at GN or IN, and the location parameter was different at GN, but not at IN. This relative similarity reflects the consistent contribution of oligochaetes, *Aricidea catherinae* and *Tharyx* spp. at GN, and of oligochaetes, *A. catherinae* and *Mediomastus ambiseta* and *Tharyx* spp. at IN. The lower position of the 3-unit curve at JC reflects a shift toward increased equitability in the distribution of taxa. In contrast, the high starting point of the 3-unit EF curve indicates the numerical dominance of the top ranked taxon, oligochaetes, during that period. The low starting point of the IN curves in both operating periods indicate that no single taxon was overwhelmingly dominant in either period (see Table 2).

Classification and Cluster Analysis

In the previous sections, temporal trends in abundance of selected species at stations in the vicinity of MNPS were related to construction and operation of the power plant. Each of these species exists in a complex community, affected by competition, predation, and other biological and physical structuring factors. Therefore, development of the benthic infaunal community at each sampling station was examined by calculating the Bray-Curtis similarity index for each pair of annual collections, using the abundance of all species that contributed at least 0.5% to total abundance, then generating a clustering dendrogram for each station (Fig. 8).

At Effluent, three groups of annual collections were apparent at a 65% level of similarity. Group I consisted of collections from 1980 through 1983, characterized by high densities of *Aricidea*

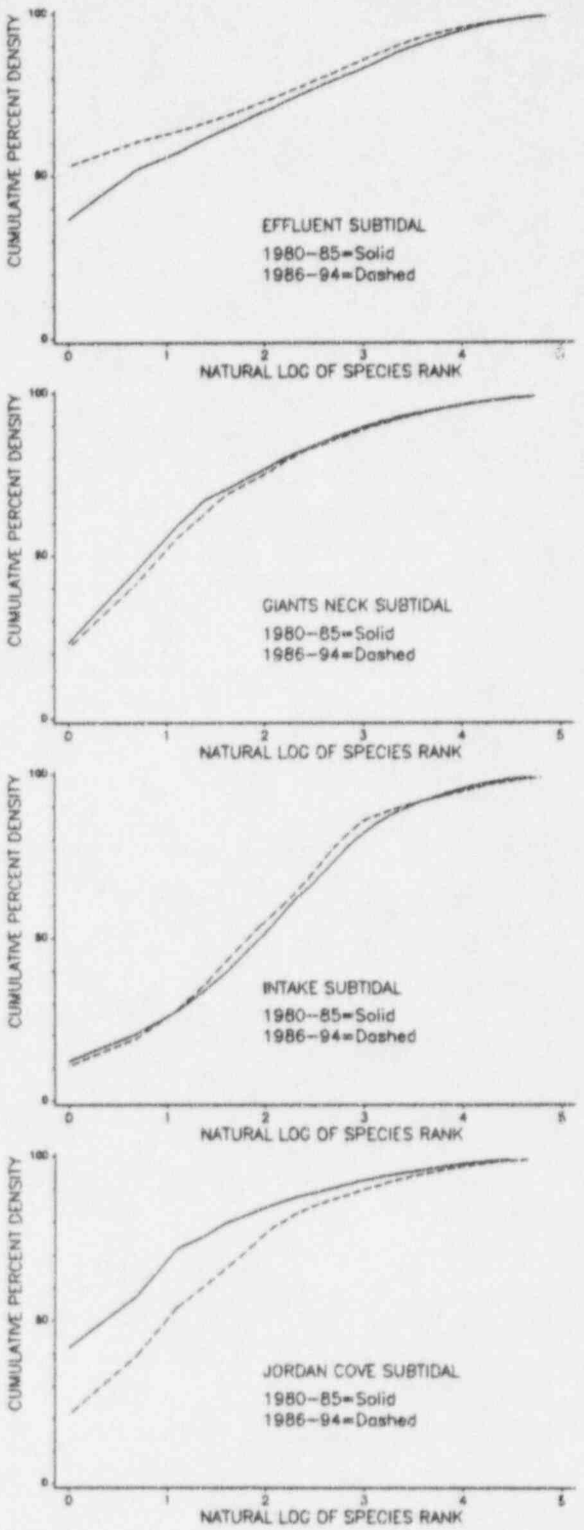

Fig. 7. Cumulative species abundance curves based on the ten most abundant organisms collected during the 2-unit (1980-1985) and 3-unit (1986-1994) operational periods at MNPS.

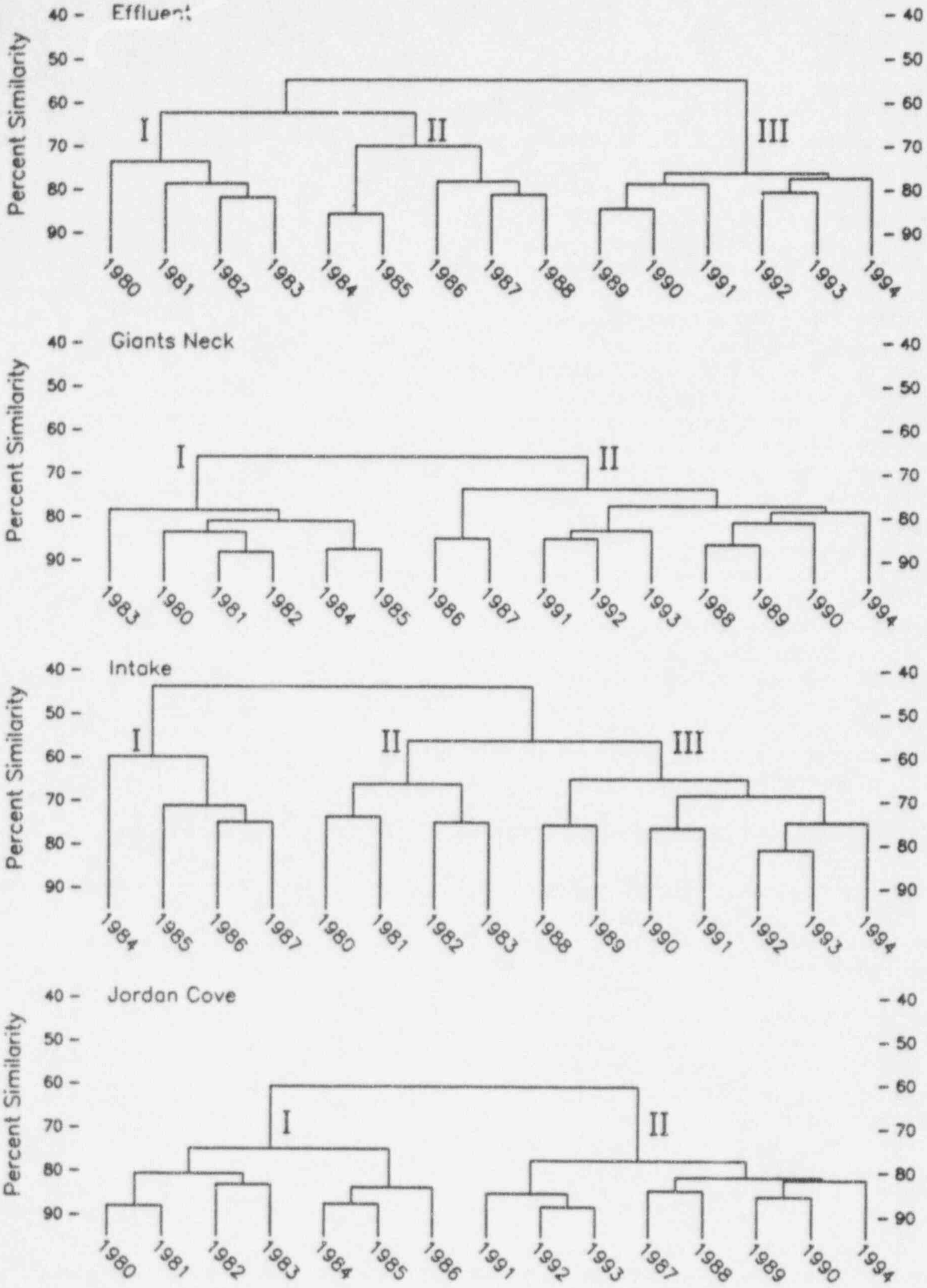

Fig. 8. Clustering dendrograms of natural log-transformed density data for annual infaunal communities at subtidal sand stations in the vicinity of MNPS.

catherinae and *Polycirrus eximius*, and relatively low numbers of oligochaetes. Group II (1984-1988) was characterized by decreased abundances of *P. eximius*, and increased abundance of oligochaetes, *Tellina agilus*, *Leptocheirus pinguis*, and *Ampelisca* spp. As discussed in other sections, this time period encompassed the period of maximum sediment scour associated with opening the second quarry cut and Unit 3 start-up. All annual collections made at Effluent since 1989 (Group III), including 1994, have exhibited high within-group similarity (>75%); the relatively stable environment in recent years has allowed development of a benthic infaunal community characterized by relatively high densities of *Parapionosyllis longicirrata*, *Prionospio steenstrupi*, *Protodorvillea gaspeensis* and *Caulleriella* spp.

Giants Neck, a reference site unaffected by MNPS operation, exhibited the highest among-group similarity, with all annual collections clustering at >65% similarity. At a 70% level, two groups were apparent; Group I (1980-1985) was characterized by high abundance of *Phoxocephalus holbolli*, *Polydora caulleryi* and *Polycirrus eximius*, and Group II (1986-1994), by high densities of *Polydora quadrilobata*, *Prionospio steenstrupi* and *Exogone dispar*, and decreased abundance of *P. holbolli*. Consistently high abundance oligochaetes and *Tharyx* spp. in all sample years contributed to the high among-group similarity.

In contrast, annual collections at Intake exhibited low among-group similarity (<45%), influenced by the dissimilarity of Group I (1984-1987) to the other years. The infaunal community in this period was strongly affected by the Unit 3 cofferdam removal and intake dredging, and was characterized by high abundance of the opportunistic polychaete *Mediomastus ambiseta*, and low densities of *Aricidea catherinae* and *Tharyx* spp. Collections from the years prior to this period (1980-1983; Group II) had included higher abundances of *A. catherinae* and *Exogone hebes*, and fewer *M. ambiseta*; collections from recent years (1988-1994; Group III) were also characterized by high densities of *A. catherinae* and *E. hebes*, but *M. ambiseta*, *Tharyx* spp. and oligochaetes were relatively abundant, as well.

Collections from Jordan Cove clustered into two groups at a 70% similarity level. Group I (1980-1986) was characterized by high abundance of *Mediomastus ambiseta* and oligochaetes, and low abundance of *Nucula proxima*, *Aricidea catherinae* and *Prionospio steenstrupi*. In contrast, Group II (1987-1994) was distinguished by relatively low densities of *M. ambiseta* and oligochaetes, and higher abundance of *N. proxima*, *A. catherinae* and *P. steenstrupi*. Additionally, *Leptocheirus pinguis*, *Scoletma tenuis* and *Protodorvillea gaspeensis* have become relatively more abundant in recent years. As discussed previously, the shifts in community structure were associated with deposition of sediment in Jordan Cove since Unit 3 began operation.

Discussion

Infaunal studies in 1994 continued to monitor soft-bottom community response to varying levels and types of impact associated with MNPS. The degree of impact at these sites can be described as occurring over a disturbance gradient. The critical baseline for this gradient, from which impacts can be assessed, is an unimpacted community (GN) responding only to naturally occurring environmental conditions. The next level of impact is community change in response to short-term disturbance events, such as dredging at IN and silt deposition at JC, after which post-impact community development and recovery under more natural conditions occurs. Impacts to the infaunal community at EF, resulting from long-term exposure to discharge scour and temperature effects, is continuous and ongoing, and defines the disturbance extreme for these studies.

Classification of local infaunal communities on this impact disturbance scale (i.e., unimpacted to continually influenced) was accomplished through identification of structuring mechanisms that produce characteristic fluctuations in species composition, abundance and dominance. However, the first step in this process, i.e., separation of naturally induced physical and biological mechanisms, including naturally varying levels of mortality, recruitment, competition and vagaries in local physico-chemical conditions is often difficult (Watling 1975; Flint and Younk 1983; Nichols 1985; Watzin 1986; Rees and Eleftheriou 1989). An attempt has been made here (through modeling and regression analysis) to more accurately separate and assess the contribution of some natural factors reported to cause differences observed in local benthic communities, and thereby help isolate other factors possibly related to construction and operation of MNPS.

Fluctuations in sediment characteristics and community composition have been observed at all study sites to some degree, with the highest overall stability in these features noted at GN. Aside from the high silt/clay content in March (and concomitant low mean grain size), sediments collected at GN in 1994 were similar to previous years. Similarly, overall community composition at GN has been remarkably consistent over the study period; this site was numerically dominated by the same five taxa (oligochaetes, *Tharyx* spp., *Aricidea catherinae*, *Mediomastus ambiseta*, and *Protodorvillea gaspeensis*) in generally the same rank order and relative abundance during both 2-unit and 3-unit operational periods (Table 2).

Along with providing documentation of long-term stability of local infaunal communities under natural conditions, data from GN have proven useful in substantiating area-wide shifts in species abundance and community structure. These included large increases in abundance of the opportunistic polychaete, *Mediomastus ambiseta*, and the amphipods, *Leptocheirus pinguis* and *Ampelisca* spp. which occurred over several years (1983-88; NUSCO 1989), as well as annual pulses in species abundance such as occurred for the spionid polychaete, *Prionospio steenstrupi* in 1992 (NUSCO 1993). These increases could not be explained by changes in site-specific sedimentary or regional climatic factors (NUSCO 1989, 1993); however, because these changes occurred at all stations, including the reference site GN, their cause was assumed to be independent of power plant construction or operation. Similarly, a notable pulse in abundance of *M. ambiseta* occurred again in 1994 at GN, JC and IN, indicating an area-wide phenomenon.

Power plant related impacts on infauna were observed at IN, JC and EF. The impacts at JC and IN were of short duration (months or years), and their overall effects have lessened over most of the 3-unit operating period. Impacts were noted earliest at IN, and were associated with disturbances resulting from dredging and coffer dam removal during Unit 3 construction from 1983-85 (NUSCO 1987). Since that time, sediments (primarily silt/clay content levels; Fig. 2) have stabilized and have become more similar to pre-impact years, and ongoing community recovery is evident. In particular, increases in the abundance of organisms common prior to 1983 (e.g., oligochaetes, *Aricidea catherinae*; Fig. 6c and g), with concomitant decreases in abundance of *Nucula proxima* (Fig. 6v) and other opportunistic species (NUSCO 1993) indicate that a recovery process has continued through 1994. However, other species which have established post-impact populations, such as *Tharyx* spp. (Fig. 6n), maintained a degree of dominance in 1994, which suggests that recovery at this site is not complete. Changes such as these are typical of those in marine benthic communities following disturbance (Kaplan et al. 1974; Swartz et al. 1980; Nichols 1985; Berge 1990).

Relatively rapid changes in sediment characteristics were also observed at JC following Unit 3 start-up, resulting in changes to the infaunal community there. In 1986, silt scoured from the area of the Unit 3 discharge settled at JC, increasing silt/clay content of sediments in this area (Fig. 2). These substratum changes resulted in decreased abundances of the previously dominant oligochaetes and the polychaetes, *Polycirrus eximius* and *Aricidea catherinae* (NUSCO 1988a). This depositional event likely occurred over a relatively short period (months), and its impact has evidently lessened since 1986. For example, abundance of *A. catherinae* continued to increase in 1994 to levels comparable to those observed during 2-unit years (Fig. 6g). Other possible signs of community recovery discussed in previous reports (e.g., rebounding abundances of oligochaetes and *P. eximius* through 1993; NUSCO 1994) were less evident after 1994 sampling (Table 2; Figs. 6d and 6q). Lower abundances of these two taxa in 1994 were probably due to the long-term persistence of some of the deposited silt/clay at JC, or to new siltation resulting from shut-down/start-up cycles of the MNPS units. Another indication of slow community recovery is the continued high abundance of the opportunistic mollusc *Nucula proxima* relative to pre-impact levels (Fig. 6w), a trend similar to that noted previously at IN. These observations are consistent with those of other researchers studying the effects of siltation on benthic infaunal communities (Rhoads and Young 1970; Jumars and Fauchald 1977; Turk and Risk 1981; Mauer et al. 1986; Emerson 1989; Brey 1991). Another notable trend in 1994 at JC is a substantial decrease in silt/clay content in the last two sampling quarters (March and June; Fig. 2), with the June value the lowest recorded in the 3-unit period. Further monitoring should document any effects this trend, if it continues, has on future infaunal community development and recovery.

Active disturbance processes continued to affect both the sediments and the infaunal community at EF. Increased grain size and decreased silt/clay levels were still evident at EF during 1994. However, it appears that in recent years, sediment, community and species parameters have stabilized under the new environmental conditions created by the 3-unit discharge. Silt/clay content levels in EF sediments were the highest observed since scouring effects were first noted and were similar to levels observed during the 2-unit period. Oligochaete abundance has generally dropped over the 3-unit period, including 1994, and is now approaching levels observed during the 2-unit period (Fig. 6a). Some relative stabilization of the sediments at EF has allowed for the return of species common during 2-unit operation, such as *Aricidea catherinae* and *Polycirrus eximius* (Fig. 6e and p). The decline at EF in 1994 of these two polychaete species, which typically maintain more stable populations at other sites, was likely due to the overall dynamic nature of environmental conditions, created primarily by the MNPS discharge, which continue to affect the infaunal community there. Another possible indication of strong influence of power plant operation on environmental conditions at EF is the absense of the opportunistic polychaete *Mediomastus ambiseta*. This species essentially exhibited an area-wide pulse, related to regional natural environmental cues that were apparently overridden by plant discharge effects at EF. The previous pulse of *M. ambiseta* was initiated in 1984 (prior to scouring related to the 3-unit discharge), and was apparent at all stations, including EF. The community at EF should continue to develop under and adjust to current operating conditions, but true recovery of this community is not expected until power plant-induced scour ceases.

Conclusions

Benthic infaunal studies continued to monitor subtidal soft bottom habitats in the vicinity of MNPS during 1994 for changes in sedimentary characteristics and infaunal community structure (total abundance, species number and species composition). The unimpacted site at GN continues to exhibit stability in terms of the sedimentary environment and infaunal community structure. Some stability in these parameters has also been noted in recent years, including 1994, at sites impacted by short-term episodic disturbance events (e.g., dredging and construction activities at IN, and siltation at JC). Community recovery is ongoing, but not complete at either site. Long-term continuous effluent scour at EF remains a dominant structuring factor on both the sedimentary environment and infaunal community, and is expected to maintain this dominant role throughout MNPS 3-unit operation.

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