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

1996 Annual Report

Prepared by: Staff of

Northeast Utilities Service Company

Environmental, Health & Safety Services

NU Environmental Laboratory

Approved by: _____

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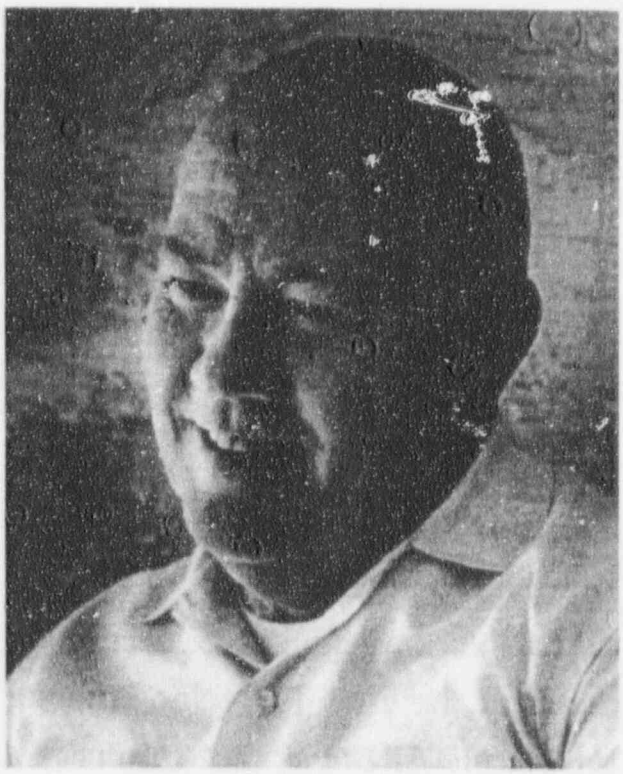

Dr. Nelson Marshall, Professor Emeritus of Oceanography
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Dedication

Nelson Marshall has spent a lifetime in the study of the Niantic River and its environs. That study began as a boy when Nelson and his family summered on the River, but for the past 40 years Nelson's study of the River has been as a professional scientist. In the early 1960s Nelson, several colleagues from Connecticut College and graduate students from the University of Rhode Island began an ecological examination of the Niantic River that focused on primary production and scallop ecology. In the mid 1960s when Northeast Utilities selected Millstone Point for location of three nuclear power plants it naturally turned to Nelson for advice on marine ecology. Thus began a relationship between Nelson and NU that has lasted for more than 30 years. Nelson was the Chairman of the Millstone Ecological Advisory Committee until 1984 when, following his retirement as Professor of Oceanography and Marine Affairs at URI, he and his wife Grace moved to the Maryland eastern shore. Fortunately for NU, Nelson has remained a vital member of the Ecological Advisory Committee to this day.

Following receipt of his Ph.D. from the University of Florida in 1941, Nelson's career took him to such institutions as the University of Miami, UCONN, William and Mary, and Alfred University. He was at URI from 1959 to 1984. Honors include Fellowship status in the American Association for the Advancement of Science and Honorary Membership in the Atlantic Estuarine Research Society. His research interests, which include coral reef and mangrove ecology and marine resource development, in addition to estuarine ecology, have taken him to Fiji, Malaysia, the Marshall Islands, the Caribbean and elsewhere. He has produced many graduate students who have gone on to excellent careers in marine science thanks to him.

In recognition for the 30 years that Nelson Marshall has served NU as ecological advisor and friend, the scientific staff of the Northeast Utilities Environmental Laboratory and the members of the Millstone Ecological Advisory Committee dedicate this 1996 Annual Report to him.

Executive Summary

Fish Ecology Studies

Studies of fish assemblages in the vicinity of MNPS were conducted to determine the effects of station operation. These effects have been defined as station-related changes in the occurrence, distribution, and abundance of fishes, which can alter community structure. Potential effects include the entrainment of early life history stages through the cooling-water system (probably the most important effect), impingement of juvenile and adult fish on the intake screens, which was mitigated by the installation of fish return sluiceways at Units 1 and 3, and changes in distribution in Jordan Cove as a result of the thermal discharge.

Trawl, seine, and ichthyoplankton (fish eggs and larvae) monitoring programs were established in 1976 to provide information for the assessment of impacts from MNPS operation. These programs provided the basis for identifying taxa potentially affected, as well as information on long-term abundance trends used to measure changes in the local populations. About 130 different fish taxa have been collected in these monitoring programs. Of these, six taxa, including American sand lance, anchovies, silversides, grubby, cunner, and tautog, were identified as having the potential to be impacted by MNPS, either by entrainment or exposure to elevated water temperatures from the plant discharge. Abundance data were analyzed separately for the two-unit (1976-85) and three-unit (1986 through 1995 or 1996, depending upon the sampling program) operational periods and for the entire 20-year data series (both periods combined) to determine if changes in abundance have occurred.

American sand lance larvae ranked third among entrained fish larvae and densities in entrainment samples have decreased after peaking in the late 1970s and early 1980s. Declines in sand lance abundance were also apparent in other areas of the Northwest Atlantic Ocean, with abundance found to be inversely correlated with that of Atlantic herring and Atlantic mackerel, both of which prey upon larval sand lance. However, abundance has increased again in recent years, although the mean larval density during 1995 was the lowest of the past 4 years. Given the large changes in abundance of this fish along the Atlantic coast, effects of MNPS on sand lance abundance is difficult to ascertain, but is probably small.

The bay anchovy is typically the most abundant ichthyoplankton species collected in estuaries within its range and was the dominant larval taxon entrained at MNPS. Recent abundance has been relatively low in comparison to the mid-1980s, but this decline occurred prior to three-unit operation and no significant trends were found for either two- or three-unit operation. The egg and larval densities and the entrainment estimate for 1995 were among the lowest of the past 20 years. The numbers of eggs and larvae entrained each year were not significantly correlated with densities found the following year, implying no direct effect of MNPS on the spawning stock of this short-lived species.

Atlantic and inland silversides are among the most common shore-zone species along the Connecticut coast. These species fluctuate in relative abundance from year to year. Typical of short-lived species, the abundance of silversides is highly variable and annual catches by trawl and seine have ranged over two orders of magnitude. Recent catches of silversides by trawl and inland silverside by seine were within historic ranges. However, the Atlantic silverside has significantly decreased in abundance during the three-unit operational period at the Jordan Cove seine site. This decrease was probably not related to thermal effects, as only a minimal (0.8°C) increase in water temperature is found at the seine station, which is less than typical summer diurnal variation on the shallow sand flats. Because catches of adults by trawl during winter did not show similar changes in abundance, MNPS likely has not affected the local Atlantic silverside population

The grubby is unique because unlike other potentially impacted species it experiences no fishing pressure and has little forage value. Both larval and adult grubby abundance indices have been relatively stable throughout the 20 years of monitoring, suggesting little plant effect.

The most abundant of the fish eggs entrained were cunner eggs, which accounted for more than 50% of all eggs collected since 1979. During three-unit operation, cunner eggs increased in abundance, with the density of cunner eggs in 1995 the third largest recorded, as was the annual entrainment estimate.

However, the densities of larvae found decreased by about 50% from 1994. Young-of-the-year cunner accounted for a higher proportion (about 70%) of fish caught by trawl since three-unit operation began. Trawl catch at a station near the MNPS intakes showed a significant decline during two-unit operation. This decrease was most likely related to the mid-1983 removal of a rock cofferdam at the Unit 3 intake structure that provided habitat for cunner. Afterwards, catches became similar to another station in nearby mid-Niantic Bay. The entrainment of eggs is the greatest potential impact of MNPS on the cunner population. However, if egg losses affected recruitment, then juvenile abundance should decrease in relation to older fish. This decrease was not apparent in the length-frequency distributions, and relative abundance of juveniles actually increased during the three-unit operational period.

The tautog was the second-most abundant egg taxon entrained, accounting for over 27% of the total eggs collected since 1979, with mean density in 1995 the largest seen since 1990. Tautog larvae, however, were not as prominent, ranking eighth in abundance. No correlation was found between eggs and larvae and no significant trends in abundance were found during either operational period. Catches of tautog by trawl were dominated by young-of-the-year. In contrast to 1994-95, when total trawl catch was the lowest in 20 years, the catch during 1995-96 was the largest. Tautog, particularly fish in size-classes that correspond to ages-3 through 5, were also taken in lobster pots. The 1996 catch at the Jordan Cove station was six times the previous high. The reasons for this increase are unclear, but may have been related to changes in prey availability.

Special studies on tautog eggs showed that large (65-80%) decreases in egg abundance occur following early evening spawning through the following morning, most probably a result of high natural mortality. Pelagic tautog eggs disperse rapidly from spawning sites by tidal transport and densities in nearshore areas are relatively uniform. Based on hydrodynamics, a conservative measure of the source area for eggs entrained at MNPS includes a radius of about 5 nautical miles. Two daily estimates of the instantaneous standing stock of tautog eggs within this area equaled or exceeded annual entrainment estimates at MNPS and, in fact, would have been even larger if high egg mortality rates had been taken into account. This implies that MNPS entrainment effects may be relatively small.

If egg losses due to entrainment affected recruitment of tautog, then juvenile abundance should also decrease and the relative abundance of older fish would appear to increase in the short term. Based on length-frequency distribution from trawl catches, the percentage of juvenile tautog increased during the three-unit operational period. Therefore, changes in the relative proportion of juveniles and adults were probably unrelated to entrainment losses. In addition, the decline in juvenile and adult tautog abundance in Long Island Sound that began in the mid-1980s coincided with the decreasing numbers of eggs collected at MNPS. If the decrease in adult numbers was caused by entrainment losses, then the reduction in egg abundance should have lagged the decline of juveniles by several years because females do not mature until age-3 or 4. Therefore, the lower abundance of tautog eggs was probably due a decline in the abundance of spawning adults from fishing rather than the operation of MNPS. At present, tautog stocks are considered overfished and because of the long life and slow growth of this species, abundance should remain depressed until fishing mortality rates are substantially reduced.

Winter Flounder Studies

The local Niantic River population of 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, extensive studies of the life history and population dynamics of this valuable sport and commercial species have been undertaken since 1976.

In contrast to the previous 2 years (1994-95), when monthly mean seawater temperatures were warmer than average, temperatures recorded at the MNPS intakes during 1996 were among the coolest of the past 21 years, particularly during spring and summer. The cold weather produced heavy ice cover in the Niantic River, delaying the start of the adult winter flounder survey until February 27. The Δ-mean trawl catch-per-unit-effort (CPUE) of fish larger than 15 cm during the spawning season was 1.6, the lowest of the series. Larger females have made up a greater proportion of the spawning stock in recent years as abundance declined to low levels. The Jolly stochastic model was applied to mark and recapture data to estimate the absolute abundance of

the adult spawning population. The abundance estimate for 1995 was 5,574 winter flounder, lower than the estimates of about 10-16 thousand for 1992-94 and considerably less than estimated population sizes during 1984-91 that ranged between 33 and 80 thousand spawners. One-third to almost two-thirds of the winter flounder found in the Niantic River during the spawning period each year were mature females. Female spawner abundance estimates ranged from 2,427 (1996) to 68,899 (1982), with corresponding total egg production from about 2.1 to 39.9 billion each year.

The low abundance of newly-hatched larvae in Niantic Bay compared to the Niantic River suggested that most local spawning occurred within the river. In addition, abundance indices of Stage 1 larvae in the river were significantly correlated with independent estimates of female spawner egg production. Densities of Stage 1 and 2 larvae in the Niantic River during 1996 were about average, except for Stage 2 larvae at a station in the upper river, which was the second highest of the series. However, abundances of Stage 3 and 4 larvae this year were at or below average at all stations. Since 1976, annual larval abundances in Niantic Bay appeared to reflect region-wide trends as they were highly correlated with abundance indices for Mount Hope Bay, MA and RI.

Smaller size-classes of larvae were dominant in the river and larger size-classes were more prevalent in the bay. The reduced cooling-water flow in 1996 resulted in larger catches of smaller larvae in MNPS entrainment samples, which could have been the result of reduced net extrusion under low flow and slower water velocity conditions.

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). Growth and development were slower than average in 1996, likely due to cooler water temperatures. Estimated mortality of larvae in the Niantic River for 1984-95 ranged from about 82 to 98% and was 94.8% in 1996. Density-dependence was examined using a function comparing mortality with egg production estimates (a measure of early stage larval abundance) and various monthly and seasonal water temperatures. The best model indicated that larval mortality increased as egg production increased and spring (April-June) water temperatures decreased.

Densities of newly metamorphosed demersal young were relatively low in 1996. Young winter flounder were particularly scarce during late summer and the median beam trawl CPUEs were among the lowest recorded since this sampling began in 1983. The Δ-mean CPUE calculated for young winter flounder taken during the late fall and early winter at the trawl monitoring program stations was 4.8 in 1995-96, the lowest value since 1976-77. This low abundance was unexpected, given the relatively high numbers of young produced in the 1995 year-class. This abundance index was significantly correlated with that of young fish taken in the Niantic River during 1994, and also indicated that the 1988 and 1992 year-classes were relatively abundant, whereas the 1993 year-class was weak. Few juveniles have been taken within the Niantic River during the adult spawning population surveys in recent years. Young-of-the-year abundance indices were either not correlated or were negatively correlated with the abundances of age-3, 4, and 5 female adult spawners. Thus, none of the early life stages was a reliable index 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, an indirect estimate of the winter flounder theoretical rate of increase (the SRR α parameter) was used for modeling winter flounder population dynamics for impact assessment. The value of α in biomass units was estimated as 5.87. The estimate of β (the second SRR parameter), which describes the annual rate of compensatory mortality as a function of stock size, has shown little annual variation since 1988. The third 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 both larval densities in Niantic Bay and plant operation. The entrainment estimate in 1996 was 53.9 million larvae, the second lowest since three-unit operation began in 1986. This was largely

attributed to plant operation as cooling-water volume during 1996 was the lowest in the three-unit operational period, with Units 1 and 2 shut down for all and Unit 3 for most of the larval winter flounder season. The decrease in cooling water use resulted in a calculated reduction in entrainment of about 72% (138 million larvae) from that expected if all three units had operated fully during the season.

The impact of larval entrainment on the Niantic River stock depends upon the fraction of the winter flounder production entrained each year. Empirical mass-balance calculations for 1984-96 showed that a large number of entrained larvae come from areas of Long Island Sound other than the Niantic River. In previous years, an estimated 14 to 38% of entrained larvae originated from the Niantic River, but the estimate for 1996 was 59%. On the other hand, the fraction of the annual river production entrained, which has ranged from 5.4 to 42.3% in previous years, was a moderate 25.7% in 1996.

A stochastic computer simulation model (SPDM) was used for long-term assessments of MNPS impact over a 100-year period (1960-2060). The winter flounder stock simulated was female spawner biomass (lbs), which is more directly related to reproductive potential than fish numbers. Conditional mortality rates corresponding to larval entrainment from mass-balance calculations and juvenile and adult impingement at MNPS were simulated according to historical information and projections; natural and fishing mortality rates (F) were provided by CT DEP. For simulation purposes, F was initially set at 0.40 in 1960 and reached a maximum of 1.33 in 1990. Based on proposed changes in fishing regulation, F was projected to decrease substantially over the next decade to 0.60 by 2006 and remain unchanged thereafter.

In the SPDM simulation, an initial stock size of 113,415 lbs was used to represent the theoretical (no fishing effects) maximum spawning potential (MSP) of the Niantic River female spawning stock. When fishing was added, the annual projections of the initially unfished stock become the baseline time-series of annual spawning biomass in the absence of any plant impact. Under the exploitation rates simulated, as provided by the CT DEP, the stochastic mean stock size of the baseline declined to 56,243 lbs by 1970 and to its lowest point of 12,880 lbs in 1993. The latter value was less than one-half of a critical stock size, defined as 25% of MSP. Following simulated reductions in fishing, however, the stock rapidly recovered. 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 MNPS impact again occurred in 1993 (10,604 lbs), whereas the greatest absolute decline relative to the baseline occurred in 2000 (a difference of 18,682 lbs). Generally, greater reductions in stock biomass resulted from fishing than from larval entrainment. The simulated spawning stock returned to within about 1,700 lbs of baseline levels in 2030, only 5 years after the scheduled termination of Unit 3 operation in 2025, and became virtually identical to the baseline in 2033.

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, stocks were likely ($p = 0.92$) greater than 40% of MSP in 1970. At the lowest point of both stock projections in the mid-1990s, all replicates of the stock projections were less than 25% of MSP. Simulated reductions in fishing allowed for a rapid increase in spawner biomass in 2000. By 2010, spawner biomass of the impacted stock was likely ($p = 0.91$) greater than 30% of MSP and had a probability of 0.42 of being larger than 40% of MSP.

This simulated recovery, however, assumed that changes in fishing regulations were implemented as scheduled and that they achieved reductions in fishing mortality rates as expected. Even with substantial reductions in fishing mortality and termination of MNPS operation, the probabilistic analysis indicated a one in three chance that the new equilibrium stock biomass would still be smaller than 40% of MSP after 2040. To date, however, the Niantic River winter flounder stock has not shown evidence of a rebound in abundance as suggested by the model. Even though fishing remains high, this population has remained resilient and very small adult spawning stocks in recent years have produced relatively large year-classes of young fish. Nevertheless, continued efforts in reducing fishing are necessary to ensure a recovery and avoid a stock collapse.

Lobster Studies

The total number of lobsters caught (all sizes) and total CPUE in 1996 was within ranges of previous years; however, CPUE of legal lobsters landed in 1996 was the lowest observed in the 3-unit period. This decline was expected because total CPUE in 1995 was the lowest observed in the nearly 20 year study period, so fewer sublegal-size lobsters were available to molt to the legal size class in 1996. There has been an overall decline in legal lobster abundance since 1978, primarily due to increased fishing rates, which have more than doubled since 1978, and to increases in minimum legal size implemented in 1989 and 1990.

Lobster catches and molting peaked earlier during the overall 3-unit period (1986-95) than during the 2-unit period (1978-85), probably owing to the regionally warmer May to August water temperatures observed in recent years. Consistent with this finding, cooler than normal water temperatures in 1996 delayed lobster catch and molting peaks. Other changes in local lobster population characteristics during 3-unit operation were related to implementation of new fishery regulations, rather than to power plant impacts. The increased proportion of berried females is associated with the increases in minimum legal size, and should increase larval production as a larger proportion of females are able to spawn before reaching legal size. Similarly, implementation of the escape vent regulation in 1984 has led to lower percentage of lobsters missing one or both claws (culls) during 3-unit operation.

The total estimated number of lobster larvae entrained through the MNPS cooling water systems during 1996 was the lowest reported since entrainment studies began in 1984. This reduction in entrainment of larvae was the result of the shutdown of MNPS during 1996. For the most part, entrainment levels have been considerably higher during 3-unit operational years relative to 2-unit years, due the additional cooling water demand of Unit 3. The long-term impact of larval entrainment at MNPS is difficult to quantify because the source of larvae entrained at MNPS is not known, and larval survival, settlement and ultimate recruitment to the fishery are not fully understood. Since lobsters require 4-5 years of growth before they become vulnerable to capture in our traps and an additional 2 years of growth to reach legal size, continued monitoring of lobsters will demonstrate the effects, if any, of 3-unit operations on the local lobster population.

Rocky Intertidal Studies

Differences among rocky intertidal stations in community composition were attributed to site-specific environmental conditions created by the influence of many interacting factors. At three of four study stations, major differences among communities (e.g., based on abundances of the dominant taxa such as barnacles, *Fucus* and *Chondrus*) were attributed to natural variability in factors that affect the degree of wave disturbance at each site. These factors include site orientation to prevailing wind-generated waves, the ability of exposed substratum (slope) to dissipate the horizontal force of those waves, and the character of that substratum (e.g., boulders, bedrock ledge).

In addition to these natural factors, impacts related to the MNPS thermal plume have created a distinctive intertidal community on the shore area immediately adjacent to the discharge outfall to the east (Fox Island). The unique algal flora at Fox Island-Exposed (FE), developed under elevated temperature conditions caused by the 3-unit thermal plume, continued to be evident in 1995-96 based on qualitative sampling. There was little change to the FE flora resulting from the extended 3-unit outage in 1996. The most notable shifts in species occurrence at FE during 3-unit operation, relative to unimpacted sites, 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.

Dominant species abundance patterns were altered by 2-cut water circulation patterns and by 3-unit operations only at FE. These changes were most notable in the low intertidal zone at FE, 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*. These populations maintained dominance within the FE intertidal community during 1996, and small populations of *Sargassum* and *Gracilaria*, found only in FE study transects, also persisted.

Elevated temperatures (2-4°C above ambient) at our *Ascophyllum* station nearest the discharge (FN), coupled with higher than normal ambient temperatures, may have created unfavorable conditions for *Ascophyllum* growth in 1995-96. *Ascophyllum* growth was significantly reduced at FN in 1995-96, relative to stations farther away. Owing to the high degree of variability associated with *Ascophyllum* growth, it is not certain whether this pattern of reduced growth is related to the temperature regime. In contrast, thermal incursion in most previous years caused growth enhancement at FN. Consistent with 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.

Eelgrass

Eelgrass (*Zostera marina*) monitoring studies during 1996 revealed relatively healthy study populations at current sampling sites. The study sites nearest to MNPS, Jordan Cove (JC) and White Point (WP), have supported stable populations since the study began in 1985. The present study site in the Niantic River (NR #4) has only been monitored since 1995; general decline of the overall population in the Niantic River has necessitated relocation of the study site three times over the 12-year study period. Variability in population parameters has been observed to some extent at all study sites and was primarily attributed to factors unrelated to MNPS operation.

Variability in eelgrass abundance and standing stock has been greatest in the Niantic River. Since 1985, this population has been characterized by isolated, often transient patches. The number and extent of these patches has declined since early study years, as eelgrass at three previous study sites has been eliminated. Eelgrass recolonization through seed germination has only been observed once, at the original study site (NR #1); this new bed declined and was eliminated after only two years. Causes for the instability of eelgrass in the Niantic River are unclear, but not related to power plant operation because this site is well beyond the zone of influence of MNPS. No indications of decline have been noted at the current eelgrass bed monitored in the Niantic River (NR #4), sampled in 1995 and 1996. It remains to be seen whether factors causing declines elsewhere in the river (perhaps water quality, disease or waterfowl grazing) will eventually impact the eelgrass bed at this site.

Eelgrass beds at the other sites, WP and JC, have persisted over the entire study period; however, analyses of some population parameters indicated moderate decline at both sites. The WP population is on the fringe of the predicted areal extent of the thermal plume, but temperature monitoring has never indicated water temperatures above ambient at that site. Furthermore, low standing stock at WP in 1996 was observed when no thermal plume was produced at MNPS due to plant shutdown. Therefore, recent indications of decline at WP were attributed to natural variability rather than a power plant impact.

While declines are apparent for the JC eelgrass population, some improvement in shoot density and standing stock has been observed in the most recent sampling years (1995 and 1996). The study population at JC is within the predicted thermal plume area, and because of shallow water depths there (≤1 m), this population is also susceptible to additional stress from solar warming in summer, sediment freezing in winter, and waterfowl grazing. Sediment transport and sand shoaling may also represent environmental stress. Elevated temperatures relative to those at the MNPS intakes have been measured directly at JC, and may have caused periodic eelgrass population declines observed there. Elevated summer temperatures (by 4-5°C) were measured in 1996 at JC, and were attributed entirely to solar warming and water circulation patterns in Jordan Cove since MNPS was shut down. Based on 1996 findings, these natural factors were largely responsible for temperature fluctuations observed historically at JC, with the MNPS thermal plume only a minor influence at most.

Benthic Infauna

Benthic infaunal studies during 1996 continued to document ongoing community changes related to

impacts from construction and operation of MNPS. This was accomplished by monitoring nearby subtidal soft-bottom habitats for changes in sedimentary characteristics and infaunal community structure (total abundance, species number and species composition). Results of these studies through the 1996 sampling year indicated that MNPS operation-related community changes continued to be observed at the three study sites nearest to MNPS.

The only study site not influenced by MNPS was the GN reference site, located well beyond the area of possible impacts. Data collected at this site continued to reflect long-term physical and biological stability; sediments collected at GN in 1996 were similar to previous years, and overall community composition was consistent over the study period. Specifically, the same four taxa (oligochaetes, *Tharyx* spp., *Aricidea catherinae* and *Mediomastus ambiseta*) have been numerically dominant at similar relative abundance levels over both 2-unit and 3-unit operational periods.

Two study sites (IN and JC) continued to reflect changes related to past isolated physical disturbances from MNPS that occurred over a relatively short duration. In recent years, sediment silt/clay content has declined to near pre-impact levels and indications of community recovery are evident through 1996. Species richness and abundances of oligochaetes and *Aricidea catherinae* (common taxa prior to 1983) have increased, while abundances of more opportunistic species (e.g., *Nucula proxima*) have declined. However, continued dominance of post-impact species, such as *Tharyx* spp., indicates the recovery is still ongoing at IN.

Habitat and community changes resulting from another past disturbance event attributed to MNPS operation were still evident at JC in 1996. Specifically, the effects of siltation at JC after the start-up of Unit 3 in 1986, associated with increased cooling water flow and sediment scour in the immediate area of the discharge, continued to be observed in 1996. Abundances of the previously dominant oligochaetes, and the polychaetes *Polycirrus eximius* and *Aricidea catherinae* quickly decreased. The impact of this siltation event has apparently lessened since 1986, as populations of some 2-unit period dominants (e.g., *A. catherinae*) have recovered. However, long-term persistence of some of the deposited silt/clay was still evident after 1995 sampling, and community recovery is evident but slow at JC.

Discharge scour directly and continuously impacts the sedimentary environment and the infaunal community at EF. Sediments in 1996 were characterized by increased sediment grain size and decreased silt/clay levels relative to 2-unit operational years. The infaunal community at EF has developed under the new, relatively stable, high current conditions in the discharge area. Populations of species common during 2-unit operation (e.g., *A. catherinae* and *P. eximius*) have returned to EF, while other species (including *Tharyx* spp.) have declined or do not occur even during periods of area-wide increase (e.g., *M. ambiseta*). Little change in the impacted infaunal community at EF was observed during 1996, when MNPS was shut down. This indicates that this community will likely persist for some time after sediment scour produced by the MNPS discharge ceases.

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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 (1996).

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).

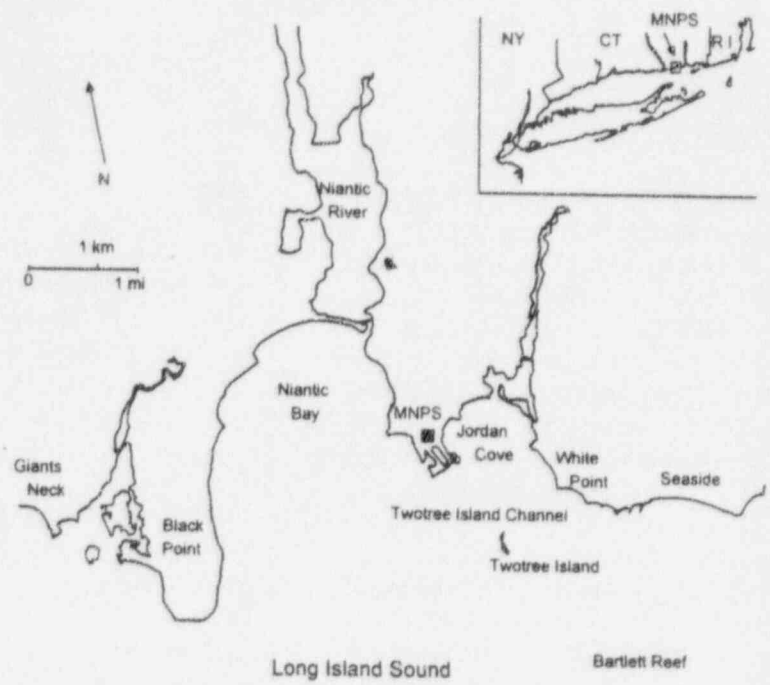


Fig. 1. The area in which biological monitoring studies are being conducted to assess the effects of the operation of MNPS.

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, 1994). 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 1996 were the lowest since Unit 3 went online: 0% for Unit 1 (shutdown on November 4, 1995), 13.7% for Unit 2 (February 20, 1996), and 24.9% for Unit 3 (March 30, 1996). All units remained shut down for all or most of 1996 for reasons that are beyond the scope of this report.

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 heated effluent into an abandoned granite quarry (ca. 3.5 ha surface area, maximum depth of approximately 30 m).

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
July 25 to August 17, 1986	Unit 3 shutdown	DNGL

TABLE 1. (cont.).

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; NUEL
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; NUEL
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; NUEL
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; NUEL
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 August 4, 1995	Unit 2 shutdown	MOSR
April 14 to June 7, 1995	Unit 3 shutdown	MOSR
November 30 to December 15, 1995	Unit 3 shutdown	MOSR
November 4, 1995 to undetermined	Unit 1 shutdown	MOSR
February 20, 1996 to undetermined	Unit 2 shutdown	MOSR
March 30, 1996 to undetermined	Unit 3 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.

The thermal discharge (about 11°C warmer than ambient under typical three-unit operation) exits the quarry through two channels (cuts), whereupon it mixes with LIS water (Fig. 2). The cuts are equipped with fish barriers made up 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 mostly with tidal currents (Fig. 3).

All hydrothermal surveys conducted at MNPS were described in detail in NUSCO (1988b).

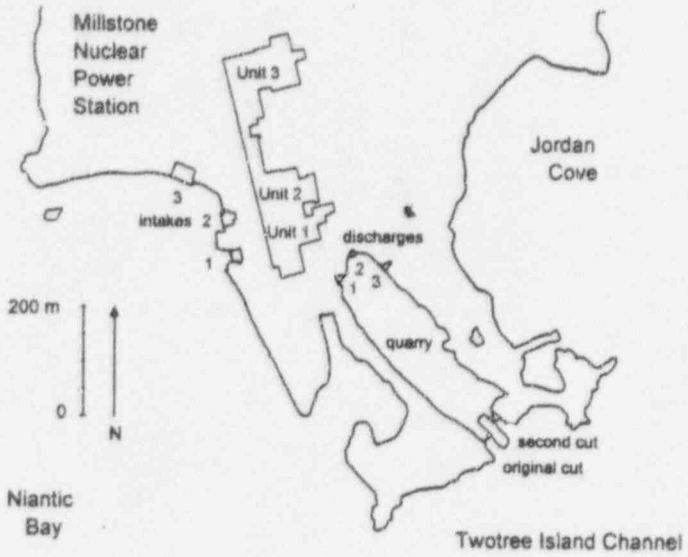


Fig. 2 The MNPS site, showing the intake and discharge of each unit, the quarry, and the two quarry discharge cuts.

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 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, 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.

References Cited

NUSCO. (Northeast Utilities Service Company). 1973. Environmental effects of site preparation and construction. Pages 4.4-1 to 4.5-1 *in* Millstone Nuclear Power Station, Unit 3, Environmental report. Construction permit stage.

NUSCO. 1978. Impingement studies. Millstone Units 1 and 2, 1977. Pages 1-1 to 4-2 *in* Annual report, ecological and hydrographic studies, 1977. Millstone Nuclear Power Station.

NUSCO. 1983. Millstone Nuclear Power Station Unit 3 environmental report. Operating license stage. Vol. 1-4.

NUSCO. 1986. The effectiveness of the Millstone Unit 1 sluiceway in returning impinged organisms to Long Island Sound. Enclosure to Letter D01185 dated May 27, 1986 from R.A. Reckert, NUSCO, to S.J. Pac, Commissioner, CT DEP. 18 pp.

NUSCO. 1988a. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to Letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP. 21 pp.

NUSCO. 1988b. Hydrothermal studies. Pages 323-355 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-87.

NUSCO. 1994. Progress report on the MNPS fish return systems. Enclosure 1 to Letter D08071 dated October 20, 1994 from D. Miller, NNECO, to T. Keeney, Commissioner, CT DEP. 11 pp.

NUSCO. 1996. Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1995. 231 pp.

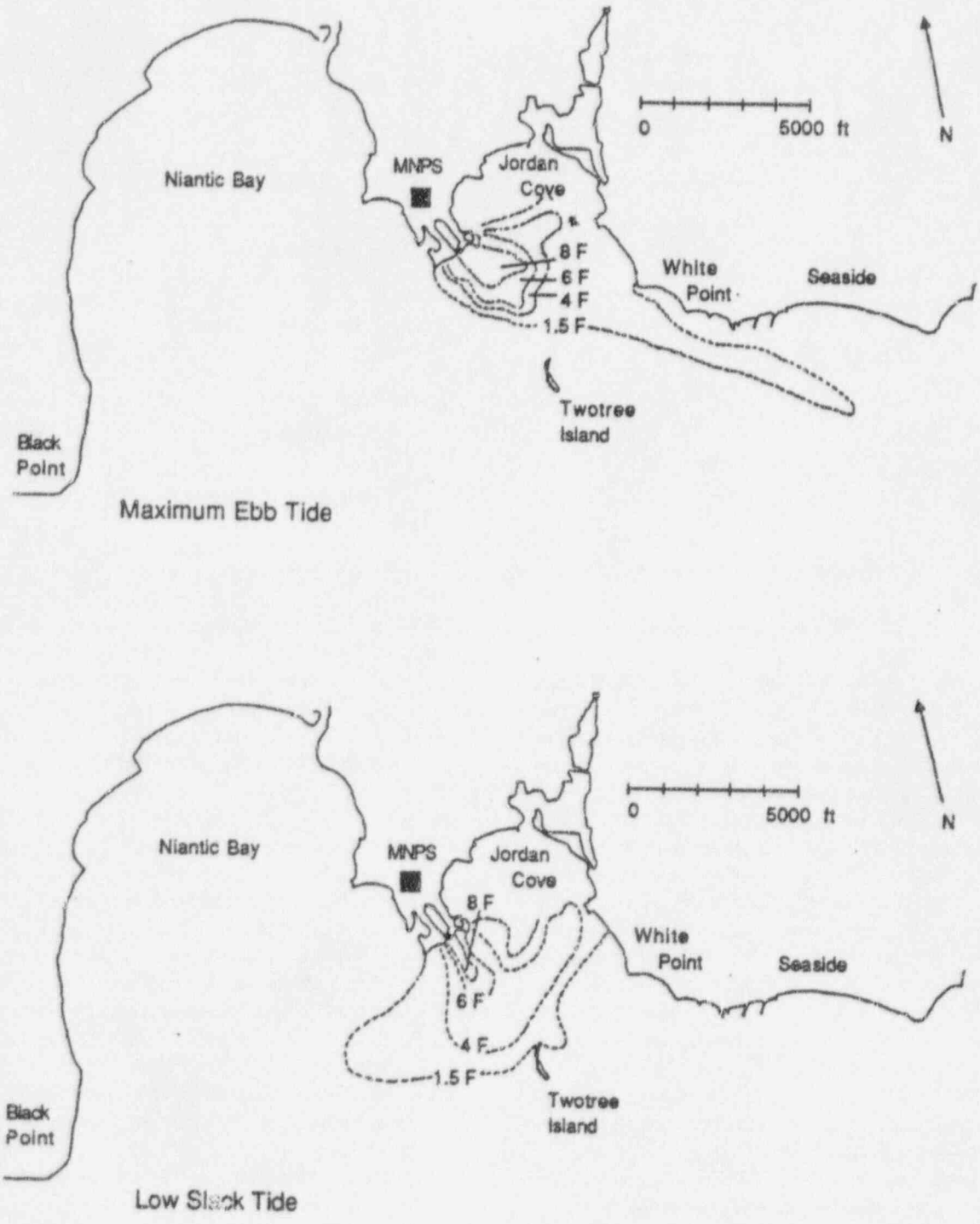
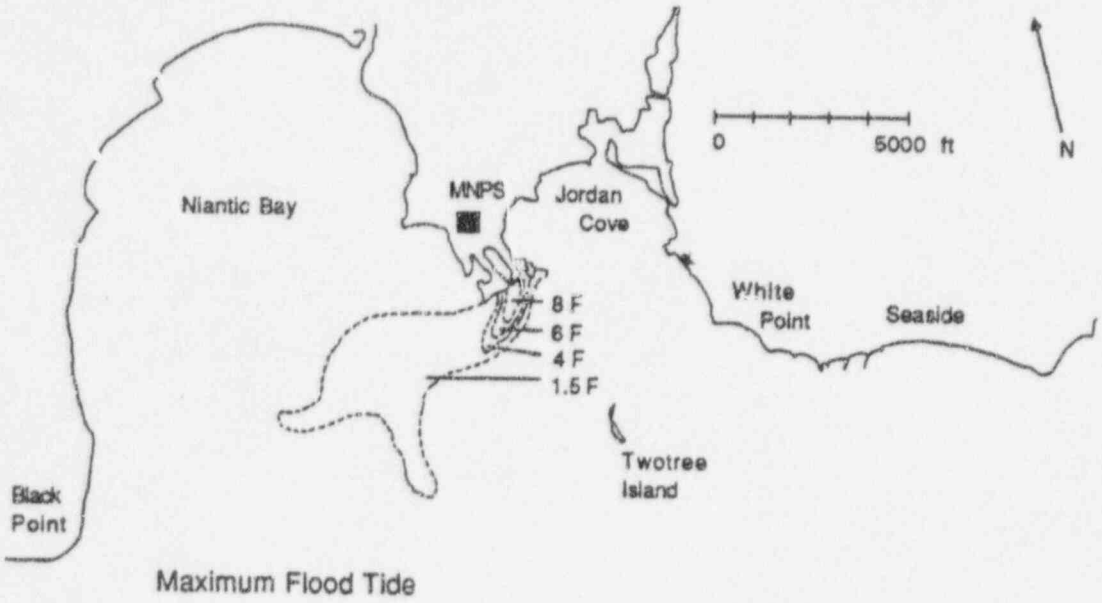


Fig. 3. Locations of selected three-unit thermal plume isotherms (1.5°F, 4°F, 6°F, and 8°F) under various tidal conditions.



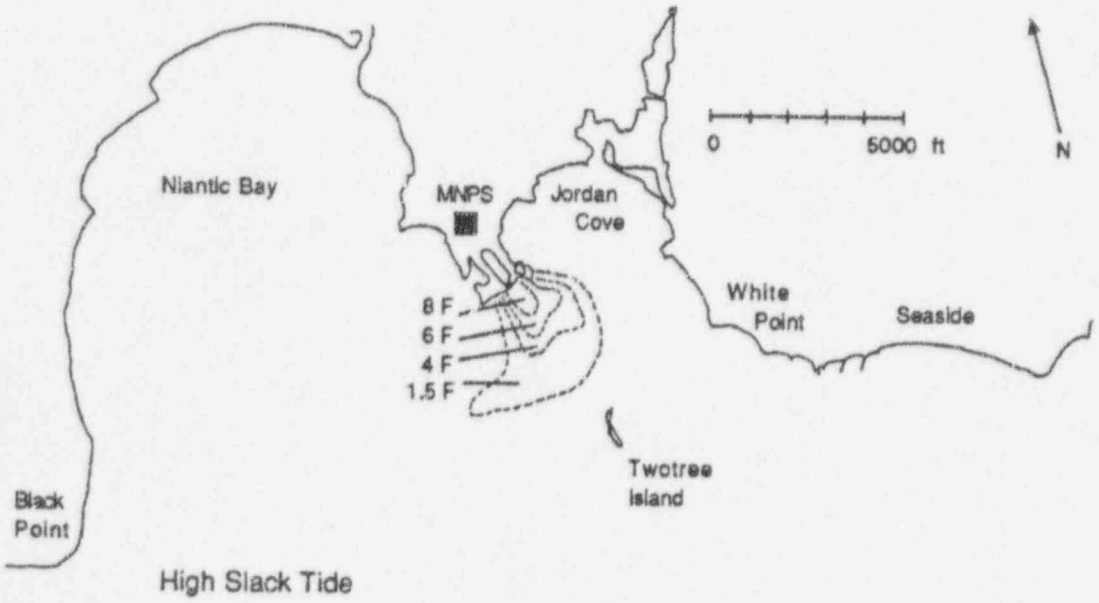


Fig. 3. (continued).

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Fish Ecology Studies

Introduction

Fish are important members of the aquatic community of eastern Long Island Sound (LIS). LIS provides a variety of habitats for fishes that in concert with a temperate climate results in a diverse assemblage of species, including residents present all year that exhibit little movement, highly migratory fish present only in certain seasons, and rarely collected transients with centers of distribution elsewhere. Fish also support commercial and sport fishing activities in Connecticut worth millions of dollars each year (Sampson 1981; Blake and Smith 1984).

The objective of the fish ecology monitoring programs at Millstone Nuclear Power Station (MNPS) is to determine whether operation of the three electrical generating units adversely affects the occurrence, distribution, and abundance of local fishes. Potential MNPS impacts include entrainment of early life history stages through the cooling-water system, impingement of juvenile and adult fish on intake screens, and changes in distribution as a result of the thermal discharge. Numbers of fish eggs and larvae entrained have been reliably estimated, but quantifying long-term effects of this impact is more difficult. Effects of entrainment mortality are influenced by biological processes, such as compensatory mortality, density-dependent growth, fecundity of individual species, population age structure, and life history strategies. Impingement can also be readily measured but, as in the case of eggs and larvae, the implications of fish removal are difficult to ascertain as adult populations are also affected by natural and fishing mortality. In any event, the impact of fish impingement at MNPS has largely been mitigated by the installation and operation of fish return sluiceways at MNPS Units 1 and 3 (NUSCO 1986, 1988c, 1994b). Changes in the thermal regime of local waters due to MNPS operation have been well-documented (NUSCO 1988b; see also the Introduction section to this report). If water temperatures exceed tolerance levels, fish may be forced to move from the area, vacating potentially important spawning or nursery grounds.

Trawl, seine, and ichthyoplankton (fish eggs and larvae) monitoring programs were established to provide information for the assessment of impacts from MNPS operation on local fish populations. These programs provided a basis for identifying taxa potentially affected, as well as information on long-term abundance trends used to measure changes in local populations and have changed over time following evaluations (NUSCO 1987, 1994a, 1995). Potentially impacted species were selected on the basis of life history characteristics, such as susceptibility of early developmental stages to entrainment at MNPS, stock structure (i.e., localized or coast-wide populations), and local distribution in relation to the thermal plume. In this report, data from June 1995 through May 1996 are summarized and compared to data previous collected from trawl, seine, and ichthyoplankton monitoring programs.

Tautog (*Tautoga onitis*) support one of the principal sport and commercial fisheries of LIS (Smith et al. 1989; ASMFC 1996), but abundance of juveniles and adults has declined since 1984 (Simpson et al. 1995). As a result of this decline, likely from overfishing on this slow-growing and long-lived fish, an interstate management plan is under development to provide for the conservation, restoration, and enhancement of the tautog stock (ASMFC 1996). Because of relatively high entrainment of tautog eggs at MNPS, concern has been raised by the Connecticut Department of Environmental Protection (CT DEP) regarding the tautog population in the vicinity MNPS. From this concern, special studies of tautog early life history were conducted in 1996. This work focused on tautog egg distribution and was based, in part, on the methodology and results of previous studies, which are summarized below in the Results and Discussion section. In addition, a preliminary assessment of tautog egg entrainment was made in terms of equivalent female spawners removed by MNPS operation. Tautog studies in 1996 and those that will be completed during subsequent years are being performed in lieu of sampling at three offshore trawl stations (BR, TT, NB; NUSCO 1996) in accordance with an agreement with the CT DEP (NNECO 1995a, 1995b, 1996).

Materials and Methods

Annual results are presented for a 12-month period that extends from June of one 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 a May 31 endpoint. When a species season of occurrence crossed a calendar year, the annual period was termed a report year (e.g., 1995-96). When a species was collected only within a calendar year, the annual period was presented as a specific year (e.g., 1996). Although methods of collection for the 1995-96 report year were essentially the same as those used in previous years, the number of stations used in the trawl monitoring program was reduced from six to three in January 1996 with the deletion of stations NB, TT, and BR (NNECO 1995a). The materials and methods that follow correspond to the 1995-96 report year.

Ichthyoplankton Program

The sampling frequency for ichthyoplankton entrained through the MNPS cooling-water system varied seasonally during 1995-96. Both day and night samples were collected twice a week during June through August, once a week in September and February; and three times a week during March through May. Only one day sample per week was collected during October through January. Generally, samples were collected each week at only one of the three plant discharges (station EN, Fig. 1), with the site of collection usually alternating weekly between Units 1 and 2, if plant operations permitted. Following the shutdown of Unit 1 in November 1995 (no circulating water pumps in operation), samples were taken exclusively at Unit 2 until mid-April of 1996, when sampling was also done at Unit 3. Beginning in early May, circulating water pumps at Units 2 and 3 were operated for only a few hours

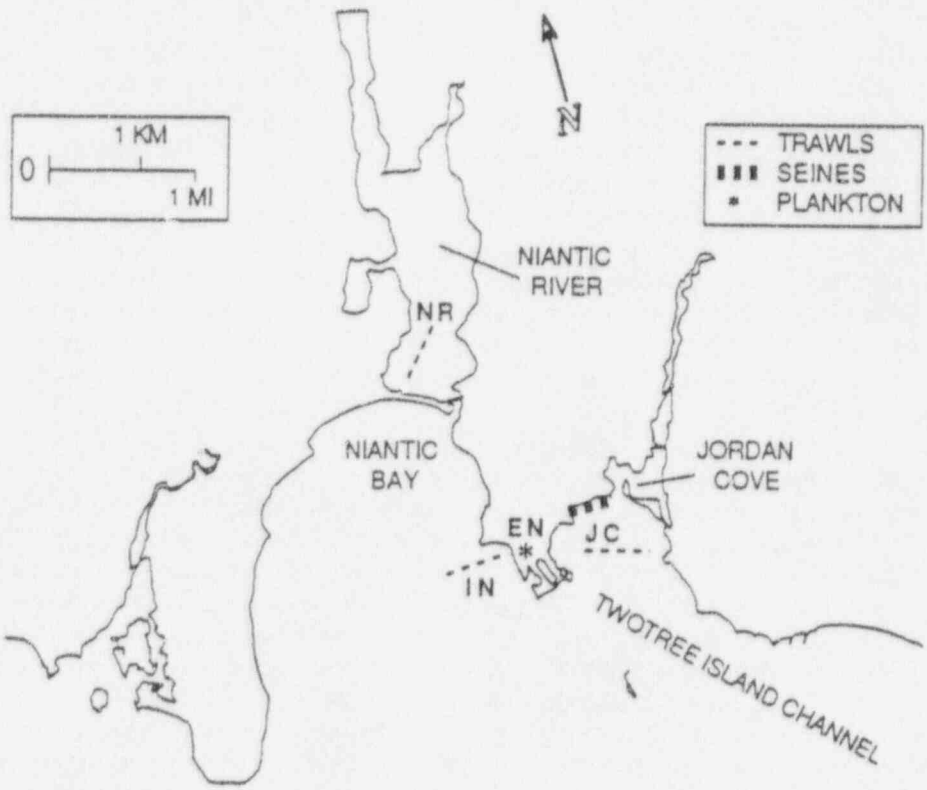

Fig. 1. Location of current trawl, seine, and ichthyoplankton sampling stations.

each week and entrainment samples were taken during the infrequent occasions when this occurred.

To collect samples from the plant discharge, a 1.0 x 3.6-m conical plankton net with 333-µm mesh was deployed with a gantry system. Four General Oceanic flowmeters (Model 2030) were mounted in the mouth of the 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 when volume was reduced) was determined from the average readings of the four flowmeters.

Ichthyoplankton samples were separated using a NOAA-Bourne splitter (Botelho and Donnelly 1978); fish eggs and larvae were removed from the samples with the aid of a dissecting microscope. 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 had been examined. Larvae were identified to the lowest practical taxon and enumerated in all samples, except for June samples, for which only two (one day and one night) samples per week were examined. Tautog, cunner (*Tautogolabrus adspersus*), and anchovy (bay anchovy, *Anchoa mitchilli* and striped anchovy, *A. hepsetus)* eggs were identified and enumerated in all samples collected from April through September. Tautog and cunner are both in the Family Labridae and their eggs are very similar in appearance. They were distinguished on the basis of a weekly bimodal distribution of egg diameters (Williams 1967). All ichthyoplankton densities were reported as a number per 500 m^3 of water filtered.

During 1996, special studies were conducted to examine the spatial distribution of tautog eggs potentially entrained at MNPS. Tautog egg entrainment is likely a function of egg distribution, that is related to spawning aggregations of adults and hydrodynamic transport during the approximate 2 to 3-day span between spawning and hatching. The vertical distribution of tautog eggs was examined using paired surface and near-bottom tows by simultaneously sampling with two 60-cm bongo samplers having 333-µm mesh nets and 22.7 kg depressors. Volume filtered was determined from a single GO flowmeter mounted in the center of each bongo opening. Sampling was conducted with one bongo system deployed just below the surface and another near the bottom. Three replicate paired tows were taken during the evening peak spawning period on both July 10 and 15 and during the mornings of July 11 and 16 (approximately 12 hours after peak spawning) for a total of 12 paired surface-bottom collections. Sampling was conducted near the time of slack tidal currents on July 10 and 11 and maximum tidal currents in both July 16 collections. The sampling was conducted at a point midway between Millstone Point and Black Point having a water depth of about 10 m. Temperature and salinity were determined at surface, midwater, and bottom depths using a YSI Model 30 meter.

To determine spatial distribution offshore of MNPS, sampling was conducted within a 5 nautical mile (n mi) radius of MNPS, which was determined as the most likely potential source area for tautog eggs entrained at MNPS. Five sampling points were located at 1 n mi intervals along three separate transects, with the point of origin for each transect at the mid-point between Millstone Point and Black Point. From the origin, the transects extended approximately southeast (SE), south (S), and southwest (SW). The rationale for selecting these stations and the precise location of each will be detailed in the Results and Discussion section. Vertical tows were taken at each station with the 60-cm bongo sampler described above. The sampler was lowered from the surface to the bottom and immediately retrieved to the surface while the boat remained at idle; water was filtered during both descent and ascent. Replicate tows were taken at shallower sampling sites until approximately 30 m^3 of seawater had been filtered (both nets combined). Material retained by both nets was combined as one sample for each station. Sampling was conducted in the morning after sunrise on July 2 and July 9. Sampling simultaneously with two boats required about 3 hours, starting about 2 hours prior to slack current, and was conducted during the period encompassing slack after ebb (July 2) and slack after flood (July 9). On both dates, each site was sampled once, except for the station at the transect origin, which was sampled at the beginning, near the middle, and at the end of each sampling period.

Trawl Program

Triplicate bottom tows were made using a 9.1-m otter trawl with a 0.6-cm codend liner. As of January 1996, demersal fishes were collected every other week throughout the year at three stations: Niantic River (station NR), Jordan Cove (JC), and Intake (IN) (Fig. 1). A standard tow was 0.69 km, but if the trawl net became loaded with macroalgae and

detritus, tow distances were shortened and catches standardized to 0.69 km by proportionally adjusting the catch. This change in sampling procedure occurred frequently in 1995-96 due to abundant concentrations of macroalgae and detritus at NR and JC. 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. Catch of tautog in lobster pots (see Lobster Studies section for sampling methods) were used to supplement the trawl abundance data for this species.

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 (standard distance of 30 m) were made parallel to the shoreline at Jordan Cove (JC) biweekly from April through November (Fig. 1). Collections were made during a period 2 hours before and 1 hour after high tide. Fish from 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 total length. Catch was expressed as number of fish per haul.

Data Analyses

Abundance Estimates

The Δ-mean was used as an index of abundance of juvenile and adult fish collected in the trawl and seine programs, and of fish eggs and larvae in the ichthyoplankton program. Beginning with this report, trawl catch data will focus on the three stations currently being sampled; catch data from the six stations previously sampled may be found in NUSCO (1996). The Δ-mean was selected because it is the best estimator of the mean for population abundance data that approximately follows the lognormal distribution and contains numerous zeros (Pennington 1983, 1986). Calculation of this index and its variance estimate was described in detail in NUSCO (1988a). Because of varying sampling frequencies, the Δ-mean indices of ichthyoplankton taxa were weighted by the largest number of samples collected in a week to standardize data across weeks and years. With species that occurred seasonally, the data used for calculating the Δ-mean were restricted by date to reduce the number of zero values in the distribution tails, which extend beyond the season boundaries. 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 nonparametric, distribution-free Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether the direction of change of an annual Δ-mean time-series represented a significant ($p \leq 0.05$) trend. Sen's (1968) nonparametric estimator of the slope was used to describe the rate of change of significant trends. This approach to trend analysis was suggested by Gilbert (1989) as particularly well-suited for analysis of environmental monitoring data because no distributional assumptions are required and because small sample sizes are acceptable. Wilcoxon's signed-ranks test (Sokal and Rohlf 1969) was used to compare the catch of tautog among lobster pot stations. Spearman's rank-order correlation (Sokal and Rohlf. 1969) was used to describe associations among various abundance indices.

Entrainment Estimates

Entrainment estimates of dominant ichthyoplankton were calculated from daily density estimates at station EN. These estimates were based on the parameters of a Gompertz function fitted to the entrainment data. The distribution of egg and larval abundance over time is usually skewed because their 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 particular form of the Gompertz function used (Gendron 1989) was:

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

where

C_t = cumulative density at time t

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

α = total or asymptotic cumulative density

p = inflection point in days since first occurrence date

κ = 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 the α, p, and κ parameters were obtained by fitting the above equation to the cumulative abundance data using nonlinear regression methods (Proc NLIN; 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 geometric mean of weekly densities was used in analyses because the data generally followed a log-normal distribution (McConnaughey and Conquest 1993) and weekly sampling frequencies varied.

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 abundance over time (abundance curve), has the form:

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

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

Species Composition

At least 130 fish taxa were collected as eggs, larvae, juveniles, and adults in the trawl, seine, and ichthyoplankton programs as part of the Fish Ecology monitoring studies for MNPS from June 1976 through May 1996. This includes fishes collected at present and former stations during this 20-year period, with 116 taxa taken by trawl, 51 by seine, and 67 collected in ichthyoplankton samples (Appendix I). For ichthyoplankton sampling at the MNPS discharges (station EN), anchovies (mostly bay anchovy) and winter flounder (*Pleuronectes americanus*) comprised about two-thirds of the larvae collected with 13 other taxa making up most of the remainder (Table 1). Cunner, tautog, and anchovies accounted for over 86% of the eggs collected. Silversides (Atlantic silverside, *Menidia menidia* and inland silverside, *M. beryllina*) dominated (80%) the seine catch at station JC (Appendix II); another 16% were killifishes (striped killifish, *Fundulus majalis* and mummichog, *F. heteroclitus*) and fourspine stickleback (*Apeltes quadracus*). Seven taxa accounted for about 82% of the total catch at the three trawl stations (Appendices III-V). These were the winter flounder, scup (*Stenotomus chrysops*), silversides (mostly Atlantic silverside), windowpane (*Scophthalmus aquosus*), grubby (*Myoxocephalus aenaeus*), skates (mostly little skate, *Raja erinacea*; also the winter skate, *R. ocellata* and clearnose skate, *R. eglanteria*), and anchovies (mostly bay anchovy). Total catch of fish over the 20-year period was very similar between IN (105,065 specimens) and NR (104,097), each of which was about 1.8 times the total catch of 57,766 fish at JC.

TABLE 1. Taxonomic composition of ichthyoplankton collected at EN (as a percentage of the total) from June 1976 through May 1996 as larvae and April 1979 through September 1995 as eggs.

Taxa	Larvae	Eggs
Anchoa spp.	52.2	5.9
Pleuronectes americanus	12.9	
Ammodytes americanus	7.9	
Myoxocephalus aenaeus	4.8	
Brevoortia tyrannus	4.6	
Pholis gunnellus	2.4	
Tautogolabrus adspersus	2.3	52.6
Tautoga onitis	2.0	27.6
Enchelyopus cimbrius	1.6	
Ulvaria subbifurcata	1.2	
Liparis spp.	1.1	
Syngnathus fuscus	1.0	
Scophthalmus aquosus	0.8	
Peprilus triacanthus	0.8	
Clupea harengus	0.6	

The temporal changes during the 20-year period in the composition of the above dominant taxa collected in trawl and ichthyoplankton programs were compared using Δ-means. Changes in the composition of seine catches were not examined because silversides have always dominated the catch. In trawl sampling, winter flounder and scup had the largest annual Δ-mean CPUE during each report year, with silversides, windowpane, and skates also relatively numerous (Table 2). Because of the elimination of

TABLE 2. The annual Δ-mean[a] CPUE (no./0.69 km) of the most abundant fish collected by trawl at JC, IN and NR for each report year from June 1976 through May 1996 (two-unit operational period: 1976-85; three-unit operational period: 1986-96).

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	94-95	95-96
P. americanus	23.9	15.6	16.7	26.6	34.8	28.9	49.4	30.6	31.3	23.5	27.3	27.3	41.0	23.1	28.4	26.7	25.3	16.9	22.6	11.4
S. chrysops	14.8	13.0	5.6	6.2	9.2	7.9	25.1	25.9	14.3	8.3	24.1	17.4	11.4	11.0	25.8	176.0	56.3	2.7	26.4	5.1
Menidia spp.	18.2	8.5	10.1	7.1	3.3	2.5	3.3	2.8	2.0	3.8	23.1	4.1	5.0	2.4	2.9	8.6	18.4	2.5	2.2	1.6
S. aquosus	1.7	1.8	0.9	1.8	1.6	1.5	2.2	3.0	2.4	2.5	3.0	4.3	3.6	4.9	3.3	2.4	3.7	6.0	3.4	3.4
M. aeneus	0.6	0.9	0.9	1.9	1.8	2.5	3.3	2.1	1.8	1.2	2.3	1.6	3.5	1.7	2.2	1.4	2.7	1.3	2.3	0.9
Raja spp.	0.7	0.6	0.4	0.4	0.8	0.6	1.0	2.6	0.7	1.8	1.8	2.2	2.6	2.4	3.4	3.2	2.2	3.2	2.1	3.5

[a] Data were seasonally restricted to June-October for *S. chrysops*, October-February for *Menidia* spp., but unrestricted (June-May) for the remaining taxa.

the three offshore trawl stations as of January 1996, skates and anchovies became less numerically dominant, whereas the grubby increased in relative proportion of the catch from the totals for six trawl stations reported in NUSCO (1996). Winter flounder, windowpane, grubby, and skates are collected throughout most of the year by trawl and their respective abundances have remained relatively stable since the early to mid-1980s. In contrast, scup, anchovies, and silversides were collected seasonally (mostly summer and fall for scup and anchovies and winter for silversides). The annual abundances of these fishes fluctuated to a greater degree because most of the catch was young-of-the-year with abundances related to variable reproductive success. Also, anchovies and silversides school and occasional large catches affected the magnitude of annual Δ-mean CPUE. In fact, Δ-means could not be computed for trawl catches of anchovies because of infrequent catches interspersed with a large number of zeroes in the time-series. The 1995-96 Δ-mean CPUE for both winter flounder (11.4) and silversides (1.6) were the lowest and for scup (5.1) the second lowest in 20 years. In contrast, the Δ-mean of 3.5 for skates was the largest annual CPUE for that species group, although annual means were less than when all six stations were included (NUSCO 1996). Trawl catch of grubby (Δ-mean CPUE of 0.9) at the three inshore stations during 1995-96 was the lowest since 1978-79.

All dominant ichthyoplankton taxa were collected seasonally at EN. Therefore, Δ-mean densities (no./500 m^3) were computed from data collected during standardized periods of occurrence for each taxon (Table 3). Cunner eggs were always the most abundant of the fish eggs collected. Tautog eggs were second-most abundant after a large decline in anchovy egg abundances that occurred during the mid-1980s. For 1995-96, abundances of tautog and anchovy eggs and most of the 14 dominant larval taxa fell within the range of abundance values for previous years. The Δ-mean density for cunner eggs of 7,126 was the largest value since 1980-81, when it was 8,223. Anchovy egg abundance remained depressed, with the Δ-mean of 153 the lowest annual density index since 1992-93.

Winter flounder larvae ranked second or third each year, except for 1992-93 and 1995-96, when larval grubby had their third highest abundance in 20 years (Δ-mean density of 85). American sand lance (*Ammodytes americanus*) larvae were abundant from 1976-77 through 1980-81 and decreased considerably until the past few years when Δ-mean densities began to increase. However, the 1995-96 Δ-mean of 18 represented a substantial decrease from last year (63). Larvae of the Atlantic menhaden (*Brevoortia tyrannus*) have increased in abundance in recent years and are becoming a dominant summer species. Larvae of other species, particularly those of cunner, tautog, rock gunnel (*Pholis gunnellus*), fourbeard rockling (*Enchelyopus cimbrius*), radiated shanny (*Ulvaria subbifurcata*), and snailfishes (*Liparis* spp.), occasionally were relatively abundant.

Entrainment Estimates

Entrainment of fish eggs and larvae in the condenser-cooling water system represents a direct impact from the operation of MNPS. The annual numbers of eggs and larvae entrained were related to their abundance at station EN and plant operations (i.e., cooling-water usage). Due to the start-up of Unit 3 in 1986, cooling-water usage increased nearly twofold upon full three-unit operations. However, this did not necessarily result in comparable increases in entrainment estimates, due to fluctuations in annual abundances of the dominant ichthyoplankton available to entrainment as indicated by abundances at station EN (Table 3). Most fish eggs collected at station EN were from cunner,

TABLE 3. The annual 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 1996 (two-unit operational period: 1976-85; three-unit operational period: 1986-96).

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	94-95	95-96
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	7,057	7,126
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	1,650	2,074
Anchoa spp.				1,447	1,245	1,080	765	2,257	4,880	145	910	89	38	54	127	476	107	542	423	153
LARVAE[b]																				
Anchoa spp.	1,152	931	483	2,168	2,430	5,768	816	1,421	502	1,102	1,244	126	359	619	1,122	799	178	233	475	181
P. americanus	106	143	114	285	129	233	297	210	180	83	109	116	203	106	99	388	21	142	224	81
A. americanus	94	318	119	111	136	21	27	18	9	3	13	41	31	24	7	18	28	43	63	18
M. aenaeus	41	38	36	38	107	72	68	50	68	34	29	95	63	30	24	58	34	48	42	85
B. tyrannus	5	4	4	0	3	1	11	23	2	41	3	2	6	72	18	97	41	9	54	66
P. gunnellus	13	13	16	13	58	27	13	14	14	22	4	26	9	6	3	15	8	28	17	41
T. adspersus	29	58	1	13	58	78	31	49	4	12	4	5	9	14	68	209	8	10	25	12
T. onitis	37	36	1	11	46	83	44	33	3	15	3	7	17	15	33	99	13	6	12	8
E. cimbrius	2	8	6	8	6	1	6	13	5	8	8	12	45	31	37	98	5	18	9	8
U. subbifurcata	5	9	14	14	16	17	6	4	60	7	9	23	41	51	34	28	2	18	8	<1
Liparis spp.	27	30	10	16	22	5	13	8	36	1	4	42	18	12	3	23	14	12	5	<1
S. fuscus	4	7	4	9	8	13	7	9	9	5	4	6	7	5	3	5	3	6	4	6
S. aquosus	10	11	1	5	5	5	2	13	3	1	4	3	5	3	4	12	2	2	3	1
P. triacanthus	14	3	1	2	11	17	9	9	1	2	3	0	9	5	29	10	2	2	5	3

[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*, 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*, and June-September for *P. triacanthus*.

tautog, and anchovies and the most numerous larvae were anchovies, winter flounder, American sand lance, grubby, and Atlantic menhaden (Table 1). The estimated number of cunner (4.9 billion) and tautog (2.7 billion) eggs entrained in 1995 fell within the range of previous years for three-unit operation (Table 4). Entrainment estimates for anchovy eggs were much lower in recent years than during a period of greater abundance that occurred during the early 1980s. In 1995, only an estimated 16 million were entrained. Larval entrainment estimates in 1995 for anchovies (186 million) and in 1996 for American sand lance (23 million) were less than long-term averages (about 579 and 65 million, respectively) that included the period prior to three-unit operations (Table 5). Entrainment estimates for grubby have been relatively consistent, with the estimate of 43 million determined for 1996 near the long-term average of 49 million. Entrainment of winter flounder larvae totaled nearly 54 million and this loss is evaluated in the Winter Flounder Studies section of this report.

Impingement

Although impingement of organisms on the intake traveling screens is a potential direct impact at MNPS, impingement monitoring on a regular basis was discontinued in 1987 (NUSCO 1988c). This monitoring reduction was implemented because impingement losses were well documented and most impingement was mitigated by the operation of return sluiceways at Units 1 and 3 (NUSCO 1986, 1988c, 1994b). Impingement at Unit 2, which does not have a return sluiceway, is routinely monitored by plant operational personnel and impingement counts are only made when a large impingement event occurs; none occurred during 1995-96. The Unit 1 sluiceway may be taken out of service during periods of high debris loading that would adversely affect plant operations. The number of days that Unit 1 sluiceway was out of service each month from 1985 through 1995 was determined from plant operational records (Appendix VI). During 1995, the latest period for which the information was available, the sluiceway was out of service for 54 days, mostly in May, August, and September.

Selection of Potentially Impacted Taxa

Additional data analyses were completed for selected taxa that were identified as potentially impacted, either because of their prevalence in entrainment samples or because of possible influence by the thermal discharge. Taxa potentially vulnerable to entrainment include American sand lance,

TABLE 4. Estimated number of cunner, tautog, and anchovy eggs entrained each year from 1979 through 1995 at MNPS and the volume of cooling water on which the entrainment estimates were based (two-unit operational period: 1979-85; three-unit operational period: 1986-95).

Year	Cunner		Tautog		Anchovies	
	No. entrained ($\times 10^6$)	Volume (m^3)[a] ($\times 10^6$)	No. entrained ($\times 10^6$)	Volume (m^3)[a] ($\times 10^6$)	No. entrained ($\times 10^6$)	Volume (m^3)[a] ($\times 10^6$)
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	2,754	1,565	1,390	1,565	17	1,537
1993	5,746	1,748	2,168	1,748	237	1,728
1994	5,982	1,726	2,162	1,726	170	1,693
1995	4,876	1,633	2,671	1,633	16	1,600

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

TABLE 5. Estimated number of anchovy, winter flounder, American sand lance and grubby larvae entrained each year from 1976 through 1996 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: 1986-96).

Year	Anchovies		Winter Flounder		American sand lance[a]		Grubby	
	No. entrained ($\times 10^6$)	Volume (m^3)[b] ($\times 10^6$)	No. entrained ($\times 10^6$)	Volume (m^3)[b] ($\times 10^6$)	No. entrained ($\times 10^6$)	Volume (m^3)[b] ($\times 10^6$)	No. entrained ($\times 10^6$)	Volume (m^3)[b] ($\times 10^6$)
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	536	1,241	182	1,175	77	2,091	58	1,118
1995	186	1,247	223	1,134	114	2,013	61	1,444
1996	-[c]	-	54	545	23	1,246	43	723

[a] Includes data from December of the previous calendar year.
[b] Volume was determined from the condenser cooling water flow at MNPS during the season of occurrence for each taxa.
[c] Not calculated because larvae occur after the end of the report period (May 1996).

anchovies, grubby, cunner, tautog, and winter flounder. The distribution of silversides in Jordan Cove may be affected by the MNPS thermal discharge. Therefore, Δ-mean densities (no./500m^3) for eggs and larvae, and Δ-mean CPUE for trawl (no./0.69 km) and seine (no./30 m) were calculated for abundant life stages of these selected taxa. Information on the winter flounder is presented in a separate section of this report (see Winter Flounder Studies) and is not included among the fishes discussed below.

American sand lance

The American sand lance is a schooling fish common in estuaries, along the coast, and in inshore waters from Labrador to Chesapeake Bay (Richards 1982; Nizinski et al. 1990). Sand lance have a life span of 5 to 9 years, but populations are dominated by the first three age groups (Reay 1970). Sexual maturation occurs at age-1 or 2 with adults spawning once a year, predominantly between November and March (Richards 1963, 1982; Scott and Scott 1988; Westin et al. 1979; Grosslein and Azarovitch 1982). Eggs are demersal and adhesive (Fritzsche 1978; Smigielski et al. 1984). Embryonic and larval development is lengthy (Smigielski et al. 1984).

American sand lance were taken in all three Fish Ecology programs, although relatively few juveniles and adults were collected by seine or trawl (Appendices I-V). Most sand lance were found as larvae in winter and spring at station EN. Larval abundance peaked in the late 1970s and early 1980s, followed by a rapid decline during the mid-1980s (Table 6; Fig. 2). Because sand lance larvae were so abundant from 1976-77 through 1980-81, larval densities in entrainment samples during the three-unit period to date have been lower than the two-unit average. The Δ-mean density of larvae at EN during 1995-96 was 18, less than one-third of the abundance index of the previous year. However, following the large decline in 1981-82, this Δ-mean is just below

TABLE 6. The annual Δ-mean[a] density (no./500 m^3) and 95% confidence interval of American sand lance larvae collected at EN from June 1976 through May 1996.

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
1994-95	63 ± 29
1995-96	18 ± 7

[a] Data seasonally restricted to December-May.

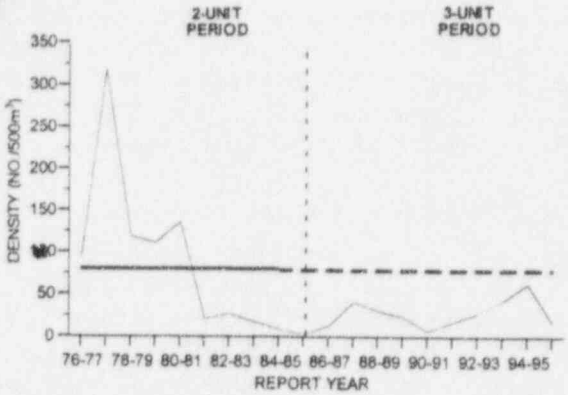

Fig. 2. Annual Δ-mean densities (no./500 m^3) of American sand lance larvae at station EN during MNPS two-unit (1976-8.) and three-unit (1986-96) operational periods. A Δ-mean density calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance.

the median value of the last 15 years. A significant decline ($p = 0.022$; Mann-Kendall test) occurred during the two-unit period (slope = −19.3); a similar negative trend was not apparent during the three-unit period. The large change in abundance was reflected in a wide range (5 to 190 million) in annual entrainment estimates, which were also dependent upon cooling-water flow during the larval season (Table 5). Plant cooling-water flow in 1996 during the period of occurrence of larval sand lance was the lowest of the three-unit period and, coupled with relatively low abundance, resulted in an entrainment estimate of only 23 million larvae.

Declines in sand lance abundance during the 1980s were also apparent in other areas of the Northwest Atlantic Ocean. Larval densities in LIS over a 32-year period (1951-83) were highest in 1965-66 and 1978-79; the latter peak was also evident throughout the entire range of American sand lance (Monteleone et al. 1987). This high abundance persisted throughout the Northwest Atlantic until 1981 and the decline that followed appeared to be inversely correlated with that of Atlantic herring and Atlantic mackerel (Nizinski et al. 1990). Sand lance likely increased in abundance due to a decrease in their predators, herring and mackerel, the abundance of which had been reduced by overfishing during the 1970s (Sherman et al. 1981). In more recent years, Atlantic mackerel, which prey heavily upon sand lance (Monteleone et al. 1987), have become more

abundant as sand lance abundance decreased. Given the large abundance changes of American sand lance along the Atlantic coast, effects of MNPS operation on this species are difficult to ascertain, but are likely small in comparison to the large-scale natural fluctuations typically associated with this fish.

Anchovies

The bay anchovy is one of the most abundant fishes found along the Atlantic Coast (McHugh 1967) and usually the dominant summer ichthyoplankton species found within its range (Leak and Houde 1987). This species ranges from Mexico to Cape Cod and occasionally into the Gulf of Maine (Hildebrand 1943; Bigelow and Schroeder 1953; Grosslein and Azarovitch 1982). Bay anchovy are common in nearshore and estuarine waters during warmer months, but move offshore in winter (Vouglitois et al. 1987). Chesapeake Bay bay anchovy were found to have little genetic variation, indicating a lack of stock structure, likely due to enormous population size and considerable movements and mixing (Morgan et al. 1995). Although the striped anchovy also occurs from Nova Scotia to Uruguay, its occurrence north of Chesapeake Bay is variable and the striped anchovy is usually found further offshore than the bay anchovy (Hoese and Moore 1977; Smith 1985). The eggs of the two species can be readily distinguished and since 1979, when eggs were first identified to species, about 96% of the eggs collected at station EN were those of the bay anchovy. Therefore, most of the anchovies collected in the Fish Ecology programs were likely bay anchovy, even if only identified to genus.

The bay anchovy can mature at 2.5 to 3 months in age and individuals spawn repeatedly during the summer (Luo and Musick 1991). In LIS, spawning takes place at depths of 20 m or less from May through September, with a peak during June and July (Wheatland 1956; Richards 1959). Spawning appears to be correlated with high zooplankton abundances (Castro and Cowen 1991; Peebles et al. 1996) and warm water temperatures (Zastrow et al. 1991). Eggs are pelagic and at 27°C hatch in about 24 hours (Kuntz 1914). Mortality rates of eggs and larvae are relatively high (Leak and Houde 1987; Houde et al. 1994; Dorsey et al. 1996), particularly from predation by ctenophores and jellyfish (Govoni and Olney 1991; Purcell et al. 1994). Juveniles and adults are important forage for many recreationally and commercially important fishes (Vouglitois et al. 1987) and also have high mortality rates (Newberger and Houde 1995).

Anchovies were collected in all three programs, but rarely by seine and only sporadically by trawl (Appendices I-V). Juvenile anchovies resulting from the summer spawn were typically captured by trawl from August through October. Most anchovies were collected in only 1 or 2 years of sampling at the three inshore trawl stations, including 1978-79 (23% of total catch at the station) and 1985-86 (52%) for IN, 1991-92 (42%) for JC, and 1989-90 (62%) for NR (Appendices III-V).

Anchovies dominated larval collections and their eggs ranked third in abundance (Table 1). Annual egg and larval abundances were correlated (Spearman's rank-order correlation coefficient $r = 0.55$; $p = 0.022$). The Δ-mean densities in 1995 for anchovy eggs (153) and larvae (181) fell within historical ranges, but were much lower than found in the early to mid-1980s (Table 7). Large annual changes in bay anchovy egg abundance were also observed in LIS during 1952-55 (Richards 1959) and in Barnegat Bay, NJ during 1976-81 (Vouglitois et al. 1987). All egg and larval densities during three-unit operational period were below the two-unit average because of a decline that occurred in the mid-1980s, prior to three-unit operation (Fig. 3).

TABLE 7. The annual Δ-mean[a] density (no./500 m^3) and 95% confidence interval of anchovy eggs and larvae collected at EN from June 1976 through May 1996.

Year	EGGS EN	LARVAE 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
1994	422 ± 356	475 ± 410
1995	153 ± 295	181 ± 117

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

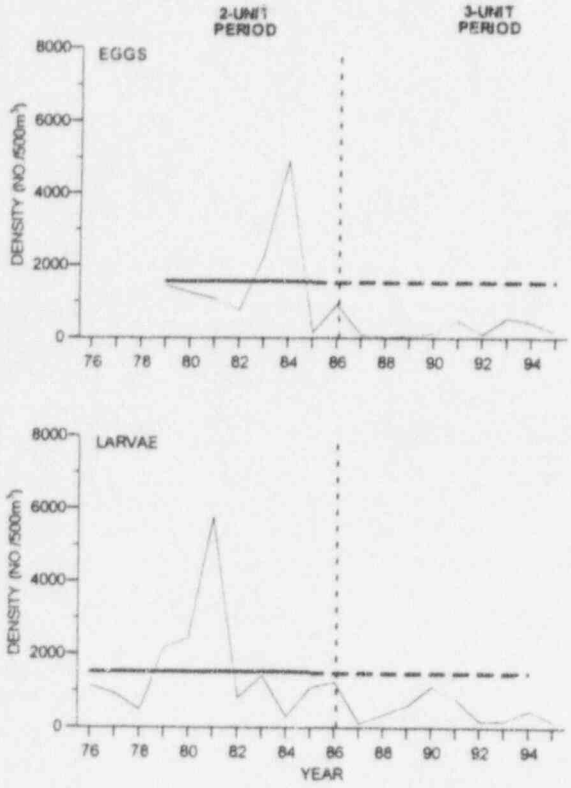

Fig. 3. Annual Δ-mean densities (no./500 m^3) of anchovy eggs and larvae at station EN during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean density calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance.

Since then, Δ-mean densities decreased by an order of magnitude and have remained relatively low, except for larvae in 1990. For both eggs and larvae, no significant trends were detected with the Mann-Kendall test during either the two- or three-unit operational periods.

The entrainment of eggs and larvae by MNPS probably represents the greatest direct operational impact on anchovies that spawn in the Millstone area. In 1995, the entrainment estimates for eggs (16 million) and larvae (186 million) were within historic ranges, but were among the lowest estimates of the past 20 years (Tables 4 and 5). Anchovies mature within a year and have a maximum life span of not more than 2 or 3 years (Stevenson 1958; Newberger and Houde 1995). Estimates of the abundance of eggs collected at station EN are representative of annual egg production. Therefore, a reduction in the adult spawning stock size due to plant operation should be readily evident if the production of eggs and larvae declines because of decreased adult abundance. Neither the number of eggs nor larvae entrained at MNPS were significantly correlated with the densities of eggs or larvae found the following year, implying no direct effect on the anchovy spawning stock. Although both egg and larval abundances declined in the mid-1980s, these decreases occurred prior to three-unit operation in 1986. In addition, the lack of negative trends during three-unit operational period suggests that MNPS has had minimal or no impact on anchovy adult abundance.

Silversides

The Atlantic and inland silversides are sympatric along most of the east coast of North America and reside in bays, estuaries, and salt marshes. The Atlantic silverside ranges from the Gulf of St. Lawrence to northern Florida (Conover 1992) and the inland silverside from Cape Cod to South Carolina (Johnson 1975). Both species are abundant, but in general the Atlantic silverside is more numerous than the inland silverside, except in low salinity waters (Bengtson 1984, 1985). Most silversides mature as yearlings and only live 1 to 2 years. Spawning begins at water temperatures of 9-12°C and occurs during the day at high tide on a semilunar cycle (Middaugh 1981; Conover and Ross 1982; Jessop 1983; Conover and Kynard 1984). Adhesive eggs are laid in shallow water on vegetation (Conover and Kynard 1984). Larvae are planktonic, but remain near spawning areas. Sex is indeterminate until fish reach 8 to 21 mm in length and sex ratio is affected by prevailing water temperature during development (Conover and Kynard 1981; Conover and Fleisher 1986). Growth of young is fast and mean lengths can exceed 90 mm by November, with fish from early spawns larger at any given age than late-spawned fish (Conover 1979; Bengtson et al. 1987). Atlantic silverside migrate offshore during winter, but remain in waters within 40 km of shore and in depths of less than 50 m (Conover and Murawski 1982). In contrast, inland silverside have small home ranges (Hoff 1972). High (>99%) overwintering mortality typically occurs (Conover and Ross 1982; Conover 1992). Both species are important trophic links between zooplankton and larger fishes and birds.

In the trawl monitoring program nearly all silversides collected have been the Atlantic silverside, with less than 0.1% identified as the inland silverside during the past 20 years. Since 1981, when the two species were first differentiated in the MNPS sampling, more than 80% of the silversides collected by seine were Atlantic silverside, although relative proportions varied from year to year. Silversides dominated summer shore-zone seine catches at JC (Appendix II) and were taken by trawl in winter (Appendices III-V), but were rarely found in ichthyoplankton collections. Potential impingement effects have been mitigated by the installation of return sluiceways. Therefore, any effect of the operation of MNPS is likely related to the influence of the thermal plume in Jordan Cove, which serves as a locally important spawning and nursery area.

Atlantic silverside were collected by trawl primarily at the three inshore stations (IN, JC, NR) currently sampled (Fig. 1). Most were taken from October through February after individuals withdrew from shoreline areas to overwinter in deeper waters. Annual trawl abundances fluctuated considerably (Table 8), with the Δ-mean CPUE for the past few years below the two-unit average at each of the stations (Fig. 4). Although recent trawl catches have been low, the only significant trend (slope of −3.2) was found for station NR ($p = 0.025$; Mann-Kendall test). During the two-unit operational period, a significantly ($p = 0.031$) negative slope of −2.0 was found for IN, probably a result of the high initial abundance found at that station in 1976-77.

TABLE 8. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of Atlantic silverside collected by trawl at selected stations from June 1976 through May 1996.

Year	IN	JC	NR
1976-77	15 ± 16	13 ± 20	77 ± 283
1977-78	29 ± 92	6 ± 612	10 ± 21
1978-79	60 ± 105	9 ± 8	2 ± 1
1979-80	42 ± 276	6 ± 17	4 ± 6
1980-81	8 ± 17	4 ± 5	3 ± 4
1981-82	6 ± 9	1 ± 0	6 ± 8
1982-83	2 ± 4	1 ± 2	12 ± 5
1983-84	2 ± 4	4 ± 1	1 ± 6
1984-85	2 ± 6	5 ± 11	1 ± 1
1985-86	7 ± 8	6 ± 8	3 ± 6
1986-87	5 ± 3	8 ± 7	110 ± 222
1987-88	3 ± 5	2 ± 2	15 ± 27
1988-89	2 ± 1	1 ± 0	25 ± 14
1989-90	1 ± 1	2 ± 2	12 ± 20
1990-91	1 ± 0	1 ± 0	17 ± 11
1991-92	12 ± 10	2 ± 1	19 ± 7
1992-93	115 ± 156	7 ± 2	24 ± 15
1993-94	5 ± 7	3 ± 2	3 ± 2
1994-95	1 ± 1	4 ± 3	2 ± 2
1995-96	2 ±1	2 ± 2	1 ± 1

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

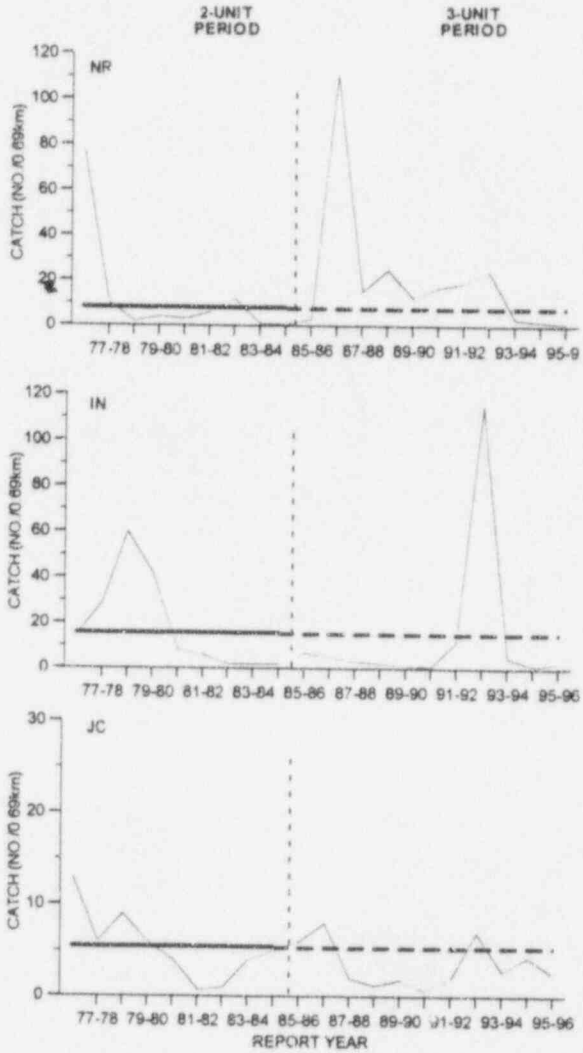

Fig. 4. Annual Δ-mean CPUE (no./0.69 km) of Atlantic silverside taken by trawl at stations NR, IN, and JC during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean CPUE calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ among the graphs).

The Δ-mean CPUE of Atlantic and inland silversides taken by seine during 1995 (87 and 60, respectively) were within historic ranges (Table 9). In 1995, the abundance of Atlantic silverside exceeded that of the inland silverside (Fig. 5). Nearly all annual Δ-mean CPUE values for inland silverside during three-unit operation were above or near the two-unit average, but abundance of Atlantic

TABLE 9. The annual Δ-mean[a] CPUE (no./30 m) and 95% confidence interval of Atlantic silverside and inland silverside collected by seine at JC from June 1981 through May 1996.

Year	Atlantic silverside	Inland silverside
1981	152 ± 251	3 ± 3
1982	114 ± 162	6 ± 16
1983	397 ± 598	88 ± 243
1984	29 ± 24	3 ± 2
1985	19 ± 12	4 ± 8
1986	172 ± 385	14 ± 21
1987	109 ± 90	3 ± 2
1988	96 ± 108	27 ± 54
1989	70 ± 93	14 ± 16
1990	83 ± 80	133 ± 234
1991	38 ± 11	74 ± 37
1992	78 ± 55	43 ± 27
1993	60 ± 73	5 ± 5
1994	37 ± 28	63 ± 64
1995	87 ± 73	60 ± 114

[a] Data seasonally restricted to June-November.

silverside has mostly been below the two-unit average (Fig. 6). These abundance trends for both species were supported by the results from a Mann-Kendall test, where the only significant trend ($p = 0.025$; slope of −18.6) was for the Atlantic silverside during three-unit operation.

To determine if a change in size distributions occurred after Unit 3 became operational, the length-frequencies for seine catches (expressed as percentages) were examined for the periods before and after three-unit operation (Fig. 7). The length-frequency distribution remained similar during these two operational periods, suggesting that no changes in growth resulted from increased thermal discharge into Jordan Cove.

The primary potential impact of the operation of MNPS on silversides is the incursion of the thermal

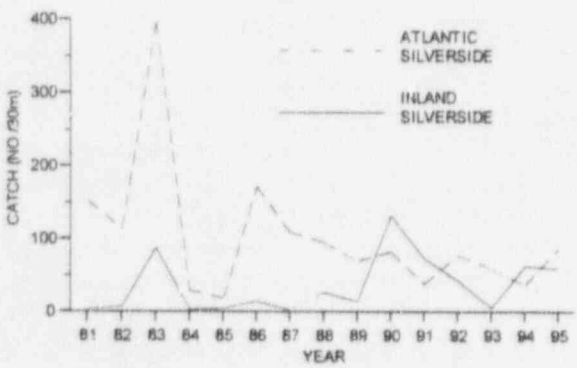

Fig. 5. Annual Δ-mean CPUE (no./30 m) of Atlantic silverside (dashed line) and inland silverside (solid line) taken by seine at station JC from 1981 through 1995.

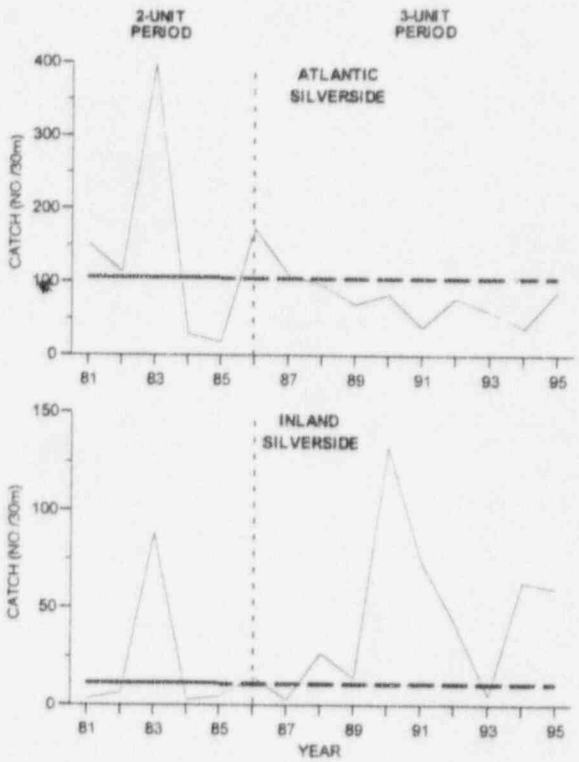

Fig. 6. Annual Δ-mean CPUE (no./30 m) of Atlantic silverside and inland silverside taken by seine at station JC during MNPS two-unit (1976-85) and three-unit (1986-95) operational periods. A Δ-mean CPUE calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ between the graphs).

plume into shore-zone area, causing potential disruption in spawning activity and in distribution of juveniles and adults. Annual abundance of inland silverside collected by seine has fluctuated over the years, but without apparent relation to operational periods. Atlantic silverside abundance in the shore-zone area showed a decreasing trend during three-unit operation. This could be related to MNPS operations, although both the predicted and measured (based on dye studies) maximum thermal increase is only 0.8°C at the site of station JC (NUSCO 1988b). Further, elevated summer temperatures in Jordan Cove appear to be more directly related to solar heating of the shallow sand flats (such as at the JC seine site) than to the MNPS thermal plume (see the Eelgrass section of this report). Therefore, it appears unlikely that a small increase in water temperature could affect the reproductive success of a species that ranges as far south as northern Florida. Also,

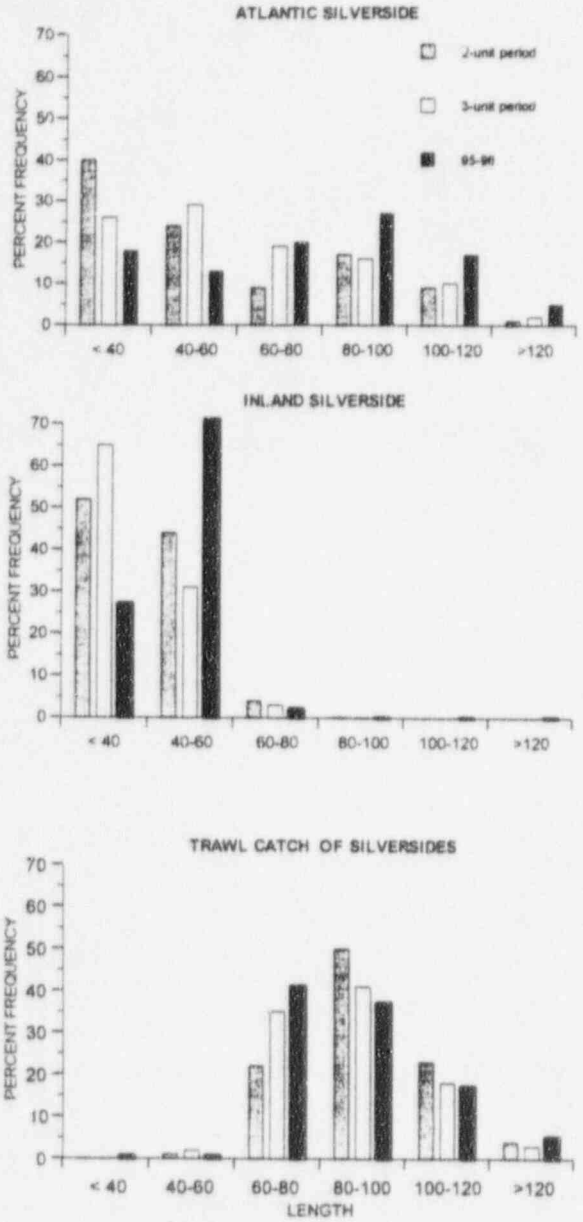

Fig. 7. Length-frequency distribution (20-mm length intervals) of Atlantic silverside and inland silverside taken by seine at station JC and Atlantic silverside taken by trawl at stations NR, IN, and JC during MNPS two-unit (1976-85) and three-unit (1986-95) operational periods and the 1995-96 report year (June-May).

abundance of adult silversides collected by the trawl monitoring program (an index of future spawners) did not show a similar pattern of abundance, suggesting that plant operation has not affected Atlantic silverside spawning population size near MNPS.

Grubby

The grubby is a demersal fish found in shallow waters along the Atlantic coast from the Gulf of St. Lawrence to New Jersey that tolerates a wide range of temperature and salinity (Bigelow and Schroeder 1953). Individuals reside in protected shallow water on mud or sand bottoms, peat reefs, and in eelgrass beds (Ennis 1969; Lazzari et al. 1989) and are found throughout the year in waters of LIS near MNPS. Similar to the winter flounder, grubby produce blood plasma antifreeze proteins and can remain active in very low water temperatures (Reisman et al. 1987). Grubby spawn throughout the winter and have a demersal, adhesive egg with an incubation time of 40 to 44 days at a water temperature of 4.6-6°C (Lund and Marcy 1975; Lazzari et al. 1989). Richards (1959) reported larvae present in LIS from February through April and Laroche (1982) noted that they are more abundant near the bottom than at the surface. A small species, the grubby has no sport or commercial value and, given its protective spines and cryptic coloration, it probably also has limited forage value. The grubby preys upon many small fishes and benthic invertebrates (Lazzari et al. 1989; Levin 1991).

The grubby is the fourth-most abundant larval fish collected at EN, accounting for 4.8% of the total from June 1976 through May 1996 (Table 1). The Δ-mean density of larvae for 1996 of 85 was within the range of historic data, but was the largest observed since 1988 (Table 10). Three-unit operational annual Δ-mean densities of larval grubby fluctuated about the two-unit average (Fig. 8). No significant temporal trend for larval grubby abundance was detected during either two- or three-unit operation when examined with the Mann-Kendall test. Despite a relatively high density in 1996, the entrainment estimate of 43 million larvae was less than the average of 49 million for these annual estimates because of the smallest cooling-water flow that occurred during the larval grubby season for any year of the three-unit period (Table 5).

Predominantly a shallow-water fish, the grubby was the fifth-most abundant fish taken by trawl at the three inshore stations; about 60% were caught at NR (Appendices III-V). In 1995-96, catches at all three stations fell below the two-unit Δ-mean average CPUE (Table 11; Fig. 9). The only significant trend found using the Mann-Kendall test was at station NR

TABLE 10. The annual Δ-mean[a] density (no./500 m^3) and 95% confidence interval of grubby larvae collected at EN from June 1976 through May 1996.

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
1995	43 ± 15
1996	85 ± 37

[a] Data seasonally restricted to February-May.

($p = 0.007$) during two-unit operation, which was increasing with a slope of 0.78.

The percent length-frequency distributions of grubby taken by trawl were similar before and after three-unit operation, although in 1995-96 more smaller fish were taken in comparison to the composite of either of the two operational periods (Fig. 10).

Because grubby eggs are adhesive, spawning in the vicinity of the MNPS intakes and subsequent hatching may directly affect the number of larvae collected at station EN. Entrainment of larvae is the primary direct plant impact on the resident grubby population. However, abundance of grubby has not declined during the three-unit operational period and this species has been among the most stable of the fishes residing near MNPS.

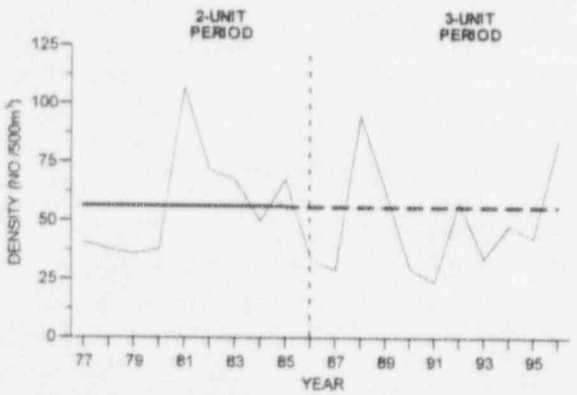

Fig. 8. Annual Δ-mean densities (no./500 m^3) of grubby larvae at station EN during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean density calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance.

TABLE 11. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of grubby collected by trawl at selected stations from June 1976 through May 1996.

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
1994-95	3.7 ± 1.6	2.9 ± 1.1	1.6 ± 0.6
1995-96	1.9 ± 1.0	0.7 ± 0.3	0.7 ± 0.2

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

Cunner

The cunner, found from Newfoundland to Chesapeake Bay (Scott and Scott 1988) is closely associated with structural habitats, such as rocks, pilings, eelgrass or mussel beds, and macroalgae. Cunner are inactive at night and when water temperatures fall below 5-8°C, they become torpid (Green and Farwell 1971; Olla et al. 1975; Dew 1976). Individuals maintain highly localized home ranges (Green 1975; Olla et al. 1975; Gleason and Recksiek 1988), may establish defended territories (Pottle and Green 1979b), and most, but not all individuals, do not undertake extensive movements (Green and Farwell 1971; Olla et al. 1979; Lawton et al. 1996). Most cunner live only 5 to 6 years, with maximum age likely about age-10, less than one-third of the life span of the closely related tautog (Dew 1976; Regan 1982).

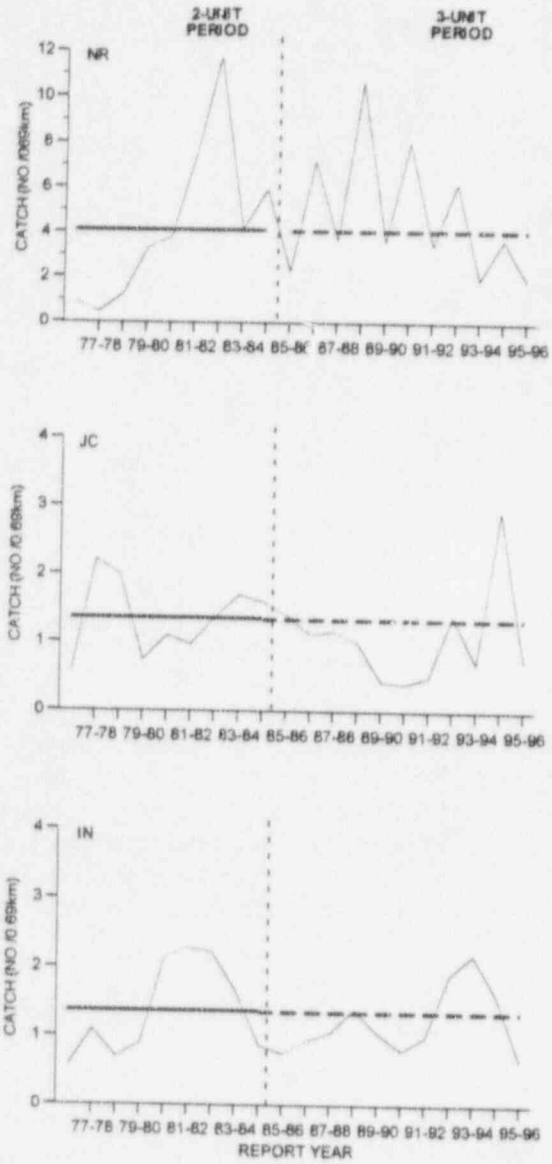

Fig. 9. Annual Δ-mean CPUE (no./0.69 km) of grubby taken by trawl at stations NR, IN, and JC during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean CPUE calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ among the graphs).

Cunner mature at age-1 to 2 and spawn during May through September from afternoon into the evening (Johansen 1925; Dew 1976; Pottle and Green 1979a; Green et al. 1985). Lawton et al. (1996) reported all cunner larger than 65 mm observed in western Cape Cod Bay to be mature. The pelagic eggs hatch in 2 to 6 days, depending upon water temperature

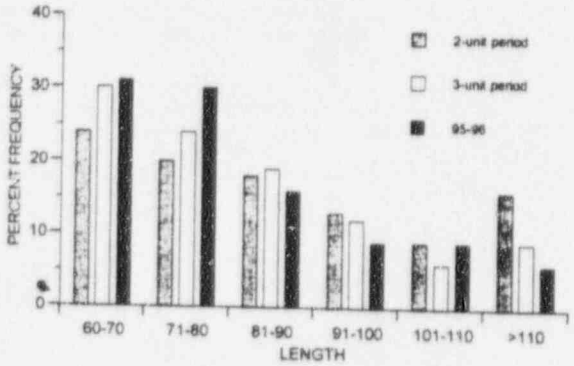

Fig. 10. Length-frequency distribution (10-mm length intervals) of grubby taken by trawl at stations NR, IN, and JC during MNPS two-unit (1976-85) and three-unit (1986-94) operational periods and the 1995-96 report year (June-May).

(Williams 1967; Dew 1976). Williams et al. (1973) noted that only about 5% of cunner eggs survive to hatching. Newly-hatched larvae are 2 to 3 mm in length, metamorphose by 10 mm, and settle into preferred habitats (Miller 1958; Levin 1991).

The cunner has no commercial value and is generally not sought after by sport fisherman, although numerous individuals are caught while fishing for other species (MacLeod 1995). Regionally, declining trends in adult cunner abundance have been observed in LIS (Smith et al. 1989), Cape Cod Bay (Lawton et al. 1994), and Mount Hope Bay (MRI 1994).

In the MNPS area, cunner eggs and larvae are present primarily from June through August. In these early developmental stages collected at station EN, cunner eggs have been most abundant of all egg taxa, whereas larvae have been less common, ranking only seventh overall (Table 1). Following a relatively large Δ-mean density of 7,057 for eggs in 1994, the Δ-mean density of 7,126 for 1995 was the third largest recorded (Table 12), as was the entrainment estimate of about 4.9 billion for 1995 (Table 4). During three-unit operation, eggs have increased in abundance from a low in 1986 and the Δ-mean in 1995 exceeded the two-unit average (Fig. 11). However, the 1995 larval Δ-mean density decreased about 50% from the previous year (Table 12). Except for a peak in 1991, annual larval abundances during three-unit operation remained below the two-unit average (Fig. 11). For larvae, no significant trends were detected using the Mann-Kendall test during either two- or three-unit operation. Cunner egg abundance varied without trend during the two-unit period, but

TABLE 12. The annual Δ-mean[a] density (no./500 m^3) and 95% confidence interval of cunner eggs and larvae collected at EN from June 1976 through May 1996.

Year	EGGS EN	LARVAE 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
1994	7,057 ± 3,315	25 ± 18
1995	7,126 ± 4,307	12 ± 9

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

showed a significant (p = 0.025) positive (slope = 277.0) trend during three-unit operation, possibly caused by the lowest density of the data series that was recorded in 1986 at the beginning of the three-unit period.

Juveniles and adult cunner are caught by trawl, mostly from spring through summer. Twice as many cunner were taken at IN (4,126) than at JC (2,063), with relatively few (416) found at NR (Appendices III-V). Relatively high Δ-mean CPUE values were recorded for IN from 1976 through 1981, followed by moderate abundance from 1982 through 1984, with annual Δ-mean CPUE becoming more similar at IN and JC (Table 13). Annual Δ-mean CPUE at both stations during three-unit operation remained considerably below the two-unit reference level (Fig. 12). The only significant decreasing trend (p = 0.002; Mann-Kendall test) occurred at station IN, with a slope of −3.0 found during two-unit operation.

To determine an age-frequency distribution of cunner collected by trawl, ages were assigned based on an age-length key provided by Serchuk (1972). Percent length-frequency distributions were determined for both the two- and three-unit periods and for 1995-96 (Fig. 13). The size distributions differed greatly between the two operational periods. Nearly 70% of the cunner caught during three-unit operation were young-of-the-year, but relatively high frequencies of older fish were taken prior to 1986. The length-frequency distribution during 1995-96

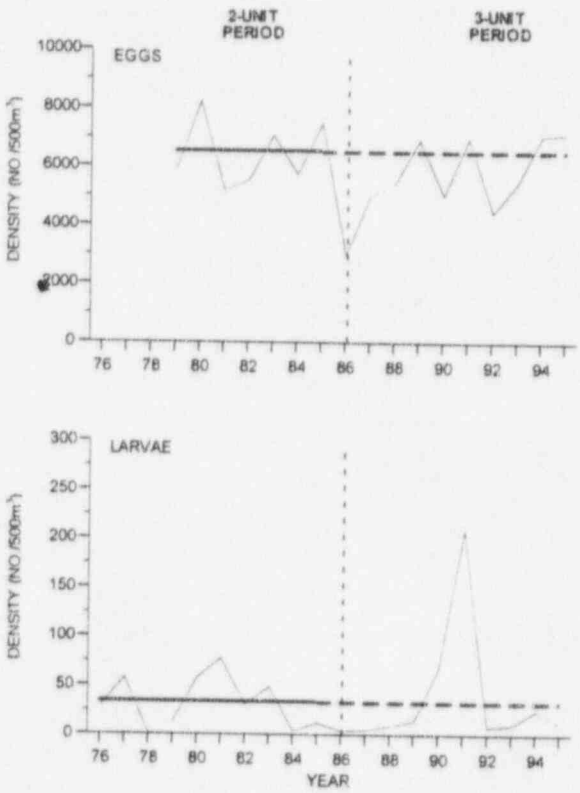

Fig. 11. Annual Δ-mean densities (no./500 m^3) of cunner eggs and larvae at station EN during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean density calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ between the graphs).

was very similar to the overall three-unit average size composition.

Although the abundance of trawl-caught cunner in the MNPS area decreased during the 1980s, particularly at IN, most of the decline occurred prior to three-unit operation. Contributing to the decrease at IN was the mid-1983 removal of a cofferdam that was in place during the construction of the Unit 3 intake structure. This rock cofferdam provided good habitat for cunner and increased their availability to sampling by trawl at the nearby IN station. After its removal, CPUE at IN decreased considerably and became more similar in magnitude to that of former trawl station NB (Fig. 14), which was located only about 500 m to the west in mid-Niantic Bay. The annual Δ-mean CPUEs at both stations from 1983 through 1994 were significantly correlated (Spearman's rank-order correlation coefficient r = 0.78; p = 0.0042).

TABLE 13. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of cunner collected by trawl at selected stations from June 1976 through May 1996.

Year	IN	JC
1976	26.0 ± 19.0	4.0 ± 2.0
1977	24.0 ± 23.0	3.0 ± 1.0
1978	6.0 ± 3.7	3.0 ± 1.4
1979	29.0 ± 23.0	9.0 ± 5.0
1980	23.0 ± 16.0	6.0 ± 2.0
1981	12.0 ± 10.0	5.0 ± 2.2
1982	5.0 ± 3.0	4.0 ± 2.0
1983	3.0 ± 1.3	4.0 ± 2.0
1984	2.0 ± 1.0	2.0 ± 1.0
1985	1.0 ± 0.6	1.0 ± 0.5
1986	0.1 ± 0.2	0.5 ± 0.4
1987	0.2 ± 0.2	0.4 ± 0.2
1988	0.3 ± 0.1	3.0 ± 3.4
1989	0.9 ± 0.4	0.8 ± 0.4
1990	0.4 ± 0.1	0.9 ± 0.2
1991	0.4 ± 0.1	2.3 ± 0.7
1992	1.0 ± 0.7	1.4 ± 0.5
1993	0.1 ± 1.1	1.4 ± 0.7
1994	0.4 ± 0.1	0.8 ± 0.5
1995	0.8 ± 0.4	1.3 ±1.1

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

The entrainment of eggs is the greatest potential impact of MNPS on the cunner population. However, if egg losses affected recruitment, then juvenile abundance should decrease in relation to older fish. This decrease was not apparent in the comparison of length-frequency distributions, which, conversely, indicated a relative increase of juveniles occurring during three-unit operation. The greatest effect of MNPS on cunner may have been the loss of habitat formerly provided by the Unit 3 cofferdam.

Tautog

General biology. The tautog ranges from New Brunswick to South Carolina, but is most common from Cape Cod to the Delaware Capes (Cooper 1965). Adult tautog prefer rocky areas and similar reef-like habitats near shore from spring through fall; juveniles are also found in eelgrass beds and among macroalgae in coves and estuaries (Tracy 1910; Bigelow and Schroeder 1953; Wheatland 1956; Cooper 1965; Briggs and O'Connor 1971; Hostetter and Munroe 1993). Tautog are active during the day, but are quiescent during night (Olla et al. 1974). During winter, adults move to deeper (25-55 m) water while juveniles remain inshore to overwinter in a torpid state (Cooper 1965; Olla et al. 1974). Tautog are long-lived with maximum age reported for males of 34 years and 22 years for females (Chenoweth 1963; Cooper 1965). Adult growth

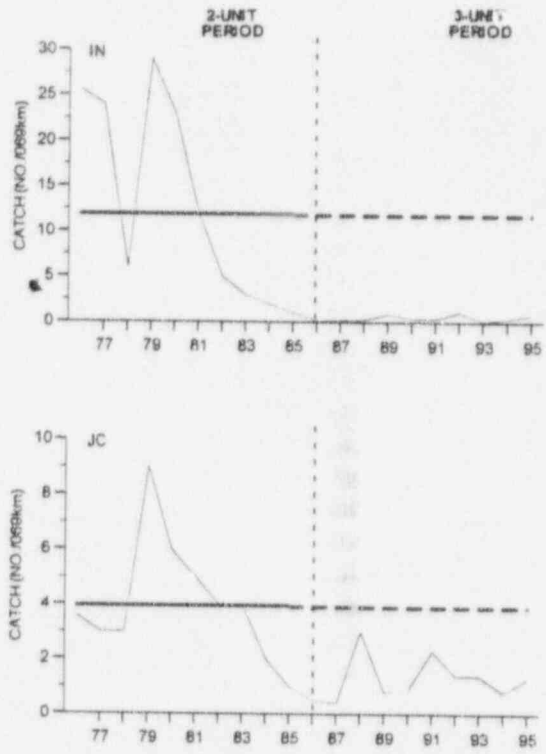

Fig. 12. Annual Δ-mean CPUE (no./0.69 km) of cunner taken by trawl at stations IN and JC during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean CPUE calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ among the graphs).

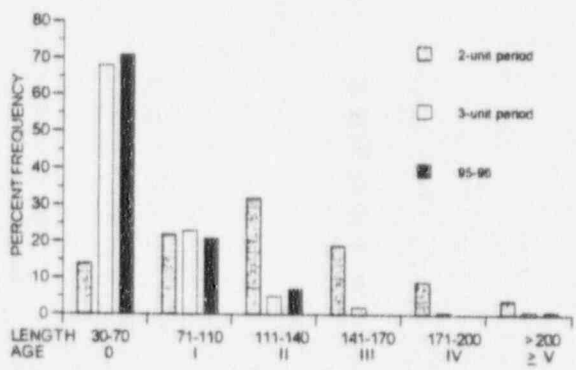

Fig. 13. Length-frequency distribution by length (mm) and age (determined from age-length key of Serchuk 1972) of cunner taken by trawl at stations IN and JC during MNPS two-unit (1976-85) and three-unit (1986-94) operational periods and the 1995-96 report year (June-May).

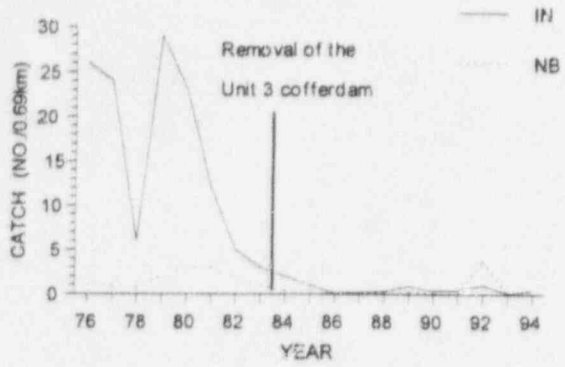

Fig. 14. Comparison of annual Δ-mean CPUE (no./0.69 km) of cunner taken by trawl at stations IN and NB from 1976 through 1994. The vertical line indicates the removal of the MNPS Unit 3 cofferdam in late summer of 1983.

rates have been estimated for several regions ranging from Narragansett Bay to Virginia (Cooper 1965; Simpson 1989; Hostetter and Munroe 1993).

Male tautog mature when 2 to 3 years old and females at age-3 to 4; fecundity at size and age was reported by Chenoweth (1963). Adults return to nearshore waters in spring prior to spawning, with a high proportion of fish returning to the same spawning area each year (Cooper 1965). Spawning occurs during afternoon or early evening hours from mid-May until mid-August in LIS (Wheatland 1956; Chenoweth 1963; Olla and Samet 1977, 1978). The pelagic eggs hatch in 42 to 45 hours at 22°C (Williams 1967; Fritzsche 1978). The pelagic larval stage lasts about 3 weeks and individuals settle on the bottom when they reach a size of about 17 mm (Sogard et al. 1992; Dorf 1994). Estimated growth rate during pre-settlement is about 0.75 mm per day and during post-settlement is about 0.5 mm per day (Sogard et al. 1992; Dorf 1994). Size at the end of the first growth season in Narragansett Bay (about 50 mm total length; Dorf 1994) was less than found in a southern New Jersey estuary (75 mm standard length; Sogard et al. 1992) and this was attributed to a longer growing season in southern waters.

Entrainment. Tautog were collected primarily as eggs in the ichthyoplankton entrainment program. Since 1979, eggs have ranked second in abundance from collections at station EN (Table 1). Tautog and cunner eggs are very similar in appearance, but their annual Δ-mean densities were not correlated (Spearman's rank-order correlation coefficient r = 0.108; p = 0.680). The 1995 Δ-mean density for tautog eggs of 2,074 fell within the range for the previous 16-year period, but was the highest abundance since 1990 (Table 14). Since the early 1990s, annual abundances have remained below the two-unit operational average (Fig. 15). During three-unit operation a significant (p = 0.040; Mann-Kendall test) decreasing (slope = −138.3) trend was detected for tautog eggs, despite increases in abundance occurring during recent years. No trend was present during the two-unit operational period.

In contrast to eggs, tautog larvae were not a predominant taxon, ranking eighth since 1976 (Table 1). Larval abundance appeared to rise and fall rapidly in relation to two peaks observed in 1981 and 1991. No correlation was found between annual abundances of tautog eggs and larvae (Spearman's rank-order correlation coefficient r = 0.101; p = 0.701) and no trends in larval abundance were found using the Mann-Kendall test during either two- or three-unit operational periods. However, abundances of tautog and cunner larvae were highly correlated (Spearman's rank-order correlation coefficient r = 0.891; p = 0.0001). Relative annual survival indices for tautog and cunner were calculated by dividing Δ-mean densities of larvae by those of eggs. The survival indices for tautog and cunner were also highly correlated (Spearman's rank-order correlation coefficient r = 0.853; p = 0.0001), indicating common processes that affected the recruitment of young for both of these wrasses.

TABLE 14. The annual Δ-mean[a] density (no./500 m^3) and 95% confidence interval of tautog eggs and larvae collected at EN from June 1976 through May 1996.

Year	EGGS EN	LARVAE 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
1994	1,650 ± 748	12 ± 8
1995	2,074 ± 704	8 ± 4

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

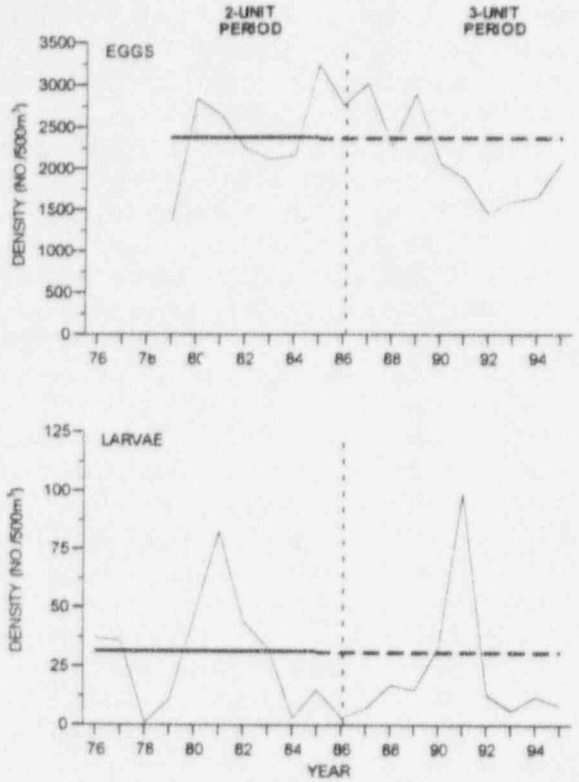

Fig. 15. Annual Δ-mean densities (no./500 m^3) of tautog eggs and larvae at station EN during MNPS two-unit (1976-85) and three-unit (1986-96) operational periods. A Δ-mean density calculated for the entire two-unit period is represented by a horizontal solid line that is extended as a dashed line through the three-unit period to serve as a reference level for abundance. (Note that the vertical scales differ between the graphs).

Trawl monitoring program. Tautog are caught infrequently by trawl because they prefer rocky or reef habitats and are less vulnerable to this sampling gear; annual Δ-mean CPUE cannot be calculated because of too many zero values. As an alternative, the annual (June-May report year) sum of catches at the trawl stations may be used as index of abundance. In contrast to 1994-95, when the total catch was the smallest recorded in the 20-year data series (NUSCO 1996), the combined catch at the three inshore stations during 1995-96 was the largest and was due to a relatively strong year-class of young (Table 15). The 128 tautog collected at NR was only one fish less than the series high in 1981-82; the 74 fish at JC also was the second highest catch for that station. However, since the mid-1980s, tautog have become less abundant at IN, which may also be related to removal of habitat (i.e., the rock cofferdam) as noted for the cunner. Although more

TABLE 15. Total annual catch of tautog collected by trawl at selected stations from June 1976 through May 1996.

Year	NR	JC	IN	Total
1976-77	39	71	63	173
1977-78	16	106	70	192
1978-79	30	59	86	175
1979-80	45	57	68	170
1980-81	25	22	47	94
1981-82	129	20	27	176
1982-83	90	37	50	177
1983-84	16	18	41	75
1984-85	11	15	46	72
1985-86	22	31	47	100
1986-87	110	57	23	190
1987-88	15	30	17	62
1988-89	57	36	42	135
1989-90	28	20	18	66
1990-91	105	40	16	161
1991-92	51	35	14	100
1992-93	24	91	9	124
1993-94	13	50	24	87
1994-95	14	20	17	51
1995-96	128	74	28	230

tautog were caught at NR, the catches in the Niantic River also varied to a greater degree (CV = 84%) than at the other two stations (both had a CV = 58%). No significant trends were found for the combined catch of tautog at the three inshore trawl stations during either the two- or three-unit operational periods. By station, the only significant trend was for JC during the two-unit operational period, when catch decreased over time (slope = –6.7; p = 0.006; Mann-Kendall test).

Length-frequency distributions of tautog caught by trawl, before and after three-unit operation and for the current year, are shown in Figure 16. Ages were

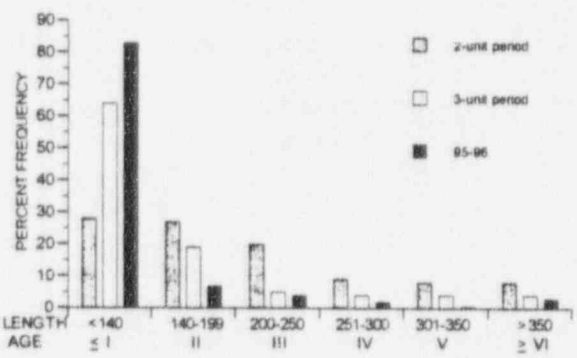

Fig. 16. Length-frequency distribution by length (mm) and age (determined from age-length key of Simpson 1989) of tautog taken by trawl at stations NR, IN, and JC during MNPS two-unit (1976-85) and three-unit (1986-94) operational periods and the 1995-96 report year (June-May).

assigned to length categories that were based on age-length data from LIS (Simpson 1989). Young-of-the-year tautog accounted for a higher proportion of the fish caught after three-unit operation began (64%) and in 1995-96 (83%) than during the two-unit period (28%), when proportionately more fish were seen in older age-classes. As noted previously, young tautog were relatively common in 1995-96.

Lobster pot sampling. Tautog have been routinely found in pots used in the lobster monitoring program (see the Lobster Studies section for details). Since 1988, these fish have been counted and measured to provide another index of tautog abundance. Total annual (May-October) catches at each of the three lobster monitoring program stations (Jordan Cove, designated herein as JC; Intake, IN; and Twotree, TT) were examined. From 1988 through 1993, annual catch was usually greatest at lobster pot station IN, followed by TT and JC (Fig. 17). However, beginning in 1994, catches at IN decreased relative to the other two stations. In 1996, catch at TT more than doubled from 1995 and at JC the number of tautog was about six times the previous high. In NUSCO (1996), catches at IN

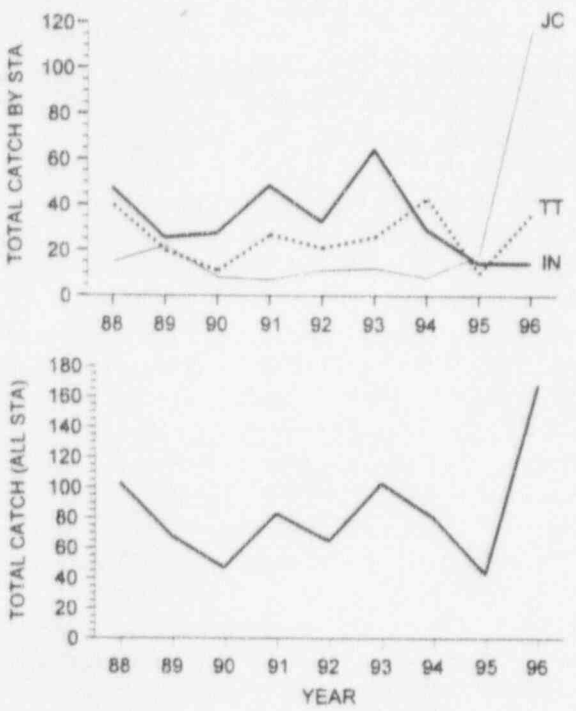

Fig. 17. Total catch of tautog in lobster pot sampling (May-October) by station (JC, IN, TT) and at all stations combined from 1988 through 1996. (Note that the vertical scales differ between the graphs).

were reported to be significantly greater than at the other two stations. However, because of the changes in abundance noted above, no significant differences were found among the stations for tautog abundance. Also, no abundance trends were detected using the Mann-Kendall test during the 1988-96 period at any particular station or for total lobster pot catch.

The reasons for the increased catch of tautog in the JC lobster pots in 1996 are unknown. A large blue mussel (*Mytilus edulis*) bed located on a ledge in the MNPS discharge area may have been greatly reduced following the shutdown of all three MNPS units as of March 30 that continued throughout 1996. Reduced prey availability as well as a lack of a thermal plume in the shallow Jordan Cove may have increased the number of tautog foraging in the cove, particularly near the rock outcrops where the lobster pots were set. In addition to greater catches during 1996, tautog entering the JC pots apparently attacked and killed or damaged a considerable number of trapped lobsters, which had unusually high injury rates this year (see Lobster Studies section for details). This was likely an indication of increased tautog attacks on lobster, perhaps in lieu of other prey.

Lobster pots should select for certain size-classes of tautog because the 2.5 cm^2 wire mesh should not retain smaller individuals when pots are hauled. Also, the 15-cm diameter of the funnel entrances restricts the entry of most larger individuals. The length-frequency distribution of tautog caught in lobster pots was dominated by fish from 200 to 349 mm, particularly at IN (Fig. 18). This size range primarily includes 3- to 5-year-old fish, ages during which both males and females become mature. Therefore, the lobster pot catches provide a reliable index of newly recruited adults. Relatively similar age structure was found at the three station locations, although fish in both smaller and larger size-classes adjacent to modal size-classes were taken more frequently at JC and TT than at IN.

Previous tautog early life history studies. Previous special studies on the hydrodynamics of LIS near MNPS and of tautog early life history (focusing predominantly on the egg stage) are summarized below with some additional comments. These studies were important to the design of the 1996 special field study that attempted to identify potential source areas for the tautog eggs entrained at MNPS.

Tautog eggs are pelagic and their dispersal from spawning sites in LIS is primarily by tidal transport. The number of eggs entrained by MNPS should be related to egg abundance in Niantic Bay, the source

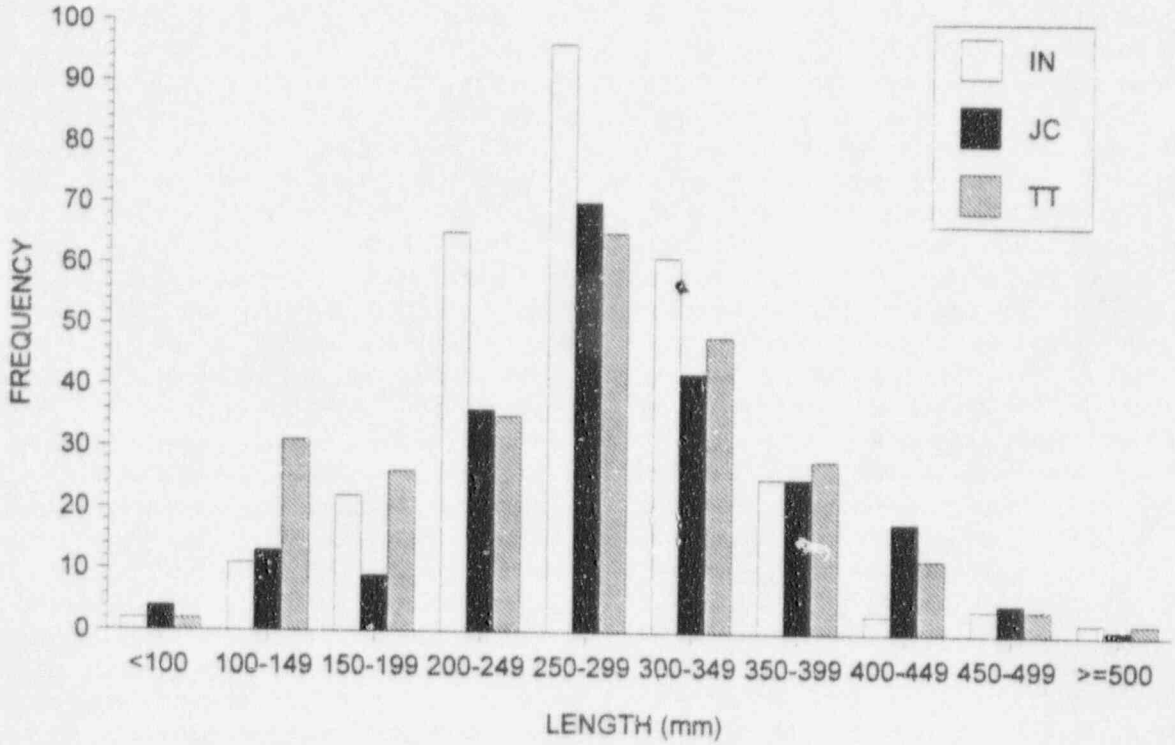

Fig. 18. Length-frequency distribution of tautog taken by lobster pot sampling at stations JC, IN, and TT from 1988 through 1996.

of condenser cooling water. Results from drogue studies in 1991 (see NUSCO 1992a for details) indicated that during ebb tide water from LIS enters Niantic Bay from the west, and conversely, during flood tide water enters from the east (Figs. 19 and 20). The distance a planktonic tautog egg is transported during a tidal stage should be directly related to tidal current velocity. Average current velocities for ebb and flood tidal stages in the area from the Connecticut River to the Thames River were estimated from information provided in tidal current tables (NOAA 1993) at five locations west of Niantic Bay and five locations to the east (Fig. 21). For simplicity, the duration of both ebb and flood stages was assumed to be 6 h. Estimated average tidal velocity was 0.94 knots during an ebb tide and 0.88 knots during a flood. Based on these velocities, a tautog egg could be transported 5.7 nautical miles (n mi) during an ebb tide and 5.3 n mi during a flood tide. Therefore, the potential source area for a tautog eggs entering Niantic Bay during a full tidal cycle lies within a radius of about 5 n mi, with a center at a mid-point between Black Point and Millstone Point. This would encompass a shoreline extending from about 2 n mi east of the Connecticut River to the Thames River. Based on this information, farfield studies of tautog egg abundance conducted during 1996 extended about 5 n mi from the mouth of Niantic Bay.

Based on MNPS monitoring data collected from 1979 through 1994, the annual temporal occurrence of tautog eggs in eastern LIS is generally from about early May through mid- to late September. This seasonal occurrence of tautog eggs is similar to that reported by Monteleone (1992) for Great South Bay, NY. The annual timing of peak spawning, as indicated by egg abundance, can be estimated from the inflection point of the Gompertz function (Eq. 1). From 1979 through 1994, peak spawning occurred during mid- to late June and appeared to be related to annual spring water temperatures. A significant ($p = 0.002$) negative relationship was found between May water temperatures and the estimated date of peak spawning (Fig. 22). The 1996 spring water temperatures were abnormally cool, with an average May water temperature at the MNPS intakes of 9.4°C, resulting in an estimated peak egg abundance during the latter portion of June to mid-July. This information was used as the basis for the timing of tautog egg studies conducted during 1996.

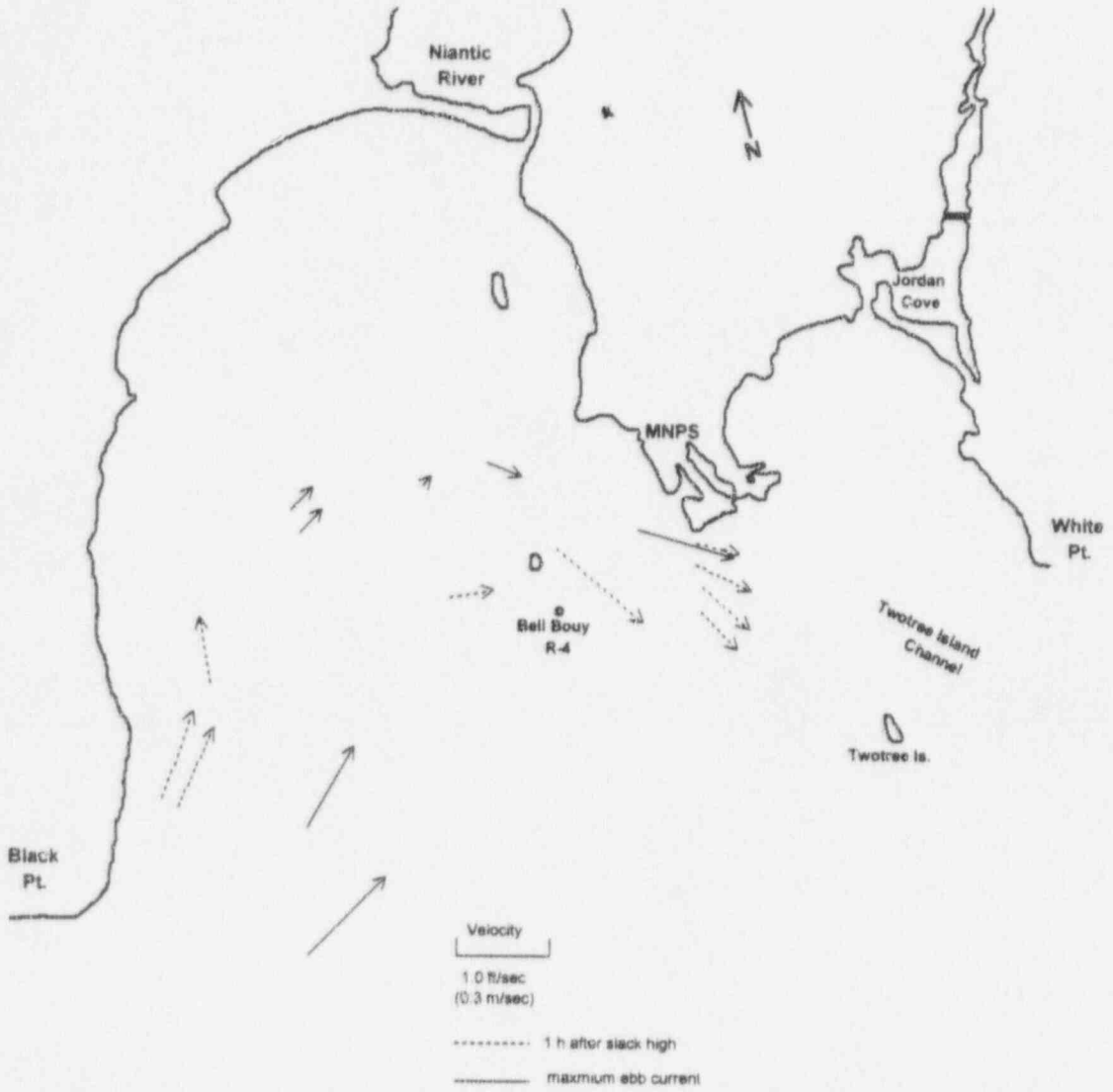

Fig. 19. Estimated tidal current direction and velocity in Niantic Bay for an ebbing tide during the first hour after high slack tide and at the time of maximum ebb current (based on results of drogue studies conducted in 1991; NUSCO 1992a). Note that the relative length of the arrows corresponds to the estimated average current velocities.

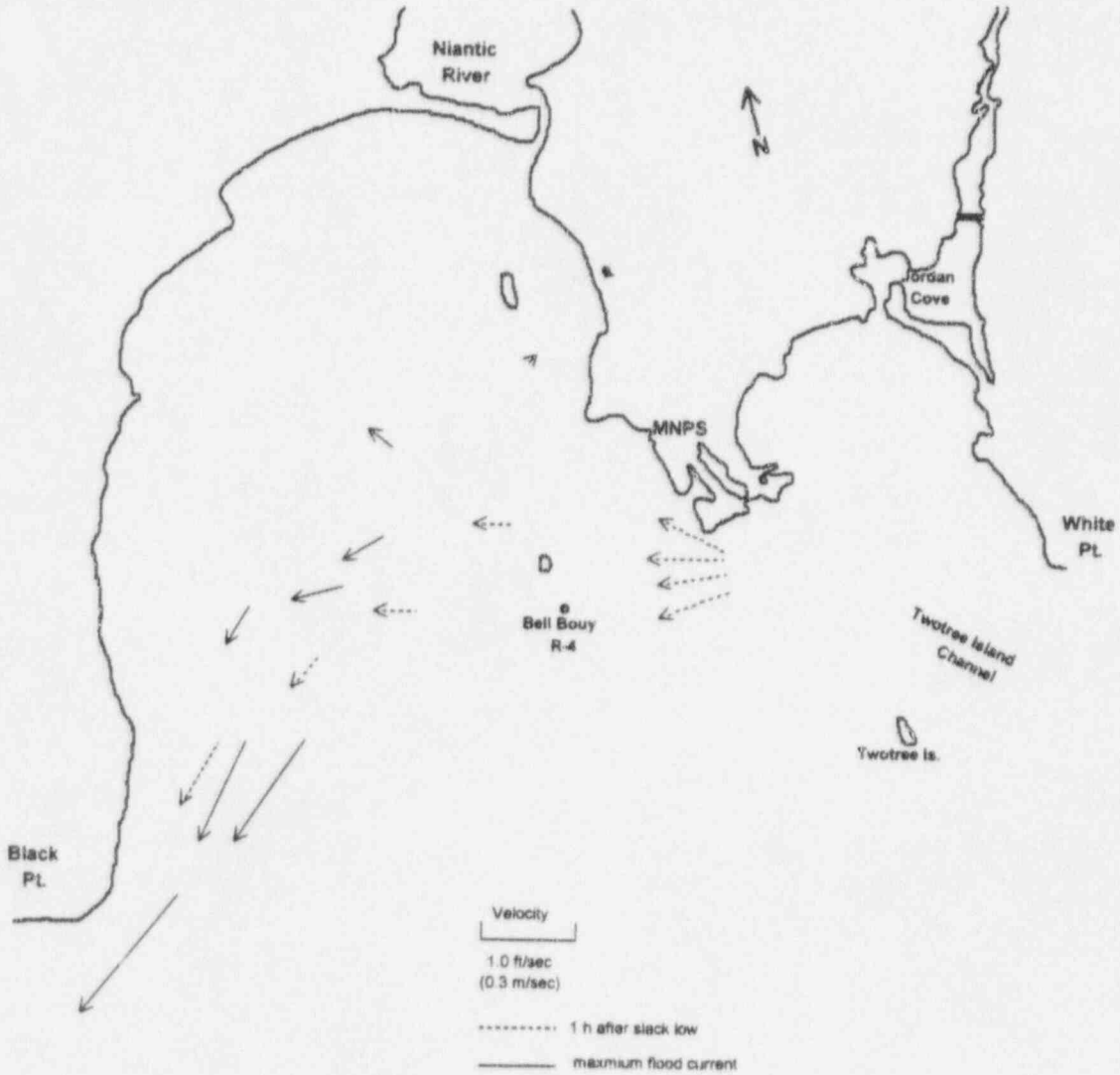

Fig. 20. Estimated tidal current direction and velocity in Niantic Bay for a flooding tide during the first hour after low slack tide and at the time of maximum flood current (based on results of drogue studies conducted in 1991; NUSCO 1992a). Note that the relative length of the arrows corresponds to the estimated average current velocities.

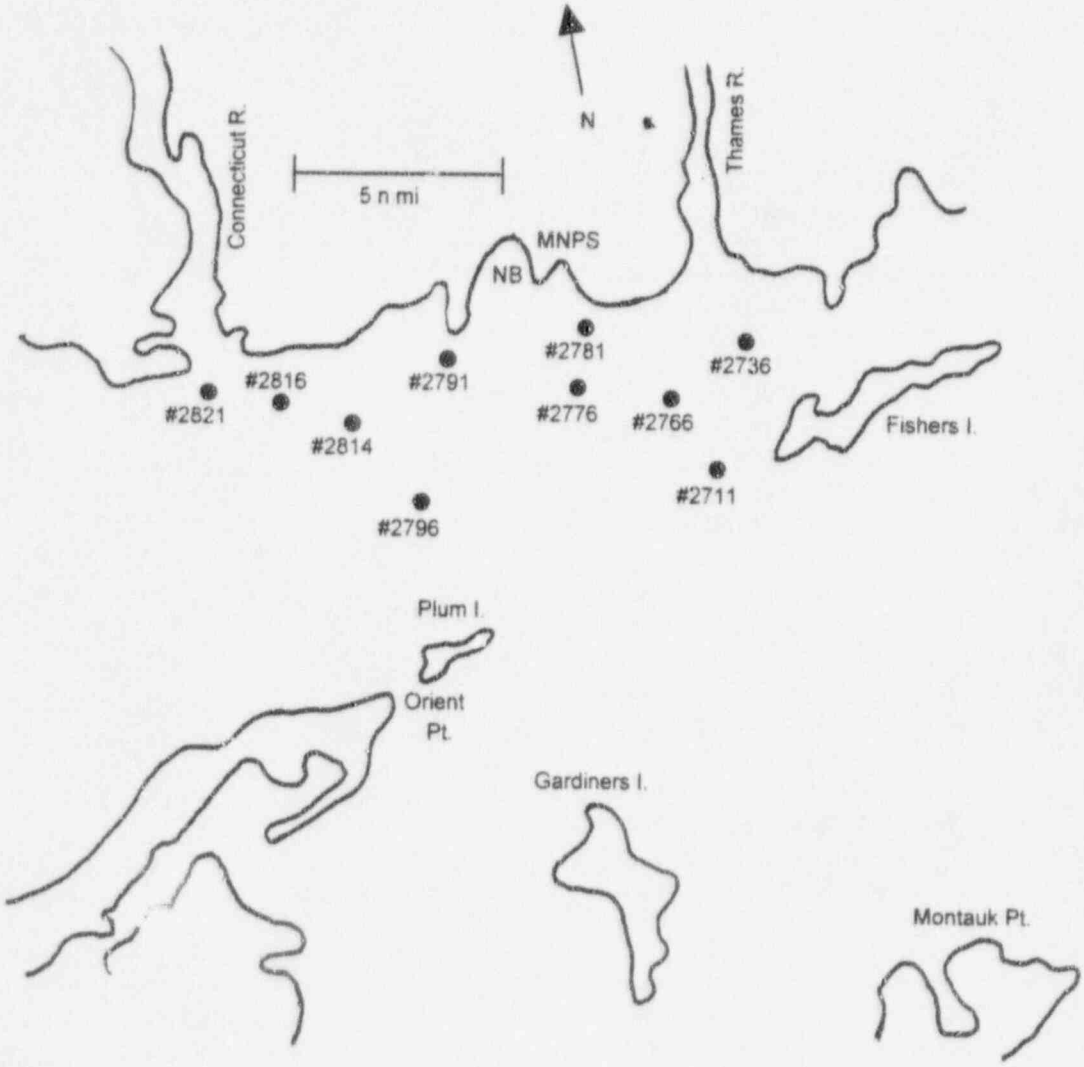

Fig. 21. Approximate location of sites used to estimate average current velocities during ebb and flood tides, including the NOAA tide current tables reference numbers (NOAA 1993).

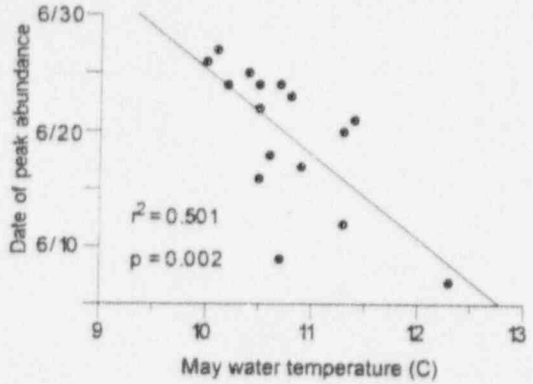

Fig. 22. Relationship between annual mean May water temperature (°C) at the MNPS intakes and the date of peak abundance of tautog eggs from 1979 through 1994.

Comparison of results from routine day and night entrainment collections at MNPS indicated that tautog eggs were more abundant at night. Therefore, 24-hour entrainment studies were conducted during 1993 to examine the diel change in abundance (NUSCO 1994a). Three 24-hour periods (June 8-9, June 15-16, and July 19-20 in 1993) were sampled at 2-hour intervals using the entrainment sampling methodology described previously.

The pattern of change in tautog egg abundance every 2 hours over a 24-hour period showed very consistent results on the three study dates (Fig. 23). In general, egg abundance peaked at about 1800 h, decreased through the night, remained relatively constant until late afternoon, and increased rapidly during the evening. This pattern indicated a short, early evening spawning period, consistent with laboratory observations of Olla and Samet (1977). The timing of peak abundance was not related to tidal stage because sampling in 1993 for the June 8-9 and June 15-16 studies occurred during opposing tidal stages. The rapid decline in abundance from 1800 to 2200 h cannot be attributed to hatching, as egg incubation takes longer than 1 day. Therefore, this decline was probably due to high natural egg mortality, likely from predation, as was suggested for eggs of the cunner, a sympatric species (Williams et al. 1973). Natural mortality, which likely accounts for the rapid decline in tautog egg abundance, from peak spawning at 1800 h through 0200 h was about 70% and through 0600 h was 80%. This information, suggesting high natural egg mortality during the first 12 hours after spawning, followed by a reasonably stable abundance, was used in designing the 1996 sampling program.

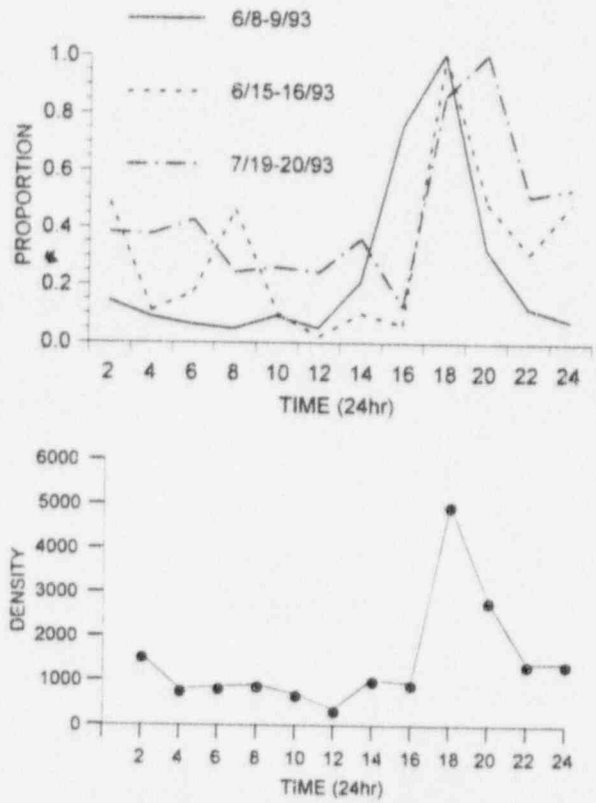

Fig. 23. Daily proportional abundance (top; expressed as sample density/maximum sample density for each date) of tautog eggs for three 24-hour studies conducted in 1993, and the geometric mean density of the three studies combined (bottom).

The nearfield spatial distribution of tautog eggs was examined from Black Point to Twotree Island Channel and in Niantic Bay in 1994 (NUSCO 1995) with five stations sampled (Fig. 24). The offshore stations (BP, LI, NB, and SS) were sampled with the 60-cm bongo system (333-μm mesh nets) using a stepwise oblique tow pattern for a 6-minute duration with equal sampling time at surface, mid, and near-bottom depths. Station EN was sampled using the previously described entrainment gantry system. 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 of tautog eggs would represent those 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. Samples were collected during the period of 0500 to 1100 h. This time period was selected because 24-hour studies conducted in 1993 showed that tautog

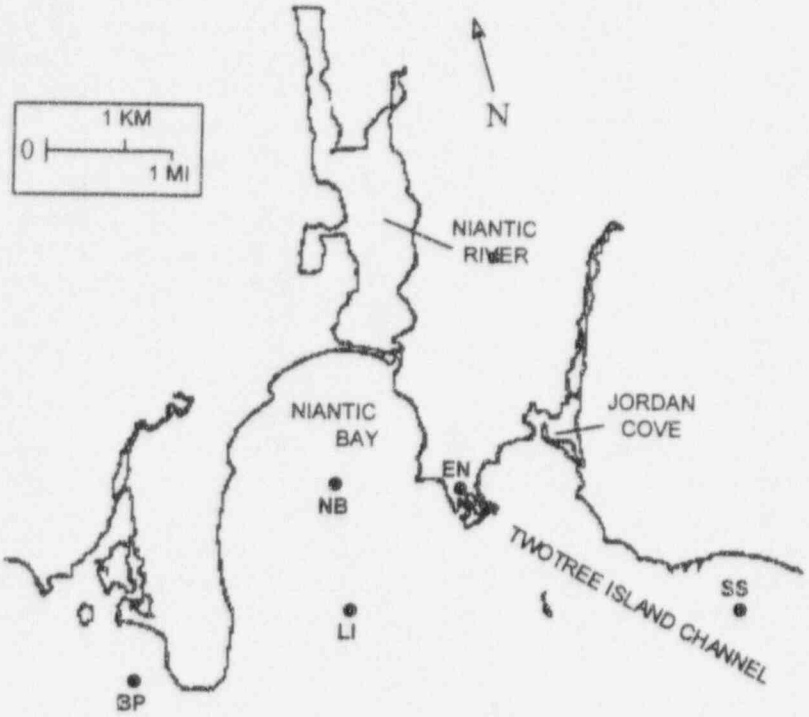

Fig. 24. Location of ichthyolankton stations (*) sampled for nearfield spatial distribution of tautog eggs during 1994.

egg densities remain relatively stable at this time of day (Fig. 23). Stations EN and NB were sampled at approximately at the same time. The collection sequence of stations sampled 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. By sampling NB second in the sequence (with EN sampled almost simultaneously), the sampling interval between EN and the other three stations was minimized. Sampling dates in 1994 were June 23 and 24 during a flood tide and June 29 and 30 during an ebb tide. These dates occurred during peak density of tautog eggs. On each sampling date, three sequences of samples were taken (LI, NB, EN, BP during an ebb tide and LI, NB, EN, SS during a flood tide), with the first sequence starting about 1 hour before maximum tidal current, the second starting at near maximum current, and the third immediately after the second was completed.

The results of this nearfield study of tautog egg abundance indicated that the geometric mean densities of tautog eggs at each station were similar and had overlapping 95% confidence intervals (Fig. 25). 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-rank test. Although the number of paired comparisons was rather low (12 pairs for NB and LI and 6 for SS and BP), no significant ($p < 0.05$) differences were detected between station EN and the other four stations. These nearfield data indicated that eggs were not concentrated near MNPS and entrainment densities of tautog eggs were representative of a more homogenous distribution, including areas outside of Niantic Bay.

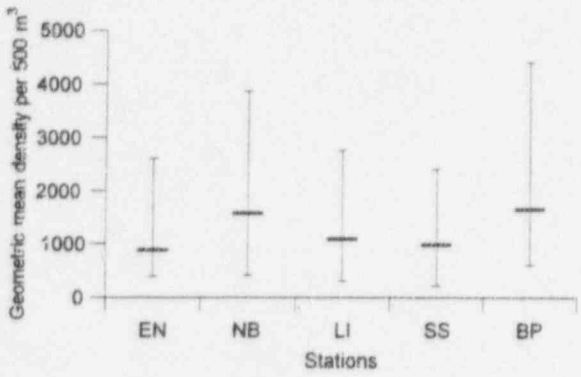

Fig. 25. Comparison of nearfield spatial distribution of tautog eggs in the vicinity of MNPS, based on geometric mean densities with 95% confidence intervals for each station sampled (see Fig. 24) during June 1994.

Tautog early life history studies in 1996. The Draft Fisheries Management Plan for Tautog (ASMFC 1996) stated that tautog eggs are primarily found near the surface, the basis for which apparently was the results of a study conducted in Narragansett Bay by Bourne and Govoni (1988). A portion of that bay is a two-layered estuary and pelagic tautog eggs may concentrate near the halocline. In contrast, the water column in eastern LIS is relatively homogenous for salinity (and temperature), with no halocline evident. However, even under these conditions (i.e., eggs tending to concentrate near the surface), the effect of winds on their transport would need to be considered along with tidal currents. Therefore, to examine the vertical distribution of tautog eggs, paired surface and near-bottom tows were taken in 1996 at a point midway between Millstone and Black Points. This location also was near the point of origin for the transects used in the farfield spatial distribution study discussed below. The abundance data for the vertical distribution study were also used to estimate tautog egg mortality given below. The densities of tautog eggs were greater at the surface than near-bottom for all 12 paired comparisons, even though vertical temperature and salinity measurements showed no apparent water column stratification (Table 16). Examination of the geometric mean densities of the three replicate pairs indicated that differences between surface and near-bottom were much greater for collections made in the evening just after spawning than for collections during the morning, approximately 12 hour after spawning (Fig. 26). Tidal currents did not appear to affect this pattern. Relatively similar egg densities were found during the evening from near-bottom collections and in the morning at both surface and near-bottom relative to densities at the surface during the evening. This pattern of densities, in addition to the large difference in tautog egg abundance between surface and near-bottom collections during evening and the reduction in density at the surface from evening to morning, suggested selectively higher mortality for eggs near the surface during the 12-hour period following spawning.

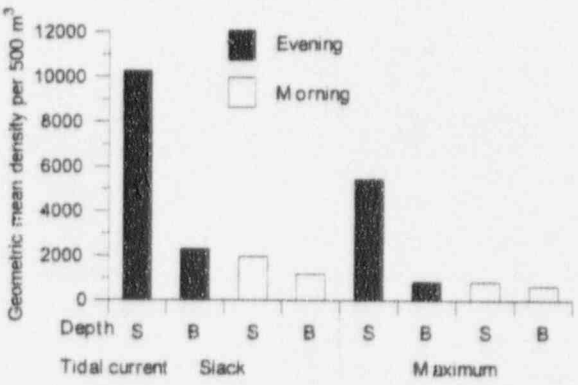

Fig. 26. Comparison of geometric mean densities of tautog eggs from paired surface and near-bottom samples collected during the evening and morning at the time of slack and maximum tidal currents from a site midway between Millstone and Black Points during July 1996.

TABLE 16. Time of sampling, tidal current stage, water temperature, and salinity when paired surface and near-bottom collections were taken to compare the vertical distribution of tautog eggs at a site midway between Millstone Point and Black Point during July 1996.

Sampling time (h)	Tidal current	Temperature (°C)			Salinity (ppt)		
		Surface	Mid	Bottom	Surface	Mid	Bottom
2005-2105	Slack	17.6	17.6	17.6	29.2	29.3	29.3
0832-0939	Slack	17.6	17.3	16.9	29.5	29.5	29.6
2005-2046	Maximum	19.1	17.5	17.0	28.0	29.0	29.4
0822-0914	Maximum	18.2	17.5	17.1	27.7	28.8	29.2

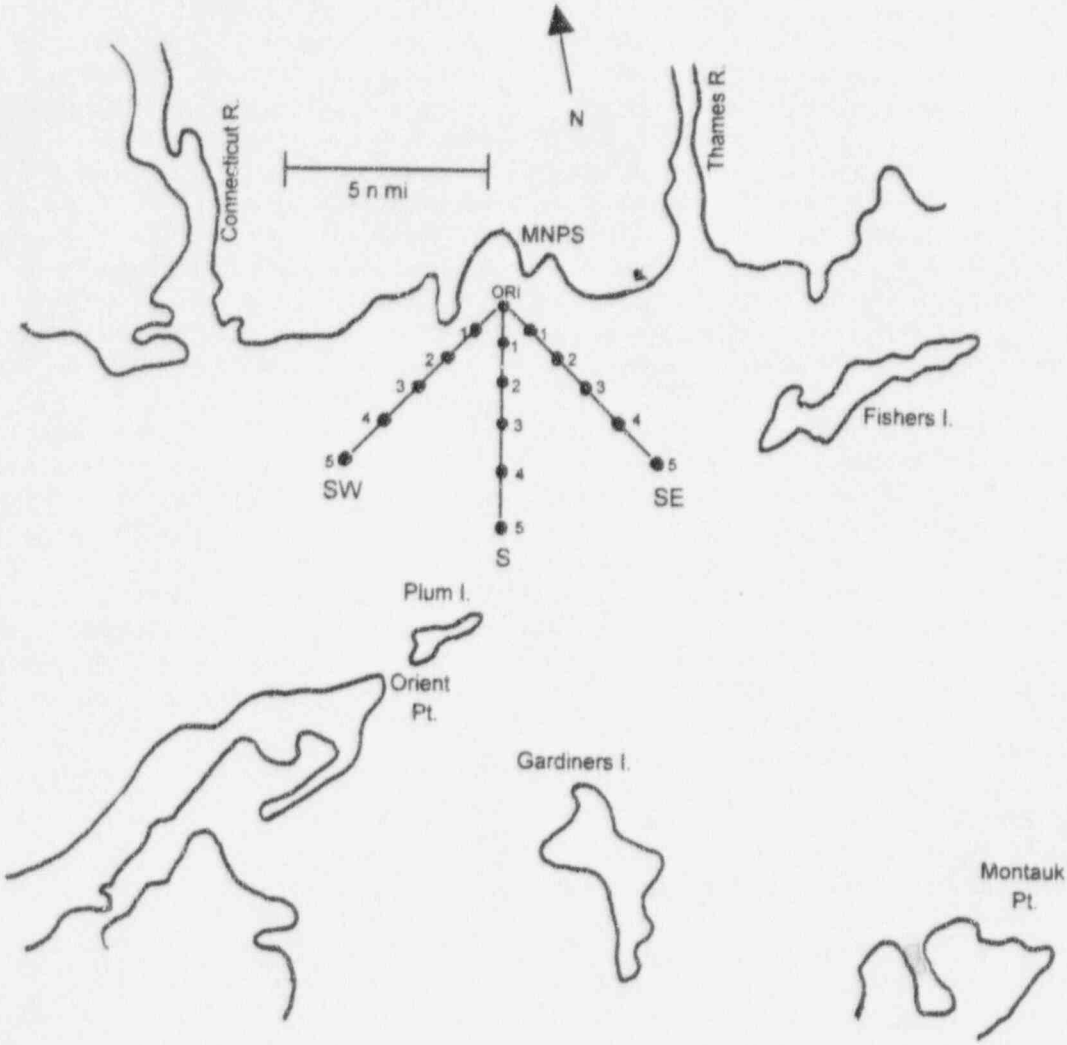

Fig. 27. Sampling sites for the farfield (within about 5 n mi of MNPS) spatial distribution study of tautog egg abundance conducted in 1996. Three transects (SW, S, SE) extended from a common origin (station ORI), with stations spaced at 1 n mi intervals.

Tautog egg mortality during about the first 12 hours after spawning may be estimated if the combination of surface and bottom densities (geometric mean of 6 samples) is assumed to be representative of abundance at the time of collections. Under this assumption, there was about 65% mortality (68.9% for slack current collections and 64.4 % for maximum current collections) during the 12-hour period. This mortality estimate, although large for a 12-hour period, was less than the approximate 80% mortality estimated from the 24-hour studies summarized above.

No information was available for tautog egg abundance further offshore of Niantic Bay than the nearfield study completed in 1994. In 1996, sampling was extended to the 5 n mi boundary discussed above to encompass a potential source area of tautog eggs entrained at MNPS. Sampling sites were at 1 n mi intervals along three separate transects (Fig. 27). The point of origin (station ORI) of each transect was at the mid-point between Millstone Point and Black Point, which was also used in the vertical distribution study. For this farfield study, sampling took place in the morning after sunrise because results of the 1993 24-hour studies showed that tautog egg densities remained relatively stable during this time of day (Fig. 23).

Abundances of tautog eggs, expressed as density per 500 m^3, for both sampling dates at station ORI were generally similar (geometric mean of three replicates), with no consistent trends evident between dates or along transects (Fig. 28). On July 2, the greatest abundances of eggs were collected along transect SE at n mi 2 through 4, but the remaining two transects had relatively similar densities. On July 9, abundances were more similar among transects with slightly higher densities at n mi 1 and 2 for transects SE and SW. For both dates no clear nearshore to offshore gradient of tautog egg densities was found.

The depths of the sampling sites varied greatly, ranging from 8 to 58 m (Tables 17 and 18). Water temperature and salinity measurements at surface, mid-depth, and bottom were similar at each site, indicating a relatively well-mixed water column throughout the 5 n mi radius from Niantic Bay. Due to the large variation in water depths among sampling sites, tautog abundance indices were recomputed to give the number of eggs under 1 m^2 of sea surface at each sampling site (Fig. 29). This abundance index suggested greater similarity among transects than the volumetric density estimates, with no obvious nearshore to offshore trends evident. This lack of trends and no indication of preferred spawning areas may have been due to the time of sampling. Results of previous 24-hour studies showed that tautog adults spawn primarily during the evening. The time period of sampling for the farfield spatial distribution was during the morning, which occurred a full tidal cycle after spawning and allowed for relatively complete mixing and distribution of eggs by tidal currents.

An instantaneous standing stock within the 5 n mi radius of Niantic Bay was calculated to estimate the number of tautog eggs that could be potentially entrained by MNPS from tidal transport. The geometric mean density of all 16 stations combined was calculated and extrapolated to a total number of eggs based on the average depth of the stations sampled. The estimated number of tautog eggs within a 5 n mi radius of Niantic Bay during the time period of sampling was 4.9×10^9 on July 2 and 3.1×10^9 on July 9. These daily egg standing stock estimates equaled or exceeded the estimated annual total number of tautog eggs entrained at MNPS since 1979, which ranged from 0.7 to 3.9×10^9 (NUSCO 1996). In addition, these standing egg stock estimates represented the number of eggs approximately 12 hours after peak spawning. Based on the results from 24-hour studies and evening-morning abundance comparisons from vertical distribution studies, the standing stock estimates only accounted for about 20 to 35% of the spawn from the previous evening because of natural mortality.

The daily standing stock estimates were also compared to an average lifetime fecundity estimate for female tautog (Table 19). Parameters used to estimate average lifetime fecundity under 1996 conditions were length at age for LIS (Simpson 1989), fecundity at age (Chenoweth 1963), fraction of mature females at age (Chenoweth 1963), and natural (M = 0.15) and fishing (F = 0.54) mortality rates from ASMFC (1996). In addition, present fishing regulations were used, including a 14 in (356 mm) legal size limit with a natural mortality of M = 0.15 and a discard mortality of F = 0.04 (D. Simpson, CT DEP, Old Lyme, CT, pers. comm.) for fish less than the legal size limit. Lifetime fecundity was estimated to be 142,655 eggs per female. The daily egg standing stock was adjusted for mortality that occurred during the 12 hours after spawning by using

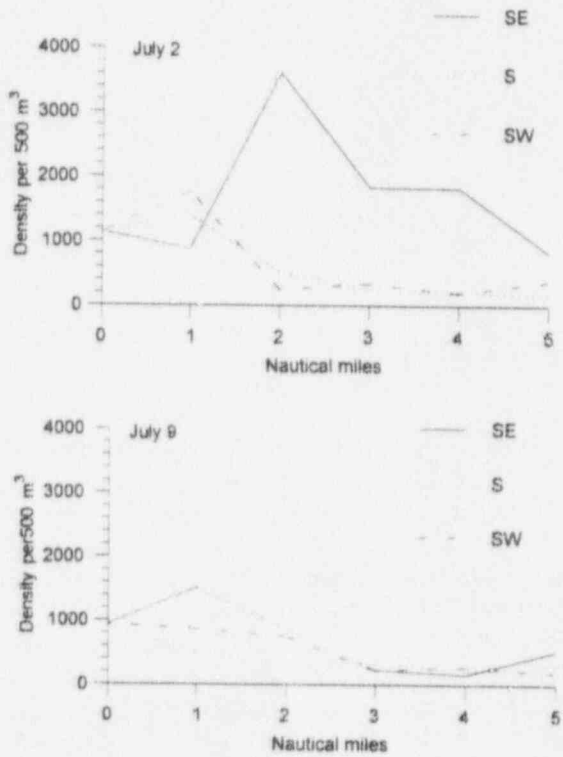

Fig. 28. Comparison of tautog egg densities along three transects sampled at 1 n mi intervals on two dates in July 1996. Nautical mile 0 is station ORI (see Fig. 27), the origin of the three transects.

TABLE 17. Station depth, water temperature and salinity for the July 2, 1996 collections to examine farfield spatial distribution of tautog eggs.

Station	Depth (m)	Temperature (°C)			Salinity (ppt)		
		Surface	Mid	Bottom	Surface	Mid	Bottom
ORI[a]	8	16.3	16.5	16.3	28.7	28.5	28.6
SE1	11	16.1	16.0	16.0	28.7	28.7	28.7
SE2	13	16.4	16.4	16.1	28.4	28.5	28.7
SE3	18	17.1	16.1	16.1	27.2	28.8	28.8
SE4	22	16.5	16.1	16.0	28.3	288.8	28.8
SE5	20	16.0	15.9	15.8	28.8	28.8	29.0
S1	20	16.3	16.2	16.1	28.4	28.5	28.7
S2	24	16.1	15.9	16.0	28.7	28.9	28.8
S3	45	16.2	15.9	-[b]	28.5	28.8	-
S4	58	16.6	16.1	-	27.6	28.3	-
S5	52	16.7	16.4	-	27.6	27.9	-
SW1	28	16.5	16.3	16.4	28.6	28.6	28.6
SW2	29	16.5	16.2	16.2	27.7	28.3	28.4
SW3	34	16.6	16.2	16.2	27.4	28.2	28.3
SW4	37	16.6	16.2	16.1	27.4	28.3	28.5
SW5	49	16.8	16.3	-	27.3	27.9	-

[a] Parameters for station ORI are a mean of three collections during the sampling date.
[b] No temperature and salinity measurements because the bottom depth was greater than the length of the probe cable.

TABLE 18. Station depth, water temperature and salinity for the July 9, 1996 collections to examine farfield spatial distribution of tautog eggs.

Station	Depth (m)	Temperature (°C)			Salinity (ppt)		
		Surface	Mid	Bottom	Surface	Mid	Bottom
ORI[a]	8	17.5	17.4	17.4	29.4	29.5	29.5
SE1	12	17.5	17.3	16.9	29.4	29.5	29.7
SE2	12	17.2	17.1	16.8	29.5	29.5	29.7
SE3	17	17.2	16.4	16.5	29.0	29.9	29.9
SE4	23	17.2	16.4	16.0	29.0	29.8	30.2
SE5	19	17.7	16.6	16.3	28.6	29.7	29.9
S1	21	17.1	16.8	16.8	29.3	29.6	29.6
S2	24	17.2	16.8	16.4	29.2	29.6	29.9
S3	47	16.8	16.1	-[b]	29.6	30.1	-
S4	55	16.9	16.4	-	29.4	29.8	-
S5	51	17.9	16.3	-	28.2	29.9	-
SW1	29	17.4	16.9	16.7	29.3	29.7	29.7
SW2	30	17.1	17.0	16.9	29.3	29.6	29.7
SW3	34	17.0	16.5	16.3	29.1	29.8	30.0
SW4	37	17.8	16.8	16.4	28.2	29.4	29.8
SW5	50	18.0	16.3	-	28.1	30.0	-

[a] Parameters for station ORI are a mean of three collections during the sampling date.
[b] No temperature and salinity measurements because the bottom depth was greater than the length of the probe cable.

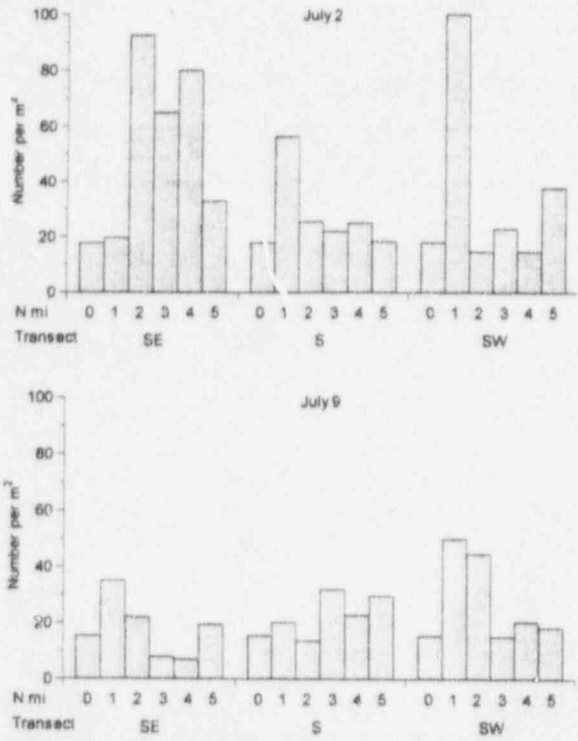

Fig. 29. Comparison of tautog egg abundances based on the number of eggs under 1 m² of sea surface along three transects sampled at 1 n mi intervals on two dates in July 1996. Nautical mile 0 is station ORI (see Fig. 27), the origin of the three transects.

65% from the vertical distribution study and 80% from the 24-hour study. Equivalent-female spawners for the July 2 study ranged from 98,139 to 171,743 and for the July 9 study ranged from 62,388 to 108,654. These numbers of equivalent-female spawners in the 5 n mi radius are conservative (i.e., low) because tautog are serial spawners and may spawn over an extended period (Olla and Samet 1977).

Summary. The greatest direct impact of MNPS on tautog stocks is most likely the entrainment of eggs. Large (65-80%) decreases in egg abundance occur following early evening spawning through the following morning, most probably from high natural mortality. Pelagic tautog eggs disperse rapidly from spawning sites by tidal transport and densities in nearshore areas are relatively uniform. Based on hydrodynamics, a conservative measure of the source area for eggs entrained at MNPS includes a radius of about 5 n mi. Two daily estimates of the instantaneous standing stock of tautog eggs within this area equaled or exceeded annual entrainment estimates at MNPS and, in fact, would have been even larger if high egg mortality rates had been taken into account. This implies that MNPS entrainment effects may be relatively small.

If egg losses due to entrainment affected recruitment, then juvenile abundance should decrease and the relative abundance of older fish would appear to increase in the short term. Based on length-frequency distribution from trawl catches, the percentage of juvenile tautog increased during the three-unit operational period. Therefore, changes in the relative proportion of juveniles and adults were probably unrelated to entrainment losses. In addition, the decline in juvenile and adult tautog abundance in Long Island Sound that began in the mid-1980s (Simpson et al. 1995) coincided with the decreasing trend in eggs collected at EN. If the decrease in adults was caused by entrainment losses, then the reduction in egg abundance should have lagged the decline of juveniles by several years because females do not mature until age-3 or 4. Therefore, the lower abundance of tautog eggs was probably due a decline in the abundance of spawning adults from fishing rather than the operation of MNPS. During the 1990s, the instantaneous fishing mortality rate for tautog was estimated at about 0.54 (annual fishing mortality of 42%) and various survey biomass indices declined by more than half from the previous decade (ASMFC 1996). At present, tautog stocks are overfished and because of the long life and slow growth of this species, abundance should remain depressed until fishing mortality is reduced to less than half of current levels.

Conclusions

Evaluation of data collected in the trawl, seine, and ichthyoplankton programs in the fish ecology studies in 1995-96 did not change conclusions made in recent years regarding the potential impact of the operation of MNPS. Examination of species composition in each program indicated similar dominant taxa as found in previous years. In trawl catches, annual abundances were more stable for resident species (e.g., grubby, skates) than for those fishes caught seasonally (e.g., anchovies, scup), probably because individuals of the latter were primarily young-of-the-year and annual reproductive success was variable. For all species, the estimated

TABLE 19. Average lifetime egg production of an age-3 female tautog spawner.

Age	Length (mm)[a]	Fecundity[b]	Survival probability[c]	Fraction mature[d]	Egg production
3	223	16,069	1.000000	0.8	12,855
4	273	26,476	0.826280	1.0	21,876
5	317	39,000	0.682738	1.0	26,627
6	354	53,519	0.564133	1.0	30,192
7	386	69,938	0.282955	1.0	19,789
8	414	88,181	0.141924	1.0	12,515
9	439	108,184	0.071186	1.0	7,701
10	460	129,894	0.035705	1.0	4,638
11	478	153,264	0.017909	1.0	2,745
12	493	178,251	0.008983	1.0	1,601
13	507	204,820	0.004505	1.0	923
14	519	232,937	0.002260	1.0	526
15	529	262,571	0.001133	1.0	298
16	538	293,697	0.000569	1.0	167
17	545	326,288	0.000285	1.0	93
18	552	360,321	0.000143	1.0	52
19	558	395,774	0.000072	1.0	28
20	562	432,628	0.000036	1.0	16
21	567	470,864	0.000018	1.0	8
22	570	510,463	0.000009	1.0	5
				Total	142,655

[a] Length at age from Simpson (1989).
[b] Fecundity at age from Chenoweth (1963).
[c] Instantaneous mortality rates (Z) were:
Natural (M) = 0.15 from ASMFC (1996)
Discard through age-6 (F) = 0.04 from Simpson (CT DEP, Old Lyme, CT, pers. comm.)
Fishing ages-7 through 22 (F) = 0.54 from ASMFC (1996).
[d] Female maturity from Chenoweth (1963).

number of eggs and larvae entrained was a function of both annual abundance and the volume of condenser cooling-water used at MNPS, with the frequency of large entrainment estimates greater during most years of three-unit operation. Entrainment estimates for several taxa and life stages (e.g., anchovy and cunner eggs; anchovy, American sand lance, and grubby larvae) decreased in 1995 or 1996 relative to the previous year for these estimates.

Detailed analyses were conducted on six taxa that were most susceptible to MNPS operational impact due to entrainment or effects of the thermal discharge. These analyses generally focused on comparing temporal trends during two- and three-unit operations. The reasons for the decline in Atlantic silverside taken by seine that occurred during three-unit operation remains unknown, but probably was not related to the MNPS thermal discharge, which affects the JC seine site only minimally. Trends of declining abundance during two-unit operation for larval American sand lance and juvenile and adult tautog and cunner were probably related to regional declines, likely resulting from increased predation or overfishing in the case of tautog. Also, removal of specific habitat (i.e., the Unit 3 cofferdam) contributed to fewer cunner and tautog being collected by trawl at station IN. The large numbers of tautog and cunner eggs entrained at MNPS did not appear to affect the future spawning stocks of these two fishes because the proportion of juvenile recruits relative to adults has increased during three-unit operation. Finally, early life history stages of both tautog and cunner appear to be affected similarly by environmental and biological processes, which seem to have greater effects than MNPS operation.

References Cited

ASMFC (Atlantic States Marine Fisheries Commission). 1996. Draft fisheries management plan for tautog. Public hearing summary. Washington, DC. 10 pp.

Bengtson, D.A. 1984. Resource partitioning by *Menidia menidia* and *Menidia beryllina* (Osteichthyes: Atherinidae). Mar. Ecol. Prog. Ser. 18:21-30.

Bengtson, D.A. 1985. Laboratory experiments on mechanisms of competition and resource partitioning between *Menidia menidia* and *Menidia beryllina* (Cope) (Osteichthyes: Atherinidae). J. Exp. Mar. Biol. Ecol. 92:1-18.

Bengtson, D.A., R.C. Barkman, and W.J. Berry. 1987. Relationships between maternal size, egg diameter, time of spawning season, temperature, and length of hatch of Atlantic silverside, *Menidia menidia*. J. Fish Biol. 31:697-704.

Bigelow, H.B., and W.C. Schroeder. 1953. Fishes of the Gulf of Maine. U.S. Fish Wildl. Serv. Bull. 53:1-577.

Blake, M.M., and E.M. Smith. 1984. A marine resources management plan for the State of Connecticut. CT Dept. Envir. Prot., Mar. Fish. 244 pp.

Botelho, V.M., and G.T. Donnelly. 1978. A statistical analysis of the performance of the Bourne plankton splitter, based on test observations. NMFS unpub. ms.

Bourne, D.W., and J.J. Govoni. 1988. Distribution of fish eggs and larvae and patterns of water circulation in Narragansett Bay, 1972-1973. Am. Fish. Soc. Symp. 3:132-148.

Briggs, P.T., and J.S. O'Conner. 1971. Comparison of shore-zone fishes over natural vegetated and sand-filled bottoms in Great South Bay. N.Y. Fish Game J. 18:15-41.

Castro, L.R., and R.K. Cowen. 1991. Environmental factors affecting the early life history of bay anchovy *Anchoa mitchilli* in Great South Bay, New York. Mar. Ecol. Prog. Ser. 76:235-247.

Chenoweth, S.B. 1963. Spawning and fecundity of the tautog, *Tautoga onitis* (Linnaeus). M.S. Thesis. Univ. of Rhode Island, Narragansett, RI. 60 pp.

Conover, D.O. 1979. Density, growth, production and fecundity of the Atlantic silverside, *Menidia menidia* (Linnaeus), in a central New England estuary. M.S. Thesis. Univ. of Massachusetts, Amherst, MA. 59 pp.

Conover, D.O. 1992. Seasonality and the scheduling of life history at different latitudes. J. Fish. Biol. 41:161-178.

Conover, D.O., and M.H. Fleisher. 1986. Temperature-sensitive period of sex determination in the Atlantic silverside, *Menidia menidia*. Can. J. Fish. Aquat. Sci. 43:514-520.

Conover, D.O., and B.E. Kynard. 1981. Environmental sex determination: interaction of temperature and genotype in a fish. Science 213:577-579.

Conover, D.O., and B.E. Kynard. 1984. Field and laboratory observations of spawning periodicity and behavior of a northern population of the Atlantic silverside, *Menidia menidia* (Pisces: Atherinidae). Envir. Biol. Fish. 11:161-171.

Conover, D.O., and S.A. Murawski. 1982. Offshore winter migration of the Atlantic silverside, *Menidia menidia*. Fish. Bull., U.S. 80:145-150.

Conover, D.O., and M.R. Ross. 1982. Patterns in seasonal abundance, growth and biomass of the Atlantic silverside, *Menidia menidia*, in a New England estuary. Estuaries 5:275-286.

Cooper, R.A. 1965. Life history of the tautog, *Tautoga onitis* (Linnaeus). Ph.D. Thesis. Univ. of Rhode Island, Narragansett, RI. 153 pp.

Dew, C.B. 1976. A contribution of the life history of the cunner, *Tautogolabrus adspersus*, in Fishers Island Sound, Connecticut. Chesapeake Sci. 14:101-113.

Dorf, B.A. 1994. Ecology of juvenile tautog (*Tautoga onitis*, Family Labridae) in Narragansett Bay, Rhode Island. Ph.D. Thesis, Univ. of Rhode Island, Narragansett, RI. 213 pp.

Dorsey, S.E., E.D. Houde, and J.C. Gamble. 1996. Cohort abundances and daily variability in mortality of eggs and yolk-sac larvae of bay anchovy, *Anchoa mitchilli*, in Chesapeake Bay. Fish. Bull., U.S. 94:257-267.

Draper, N., and H. Smith. 1981. Applied regression analysis. John Wiley and Sons, New York. 709 pp.

Ennis, G.P. 1969. Occurrences of the little sculpin, *Myoxocephalus aeneus*, in Newfoundland waters. J. Fish. Res. Board Can. 26:1689-1694.

Fritzsche, R.A. 1978. Development of fishes of the Mid-Atlantic Bight. An atlas of egg, larval and juvenile stages. Vol. V. Chaetodontidae through Ophidiidae. Power Plant Project, Off. Biol. Serv., U.S. Fish Wildl. Serv., U.S. Dept. of the Interior, FWS/OBS-78/12. 340 pp.

Gendron, L. 1989. Seasonal growth of the kelp, *Laminaria longicruris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. Bot. Mar. 32:345-354.

Gilbert, R.O. 1989. Statistical methods for environmental pollution monitoring. Van Nostrand-Reinhold Co., New York. 320 pp.

Gleason, T., and C. Recksiek. 1988. Synopsis of biological data for the cunner *Tautogolabrus adspersus* (Walbaum). Univ. of Rhode Island. Contrib. 240 of the RI Exp. Sta.

Govoni, J.J., and J.E. Olney. 1991. Potential predation on fish eggs by the lobate ctenophore *Mnemiopsis leidyi* within and outside the Chesapeake Bay plume. Fish. Bull., U.S. 89:181-186.

Green, J.M. 1975. Restricted movements and homing of the cunner *Tautogolabrus adspersus*. Can. J. Zool. 53:1427-1431.

Green, J.M., and M. Farwell. 1971. Winter habits of the cunner, *Tautogolabrus adspersus* (Walbaum 1792), in Newfoundland. Can. J. Zool. 49:1497-1499.

Green, J.M., G. Martel, and E.A. Kingsland. 1985. Foraging time allocation in a territorial fish: influence of reproductive activities. Mar. Ecol. Prog. Ser. 24:23-26.

Grosslein, M.D., and T.R. Azarovitch. 1982. Fish distribution. MESA New York Bight Atlas Monogr. 15. New York Sea Grant Institute, Albany, NY. 182 pp.

Hildebrand, S.F. 1943. A review of the American anchovies (Family Engraulidae). Bull. Bingham Oceanogr. Coll. 8:1-165.

Hoese, H.D., and R.H. Moore. 1977. Fishes of the Gulf of Mexico. Texas A&M Univ. Press, Coll. Sta. 327 pp.

Hoff, J.G. 1972. Movements of adult tidewater silverside, *Menidia beryllina* (Cope), tagged in New England waters. Am. Midl. Nat. 88:499-502.

Hollander, M. and D.A. Wolfe. 1973. Nonparametric statistical methods. John Wiley and Sons, New York. 503 pp.

Hostetter, E.B., and T.A. Munroe. 1993. Age, growth, and reproduction of tautog, *Tautoga onitis* (Labridae: Perciformes) from coastal waters of Virginia. Fish. Bull., U.S. 91:45-64.

Houde, E.D., J.C. Gamble, S.E. Dorsey, and J.H. Cowan, Jr. 1994. Drifting mesocosms: the influence of gelatinous zooplankton on mortality of bay anchovy, *Anchoa mitchilli*, eggs and yolk-sac larvae. ICES J. Mar. Sci. 51:383-394.

Jessop, B.M. 1983. Aspects of the life history of the Atlantic silverside (*Menidia menidia*) of the Annapolis River, Nova Scotia. Can. Ms. Rep. Fish. Aquat. Sci. 1694. 41 pp.

Johansen, F. 1925. Natural history of the cunner (*Tautogolabrus adspersus* Walbaum). Contrib. Can. Biol. 2:423-468.

Johnson, M.S. 1975. Biochemical systematics of the atherinid genus *Menidia*. Copeia 1975:662-691.

Kuntz, A. 1914. The embryology and larval development of *Bairdiella chrysura* and *Anchovia mitchilli*. U.S. Bur. Fish., Bull. (1913) 33:1-19.

Laroche, J.L. 1982. Trophic patterns among larvae of five species of sculpins (Family: Cottidae) in a Maine estuary. Fish. Bull., U.S. 80:827-840.

Lawton, R.P., B.C. Kelly, V.J. Malkoski, J.H. Chisholm, P.Nitschke, B. Starr, and E. Casey. 1994. Semi-annual report on monitoring to assess impact of Pilgrim Nuclear Power Station on marine fisheries resources of Western Cape Cod Bay *in* Marine Ecology studies related to the operation of Pilgrim Station. Semi-annual Rep. No. 44. Boston Edison Company.

Lawton, R., B. Kelly, V. Malkoski, J. Chisholm, P.Nitschke, and J. Boardman. 1996. Annual report on assessment and mitigation of impact of the Pilgrim Nuclear Power Station on finfish populations in Western Cape Cod Bay *in* Marine Ecology studies related to the operation of Pilgrim Station. Semi-annual Rep. No. 47. Boston Edison Company. 86 pp.

Lazzari, M.A., K.W. Able, and M.P. Fahay. 1989. Life history and food habits of the grubby, *Myoxocephalus aenaeus* (Cottidae), in a Cape Cod estuary. Copeia 1989:7-12.

Leak, J.C., and E.D. Houde. 1987. Cohort growth and survival of bay anchovy, *Anchoa mitchilli*, larvae in Biscayne Bay, Florida. Mar. Ecol. Prog. Ser. 37:109-122.

Levin, P.S. 1991. Effects of microhabitat on recruitment variation in a Gulf of Maine reef fish. Mar. Ecol. Prog. Ser. 75:183-189.

Lund, W.A., and B.C. Marcy, Jr. 1975. Early development of the grubby, *Myoxocephalus aenaeus* (Mitchill). Biol. Bull. 149:373-383.

Luo, J., and J.A. Musick. 1991. Reproductive biology of bay anchovy in Chesapeake Bay. Trans. Am. Fish. Soc. 120:701-710.

MacLeod, R.E. 1995. Job 1: marine angler survey. *In* A study of marine recreational fisheries in Connecticut. Federal aid to sports fish restoration. F54R final report. March 1, 1989-February 28,

1995. CT Dept. Envir. Prot., Bur. Nat. Res., Fish. Div. 25 pp.

McConnaughey, R.A., and L.L. Conquest. 1993. Trawl survey estimation using a comparative approach based on lognormal theory. Fish. Bull., U.S. 91:107-118.

McHugh, J.L. 1967. Estuarine nekton. Pages 581-620 *in* G.H. Lauff, ed. Estuaries. Amer. Assoc. Advan. Sci. Publ. 83.

Middaugh, D.P. 1981. Reproductive ecology and spawning periodicity of the Atlantic silverside, *Menidia menidia* (Pisces: Atherinidae). Copeia 1981:766-776.

Miller, D. 1958. A key to some of the more common larval fishes of the Gulf of Maine. Woods Hole Lab. Ms. Rep. 58-1. 56 pp.

Monteleone, D.M. 1992. Seasonality and abundance of ichthyoplankton in Great South Bay, New York. Estuaries 12:230-238.

Monteleone, D.M., and W.T. Peterson. 1987. Interannual fluctuations in the density of sand lance, *Ammodytes americanus*, larvae in Long Island Sound, 1951-1983. Estuaries 15: 246-254.

Morgan, R.P. II, B.M. Baker, and J.H. Howard. 1995. Genetic structure of bay anchovy (*Anchoa mitchilli*) populations in Chesapeake Bay. Estuaries 18:482-493.

MRI (Marine Research, Inc.). 1994. Brayton Point Station biological and hydrological report. January-December 1993. Submitted to New England Power Co.

Newberger, T.A., and E.D. Houde. 1995. Population biology of bay anchovy *Anchoa mitchilli* in the mid Chesapeake Bay. Mar. Ecol. Prog. Ser. 116:25-37.

Nizinski, M.S., B.B. Collette, and B.B. Washington. 1990. Separation of two species of sand lance, *Ammodytes americanus* and *A. dubius*, in the Western North Atlantic. Fish. Bull., U.S. 88:241-255.

NNECO (Northeast Nuclear Energy Company). 1995a. Ecological studies proposed for 1996 at Millstone Nuclear Power Station. Enclosure 1 to Letter D08923 dated July 25, 1995 from D.B. Miller, NNECO, to S.J. Holbrook, Commissioner, CT DEP. 33 pp.

NNECO. 1995b. Letter D09375 dated November 29, 1995 from D.B. Miller, NNECO, to S.J. Holbrook, Commissioner, CT DEP.

NNECO. 1996. Ecological studies proposed for 1997 at Millstone Nuclear Power Station. Enclosure 1 to Letter D10084 dated July 15, 1996 from D.B. Miller, NNECO, to S.J. Holbrook, Commissioner, CT DEP. 12 pp.

NOAA. 1993. Tide current tables, 1994 Atlantic coast of North America. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service. 201 pp.

NUSCO (Northeast Utilities Service Company). 1986. The effectiveness of the Millstone Unit 1 sluiceway in returning impinged organisms to Long Island Sound. 18 pp.

NUSCO. 1987. Fish ecology. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1987. 102 pp.

NUSCO. 1988a. Delta distribution. Pages 311-320 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1987.

NUSCO. 1988b. Hydrothermal studies. Pages 323-355 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1987.

NUSCO. 1988c. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to Letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP. 21 pp.

NUSCO. 1992a. Niantic Bay current studies. Pages 317-331 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1991.

NUSCO. 1992b. Progress report on the MNPS fish return systems. Enclosure 2 to letter D05905 dated October 22, 1992 from J.F. Opeka, NUSCO, to T. Keeney, Commissioner, CT DEP.

NUSCO. 1994a. Fish ecology. Pages 113-132 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1993.

NUSCO. 1994b. Progress report on the MNPS fish return systems. Enclosure 1 to letter D08071 dated October 20, 1994 from D. Miller, NNECO, to T. Keeney, Commissioner, CT DEP.

NUSCO. 1995. Fish ecology. Pages 93-121 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1994.

NUSCO. 1996. Fish ecology. Pages 199-231 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, CT. Annual report, 1995.

Olla, B.L., A.J. Bejda, and A.D. Martin. 1974. Daily activity, movements, feeding, and seasonal occurrence in the tautog, *Tautoga onitis*. Fish. Bull., U.S. 72:27-35.

Olla, B.L., A.J. Bejda, and A.D. Martin. 1975. Activity, movements, and feeding behavior of the cunner, *Tautogolabrus adspersus*, and comparison of food habits with young tautog, *Tautoga onitis*, off Long Island, New York. Fish. Bull., U.S. 73:895-900.

Olla, B.L., A.J. Bejda, and A.D. Martin 1979. Seasonal dispersal and habitat selection of cunner, *Tautogolabrus adspersus*, and young tautog, *Tautoga onitis*, of Long Island, New York. Fish. Bull., U.S. 77:255-262.

Olla, B.L., and C. Samet. 1977. Courtship and spawning behavior of the tautog, *Tautoga onitis* (Pisces: Labridae), under laboratory conditions. Fish. Bull., U.S. 75:585-599.

Olla, B.L., and C. Samet. 1978. Effects of elevated temperature on early embryonic development of the tautog, *Tautoga onitis*. Trans. Am. Fish. Soc. 107:820-824.

Peebles, E.B., J.R. Hall, and S.G. Tolley. 1996. Egg production by the bay anchovy *Anchoa mitchilli* in relation to adult and larval prey fields. Mar. Ecol. Prog. Ser. 131:61-73.

Pennington, M. 1983. Efficient estimators of abundance for fish plankton surveys. Biometrics 39:281-286.

Pennington, M. 1986. Some statistical techniques for estimating abundance indices from trawl surveys. Fish. Bull., U.S. 84:519-525.

Pottle, R.A., and J.M. Green. 1979a. Field observations on the reproductive behaviour of the cunner, *Tautogolabrus adspersus* (Walbaum), in Newfoundland. Can. J. Zool. 57:247-256.

Pottle, R.A., and J.M. Green. 1979b. Territorial behaviour of the north temperate labrid, *Tautogolabrus adspersus*. Can. J. Zool. 57:2337-2347.

Purcell., J.E., D.A. Nemazie, S.E. Dorsey, E.D. Houde, and J.C. Gamble. 1994. Predation mortality of bay anchovy *Anchoa mitchilli* eggs and larvae due to scyphomedusae and ctenophores in Chesapeake Bay. Mar. Ecol. Prog. Ser. 114:47-58.

Reay, P.J. 1970. Synopsis of biological data on North Atlantic sand eels of the genus *Ammodytes*. (*A. tobianus*, *A. dubius*, *A. americanus* and *A. marinus*). FAO Fish. Synop. No. 82. 28 pp.

Regan, J.D., W.L. Carrier, C. Samet, and B.L. Olla. 1982. Photoreactivation in two closely related marine fishes having different longevities. Mech. Ageing Devel. 18:59-66.

Reisman, H.M., G.L. Fletcher, M.H. Kao, and M.A. Shears. 1987. Antifreeze proteins in the grubby sculpin, *Myoxocephalus aenaeus* and the tomcod, *Microgadus tomcod*: comparisons of seasonal cycles. Envir. Biol. Fish. 18:295-301.

Richards, S.W. 1959. Pelagic fish eggs and larvae of Long Island Sound. Bull. Bingham Oceanogr. Coll. 17:95-124.

Richards, S.W. 1963. The demersal fish population of Long Island Sound. Bull. Bingham Oceanogr. Coll. 18:1-101.

Richards, S.W. 1982. Aspects of the biology of *Ammodytes americanus* from the St. Lawrence River to Chesapeake Bay, 1972-75, including a comparison of the Long Island Sound postlarvae with *Ammodytes dubius*. J. Northw. Atl. Fish. Sci. 3:93-104.

Sampson, R. 1981. Connecticut marine recreational fisheries survey 1979-1980. CT Dept. Envir. Prot., Mar. Fish. 49 pp.

SAS Institute, Inc. 1985. SAS user's guide: statistics. Version 5 ed. SAS Institute Inc. Cary, NC. 956 pp.

Scott, W.B., and M.G. Scott. 1988. Atlantic fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219. 731 pp.

Sen, P.K. 1968. Estimator of the regression coefficient based on Kendall's tau. Amer. Stat. Assoc. 63:1379-1389.

Serchuk, F.M. 1972. The ecology of the cunner, *Tautogolabrus adspersus* (Walbaum) (Pisces: Labridae), in the Weweantic River Estuary, Wareham, Massachusetts. M.S. Thesis, Univ. of Massachusetts, Amherst, MA. 111 pp.

Sherman, K., C. Jones, L. Sullivan, W. Smith, P. Berrien, and L. Ejsymont. 1981. Congruent shifts in sand eel abundance in western and eastern North Atlantic ecosystems. Nature (London) 291:486-489.

Simpson, D.G. 1989. Population dynamics of the tautog, *Tautoga onitis*, in Long Island Sound. M.S. Thesis. Southern Connecticut State Univ., New Haven, CT. 65 pp.

Simpson, D.G., M.W. Johnson, and K. Gottschal. 1995. Job 2: Marine finfish survey. Pages 27-53 *in* A study of marine recreational fisheries in Connecticut. CT Dept. Envir. Prot., Mar. Fish.

Smigielski, A.S., T.A. Halavik, L.J. Buckley, S.M. Drew, and G.C. Laurence. 1984. Spawning, embryo development and growth of the American

sand lance *Ammodytes americanus* in the laboratory. Mar. Ecol. Prog. Ser. 14:287-292.

Smith, C.L. 1985. The inland fishes of New York State. NY State Dept. Envir. Conserv. Albany, NY. 522 pp.

Smith, E.M., E.C. Mariani, A.P. Petrillo, L.A. Gunn, and M.S. Alexander. 1989. Principal fisheries of Long Island Sound, 1961-1985. CT Dept. Envir. Prot., Mar. Fish. 47 pp.

Sogard, S.M., K.W. Able, and M.P. Fahay. 1992. Early life history of the tautog *Tautoga onitis* in the Mid-Atlantic Bight. Fish. Bull., U.S. 90:529-539.

Sokal, R.R. and F.J. Rohlf. 1969. Biometry. W.H. Freeman and Company, San Francisco. 775 pp.

Stevenson, R.A. 1958. The biology of the anchovies *Anchoa mitchilli* and *Anchoa hepsetus* in Delaware Bay. M.S. Thesis. Univ. Delaware, Newark, DE. 56 pp.

Tracy, H.C. 1910. Annotated list of the fishes known to inhabit the waters of Rhode Island. R.I. Ann. Rep. Comm. Inland Fish. 40:35-176.

Vouglitois, J.J., K.W. Able, R.J. Kurtz, and K.A. Tighe. 1987. Life history and population dynamics of the bay anchovy in New Jersey. Trans. Am. Fish. Soc. 116:141-153.

Westin, D.T., K J. Abernethy, I.E. Meller, and B.A. Rogers. 1979. Some aspects of biology of the American sand lance, *Ammodytes americanus*. Trans. Am. Fish. Soc. 108:328-331.

Wheatland, S.B. 1956. Oceanography of Long Island Sound. 1952-1954. II. Pelagic fish eggs and larvae. Bull. Bingham Oceanogr. Coll. 15:234-314.

Williams, G.C. 1967. Identification and seasonal size changes of eggs of the labrid fishes, *Tautogolabrus adspersus* and *Tautog onitis*, of Long Island Sound. Copeia 1967:452-453.

Williams, G.C., D.C. Williams, and R.J. Miller. 1973. Mortality rates of planktonic eggs of the cunner, *Tautogolabrus adspersus* (Walbaum), in Long Island Sound. Pages 181-195 *in* A. Pacheco, ed. Proceedings of a workshop on egg, larval and juvenile stages of fish in Atlantic coast estuaries. Nat. Mar. Fish. Serv., Mid. Atl. Coast. Fish. Ctr. Tech. Pub. No. 1.

Zastrow, C.E., E.D. Houde, and L.G. Morin. 1991. Spawning, fecundity, hatch-date frequency and young-of-the-year growth of bay anchovy *Anchoa mitchilli* in mid-Chesapeake Bay. Mar. Ecol. Prog. Ser. 73:161-171.

APPENDIX I. List of fishes collected in the Fish Ecology sampling programs (June 1979-May 1995; all stations).

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	•		•
Chaetodon 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 cornetfish	•		
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	•	•	•
Gobiidae	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 crumenophthalmus	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 seine at station JC during each report year from June 1976 through May 1996.

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	94-95	95-96	Total
Number of samples	24	24	24	24	24	24	33	42	54	36	36	60	60	60	60	63	45	45	42	42	822
Taxon[a]																					
Menidia spp.	37,264	15,962	505	443	5,834	1,587	2,006	7,667	1,056	484	3,103	4,174	3,548	2,369	5,611	3,585	4,504	2,234	2,922	4,445	109,303
Fundulus spp.	1,634	714	706	472	515	308	640	667	1,312	759	80	364	2,294	655	831	1,152	310	300	1635	609	15,957
A. quadracus	463	592	257	264	48	93	88	1,827	167	104	296	92	144	301	125	1,076	55	36	29	9	6,066
C. variegatus	42	284	35	16	7	33	133	27	25	23	1	2	10	2	14	1,169	8	0	55	11	1,897
B. tyrannus	0	0	0	0	0	0	1	1	0	0	0	3	2	25	510	171	2	47	14	881	1,657
P. saltatrix	1	0	1	1	0	1	135	0	2	0	0	1	1	3	797	2	0	10	4	3	962
P. pungitius	2	0	6	1	3	2	3	295	7	3	8	2	12	5	4	4	0	0	0	0	357
G. aculeatus	8	141	13	2	2	2	2	49	5	3	14	2	32	0	0	2	0	0	1	0	278
S. fulfus	1	1	6	1	1	2	12	9	9	5	1	8	12	5	8	8	5	2	13	12	121
M. cephalus	0	0	3	1	41	1	4	4	1	0	0	4	39	0	0	0	1	0	0	0	99
M. curema	0	0	0	0	0	0	0	1	9	0	0	0	42	1	9	1	0	1	30	0	94
Gadidae	0	0	9	0	20	12	11	6	0	2	3	0	1	1	0	6	1	0	1	0	67
L. parva	1	0	0	0	0	0	0	2	0	1	0	15	10	2	0	32	3	0	0	1	67
P. americanus	1	1	0	0	6	0	0	2	4	7	4	0	0	3	10	3	2	0	3	0	46
C. harengus	0	0	0	0	0	0	2	0	0	0	30	0	6	1	0	0	0	0	0	0	39
G. wheatlandi	0	0	0	0	0	0	4	2	6	11	2	6	1	3	0	1	0	2	0	0	38
T. falcatus	0	0	1	0	1	0	0	0	0	0	0	0	22	6	0	0	0	0	0	2	32
A. rostrata	9	4	12	2	2	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	31
T. onitis	0	0	0	0	0	0	4	0	0	0	0	0	0	0	7	1	0	0	4	4	20
Anchoa spp.	0	0	0	0	2	0	7	2	0	0	0	0	0	0	3	2	0	1	0	2	19
O. mordax	0	0	0	0	0	0	0	0	0	2	0	0	18	0	0	0	0	0	0	0	18
M. aeneus	0	6	1	0	0	0	1	0	0	2	2	0	1	0	0	0	1	2	3	1	14
C. hippos	0	0	1	0	0	1	0	0	0	0	0	0	3	1	3	0	1	1	2	0	13
A. pseudoharengus	0	0	0	0	0	0	0	0	5	0	0	0	0	3	0	0	0	0	0	2	10
L. apodus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	7
T. adspersus	0	0	2	0	0	0	3	0	0	0	0	0	0	0	1	0	0	0	0	0	6
A. americanus	0	0	0	1	0	0	1	0	0	0	0	1	0	0	1	0	0	1	0	0	5
A. aestivalis	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	0	4
S. marina	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	2	4
S. vomer	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	3
C. regalis	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
Men. setosus	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
																					0
TOTAL	39,426	17,700	1,359	1,204	6,482	2,043	3,059	10,562	2,609	1,404	3,544	4,674	6,198	3,388	7,935	7,209	4,894	2,638	4,725	5,984	137,237

[a] Fish identified to the lowest practical taxon.

APPENDIX III. Total number of samples collected and number of fish caught by trawl at station IN during each report year from June 1976 through May 1996.

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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	1566
Taxon[a]																					
P. americanus	924	1338	1165	2078	2143	1908	2332	2182	1452	1807	1499	1844	2410	1259	1531	1008	1621	1532	2160	1078	33,271
S. chrysops	693	1465	388	425	903	800	1436	1843	1264	391	1528	1315	996	747	1369	7903	3749	277	1848	282	29,622
S. aquosus	203	294	107	176	150	161	216	294	199	312	322	761	591	682	357	308	616	1000	845	558	8,152
Menidia spp.	287	718	805	728	190	146	75	115	80	144	128	53	60	26	24	261	1915	107	112	82	6,056
Raja spp.	97	90	44	91	144	112	191	358	136	312	344	385	473	448	469	543	299	484	401	556	5,977
T. adspersus	632	666	227	1022	596	342	207	76	68	27	9	9	12	35	31	17	95	8	31	21	4,126
Anchoa spp.	165	58	806	0	44	354	1	20	13	1799	95	41	11	1	3	16	2	2	3	12	3,446
Gadidae	18	69	63	62	423	315	140	194	94	135	35	373	21	20	17	24	36	23	26	50	2,138
M. aeneus	45	87	56	72	162	176	208	242	76	59	126	85	111	84	62	47	122	90	147	31	2,088
P. triacanthus	5	2	12	3	4	9	4	7	0	3	10	5	929	10	328	60	24	16	1	7	1,439
P. dentatus	75	40	16	9	24	49	37	53	80	39	107	121	61	10	63	75	115	108	94	69	1,245
Prionotus spp.	42	30	30	46	66	72	31	67	38	31	104	27	36	215	76	25	19	42	98	136	1,231
C. striata	8	2	0	3	5	39	13	24	25	43	241	8	32	46	49	35	47	5	277	34	936
M. bilinearis	101	15	2	36	109	48	38	52	26	38	44	4	23	51	47	47	73	5	31	23	813
Urophycis spp.	2	5	7	5	21	23	182	45	19	29	11	26	49	25	59	13	43	59	93	44	760
T. costis	63	70	86	68	47	27	50	41	46	47	23	17	42	18	16	14	9	24	17	28	753
E. microstomus	6	0	0	0	1	17	4	15	14	34	107	39	59	12	85	82	86	31	96	36	724
P. gunnellus	54	28	11	7	35	25	36	27	23	12	21	15	24	22	6	7	12	12	27	14	418
H. americanus	7	5	11	19	62	96	115	60	16	7	1	1	0	2	2	1	0	0	0	5	410
O. mordax	1	6	14	0	9	19	29	2	4	16	8	4	4	4	16	6	134	1	12	7	296
A. pseudoharengus	1	216	8	3	0	4	0	1	0	3	4	1	1	0	0	2	7	17	3	1	272
S. fuscus	0	14	9	12	24	18	12	12	25	11	7	15	13	7	5	9	10	11	8	7	229
Clupeidae	1	0	0	0	0	0	0	0	0	110	0	0	0	0	0	0	0	0	2	0	113
S. maculatus	2	1	0	0	1	0	0	2	0	0	1	0	4	3	8	22	16	1	7	5	73
C. lumpus	8	1	6	16	4	0	4	0	4	0	0	7	2	0	4	0	1	0	0	0	57
C. regalis	1	8	0	1	0	3	1	0	0	1	4	2	10	0	1	1	1	0	1	2	37
B. tyrannus	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	5	24	0	3	0	35
P. oblongus	1	0	1	6	3	1	1	2	5	3	7	1	2	0	0	0	1	2	1	1	32
Liparis spp.	1	3	4	2	2	2	5	1	3	0	0	0	4	0	0	0	0	0	0	0	27
M. americana	2	7	1	0	6	1	0	0	0	0	0	0	1	2	0	2	0	1	1	1	25
A. sapidissima	15	2	0	0	4	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	24
M. hispidus	2	0	2	0	0	0	1	0	3	1	0	0	1	2	2	0	0	3	1	0	18
D. volitans	1	0	0	0	0	0	0	1	0	0	1	4	1	2	3	0	1	0	0	2	16
H. erectus	0	0	0	0	0	0	0	0	0	2	1	0	1	0	2	6	0	0	0	4	16
M. octodecemspinosus	0	0	2	2	1	6	3	1	0	0	0	1	0	0	0	0	0	0	0	0	16
A. aestivalis	0	0	1	0	0	0	0	9	0	0	2	1	0	0	0	0	1	0	0	0	14
G. aculeatus	1	0	1	1	0	2	1	1	1	0	0	0	0	0	0	1	2	1	0	1	13
O. tau	1	0	0	1	4	1	1	0	2	0	1	1	0	0	0	0	0	1	0	0	13
M. saxatilis	0	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	5	1	2	11
M. aegleftnus	0	0	0	0	0	0	2	0	0	0	1	0	3	0	0	0	0	5	0	0	11
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	7	1	0	0	0	10
A. americanus	0	2	0	1	1	0	2	0	0	2	1	0	0	0	0	0	0	0	0	0	9
A. rostrata	1	0	1	0	0	0	1	0	2	1	0	1	0	0	0	1	0	0	1	0	9
F. tabacaria	1	1	0	0	1	0	0	0	0	0	0	0	0	1	3	1	0	1	0	0	9
A. schoepfi	0	0	1	0	0	0	0	0	1	1	0	0	0	2	1	0	1	0	0	1	8
P. cruentatus	0	0	0	0	0	0	0	0	3	0	0	0	2	2	1	0	0	0	0	0	8

APPENDIX III. (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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	41	78	78	78	78	78	78	78	78	78	78	81	78	78	1566
Taxon[a]																					
C. harengus	0	1	0	0	0	0	0	2	1	0	0	0	0	0	0	1	0	1	0	0	6
L. xanthurus	0	0	0	0	0	0	1	0	0	2	0	0	3	0	0	0	0	0	0	0	6
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	6
P. arenatus	0	0	0	0	0	0	0	0	2	0	0	0	0	0	3	0	0	0	0	0	5
C. hippos	0	0	0	0	0	0	1	0	0	1	0	0	0	2	0	0	0	0	0	0	4
M. canis	0	0	1	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	4
A. quadracus	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	2
C. crysos	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	2
E. cimbrius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	2
Gobiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	2
L. americanus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	2
M. aeratus	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	2
P. alta	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	2
S. foetens	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	2
Lactophrys spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
P. saltatrix	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
P. pungitius	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. scombrus	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. vomer	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
T. falcatus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
T. lathami	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
T. maculatus	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1
U. subbifurcata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
TOTAL	3468	5246	3889	4890	5190	4779	5382	5752	3727	5427	4796	5169	6002	3742	4647	10551	9080	3876	6350	3102	105,065

[a] Fish identified to the lowest practical taxon.

APPENDIX IV. Total number of samples collected and number of fish caught by trawl at station JC during each report year from June 1976 through May 1996.

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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	77	78	81	78	1565
Taxon[a]																					
P. americanus	1,422	991	1,262	1,039	885	385	1,[illegible]	[illegible]02	795	1,064	781	617	660	648	551	540	640	656	598	564	17,065
S. chrysops	441	250	162	214	65	12	713	374	26	332	426	196	77	205	608	4501	628	17	384	132	9,563
Menidia spp.	479	183	261	446	202	22	73	251	152	161	1491	232	45	50	23	238	220	91	35	96	4,751
Anchoa spp.	285	9	32	4	6	6	4	24	25	326	283	164	256	492	15	1407	4	1	1	16	3,360
Gadidae	38	116	68	75	420	341	171	131	134	186	97	88	16	17	61	58	67	331	52	69	2,536
Raja spp.	63	64	42	19	45	25	86	239	41	109	84	111	130	108	334	191	196	238	102	209	2,416
G. aculeatus	9	0	0	54	15	1	3	210	933	28	346	385	30	0	1	34	27	111	1	3	2,191
S. aquosus	135	110	87	108	65	30	95	199	108	155	107	65	74	98	82	64	119	148	77	137	2,063
T. adspersus	97	78	90	232	191	263	209	120	73	23	28	15	148	34	55	88	162	50	61	46	2,063
F. punctatus	20	55	45	37	171	189	137	55	55	116	78	107	278	40	42	78	48	104	220	174	2,049
M. aeneus	33	115	96	46	70	113	98	133	82	76	57	97	159	26	27	45	102	111	137	59	1,682
O. mordax	47	164	51	0	72	11	2	21	216	274	224	227	89	5	8	3	7	1	0	6	1,428
P. dentatus	80	35	19	16	9	9	65	57	149	53	157	85	35	10	68	75	110	33	33	40	1,138
S. fluvus	7	13	15	37	27	39	65	84	124	50	57	72	136	12	18	21	67	77	27	92	1,040
T. onitis	71	106	59	57	22	20	37	18	15	31	57	30	36	20	40	35	91	50	20	74	889
Urophycis spp.	8	22	19	17	9	21	81	45	42	37	5	14	23	18	101	15	53	37	87	36	690
B. tyrannus	0	0	0	0	0	0	0	0	0	0	0	1	0	520	3	3	5	4	0	0	536
H. americanus	2	8	3	36	49	82	145	80	28	8	1	1	0	0	1	0	0	0	0	0	444
E. microstomus	11	0	0	1	1	3	5	8	8	22	35	14	4	4	54	23	54	25	29	41	342
A. quadracus	1	3	1	0	3	1	0	5	21	13	85	23	6	0	0	0	19	87	1	2	271
M. bilineatis	65	2	1	4	41	4	9	4	4	5	3	0	0	4	3	9	41	1	3	3	206
C. lumpus	11	10	16	39	4	0	7	0	18	1	1	31	4	1	2	1	15	7	1	2	171
C. striata	3	0	0	0	0	6	1	6	1	4	45	0	1	3	5	10	9	2	64	4	164
Prionotus spp.	23	7	5	5	4	3	7	1	9	3	15	0	4	7	5	8	12	13	2	3	136
P. triacanthus	1	1	2	0	4	0	0	11	0	0	5	1	26	4	14	3	1	0	0	0	73
C. harengus	0	0	0	0	0	0	0	0	0	63	0	2	0	0	1	3	0	1	0	0	70
H. erectus	0	0	0	0	0	0	0	0	3	1	8	8	3	1	6	20	3	0	0	5	58
A. rostrata	1	3	0	1	1	8	9	3	5	4	0	1	1	1	0	5	0	0	0	0	43
F. tabacaria	1	1	0	0	2	0	1	0	8	1	2	0	0	1	11	5	5	2	1	0	41
Liparis spp.	1	1	1	2	1	3	3	1	5	0	1	1	2	4	0	0	0	0	1	0	27
S. maculatus	3	2	1	0	0	0	0	1	3	1	0	0	0	2	3	3	0	0	0	5	24
C. regalis	1	11	1	0	0	5	0	0	0	0	2	0	1	1	0	0	0	0	1	0	23
A. americanus	3	2	6	0	6	1	0	0	1	0	0	0	0	0	0	0	1	0	0	1	21
M. hispidus	0	1	2	0	0	0	6	0	2	4	1	0	0	1	2	1	0	1	0	0	21
S. scapinnis	0	0	0	0	0	0	0	0	0	0	0	0	16	0	0	0	0	0	0	0	16
A. pseudoharengus	1	1	0	0	0	0	0	7	0	0	0	0	0	0	1	2	0	0	0	0	10
A. schoepfi	0	1	0	1	1	0	0	1	0	1	1	0	2	1	1	0	0	0	0	0	10
M. americana	1	4	0	0	0	0	1	0	0	0	0	0	2	0	0	0	0	2	0	0	10
M. canis	2	1	1	0	0	0	0	0	0	0	0	0	2	0	4	0	0	0	0	0	10
P. pungitius	0	0	0	0	0	0	0	0	5	0	5	0	0	0	0	0	0	0	0	0	10
O. tau	1	0	0	0	0	0	0	2	0	1	2	0	2	1	0	0	0	0	0	0	9
A. tapidissimo	1	0	0	0	0	1	0	6	0	0	0	0	0	0	0	0	0	0	0	0	8
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	4	0	0	1	0	8
G. wheatlandi	0	0	0	0	0	1	1	1	0	1	2	0	0	0	0	0	1	0	0	0	7
Lactophrys spp.	0	0	0	0	0	0	0	0	2	0	0	0	0	0	1	3	0	1	0	0	7

APPENDIX IV. (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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	77	78	81	78	1565
Taxon[a]																					
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	3	0	1	1	0	0	0	0	5
S. borealis	0	0	0	0	0	0	0	1	1	2	0	1	0	0	0	0	0	0	0	0	5
T. maculatus	3	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	5
Gobiidae	0	0	0	0	0	0	0	1	0	1	0	0	0	0	2	0	0	0	0	0	4
L. xanthurus	1	1	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	4
T. lathami	0	0	0	1	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	4
M. octodecemspinosus	0	0	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
P. saltatrix	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	3
P. arenatus	0	0	0	0	0	1	0	0	0	0	1	0	0	1	0	0	0	0	0	0	3
U. subbifurcata	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
C. ocellatus	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	2
D. volitans	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	2
E. cimbrius	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	2
Gasterosteidae	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
M. saxatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	2
P. crientatus	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	2
A. aestivalis	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
A. mediocris	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
A. maculatus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. oceanicus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
D. centrarcus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
D. macarellus	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
Fundulus spp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
L. americanus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
M. aeglefinus	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
Monacanthus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1
M. auratus	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. vomer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. marina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
T. myops	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	1
U. parvus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
TOTAL	3,376	2,375	2,351	2,493	2,394	1,609	3,679	3,400	3,098	3,160	4,496	2,592	2,277	2,341	2,158	7,298	2,707	2,202	1,941	1,819	57,766

[a] Fish identified to the lowest practical taxon.

APPENDIX V. Total number of samples collected and number of fish caught by trawl at station NR during each report year from June 1976 through May 1996

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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	1,566
Taxon[a]																					
P. americanus	2,775	1,191	1,145	2,714	3,863	3,823	7,569	3,543	4,890	2,472	4,108	3,463	6,763	3,621	4,504	4,916	5,214	1,400	2,185	869	71,028
Menidia spp	1,203	315	58	121	73	159	315	49	15	93	1,549	318	639	220	317	431	481	75	97	34	6,562
M. aeneus	57	30	59	161	201	349	587	232	274	174	490	205	631	374	492	273	451	142	299	167	5,648
S. aquosus	76	40	32	134	158	165	230	225	244	136	302	236	173	342	363	234	225	292	94	97	3,798
P. dentatus	41	29	16	14	21	100	82	63	163	75	226	214	191	20	149	173	248	105	165	68	2,163
S. fuscus	20	11	13	31	61	178	132	80	70	117	114	177	128	55	112	90	69	182	87	88	1,815
A. quadracus	8	3	22	27	190	764	76	6	91	115	22	29	25	11	18	100	50	3	4	4	1,568
G. aculeatus	16	12	44	22	186	99	58	7	162	86	8	19	63	8	14	410	135	20	60	3	1,432
Prionotus spp	97	21	2	6	4	63	200	2	2	3	44	1	19	147	98	267	37	12	262	53	1,340
Anchoa spp	10	195	0	2	0	0	11	2	5	16	8	11	168	734	2	12	1	0	12	2	1,191
B. tyrannus	0	13	1	0	1	1	0	1	0	31	10	3	0	788	2	182	23	17	41	0	1,114
T. anitis	39	16	30	45	25	129	90	16	11	22	110	15	57	28	105	51	24	13	14	128	968
Gadidae	5	9	11	8	152	185	71	43	14	209	12	5	32	29	29	12	8	16	20	44	914
S. chrysops	58	10	11	5	6	38	27	57	2	3	22	15	3	5	10	175	119	0	83	38	687
P. gunnellus	0	1	10	4	14	29	69	32	21	10	18	41	47	58	83	30	48	44	26	26	611
O. tau	96	21	7	17	27	34	24	21	22	31	53	50	56	29	55	17	5	9	18	11	603
T. adspersus	14	4	7	11	7	91	58	60	16	15	38	14	9	15	9	8	10	9	13	8	416
O. mordax	55	86	2	0	6	13	30	1	3	37	4	5	34	4	7	3	10	2	1	2	305
Clupeidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	258	259
A. rostrata	16	11	7	4	8	24	14	21	15	27	26	20	14	4	15	2	2	4	10	7	251
C. striata	18	0	0	1	2	3	2	0	1	13	107	0	3	2	20	39	1	0	9	6	227
Alosa spp	0	0	0	0	0	0	0	0	0	0	0	1	0	3	14	11	2	16	111	0	158
S. maculatus	11	5	0	0	7	11	12	7	2	1	1	0	2	4	13	22	3	2	17	15	135
Urophycis spp	0	0	0	0	1	2	7	4	3	10	1	22	11	6	9	4	2	5	18	16	121
Gobiidae	3	0	0	0	4	0	0	2	9	6	2	5	10	2	19	15	1	25	1	3	107
A. americanus	1	0	75	1	0	2	14	1	0	0	0	0	1	0	1	0	0	0	0	0	96
H. americanus	0	0	0	3	7	6	20	35	9	2	0	0	0	0	0	0	0	0	1	1	84
A. pseudoharengus	0	47	0	2	2	2	3	1	2	1	3	0	1	0	3	0	0	0	1	0	68
H. erectus	0	0	0	0	0	0	0	1	1	1	10	3	2	2	8	34	1	0	0	0	63
Raja spp	0	0	0	0	2	1	1	3	0	2	0	0	4	2	4	0	20	2	6	2	49
E. microstomus	0	0	0	0	1	0	0	3	0	1	4	3	4	1	3	7	8	1	5	1	42
A. aestivalis	0	7	1	9	0	0	0	1	2	0	0	0	0	0	0	9	0	0	1	7	37
M. americana	3	0	1	5	1	1	0	0	0	0	0	0	2	9	0	0	1	2	0	0	25
A. tapirina	2	0	0	4	1	1	0	9	0	0	0	0	0	0	0	0	0	0	0	0	17
D. volitans	2	0	0	0	0	1	3	0	0	1	1	0	0	0	0	3	2	2	1	1	17
Morone saxatilis	0	0	2	1	0	1	1	0	0	1	0	0	0	4	1	0	0	2	2	0	15
F. tabacaria	0	1	0	0	0	0	0	0	0	0	0	0	1	1	7	3	0	0	1	0	14
M. bilineatis	0	0	0	0	0	0	3	0	0	0	0	1	1	1	0	0	4	0	0	4	14
Liparis spp	0	0	0	0	0	1	0	0	0	0	0	0	0	9	0	1	0	1	1	0	13
P. triacanthus	1	0	0	0	0	0	0	2	0	0	0	0	0	8	0	0	2	0	0	0	13
Fundulus spp	0	0	0	0	0	5	2	0	0	1	1	0	1	0	1	0	0	1	0	0	12
P. oblongus	0	0	0	2	0	0	0	0	0	0	0	0	2	7	0	1	0	0	0	0	12
Gasterosteidae	0	0	0	11	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	11

APPENDIX V. (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	94-95	95-96	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	1,566
Taxon[a]																					
C. lumpus	0	0	1	1	0	0	1	0	1	0	0	2	0	0	0	2	1	1	0	0	10
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	3	0	0	3	0	0	0	0	0	6
P. saltatrix	0	1	0	0	0	0	2	0	0	0	0	1	0	0	1	0	0	0	1	0	6
S. borealis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	6
Ment. saxatilis	0	1	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	2	5
C. harengus	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	0	0	0	4
C. oceanicus	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0	1	0	0	0	0	4
S. fontens	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	4
M. hispidus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	3
P. pungitius	0	0	0	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	3
C. ocellatus	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	2
G. wheatlandi	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	2
Lactophrys spp.	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	2
P. ferrugineus	0	0	0	0	0	0	0	0	.	0	0	0	0	0	0	2	0	0	0	0	2
M. cephalus	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
T. maculatus	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	2
B. ocellatus	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1
C. regalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
C. variegatus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Monocanthus spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1
M. canis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Myoxocephalus spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1
P. alta	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
P. arenatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
S. trutta	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. vetifer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. setapinis	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1
TOTAL	4,630	2,081	1,559	3,367	5,033	5,282	9,716	4,531	6,055	3,716	7,297	4,884	9,106	6,555	6,494	7,541	7,210	2,405	3,669	1,966	104,097

[a] Fish identified to lowest practical taxon.

APPENDIX VI. Total number of days the MNPS Unit 1 sluiceway was not in service by month from 1985 through 1995.

Month	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995
January	0	0	0	0	2	0	[b]	0[a]	4	4[a]	3
February	0	0	0	0	6	0	[b]	0[a]	2	16[a]	5
March	0	0	1	0	31	2	[b]	10	7	0[a]	0
April	0	0	21	0	15[a]	3[a]	a,b	0	0	12	5
May	3	8	2	0	31[a]	18	a,b	0	0	0	10
June	4	1	6[a]	8	6	1[a]	a,c	0	0	6	1
July	2	6	31[a]	10	3	0	0[a]	0[a]	2	3	0
August	8	21	10[a]	3	6	4	15[a]	6[a]	4	4	9
September	28	3	4	18	11	3[a]	22	9	5	2	17
October	7	5	0	15	0	18[a]	4[a]	4	2	2	4
November	30[a]	0	3	3	2	6	4[a]	6	4	20	0[a]
December	21[a]	0	0	0	0	8	0[a]	7	3	5	0[a]

[a] Unit 1 shutdown.
[b] No information available (NUSCO 1992b).
[c] No information for June 1-15, 1991; sluiceway in service from June 16-30, 1991.

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Winter Flounder Studies

Introduction

The winter flounder (*Pleuronectes americanus*) has been the object 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). Its seasonal movement patterns and reproductive activity are 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 reduced. 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 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 a few remain in estuaries, apparently avoiding 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 for this species 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, 1988a, 1994b). Unlike many marine fishes, mortality of entrained winter flounder larvae potentially has greater significance as it is a product of local spawning with geographically isolated stocks associated with specific estuaries or 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. Although the 1996 spawning season was the eleventh year in which winter flounder could have been impacted by the operation of all three MNPS units, the plant was shut down during most of this period, resulting in the smallest cooling-water flow since 1985, before Unit 3 went on-line.

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 and obtain annual estimates of the spawning stock. This report summarizes data collected during 1996 and updates results reported previously in NUSCO (1996). 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

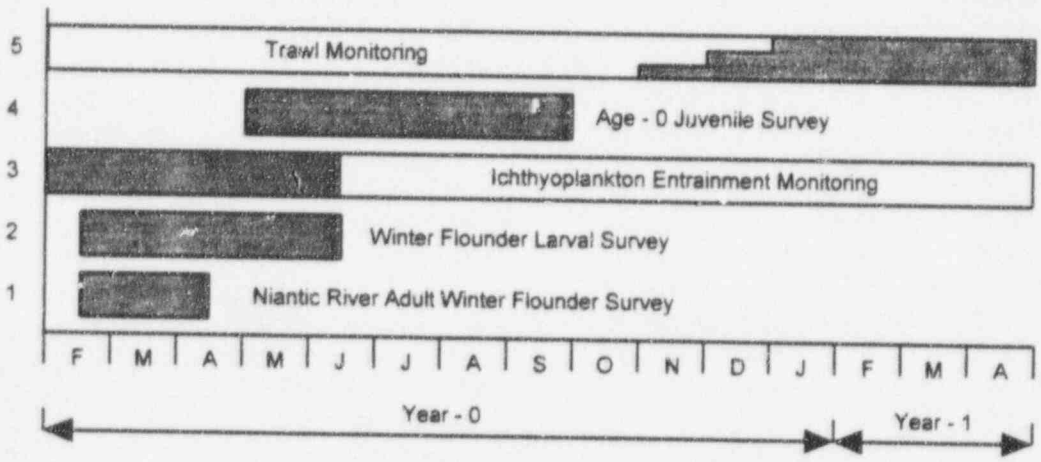
SPDM can simulate the long-term effects of historical and projected rates of fishing mortality and simultaneous plant operation, resulting in annual mortalities from impingement of juveniles and adults and the entrainment of larvae through the MNPS cooling-water system. Results of SPDM simulations and a probabilistic risk analysis help to assess the effects of MNPS operation on the Niantic River winter flounder population through the year 2060, well after the scheduled shutdown of 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 programs provide information 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 (°C) 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. Water temperature and salinity measurements at adult (surface and bottom) and larval (surface, midwater, and bottom) sampling stations were recorded using a Rosemont RS5-3 Portable Salinometer or a YSI Model 30 Salinity/Temperature/Conductivity meter. Temperature at juvenile winter flounder stations in the Niantic River was taken with a mercury thermometer.



1. February-April sampling (spawning season) for adults and juveniles throughout the Niantic River.
2. February-June larval sampling at three sations in the Niantic River and one in Niatic Bay.
3. Year-round monitoring of all ichthyoplankton at the MNPS discharges.
4. Late May-September sampling of age-0 juveniles at two stations in the Niantic River.
5. Year-round monitoring of all benthic fishes at six (1976-95) or three (1996) stations near MNPS (juvenile catch data come from two stations in November, three to four in December, and three to six in January).

Figure 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).

Adult Winter Flounder Sampling

Sampling methodology for the adult winter flounder spawning surveys in the Niantic River has remained essentially unchanged since 1982. Surveys usually begin between mid-February and mid-March, after most ice cover disappears from the river, and continues into April. Sampling ceases when the proportion of reproductively active females decreases 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. 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. Gender and reproductive condition of larger winter flounder was determined by either observing eggs or milt, or as suggested by Smigielski (1975), 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 have been sampled at the MNPS discharges (station EN, Fig. 3) since 1976. In most years, collections usually alternated between the

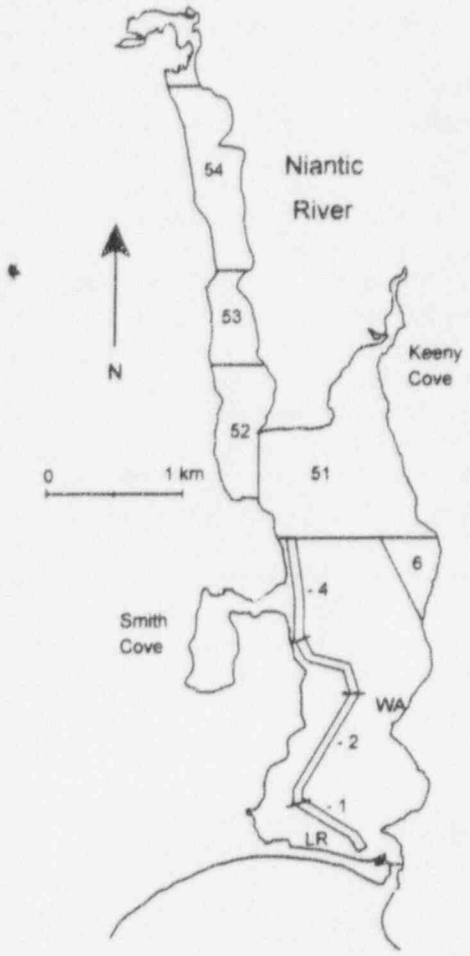


Fig. 2. Location of stations sampled in the Niantic River during 1996 for adult winter flounder from February 27 through April 3 (numbers) and age-0 winter flounder from May 22 through September 25 (letters).

discharges of Units 1 and 2, depending upon plant operation and the resulting water flow from the condenser cooling-water pumps. This year, because of the shutdown of Unit 1 (no pumps in operation) in November 1995, most collections early in the larval winter flounder season were made at Unit 2 and later alternated between Units 2 and 3. Most of these samples were collected with only one or two circulating water pumps in operation. 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 the average of four volume estimates from the flowmeters. The net was usually deployed for 3 to 4 min (filtering about 200 m^3), with variation in sampling time dependent upon the number of circulating water pumps in operation and tidal stage. Sampling frequencies and volume filtered have varied since 1976 (NUSCO 1987, 1994a). In 1996, sampling was conducted during both day and night once per week in February and 3 days and nights per week during March through early May and the last 3 weeks of June. During the last 4 weeks of May and the first week of June, only one day sample was collected each week, as unlike previous outages, the circulating water pumps were shut off for most of the time. 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 for the remainder of the season. Volume of water filtered was determined from 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 min (filtering about 120 m^3). One of the duplicate samples from the bongo sampler was retained for laboratory processing. When present, jellyfish medusae at the river stations were removed from the samples using a 1-cm mesh sieve and their mass estimated volumetrically to the nearest 100 mL.

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 (NUSCO 1987). Larval sampling at the Niantic River stations usually begins in early to mid-

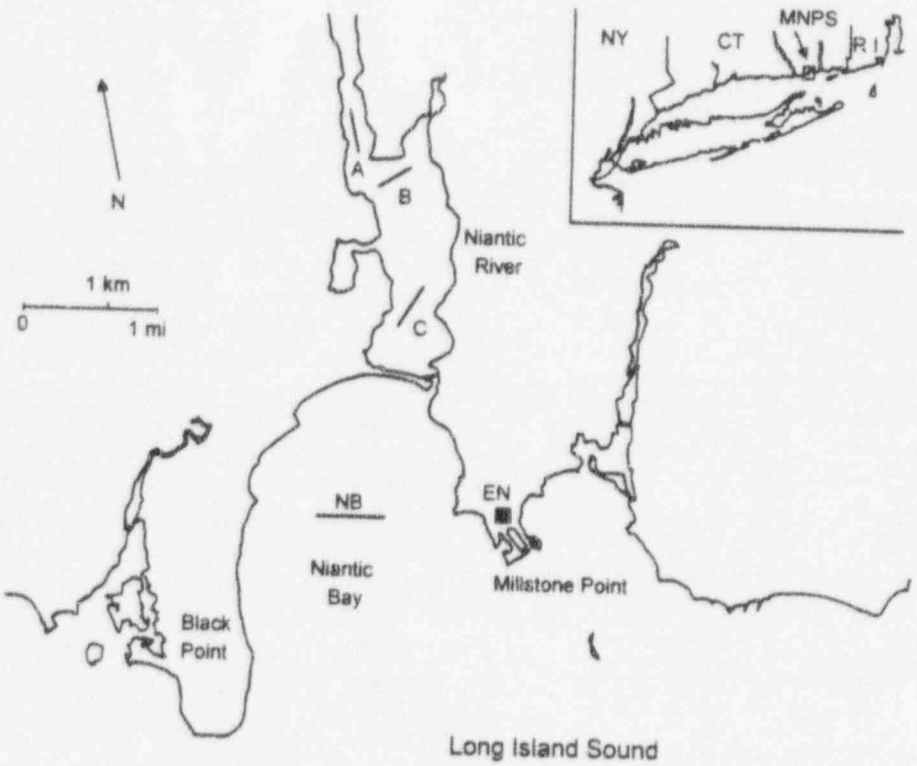


Fig. 3. Location of stations (denoted by letters) sampled for larval winter flounder during 1996.

February. In 1996, collections were only taken at stations NB and C during most of February because ice conditions prevented sampling in the upper river until February 23. Daytime tows were made within 1 hour of low slack tide through the end of March. 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 during 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 through March and at night from April throu h the remainder of the larval season.

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 data source is the Niantic River adult spawning abundance surveys, during which winte. 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 are recorded as 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 beginning in 1993, 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 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 frames with 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, only three tows were taken at a station because of bad weather or damage to the net. 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 densities of young were high, 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. For some indices, a long-term mean was calculated with variability described by the coefficient of variation (CV; standard deviation divided by the mean).

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 determined by trawl catch-per-unit-effort (CPUE). An annual relative abundance index with 95% confidence interval (CI) was calculated by using the Δ-mean CPUE (NUSCO 1988c) following data standardization. This represents a departure from the median CPUE abundance index previously used and will be discussed in more detail below. Components of standardization for CPUE calculation included tow length, tow duration, weekly effort, and fish length and gender categories. Tow distance had been measured using radar, but more recently with LORAN or differential Global Positioning System. Distance was fixed in 1983 (with exceptions noted below) 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. In most years, but especially during 1987 and 1989-91, tows one-half or two-thirds of this length were occasionally 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 program. In 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.

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; 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 fish 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 as necessary the median CPUE value for a given week such that the number of tows used in calculating CPUE was the same for each week sampled.

A second relative index of abundance was based on the size distribution of female fish from adult spawning survey catches standardized by variable weekly and yearly effort (i.e., number of tows). Catches were adjusted by sampling effort to insure that each size 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 (see NUSCO 1989 for more details). To avoid confusion with the CPUE index, this adjusted catch is referred to as "annual standardized catch" throughout the remainder of this report. The annual standardized catch of females 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 among the most appropriate ones 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 caused by brands used during early surveys. Estimates of annual population size (N) and other model parameters, including survival (ϕ), recruitment (B), and sampling intensity (p), were made 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 1996 were determined by assuming that the relative values represented 4.0% 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. Yolk-sac present or eyes not pigmented (yolk-sac larvae);

Stage 2. Eyes pigmented, no yolk-sac present, no fin ray development, and no flexion of the notochord;

Stage 3. Fin rays present and flexion of the notochord began, but left eye not yet migrated to the midline;

Stage 4. Left eye reached the midline, but juvenile characteristics not present;

Stage 5. Transformation to juvenile stage complete and intense pigmentation present near the base of the caudal fin.

Larval data analyses were based on standardized densities (number·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 and years. 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. During May and June of 1996, when night collections could not be taken at station EN due to insufficient water flow, data from night collections at station NB were used for abundance and entrainment estimates.

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% CIs 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 from 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:

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

where d_t = density at time t and all the 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 then 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):

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

where L = annual larval abundance of larvae 7-mm and larger at EN as estimated by α (see Equation 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 the correlation coefficient r 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 its 95% CI.

Abundance, Growth, and Mortality of Juveniles in Summer

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 computing mean CPUE for each day and station; density was expressed as the number per 100 m^2 of bottom. A median CPUE abundance index was determined for each half-season, with late May through July denoting the early season and August-

September the late season. A 95% CI was calculated for each median CPUE using a distribution-free method based on order statistics (Snedecor and Cochran 1967).

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 smaller 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 with a common birthdate. A catch curve was constructed such that the natural logarithm of density was plotted against age (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 $Z \cdot (30.4 / 7)$.

The relationship between growth and abundance of young and water temperature was examined using multiple linear regression (SAS Institute Inc. 1985) and functional regression methods that were described above for larval winter flounder.

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 selection 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 1988c). Beginning in January 1996, stations BR, TT, and NB were deleted (see Fish Ecology section for details), resulting in a sample size of 28 tows for the 1995-96 Δ-mean.

The annual Δ-mean 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. Two Δ-mean indices were calculated, one for stations in the lower Niantic River navigational channel (1 and 2) and, when sufficient data were available, one for all river stations combined. For comparative purposes, an annual Δ-mean abundance index of juvenile fish of similar size was also determined using catch data from the five (or, in 1996, the two) trawl monitoring program stations outside of the Niantic River during the period of January through April, with an annual sample size of 45 collections (18 in 1996), which temporally overlapped the adult spawning surveys.

Stock and Recruitment Relationship and Biological Reference Points

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 lifetime 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 are 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 (see Adult Spawning Stock Size and Egg Production section above). The recruitment index was determined by applying an age-length key to the annual standardized catches of females partitioned into length categories. Based on a re-examination of data, the age-length key used this year differed from the one used previously (described in NUSCO 1989) and will be discussed in more detail in Results and Discussion. 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 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 1992 to its terminal age (e.g., 2007 for the 1992 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). These were the same mortality rates used in the stochastic population dynamics model, discussed below. An instantaneous natural mortality rate (M) for winter flounder was assumed constant at 0.25 over all years. 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 1992 year-class (i.e., age-4 females taken in 1996). 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 current mean fecundity).

Stock and recruitment parameters. 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 1988b, 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 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:

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

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 (1977-92) 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.

Biological reference points. 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:

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

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:

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

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

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

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

$$P^{*}_{rep} = (\log_e[\alpha]) / \beta \quad (11)$$

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

Although Equations 9 through 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-structure 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 provide 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 measured volume of cooling water used by the three MNPS units. The Gompertz density function (Eq. 3) was fitted to larval data and daily densities (number·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 larval entrainment as a result of the 1996 shutdowns at all MNPS generating units was estimated by simulating full cooling-water flows at each unit with weekly winter flounder larval entrainment densities. The difference between estimates based on the actual and simulated flows was the avoided larval entrainment attributed to the shutdowns.

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 is related to the number of larvae in Niantic Bay originating 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 from 1984 through 1996. 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, advection into the Niantic River during a flood tide, entrainment at MNPS, and flushing from the bay into LIS. Estimates could be made for the number of larvae lost through natural mortality, advected into 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 an unknown termed *Source or Sink* in the mass-balance calculations. Thus, the form of the mass-balance equation was:

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

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 by 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 out 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:

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

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

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

Thus:

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

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 x 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 by V. Crecco (CT DEP, Old Lyme, CT, pers. comm.), and for 1991 through 1996 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 imprecise, sensitivity analysis on the mass-balance calculations (NUSCO 1991b) indicated that larval mortality was the least sensitive parameter in Equation 15 above. 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 (*Mort*) was summed for each 5-day period.

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) was compared to the estimated daily densities at station C. The average density of larvae flushed from the Niantic River was estimated from the functional regression equation:

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

The 95% CI for the slope ($r = 0.969$; $p = 0.001$) 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 was conducted again 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 functional regression equation:

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

The 95% CI for the slope ($r = 0.705$; $p = 0.001$) 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 5-day net loss or gain of larvae to Niantic Bay from LIS 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. Results from mass-balance calculations by developmental stage were used to estimate the number of larvae entrained at MNPS each year from the Niantic River. If *FromNR* can support the number of larvae entrained by MNPS, then the *Source or Sink* term is negative (i.e., no import) to balance the equation. These larval losses were then used to calculate conditional mortality rates for Niantic River larvae for under both actual operating conditions and projected full MNPS three-unit operation. Their derivation will be provided in greater detail in the following section and later in the Results and Discussion section.

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 (Eq. 7) fitted to the data, even though the SRR equation 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 growth and reproduction. These processes occur 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; Saila and Lorda 1977; 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 components of the computer program used for the SPDM. Components depicted by solid-line boxes constitute the model presently in use, while the box with dashed lines corresponds to the mass-balance calculations dealing with spatial larval distribution and entrainment loss estimates, which are not an integral part of the model. 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.

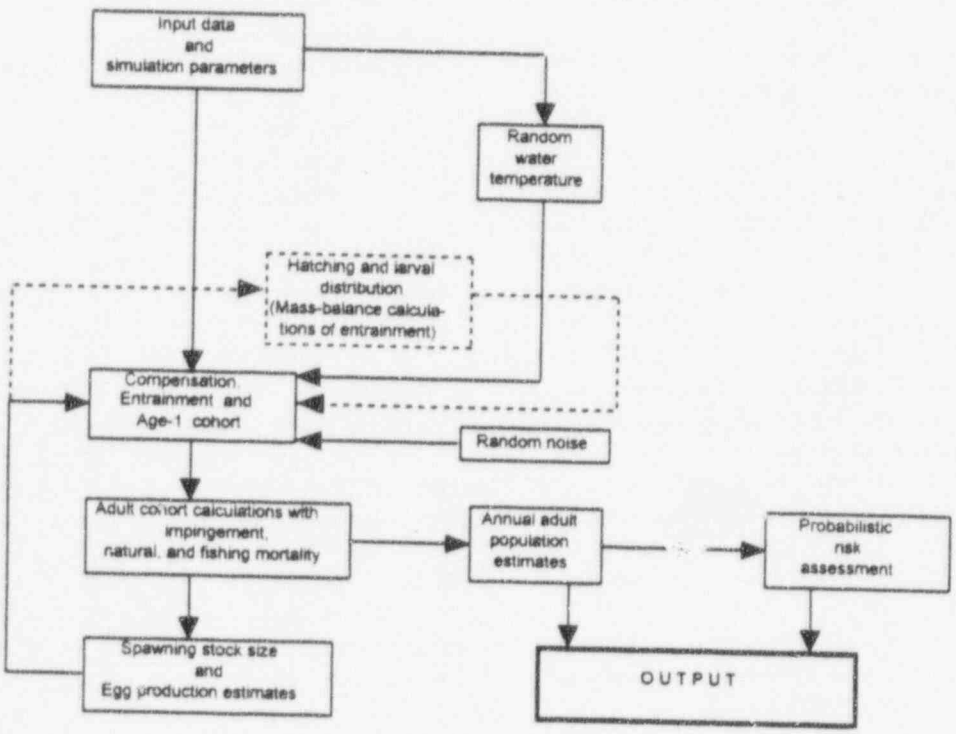


Fig. 4. Diagram of NUSCO stochastic population dynamics computer model for assessing the long-term effect of larval winter flounder entrainment at MNPS. Dashed boxes and arrows refer to components and calculations which are not an integral part of the model.

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 used in the present model was:

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

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 (see Eq. 7), but with α estimated independently of the stock and recruitment data; 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 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 their spawning potential throughout their lifetimes. 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 (n_t) and annual environmental variability in the form of random deviations from the long-term mean February water temperature ($\phi \cdot WT_t$). 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 random noise term n_t in Equation 18 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% CI 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 1977-92.

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 of the SPDM 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); larval entrainment conditional mortality rates (i.e., ENT, the fraction of the annual production of Niantic River winter flounder removed as a result of power plant operation) estimated from the mass-balance calculations described above; 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 (P. Howell, CT DEP, Old Lyme, CT, pers. comm.) and differed from those given in NUSCO (1995a). These exploitation rates took into account length-limit regulations in effect from 1982-96 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 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) 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%. Values of F used in the simulations were stepped up from 0.40 in the 1960s to a peak of 1.33 in 1990 (Fig. 5), which reflected an historical increase in fishing and the current high exploitation of winter flounder. F was subsequently reduced to meet a targeted values of 0.90 in the late 1990s.

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. Connecticut winter flounder regulations in effect for the commercial and sport fisheries since 1982.

Period	Minimum length limit (in) Commercial fishery	Sport fishery	Minimum length limit (mm) 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)
1988-1989	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
1990-93[b]	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
1994[c]	11	11	305	279	Mar 1 - Apr 14 (in all state waters)
1995 (Jan-Sep)	12	11	305	279	Mar 1 - Apr 14 (in all state waters)[d]
1995 (Oct-Dec)	12	12	305	305	None
≥ 1996	12	12	305	305	None

[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] On January 1, 1989, the minimum trawl codend mesh size for the commercial fishery was established at 4.5 inches for November 15-May 14 and 3 inches for May 15-November 14.

[c] On April 22, 1994, the minimum trawl codend mesh was established at 4.5 inches for November 15-June 30 and 4 inches for July 1-November 14. On November 15, 1994, the minimum trawl codend mesh size was increased to 5.5 inches for November 15-June 30. An 8 fish creel limit was established for the sport fishery.

[d] Closed season rescinded as of September 25, 1995, but creel limit of 8 fish remained in effect.

TABLE 3. Vulnerability factors[a] for eastern LIS winter flounder by age-class[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, as used in the Niantic River winter flounder population dynamics simulation model.

Period	Commercial fishery Age-classes: 1	2	3	4	5+	Sport fishery Age-classes: 1	2	3	4	5+	Total fishery Age-classes: 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
1988-89	0.00	0.21	0.57	0.60	0.50	0.06	0.06	0.40	0.40	0.40	0.06	0.27	0.97	1.00	1.00
1990-93	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
1994	0.00	0.12	0.57	0.60	0.60	0.06	0.06	0.30	0.36	0.40	0.06	0.18	0.87	0.96	1.00
1995	0.00	0.01	0.25	0.49	0.60	0.06	0.06	0.30	0.36	0.40	0.06	0.07	0.55	0.85	1.00
≥ 1996	0.00	0.01	0.25	0.49	0.60	0.06	0.06	0.07	0.30	0.40	0.06	0.07	0.32	0.79	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.

0.70 during 2000-2005, and 0.60 thereafter. 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

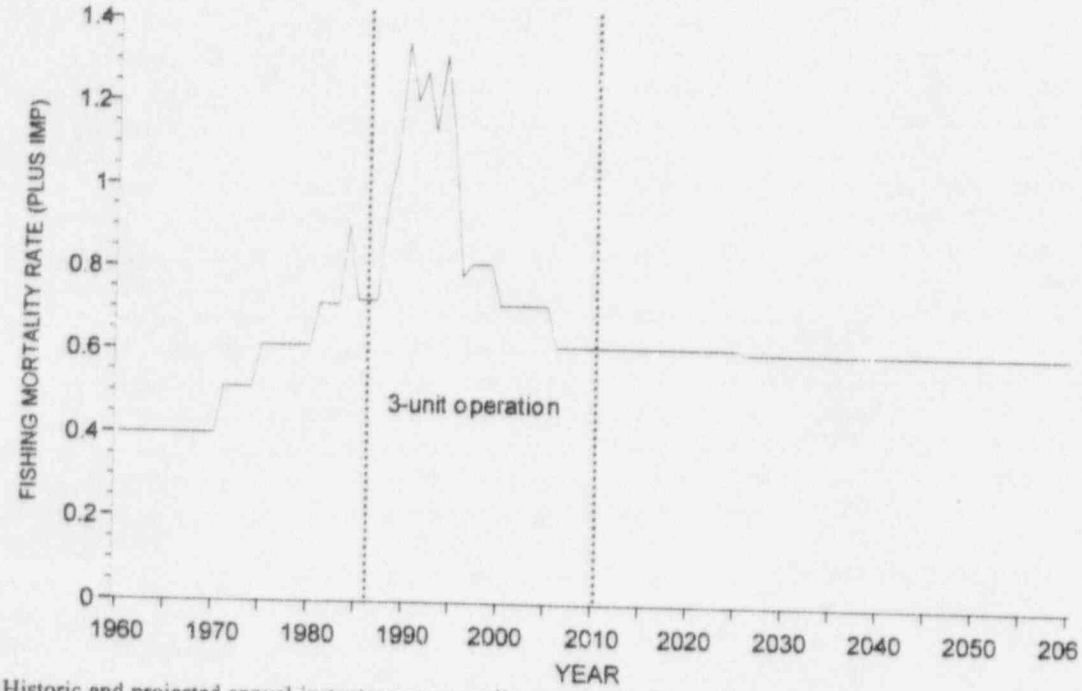


Fig. 5. Historic and projected annual instantaneous mortality rate due to fishing (F), as determined in consultation with the CT DEP, plus a small (0.01) component accounting for impingement mortality (IMP) at MNPS as implemented in the SPDM simulations.

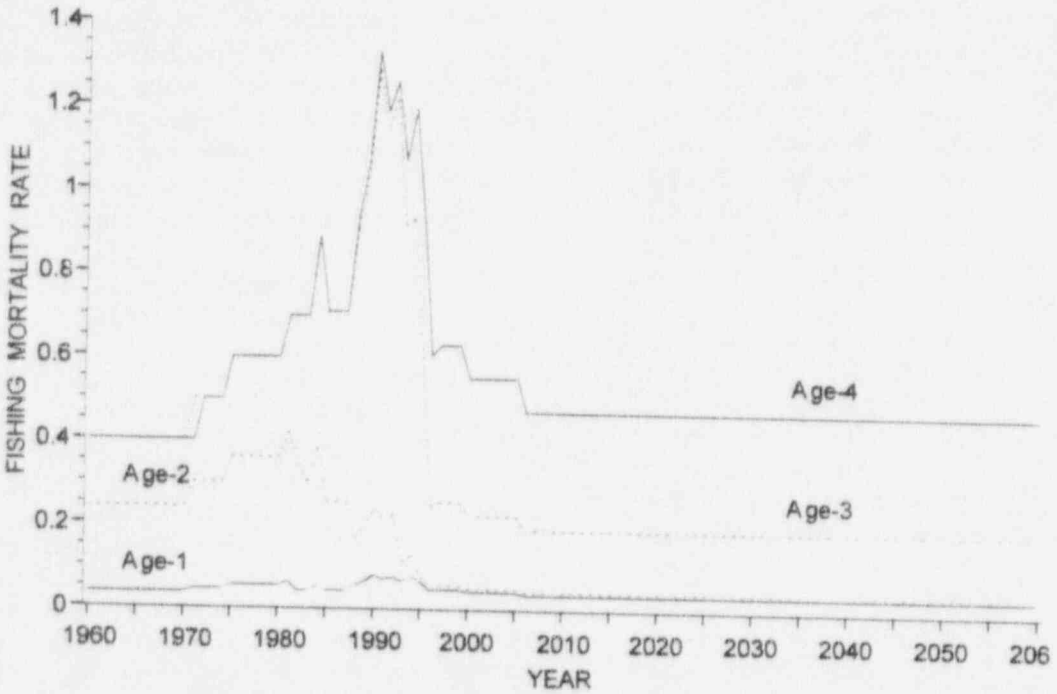


Fig. 6. Estimated reductions in instantaneous fishing mortality rate F (including discard mortality) of age-1 through 4 winter flounder as a result of actual or planned regulations imposed by the CT DEP on the winter flounder commercial and sport fisheries.

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 1996 were estimated directly using the mass-balance calculations under actual MNPS operating conditions for use in the SPDM simulations. Values of ENT determined for other years (1971-83; 1997-2025) were varied stochastically by randomly choosing one of the annual values that was adjusted for full MNPS three-unit operation; this selection process was done by re-sampling with replacement using uniform probabilities. Estimates of ENT were made by assuming that all three units used cooling water pumped at maximum capacity (11.1×10^6 $m^3 \cdot day^{-1}$). 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 1996 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 during 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 1971-95 were re-used to predict entrainment for 1997-2025 by re-sampling the historic flows with replacement using uniform probabilities to randomize the process. This approach assumed that the existing 26-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. The 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 of the 1960-2060 stock projection series consisted of 100 replicates, which were judged to be sufficient given the 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 for each year in the projection. All stock projections are given in units of spawning biomass (lbs) because overfishing criteria often rely on assessments of biomass, which tend to be more conservative than those based on fish numbers. Furthermore, 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 (i.e., the impacted stock). 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.

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	0.10	0.554	223,735
Age-4 females	0.38	0.811	378,584
Age-5 females	0.98	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	4		
Instantaneous mortality rates M and F at age-1	0.50	0[a]	
Instantaneous mortality rates M and F at age-2 and older	0.25	0	
Initial number of female spawners	72,239[b]		
Biomass of female spawners	113,415 lbs		
Mean fecundity of the stock (eggs per female spawner)	972,205[c]		
α from the three-parameter SRR for the virgin (F = 0) stock (numbers of fish)	5.87[d]		
β from the three-parameter SRR	2.450×10^{-5}		
ϕ from the three-parameter SRR	-0.418		
Mean February (1977-92) water temperature (°C)	2.81		
standard deviation	1.22		
minimum temperature	0.36		
maximum temperature	4.76		
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[e]		

[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] Calculated for the Niantic River winter flounder female spawning stock at equilibrium in the absence of fishing (see Table 33 in Results and Discussion).

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

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

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 1996.

	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
1995	29.04	99.52	21.61	57.44	37.35	60.33
1996	0.72	2.47	14.44	38.38	31.05	50.15

[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.

Results and Discussion

Seawater Temperature

In contrast to late 1994 and most of 1995, monthly mean seawater temperatures recorded at the MNPS intakes during 1996 were among the coolest since sampling began in 1976 (Table 6). The winter of 1995-96 began with the coldest water temperature (6.89°C) in December since 1989. January (3.55°C), February (2.12°C), and March (2.87°C) were also below average. April (5.38°C) and May (9.36°C) had the coolest water temperatures since 1978, June (14.40°C) since 1982, and July (17.93°C) and August (19.44°C) since 1983. This trend continued during late summer and fall, with September (18.86°C) and October (15.35°C) about 0.8°C cooler than average. The mean for November of 10.34°C was the third coldest after 1976 and 1980. Although cooler than average, the December mean of 6.99°C, however, was not exceptionally low.

By season, winter (2.86°C) was cooler than average, spring (9.71°C) the second coolest after 1978, and summer (18.74°C) the coolest of the 21-

TABLE 6. Monthly and annual mean seawater temperature (°C) from January 1976 through December 1996 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	.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	7.95	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
1995	6.60	4.11	5.14	7.82	10.98	15.28	19.30	21.06	20.43	18.33	13.41	6.89	12.51
1996	3.55	2.12	2.87	5.38	9.36	14.40	17.93	19.43	18.86	15.35	10.34	6.99	10.58
Overall mean	4.04	2.82	3.82	6.61	10.63	15.21	18.82	20.41	19.64	16.14	11.91	7.47	11.51
CV (%)	38	41	26	12	6	4	3	3	3	6	9	17	-

year series (Table 7). At 10.90°C, the fall of 1996 ranked as the fourth coolest. Because of these water temperatures, the annual mean for 1996 was 10.58°C, nearly 1°C cooler than the long-term average of 11.51°C, and was the coldest in 21 years. These relatively extreme water temperatures in 1996 likely had important effects on adult spawning, as well as larval growth, development, and mortality, and the settlement, growth, and mortality of demersal young. Monthly mean temperatures were most variable during January through March (monthly CV = 26-41%; Table 6), the period when winter flounder spawning and early larval development occurs and most stable (CV = 3-6%) from May through October, when collections of winter flounder were dominated by young and other immature fish.

The mean temperatures given above reflected water temperatures in Niantic Bay, where the MNPS intakes are located. Water temperature in the Niantic River usually has a wider annual range, with somewhat colder temperatures in winter and warmer in summer. During March, when considerable spawning, egg incubation, and larval development takes place, mean water temperature in the Niantic River was determined from readings taken occasionally during the annual adult winter flounder surveys. From 1976 through 1996, these means differed from those recorded at MNPS by 0.6°C or less. For 10 of the years the water temperature in March was slightly cooler in the river than in Niantic Bay and in 8 of the years, including 1996 (+0.02°C) the river was slightly warmer; data were insufficient to calculate comparative means during three surveys.

Adult Winter Flounder

Relative Annual Abundance

The cold winter temperatures during the winter of 1995-96 produced heavy ice cover in the Niantic River as far south as Smith Cove (Fig. 2), which persisted until late February, when air temperatures exceeding 10°C coincided with heavy rains. The adult winter flounder survey finally began on February 27 and sampling continued for 6 weeks until April 3 (Table 8). By this time, few fish remained in spawning condition, as illustrated by the percentage of females 26 cm and larger that were gravid (Fig. 7). The pattern in the decline of gravid

TABLE 7. Seasonal[a] mean seawater temperature (°C) for 1976 through 1996 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
1995	5.31	11.35	20.26	12.87
1996	2.86	9.71	18.74	10.90
Overall mean	3.58	10.82	19.63	11.85
CV (%)	30	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.

females was similar to 1995, but more of the gravid females were found each week in 1996 until early April. This likely reflected colder water temperatures this year. Even so, most spawning apparently occurred earlier in the season under the ice cover, because more than half of the females were spent at the start of the survey in late February.

Relative annual abundance of spawning winter flounder in the Niantic River was measured by otter trawl CPUE. More than one-third of the tows made during 1996 had no winter flounder larger than 15 cm (Fig. 8). Because of the increasing frequency of zero catches in recent years, the relative abundance index was changed in this report from a median to a Δ-mean CPUE (NUSCO 1988c). The Δ-mean index of abundance is the best estimator of the population mean when the data come from a distribution that contains numerous zero values (as it has for the adult winter flounder surveys during the past few years) and the distribution of the non-zero values is approximately lognormal (Pennington 1983, 1986). The Δ-mean CPUE of winter flounder larger than 15 cm in 1996 was 1.6, which was the lowest CPUE of the 21-year time-series (Fig. 9; Table 9). The small CPUE values for 1992-96 reflected extremely low adult stock sizes present in recent years. The Δ-mean CPUE was highly correlated (Spearman's rank-order correlation coefficient $r = 0.9675$; $p = 0.0001$) with the median CPUE values (Fig. 10). The Δ-mean index was slightly greater in magnitude than the median for all years, with largest differences occurring during 1976-80.

Female winter flounder taken during the 1996 survey were mostly larger than 31 cm, with fish 45 cm or larger relatively common in comparison to all but the smallest (20.0-21.5 cm) fish (Fig. 11). A comparison of the annual standardized catch of females from 1993 through 1996 showed the scarcity of all sizes of winter flounder this year (Fig. 12). The decline in winter flounder abundance was even more striking when catches from 1981 (largest CPUE since 1976; Table 9), 1985, 1990, and 1996 were compared (Fig. 13). Large decreases in abundance have occurred for all size-classes of female winter flounder, with the exception of the very largest females. Although not abundant, larger fish in the

TABLE 8. Annual Niantic River winter flounder[a] population surveys during the spawning season from 1976 through 1996.

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
1995	February 28 - April 6	6
1996	February 27 - April 3	6

[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.

Niantic River now make up a relatively larger proportion of female winter flounder than in previous years. The large numbers of females from 23 to 34 cm collected in 1981 were likely 3- and 4-year old fish from the very strong year-classes produced in

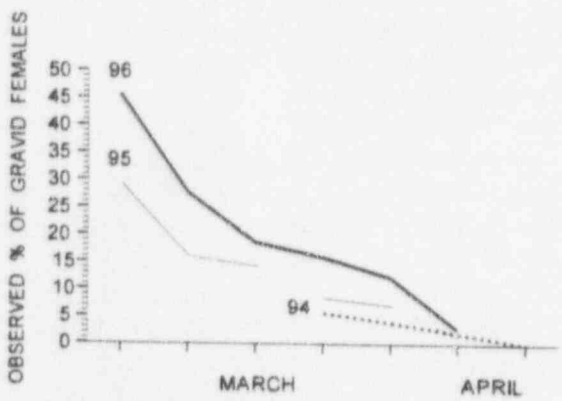


Fig. 7 Weekly percentage of Niantic River female winter flounder larger than 26 cm that were gravid during the 1994-96 adult population abundance surveys.

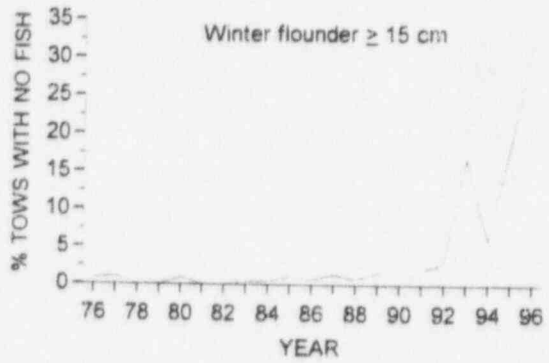


Fig. 8. Percentage of tows with no fish larger than 15 cm collected in the Niantic River by year from 1976 through 1996.

1977 and 1978. Despite relatively high abundance of age-0 fish produced in 1988 and 1992, female winter flounder from 20 to 30 cm in length have been relatively scarce in the spawning survey catches in recent years.

Also noted during the 1996 spawning survey was the collection of five partially eaten winter flounder carcasses and one wounded fish that, based on canine tooth holes in several specimens, were apparently preyed upon by one or more harbor seals (*Phoca vitulina concolor*). Seals were first noted in the Niantic River during the 1993 winter flounder spawning season, with sightings increasing during each succeeding year. In some years, seal predation may represent an increased mortality risk to Niantic River winter flounder spawners, particularly since population size has become depressed.

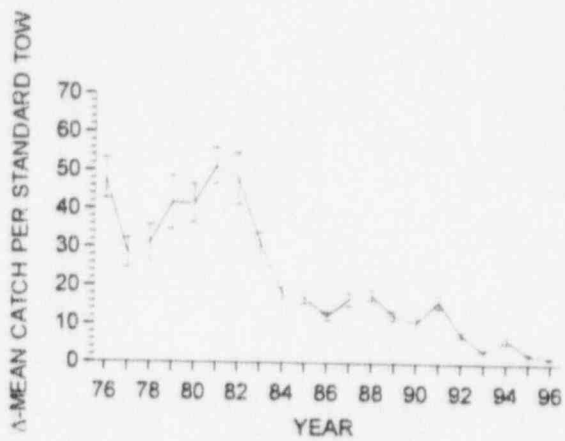


Fig. 9. Annual Δ-mean CPUE and 95% confidence interval of Niantic River winter flounder larger than 15 cm from 1976 through 1996.

TABLE 9. Annual 9.1-m otter trawl adjusted Δ-mean CPUE[a] of winter flounder larger than 15 cm[b] taken throughout the Niantic River during the 1976 through 1996 adult population abundance surveys.

Survey year	Weeks used for CPUE computation[c]	Tows acceptable for CPUE[d]	Adjusted number of tows used[c]	Non-zero observations	Δ-mean CPUE estimate	Standard error	95% confidence interval for Δ-mean CPUE
1976	7	132	224	223	48.0	2.7	42.7 - 53.2
1977	6	183	228	226	28.6	1.9	24.9 - 32.4
1978	6	135	162	162	31.2	2.4	26.5 - 35.9
1979	5	116	140	140	41.5	3.6	34.6 - 48.5
1980	5	112	145	144	41.6	2.5	36.6 - 46.5
1981	7	171	231	231	51.4	2.4	46.7 - 56.0
1982	5	116	150	150	48.1	3.4	41.4 - 54.8
1983	7	232	238	237	31.4	1.3	28.8 - 33.9
1984	7	244	287	286	18.4	0.7	17.1 - 19.7
1985	7	267	280	277	17.1	0.7	15.8 - 18.5
1986	7	310	336	334	12.3	0.5	11.3 - 13.4
1987	5	233	239	236	16.8	0.9	15.0 - 18.6
1988	6	287	312	310	17.9	0.7	16.6 - 19.3
1989	6	231	271	267	12.5	0.6	11.4 - 13.7
1990	7	260	315	314	10.7	0.5	9.8 - 11.7
1991	6	296	330	324	16.3	0.9	14.5 - 17.8
1992	7	377	406	395	7.7	0.3	7.0 - 8.3
1993	7	287	392	344	3.3	0.2	3.0 - 3.7
1994	4	184	212	201	6.4	0.5	5.5 - 7.3
1995	6	316	342	284	2.6	0.1	2.4 - 2.9
1996	6	310	342	242	1.6	0.1	1.4 - 1.8

[a] Catch per standardized tow (see Materials and Methods); differs from NUSCO (1996) because median CPUE was replaced by a Δ-mean as the index of abundance.
[b] Mostly age-2 and older fish.
[c] Effort equalized among weeks; during several years weeks with very low effort were not used for computing CPUE.
[d] Only tows of standard time or distance were considered.

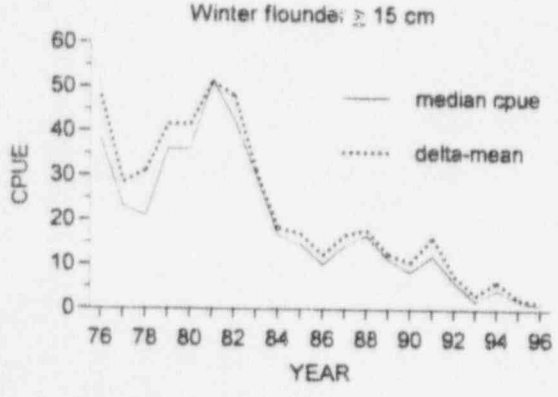


Fig. 10. Comparison between the annual median and Δ-mean CPUEs of winter flounder larger than 15 cm from 1976 through 1996.

Absolute Abundance Estimates

Adult winter flounder distribution in the Niantic River during the 6 weeks of sampling was not consistent in either time or place. Few (n = 36) fish were captured and marked during the week of March 18 in comparison to weeks before (50-61) or immediately after (61). A total of 118 fish was marked during the last week of the survey (April 1 and 3), which may have reflected fish moving into deeper waters from shallow flats not sampled as waters warmed from about 3 to 5°C from late March through early April. As found during other recent years, most adults were concentrated in relatively small areas, including the upper river arm (stations 52-54), particularly at station 54 (Fig. 2). As noted during 1995, catches at station 51 remained

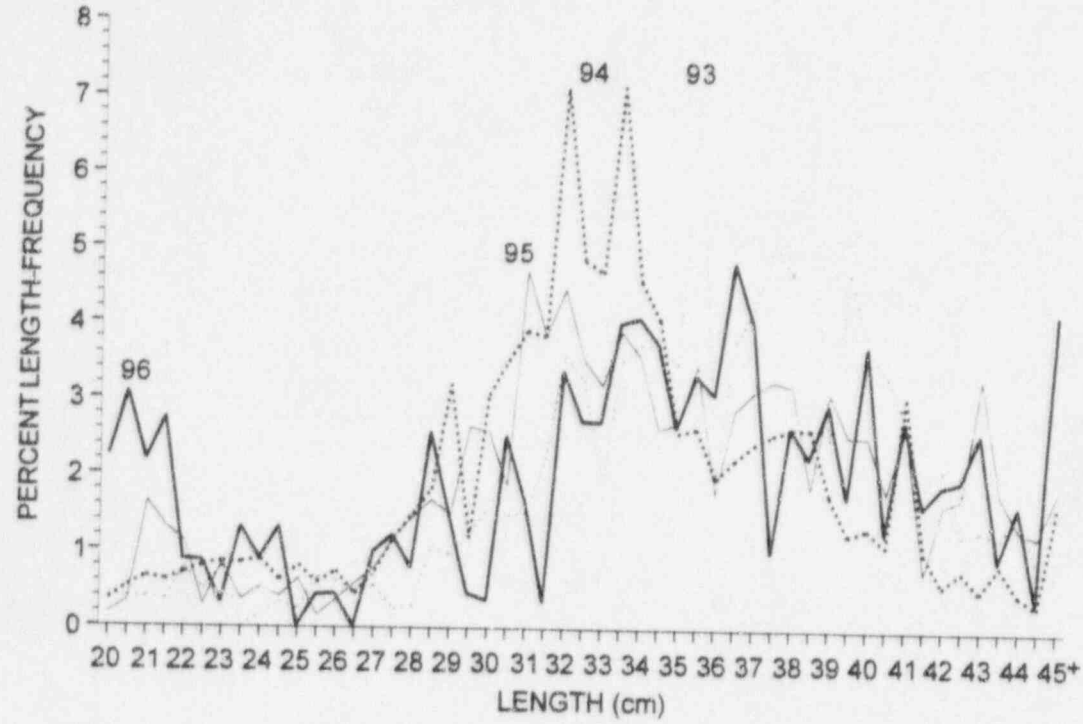


Fig. 11. Comparison of percent length-frequency distributions of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season from 1993 through 1996.

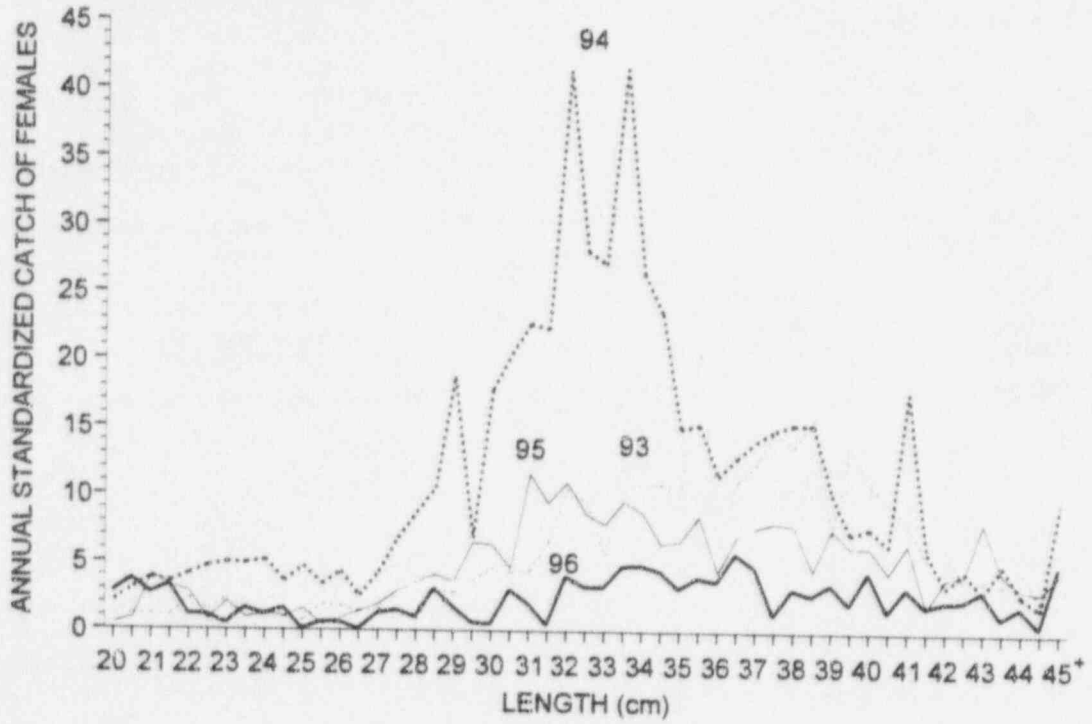


Fig. 12. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic river during the spawning season from 1993 through 1996.

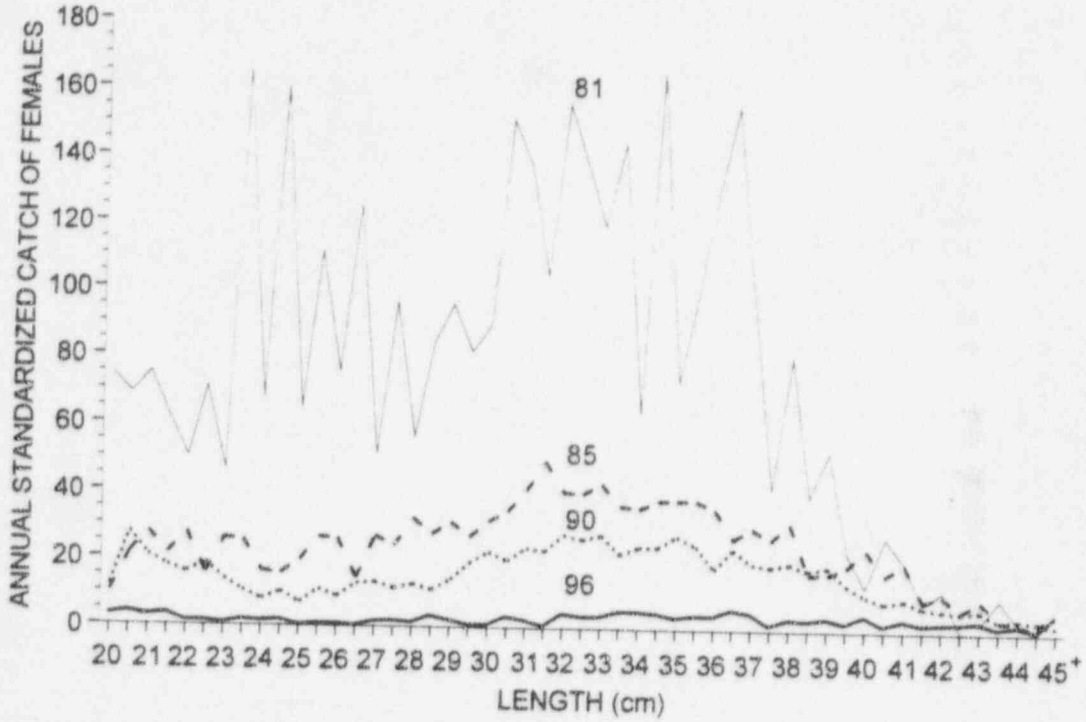


Fig. 13. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season in 1981, 1985, 1990, and 1996.

particularly poor this year, with the few fish collected there mostly found near the eastern shoreline.

Absolute abundance of winter flounder larger than 20 cm (N) spawning in the Niantic River was estimated using mark and recapture data with the Jolly (1965) model. Estimates of survival (Φ), recruitment (B), and sampling intensity (p) were also generated by this model. Because of continued low abundance of Niantic River winter flounder population, this year only 376 fish 20 cm and larger were marked with a freeze brand and released (Table 10). This was the lowest total of the time-series and only

TABLE 10. Mark and recapture data from 1983 through 1996 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	94	95
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		
1995	682	601	681	81	0	0	0	0	0	1	1	2	8	8	18	43	
1996	379	341	376	38	0	0	0	0	0	0	0	2	2	5	5	4	20

55% of the 1995 total of 681 fish branded, the previous low. Because so few fish were captured, only 38 previously-marked fish were collected in 1996. About half (n = 20) the recaptures had been marked in 1995, with most others from 1992-94.

The mark-recapture data from 1996 provided an initial abundance estimate for 1995 of 5,544 winter flounder (Table 11); this value and those of other recent years will be subject to change as additional marked fish are found during 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% CIs 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 estimated as 0.122 for 1995, which was the second highest estimate for this parameter, perhaps indicating relatively intense sampling on fish concentrated in relatively few small

TABLE 11. Estimated abundance[a] of winter flounder larger than 20 cm taken during the spawning season in the Niantic River from 1984 through 1995 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	0.596	0.062	0.474 - 0.718
1988	66,688	7,244	52,491 - 80,885	0.453	0.048	0.360 - 0.546
1989	41,744	4,730	32,474 - 51,014	0.391	0.041	0.310 - 0.472
1990	32,983	3,778	25,577 - 40,389	0.844	0.096	0.656 - 1.032
1991	61,131	7,248	46,925 - 75,336	0.200	0.025	0.151 - 0.249
1992	16,153	2,057	12,122 - 20,184	0.445	0.075	0.298 - 0.593
1993	10,435	1,830	6,849 - 14,022	0.481	0.103	0.278 - 0.683
1994	16,094	3,437	9,357 - 22,831	0.306	0.093	0.123 - 0.488
1995	5,544	1,668	2,274 - 8,814			
Mean	42,754	1,837	39,153 - 46,355	0.468	0.013	0.444 - 0.492

Year	Sampling intensity (p)	Standard 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	21,472	6,379	8,969 - 33,975
1988	0.065	0.0071	0.051 - 0.079	11,524	3,663	4,344 - 18,704
1989	0.067	0.0077	0.052 - 0.082	16,692	3,074	10,667 - 22,716
1990	0.069	0.0080	0.054 - 0.085	33,311	5,453	22,624 - 43,998
1991	0.071	0.0084	0.054 - 0.087	3,925	1,440	1,102 - 6,748
1992	0.145	0.0186	0.108 - 0.181	3,245	1,104	1,082 - 5,408
1993	0.094	0.0166	0.061 - 0.126	11,083	2,713	5,765 - 16,401
1994	0.064	0.0137	0.037 - 0.090	621	887	-1,116 - 2,359
1995	0.122	0.0365	0.050 - 0.193			
Mean	0.075	0.0043	0.067 - 0.084	19,419	856	17,741 - 21,096

[a] Estimates may vary from those reported in NUSCO (1996) because of mark and recapture data added from the 1996 adult winter flounder population survey.

areas of the river. 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), although 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. Estimates of p only approximated or exceeded 0.10 this year, in 1992 (0.145) and 1993 (0.094). Loss of information because brands were missed, or due to 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).

Although the Jolly estimates are subject to considerable error, annual Δ-mean CPUE and Jolly abundance estimates were significantly (Spearman's rank-order correlation coefficient r = 0.9021; p = 0.0001) correlated (Fig. 14). Thus, based on a Δ-mean CPUE of 1.6 for 1996, absolute abundance of winter flounder may have been less than 5,000 fish. By extrapolation, abundance in 1981 could have exceeded 150 to 200 thousand winter flounder.

Estimates of survival (Φ) have varied considerably from year to year (0.200 - 0.844; Table 11).

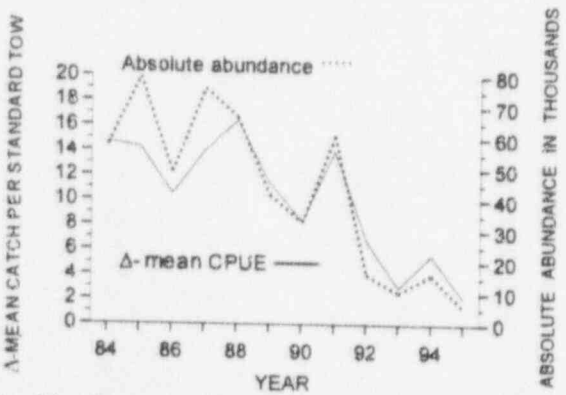


Fig. 14. Comparison between the estimates of absolute abundance in thousands of winter flounder larger than 20 cm in the Niantic River during the spawning season (dashed line) and the corresponding Δ-mean CPUE (solid line) from 1984 through 1995.

Estimated recruitment (B) values were low in 1991 (3,925) and 1992 (3,245), increased to 11,083 for 1993, and fell to a particularly low estimate of 621 fish for 1994. However, estimates for both these 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). Estimates of B were also relatively imprecise. 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 in recent years, however, appeared to accurately reflect weak recruitment of winter flounder.

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 1984-95 were compared to total abundance estimates from the Jolly model. The relative numbers of females and eggs produced each year, as determined from the standardized catch estimates, were conservatively assumed to represent about 4% of the absolute values; the range for annual values was 2.7 - 6.3%. Thus, a multiplier of 25 was used to scale standardized catch indices to absolute numbers of female winter flounder spawning in the Niantic River that are given below. In using this scaling factor it was assumed that ratios of annual standardized catch to absolute abundance during 1977 through 1983 would have been similar to those for 1984-95, had estimates of absolute abundance been available for the earlier period.

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 1996 spawning season in the Niantic River was 1.78 females for each male (Table 12). This sex ratio is larger than the long-term average for the 20-year time-series of 1.44, but less than the ratio of 2.70:1 found during 1995, which was the largest one found since 1977. Ratios of 1.50 to 2.33 in favor of females were reported by Saila (1962a, 1962b) and by Howe and Coates (1975) for

TABLE 12. Female to male sex ratios of winter flounder taken during the spawning season in the Niantic River from 1977 through 1996.

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	1.50
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
1995	2.70	2.70
1996	1.78	1.78
Geometric mean	1.44	1.44

other winter flounder populations in southern New England. Witherell and Burnett (1993) also found 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 1988b). 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 one-half to 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.

During each year, the proportion of females estimated to be mature in each 0.5-cm length increment was used with the annual standardized catch of females to obtain annual abundance indices for female winter flounder. Annual estimates of female spawner abundance and egg production were generated by multiplying relative standardized catch estimates for each by 25 (see Absolute Abundance Estimates, above). This multiplier has decreased from values of 26.316 and 28.571 used in recent years (NUSCO 1992a, 1993, 1994a, 1995a, 1996). This has resulted in reductions of approximately 5 to 13% for estimates of absolute female stock size and total egg production previously reported. Estimates of female stock size ranged between 2,427 (1996) and 68,899 (1982) fish (Table 13). Mature females generally comprised approximately one-third to one-half of each annual total, with 1995 having the highest fraction of mature females at 63%, which was related to the highly skewed sex ratio and proportionately larger fish found during that year. Despite this preponderance of larger mature females, the number of spawners has been very low in recent years because of low overall abundance of winter flounder. The total number of female spawners was used as an estimate of parental stock size for the SRR (see below).

Annual egg production estimates ranged from about 2.1 to 39.9 billion (Table 13). Differences in percent maturity due to changes in length-frequency distributions somewhat affected mean 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. Total egg production was greatest from 1981 through 1983 because of peak population abundance and moderate mean fecundity. Estimates were also relatively high in 1988, 1989, and 1991 as proportionally older and larger females dominated a moderately-sized reproductive stock. Total fecundity decreased to relatively low values of 2.1 - 8.2 billion during 1993-96 because of very low abundance.

Female size and time of spawning affects various reproductive parameters, including egg size, fecundity, and viability; embryos deposited earlier in the season appear to have better survival than eggs

TABLE 13. Relative and absolute annual standardized catch of female winter flounder spawners and corresponding egg production in the Niantic River from 1977 through 1996[a].

Survey year	Relative index of spawning females[b]	% mature females[c]	Average fecundity[d]	Relative index of total egg production[e]	Total female stock size[c]	Total egg production ($X 10^9$)[e]
1977	889	36	446,374	394.2	22,226	9.854
1978	1,415	51	506,220	716.2	35,368	17.904
1979	1,129	38	474,665	535.7	28,217	13.394
1980	916	35	464,104	425.0	22,893	10.625
1981	2,683	45	515,241	1,382.3	67,070	34.557
1982	2,756	49	578,530	1,594.4	68,899	39.860
1983	1,873	46	577,307	1,081.2	46,821	27.299
1984	872	40	574,214	500.7	21,801	12.518
1985	931	43	607,083	564.9	23,264	14.123
1986	654	42	666,312	436.1	16,361	10.902
1987	852	39	623,254	530.9	21,295	13.272
1988	1,278	53	677,596	865.7	31,939	21.642
1989	983	52	727,934	715.4	24,570	17.885
1990	580	42	637,693	370.1	14,510	9.253
1991	1,060	47	602,499	638.7	26,502	15.968
1992	533	52	732,366	390.7	13,336	9.767
1993	273	54	816,797	223.4	6,837	5.585
1994	507	55	649,622	329.4	12,676	8.234
1995	218	63	775,416	169.3	5,458	4.232
1996	97	52	844,911	82.0	2,427	2.051

[a] Some estimates differ slightly from those reported in NUSCO (1996) because of changes in the length-age key used
[b] Based on proportion of the relative annual standardized catches of winter flounder that were mature females.
[c] As a proportion of all winter flounder 20 cm or larger.
[d] Total egg production divided by the number of spawning females.
[e] A relative index for year-to-year comparisons and not an absolute estimate of production.
[c] Calculated on the assumption that the relative annual standardized catches were approximately 4.0% of absolute values; estimates vary by approximately 8% or less from those reported in NUSCO (1996) because of a 0.2% increase in the scaling factor used.

produced by smaller fish late in the season (Buckley et al. 1991). 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). 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 1981). However, highest densities occurred in summer, after the winter flounder spawning season. Based on a comparison of estimates of egg production and abundance of Stage 1 larvae (discussed below), egg mortality may be considerable in the Niantic River.

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 α parameter estimates, larval abundance during 1996 in both the river

(stations A, B, and C combined) and the bay (stations EN and NB combined) were about average for the 14-year series (Table 14). In general, annual abundances in the bay varied less than in the river. In 1985, 1987, 1988, 1989, 1995, and 1996, larval abundance was at least four times greater in the river than in the bay. No consistent relationship was found between the indices of annual abundances in the two areas (Spearman's rank-order correlation coefficient r = 0.455; p = 0.102). This lack of a relationship has two possible causes. First, if many of the larvae in the bay came from the river, then annual larval mortality rates prior to the period when larvae were flushed from the river to the bay were highly variable. Second, the Niantic River may not be the only source of larvae entering the bay (NUSCO 1992a, 1992b, 1993, 1994a, 1995a, 1996) and this possibility 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 correlated (Spearman's rank-order correlation coefficient r = 0.635; p = 0.002) with annual abundance indices in Mount Hope Bay, MA and RI (Marine Research, Inc. 1992; M. Scherer, Marine Research, Inc., Falmouth, MA., pers. comm.). As was found for the comparison between Niantic River and Bay, no relationship was found between the abundances in the Niantic River (1983-96) and Mount Hope Bay (Spearman's rank-order correlation coefficient r = 0.108; p = 0.714) This suggested that Mount Hope Bay, similar to Niantic Bay, is not a preferred winter flounder spawning area, as discussed below.

Annual spatial abundances of the first four larval developmental stages were based on cumulative weekly geometric means (Figs. 15 and 16). The abundance distribution of Stage 5 fish (i.e., newly transformed juveniles) was not examined because so few were collected by ichthyoplankton gear. Cumulative density data (the running sum of the weekly geometric means) were 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 (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 1996 in the river was about average compared to the previous 13-year period of sampling at all three stations. (Fig. 15). 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

TABLE 14. Index of annual larval winter flounder abundances and 95% confidence intervals for the Niantic River and Bay, based on the α parameter from the Gompertz function for 1983 through 1996.

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)
1995	14,075	(13,416 - 14,735)	3,091	(2,966 - 3,216)
1996	9,916	(9,631 - 10,202)	1,690	(1,535 - 1,844)

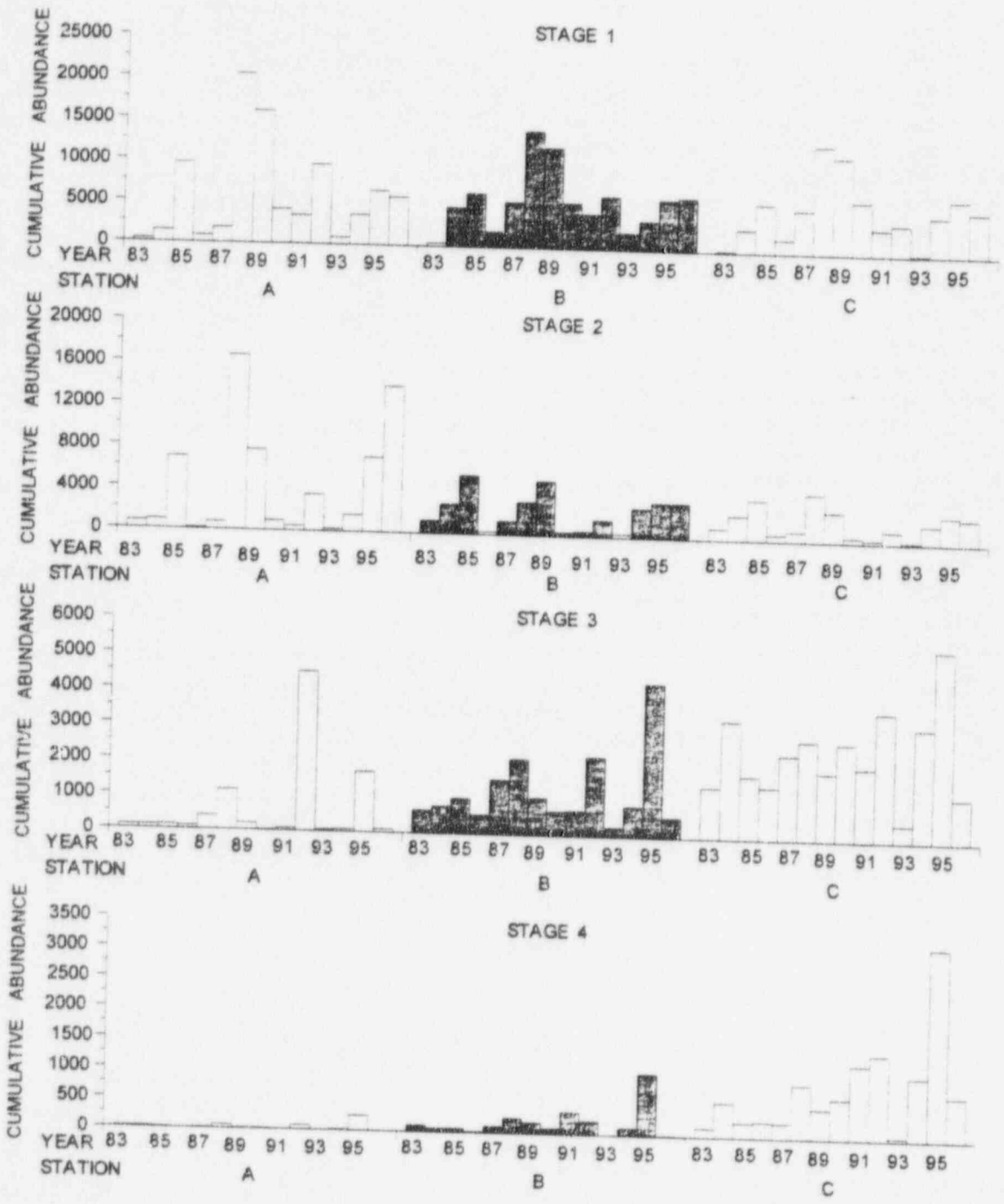


Fig. 15. Index of cumulative density for each developmental stage of larval winter flounder at the Niantic River stations A, B, and C from 1983 through 1996. (Note that the vertical scales differ among the graphs).

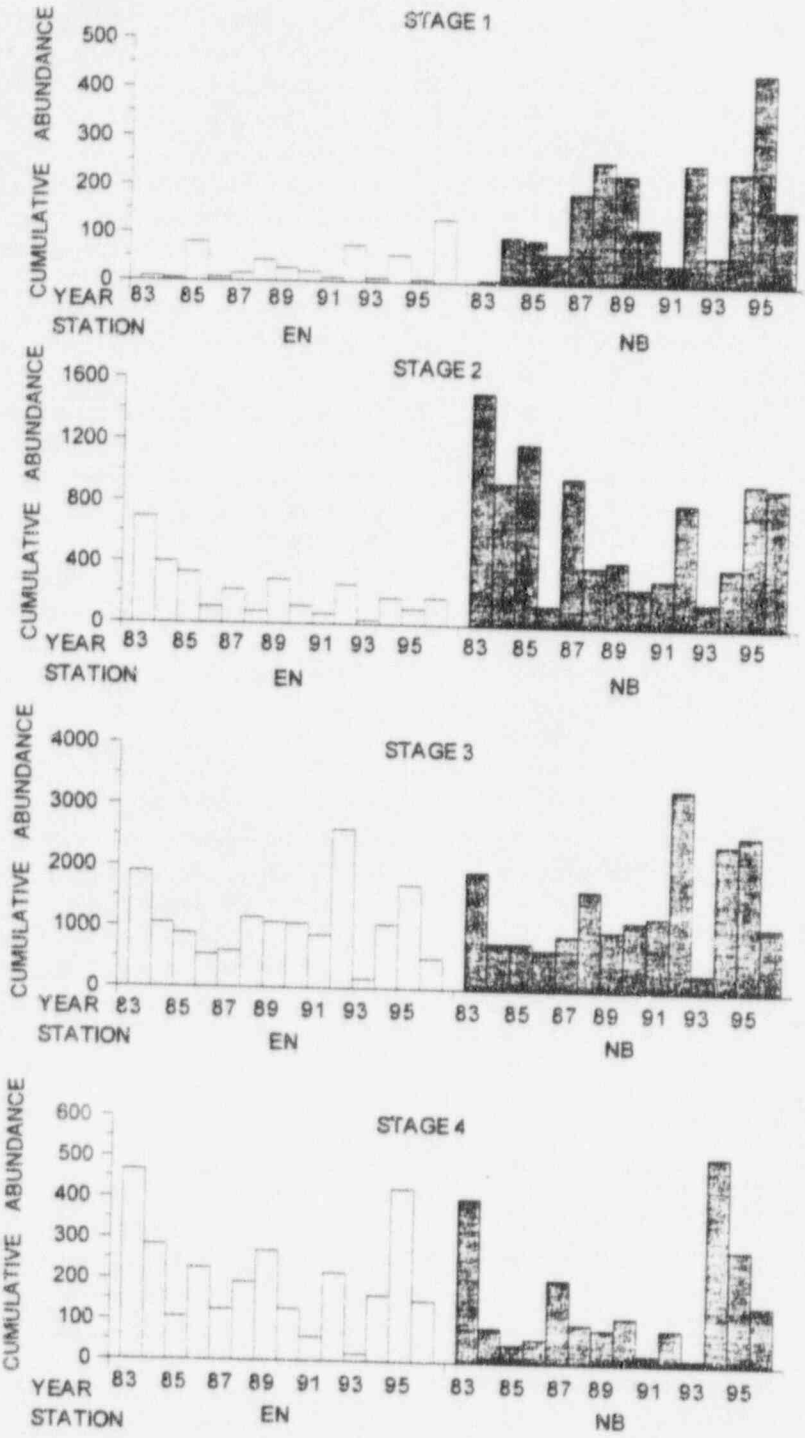


Fig. 16. Index of cumulative density for each developmental stage of larval winter flounder at the Niantic Bay stations EN and NB from 1983 through 1996. (Note that the vertical scales differ among the graphs).

demersal and adhesive and the duration of Stage 1 is short (about 10 days), the homogenous distribution suggested either that spawning was not restricted to a specific area of the river or 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 (Fig. 16) was low in comparison to the river (Fig. 15), indicating that little, if any, spawning occurred in the bay. Except for 1985 and 1996, 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. Additional evidence for possible net extrusion at EN is discussed in the Development and Growth section that follows. At station NB, ranks of annual abundance indices were similar to those of the river stations; 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 for station EN with stations C and NB (Table 15).

Stage 2 abundance in 1996 at stations B and C was about average, but at station A was the second greatest (Fig. 15). In general, annual ranks of Stage 2 abundance at the three river stations were similar to those of Stage 1. 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. 16). Annual Stage 2 abundances were consistently greater at station NB than at EN, unlike annual Stage 1 abundances in 1985 and 1996, which were similar. Significant ($p \leq 0.05$) positive correlations of abundance were found among all river stations and between stations EN and NB (Table 15).

The later developmental stages (3 and 4) of winter flounder larvae were usually 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. 15 and 16). In comparison to other years, abundances of Stage 3 and 4 larvae during 1996 were at or below average at all stations. For Stage 3, discrete spatial relationships for annual abundances were found between adjacent stations; correlations were significant ($p \leq 0.05$) between stations A and B; B and C; C and both NB and EN; and NB and EN (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.

Annual abundance of newly hatched winter flounder larvae should relate to adult spawning (i.e., 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 ($r = 0.625$; $p = 0.022$) between egg production and Stage 1 abundance (Fig. 17). The abundance of newly hatched larvae was directly related to the adult egg production under the assumption that egg hatchability was similar among years. However, Stage 1 abundance for both 1995 and 1996 appeared to be greater than expected from this relationship, suggesting a greater egg hatchability or larval survival during the past 2 years, which had the lowest estimates of egg production.

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 were not estimated for bay stations because during several years this larval stage was rarely collected outside of the Niantic River. In 1996, peak abundance for both Stages 1 and 2 larvae in the river occurred on the third latest dates of the 14-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. Buckley et al. (1990) reported that egg developmental time was inversely related to water temperature during oocyte maturation and egg incubation. Colder

TABLE 15. Matrix of Spearman's rank-order correlations among stations for the indices of annual cumulative abundance of each developmental stage of larval winter flounder from 1983 through 1996.

Stage	Station	B	C	EN	NB
1	A	0.9429[a] 0.0001 **	0.8813 0.0001 **	0.5919 0.0258 *	0.7099 0.0045 **
	B		0.8637 0.0001 **	0.5655 0.0351 *	0.6396 0.0138 *
	C			0.4158 0.1392 NS	0.7187 0.0038 **
	EN				0.2992 0.2987 NS
2	A	0.8725 0.0001 **	0.8857 0.0001 **	0.1517 0.6048 NS	0.3011 0.2955 NS
	B		0.9385 0.0001 **	0.3802 0.1799 NS	0.5648 0.0353 *
	C			0.3670 0.1967 NS	0.4725 0.0880 NS
	EN				0.7275 0.0032 **
3	A	0.8374 0.0002 **	0.5121 0.0612 NS	0.5780 0.0304 *	0.3539 0.2145 NS
	B		0.7758 0.0011 **	0.6352 0.0147 *	0.4857 0.0783 NS
	C			0.6132 0.0197 *	0.7714 0.0012 **
	EN				0.7143 0.0041 **
4	A	0.5823 0.0289 *	0.4000 0.1564 NS	0.5956 0.0246 *	0.5050 0.0655 NS
	B		0.7099 0.0045 **	0.2791 0.3338 NS	0.2904 0.3138 NS
	C			0.1780 0.5426 NS	0.2772 0.3373 NS
	EN				0.4621 0.0962 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 $p \leq 0.05$, ** - significant at $p \leq 0.01$).

than average winter water temperatures in 1996 (Tables 6 and 7) could have lengthened egg development time. A comparison between the 1983-96 February water temperatures and the annual dates of Stage 1 peak abundance in the river showed a significant negative relationship (Spearman's rank-order correlation coefficient $r = -0.660$; $p = 0.010$). The later dates of peak abundance in 1996 were also

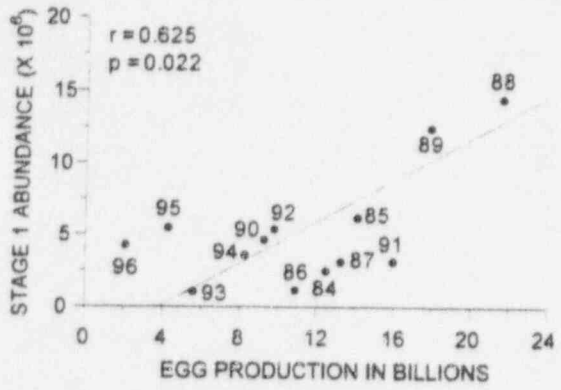


Fig. 17. The relationship (functional regression) between the index of annual Stage 1 abundance in the Niantic River (α parameter of the Gompertz function) and estimated annual egg production from 1984 through 1996.

evident for Stages 3 and 4 larvae in the river and Stages 2, 3, and 4 larvae in the bay. The later peaks of abundance in 1996 for older developmental stages could have resulted from a later hatch or possibility that colder than normal spring water temperatures in the bay (Table 7) may have slowed larval developmental rates. The relationship between water temperature and larval developmental rate is discussed in more detail below.

Development and Growth

The length-frequency distribution of each larval stage has remained relatively consistent since developmental stage determination began in 1983 (NUSCO 1987, 1988b, 1989, 1990, 1991b, 1992a, 1993, 1994a, 1995a, 1996). Stage-specific length-frequency distributions by 0.5-mm size-classes in 1996 showed some separation in predominant size-classes by developmental stage (Fig. 18). Stage 1 larvae were primarily (73%) in the 2.5 to 3.0-mm size-classes, 91% of Stage 2 were 2.5 to 4.0 mm, 91% of Stage 3 were 4.0 to 7.5 mm, and 83% of Stage 4 were 6.5 to 8.0 mm. These results were consistent from year to year and 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; Bertram et al. 1996). 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 1996 (Fig. 19). Smaller size-classes predominated in the river. In 1996, about 79% of river larvae were in the 3.5-mm and smaller size-classes, which was similar to the historical (1983-95 combined) length-frequency distribution, showing 81% of larvae in the same size-classes. The size-class distribution for the bay in 1996 was not consistent with previous findings (NUSCO 1987, 1988b, 1989, 1990, 1991b, 1992a, 1993, 1994a, 1995a, 1996). For the bay in 1996, the length-frequency distribution was bimodal with the greatest frequencies in the 2.5 to 3.0-mm size-classes (30%) and the 6.5-mm size-class (9%). Typically, the length-frequency distribution in the bay (1983-95 combined) was unimodal, with the greatest frequency (12%) at about the 6.0-mm size-class. Small larvae accounted for only 12% of the total. The length-frequency-distribution for the bay was described from a combination of samples collected at stations EN and NB; a majority of the samples were from EN. Samples at station EN were collected using a 333-µm mesh net. Previous studies indicated that smaller larvae, primarily Stage 1, can be extruded through this size net (NUSCO 1987). Larval extrusion through net mesh is related the size of the mesh opening and also may be related to the velocity of water filtered by the net. Historically (1983-95 combined), about 87% of the Stage 1 larvae from station EN were collected in March and April. During this period in 1996, all collections were made at the discharges of Unit 2 and 3 under reduced circulating water flow conditions (Unit 1 was not operating). The percentage of nominal circulating water flow at 100% capacity (Table 5) for the dates sampled in 1996 during March and April were 47% for Unit 2 and 24% for Unit 3. Reduced flow resulted in a lower cross-net water velocity that appeared to increase the number of Stage 1 larvae retained. Generally, the average flow (as a percentage of nominal) on sampling dates during Mach and April from 1983 through 1995 ranged from about 90 to 100% at all units.

Additional evidence for the greater retention of Stage 1 larvae at station EN was the similar estimated annual abundance at both EN and NB (Fig. 16). A

TABLE 16. Estimated annual dates of peak abundance of larval winter flounder for each development stage in the Niantic River and Bay and the number of days corresponding to the 95% confidence interval from 1983 through 1996.

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 6 (3)	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)
1995	March 2 (3)	March 7 (5)	April 20 (2)	May 4 (2)
1996	March 14 (6)	March 21 (4)	April 19 (8)	May 17 (5)
		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)
1995	-	April 4 (5)	April 21 (4)	April 28 (3)[b]
1996	-	April 7 (4)	April 30 (2)	May 24 (9)

[a] Due to low abundance during the 1993 sampling, the Gompertz function could not be fitted to the data.
[b] Corrected from NUSCO (1996).

review of previous annual length-frequency distributions for the bay indicated a similar bimodal distribution in 1985, with peaks occurring at the 2.5 to 3.0-mm (23%) and the 6.0-mm (11%) size-classes. On collection dates during March and April of 1985 at station EN, the flow at Unit 2 was 48% of nominal; about one-third of the samples were collected at this unit. As was found for 1996, Stage 1 abundance in 1985 at stations EN and NB was similar (Fig. 16). This suggested that some smaller larvae, primarily Stage 1, may be extruded through the entrainment sampling net under normal flow and discharge velocity conditions. However, for entrainment impact assessment, the possible undersampling of Stage 1 larvae due to net extrusion was corrected in production loss estimates that are discussed below.

An increase in frequency of larger size-classes in the river was not as apparent during 1996 as was reported in some other years (NUSCO 1987, 1988b, 1989, 1991b, 1992a, 1993, 1994a, 1995a). The previous findings suggested that some older larvae were imported into the river. Import of larger size-classes was also apparent in the length-frequency distribution at a station located in the river mouth which was sampled in 1991-93 during maximum flood current (NUSCO 1994a).

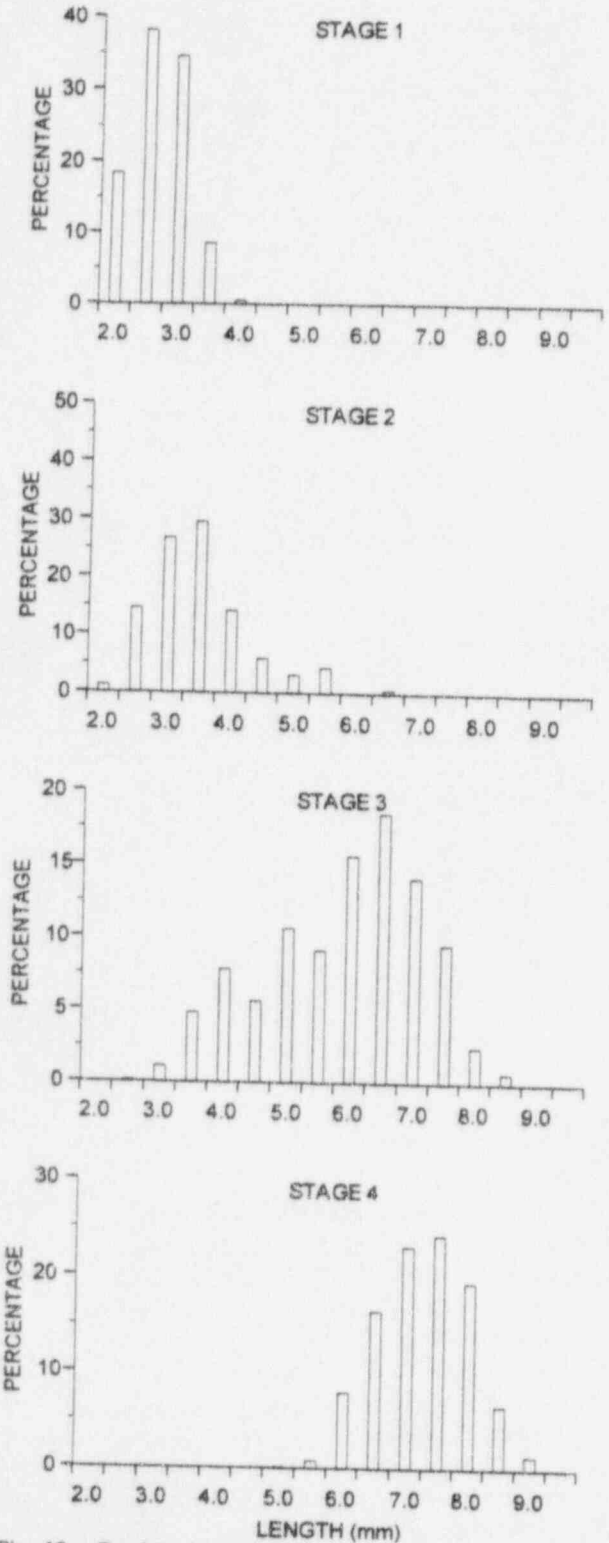


Fig. 18. Combined length-frequency distribution of larval winter flounder by developmental stage at all stations sampled in the Niantic River and Bay during 1996. (Note that the vertical scales differ among the graphs).

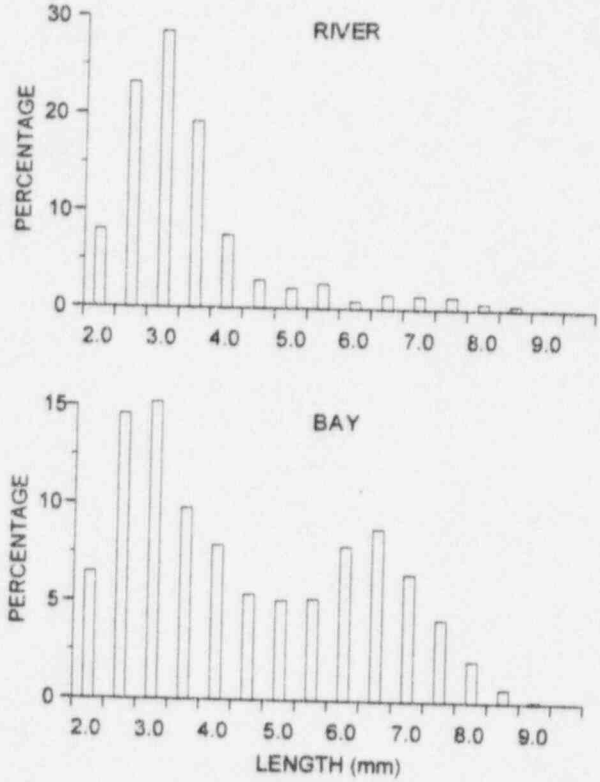


Fig. 19. Length-frequency distribution of larval winter flounder in the Niantic River and Bay during 1996. (Note that the vertical scales differ between the graphs).

Length-frequency data from entrainment collections taken from 1976 through 1996 (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 1988b). The linear portion of the sigmoid curve usually occurred in the middle of the larval season and growth rates were estimated by fitting a linear model to individual larval length measurements during this time period. This model adequately described growth and all slopes (growth rate as $mm \cdot day^{-1}$) 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 $mm \cdot day^{-1}$, with 1996 less than average. To validate this 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.

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 from 1976 through 1996. 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·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
1995	March 5 - April 30	0.088	0.081 - 0.094	5.8
1996	March 24 - June 16	0.060	0.056 - 0.063	5.2

[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.

In laboratory studies, water temperature affected the growth rate of winter flounder larvae (Laurence 1975; NUSCO 1988b). 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). The mean temperatures used may be not be indicative of the actual annual seasonal water temperatures because annual starting points varied from February 28 (1993) to April 3 (1977). A positive exponential relationship was found between growth rate and water temperature with the point for 1996 falling nearly on the line described by the relationship (Fig. 20). 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.

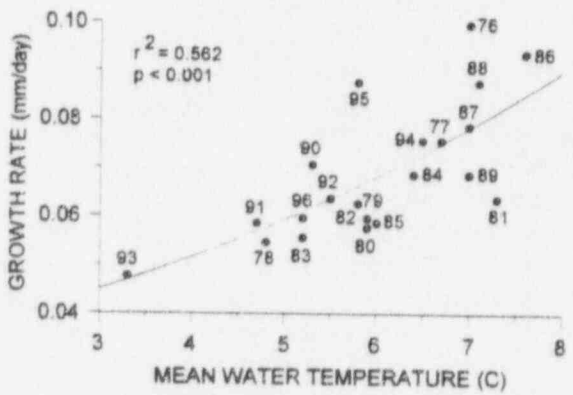


Fig. 20. The exponential relationship between mean water temperature T (°C) and the estimated growth rate G (mm per day) of winter flounder larvae at station EN from 1976 through 1996 ($G = 0.030\, e^{0.138T}$).

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. 21). Again, there was a positive relationship with larger mean lengths associated with warmer March temperatures.

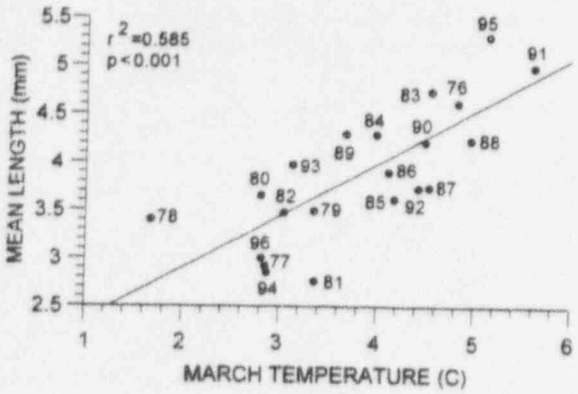


Fig. 21. The relationship between annual mean March water temperature (°C) and the mean length of winter flounder larvae during April 1-15 at station EN for 1976 through 1996.

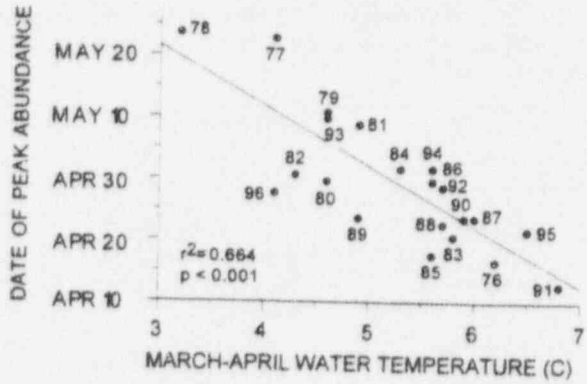


Fig. 22. The relationship between March-April mean water temperature (°C) and the annual date of peak abundance (estimated from the Gompertz function) of winter flounder larvae at station EN from 1976 through 1996.

As concluded previously from comparisons of annual length-frequency distribution and developmental stages, growth and larval development are 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 with the mean water temperature in March and April; earlier dates of peak abundance were associated with warmer mean water temperatures (Fig. 22). 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 20-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. The average March and April water temperatures in 1996 suggested a later date of peak abundance than calculated, but the earlier estimated date of peak abundance may be related to the greater abundance of early larvae (i.e., in Stage1 of development), possibly due to the reduction in net extrusion that was discussed above. Despite the wide range in annual growth rates, a consistent relationship was found between length-frequency distribution and stage of development (Fig. 18). This 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 of one another.

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. 15). Estimated growth rates for larvae in the river were generally greater than for larvae from the bay, with the 1996 rate for river larvae close to average for the 14-year series (Table 18). A linear model again provided a good fit and slopes (growth rates as mm·day^{-1}) were 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 1988b).

A laboratory study with larval winter flounder held at 8°C showed a decrease in growth as prey densities decreased, suggesting density-dependent growth due to food availability (Laurence 1977). To examine density-dependent growth in the Niantic River, annual growth rate was compared to the abundance index for Stage 2 larvae. This method also assumed that prey availability was similar from year-to-year. The annual index of Stage 2 larval abundance was the α parameter (Eq. 2) for all three river stations (Table 18). The abundance of Stage 2 was examined because during this developmental stage larvae begin to feed. A density-dependent relationship was apparent during previous years (NUSCO 1990, 1991b, 1992a, 1993), but since 1993 the relationship

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 from 1983 through 1996. 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 ($mm \cdot day^{-1}$)	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
1995	March 12 - April 30	0.130	0.121 - 0.140	6.8	2,023
1996	March 24 - May 19	0.096	0.092 - 0.099	6.7	3,677

[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).

was no longer significant ($p = 0.562$) when tested with 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 the regression analysis, it was determined that the two independent variables were not correlated (Spearman's rank-order correlation coefficient $r = 0.141$; $p = 0.631$). The multiple regression including 1996 data was significant ($p = 0.035$; $r^2 = 0.456$) with the slopes being positive for temperature and negative for Stage 2 abundance, although the Stage 2 slope was not significant ($p = 0.141$). This relationship suggested that winter flounder growth in the Niantic River may be a function of both water temperature and larval density. The varying results from year-to-year suggested that factors affecting growth, such as prey abundance, for which no information was collected, may be more complex than just water temperature and larval abundance. 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 may also be affected by larval density as well as by water temperature.

Mortality

From 1984 through 1996, total instantaneous mortality (Z) for larvae in the Niantic River from hatching to just prior to metamorphosis ranged between 82.4 and 97.9% (Table 19). Estimated larval mortality in 1996 of 2.96 was greater than the 12-year mean value of 2.67. Based on larval length-frequency distributions in the river during 1996 (Fig. 19) and previous years, most winter flounder larval mortality occurred between the 3.0- to 4.0-mm size-classes. A 74% decline in occurrence frequency was found in 1996 between these two size-classes, which included yolk-sac (Stage 1) and first-feeding Stage 2 larvae. This initially large decline was followed by

TABLE 19. Estimated larval winter flounder total instantaneous mortality rate from hatching to the 7-mm size-class from 1984 through 1996.

Year	Abundance index: Newly hatched	Abundance index: 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
1995	13,088	1,536	88.3	2.14
1996	11,151	576	94.8	2.96
				mean = 2.69

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 the Mystic River, CT. 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 1996 of 0.096 mm·day^{-1} (Table 18), a larva would require about 10 days to grow from 3 to 4 mm. The decline in abundance between these two size-classes would be equivalent to a mortality of about 12.6%·day^{-1}, which is less than the rate reported by Pearcy (1962) and for the previous 3 years in the Niantic River, which ranged from 14.6 to 19.9% (NUSCO 1994a, 1995a, 1996). 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 winter flounder development. Hjorleifsson (1992) 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, such as cnidarians.

Jellyfish predation can affect the abundance of flatfish larvae. Evidence of a causal predator-prey relationship on larvae of plaice and European flounder (*Platichthys flesus*) by the scyphomedusan *Aurelia 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). Laboratory studies showed that successful capture of plaice larvae increased as medusal size of *A. aurita* increased (Bailey and Batty 1984). Pearcy (1962) stated that *Sarsia tubulosa* medusae were important predators of larval winter flounder in the Mystic River 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 1988b).

During 6 of the 14 years (1983, 1984, 1986, 1989, 1990, and 1994), the weekly mean larval abundance of larvae at station A was negatively correlated ($p \leq 0.05$; Spearman's rank-order correlation coefficients having a range of –0.736 to –0.927) with weekly mean jellyfish volume during the period when both

medusae and larvae were collected. In 1996, jellyfish appeared later in samples, although volumes quickly increased to the average of previous years (Fig. 23) and peaked later in May. This year, however, the temporal distribution of jellyfish and winter flounder larvae overlapped very little. Peak abundance of winter flounder larvae at station A occurred from mid-March through early April and numbers decreased rapidly thereafter. Only a few larvae were collected in mid-April when jellyfish were first taken, so the disappearance of larvae at station A occurred before jellyfish became abundant. Therefore, the decline in larval abundance in 1996 was likely due to gradual flushing from the upper portion of the river and not to jellyfish predation.

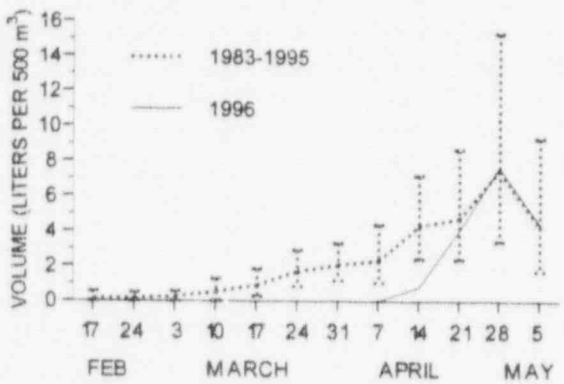


Fig. 23. Comparison of weekly mean volumes of *Cyanea* sp. at station A in the Niantic River during 1983 through 1995 (with 95% confidence intervals) with weekly volumes for 1996.

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 1995 at station EN was selected as a measure of larval recruitment, even though many of these larvae may be progeny of other spawning stocks (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 natural 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. 24). A significant ($r = -0.671$; $p = 0.001$) negative relationship was found, suggesting compensatory mortality. The 1982, 1993, and 1995 data points showed the largest deviation from the functional regression line, showing poorer recruitment in 1993 than expected and better recruitment in 1982 and 1995. The 1996 point was consistent with this relationship in that a relatively high larval recruitment index was found following relatively low egg production. Because evidence existed that many of the winter flounder larvae collected at station EN did not originate from the Niantic River, the compensatory relationship suggested that annual egg production estimates for the Niantic River were consistent with regional trends in winter flounder egg production.

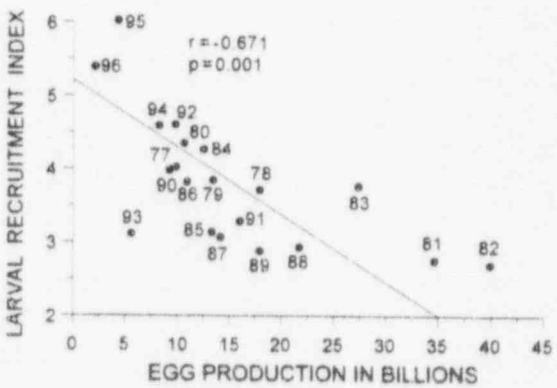


Fig. 24. The relationship (functional regression) between the annual winter flounder egg production in the Niantic River and the larval recruitment index (logarithm of the ratio of the annual abundance of 7 mm and larger larvae to the egg production) at station EN from 1976 through 1996.

Because the egg production estimate was used in calculating the larval recruitment index above, a possibility existed of introducing a spurious 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 (Z) from the river (Table 19) were compared to estimates of egg production (Table 13). To determine if density-dependent mortality could be

identified in the larval stage, the values of Z were compared to egg production estimates. In addition, the effect of water temperature on larval mortality was examined. A multiple regression model used egg production with various combinations of seasonal water temperature recorded at MNPS. The combinations included monthly means for March though May, combinations of March and April and April and May, and seasonal means for winter (January-March) and spring (April-June). The best model indicated that larval mortality increased as egg production ($p = 0.014$) increased and spring temperature ($p = 0.007$) decreased (Fig. 25). This suggested that density-dependent larval mortality may occur in the Niantic River that is further moderated by spring water temperatures. The effect of temperature on mortality may be due to its positive relationship to rates of larval growth and development. For example, the low mortality rate in 1995 was likely related to the lowest annual egg production estimate and relatively warm spring temperatures and the relatively high rate in 1988 to high egg production and cool temperatures (Tables 7, 13, and 19).

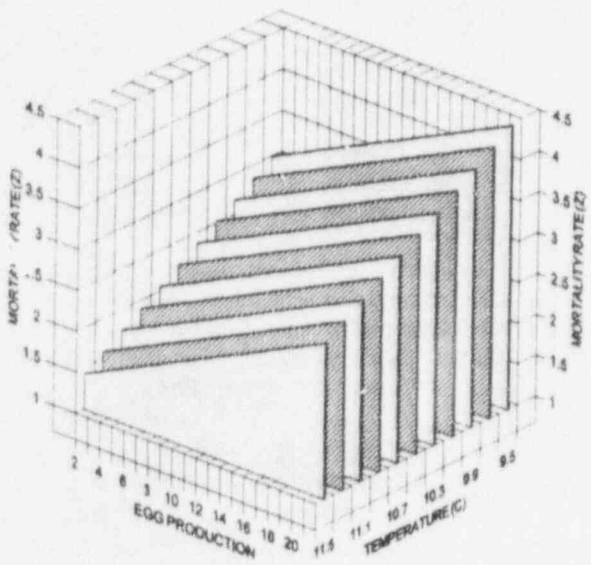


Fig. 25. Relationship between the instantaneous larval mortality rate (Z) and annual winter flounder egg production in the Niantic River and spring (April-June) mean water temperature (°C) at the MNPS intakes from 1984 through 1996 ($Z = 10.10.295 + 0.074 \cdot$egg production $- 0.734 \cdot$spring water temperature).

Juvenile Winter Flounder

Age-0 Juveniles during Summer

Abundance. Abundance of demersal young-of-the-year winter flounder was determined using a 1-m beam trawl. Although beam trawls are much more efficient than small otter trawls for collecting juvenile flatfish (Kuipers et al. 1992), the densities of young winter flounder reported herein 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 much of 1993 and occasionally in other years, were not found this year.

In contrast to the high densities of Stages 3 and 4 larvae and metamorphosing young found during 1994 and 1995, larvae were relatively less common in 1996 (Figs. 15 and 16). Following metamorphosis, numbers of young settling in the Niantic River were also relatively low in 1996. Densities of about 70 fish per 100 m^2 were found in mid-June at WA, but at all other times densities were less than 25 fish per 100 m^2 (Fig. 26). Young winter flounder were particularly scarce during late summer (August and September). Unlike most other years, abundance of young in 1996 was usually greater at WA than at LR. The CPUE abundance index for early summer at LR ($8.8 \cdot 100m^{-2}$) was the lowest of the time-series dating back to 1983 (Table 20). The index for WA (21.7) was the fourth lowest. The CPUE of 3.0 for the second half of summer at LR was again the lowest value for all years sampled there, whereas at WA the CPUE of 6.2 was only greater than during 1989 and 1993.

Overall, catches at both NR stations were considerably less than during 1995 and the year-class produced during 1996 was among the least abundant since 1984 (Fig. 27). A comparison of early and late season median CPUE values illustrated that initially strong sets of young during early summer may not necessarily result in high densities of fish at the end

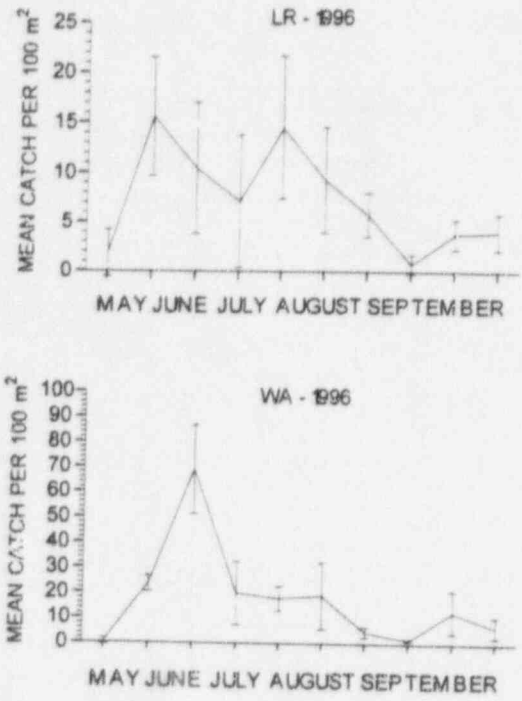


Fig. 26. Weekly mean CPUE (±2 standard errors) of age-0 winter flounder taken at Niantic River stations LR and WA by 1-m beam trawl during 1996.

of summer (Fig. 28). These differences were related to variation in mortality rates (discussed below). Most fish were produced in the Niantic River during 1988, 1992, and 1995, with year-classes from the mid-1980s, 1989, 1993, and 1996 particularly weak.

Growth. Increases in mean length over time were used to express growth of age-0 winter flounder. During 1996, growth rate was greatest in early summer with smaller increases observed in biweekly means during late summer (Fig. 29). Mean lengths were virtually identical at LR and WA until early July, when fish at LR began to show larger increases. The difference between mean length of fish at the two stations was about 20 mm in September. Fast growth after settlement followed by a rapid decline in growth rate was reported for young winter flounder in New Jersey bays by Sogard and Able (1991), who found nearly imperceptible growth by the time young reached 50 mm in length. 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 1996 was 60 mm, but at WA was only 47 mm (Table 21). This was the largest difference in mean length found between the two stations since 1988, when it was 15 mm. Growth was probably affected by water temperature, which increases respiratory and other metabolic demands. Even though faster growth occurs in warm waters, optimal growth temperatures for young winter flounder 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 cooler North Sea beaches. As noted above, water temperatures in Niantic Bay during the summer of 1996 were the coolest in 21 years. Some of the differences in mean lengths noted each year between LR and WA (Table 21) were also likely due to water temperature, as WA was generally warmer (ca. 0.5 - 1.0°C) than LR. However, no consistent differences were found between these stations, with differences in mean length during some years relatively substantial (8 - 16 mm), but small (-2 - 4 mm) in some other years.

Relatively large mean lengths associated with low abundance in some years and small mean lengths in years when fish were abundant suggested a component of growth that was density-dependent. Numerous accounts on the growth of age-0 plaice were inconclusive on the effects of density and other factors influencing growth. Bergman et al. (1988) and Zijlstra et al. (1982) re-examined reports of density-dependent growth in British waters by Steele and Edwards (1970), Lockwood (1972), and Rauck and Zijlstra (1978). The reviewers 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.

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 1996.

Survey year[a]	Station	Season[c]	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
1995	LR	Early	20	87.5	52.5 - 140.0	1.82
	LR	Late	20	15.8	12.0 - 26.7	1.96
	WA	Early	20	116.3	85.0 - 137.5	2.31
	WA	Late	20	55.0	28.3 - 70.0	0.59
1996	LR	Early	20	8.8	5.0 - 15.0	0.27
	LR	Late	20	3.0	3.0 - 6.0	1.42
	WA	Early	20	21.7	11.7 - 27.5	1.30
	WA	Late	20	6.2	3.0 - 7.0	1.76

[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.

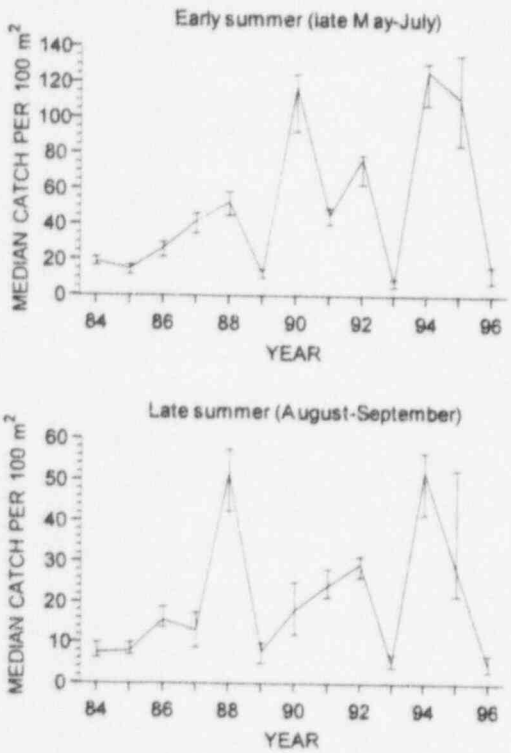


Fig. 27. Early and late summer seasonal 1-m beam trawl median CPUE and 95% confidence interval at Niantic River stations LR and WA combined from 1984 through 1996.

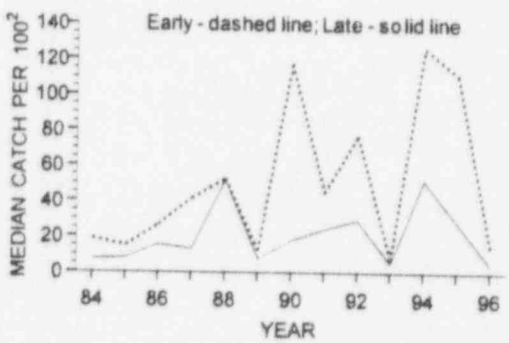


Fig. 28. Comparison between the early and late summer seasonal 1-m beam trawl median CPUEs at Niantic River stations LR and WA combined from 1984 through 1996.

The effects of density (median CPUE during July and August) and water temperature (cumulative degree-days during the period of May 15 through September 30) on growth (mean lengths achieved during late summer) at each station were examined using a multiple linear regression model. Water temperature did not significantly affect growth. Using functional regression, growth of young at LR, however, was found to be significantly ($r = -0.587$; $p = 0.027$) negatively correlated with densities of fish during late summer (Fig. 30). The relationship between abundance and mean length was only significant ($r = -0.779$; $p = 0.005$) for station WA if data outliers from 1988 and 1991 were excluded. The pattern of

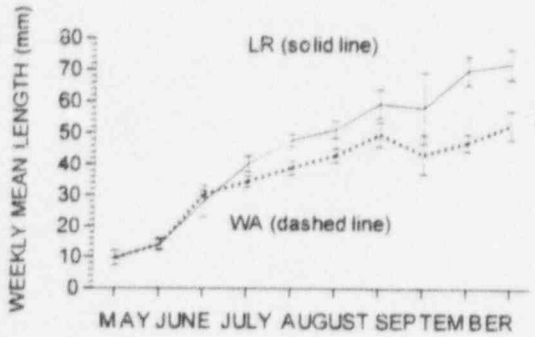


Fig. 29. Weekly mean length (±2 standard errors) of age-0 winter flounder taken at Niantic River stations LR and WA by 1-m beam trawl during 1996.

abundance for age-0 winter flounder during 1988 was unlike all other years sampled since 1983. Abundance rose to moderately high levels by mid-June and remained fairly level, without the normal decrease seen in all other years (NUSCO 1989). In 1991, water temperature, as indicated by degree-days from May 15 through September 30, was the warmest of the 14-year series, indicating a possible threshold temperature effect for that year at WA. The inclusion of data from these 2 years, however, did not make any difference in the relationship between density and growth at LR.

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.

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 found,

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 1996.

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

66	61	60	59	58	57	56	55	51	51	51	50
LR	LR	LR	LR	LR	LR	LR	WA	WA	LR	WA	WA
83	93	96	84	89	85	91	91	93	88	88	89

Mean length by station and year (continued):

48	47	47	46	46	45	44	43	43	42	42	42	40	39	38
LR	WA	WA	LR	LR	LR	WA	LR	WA	WA	WA	WA	LR	WA	WA
92	96	87	95	86	87	92	90	84	85	90	86	94	94	95

Difference in mm between the late seasonal means at LR and WA:

Year	84	85	86	87	88	89	90	91	92	93	94	95	96
Difference (LR - WA)	16	15	4	-2	0	8	1	1	4	10	1	8	13

[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.

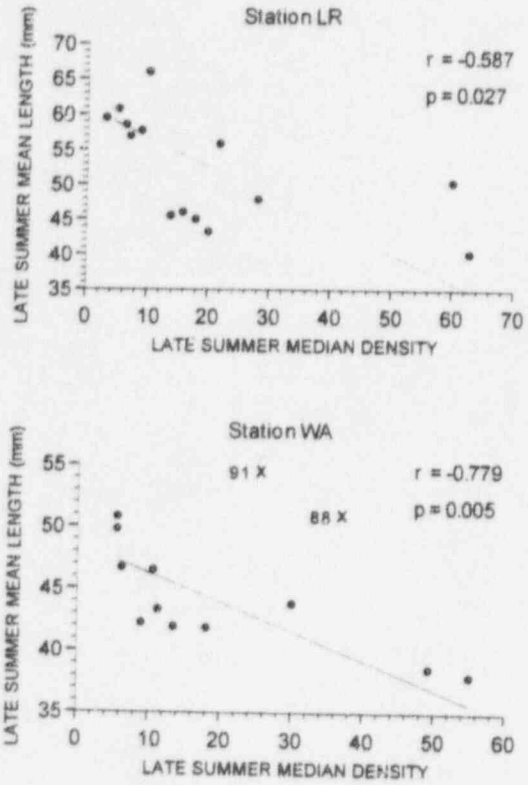


Fig. 30. The relationship (functional regression) between the density (median catch per 100 m^2) and mean length of age-0 winter flounder during late summer (August-September) at stations LR and WA in the Niantic River. For WA, data points designated by an 'x' for 1988 and 1991 were not included in the regression.

particularly for 1994 and 1995 (Table 22). The r^2 value of 0.47 for station LR in 1996 was the lowest determined since 1984 and was indicative of relatively high variance in abundance this year. No mortality estimates were made for LR and WA during the high abundance year of 1988 as slopes of these catch curves were not significantly different from zero as they were for WA in 1986, 1993, and 1996 because of considerable variation in weekly abundance. The Z_{mo} estimate for station LR in 1996 was 0.504 (equivalent to a survival rate S_{mo} of 60.4. Long-term averages of Z_{mo} at LR and WA were 0.631 and 0.557, respectively, equivalent to survival rates of 53.2% and 57.3%. 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). Annual variation in mortality rate resulted in the differences observed in year-class abundance for Niantic River winter flounder. Notably, little observed mortality in 1988 meant that a modest set of young resulted in a relatively strong year-class, whereas high mortality during early summer in 1990 and 1994 reduced initially high numbers considerably by late summer in both years.

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. 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, rather than from off-station emigration. Because of the apparent lack of production of young in Niantic Bay in comparison to the Niantic River, no further sampling was conducted in the bay.

The cause of high mortality shortly before or after settlement of young in the Niantic River 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, 1995) 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, 1995) found that the size of age-0 winter flounder significantly affected their probability of predation by sevenspine bay shrimp (*Crangon septemspinosa*), with predation greatest at settlement for the smallest fish. Mortality decreased with size and young apparently outgrew predation by shrimp when they reached 17 to 20 mm in length, which meant that fish would have to double in length after settlement before attaining a size refuge from

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 1996.

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
1995		9	-0.203 **	0.010	0.98		9	-0.138 **	0.018	0.89
1996		9	-0.116 *	0.046	0.47		8	NS	-	-

		Mortality (Z_{mo})	Survival (S_{mo})		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%
1995		0.883	41.4%		0.600	54.9%
1996		0.504	60.4%		-	-
	Mean	0.631	53.2%	Mean	0.557	57.3%
	SD	0.272		SD	0.235	
	CV	43%		CV	42%	

[a] Weekly sampling during 1984-92 and biweekly sampling during 1993-96. 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.

shrimp. Predation was also related to shrimp density and steadily increased until reaching an asymptote at shrimp densities $\geq 10.6 \cdot m^{-2}$ (Witting and Able 1995). 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 their 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). High (> $2 \cdot m^{-2}$) densities of young winter flounder at LR during 1990, 1994, and 1995 produced the steepest declines in abundance at that station. However, mortality rate at WA was also high in 1990, despite having only moderate densities of young there. Furthermore, mortality rates during 1994 and 1995 at WA were about average, despite having the highest observed abundances at that station. Estimated production of young was high in 1988 because of apparently very low mortality, but no sharp peaks in abundance were observed that year and densities generally remained below $1 \cdot m^{-2}$. 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). Mortality rates for demersal age-0 winter flounder only occasionally (e.g., 1988, 1990) showed large deviations from average. However, these occurrences, in addition to events occurring during the larval phase of winter flounder life history, determine the potential for a year-class to become exceptionally strong or weak.

Age-0 Juveniles during 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 susceptible to 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 1988c) 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. In January 1996, stations BR, TT, and NB were deleted from the TMP (see Fish Ecology section). Therefore, sample size was reduced from 42 for 1976-77 through 1994-95 to 28 for 1995-96.

The months of November through February form a transitional period that follows the 1-m beam trawl sampling of young in summer and precedes the catch of this cohort of fish as age-1 juveniles during the intensive late February-early April adult winter flounder survey in the Niantic River. Based on the availability of data for this report, the most recent Δ-mean CPUE is for the 1995 year-class (Table 23). The Δ-mean CPUE of 4.8 for 1995-96 was the lowest of the time-series, which was unexpected, given the relatively high abundance of the 1995 year-class (Table 20; Fig. 27). In 1994-95, the Δ-mean CPUE of 31.7 did reflect the strong 1994 year-class, however. Also in recent years, strong production of young in 1988 and 1992 as well as the weak 1993 year-class were also evident in the corresponding Δ-means of 29.6, 31.1, and 7.4, respectively.

Since 1983, when data were first available from Niantic River beam trawl sampling, the fall-early winter Δ-means were compared to a 1-m beam trawl median CPUE from late summer using data combined from both stations LR and WA (Fig. 31). These abundance indices track each other closely and are significantly correlated (Spearman's rank-order correlation coefficient $r = 0.6667$; $p = 0.0171$). The relationship between the TMP Δ-mean CPUE and the Δ-mean CPUE of winter flounder smaller than 15 cm taken in the Niantic River during the subsequent (late February-early April) adult winter flounder survey (see below) was negative ($r = -0.3524$; $p = 0.1275$; Fig. 32), although no longer significant with the inclusion of data from this year. More juvenile winter flounder of the 1985 and earlier year-classes were taken in the river than at the six TMP stations (five of which are outside of the Niantic River) during the preceding months. Since the 1988 year-class was produced, abundance of young has been higher in the TMP, with the exception of the 1993 and 1995 year-classes, when smaller differences were observed. The reason for the relative shift in magnitude for these abundance indices is unknown. The numbers of young taken during late fall and early winter by the TMP should be a predictor of age-1 fish abundance in the Niantic River during late winter and early spring. However, this presumes that the relative distribution of fish both inside and outside the river remains constant each year, which likely does not occur.

Relationships among abundance indices of juvenile winter flounder may have been obscured by differences in sampling gear used and variations in fish behavior in response to environmental conditions. Major biases in abundance estimation

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 1995-96.

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
1994-95	42	41	31.7	7.3 - 56.1
1995-96	28	25	4.8	1.0 - 8.6

[a] Data restricted during 1976-77 through 1994-95 to November-February for NR and JC, December-February for IN and NB, and Janua February for TT and BR and during 1995-96 to November-February for NR and JC and December-February for IN.
[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.

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

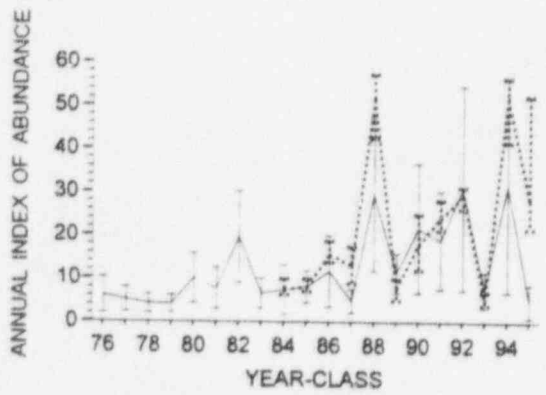


Fig. 31. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the 1983-95 late summer Niantic River (stations LR and WA combined) age-0 1-m beam trawl median CPUE (dashed line) for the 1976-95 year-classes.

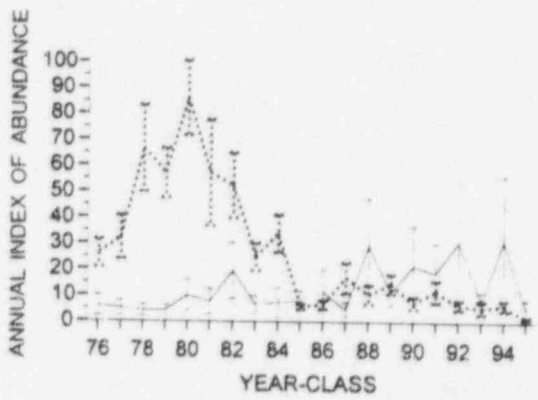


Fig. 32. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the Niantic River (stations 1 and 2) spawning survey Δ-mean CPUE (dashed line) of winter flounder smaller than 15 cm for the 1976-95 year-classes.

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 sizes achieved by age-0 winter flounder each year (Table 21) may have differentially biased the trawl sampling. The fixed locations of the otter trawl sampling stations in relation to the habitat available to juveniles also may have affected catches and the reduction in the offshore TMP stations may also have affected the comparisons.

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 CIs 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, a mixture of juveniles from a number of sources most likely was present throughout LIS during the winter, which would have influenced measures of abundance because of potential variable contributions from different stocks. These factors all contributed to weakening the strength of correlations among juvenile winter flounder abundance indices.

Age-1 Juveniles during Late Winter

Small (<15 cm)winter flounder, which include mostly age-1 fish spawned during the previous year, are incidentally captured each year during the February-April adult winter flounder surveys in the Niantic River. As for the adult CPUE, because of increasing percentages of tows with no fish smaller than 15 cm (Fig. 33), the annual age-1 juvenile winter flounder abundance index was changed from a median to a Δ-mean estimator this year. During 1996, 58% of the tows made throughout the Niantic River and 45% of the tows made in the river channel had no small winter flounder, by far the largest numbers of zero catches. Also, before calculating CPUE, adjustments were made to the catch data that were similar to those previously discussed for adult fish. In some annual comparisons, data were restricted to stations 1 and 2 in the navigational

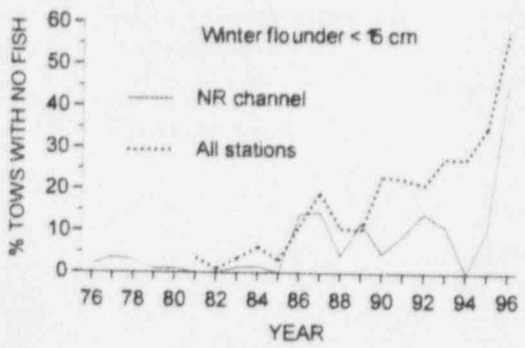


Fig. 33. Percentage of tows with no fish smaller than 15 cm collected in the Niantic River at all stations or in the navigational channel (stations 1 and 2) by year from 1976 through 1996.

channel (Fig. 2) because the distribution of small winter flounder generally varied more so than for adult fish and, moreover, no tows were made in the upper river from 1977 through 1980. The Δ-mean CPUE was highly correlated (Spearman's rank-order correlation coefficient r = 0.9786; p = 0.0001) with median CPUE values (Fig. 34). The Δ-mean index was slightly greater in magnitude than the median for all years, with largest differences occurring during 1976-83.

The Δ-mean CPUE for age-1 juveniles taken in the navigational channel of the lower Niantic River during 1996 was 1.6, the lowest value of any year since 1976 (Table 24). When tows from throughout the river were considered in the calculation, the median CPUE was 0.8, which again was a series low and continued the trend in relatively small CPUE for this time-series (Table 25).

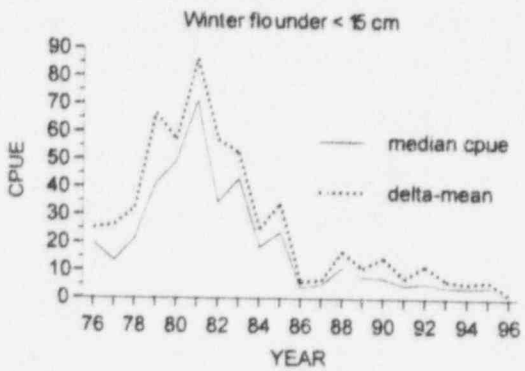


Fig. 34 Comparison of annual median and Δ-mean CPUEs for winter flounder smaller than 15 cm taken in the Niantic River at stations 1 and 2 from 1976 through 1996.

TABLE 24. Annual 9.1-m otter trawl adjusted Δ-mean CPUE[a] of winter flounder smaller than 15 cm[b] taken in the navigational channel of the lower Niantic River during the 1976 through 1996 adult population abundance surveys.

Survey year	Weeks used for CPUE computation[c]	Tows acceptable for CPUE[d]	Adjusted number of tows used[c]	Non-zero observations	Δ-mean CPUE estimate	Standard error	95% confidence interval for Δ-mean CPUE
1976	7	98	154	152	25.2	1.9	21.5 - 28.9
1977	6	166	228	222	26.3	2.8	20.9 - 31.7
1978	6	129	156	152	32.5	4.3	24.1 - 40.9
1979	5	107	135	134	66.7	8.6	49.9 - 83.4
1980	5	110	145	144	57.2	4.9	47.7 - 66.7
1981	7	93	140	140	86.2	7.4	71.7 - 100.7
1982	5	50	70	70	57.4	10.4	37.0 - 77.8
1983	7	77	77	76	52.5	6.4	39.9 - 65.0
1984	7	72	77	76	25.0	2.8	19.6 - 30.4
1985	7	82	84	84	34.0	3.7	26.7 - 41.3
1986	7	72	119	109	6.0	0.6	4.8 - 7.1
1987	5	41	50	44	6.6	0.9	4.9 - 8.3
1988	6	49	54	52	17.0	3.1	11.0 - 23.1
1989	6	50	56	50	10.6	1.9	6.9 - 14.3
1990	7	65	91	88	14.7	1.9	10.9 - 18.5
1991	6	45	60	56	7.4	1.2	5.0 - 9.8
1992	7	35	49	44	11.9	2.1	7.8 - 16.1
1993	7	36	49	45	6.6	1.0	4.6 - 8.5
1994	4	22	24	24	5.6	1.3	3.1 - 8.1
1995	6	39	54	50	6.4	1.1	4.3 - 8.4
1996	6	49	60	38	1.6	0.3	1.0 - 2.2

[a] Catch per standardized tow (see Materials and Methods); differs from NUSCO (1996) because median CPUE was replaced by a Δ-mean as the index of abundance.

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

[c] Effort equalized among weeks; during several years weeks with very low effort were not used for computing CPUE.

[d] Only tows of standard time or distance were considered.

Distribution of juvenile winter flounder largely influences their availability to sampling and apparently differs from year to year, probably as a result of variable environmental conditions, which includes water temperature and winter storm events. The relative abundance of small winter flounder has not been consistent between Niantic Bay and River from year to year (NUSCO 1993). 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 (five for 1976-95; two, IN and JC, for 1996). This time span overlapped the spawning period and also served to increase sample size. The TMP Δ-mean was then compared to the Δ-mean CPUE for fish found within the river during the spawning season (Fig. 35). 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 age-1 winter flounder in the river declined to low levels in recent years, relative abundance of these fish in Niantic Bay has increased. Except for 1988, the CPUE of fish found in the bay from 1986 through 1996 was greater than that of fish taken in the river. The catch outside the river in 1995 was the highest of the time-series, indicating that most fish from the strong 1994 year-class did not remain within or re-enter the Niantic River during the spawning season. However, the low CPUEs both inside and outside the river during the fall and winter of 1995-96 did not appear to reflect the relative abundance of fish produced in the Niantic River during 1995.

A small CPUE value determined for the Niantic River may not necessarily represent low abundance of a year-class. 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

TABLE 25. Comparison of annual 9.1-m otter trawl adjusted Δ-mean 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 1996 adult population abundance surveys.

	Navigational channel only:			Entire area of river sampled:		
Survey year[b]	Adjusted number of tows used[c]	Non-zero observations	Δ-mean CPUE estimate	Adjusted number of tows used[c]	Non-zero observations	Δ-mean CPUE estimate
1976	154	152	25.2	224	219	20.5
1977	228	222	26.3	Insufficient tows made in upper river		
1978	156	152	32.5	Insufficient tows made in upper river		
1979	135	134	66.7	Insufficient tows made in upper river		
1980	145	144	57.2	Insufficient tows made in upper river		
1981	140	140	86.2	231	225	45.5
1982	70	70	57.4	150	149	27.8
1983	77	76	52.5	238	230	24.9
1984	77	76	25.0	287	272	10.4
1985	84	84	34.0	280	272	18.7
1986	119	109	6.0	336	301	6.4
1987	50	44	6.6	239	198	4.2
1988	54	52	17.0	312	281	6.5
1989	56	50	10.6	271	246	8.7
1990	91	88	14.7	315	255	3.9
1991	60	56	7.4	330	263	2.4
1992	49	44	11.9	406	327	4.0
1993	49	45	6.6	392	312	3.1
1994	24	24	5.6	212	163	2.9
1995	54	50	6.4	342	232	1.8
1996	60	38	1.6	342	139	0.8

[a] Catch per standardized tow (see Materials and Methods) differs from NUSCO (1996) because median CPUE was replaced by a Δ-mean as the index of abundance.

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

[c] Effort equalized among weeks; during several years weeks with very low effort were not used for computing CPUE. Only tows of standard time or distance were considered.

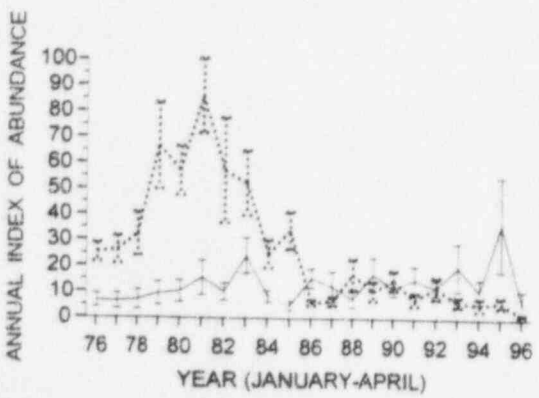


Fig. 35. Comparison between the annual January-April Δ-mean CPUE (solid line) at all trawl monitoring program stations except NR and the Niantic River (stations 1 and 2) spawning survey Δ-mean CPUE (dashed line) for winter flounder smaller than 15 cm from 1976 through 1996.

of age-1 juveniles, perhaps as a result of variable environmental conditions influencing their behavior and availability to sampling, the abundance indices determined from data taken during the TMP and the adult spawning surveys remain generally unreliable predictors of future population size. For example, the low abundance of juvenile winter flounder found during the fall-winter season of 1995-96 may have been the result of their movement into deeper water beyond the sampling stations because of colder water temperatures that occurred during these months (Table 6).

Comparisons among Life-Stages of Winter Flounder Year-Classes

Abundance indices for various life-stages of the 1976 through 1996 year-classes of Niantic River

winter flounder given previously in this report are summarized in Table 26. Coefficients of variation (CV) were used to examine annual variability in these abundance indices (Table 27). Changes (mostly increases) from CV values given in NUSCO (1996) differed by 7% or less. Variability in abundance was lowest (CV = 70%) for the number of females spawning in the Niantic River and for associated egg production (64%). In the first three adult female age-classes, variability decreased from age-3 (102%) to age-4 (84%) and age-5 (81%). This likely reflected variation in recruitment of year-classes as well as variable 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.

CVs for larval abundance indices were 75% for Stage 1, 92% for Stage 2, 98% for Stage 3, and 155% for Stage 4. The α parameter for Stage 4 larvae in 1995 was about 3.75 times larger than the next highest value and, as a consequence, the CV for this measure of abundance nearly doubled from a value of 78% determined in 1994. Variability in winter flounder abundance decreased to 82% following larval metamorphosis and settlement of demersal young, but became relatively higher (95%) for age-0 young in late summer, illustrating effects of variable mortality rates during this life stage. The CV decreased to 74% after young left shallow inshore waters during fall and early winter. Another increase was seen in CV for age-1 juveniles in the Niantic River during the adult surveys (CV = 93%) that was probably related to the previously discussed annual differences in distribution related to behavior as much as from actual variation in year-class strength. Greatest variation in abundance during early life history was expected for Stage 2 larvae, because much of the compensatory mortality was believed to occur during this stage of development. However, considerable variation was apparent in other larval and juvenile life history stages, which indicated that compensation likely takes place throughout the early life history of winter flounder.

TABLE 26. Comparison of indices of abundance of various life-stages of winter flounder for the 1976 through the 1996 year-classes[a].

	Adult indices		Larval indices							Juvenile indices	
			Niantic River stations (Feb-Jun)				MNPS	Lower	Lower	River/bay	Age-1
Year-class	Female spawners (Feb-Apr)	Annual egg prod.	Stage 1 (3 mm)	Stage 2 (3.5 mm)	Stage 3 (6 mm)	Stage 4 (7.5 mm)	(EN) (≥7 mm)	river (May-Jul)	river (Aug-Sep)	Δ-mean (Nov-Feb)	CPUE (Feb-Apr)
76	-	-	-	-	-	-	854	-	-	6.1	26.3
77	889	394.2	-	-	-	-	567	-	-	5.1	32.5
78	1,415	716.2	-	-	-	-	754	-	-	4.2	66.7
79	1,129	535.7	-	-	-	-	641	-	-	4.2	57.2
80	916	425.0	-	-	-	-	845	-	-	10.1	86.2
81	2,683	1,382.3	-	-	-	-	561	-	-	7.7	57.4
82	2,756	1,594.4	-	-	-	-	610	-	-	19.6	52.5
83	1,873	1,081.2	-	749	408	56	1,215	32.7	10.0	6.6	25.0
84	872	500.7	2,601	1,501	573	67	917	18.8	6.3	7.4	34.0
85	931	564.9	6,260	4,676	584	35	312	13.3	7.0	8.1	6.0
86	654	436.1	1,279	176	301	24	510	33.8	13.8	11.7	6.6
87	852	530.9	3,218	829	1,036	48	315	59.2	17.9	4.8	17.0
88	1,278	865.7	14,491	4,469	1,531	210	419	61.3	60.0	29.6	10.6
89	983	715.4	12,463	3,976	589	73	327	17.5	8.8	11.3	14.7
90	580	370.1	4,728	355	258	57	508	156.3	20.0	21.7	7.4
91	1,060	638.7	3,248	252	343	112	439	77.5	21.7	19.0	11.9
92	533	390.7	5,476	1,367	2,339	195	1,003	90.0	28.1	31.1	6.6
93	273	223.4	1,187	133	111	6[b]	130	10.6	5.0	7.4	5.6
94	507	329.4	3,692	1,248	429	90	834	128.8	62.9	31.7	6.4
95	218	169.3	5,580	2,023	2,615	787	1,804	87.5	15.8	4.8	1.6
96	97	82.0	4,373	3,677	265	31	462	8.8	3.0	-	-

[a] Differs from NUSCO (1996) for female spawners and annual egg production because of changes in the length-age key used and in age-1 CPUE because median CPUE was replaced by a Δ-mean as the index of abundance.

[b] 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.

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

Life stage	Abundance index used	Number of observations	CV
Female spawners	Annual standardized catch	29	70%
Age-3 females	Annual standardized catch	18	102%
Age-4 females	Annual standardized catch	17	84%
Age-5 females	Annual standardized catch	16	81%
Eggs	Egg production index	20	64%
Stage 1 larvae	α parameter of Gompertz function	13	75%
Stage 2 larvae	α parameter of Gompertz function	14	92%
Stage 3 larvae	α parameter of Gompertz function	14	98%
Stage 4 larvae	α parameter of Gompertz function	14	155%
Age-0 young	Median CPUE at station LR (May-July)	14	82%
Age-0 young	Median CPUE at station LR (August-Sept)	14	95%
Age-0 young	Fall-winter Δ-mean at trawl stations	20	74%
Age-1 juveniles	Δ-mean CPUE of fish < 15 cm in Niantic River	20	93%

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

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%, which is consistent with values of Niantic River winter flounder. The CV of European flounder abundance decreased from 172% (n = 9) in the larval stage to 99% (n = 8) in newly settled young to 80% (n = 8, 12) in young during September and again 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%). Smaller variation was found in post-larval young during mid-summer (9, 30%) and age-2 recruits (9, 35%), which is less than was found for winter flounder. He attributed the decline in variation of abundance for 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 found in The Netherlands was stabilized between years as a result of density-dependent regulatory processes (i.e., shrimp predation) on newly metamorphosed fish. In contrast, year-class strength of plaice in Swedish bays varied to a greater degree (CVs = 67-118%), which was thought to be related to temperature effects during the larval stage and more variable crustacean predation on newly metamorphosed young in more northerly waters than The Netherlands (Pihl 1990; Pihl and van der Veer 1992). Thus, population regulation in flatfishes may be coarsely determined in earliest life history by variable survival of eggs and larvae and then fine-tuned by density-dependent mortality of newly metamorphosed juveniles (van der Veer and Bergman 1987).

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. Indices of female spawners and egg production were highly correlated (Spearman's rank-order correlation coefficient r = 0.9264; p = 0.0001), 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 also found among most larval stage abundance indices. Niantic River Stage 4 larval abundance was also significantly correlated with age-0 juveniles collected during early (r = 0.6528; p = 0.0114) and late (r = 0.6747; p =

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

Index[a]	Adult egg production	Stage 1 larvae	Stage 2 larvae	Stage 3 larvae	Stage 4 larvae	Larvae (≥7 mm) at MNPS discharge
Female spawners	0.9624[b] 0.0001 ** 20	0.2967 0.3249 NS 13	0.1253 0.6696 NS 14	0.1736 0.5528 NS 14	0.1692 0.5630 NS 14	0.0211 0.9298 NS 20
Adult egg production		0.3462 0.2466 NS 13	0.1692 0.5630 NS 14	0.2571 0.3748 NS 14	0.1692 0.5630 NS 14	-0.0767 0.7479 NS 20
Stage 1 larvae			0.8187 0.0006 ** 13	0.5934 0.0325 * 13	0.5879 0.0346 * 13	0.0165 0.9574 NS 13
Stage 2 larvae				0.6176 0.0186 * 14	0.3626 0.2026 NS 14	-0.0550 0.8520 NS 14
Stage 3 larvae					0.6967 0.0056 ** 14	0.2440 0.4006 NS 14
Stage 4 larvae						0.4593 0.0985 NS 14

[a] Indices used correspond to those given on Tables 26 and 27.

[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).

0.0081) summer (Table 29). Age-0 juvenile abundance during late summer and late fall-early winter was also correlated (Table 29; Fig. 31). The densities of larger (≥ 7 mm) larvae collected in entrainment samples at MNPS were not significantly correlated with most adult, larval, or juvenile abundance indices, although some correlations have strengthened over time (Tables 28 and 29). As discussed previously, although not significantly correlated, the abundance the Niantic River spawning survey age-1 Δ-mean CPUE was inverse with that of young during fall and early winter in the TMP (Fig. 32). This was probably mostly related to changes in distribution rather than an indication of any compensatory mortality.

Niantic River winter flounder are not fully recruited until age-5, if catch indices are assumed to be representative of annual relative abundances (NUSCO 1990). Thus, age-3 or age-4 fish probably should not be used as an index of year-class strength because only a fraction of these fish likely are present on the spawning grounds each year. Furthermore, the presence of immature fish may vary from year to year because of environmental conditions. Several significant correlations were found between the abundances of ages-3 through 5 female spawners and those of juvenile winter flounder and larvae 7-mm and larger taken at MNPS (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, but not with age-5 females. Significant negative correlations were found between ages-3 through 5 females and the age-0 fall-winter Δ-mean CPUE. However, the form of these relationships is unclear and may be a statistical artifact (Fig. 36). Persistence of negative correlations in future years perhaps result from unknown processes operating after winter flounder become age-1 that produce fewer age-5 adult recruits from more abundant year-classes of juveniles.

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
Larvae (≥7 mm) at Millstone discharge	0.4593[b] 0.0985 NS 14	0.4769 0.0846 NS 14	0.2528 0.3833 NS 14	-0.1536 0.5180 NS 20	0.2377 0.3129 NS 20
Niantic River Stage 4 larvae		0.6528 0.0114 * 14	0.6747 0.0081 ** 14	0.3333 0.2657 NS 13	-0.0138 0.9644 NS 13
Lower river early age-0 juveniles			0.8901 0.0001 ** 14	0.5179 0.0698 NS 13	-0.1238 0.6870 NS 13
Lower river late age-0 juveniles				0.6667 0.0128 * 13	-0.0578 0.8513 NS 13
Fall-early winter river-bay age-0 juveniles					-0.3524 0.1275 NS 20

[a] Indices used correspond to those given on Tables 26 and 27.

[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).

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.

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 section, 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 1996 adult winter flounder survey along with a change in the scaling factor used, updated fishing mortality estimates provided by CT DEP, and a change in the age-length key used resulted in some differences between current estimates of spawners and recruits and those reported in NUSCO (1996). Ages were formerly assigned to female winter flounder using an age-length key described in NUSCO (1989). Briefly, this key made use of derived probability density functions approximated by fitting a two-parameter Gompertz cumulative function (Draper and Smith 1981) to Niantic River female winter flounder age and length data. However, it appeared that this method did not adequately reflect the proportions of females in ages 3 through 5. A re-examination of the data resulted in the creation of a revised age-length key that more closely matched empirical age-length data (Fig. 37).

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) for this model was estimated as 1.42 with a standard error of 0.525 (37% of the parameter value). The two-parameter model estimates were used as initial values for fitting the three-parameter SRR

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

Index[a]	Larvae (≥7 mm) at MNPS discharge	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]	0.3375[c] 0.1708 NS 18	-0.1273 0.7092 NS 11	0.0818 0.8110 NS 11	-0.5227 0.0260 * 18	0.7734 0.0002 ** 18
Age-4 female spawners[b]	0.2500 0.3332 NS 17	-0.3818 0.2763 NS 10	-0.5273 0.1173 NS 10	-0.8866 0.0001 ** 17	0.6806 0.0026 ** 17
Age-5 female spawners[b]	0.7088 0.0021 ** 16	-0.5333 0.1392 NS 9	-0.5833 0.0992 NS 9	-0.6534 0.0061 ** 16	0.4588 0.0738 NS 16

[a] Early life history indices used correspond to those given on Tables 26 and 27.

[b] Determined by applying an age-length key (NUSCO 1989) 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).

model with temperature effects (Eq. 7). The three-parameter SRR explained 71% of the variability associated with the recruitment index. Relationships resulting from fits of both Ricker models are shown as the curved lines in the central portion of Figure 38 as follows: the unadjusted SRR (two-parameter model; Eq. 7) is shown as the thinner solid line and the three-parameter model (SRR adjusted for T_{Feb}) is represented by the thicker solid line. The outermost two dashed lines illustrate low recruitment in the warmest year (1991; T_{Feb} = +1.95) and high recruitment in the coldest year (1977; T_{Feb} = –2.45).

Using the three-parameter model, α was estimated at 1.473, with a standard error of 0.306, which is 21% of the parameter value (Table 32). This was the lowest estimate of α made to date, reflecting low recruitment in the early 1990s; previous values ranged from 1.710 through 2.646 (NUSCO 1990, 1991b, 1992a, 1993, 1994a, 1995a, 1996). Variation in α estimates 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 1988-92 data points on the estimate of α were illustrative of higher recent exploitation and poor recruitment (Fig. 38). Because of the relatively high abundance of juvenile winter flounder from the 1988 year-class, numbers of females were expected to increase during 1992-94 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 (Fig. 5).

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 value of 2.450×10^{-5} was indicative of the consistency found for β parameter estimates (range of $2.140 - 2.583 \times 10^{-5}$; NUSCO 1990, 1991b, 1992a, 1993, 1994a, 1995a, 1996). The parameter ϕ, which is an estimate of the effect of February temperature deviations (T_{Feb}) from the 1977-92 mean of 2.81°C, was –0.418, which was quite similar to the estimate of –0.415 given in NUSCO (1996). Other estimates of ϕ ranged from –0.412 through –0.259 (NUSCO 1990, 1991b, 1992a, 1993, 1994a, 1995a). The values for ϕ are negative and although the reasons for the apparent relationship between winter flounder recruitment and February temperatures remain unknown. February coincides with most spawning, egg incubation, and hatching, which along with larval growth are all temperature-dependent. Buckley et al. (1990) noted that the winter flounder reproductive process appears optimized for cold winter temperatures that are followed by a gradual spring warming. Adult acclimation temperatures and egg and larval incubation temperature affected larval size and biochemical composition. Cold winters and

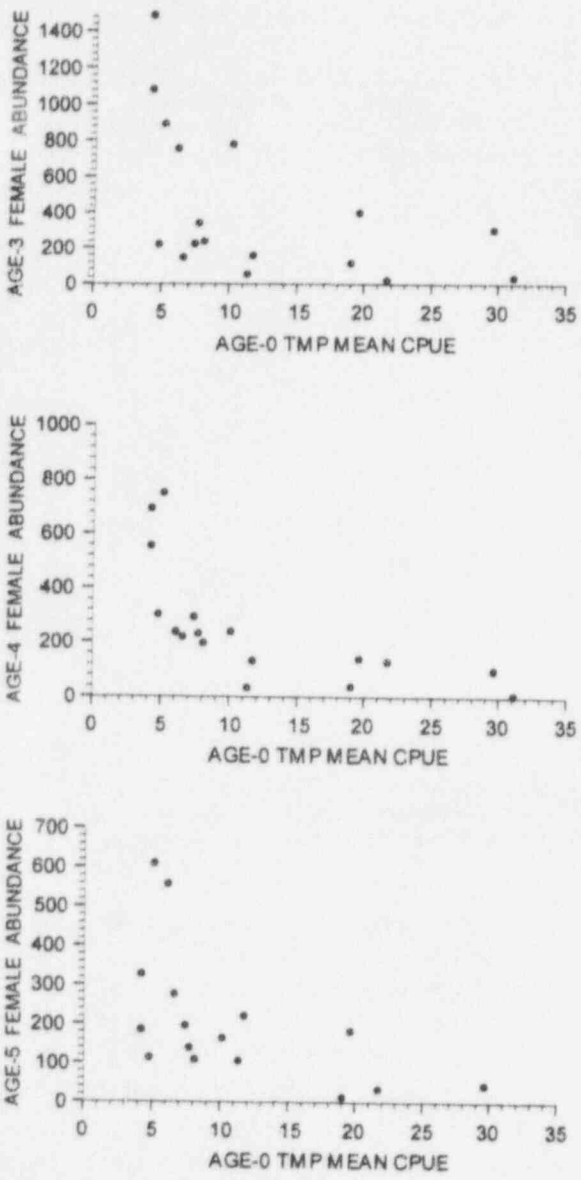


Fig. 36. Comparison between the late fall-early winter seasonal Δ-mean CPUE of age-0 winter flounder (TMP data) and the relative annual abundance of ages-3, 4, and 5 Niantic River female winter flounder. (Note that the vertical scales differ among the graphs).

warm springs produced large larvae that were in the best condition at first feeding, which favored high survival and partly explained 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, which is generally consistent over time each year. Therefore, a 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 parameters directly estimated from stock-recruitment data (Fig. 38), Table 32 includes four derived biological reference points. Ricker's stock-at-replacement, or P_{rep} (Eq. 11), was estimated at 72,239 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, is 113,415 lbs. Stocks with biomass less than the critical size of 25% of MSP (here, 28,354 lbs) are considered to be overfished (Howell et al. 1992). The present equilibrium size $P_{E(F)}$ (Eq. 9) of 15,808 spawners refers to the sustainable or equilibrium size to which the stock could converge if present (through 1992) exploitation and other conditions remained unchanged. The calculated (Eq. 10) value of F that would achieve equilibrium stock size was 1.38, which is consistent with the high estimates of F made in recent 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 were used in this study as a comparison to 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 underestimated 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

TABLE 31. Annual Niantic River winter flounder stock-recruitment data based on indices of egg production for the 1977 through the 1992 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	17,565	51,250	2.92	0.36	-2.45
1978	31,915	39,781	1.25	1.09	-1.72
1979	23,874	32,827	1.38	1.48	-1.33
1980	18,939	26,153	1.38	2.38	-0.43
1981	61,599	24,345	0.40	2.63	-0.18
1982	71,052	30,997	0.44	1.56	-1.25
1983	48,182	30,205	0.63	3.74	0.93
1984	22,315	23,537	1.05	4.02	1.21
1985	25,175	22,998	0.91	2.36	-0.45
1986	19,433	20,002	1.03	3.38	0.57
1987	23,659	16,737	0.71	3.27	0.46
1988	38,577	14,137	0.37	2.67	-0.14
1989	31,881	9,066	0.28	3.24	0.43
1990	16,494	6,293	0.38	4.28	1.47
1991	28,463	3,600	0.13	4.76	1.95
1992	17,409	1,625	0.09	3.68	0.87
Mean	31,033	22,097	0.60	2.81	
CV	52%	61%		43%	

[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 25 to convert relative abundance to an absolute population size. Indices of female spawners and recruits differ from those reported in NUSCO (1996) because of data added from the 1996 adult winter flounder population survey and changes in the mortality rates, age-length key, and in the scaling factor used in the calculations.

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, biases caused by changing fishing rates are avoided and independent means of validating SRR-based estimates are provided. 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). The value of 3.74 was re-scaled for numbers of fish on the basis of the following relationship:

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

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). These calculations used population data previously reported (NUSCO 1990) and an M of 0.25 (NUSCO 1996). Small changes in the fraction mature from a review of the age-length data previously discussed resulted in a mean weight of 1.57 lbs per female spawner for the Niantic River unfished winter flounder stock and a mean fecundity of 972,205 eggs per female spawner (Table 33).

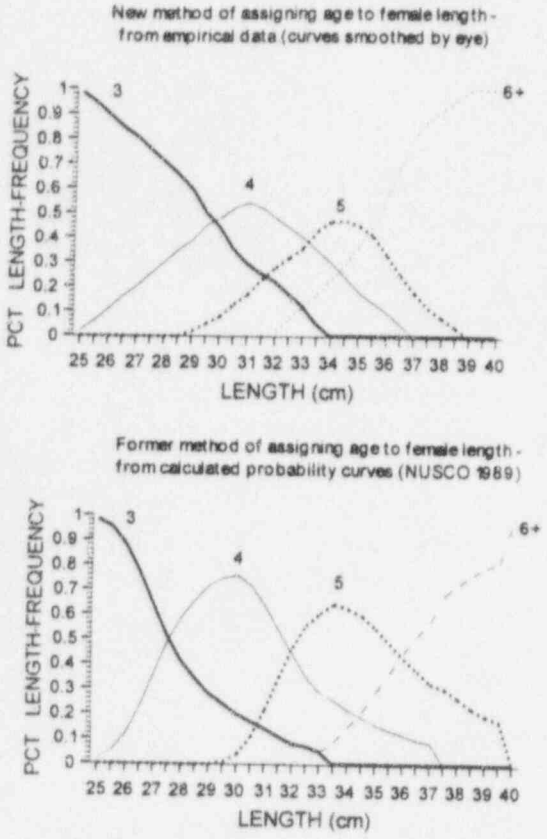


Fig. 37. Comparison of female winter flounder length-age distribution as presently determined (from empirical data with curves smoothed by eye) and as formerly determined (calculated from probability curves; NUSCO 1989).

Using the derived mean weight, the re-scaled α parameter for this study was obtained as:

$$\alpha_0 = \alpha_{biomass} \cdot (\text{mean weight}) = 3.74 \cdot (1.57 \text{ lbs}) = 5.87. \quad (20)$$

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). Using field data, the slope of the SRR at the origin decreases with increasing exploitation rates and α can be considered as the "remaining growth potential" or "growth reserve" of the stock. Consequently, the large difference between the derived value of α (5.87) and regression estimates of α based on field data (e.g., 1.47; Table 32) reflects the difference in growth reserves between unfished and highly exploited stocks of winter flounder. Using an unfished stock as a starting point for a population dynamics simulation has other advantages, depending upon the particular scenario selected. For example, the 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 both 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, mostly (> 90%) during April and May. The entrainment estimate for 1996 of 53.9 million was the second lowest since three-unit operation began in 1986 (Table 34). This largely can be attributed to plant operation because the cooling water volume during 1996 was the lowest of the three-unit operational period. The α parameter, an index of larval abundance (Eq. 2), was 1,388 in 1996, which was lower than the 21-year average of 1,685.

Due to refueling outages, Units 1 and 2 did not operate during the entire larval winter season and Unit 3 was shut down after March 30 (Fig. 39). The decrease in cooling water usage resulted in a calculated reduction in larval entrainment of about 72% (equal to 138 million larvae) from that expected if all three units had operated fully during this period.

As in previous years, Stage 3 larvae predominated in entrainment collections. In 1996, the percentages of each developmental stage entrained were 25% for Stage 1, 30% for Stage 2, 34% for Stage 3, and 11% for Stage 4. Percentages for 1983-95 combined were 3% for Stage 1, 20% for Stage 2, 65% for Stage 3, and 12% for Stage 4 of development. Compared to

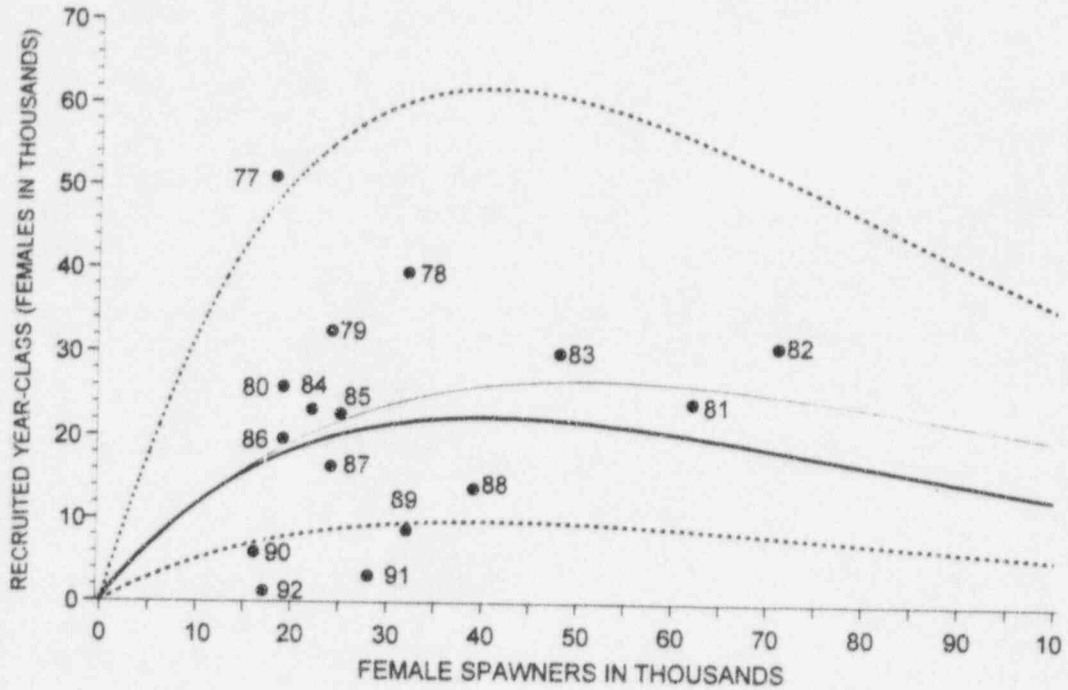


Fig. 38. The Ricker SRRs of Niantic River winter flounder (see text for explanation of the four curves plotted). Calculated recruitment indices of the 1977 through the 1992 year-classes are shown.

previous years there was a noticeably large increase for developmental Stage 1 in 1996. As noted previously, the greater frequency of small larvae collected in entrainment samples in 1996 was likely due to collections made under reduced cooling-water flow, which implies that many smaller larvae likely are extruded through the entrainment sampling net under normal operation with higher flows.

TABLE 32. Parameters of the Ricker stock-recruitment model fitted to data for Niantic River female winter flounder spawners from 1977 through 1992 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)	5.87	-	-
α (current compensatory reserve)	1.473[b]	0.306	4.82 **
β (stock-dependent compensatory rate)	2.450×10^{-5}	4.88×10^{-6}	5.02 **
ϕ (environmental [temperature] effect)	-0.418	0.075	5.59 **
Derived points of reference:		Scaled as numbers of fish	Scaled as biomass (lbs)
Unfished stock equilibrium size (P_{rep}; called maximum spawning potential by Howell et al. 1992)		72,239[c]	113,415
Present (through 1992) equilibrium size ($P_{E(F)}$)		15,808[c]	15,808
F for $P_{E(F)}$ = 15,808 female spawners		1.38	-
Estimate of critical stock size (25% of maximum spawning potential)		-	28,354

[a] (t-test for H_0: parameter estimate = 0); t =13 with df = n-3 = 13 rejects H_0 at $p \leq 0.01$. R^2 for the fit to the model was 0.71.
[b] Includes the effects of recent exploitation rates.
[c] Mean weight of a female spawner for the unfished stock is 1.57 lbs and for the current exploited stock is 1 lb.

TABLE 33. Biomass calculations for the Niantic River winter flounder female spawning stock at equilibrium based on an instantaneous natural mortality rate of M = 0.25 and an instantaneous fishing mortality rate of F = 0 (i.e., an unfished 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	778.80	0.10	77.88	0.554	223,735	43.15	17.424
4	606.53	0.38	230.48	0.811	378,584	186.92	87.257
5	472.37	0.98	462.92	1.088	568,243	503.66	263.053
6	367.88	1.00	367.88	1.377	785,897	506.57	289.116
7	286.50	1.00	286.50	1.645	1,004,776	471.29	287.868
8	223.13	1.00	223.13	1.873	1,201,125	417.92	268.007
9	173.77	1.00	173.77	2.057	1,366,951	357.44	237.535
10	135.34	1.00	135.34	2.203	1,502,557	298.15	203.356
11	105.40	1.00	105.40	2.304	1,598,597	242.84	168.492
12	82.08	1.00	82.08	2.390	1,682,208	196.17	138.076
13	63.93	1.00	63.93	2.461	1,754,800	157.33	112.184
14	49.79	1.00	49.79	2.516	1,809,000	125.27	90.070
15	38.77	1.00	38.77	2.552	1,845,800	98.94	71.562
Total	4,384.29		2,297.87			3,605.65	2,234.000

Mean weight per mature female fish = (3,606 lbs ÷ 2,298 mature females) = 1.57 lbs (~38.6 cm fish)
Mean fecundity (unfished stock) = 972,205 eggs per female spawner

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 1996 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	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
1995	2,571	47	2,470 - 2,671	222.9	1,133.8
1996	1,388	78	1,222 - 1,554	53.9	544.7

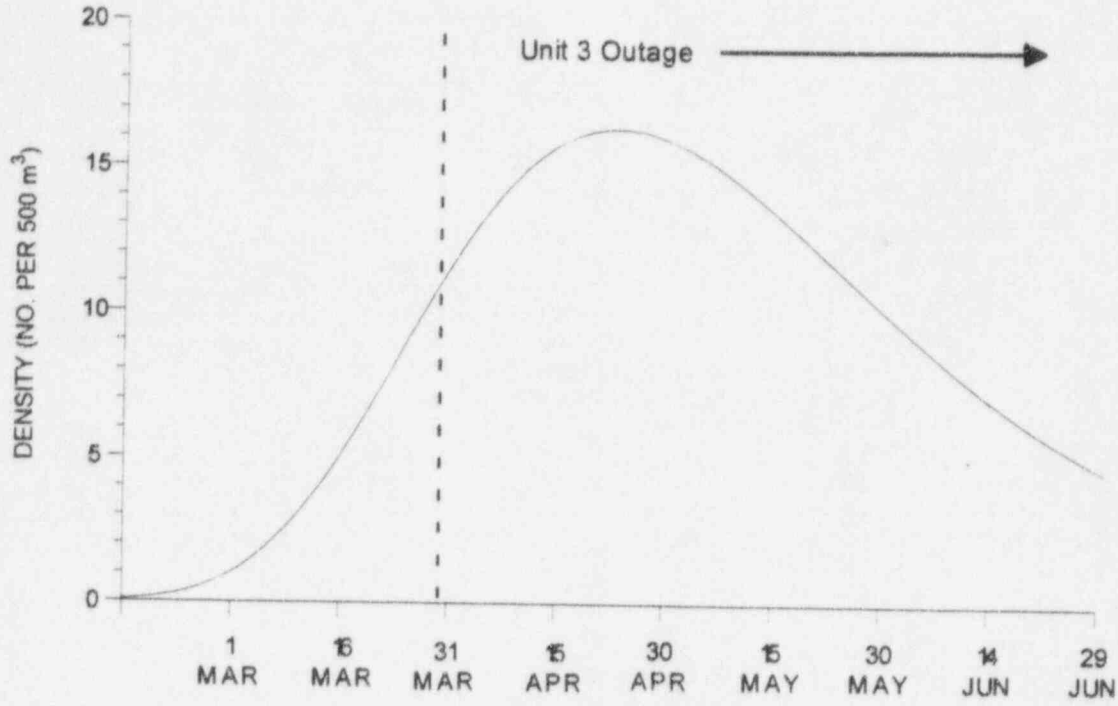


Fig. 39. Abundance curve of entrained larval winter flounder in relation to the period of MNPS unit outages during 199[illegible]. Unit 1 (shutdown on November 4, 1995) and Unit 2 (February 20, 1996) did not operate during the entire larval winter flounder season.

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 positively correlated with two abundance indices of juvenile winter flounder, which were the median CPUE of age-0 fish taken in both early and late summer at station LR (Table 35). Entrainment estimates were also significantly negatively correlated with a calculated apparent larval survival rate (defined as the Δ-mean CPUE of

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	0.6484[b] 0.0121 * 14	0.6659 0.0093 ** 14	0.4349 0.0553 NS 20	-0.2912 0.2129 NS 20	-0.5386 0.0143 * 20

[a] Indices used correspond to those given on Tables 26, 27, and 34, except for the apparent survival rate, which is the Δ-mean CPUE of age-1 winter flounder taken in the Niantic River during adult population surveys divided by the α abundance index of 7 mm and larger larvae at the MNPS discharge (entrainment station EN).

[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).

age-1 winter flounder taken in the Niantic River during the adult population surveys divided by the α abundance index of winter flounder larvae 7 mm and larger taken at the MNPS discharge for the same year-class). Although significant, the form of the relationships between the entrainment estimates with age-0 abundance and the apparent survival rate was not obvious (Fig. 40). 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 abundance of Stage 4 larvae or the age-0 indices (Table 29). The significant correlation coefficients found between entrainment and age-0 abundance were positive, implying no apparent entrainment effect. However, 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, which are lacking (Table 30).

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 Bay from LIS. Other stocks are known to spawn both to the east and west of the bay and results from tidal studies also indicate 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 conducted from 1984 through 1996; 11 of these years (1986-96) occurred during three-unit operation. The results for each 5-day period of the 1996 larval winter flounder season are provided as an example of these calculations (Table 36). Results for other years were given in NUSCO (1993, 1994a, 1995a, 1996).

During the 1996 larval season, the sign of the term *5-day change* (i.e., the difference in the number of larvae in Niantic Bay at the beginning and end of each 5-day period) shifted from positive to negative when the abundance of larvae in the bay began to decline on April 20 (Table 36). A negative *Source/Sink* term indicated a net loss of larvae from Niantic Bay during the first part of the larval season,

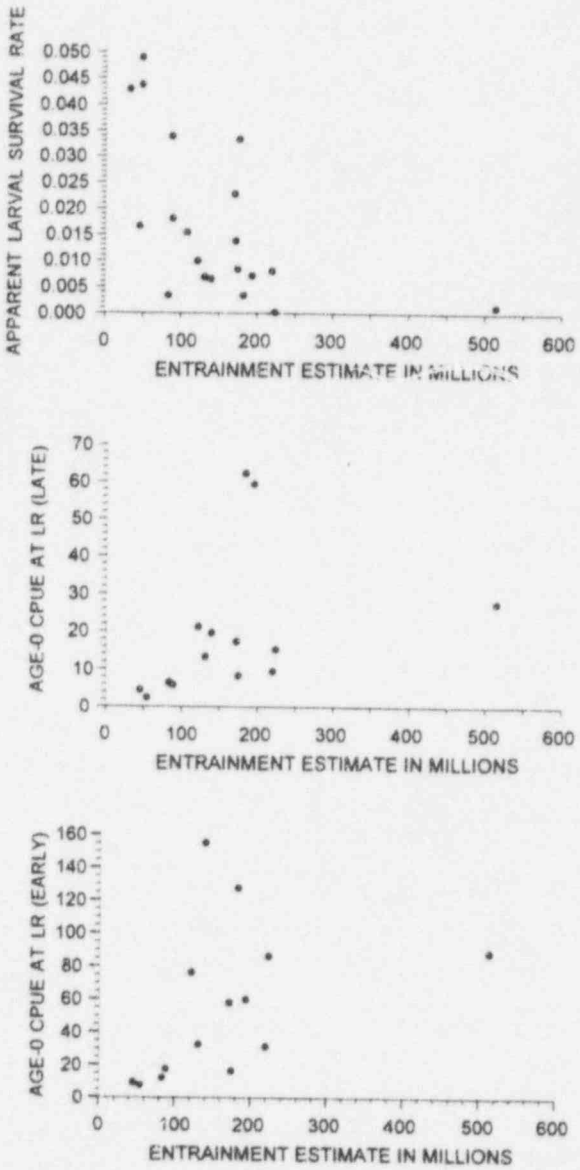


Fig. 40. Comparison between the annual entrainment estimate of winter flounder larvae at MNPS and the apparent survival rate of age-1 winter flounder (age-1 Δ-mean CPUE in the Niantic River during the adult winter flounder survey divided by the α parameter for larvae ≥7 mm at EN; 1976-95 year-classes), and the CPUE of age-0 winter flounder taken during both early and late summer (1983-96 year-classes).

but beginning with the 5-day period starting on April 5, the *Source or Sink* term became positive. This indicated that larvae from other sources (i.e., LIS) were required to support the change in larval abundance and balance the equation. Timing of this

TABLE 36. Results of mass-balance calculations for each 5-day period in 1996.

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]	0.1	0.0	0.6	0.0	-0.5
2-20	0.0	0.2	0.0	1.5	6.7	5.4
2-25	0.1	0.4	0.0	5.5	6.8	1.8
3-01	0.4	0.7	0.1	14.5	7.0	-6.3
3-06	0.8	1.3	0.3	26.6	7.6	-16.7
3-11	1.4	2.1	0.7	37.1	8.7	-24.1
3-16	2.1	3.1	1.3	42.2	10.6	-25.2
3-21	2.6	4.3	2.7	41.8	13.1	-19.0
3-26	2.8	5.8	3.1	37.3	16.0	-9.7
3-31	2.6	3.9	4.6	31.1	19.0	-1.0
4-05	2.1	3.8	5.1	24.7	21.6	7.9
4-10	1.4	3.7	5.6	18.9	23.6	15.3
4-15	0.6	3.3	5.1	14.2	24.8	19.6
4-20	-0.1	3.6	3.9	10.5	25.1	22.0
4-25	-0.6	4.3	3.6	7.7	24.8	24.3
4-30	-1.0	3.9	3.2	5.7	24.0	24.4
5-05	-1.3	1.3	2.9	4.2	22.7	21.4
5-10	-1.4	0.9	2.6	3.1	21.2	20.2
5-15	-1.5	0.9	2.3	2.3	19.7	19.1
5-20	-1.4	0.5	2.1	1.8	18.1	17.5
5-25	-1.3	0.5	1.7	1.4	16.6	16.0
5-30	-1.2	0.9	1.5	1.1	15.2	15.3
6-04	-1.1	0.7	1.3	1.0	14.0	13.9
6-09	-0.9	1.0	1.1	0.8	12.9	13.1
6-14	-0.8	1.1	0.9	0.7	11.9	12.3
6-19	-0.7	1.0	0.7	0.7	11.1	11.4
6-24	-0.6	0.7	0.6	0.6	10.3	10.5

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

change in the *Source or Sink* term was similar to previous years (NUSCO 1993, 1995a, 1996), except for an earlier date of February 25 in 1993 (NUSCO 1994a). Considerably fewer larvae were entrained (*Ent*) than were imported from LIS (i.e., positive *Source or Sink*), starting in early April. Also in 1996, the weekly estimates of *Ent* were comparable or smaller than losses due to natural mortality (*Mort*) and until early May, were considerably smaller than the number flushed from the river into the bay (*FromNR*).

During each 5-day period of the season, the proportion of entrained larvae from the Niantic River was estimated from the 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 the same 5-day period to estimate the number entrained from the Niantic River. During any 5-day period 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 during that time were assumed to have originated from the Niantic River. This estimate was conservative because the results of a dye study and larval dispersal modeling (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 then determined by summing over all 5-day periods. In 1996, the estimated number of 31.6 million larvae entrained that were from the river was the second smallest of the 13-year time-series (Table 37). However, the percent entrainment attributed to the Niantic River in 1996 was 58.6%, by far the largest estimate, as in 1984-95 approximately 14 to 38% of winter flounder larvae originated from the

TABLE 37. Estimates of the 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 from 1984 through 1996.

Year	Total entrainment ($X 10^6$)	Niantic River larval entrainment ($X 10^6$)	% 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	52.0	28.6
1995	222.9	80.4	36.1
1996	53.9	31.6	58.6

Niantic River, with the remainder coming from other sources. The large value for 1996 was a result of the conservative nature of the mass-balance calculation in that all larvae were assigned to the Niantic River unless other factors in the model could have accounted for them. The model was also conceptualized for normal MNPS operating conditions and under the unusually low flow at MNPS in 1996, the model appeared to overestimate the contribution of larvae from the river. This led to an unrealistically high estimate of Niantic River larval entrainment that occurred concurrently with the second lowest estimate of larval entrainment.

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 total of each larval stage entrained from either source was estimated by summing over all 5-day periods (Table 38). In most years, Stage 3 was the predominant stage entrained whose source was attributed to the Niantic River. However, in 1996 developmental stages 1 and 2 predominated entrained larvae attributed to the Niantic River. Consistent with previous years, older Stage 3 and 4 larvae entrained were determined to originate mostly from other sources.

The interpretation of mass-balance calculation results has been substantiated by results from ongoing sampling and during several special studies. Some of the larger larvae from other areas may enter the Niantic River during a flood tide and caused the increased frequency noted in larger size-classes during some years. In special bay-wide sampling in April and May of 1991 (NUSCO 1992a), 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 tide than during 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 different larval stages from the river were compared to annual abundance estimates for each stage in the Niantic River. The latter were computed by applying mortality coefficients to each stage of early life history, beginning with total annual egg production (Table 13), which differed somewhat from estimates reported in previous years (see Spawning Stock Size and Egg Production). This process allowed the

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 1996.

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	2.9	12.8	29.9	6.4
	Other	2.7	25.5	84.5	17.4
1995	Niantic River	0.6	7.1	57.5	15.3
	Other	1.1	14.1	109.0	18.2
1996	Niantic River	13.7	13.2	4.3	0.3
	Other	1.7	4.3	12.0	4.1

determination of percent production loss of larvae 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, which was verified by sampling this year. As noted above, 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 estimates of Stage 1 larvae at station C were used to determine 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 in 1996 was 25.7%,

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 from 1984 through 1996. Numbers of larvae from the Niantic River were based on the most recent mass-balance calculations.

Year	Stage of development	Niantic River abundance[a] ($X 10^6$)	Actual MNPS operating conditions: Entrainment from the Niantic River[b] ($X 10^6$)	Actual MNPS operating conditions: % of the production[c]	Projected full[d] MNPS three-unit operating conditions: Entrainment from the Niantic River ($X 10^6$)	Projected full[d] MNPS three-unit operating conditions: % of the production[c]
1984	Stage 1	2502	10.3	0.4	22.6	0.9
	Stage 2	599	15.4	2.6	32.7	5.5
	Stage 3	294	14.4	4.9	34.9	11.9
	Stage 4	205	3.2	1.6	9.0	4.4
	Total		43.3	**9.4**	99.2	**22.6**
1985	Stage 1	2823	15.6	0.6	44.2	1.6
	Stage 2	676	17.9	2.6	44.7	6.6
	Stage 3	332	7.1	2.1	15.2	4.6
	Stage 4	232	0.4	0.2	1.0	0.4
	Total		41.0	**5.5**	105.1	**13.2**
1986	Stage 1	2179	11.6	0.5	14.4	0.7
	Stage 2	612	7.7	1.3	8.5	1.4
	Stage 3	319	15.9	5.0	15.7	4.9
	Stage 4	223	4.5	2.0	5.1	2.3
	Total		39.7	**8.8**	43.7	**9.3**
1987	Stage 1	2653	34.4	1.3	39.8	1.5
	Stage 2	745	15.6	2.1	18.1	2.4
	Stage 3	389	24.5	6.3	25.4	6.5
	Stage 4	271	1.9	0.7	2.0	0.7
	Total		76.4	**10.4**	85.3	**11.2**
1988	Stage 1	4326	83.7	1.9	92.1	2.1
	Stage 2	647	9.8	1.5	10.3	1.6
	Stage 3	233	25.3	10.8	27.1	11.6
	Stage 4	168	1.6	1.0	1.7	1.0
	Total		120.4	**15.2**	131.2	**16.3**
1989	Stage 1	3575	66.5	1.9	84.3	2.4
	Stage 2	499	11.5	2.3	14.3	2.9
	Stage 3	164	19.7	12.0	24.1	14.7
	Stage 4	110	0.5	0.5	0.7	0.6
	Total		98.2	**16.6**	123.4	**20.5**
1990	Stage 1	1850	33.2	1.8	36.7	2.0
	Stage 2	760	6.4	0.8	7.6	1.0
	Stage 3	209	28.5	13.6	32.4	15.5
	Stage 4	180	3.8	2.1	4.3	2.4
	Total		71.9	**18.4**	81.0	**20.9**
1991	Stage 1	3192	8.0	0.3	13.0	0.4
	Stage 2	2227	3.7	0.2	5.2	0.2
	Stage 3	677	27.5	4.1	36.2	5.3
	Stage 4	549	4.9	0.9	6.4	1.2
	Total		44.1	**5.4**	60.8	**7.1**

TABLE 39. (cont.).

Year	Stage of development	Niantic River abundance[a] (X 10^6)	Actual MNPS operating conditions: Entrainment from the Niantic River[b] (X 10^6)	Actual MNPS operating conditions: % of the production[c]	Projected full[d] MNPS three-unit operating conditions: Entrainment from the Niantic River (X 10^6)	Projected full[d] MNPS three-unit operating conditions: % of the production
1992	Stage 1	1952	23.0	1.2	28.6	1.5
	Stage 2	818	10.4	1.3	11.8	1.4
	Stage 3	301	57.2	19.0	64.4	21.4
	Stage 4	241	9.0	3.7	10.5	4.3
	Total		99.6	**25.2**	115.3	**28.7**
1993	Stage 1	1116	11.7	1.0	13.3	1.2
	Stage 2	577	1.2	0.2	1.3	0.2
	Stage 3	104	4.1	3.9	4.2	4.0
	Stage 4	73	0.5	0.7	0.5	0.7
	Total		17.5	**5.9**	19.3	**6.1**
1994	Stage 1	1646	27.4	1.7	36.1	2.2
	Stage 2	903	12.8	1.4	16.2	1.8
	Stage 3	453	29.9	6.6	38.6	8.5
	Stage 4	394	6.4	1.6	7.5	1.9
	Total		76.5	**11.3**	98.4	**14.4**
1995	Stage 1	846	39.6	4.7	44.2	5.2
	Stage 2	534	7.1	1.3	10.1	1.9
	Stage 3	222	57.5	26.0	84.3	38.1
	Stage 4	148	15.3	10.3	23.6	15.9
	Total		119.5	**42.3**	162.2	**61.1**
1996	Stage 1	410	24.3	5.9	39.0	9.5
	Stage 2	145	13.2	9.1	25.5	17.6
	Stage 3	45	4.3	9.7	12.2	27.4
	Stage 4	30	0.3	1.0	1.9	6.2
	Total		42.1	**25.7**	78.6	**60.7**
Geometric mean				**12.7**		**17.5**

[a] Abundance estimates for 1984-89 were from Crecco and Howell (1990), for 1990 from V. Crecco (CT DEP, Old Lyme, CT, pers. comm.), and those for 1991-96 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] Values changed from those reported in NUSCO (1996), based on revised estimates of total egg production (see Table 13).

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

which was the second largest conditional mortality rate (ENT) of the 13-year period. This large production loss estimate was mostly due to the entrainment of Stage 2 and 3 larvae attributed to the Niantic River stock, which, as discussed above, is likely an overestimate because of the conservative nature of the mass-balance calculation. Values of ENT were about 5% higher than reported in NUSCO (1996) due to a decrease in estimation of annual egg production. The geometric mean of the time-series was 12.7%.

The entrainment estimate for 1996 was the second lowest since three-unit operation began in 1986, which was related to moderate larval abundance and lowest entrained seawater volume since 1984 (Table 34). Because annual egg production in the river during both 1995 and 1996 were the lowest of all the estimates (Table 13), high larval abundance suggested substantial survival that resulted from several factors. Egg hatchability for these years was apparently higher than usual with more Stage 1 larvae than would have been expected (Fig. 17). The

larval mortality rate in the river during 1995 was among the lowest found, but in 1996 was greater than the long-term average (Table 19). The larval recruitment indices at station EN (Fig. 24) for both years were the highest calculated. Larval development is directly related to growth rate and shorter larval periods likely result in better survival (Houde 1987). For example, during 1995, larval growth and development were among the fastest in comparison to previous years, as indicated by greater annual growth rates in the bay and river (Tables 17 and 18). This resulted in a large mean larval length at station EN during the first part of April and a relatively early date of peak abundance (Figs. 21 and 22). Both growth and development appeared to be related to the warmer than average water temperature in 1995 during the larval season (Table 6), with high larval survival apparently resulting in large numbers of Stage 3 and 4 larvae available for entrainment. The relatively large entrainment estimates in comparison to low egg production resulted in a high larval production loss in 1995. Conversely, the mean temperature in spring of 1996 was the second coldest since 1976 (Table 7) with a late date for peak abundance (Table 16) and a longer exposure of larvae to predators, which probably reduced some of the advantages of relatively low larval densities.

The mass-balance calculations given above were based on actual daily condenser cooling-water volumes. To determine annual percentages of the Niantic River winter flounder production that would have been entrained since 1984 under simulated full (100% capacity) three-unit operation, the calculations were recomputed based on a maximum daily condenser cooling-water volume of 11.1 million $m^3 \cdot day^{-1}$ (Table 39). To have a longer time-series, three-unit operation was simulated to include 1984 and 1985, prior to Unit 3 start-up. Estimated reductions in year-class strength under three-unit operation ranged from 7.1 to 61.1% (geometric mean = 17.5%), with the highest values found in 1995 and 1996. The annual estimates of ENT 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 on-line, 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., ENT, the percent Niantic River annual larval production loss) assumed for each year of the simulation. Values of ENT during 1984-96 were based on known rates of MNPS cooling-water flow (Table 5) and calculated entrainment of Niantic River winter flounder larvae as derived from the mass-balance calculations discussed above (Table 39). Larval losses for 1971-83 were simulated by modifying a randomly chosen value of ENT by known condenser cooling-water flows at MNPS for each of those years. Similarly, entrainment rates for 1997 through 2025, which also depended upon a unit retirement schedule (Table 1), were estimated by randomly selecting both records of cooling-water flows for each unit during 1974-96 (Table 5) and the historic time-series values of ENT for full three-unit operation (Table 39). The flows were used to adjust the values of ENT to simulate year-to-year variation in cooling-water use during the larval winter flounder season. In this simulation, MNPS units were assumed to operate during a larval winter flounder season in the future as they had in the past. However, neither the estimate of ENT nor unit-specific cooling-water flows for 1996 were used in the random selection process because the simultaneous shutdown of all three units for nearly all of the larval winter flounder season resulted in atypically low flows (Table 5) that in all likelihood will not reoccur after 1997 as well as an overly conservative (i.e., high) estimate of ENT. All values of ENT calculated or selected for use in the SPDM simulation presented in this particular report are given in Table 40.

A combined mortality of fishing (F) and impingement (IMP) was used in the simulations only during 1971-2025, years corresponding to actual or expected MNPS operation. Expected changes in the values of F over time were determined after consultation with DEP Marine Fisheries (P. Howell, CT DEP, Old Lyme, CT, pers. comm.) and reflected recent estimates of mortality and changes in regulations designed to considerably reduce F in the future (Table 2). Nominal fishing mortality rates were initially set at F = 0.40, remained unchanged through the 1960s, increased abruptly to 0.50 in

TABLE 40. Schedule of conditional entrainment (ENT values), fishing (F) mortalities with adjustments for impingement (IMP), and fishing discard mortalites as implemented in the 1996 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	0.036	0.240	0.400	0.400
1	1961	0.0	0.40	0.036	0.240	0.400	0.400
2	1962	0.0	0.40	0.036	0.240	0.400	0.400
3	1963	0.0	0.40	0.036	0.240	0.400	0.400
4	1964	0.0	0.40	0.036	0.240	0.400	0.400
5	1965	0.0	0.40	0.036	0.240	0.400	0.400
6	1966	0.0	0.40	0.036	0.240	0.400	0.400
7	1967	0.0	0.40	0.036	0.240	0.400	0.400
8	1968	0.0	0.40	0.036	0.240	0.400	0.400
9	1969	0.0	0.40	0.036	0.240	0.400	0.400
10	1970	0.0	0.40	0.036	0.240	0.400	0.400
11	1971	0.1530 X ENT = 2.122	0.51	0.045	0.300	0.500	0.500
12	1972	0.2262 X ENT = 1.606	0.51	0.045	0.300	0.500	0.500
13	1973	0.0767 X ENT = 1.250	0.51	0.045	0.300	0.500	0.500
14	1974	0.1895 X ENT = 1.423	0.51	0.045	0.300	0.500	0.500
15	1975	0.2262 X ENT = 2.104	0.61	0.054	0.360	0.600	0.600
16	1976	0.4421 X ENT = 3.139	0.61	0.054	0.360	0.600	0.600
17	1977	0.4232 X ENT = 2.582	0.61	0.054	0.360	0.600	0.600
18	1978	0.3018 X ENT = 4.346	0.61	0.054	0.360	0.600	0.600
19	1979	0.3133 X ENT = 4.136	0.61	0.054	0.360	0.600	0.600
20	1980	0.4810 X ENT = 9.861	0.61	0.054	0.360	0.600	0.600
21	1981	0.2873 X ENT 8.246	0.71	0.063	0.420	0.700	0.700
22	1982	0.4857 X ENT = 2.963	0.71	0.042	0.343	0.700	0.700
23	1983	0.4675 X ENT = 5.236	0.71	0.042	0.301	0.700	0.700
24	1984	9.4	0.90	0.053	0.383	0.890	0.890
25	1985	5.5	0.72	0.043	0.256	0.710	0.710
26	1986	8.8	0.72	0.043	0.256	0.710	0.710
27	1987	10.4	0.72	0.043	0.256	0.710	0.710
28	1988	15.2	0.95	0.056	0.254	0.912	0.940
29	1989	16.6	1.07	0.064	0.286	1.028	1.060
30	1990	18.4	1.34	0.080	0.239	1.290	1.330
31	1991	5.4	1.20	0.071	0.214	1.154	1.190
32	1992	25.2	1.27	0.076	0.227	1.222	1.260
33	1993	5.9	1.13	0.067	0.202	1.086	1.120
34	1994	11.3	1.31	0.078	0.234	1.131	1.248
35	1995	42.3	1.11	0.066	0.077	0.605	0.935
36	1996	25.7	0.78	0.046	0.054	0.246	0.608
37	1997	U1, U2, U3 flow X ENT = 8.323	0.81	0.048	0.056	0.256	0.632
38	1998	U1, U2, U3 flow X ENT = 10.905	0.81	0.048	0.056	0.256	0.632
39	1999	U1, U2, U3 flow X ENT = 5.623	0.81	0.048	0.056	0.256	0.632
40	2000	U1, U2, U3 flow X ENT = 7.532	0.71	0.042	0.049	0.224	0.553
41	2001	U1, U2, U3 flow X ENT = 14.938	0.71	0.042	0.049	0.224	0.553
42	2002	U1, U2, U3 flow X ENT = 18.361	0.71	0.042	0.049	0.224	0.553
43	2003	U1, U2, U3 flow X ENT = 16.354	0.71	0.042	0.049	0.224	0.553
44	2004	U1, U2, U3 flow X ENT = 14.228	0.71	0.042	0.049	0.224	0.553
45	2005	U1, U2, U3 flow X ENT = 9.355	0.71	0.042	0.049	0.224	0.553
46	2006	U1, U2, U3 flow X ENT = 5.623	0.61	0.036	0.042	0.192	0.474
47	2007	U1, U2, U3 flow X ENT = 17.541	0.61	0.036	0.042	0.192	0.474
48	2008	U1, U2, U3 flow X ENT = 23.228	0.61	0.036	0.042	0.192	0.474
49	2009	U1, U2, U3 flow X ENT = 15.072	0.61	0.036	0.042	0.192	0.474
50	2010	U1, U2, U3 flow X ENT = 10.799	0.61	0.036	0.042	0.192	0.474
51	2011	U2, U3 flow X ENT = 7.687	0.61	0.036	0.042	0.192	0.474
52	2012	U2, U3 flow X ENT = 14.083	0.61	0.036	0.042	0.192	0.474
53	2013	U2, U3 flow X ENT = 12.774	0.61	0.036	0.042	0.192	0.474

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
54	2014	U2, U3 flow X ENT = 4.607	0.61	0.036	0.042	0.192	0.474
55	2015	U2, U3 flow X ENT = 13.625	0.61	0.036	0.042	0.192	0.474
56	2016	U3 flow X ENT = 2.819	0.61	0.036	0.042	0.192	0.474
57	2017	U3 flow X ENT = 6.500	0.61	0.036	0.042	0.192	0.474
58	2018	U3 flow X ENT = 13.115	0.61	0.036	0.042	0.192	0.474
59	2019	U3 flow X ENT = 6.277	0.61	0.036	0.042	0.192	0.474
60	2020	U3 flow X ENT = 24.629	0.61	0.036	0.042	0.192	0.474
61	2021	U3 flow X ENT = 4.179	0.61	0.036	0.042	0.192	0.474
62	2022	U3 flow X ENT = 2.819	0.61	0.036	0.042	0.192	0.474
63	2023	U3 flow X ENT = 11.396	0.61	0.036	0.042	0.192	0.474
64	2024	U3 flow X ENT = 6.277	0.61	0.036	0.042	0.192	0.474
65	2025	U3 flow X ENT = 2.787	0.61	0.036	0.042	0.192	0.474
66	2026	0.0	0.60	0.036	0.042	0.192	0.474
67	2027	0.0	0.60	0.036	0.042	0.192	0.474
68	2028	0.0	0.60	0.036	0.042	0.192	0.474
69	2029	0.0	0.60	0.036	0.042	0.192	0.474
70	2030	0.0	0.60	0.036	0.042	0.192	0.474
71	2031	0.0	0.60	0.036	0.042	0.192	0.474
72	2032	0.0	0.60	0.036	0.042	0.192	0.474
73	2033	0.0	0.60	0.036	0.042	0.192	0.474
74	2034	0.0	0.60	0.036	0.042	0.192	0.474
75	2035	0.0	0.60	0.036	0.042	0.192	0.474
76	2036	0.0	0.60	0.036	0.042	0.192	0.474
77	2037	0.0	0.60	0.036	0.042	0.192	0.474
78	2038	0.0	0.60	0.036	0.042	0.192	0.474
79	2039	0.0	0.60	0.036	0.042	0.192	0.474
80	2040	0.0	0.60	0.036	0.042	0.192	0.474
81	2041	0.0	0.60	0.036	0.042	0.192	0.474
82	2042	0.0	0.60	0.036	0.042	0.192	0.474
83	2043	0.0	0.60	0.036	0.042	0.192	0.474
84	2044	0.0	0.60	0.036	0.042	0192	0.474
85	2045	0.0	0.60	0.036	0.042	0.192	0.474
86	2045	0.0	0.60	0.036	0.042	0.192	0.474
87	2047	0.0	0.60	0.036	0.042	0.192	0.474
88	2048	0.0	0.60	0.036	0.042	0.192	0.474
89	2049	0.0	0.60	0.036	0.042	0.192	0.474
90	2050	0.0	0.60	0.036	0.042	0.192	0.474
91	2051	0.0	0.60	0.036	0.042	0.192	0.474
92	2052	0.0	0.60	0.036	0.042	0.192	0.474
93	2053	0.0	0.60	0.036	0.042	0.192	0.474
94	2054	0.0	0.60	0.036	0.042	0.192	0.474
95	2055	0.0	0.60	0.036	0.042	0.192	0.474
96	2056	0.0	0.60	0.036	0.042	0.192	0.474
97	2057	0.0	0.60	0.036	0.042	0.192	0.474
98	2058	0.0	0.60	0.036	0.042	0.192	0.474
99	2059	0.0	0.60	0.036	0.042	0.192	0.474
100	2060	0.0	0.60	0.036	0.042	0.192	0.474

[a] ENT values for 1984-96 were estimates made under actual MNPS operating conditions as shown on Table 39. For 1971-83 and 1997-2025, ENT values were randomly selected from projected rates determined from mass-balance calculations for full three-unit operation during 1984-96 (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 (except for 1996) were used for 1997-2025. The values of ENT given in the table above were used in the SPDM simulations for this particular report only.

[b] F values were obtained from the DEP (P. Howell, 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).

1971, to 0.60 in 1975, and to 0.70 in 1981 (Table 40; Fig. 5). Fishing mortality increased more rapidly in subsequent years to a maximum of 1.33 in 1990. The values of F also included an additional mortality of 0.01 to account for average impingement losses (IMP) during the years of MNPS operation (NUSCO 1992a). As a result of already implemented or proposed regulatory changes to the fisheries in 1996 and thereafter, F was projected to decrease to 0.80 in the late 1990s, to 0.70 during 2000-05, stabilize at 0.60 in 2006, and remain unchanged throughout the rest of the simulation period.

Simulation results. Three stochastic time-series of female spawning stock sizes were generated during the three 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.

Baseline stock projections include fishing, but no power plant effects (Fig. 41A). Thus, this time-series was used as the reference against which the impacted stock projections were compared so that past and projected trends of Niantic River winter flounder abundance would be taken into account. Based on the age and size structure of an unfished female winter flounder stock at equilibrium, the unfished stock size used initially in the simulations was 113,415 lbs (value of P_{rep}), which was equivalent to 72,239 female spawners (Tables 4, 32, and 33). This initial stock size represented the maximum spawning potential (MSP) for the unfished Niantic River female spawning stock. The geometric mean estimate of MSP from the SPDM simulations was 113,360 lbs, which was remarkably similar to the deterministic estimate of P_{rep} (Table 32) used to initiate the model runs. By 1970, the stochastic mean size of the exploited stock under the starting nominal fishing rate of F = 0.40 was quickly reduced to 56,243 lbs. The simulated baseline (the solid line in Figs. 41A and 42) responded as expected to the high rates of fishing through the mid-1990s as the stock steadily declined to its lowest point of 12,880 lbs in 1993. Stock biomass increased only slightly to 14,405 lbs in 1994 and 14,108 lbs in 1995. The annual estimated biomass from 1992 through 1995 was only about half or less of the critical stock size (defined as a stock biomass equal to 25% of the MSP) of 28,340 lbs, shown as the dashed line in Figures 41A and B; this critical stock size will be discussed in greater detail below. Allowing for natural variation in the simulation, even the largest values of random stock sizes for 1991-93 were below 25% of MSP and the minimum value found in 1995 was only 6.5% of MSP. The simulation illustrated that the baseline population could fall (with varying probabilities) below the critical stock size at any time from 1975 through 1996. However, if reductions in F are realized 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 (Figure 41B), which is also shown as the dashed line in Figure 42. The impacted series corresponds to projections of the baseline stock, 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. 42). The lowest projected stock biomass (10,604 lbs) was again reached in 1993, whereas the greatest absolute decline relative to the baseline occurred in 2000 (a difference of 18,682 lbs), when the effects of reductions in F beginning in 1996 propagated through the spawning population. From this point, biomass of the impacted stock generally paraileled that of the baseline, except for several periods. Relatively large (ENT > 20) production loss estimates were selected for 2008 (23.228) and 2020 (24.629) in the SPDM simulation time-series (Table 40). These production losses were reflected by decreases in biomass found mostly in 2012-14 and 2023-25. Differences between the baseline and impacted projections were 6,756 to 7,843 and 4,484 to 5,787 lbs, respectively, for those years before beginning to recover under more moderate estimates of ENT. The large value of ENT chosen for 2020, however, had a smaller effect as only Unit 3 remained on-line by then and flow through MNPS had been reduced by about half.

As MNPS units were retired, impacted stock size began to approach that of the baseline. Impacted stock biomass was less than 1,700 lbs of the baseline in 2030 (5 years after the end of Unit 3 operation in 2025) and became virtually identical to it by 2033. These projections are only realistic for the fishing rates simulated, but actual winter flounder abundance

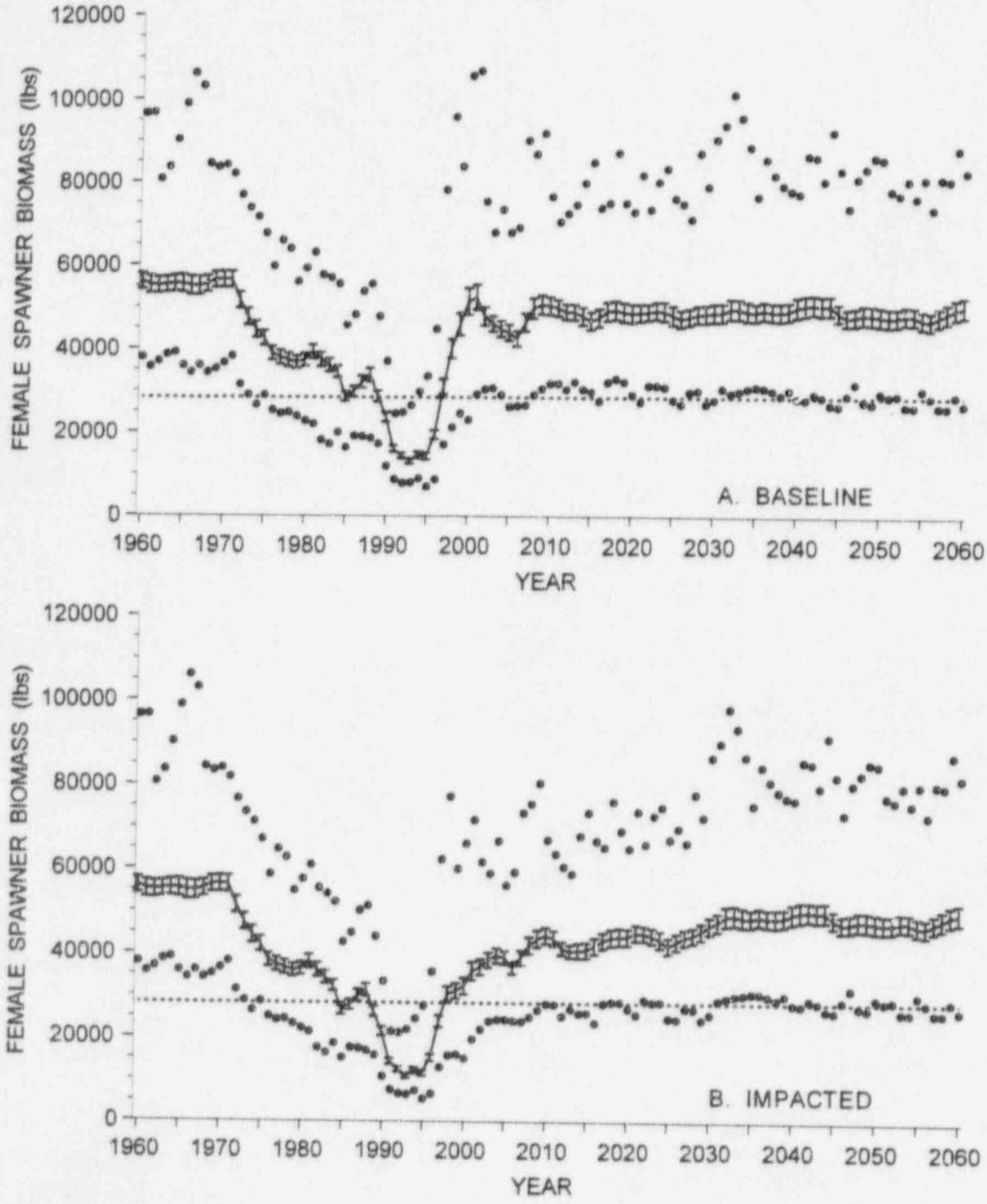


Fig. 41. The stochastic variability associated with projected Niantic River female winter flounder stocks expressed as biomass in lbs for: A. The baseline stock with simulated sport and commercial fishing rates, but no effects from MNPS operation, and B. The impacted stock with both fishing effects and 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 42. The symbols above and below each solid line correspond to the largest and smallest stocks among the 100 replicates generated each year. The horizontal dashed line represents the critical stock size (here, 28,340 lbs), defined in Howell et al. (1992) as stock biomass equal to 25% of the maximum spawning potential.

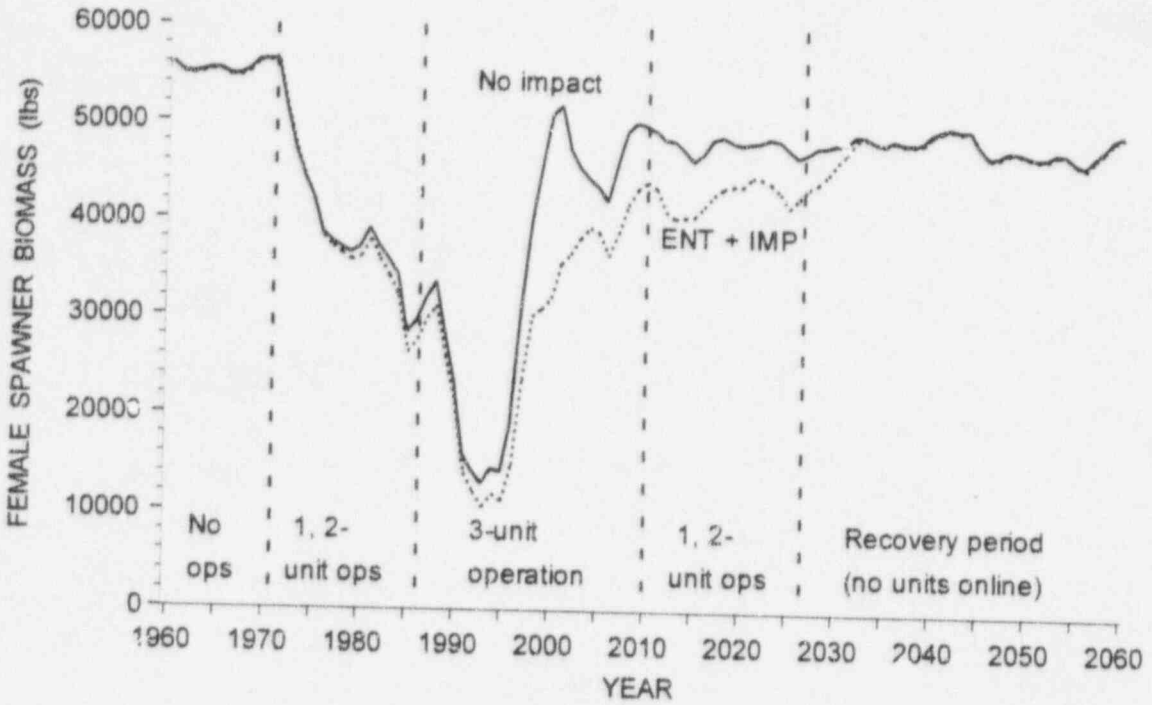


Fig. 42. Results of the SPDM simulation showing the combined effects of fishing and calculated larval entrainment rate and fish impingement (dashed line labeled "ENT + IMP") on the biomass in lbs of Niantic River female winter flounder spawning stock. Entrainment rates changed annually according to the 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.

could depart considerably from predictions if fishing rates and other simulated conditions are not matched by actual conditions. For example, should fishing rates remain high into the late 1990s, the difference between the lowest points in the baseline and impacted stock series would become wider and recovery would take longer, assuming that fishing would eventually decrease to projected levels.

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 individuals 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 fish. 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. 43). Most biomass reductions were attributed to fishing. However, as fishing mortality was reduced and stock biomass increased, reductions in winter flounder population size caused by larval entrainment at MNPS became larger relative to the baseline until MNPS units ceased operation.

Stock sizes projected for each simulation scenario at nine selected points in time are given in Table 41; losses relative to the theoretical unfished stock for each particular year are shown as percentages in this table. Stock sizes representing the 5th and 95th percentiles for the 100 Monte Carlo replicates generated for each year are also given. The theoretical unfished stock in each of the years shown varied little and averaged about 121,500 lbs. Prior to MNPS operation in 1970, the baseline and the impacted stocks were identical (56,993 lbs) and represented about 47% of the unfished stock. By

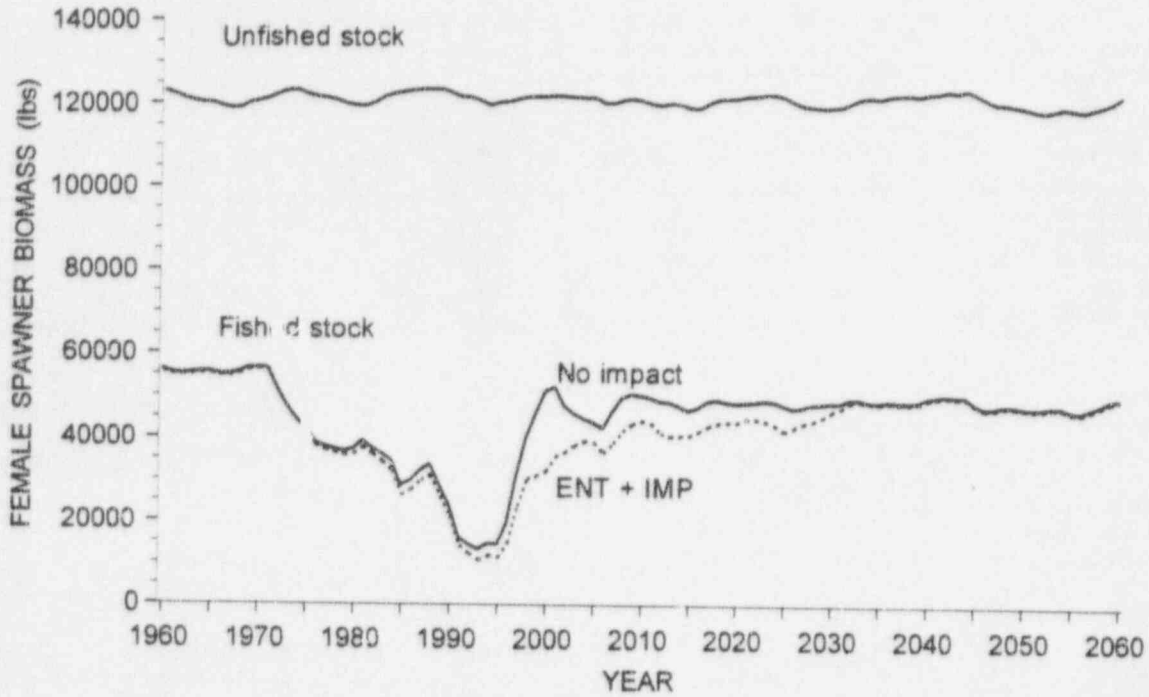


Fig. 43. Comparison of Niantic River winter flounder female stock biomass in lbs after effects of fishing (solid line labeled "No impact") and MNPS operation under calculated entrainment and impingement rates (dashed line labeled "ENT + IMP") with the theoretical (SRR-based estimate) unfished stock. All stock sizes are averages of 100 Monte Carlo replicates.

1990, winter flounder spawning stocks under full MNPS three-unit operation declined to about 38% of the 1970 size. However, this was mostly the result of increased fishing as the impacted stock was only 2,164 lbs less than the baseline. As noted previously, smallest stock sizes were predicted for 1993 as a result of high rates of exploitation during the early 1990s. In 1996, the baseline and impacted stocks increased 49% and 41%, respectively, relative to 1993, but biomass remained at only about 16% and 13% of the unfished stock. In following years, the baseline stock responded more rapidly to decreased fishing than the impacted stock, increasing to 40 to 43% of the unfished population after 1998. In contrast, the impacted stock remained at 27 to 40% of the unfished stock through the remaining years of three-unit operation. The gap between the two populations began to close in 2010 and, as noted above, became almost identical in 2033, after which the impacted stock could be considered fully recovered.

Probabilistic assessment of MNPS effects. The stochastic variability associated with stock projections for the baseline and impacted stocks (Fig. 41) 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, including stock sizes both smaller and larger than the mean. To assess effects of MNPS operation, the probability that the Niantic River female winter flounder spawning stock would fall below three selected reference sizes was determined directly from the frequency distribution of 100 replicates of selected annual stock sizes. 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). A stock that has been reduced to less than 25% of the MSP is considered overfished and its continued maintenance is questionable; spawner abundance may decline to even lower levels. Alternatively, conservative fishing rates that preserve 40% of MSP allow for sustainability of stocks and maximize yield to fisheries in the long term. According to the management plan for eastern LIS, values of F ranging from 0.37 to 0.68 would be necessary to achieve maximum yield, depending

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 41 and 42). Expected mean stock sizes are geometric means of 100 Monte Carlo replicates and the fifth and ninety-fifth percentiles of stock sizes of the 100 replicates of each year are given.

Type of population simulated	1970	1980	1990	1996	2000	2010	2020	2030	2040
Theoretical unfished stock[a]									
Geometric mean	121,067	120,051	123,110	120,929	122,600	121,375	121,927	119,712	122,755
5th percentile	103,757	96,231	101,968	99,563	95,159	99,710	99,271	98,661	98,845
95th percentile	141,788	146,014	152,778	148,386	157,449	156,959	150,488	145,411	147,939
Baseline[b]									
Geometric mean	56,993	37,656	24,042	19,828	52,980	50,746	48,986	49,147	50,544
5th percentile	43,807	26,050	16,038	13,073	31,248	36,467	33,737	34,911	32,935
95th percentile	76,133	49,136	33,909	28,303	85,365	68,323	69,967	69,162	69,290
Baseline mean as a % of the unfished stock	47.1%	31.4%	19.5%	16.4%	43.2%	41.8%	40.2%	41.1%	41.2%
Impact (ENT + IMP)[c]									
Geometric mean	56,993	36,834	21,878	15,449	33,439	45,043	44,735	47,468	50,539
5th percentile	43,807	25,417	14,533	10,432	20,232	31,725	30,086	34,056	32,940
95th percentile	76,133	47,919	30,825	22,494	54,072	61,528	62,857	67,072	69,170
Impacted mean as a % of the baseline stock	100%	97.8%	91.0%	77.9%	63.1%	88.8%	91.3%	96.6%	100%
Impacted mean as a % of the unfished stock	47.1%	30.7%	17.8%	12.8%	27.3%	37.1%	36.7%	40.0%	41.2%

[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.

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.

In 1970, both the baseline and impacted stocks were likely ($p = 0.92$) larger than 40% of MSP (Table 42). However, by 1980 both stocks were probably ($p = 0.85, 0.86$, respectively) smaller than 40% of MSP and had increased probabilities (0.32, 0.38) of falling below 30% of MSP. In 1990, the stocks were almost certainly ($p = 0.94$) less than 30% of MSP and likely ($p = 0.81, 0.87$) less than 25% of MSP. At the lowest points of both stock projections in the mid-1990s, all replicates were below 25% of MSP ($p = 0.99, 1.00$). Relatively large reductions in fishing rates in the late 1990s allowed for an increase in spawning biomass of the baseline stock above 25% of MSP to more optimal stock sizes by 2000. However, the impacted stock still had a relatively high probability of being less than 25% and 30% of MSP (0.35, 0.62, respectively). By 2010, spawning biomass of the impacted stock was likely ($p = 0.91$) greater than 30% of MS and had a probability of 0.42 of being greater than 40% of MSP. In 2020 and 2030, both the baseline and impacted stocks were most likely greater than 30% of MSP, although the chances of being greater than 40% of MSP did not improve substantially. SPDM output shows the Niantic River stock stabilizing at a biomass of about 47 to 49 thousand lbs following the shutdown of MNPS in 2025. 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 as in 2040 there still was a one in three chance of stock sizes being smaller than this reference level.

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 113,360 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	1996	2000	2010	2020	2030	2040
Baseline[b]	25% of MSP	0.00	0.08	0.81	0.94	0.01	0.00	0.00	0.01	0.01
Impacted[c]	25% of MSP	0.00	0.10	0.87	0.98	0.35	0.00	0.01	0.01	0.00
Baseline	30% of MSP	0.00	0.32	0.94	0.97	0.08	0.03	0.05	0.03	0.08
Impacted	30% of MSP	0.00	0.38	1.00	0.99	0.62	0.09	0.10	0.04	0.07
Baseline	40% of MSP	0.08	0.85	0.99	0.99	0.36	0.36	0.40	0.39	0.34
Impacted	40% of MSP	0.08	0.86	1.00	1.00	0.88	0.58	0.58	0.47	0.34

[a] Corresponds to reference stock sizes given in Howell et al. (1992) of 25%, 30%, and 40% of the MSP (28,340 lbs, 34,008 lbs, and 45,344 lbs, respectively).

[b] Fishing effects, but no MNPS impact.

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

Conclusions

Abundance of adult winter flounder spawners in the Niantic River has been depressed since 1992. The current size structure of the spawning population is heavily skewed towards larger fish and may be an indication of potential stock collapse. Nevertheless, despite projected low egg production, densities of larvae found in Niantic River and Bay remained relatively high, as did entrainment of larvae at MNPS. High larval densities were likely the consequence of both high egg hatchability and successful larval survival. Warm water temperatures during winter promotes relatively fast growth and development of larvae and probably enhances egg and larval survival. Conversely, cooler temperatures retard larval growth and development and likely increase mortality by exposing larvae to increased predation while part of the plankton. This is apparently contradictory to other findings, as stock and recruitment analyses suggest that largest year-classes are produced in cold years. However, this temperature effect may only operate during abnormally cold years (e.g., 1977-78) as spawning, hatching, and larval development periods are considerably lengthened, thereby reducing density-dependent risks of predation and starvation and also limiting predator activity. Further, very cold winters appear to also negatively affect the abundance of demersal predators (e.g., sevenspine bay shrimp) of newly metamorphosed settled juveniles, another life stage critically important to winter flounder year-class formation.

MNPS operations were considerably reduced this year, resulting in the lowest cooling-water flows in 15 years. However, because of the conservative assumptions of the mass-balance calculations used to estimate production loss of larvae from the Niantic River (i.e., the conditional entrainment mortality rate), the value for 1996 was unrealistically high. This was considered to be an overestimate because the mass-balance model was not developed for the abnormally low flow conditions encountered at MNPS during 1996. Nevertheless, relatively weak larval production and settlement of young in the Niantic River resulted in a poor winter flounder year-class produced during 1996.

The NUSCO stochastic population dynamics model (SPDM) was used to assess the long-term effects of MNPS operation (predominantly entrainment) on Niantic River winter flounder concurrently with relatively high rates of fishing mortality. Annual larval production loss estimates of 7.1 to 61.1%, calculated for projected full MNPS three-unit operation, were used in the SPDM, along with the annual cooling-water flow histories for each MNPS unit. Simulations illustrated that fishing alone reduced the stock to less than half of the unfished equilibrium spawner biomass (i.e., MSP or maximum

spawning potential) from about 113 thousand lbs in 1960 to 56 thousand lbs by 1970. Increases in fishing during the 1980s and 1990s further eroded the baseline stock projection (with fishing but no plant effects) to a level of about 13 thousand lbs during the mid-1990s. MNPS impact decreased the winter flounder female spawner stock by an additional several thousand lbs, with biomass reduced to as low as 9% of MSP. However, projected substantial reductions in fishing beginning in the late 1990s allowed the simulated stocks to recover quickly and MNPS impact became proportionately larger in terms of absolute loss of biomass as the population rebounded. Even so, the baseline and impacted biomass time-series became identical within a few years of the cessation of MNPS operation in 2025 and stock biomass leveled off at about 40% of the unfished population.

Recovery of the Niantic River winter flounder stock from present low levels depends upon fishing mortality being reduced as proposed. The SPDM outputs have projected this upturn each year since relatively optimistic forecasts of future fishing mortality rates were first presented in NUSCO (1993). However, these projections have not yet been matched by reality. The lack of a recovery despite efforts to reduce fishing and the inability to correlate abundance indices of juvenile winter flounder with adults from specific year-classes may be analogous to the example of Atlantic cod (*Gadus morhua*) in Canada, although certainly to a lesser degree. Meyers et al. (1997) suggested that an unaccounted fraction of mortality from increasing discards of juvenile cod occurred as fishing mortality of adult stocks increased. As cod populations declined, fishing mortality of adults not only kept increasing, mortality of younger fish also increased. This reduced recruitment exacerbated the situation until the commercial fishery had to be closed. A similar process, however, remains to be demonstrated for the winter flounder. Even though fishing rates remain high at present, the Niantic River population of winter flounder has remained resilient and very small adult spawning stocks in some recent years have produced relatively abundant year-classes of young fish. Continued efforts in reducing fishing mortality are necessary, however, to ensure a recovery and avoid a stock collapse.

References Cited

Al-Hossaini, M., Q. Liu, and T.J. Pitcher. 1989. Otolith microstructure indicating growth and mortality among plaice, *Pleuronectes platessa* L., post-larval sub-cohorts. J. Fish Biol. 35(Suppl. A):81-90.

Anderson, J.T. 1988. A review of size dependent survival during pre-recruit stages of fishes in relation to recruitment. J. Northw. Atl. Fish. Sci. 8:55-66.

Arnason, A.N., and K.H. Mills. 1981. Bias and loss of precision due to tag loss in Jolly-Seber estimates for mark-recapture experiments. Can. J. Fish. Aquat. Sci. 38:1077-1095.

Bailey, K.M., and E.D. Houde. 1989. Predation on eggs and larvae of marine fishes and the recruitment problem. Adv. Mar. Bio. 25:1-83.

Bailey, K.M., and R.S. Batty. 1984. Laboratory study of predation by *Aurelia aurita* on larvae of cod, flounder, plaice and herring: development and vulnerability to capture. Mar. Biol. (Berl.) 83:287-291.

Bannister, R.C.A., D. Harding, and S.J. Lockwood. 1974. Larval mortality and subsequent year-class strength in the plaice (*Pleuronectes platessa* L.). Pages 21-38 *in* J.H.S. Blaxter, ed. The early life history of fish. Springer-Verlag, New York.

Begon, M. 1979. Investigating animal abundance: capture-recapture for biologists. University Park Press, Baltimore. 97 pp.

Berghahn, R. 1986. Determining abundance, distribution, and mortality of 0-group plaice (*Pleuronectes platessa* L.) in the Wadden Sea. J. Appl. Ichthyol. 2: 11-22.

Berghahn, R. 1987. Effects of tidal migration on growth of 0-group plaice (*Pleuronectes platessa* L.) in the North Frisian Wadden Sea. Meeresforsch. 31:209-226. (Not seen, cited by Karakiri et al. 1989).

Bergman, M.J.N., H.W. van der Veer, and J.J. Zijlstra. 1988. Plaice nurseries: effects on recruitment. J. Fish Biol. 33 (Suppl. A): 210-218.

Bertram, D.F., R.C. Chambers, and W.C. Leggett. 1993. Negative correlations between larval and juvenile growth rates in winter flounder: implications of compensatory growth for variation in size-at-age. Mar. Ecol. Prog. Ser. 96:209-215.

Bertram, D.F., T.J. Miller, and W.C. Leggett. 1996. Individual variation in growth and development during the early life stage of winter flounder, *Pleuronectes americanus*. Fish. Bull., U.S. 95:1-10.

Bigelow, H.B., and W.C. Schroeder. 1953. Fishes of the Gulf of Maine. U.S. Fish Wildl. Serv. Bull. 53:1-577.

Bishop, J.A., and P.M. Sheppard. 1973. An evaluation of two capture-recapture models using the technique of computer simulation. Pages 235-253 *in* M.S. Bartlett and R.W. Hiorns, eds. The mathematical theory of the dynamics of biological populations. Academic Press, London.

Boudreau, P.R., and L.M. Dickie. 1989. Biological model of production based on physiological and ecological scaling of body size. Can. J. Fish. Aquat. Sci. 46:614-623.

Buckley, L.J. 1980. Changes in ribonucleic acid, deoxyribonucleic acid, and protein content during ontogenesis in winter flounder, *Pseudopleuronectes americanus*, and effect of starvation. Fish. Bull., U.S. 77:703-708.

Buckley, L.J. 1982. Effects of temperature on growth and biochemical composition of larval winter flounder *Pseudopleuronectes americanus*. Mar. Ecol. Prog. Ser. 8:181-186.

Buckley, L.J., A.S. Smigielski, T.A. Halavik, and G.C. Laurence. 1990. Effects of water temperature on size and biochemical composition of winter flounder *Pseudopleuronectes americanus* at hatching and feeding initiation. Fish. Bull., U.S. 88:419-428.

Buckley, L.J., A.S. Smigielski, T.A. Halavik, E.M. Caldarone, B.R. Burns, and G.C. Laurence. 1991. Winter flounder *Pseudopleuronectes americanus* reproductive success. II. Effects of spawning time and female size on size, composition and viability of eggs and larvae. Mar. Ecol. Prog. Ser. 74:125-135.

Burton, M.P., and D.R. Idler. 1984. The reproductive cycle in winter flounder *Pseudopleuronectes americanus* (Walbaum). Can. J. Zool. 62:2563-2567.

Carothers, A.D. 1973. The effects of unequal catchability on Jolly-Seber estimates. Biometrics 29:79-100.

Chambers, R.C., and W.C. Leggett. 1987. Size and age at metamorphosis in marine fishes: an analysis of laboratory-reared winter flounder (*Pseudopleuronectes americanus*) with a review of variation in other species. Can. J. Fish. Aquat. Sci. 44:1936-1947.

Chambers, R.C., W.C. Leggett, and J.A. Brown. 1988. Variation in and among early life history traits of laboratory-reared winter flounder *Pseudopleuronectes americanus*. Mar. Ecol. Prog. Ser. 47:1-15.

Christensen, S.W., and C.P. Goodyear. 1988. Testing the validity of stock-recruitment curve fits. Am. Fish. Soc. Monogr. 4:219-231.

Cormack, R.M. 1968. The statistics of mark-recapture methods. Oceanogr. Mar. Biol. Ann. Rev. 6:455-506.

Crawford, R.E. 1990. Winter flounder in Rhode Island coastal ponds. Rhode Island Sea Grant, Univ. of Rhode Island, Narragansett, RI. RIU-G-90-001. 24 pp.

Crawford, R.E., and C.G. Carey. 1985. Retention of winter flounder larvae within a Rhode Island salt pond. Estuaries 8:217-227.

Crecco, V.A., and P. Howell. 1990. Potential effects of current larval entrainment mortality from the Millstone Nuclear Power Station on the winter flounder, *Pseudopleuronectes americanus*, spawning population in the Niantic River. Connecticut Dept. Envir. Prot., Bu. Fish., Spec Pub. 37 pp.

Crecco, V.A., and T. Savoy. 1987. Fishery management plan for the American shad in the Connecticut River. Connecticut Dept. Envir. Prot., Bu. Fish., Spec. Pub. 140 pp.

Cushing, D.H. 1971. The dependence of recruitment on parent stock in different groups of fish. J. Cons. int. Explor. Mer 33:340-362.

Cushing, D.H. 1974. The possible density-dependence of larval mortality and adult mortality in fishes. Pages 103-111 *in* J.H.S. Blaxter, ed. The early life history of fish. Springer-Verlag, New York.

Cushing, D.H., and J.G.K. Harris. 1973. Stock and recruitment and the problem of density dependence. Rapp. P.-v. Réun. Cons. int. Explor. Mer 164:142-155.

Cushing, D.H., and J.W. Horwood. 1977. Development of a model of stock and recruitment. Pages 21-35 *in* J.H. Steele, ed. Fisheries mathematics. Academic Press, New York.

DeBlois, E.M., and W.C. Leggett. 1991. Functional response and potential impact of invertebrate predators on benthic fish eggs: analysis of the

Calliopius laeviusculus-capelin (*Mallotus villosus*) predator-prey system. Mar. Ecol. Prog. Ser. 69:205-216.

Dimou, N.K., and E.E. Adams. 1989. Application of a 2-D particle tracking model to simulate entrainment of winter flounder larvae at the Millstone Nuclear Power Station. Energy Laboratory Report No. MIT-EL 89-002. Massachusetts Institute of Technology, Cambridge, MA. 73 pp.

Draper, N., and H. Smith. 1981. Applied regression analysis. John Wiley and Sons, New York. 709 pp.

Dunn, R.S. 1970. Further evidence for a three-year oocyte maturation time in the winter flounder (*Pseudopleuronectes americanus*). J. Fish. Res. Board Can. 27:957-960.

Dunn, R.S., and A.V. Tyler. 1969. Aspects of the anatomy of the winter flounder ovary with hypotheses on oocyte maturation time. J. Fish. Res. Board Can. 26:1943-1947.

Garrod, D.J., and B.W. Jones. 1974. Stock and recruitment relationships in the Northeast Arctic cod stock and the implications for the management of the stock. J. Cons. int. Explor. Mer 36:35-41.

Gendron, L. 1989. Seasonal growth of the kelp *Laminaria longicruris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. Bot. Mar. 32:345-354.

Gibson, M.R. 1987. Preliminary assessment of winter flounder (*Pseudopleuronectes americanus*) stocks in Rhode Island waters. Rhode Island Div. Fish Wildl., Res. Ref. Doc. 87/7. 51 pp.

Gibson, M.R. 1989. Stock-recruitment relationships for winter flounder in the S. New England area and revised fishery reference points. Rhode Island Div. Fish Wildl., Res. Ref. Doc. 89/9. 10 pp + 5 fig.

Gilbert, R.O. 1973. Approximations of the bias in the Jolly-Seber capture-recapture model. Biometrics 29:501-526.

Goodyear, C.P. 1977. Assessing the impact of power plant mortality on the compensatory reserve of fish populations. Pages 186-195 *in* W. Van Winkle, ed. Proceedings of the conference on assessing the effects of power-plant-induced mortality on fish populations. Pergamon Press, New York.

Goodyear, C.P. 1980. Compensation in fish populations. Pages 253-280 *in* C.H. Hocutt and J.R. Stauffer, eds. Biological monitoring of fish. Lexington Books, Lexington, MA.

Goodyear, C.P., and S.W. Christensen. 1984. Bias-elimination in fish population models with stochastic variation in survival of the young. Trans. Am. Fish. Soc. 113:627-632.

Hess, K.W., M.P. Sissenwine, and S.B. Saila. 1975. Simulating the impact of entrainment of winter flounder larvae. Pages 1-30 *in* S.B. Saila, ed. Fisheries and energy production: a symposium. D.C. Heath and Co., Lexington, MA.

Hightower, J.E., and R.J. Gilbert. 1984. Using the Jolly-Seber model to estimate population size, mortality, and recruitment for a reservoir fish population. Trans. Am. Fish. Soc. 113:633-641.

Hjorleifsson, E. 1992. Abundance, condition, growth and mortality of winter flounder (*Pleuronectes americanus*, Walbaum) larvae in Narragansett Bay during spring of 1988. Ph.D. Dissertation. Univ. of Rhode Island, Narragansett, RI. 259 pp.

Hjort, J. 1926. Fluctuations in the year classes of important food fishes. J. Cons. int. Explor. Mer 1:5-38. (Not seen, cited by May 1974).

Hoenig, J.M., D.M. Heisey, W.D. Lawing, and H.D. Schupp. 1987. An indirect rapid methods approach to assessment. Can. J. Fish. Aquat. Sci. 44 (Suppl 2):324-338.

Houde, E.D. 1987. Fish early life history dynamics and recruitment variability. Am. Fish. Soc. Symposium 2:17-29.

Houde, E.D. 1989. Subtleties and episodes in the early life history of fishes. J. Fish Biol. 35(Suppl. A):29-38.

Hovenkamp, F., and J.IJ. Witte. 1991. Growth, otolith growth and RNA/DNA ratios of larval plaice *Pleuronectes platessa* in the North Sea 1987 to 1989. Mar. Ecol. Prog. Ser. 70:105-116.

Howe, A.B., and P.G. Coates. 1975. Winter flounder movements, growth and mortality off Massachusetts. Trans. Am. Fish. Soc. 104:13-29.

Howell, P., A. Howe, M. Gibson, and S. Ayvazian. 1992. Fishery management plan for inshore stocks of winter flounder. Fisheries management rep. no. 21 of the Atlantic States Marine Fisheries Commission. 138 pp.

Howell, W.H., and R. Langan. 1987. Commercial trawler discards of four flounder species in the Gulf of Maine. N. Am. J. Fish. Man. 7:6-17.

Howell, W.H., and R. Langan. 1992. Discarding of commercial groundfish species in the Gulf of

Maine shrimp fishery. N. Am. J. Fish. Man. 12:568-580.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with death and immigration stochastic model. Biometrika 52:225-247.

Karakiri, M., R. Berghahn, and H. von Westernhagen. 1989. Growth differences in 0-group plaice *Pleuronectes platessa* as revealed by otolith microstructure analysis. Mar. Ecol. Prog. Ser. 55:15-22.

Klein-MacPhee, G. 1978. Synopsis of biological data for the winter flounder, *Pseudopleuronectes americanus* (Walbaum). NOAA Tech. Rep. NMFS Circ. 414. 43 pp.

Kollmeyer, R.C. 1972. A study o' ne Niantic River estuary, Niantic, Connecti- nal report phases I and II, physical aspects o1 the Niantic River estuary. Rep. No. RDCGA 18. U.S. Coast Guard Academy, New London, CT. 78 pp.

Kuipers, B., B. MacCurrin, J.M. Miller, H.W. van der Veer, and J. IJ. Witte. 1992. Small trawls in juvenile flatfish research: their development and efficiency. Neth. J. Sea Res. 29:109-117.

Laurence, G.C. 1975. Laboratory growth and metabolism of the winter flounder *Pseudopleuronectes americanus* from hatching through metamorphosis at three temperatures. Mar. Biol. (Berl.) 32:223-229.

Laurence, G.C. 1977. A bioenergetic model for the analysis of feeding and survival potential of winter flounder, *Pseudopleuronectes americanus*, larvae during the period from hatching through metamorphosis. Fish. Bull., U.S. 75:529-546.

Lobell, M.J. 1939. A biological survey of the salt waters of Long Island, 1938. Report on certain fishes. Winter flounder (*Pseudopleuronectes americanus*). Suppl. 28th Ann. Rep., N.Y. Cons. Dep., Pt. 1:63-96.

Lockwood, S.J. 1972. The settlement, distribution and movements of 0-group plaice (*Pleuronectes platessa* L.) in Filey Bay, Yorkshire. J. Fish. Biol. 6:465-477.

Lockwood, S.J. 1980. Density-dependent mortality in 0-group plaice (*Pleuronectes platessa* L.) populations. J. Cons. int. Explor. Mer 39:148-153.

Longhurst, A. 1983. Benthic-pelagic coupling and export of organic carbon from a tropical Atlantic continental shelf, Sierra Leone. Est. Coast. Shelf Sci. 17:261-285.

Lorda, E.C., and V.A. Crecco. 1987. Stock-recruitment relationship and compensatory mortality of American shad in the Connecticut River. Am. Fish. Soc. Symposium 1:469-482.

Manly, B.J.F. 1971. A simulation of Jolly's method for analysing capture-recapture data. Biometrics 27:415-424.

Marine Research, Inc. 199-. Brayton Point investigations semi-annual report. January-June 1992. Submitted to New England Power Co.

Marshall, N., and S.D. Hicks. 1962. Drift of medusae and their distribution in relation to the hydrography of the Niantic River, Connecticut. Limnol. Oceanogr. 7:268-269.

May, R.C. 1974. Larval mortality in marine fishes and the critical period concept. Pages 3-20 *in* J.H.S. Blaxter, ed. The early life history of fish. Springer-Verlag, New York.

McConnaughey, R.A., and L.L. Conquest. 1993. Trawl survey estimation using a comparative approach based on lognormal theory. Fish. Bull., U.S. 91:107-118.

McCracken, F.D. 1963. Seasonal movements of the winter flounder, *Pseudopleuronectes americanus* (Walbaum), on the Atlantic coast. J. Fish. Res. Board Can. 20:551-586.

Miller, J.M., J.S. Burke, and G.R. Fitzhugh. 1991. Early life history patterns of Atlantic North American flatfish: likely (and unlikely) factors controlling recruitment. Neth. J. Sea Res. 27:261-275.

Morrison, J.A., I.R. Napier, and J.C. Gamble. 1991. Mass mortality of herring eggs associated with a sedimenting diatom bloom. ICES J. Mar. Sci. 48:237-245.

Myers, R.A., J.A. Hutchings, and N.J.Barrowman. 1997. Why do fish stocks collapse? The example of cod in Atlantic Canada. Ecol. Appl. 7:91-106.

Myers, R.A., and N.G. Cadigan. 1993a. Density-dependent juvenile mortality in marine demersal fish. Can. J. Fish. Aquat. Sci. 50:1576-1590.

Myers, R.A., and N.G. Cadigan. 1993b. Is juvenile mortality in marine demersal fish variable? Can. J. Fish. Aquat. Sci. 50:1591-1598.

Nichols, J.D., B.R. Noon, S.L. Stokes, and J.E. Hines. 1981. Remarks on the use of capture-recapture methodology in estimating avian population size. Studies in Avian Biol. 6:121-136. (Not seen, cited by Hightower and Gilbert 1984).

NUSCO (Northeast Utilities Service Company). 1976. Environmental assessment of the condenser cooling water intake structures (316b demonstration). Vol. 1 and 2.

NUSCO. 1981. Plankton ecology. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report, 1980. 40 pp.

NUSCO. 1985. Winter flounder studies. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report, 1984. 74 pp.

NUSCO. 1986a. Winter flounder studies. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report, 1985. 69 pp.

NUSCO. 1986b. The effectiveness of the Millstone Unit 1 sluiceway in returning impinged organisms to Long Island Sound. 18 pp.

NUSCO. 1987. Winter flounder studies. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of studies prior to Unit 3 operation. 151 pp.

NUSCO. 1988a. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to Letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP. 21 pp.

NUSCO. 1988b. Winter flounder studies. Pages 149-224 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-1987.

NUSCO. 1988c. The usage and estimation of DELTA means. Pages 311-320 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-1987.

NUSCO. 1989. Winter flounder studies. Pages 239-316 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1988.

NUSCO. 1990. Winter flounder studies. Pages 9-77 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1989.

NUSCO. 1991a. Evaluation of the larval winter flounder sampling program in the Niantic River. Enclosure to letter D04343 dated January 23, 1991 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP.

NUSCO. 1991b. Winter flounder studies. Pages 9-86 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1990.

NUSCO. 1992a. Winter flounder studies. Pages 7-109 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1991.

NUSCO. 1992b. Niantic Bay current studies. Pages 317-331 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1991.

NUSCO. 1993. Winter flounder studies. Pages 191-269 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1992

NUSCO. 1994a. Winter flounder studies. Pages 141-228 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station. Annual report 1993.

NUSCO. 1994b. Progress report on the MNPS fish return systems. Enclosure 1 to letter D08071 dated October 20, 1994 from D. Miller, NNECO, to T. Keeney, Commissioner, CT DEP.

NUSCO. 1995a. Winter flounder studies. Pages 9-92 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station. Annual report 1994.

NUSCO. 1995b. The 1994-95 winter-spring refueling outages at Millstone Nuclear Power Station Units 2 and 3 relative to the larval winter flounder season. Enclosure 1 to letter D08983 dated August 3, 1995 from D. Miller, NNECO, to T. Keeney, Commissioner, CT DEP.

NUSCO. 1996. Winter flounder studies. Pages 109-197 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station. Annual report 1995.

O'Brien, L., J. Burnett, and R.K. Mayo. 1993. Maturation of nineteen species of finfish off the northeast coast of the United States, 1985-1990. NOAA Tech. Rep. NMFS 113. 66 pp.

Olla, B.L., R. Wicklund, and S. Wilk. 1969. Behavior of winter flounder in a natural habitat. Trans. Am. Fish. Soc. 98:717-720.

Parrish, B.B. 1963. Some remarks on the selection processes in fishing operations. Int. Comm. Northwest Atl. Fish. Spec. Pub. 5:166-170.

Pearcy, W.G. 1962. Ecology of an estuarine population of winter flounder *Pseudopleuronectes americanus* (Walbaum). Bull. Bingham Oceanogr. Coll. 18(1):1-78.

Pennington, M. 1983. Efficient estimators of abundance for fish plankton surveys. Biometrics 39:281-286.

Pennington, M. 1986. Some statistical techniques for estimating abundance indices from trawl surveys. Fish. Bull., U.S. 84:519-525.

Perlmutter, A. 1947. The blackback flounder and its fishery in New England and New York. Bull. Bingham Oceanogr. Coll. 11:1-92.

Pihl, L. 1990. Year-class strength regulation in plaice (*Pleuronectes platessa* L.) on the Swedish west coast. Hydrobiologia 195:79-88.

Pihl L., and H.W. van der Veer. 1992. Importance of exposure and habitat structure for the population density of 0-group plaice, *Pleuronectes platessa* L., in coastal nursery areas. Neth. J. Sea Res. 29:145-152.

Pollock, K.H., J.D. Nichols, C. Brownie, and J.E. Hines. 1990. Statistical inference for capture-recapture experiments. Wildl. Monogr. 107. 97 pp.

Poxton, M.G., A. Eleftheriou, and A.D. McIntyre. 1982. The population dynamics of 0-group flatfish in the Clyde Sea area. Est. Coast. Shelf Sci. 14:265-282.

Poxton, M.G., and N.A. Nasir. 1985. The distribution and population dynamics of 0-group plaice (*Pleuronectes platessa* L.) on nursery grounds in the Firth of Forth. Est. Coast. Shelf Sci. 21:845-857.

Rauck, G., and J.J. Zijlstra. 1978. On the nursery aspects of the Wadden Sea for some commercial fish species and possible long-term changes. Rapp. P.-v. Réun. Cons. int. Explor. Mer 172:266-275. (Not seen, cited by Zijlstra et al. 1982).

Reed, M., M.L. Spaulding, E. Lorda, H. Walker, and S.B. Saila. 1984. Oil spill fishery impact assessment modeling: the fisheries recruitment problem. Est. Coast. Shelf Sci. 19:591-610.

Ricker, W.E. 1954. Stock and recruitment. J. Fish. Res. Board Can. 11:559-623.

Ricker, W.E. 1973. Linear regressions in fishery research. J. Fish. Res. Board Can. 30:409-434.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Board Can. 191. 382 pp.

Ricker, W.E. 1984. Computation and uses of central trend lines. Can. J. Zool. 62:1897-1905.

Roff, D.A. 1973. On the accuracy of some mark-recapture estimators. Oecologica (Berl.) 12:15-34.

Rogers, C.A. 1976. Effects of temperature and salinity on the survival of winter flounder embryos. Fish. Bull., U.S. 74:52-58.

Rogers, S.I., and S.J. Lockwood. 1989. Observations on the capture efficiency of a two-metre beam trawl for juvenile flatfish. Neth. J. Sea Res. 23:347-352.

Rothschild, B.J., and G.T. DiNardo. 1987. Comparison of recruitment variability and life history data among marine and anadromous fishes. Pages 531-546 *in* M.J. Dadswell, R.J. Klauda, C.M. Moffitt, R.L. Saunders, R.A. Rulifson, and J.E. Cooper, eds. Common strategies of anadromous and catadromous fishes. Am. Fish. Soc. Sym. 1.

Roughgarden, J. 1979. Evolutionary ecology of single populations. Pages 295-408 *in* The theory of population genetics and evolutionary ecology: an introduction. MacMillan Publishing Company, Inc., New York.

Rubinstein, R.Y. 1981. Simulation and the Monte Carlo method. John Wiley and Sons, New York. 278 pp.

Saila, S.B. 1961. A study of winter flounder movements. Limnol. Oceanogr. 6:292-298.

Saila, S.B. 1962a. The contribution of estuaries to the offshore winter flounder fishery in Rhode Island. Proc. Gulf Caribb. Fish. Inst. 14th Annu. Sess. 1961:95-109.

Saila, S.B. 1962b. Proposed hurricane barriers related to winter flounder movements in Narragansett Bay. Trans. Am. Fish. Soc. 91:189-195.

Saila, S.B., and E. Lorda. 1977. Sensitivity analysis applied to a matrix model of the Hudson River striped bass population. Pages 311-332 *in* W. Van Winkle, ed. Assessing the effects of power-plant-induced mortality on fish populations. Pergamon Press, New York.

SAS Institute Inc. 1985. SAS user's guide: statistics. Version 5 edition. SAS Institute Inc., Cary, NC. 956 pp.

Scarlett, P.G., and R.L. Allen. 1992. Temporal and spatial distribution of winter flounder (*Pleuronectes americanus*) spawning in Manasquan River, New Jersey. Bull. N.J. Acad. Sci. 37:13-17.

Scott, W.B., and M.G. Scott. 1988. Atlantic fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219. 731 pp.

Simpson, D.G. 1989. Codend selection of winter flounder *Pseudopleuronectes americanus*. NOAA Tech. Rep. NMFS 75. 10 pp.

Sissenwine, M.B. 1984. Why do fish populations vary? Pages 59-94 *in* R.M. May, ed. Exploitation of marine communities. Springer-Verlag, New York.

Smigielski, A.S. 1975. Hormonal-induced ovulation of the winter flounder, *Pseudopleuronectes americanus*. Fish. Bull., U.S. 73:431-438.

Smith, E.M., E.C. Mariani, A.P. Petrillo, L.A. Gunn, and M.S. Alexander. 1989. Principal fisheries of Long Island Sound, 1961-1985. Connecticut Dept. Envir. Prot., Bu. Fish., Mar. Fish. Program. 47 pp. + app.

Smith, T.D. 1988. Stock assessment methods: the first fifty years. Pages 1-33 *in* J.A. Gulland, ed. Fish population dynamics (second ed.). John Wiley and Sons, New York.

Smith, W.G., J.D. Sibunka, and A. Wells. 1975. Seasonal distributions of larval flatfishes (Pleuronectiformes) on the continental shelf between Cape Cod, Massachusetts and Cape Lookout, North Carolina, 1965-1966. NOAA Tech. Rep. NMFS SSRF-691. 68 pp.

Snedecor, G.W., and W.C. Cochran. 1967. Statistical methods. The Iowa State University Press, Ames, IA. 593 pp.

Sogard, S.M. 1990. Parameters of habitat quality for epibenthic fishes and decapod crustaceans in New Jersey estuaries. Ph.D. dissertation, Rutgers University, New Brunswick, NJ. 195 pp. (Not seen, cited by Sogard and Able 1992).

Sogard, S.M., and K.W. Able. 1992. Growth variation of newly settled winter flounder (*Pseudopleuronectes americanus*) in New Jersey estuaries as determined by otolith microstructure. Neth. J. Sea Res. 29:163-172.

Southwood, T.R.E. 1978. Ecological methods. Halstead Press, New York. 523 pp.

Spaulding, M.L., S.B. Saila, E. Lorda, H. Walker, E. Anderson, and J.C. Swanson. 1983. Oil-spill fishery impact assessment model: application to selected Georges Bank fish species. Est. Coast. Shelf Sci. 16:511-541.

Steele, J., and R.R.C. Edwards. 1970. The ecology of 0-group plaice and common dabs in Loch Ewe. IV. Dynamics of the plaice and dab populations. J. Exp. Mar. Biol. 4:174-187.

Stuart, A., and J.K. Ord. 1987. Kendall's advanced theory of statistics. Vol. I. Distribution theory. Oxford University Press, New York. 604 pp.

Townsend, D.W., and L.M. Cammen. 1988. Potential importance of the timing of spring plankton blooms to benthic-pelagic coupling and recruitment of juvenile demersal fishes. Biol. Oceanogr. 5:215-229.

Vaughan, D.S. 1981. An age structure model of yellow perch in western Lake Erie. Pages 189-216 *in* D.G. Chapman and V.F. Gallucci, eds. Quantitative population dynamics. International Co-operative Publishing House, Fairland, MD.

Veer, H.W. van der. 1985. Impact of coelenterate predation on larval plaice *Pleuronectes platessa* and flounder *Platichthys flesus* stock in the western Wadden Sea. Mar. Ecol. Prog. Ser. 25:229-238.

Veer, H.W. van der. 1986. Immigration, settlement, and density-dependent mortality of a larval and early postlarval 0-group plaice (*Pleuronectes platessa*) population in the western Wadden Sea. Mar. Ecol. Prog. Ser. 29:223-236.

Veer, H.W. van der, and M.J.N. Bergman. 1987. Predation by crustaceans on a newly settled 0-group plaice *Pleuronectes platessa* population in the western Wadden Sea. Mar. Ecol. Prog. Ser. 35:203-215.

Veer, H.W. van der, M.J.N. Bergman, R. Dapper, and J. IJ. Witte. 1991. Population dynamics of an intertidal 0-group flounder *Platichthys flesus* population in the western Dutch Wadden Sea. Mar. Ecol. Prog. Ser. 73:141-148.

Veer, H.W. van der, L. Pihl, and M.J.N. Bergman. 1990. Recruitment mechanisms in North Sea plaice *Pleuronectes platessa*. Mar. Ecol. Prog. Ser. 64:1-12.

Wigley, S.E., and W.L. Gabriel. 1991. Distribution of sexually immature components of 10 northwest Atlantic groundfish species based on Northeast Fisheries Center bottom trawl surveys 1968-86. NOAA Tech. Mem. NMFS-F/NEC-80. 17 pp.

Williams, P.J., and J.A. Brown. 1992. Development changes in the escape response of larval winter flounder *Pleuronectes americanus* from hatch through metamorphosis. Mar. Ecol. Prog. Ser. 88:185-193.

Witherell, D.B., and J. Burnett. 1993. Growth and maturation of winter flounder, *Pleuronectes americanus*, in Massachusetts. Fish. Bull., U.S. 91:816-820.

Witting, D.A., and K.W. Able. 1993. Effects of body size on probability of predation for juvenile summer and winter flounder based on laboratory experiments. Fish. Bull., U.S. 91:577-581.

Witting, D.A., and K.W. Able. 1995. Predation by sevenspine bay shrimp *Crangon septemspinosa* on winter flounder *Pleuronectes americanus* during settlement: laboratory experiments. Mar. Ecol. Prog. Ser. 123: 23-31.

Zijlstra, J.J., R. Dapper, and J. IJ. Witte. 1982. Settlement, growth and mortality of post-larval plaice (*Pleuronectes platessa* L.) in the western Wadden Sea. Neth. J. Sea Res. 15:250-272.

Zijlstra, J.J., and J. IJ. Witte. 1985. On the recruitment of 0-group plaice in the North Sea. Neth. J. Zool. 35:360-376. (Not seen, cited by van der Veer and Bergman 1987).

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Lobster Studies

Introduction

The principal fishery for American lobster, *Homarus americanus*, in the United States extends from coastal Maine through southern New England and Long Island Sound (LIS). The American lobster supports the most valuable single species commercial fishery in the Northeast United States. In LIS, annual landings have ranged from 0.8 to 2.7 million pounds since 1978 and yielded between 2.4 and 8.4 million dollars to lobstermen employed in the fishery (Blake and Smith 1984; Smith et al. 1989; Connecticut Department of Environmental Protection CT DEP, Marine Fishery Statistics). Between 25 and 30% of the total Connecticut landings during 1996 were made in New London county, which includes the Millstone Point area. Fishing effort is intense throughout the range of the species and recent stock assessments have warned that the lobster resource is overfished and vulnerable to collapse (NMFS 1996). The intense exploitation of lobsters throughout their range has raised concerns over possible impacts of increased fishing mortality rates on egg production and recruitment to coastal populations (Anthony and Caddy 1980). In response to concerns raised by fishery managers, biologists, and lobstermen, the New England Fishery Management Council recommended an increase of the minimum legal size of lobsters to improve larval production and subsequent recruitment. Another regulation was implemented throughout the lobster fishery in the mid-1980s to improve lobster survival by requiring escape vents in wire lobster traps which allow escape of sublegal-sized lobsters. As a result of these management policies, the sustainability of the lobster resource should be enhanced. The new lobster fishery regulations implemented in Connecticut increased the minimum legal size (carapace length) from 81.0 mm ($3\ ^{3}/_{16}$ in) in 1988 to 81.8 mm in 1989 ($3\ ^{7}/_{32}$ in) and to 82.6 mm ($3\ ^{1}/_{4}$ in) in 1990. Federal regulations required lobster producing states to increase the minimum legal size to 84.1 mm ($3\ ^{5}/_{16}$ in). Lobstermen successfully lobbied to hold the minimum legal size at 82.6 mm. However, overwhelming scientific evidence indicates that fishing mortality rates throughout the range of lobsters should be reduced. Fishery managers, regulators and the lobster industry are currently examining other fishing effort reduction techniques, such as trap limitations, moratorium on new licenses, quotas, or closed seasons and areas, to protect the lobster resource. Lobsters in the Millstone Point area are heavily exploited, with over 90% caught in the first year after molting to legal size. The local lobster population has been studied extensively since 1978 to determine if operation of the Millstone Nuclear Power Station (MNPS) has caused changes beyond those expected from natural variability and the high level of fishing.

The potential impacts of power plant operations on the local population 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 could alter recruitment patterns. Also, heated effluent may affect the distribution or behavior of lobsters in the discharge area.

The objectives of the lobster program 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. Since 1984, studies have been conducted during the hatching season to estimate the number of lobster larvae entrained through the cooling water systems. Impacts associated with plant operations on the local lobster population were assessed by comparing results of the 1996 study to other 3-unit operational study years (1986-1995) 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 (1986-1996) to assess impacts associated with the addition of a third unit at Millstone. These results were compared, when appropriate, to other studies conducted in LIS and throughout the range of the American lobster.

Materials and Methods

Description of methods used to conduct lobster population studies can be found in NUSCO (1982, 1987a). 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

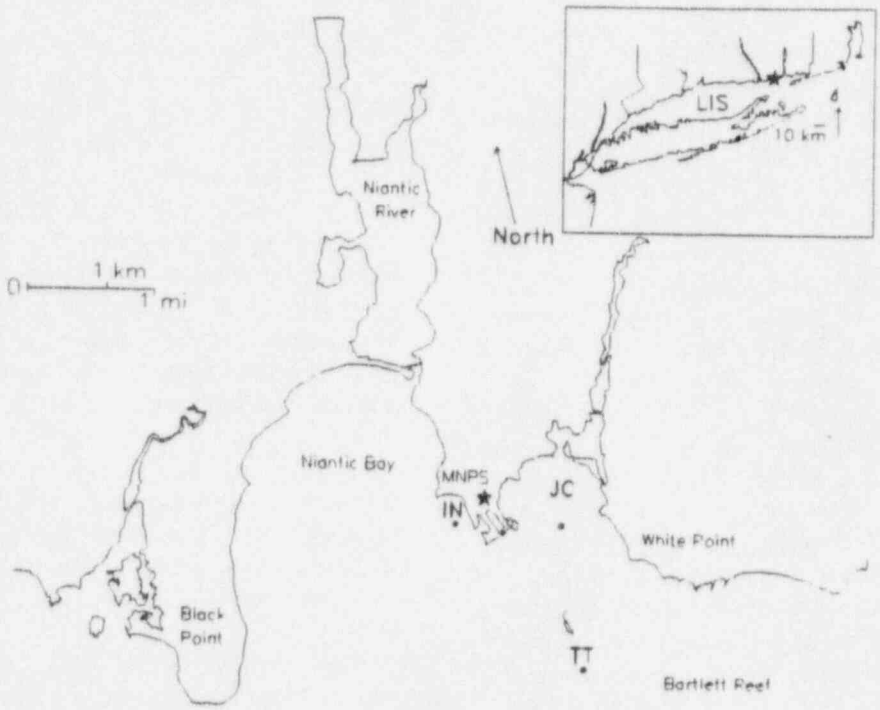


Fig. 1. Location of the Millstone Nuclear Power Station (MNPS), and the three lobster sampling stations (•). JC=Jordan Cove, IN=Intake, TT=Twotree.

both ends, were used to collect lobsters from May through October. Pot-trawls were set in three rocky areas in the vicinity of MNPS (Fig. 1). Pots set in Jordan Cove (average depth 6 m) were 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 or during periods of inclement weather the pots were hauled twice per week. On each sampling trip, surface and bottom water temperatures and salinities were recorded at each station. Lobsters were removed from the pots, banded to restrain chelipeds, transported to the laboratory, and kept in a tank supplied with a continuous flow of seawater. After lobster removal, pots were rebaited 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) to the nearest 0.1 mm, 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 and severely injured or newly molted (soft) lobsters were released untagged after recording the above data.

Beginning in 1981, the size at which females become sexually mature was estimated by measuring the maximum outside width of the second abdominal segment of all females to the nearest 0.1 mm. Female size at sexual maturity was estimated by calculating the ratio of abdominal width to carapace length and plotting that ratio against carapace length (Skud and Perkins 1969; Krouse 1973).

Lobster larvae have been sampled from 1984 to 1996 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. The volume of cooling water 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. From 1984 to 1993, eight lobster larvae entrainment samples (four day and four night) were collected each week; beginning in 1994, the sampling frequency was reduced to six samples per week (three day and three night). Each sample was sorted immediately, or placed in a 1.0 mm mesh sieve and kept for less than 24 h in tanks supplied with a continuous flow of seawater. 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 (I-IV) according to the criteria established by Herrick (1911). The abundance of larvae 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 the total number of larvae entrained, the Δ-mean density was scaled by the total volume of water pumped through the plants 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. Possible impacts 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 computed to analyze trends in CPUE. The geometric mean is better suited for constructing asymmetric confidence intervals for skewed data (McConnaughey and Conquest 1993). Annual geometric mean CPUEs were calculated for all lobster sizes. The annual abundance (CPUE) of legal-size lobsters in the MNPS area 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 contain 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. Slopes of significant trends were calculated using Sen's estimator of the slope (Sen 1968).

The influence of water temperature on lobster molting was examined by estimating the time when lobster molts peaked each year and correlating the annual molt peaks with water temperature. Molting peaks were derived using the inflection point 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 had the form:

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

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

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

Results and Discussion

Water Temperature

Mean monthly surface and bottom water temperatures measured at each station during 2-unit, 3-unit, and 1996 studies are presented in Table 1. Water temperatures during the 1996 lobster study were among the coldest observed since the study began in 1978. Overall mean surface and bottom water temperatures from May through October in 1996 were consistently lower than overall means reported for each station during previous 2- and 3-unit studies. Average May-October surface water temperature during 1996 ranged from 15.8°C at Twotree to 16.1°C at Intake, which was lower than the ranges reported in previous 2-unit (16.3-17.3°C) and 3-unit studies (16.4-18.6°C; Table 1). Bottom water temperatures were also below average from May to October 1996 (range of overall averages 15.6-15.8°C), when compared to previous 2-unit (15.9-16.3°C) and 3-unit studies (16.0-16.6°C). Similarly, at each of the three stations, the monthly mean surface and bottom water temperature values for 1996 were lower than the monthly means reported during the period of 2-unit and 3-unit operation. On average, surface water temperatures in previous 2- and 3-unit studies reached 20.0°C or more at each station in August, however in 1996 the monthly mean surface water temperature reached only 19.2°C at Jordan Cove and Intake and 18.9°C at Twotree.

Historically, surface water temperatures were highest at the stations closest to the discharge (Jordan Cove and Intake) and were slightly warmer during 3-unit studies when compared to 2-unit studies. No MNPS Units were in operation during the 1996 lobster study and, as a result, the monthly and overall surface water temperatures at Jordan Cove and Intake were lower than those observed previously. At Jordan Cove, surface water temperature averaged 17.3 and 18.6°C from May to October during previous 2- and 3-unit studies, respectively; in 1996, surface water temperature averaged 16.0°C (Table 1). At Intake, previous 2- and 3-unit surface temperatures averaged 16.9 and 17.4°C, respectively, while those during 1996 averaged 16.1°C from May to October. The Twotree station, located 1.5 km offshore, is not influenced by the MNPS thermal plume and is less subject to natural cooling and heating; overall surface water temperatures from May to October were similar during 1996, 2- and 3-unit studies (15.8, 16.3 and 16.4°C, respectively). Field temperature data from the lobster study 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 during an ebb tide. At 600 m from the discharge, a 0.8°C isotherm extends to a depth of 3 to 5 m (NUSCO 1988c), which could reach the bottom at some pot-trawls in the Jordan Cove and Intake sites.

TABLE 1. Mean monthly surface and bottom water temperatures (°C) measured at each station during 2-unit (1979-85), 3-unit (1986-95) and 1996 studies.

	Surface			Bottom		
	2-Unit	3-Unit	1996	2-Unit	3-Unit	1996
			Jordan Cove			
MAY	10.2	12.0	9.4	9.2	9.8	8.9
JUN	15.1	17.0	14.0	13.9	14.4	13.7
JUL	19.5	20.8	17.5	18.0	18.3	17.2
AUG	21.2	22.3	19.2	19.9	20.1	18.8
SEP	20.3	21.6	18.8	19.2	19.3	18.8
OCT	16.8	17.6	15.8	16.0	16.2	15.8
Overall	17.3	18.6	16.0	16.1	16.3	15.7
			Intake			
MAY	10.1	10.9	9.7	9.3	10.0	9.0
JUN	14.9	15.7	14.2	14.1	14.7	13.9
JUL	19.2	19.6	17.8	18.3	18.7	17.1
AUG	20.7	21.1	19.2	20.1	20.3	19.0
SEP	19.8	20.1	18.9	19.4	19.5	18.8
OCT	16.1	16.7	15.8	15.9	16.3	15.8
Overall	16.9	17.4	16.1	16.3	16.6	15.8
			Twotree			
MAY	9.4	9.8	9.0	8.9	9.4	8.7
JUN	14.2	14.6	14.0	13.7	14.1	13.6
JUL	18.3	18.5	17.3	17.7	17.9	17.1
AUG	20.0	20.2	18.9	19.6	19.8	18.7
SEP	19.3	19.3	18.9	19.1	19.1	18.8
OCT	16.0	16.1	16.0	15.9	16.0	16.0
Overall	16.3	16.4	15.8	15.9	16.0	15.6

TABLE 2. Catch statistics of lobsters caught in wire[a] pots from 1978 to 1996.

	Total number caught	Number pots hauled	Geometric mean total CPUE	95% C.I.	Δ-mean legal CPUE[b] ≥ 81.0	≥ 81.8	≥ 82.6	95% C.I. ≥ 81.0 mm
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
1995	6435	4317	1.261	1.152 - 1.380	0.180	0.106	**0.080**	0.158 - 0.202
1996	7531	4249	1.587	1.466 - 1.718	0.152	0.091	**0.067**	0.134 - 0.170
2-Unit (1978-85)	40261	25022	1.364	1.337 - 1.403	0.134	0.100	0.085	0.127 - 0.141
3-Unit (1986-96)	91463	47354	1.762	1.728 - 1.796	0.148	0.089	0.068	0.143 - 0.154

[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 1982-96.

[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).

Abundance and Catch-per-Unit-Effort

The total number of lobsters caught during 1996 was 7,531, which was within the range of previous 3-unit studies (6,435-11,438); this total was also within the range of 2-unit studies, when 20 wire traps were used at each site (6,376-9,109; Table 2). The geometric mean total CPUE for 1996 of 1.587 lobsters/pot was also within the range of previous 3-unit (1.261-2.457) and 2-unit studies (0.904-2.006). Relative to the 1995 lobster catch, which was the lowest observed in more than 10 years, the number of lobsters caught and total CPUE increased in 1996 and reversed a trend of declining catches observed since 1992. Although total CPUE during 3-unit studies was higher (1.762 lobsters/pot) than during 2-unit studies (1.364), no significant trends were identified in the time series of total CPUE data collected since 1978.

The Δ-mean CPUE of legal-size lobsters (≥ 82.6 mm) was 0.067 in 1996, which was the lowest value reported since the legal size was increased to 82.6 mm in 1990 (Δ-mean range=0.071-0.091; Table 2). Legal CPUE in 1996 was also less than the values during other years of 3-unit operation, when the legal size was 81.0 mm (1986-88 Δ-mean range=0.079-0.086), but was higher than the 1989 mean (0.065) when legal size was 81.8 mm. The majority of the annual legal CPUE values during 3-unit studies, including 1996, were lower than any legal CPUE reported in 2-unit studies (1978-85 Δ-mean range=0.098-0.173). Legal catches have steadily decreased since 1978, showing a significant declining trend (slope=-0.004, p=0.001). The decline in legal CPUE is most likely due to the fourfold increase in fishing effort since the 1970s (ASMFC 1996) and, more recently, to the increases in minimum legal size in 1989 and 1990. Our studies have shown that the magnitude of legal catches in any year is highly dependent on the abundance of sublegal-sized lobsters the year before; more than 90% of lobsters in the legal size-class had recently molted from the sublegal size-class. Since total CPUE in 1995 was the lowest observed in more

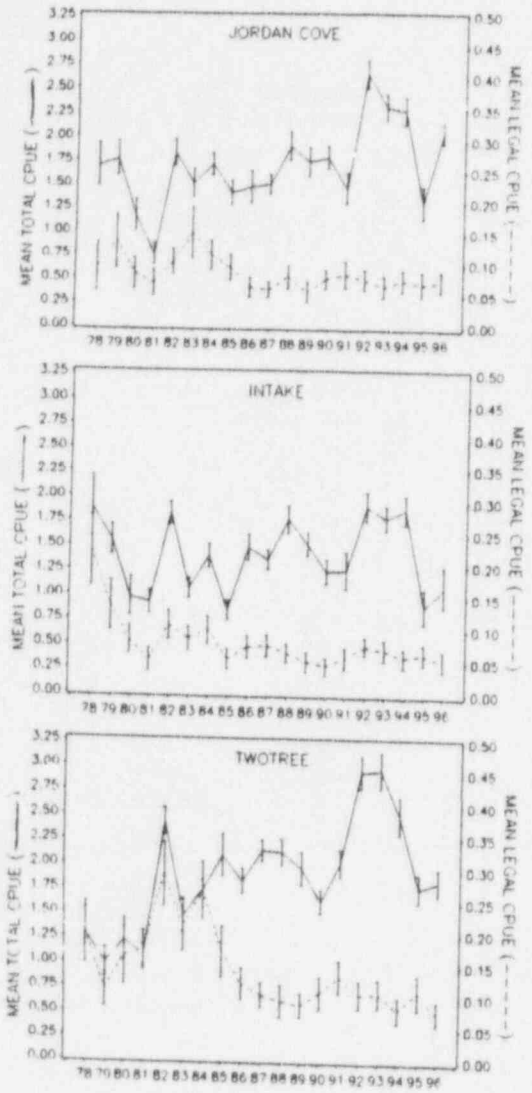


Fig. 2. Mean total CPUE (geometric mean ± 95% C.I.) and mean legal CPUE (Δ-mean ± 95% C.I.) of lobsters caught at each station from 1978 to 1996 (minimum legal size increased from 81.0 mm to 81.8 mm in 1989 and to 82.6 mm in 1990).

than 10 years, fewer legal-size lobsters were expected to be caught in 1996, because fewer sublegal-sized lobsters were available to molt to legal size.

Annual CPUE values for all sizes of lobster and legal-size lobster are presented for each station in Figure 2. The highest total CPUE during 1996 occurred at Jordan Cove (2.036), followed by Twotree (1.830) and Intake (1.073). Total CPUE values during 1996 at each of the three stations were within the range of previous 3-unit studies (Jordan Cove 1.289-2.642; Twotree 1.627-2.957; Intake 0.880-1.908). At Jordan Cove, CPUE during 1996 was higher than the range of CPUEs reported during 2-unit studies (0.153-1.402), whereas at Twotree and Intake, 1996 total CPUEs were within the range of 2-unit studies. No significant trends were identified in the time-series of total CPUE data at the nearshore Jordan Cove and Intake stations, although at Twotree total CPUE significantly increased from 1978 to 1996 (slope=0.062, p=0.011).

During 1996, legal CPUE (lobster ≥ 82.6 mm) was highest at Twotree and Jordan Cove (0.076 and 0.073, respectively) and lowest at Intake (0.052; Fig. 2). Legal CPUE at Twotree during 1996 was the lowest value reported in 3-unit studies (previous range 0.082-0.133), while legal CPUE values for 1996 at Jordan Cove and Intake were within the range of previous 3-unit studies. Since 1978, legal catches have significantly declined at each station (Jordan Cove slope=-0.002, p=0.013; Intake slope=-0.003, p=0.005; Twotree slope=-0.006, p=0.002).

The monthly patterns of total and legal lobster abundance during 2-unit, 3-unit and 1996 studies are presented in Figure 3. Total lobster CPUE was generally highest in June or July, and lowest in October during previous 2- and 3-unit studies. In contrast, the 1996 total CPUE was highest in October (1.871) and lowest in July (1.299; Fig. 3). Values for total CPUE during the summer of 1996 (June, July, and August) were among the lowest observed since the study began in 1978. Legal-size lobster CPUE peaked in July 1996, which was the same month that legal catches peaked during combined 2- and 3-unit study years. The reason for the shift in the pattern of monthly total CPUE is unclear, although it may be related to cooler water temperatures observed during 1996. The 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 1996). Cooler sea temperatures during 1996, relative to previous studies (Table 1), may be responsible for the lower summer catches and delayed peak in total lobster abundance observed in 1996.

Besides lobsters, pots often catch other organisms, which have been shown to influence lobster CPUE in previous years (NUSCO 1987a, 1996). Incidental catches of all species at each station were used as

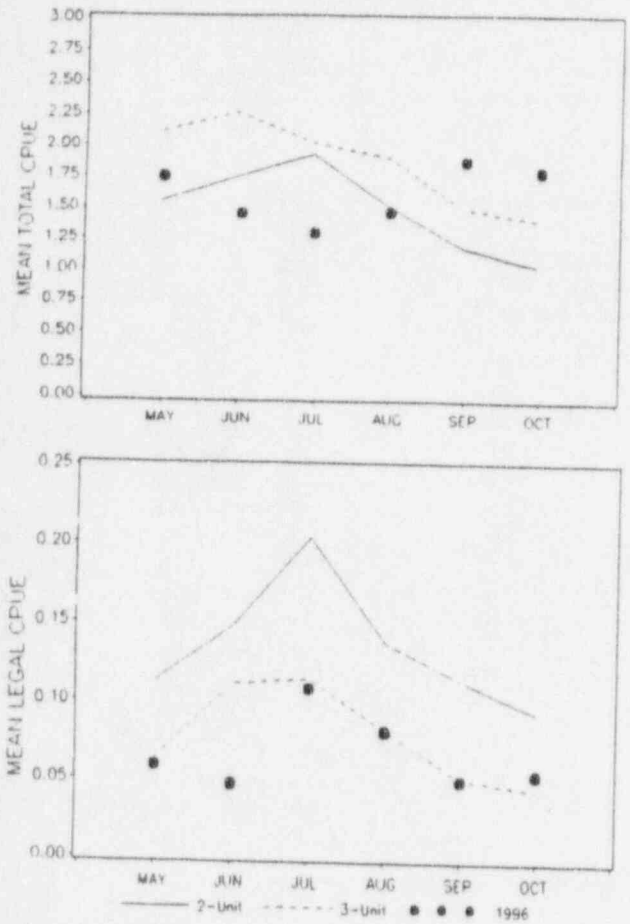


Fig. 3. Monthly mean total CPUE and Δ-mean legal CPUE during 2-unit (1979-85), 3-unit studies (1986-96) and during 1996.

covariates to identify species which significantly ($p<0.05$) influenced lobster catch (Table 3). During 1996, lobster CPUE was influenced by catches of spider crabs at all stations; the total spider crab catch of 24,572 was the second largest observed in the studies (range 1,344-31,480). Rock crabs were also found to influence the catch of lobsters at Twotree during 1996. These species have had a significant influence on lobster CPUE in previous study years. The reason for the substantial increase in spider crab abundance over the past 6 years (>10,000/yr) is unknown, although similar increases in spider crab catches have been observed in the trawl monitoring program. The incidental catches of rock crabs and spider crabs have been reported to significantly affect lobster catch in other studies (Richards et al. 1983; Richards and Cobb 1987).

TABLE 3. Total number of lobsters and incidental catch of other species caught in traps.

	Range (1984-1995)	1996
Lobster	7014-11438	7531
Rock, Jonah crab	79-2033*	848*
Spider crab	1344-31480*	24572*
Hermit crab	217-721*	306
Blue crab	21-148	22
Winter flounder	8-45*	12
Summer flounder	4-60*	14
Skates	14-54	30
Oyster toadfish	7-76	8
Scup	27-288	21
Cunner	48-207	41
Tautog	39-250*	167
Sea raven	0-20	0
Whelks	27-178*	37

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

Pot catches of tautog increased substantially from 1995 (n=48) to 1996 (n=167), particularly at Jordan Cove, where 117 tautog were caught in traps during 1996. Most were caught in August (n=25), September (n=35), and October (n=52). Although covariance analysis did not identify this species as having a significant influence on the catchability of lobsters, the presence of tautog in traps containing lobsters had a profound effect on lobster survival at Jordan Cove from August to October. The percentage of lobsters that were dead or not tagged due to injuries and damages was 16, 12, and 37% at Jordan Cove in August, September, and October, respectively. Typically, in any given year, the percentage of lobsters that are not tagged due to injuries or damage is highest (but still<5%) during the June-July molt when animals are soft-shelled. Tautog are known predators of lobsters (Bigelow and Schroeder 1953; Auster 1985; Cobb et al. 1986) and the reason for the increased abundance in Jordan Cove during 1996 is unclear, although a large year-class of age 2 fish was observed in trawl catches in 1994-95. The abundance of mussels, a common food item of tautog, in the discharge area was greatly reduced during 1996 due to MNPS shutdown. The reduced availability of prey for tautog may have forced fish to search for food in nearby Jordan Cove, and could have been responsible for the increased predation observed by tautog on lobsters in Jordan Cove during 1996. Furthermore, with no thermal discharge entering Jordan Cove due to the shutdown of MNPS during 1996, temperature conditions were favorable for tautog to inhabit the rocky outcrop where pots are set in Jordan Cove.

TABLE 4. Summary of lobster carapace length statistics in wire pot catches from May through October, 1978-1996.

	N[a]	Carapace length (mm) Range	Mean±95% CI	Percentage of legal sizes[b] ≥ 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
1995	5472	37-101	71.9 ± 0.20	10.0	7.3	5.7
1996	6634	16-96	70.0 ± 0.19	7.1	5.0	3.8
2-Unit (1978-85)	34014	38-121	71.3 ± 0.07	7.5	6.3	5.3
3-Unit (1986-96)	79262	16-107	70.3 ± 0.05	6.3	4.6	3.5

[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).

Population Characteristics

Size Frequency

The mean carapace length (CL) of all lobsters during 1996 was 70.0 mm, which was within the range of values reported in previous 3-unit studies (69.5-71.9 mm), but smaller when compared to mean CLs reported during 2-unit studies (70.7-71.8 mm; Table 4). The overall mean CL during 3-unit studies was smaller (70.3 mm) than during 2-unit studies (71.3 mm). The percentage of legal-size lobsters (≥ 82.6 mm) during 1996 was 3.8%, which was within the range reported in previous 3-unit studies (3.1-5.7%; Table 4) when the legal sizes were ≥ 82.6 mm. However, due to the increases in minimum legal size in 1989 and 1990, the percentage of legal-size lobsters during 1996 was lower than the range reported in 2-unit studies when legal size was ≥ 81.0 (5.9-9.1%). The percentages of legal lobsters in our catch have significantly declined since 1978 (slope=-0.275, p=0.002), which can be attributed to increases in minimum legal size and to increased fishing effort, which has more than doubled since 1978 (Blake 1991; NMFS 1993).

When the three stations were compared, the mean CL of lobsters caught during 1996 was largest at Twotree (70.9 mm) and smallest at Jordan Cove (68.8 mm; Table 5). The value at Jordan Cove was the smallest reported in our lobster studies (previous range 69.0-71.3), while those at Intake (70.3 mm) and Twotree (70.9 mm) were within the range of 3-unit studies (68.9-71.5 mm and 70.0-72.5 mm, respectively). Mean sizes at all stations during 1996 were smaller than the range of mean sizes reported in 2-unit studies. In contrast to previous study years, when Twotree yielded the highest percentage of legal-size lobsters, the percentage during 1996 was higher at Intake (4.2%), followed by Twotree (3.8%) and Jordan Cove (3.5%). Percentages of legal-size at Intake and Jordan Cove were within the range of both 2- and 3-unit studies (Table 5). At Twotree the

TABLE 5. Summary of lobster carapace length statistics in wire pot catches at each station from May through October, during 2-Unit (1978-1985), 3-Unit (1986-1995) and 1996 studies.

	Mean carapace length (mm)*	Percentage of legals (≥82.6 mm)
JORDAN COVE		
2-Unit range	69.8 - 71.1	2.5 - 5.9
3-Unit range	69.0 - 71.3	2.1 - 5.1
1996 mean	68.8	3.5
INTAKE		
2-Unit range	69.2 - 71.8	2.9 - 5.7
3-Unit range	68.9 - 71.5	1.8 - 5.8
1996 mean	70.3	4.2
TWOTREE		
2-Unit range	71.3 - 73.7	4.4 - 10.4
3-Unit range	70.0 - 72.5	2.6 - 6.2
1996 mean	70.9	3.8

* Recaptures not included.

percentage of legal-size lobsters caught during 1996 was within the range of 3-unit studies, but lower than the range reported in 2-unit studies.

Sex Ratios

The sex ratio of lobsters collected during 1996 was 0.48 females per male, the lowest ratio ever observed in these studies (1978-95 range 0.71-0.97; Table 6). Female to male sex ratios at each of the three stations were also the lowest observed in 19 years of study. Twotree, which typically has yielded more females than males, had only 0.79 female per male, compared to the range of 0.90 to 1.59 reported previously. Even lower ratios were found at Jordan Cove (0.32) and Intake (0.37) during 1996 and each of these values were well below the range reported since 1978 (0.43-0.79 and 0.47-0.97, respectively). The occurrence of more females at Twotree than at other stations has been consistent since 1975 (Keser et al. 1983). The overall female to male sex ratio during 3-unit studies (0.76) was lower than during 2-unit studies (0.86) and has significantly declined since 1978 (slope=-0.01, p=0.002); significant declines were also noted at Jordan Cove (slope=-0.016, p=0.001) and Intake (slope=-0.019, p=0.001). The cause for the decline in female to male sex ratios is unclear, although it may be related to increased fishing. In the mid-1970s, Smith (1977) reported female to male sex ratios in the LIS commercial fishery ranging between 1.06 and 1.81.

TABLE 6. Female to male sex ratios* of lobsters caught in wire pots from May through October, 1978-1996.

	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
1995	0.53	0.61	0.93	0.71
1996	0.32	0.37	0.79	0.48
2-Unit 78-85	0.67	0.72	1.21	0.86
3-Unit 86-96	0.53	0.60	1.17	0.76

* Recaptures not included.

More recently, sex ratios of lobsters caught in eastern LIS commercial traps were higher, ranging between 2.61 and 6.29 females per male (Blake 1988). Throughout the range of lobster, many researchers have reported sex ratios close to 1:1 for 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

Several methods were used to determine the size at which females become sexually mature. The most obvious indication of female lobster maturity is the presence of external eggs (berried). The smallest berried female collected in studies was 60 mm and further examination of the size distribution of berried females indicated that 50% of the berried females were smaller than 76 and 78 mm during 2- and 3-unit studies, respectively (Fig. 4.). Another technique, first described by Templeman (1935), is

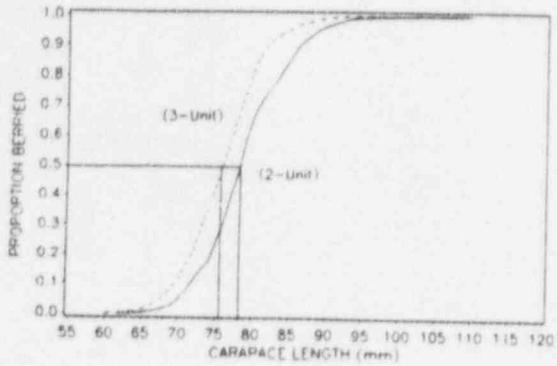
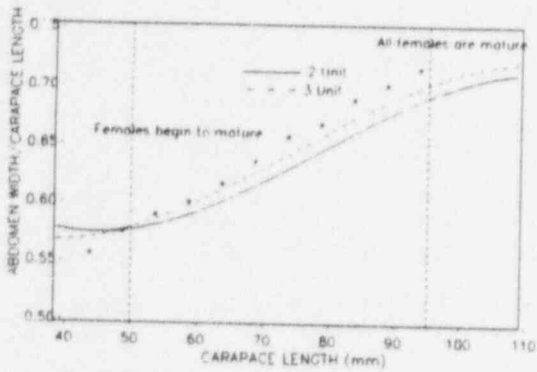


Fig. 4. Proportion of berried females at various sizes collected during 2-unit (——) and 3-unit studies (- - -).



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: $y=1.05-(2.47*10^{-2})x+(3.88*10^{-4})x^2-(1.77*10^{-6})x^3$, $r^2=.32$

Fig. 5. Morphometric relationship between the abdominal width to carapace length ratio (y) and the carapace length (x) for female lobsters during 2-unit (——) and 3-unit studies (- - -) and during 1996 (o o o).

based on abdominal width measurements of females, which markedly increases during maturation. Calculating the abdominal width to CL ratio and comparing it to 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 and plotted against the carapace length of lobsters collected during 2-unit (1981-85) and 3-unit (1985-96) operation and for 1996 alone (Fig. 5). During 1996, females began to mature at about 55 mm CL, and all females larger than 90 mm CL were mature. The close correspondence between the 2- and 3-unit curves in Figure 5 indicates that female size at sexual maturity was similar during both operational periods. The small size of berried females collected in 2-unit (62 mm) and 3-unit studies (60 mm) supported the results of the morphometric relationship between the abdominal width and carapace length. These individuals were between 50-55 mm CL when oviposition first occurred (assuming 14% growth per molt). 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. The New York State Department of Environmental Conservation has found berried females as small as 56 mm CL in their monitoring studies of the LIS lobster fishery (K. Graulich, NYDEC, pers. comm.). Blake (1994) used a technique described by Aiken and Waddy (1982) to estimate sexual maturity of females in LIS and found that half of the females in LIS could extrude eggs at about 73 mm CL. In contrast to LIS, female lobsters in the Gulf of Maine 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). Earlier maturation of females in LIS than in the Gulf of Maine was attributed to warmer LIS water temperatures (Smith 1977; Aiken and Waddy 1980).

The percentage of berried females collected during 1996 was 8.6%, which was within the range of previous 3-unit studies (3.8-12.2%) and higher than the range during 2-unit studies (3.1-6.2%; Table 7). Since 1978, the percentage of berried females has been highest at Twotree, during 1996, 13.3% of the females collected at Twotree were berried, followed by Jordan Cove (4.3%) and Intake (3.3%). The percentages at each of the three stations during 1996 were within the range of previous 3-unit studies, but higher than the ranges reported during 2-unit studies. Overall, berried females were more abundant during 3-unit operation (8.2%) than they were during 2-unit operation (4.3%). The mean CL of berried females collected during 1996 was 76.6 mm, which was within the range of means reported in previous 3-unit studies (75.3-78.1 mm), but smaller than the range of mean CLs reported during 2-unit studies (77.0-81.2 mm). The average size of berried females was lower during 3-unit studies (76.4 mm) than during 2-unit studies (79.4 mm) and reflected the larger proportion of sublegal-sized berried females collected since 1986. Only 9% of the berried females collected during 3-unit studies were above the legal size compared to 32% during 2-unit studies. The reduction in percentage of berried

TABLE 7. Percentage of berried females caught at each station and annual carapace length statistics from 1978-96.

	Percentage of Berried Females				Carapace Length (mm)			
	All stations	Jordan Cove	Intake	Twotree	N[a]	Range	Mean ± 95% C.I.	Percent sublegal[b]
1978	3.4	1.4	2.6	5.3	58	74 - 88	80.1 ± 1.04	73
1979	3.1	1.9	2.7	7.2	70	64 - 93	80.5 ± 1.28	59
1980	3.3	3.5	1.8	5.6	71	66 - 93	79.1 ± 1.27	70
1981	4.2	1.6	2.7	7.1	82	69 - 97	81.2 ± 1.35	55
1982	3.1	0.8	0.9	6.1	108	64 - 99	80.0 ± 1.08	60
1983	4.7	2.1	3.2	8.5	123	66 - 103	80.5 ± 1.04	63
1984	6.2	3.6	3.5	10.6	173	62 - 95	79.1 ± 0.87	69
1985	6.2	3.5	4.5	8.5	171	63 - 94	77.0 ± 0.81	82
1986	4.8	3.0	2.3	8.0	135	65 - 94	78.0 ± 0.95	77
1987	5.7	3.2	1.9	9.6	158	62 - 90	76.5 ± 0.67	92
1988	3.8	2.4	1.9	6.4	124	63 - 90	76.9 ± 0.82	89
1989	5.4	2.8	3.3	8.2	161	65 - 98	77.3 ± 0.78	85
1990	6.6	2.7	4.0	11.2	165	65 - 102	78.1 ± 0.82	87
1991	8.2	3.2	1.5	13.5	226	62 - 96	78.0 ± 0.75	82
1992	12.1	3.4	1.7	19.3	491	60 - 93	75.3 ± 0.44	94
1993	12.2	3.1	2.7	19.4	476	62 - 90	75.6 ± 0.43	93
1994	10.8	6.1	4.7	16.9	372	61 - 91	75.9 ± 0.52	93
1995	9.6	5.9	5.9	13.4	218	64 - 91	76.3 ± 0.61	94
1996	8.6	4.3	3.3	13.3	185	63 - 91	76.6 ± 0.78	89
2-Unit 78-85	4.3	2.0	2.2	7.1	856	62 - 103	79.4 ± 0.39	68
3-Unit 86-96	8.2	3.6	2.9	13.3	2711	60 - 102	76.4 ± 0.19	91

[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).

females above the legal size is due primarily to the increases in minimum legal size in 1989 and 1990, although the high rates of fishing in LIS 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, is due to females that become mature and spawn at sizes well below the legal size, thereby providing a buffer against recruitment failure (Graulich 1996). However, the fecundity of the stock may be lower, as a result of relying on smaller berried females to sustain recruitment, which could affect the long-term health of the LIS fishery.

Molting and Growth

Lobster growth is a function of size and weight increment per molt and molt frequency; water temperature is one of the most important factors that regulates these processes (Aiken 1980). The majority of molting lobsters were caught from late spring to early summer (i.e., the end of May to early July). In previous years, when sampling was conducted through November, a second peak in the catch of molting lobsters was observed in autumn (Keser et al. 1983). The 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 and during 1996 (Fig. 6). 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, suggesting that molting occurred earlier in the years when May water temperatures were warmer than average. Conversely, peaks occurred later when water temperatures were colder than average. The average bottom water temperature during May of 2-unit studies was 9.2°C and molting peaked on 27 June (Fig. 6), which was a week later than the date of peak molting during 3-unit studies (20 June) when the average bottom water temperature was warmer (9.7°C). During

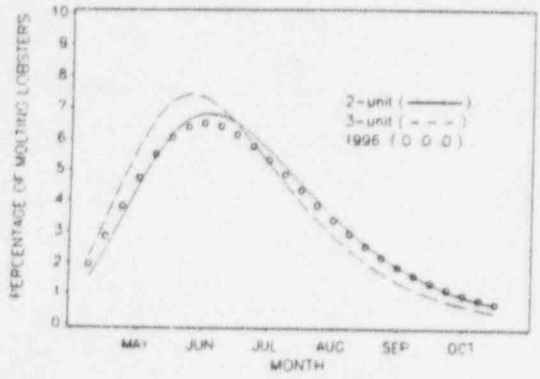


Fig. 6. Molt frequency curves based on the Gompertz function fitted to data on lobsters caught during 2-unit (1978-85; —) and 3-unit studies (1986-96; - - -) and during 1996 (o o o).

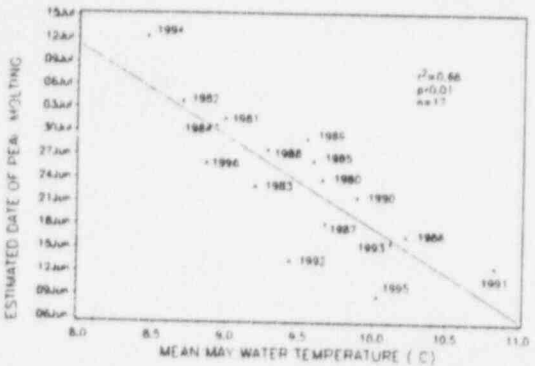


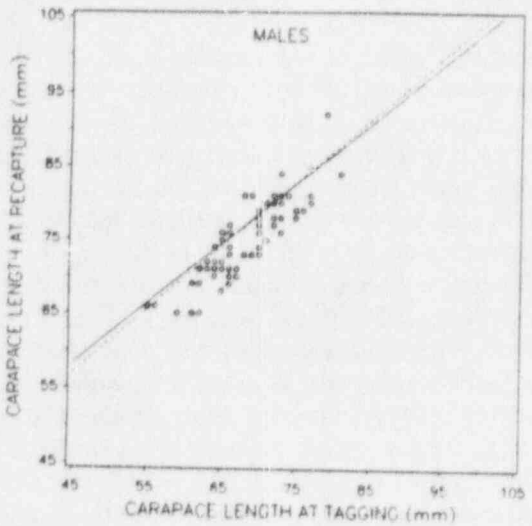
Fig. 7. Relationship between the date of peak molting of lobsters (parameter t from the Gompertz function) and annual mean bottom water temperature during May.

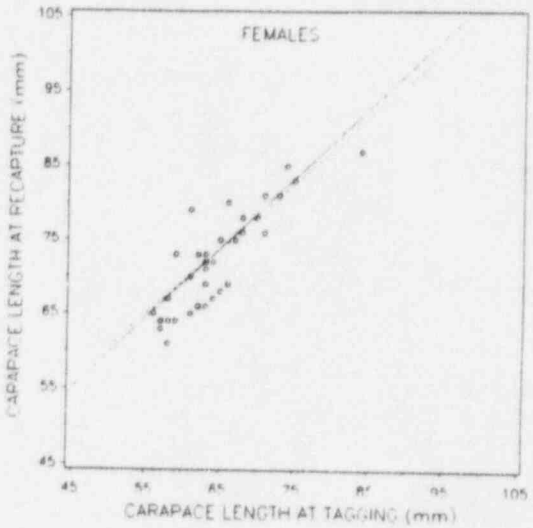
1996, molting peaked on 25 June when bottom water temperature during May averaged 8.9°C. The earliest molting peaks (first and second weeks of June) occurred when bottom water temperatures during May averaged >10.0°C (Fig. 7). Templeman (1936) found molting was delayed 1 week for every 1°C reduction in water temperature. Aiken and Waddy (1980) described the influence of varying water temperature on the molt cycle and found that at 10°C lobsters quickly entered the premolt stage and progressed to ecdysis.

Growth per molt was determined from lobster tagging studies by comparing CL measurements at the time of tagging with those from recaptured lobsters 1 year later. Simple linear regressions of pre-molt (tag-size) and post-molt (recapture-size) sizes best describe growth for the size range of lobsters caught in our studies (Wilder 1953; Kurata 1962; Mauchline 1976). Regression plots and parameter estimates of growth for males and females caught during 1996 and in 2-unit and 3-unit studies are presented in Figure 8. During 1996, average growth increments of both sexes were less than the average increments reported in previous 2- and 3-unit studies. Males grew an average of 6.6 mm per molt during 1996, compared to 8.9 and 8.6 mm during 2- and 3-unit studies, respectively. Average female growth of 7.7 mm was slightly higher than males during 1996, but below the average increment of 8.7 mm reported during both 2- and 3-unit studies. The mean growth increment at Twotree of 8.1 mm during 1996 was within the range of values reported in previous 2- and 3-unit studies (8.0-11.1 mm and 7.2-10.2 mm, respectively; Table 8). However, 1996 growth increments at Jordan Cove (6.2 mm) and Intake (5.6 mm) were the lowest observed since the study began (previous range 7.3-9.1 mm and 7.0-9.5 mm, respectively). The below-average growth increments were most likely related to the cooler seawater temperatures observed throughout the 1996 study period. The average percent growth of males and females during 3-unit studies of 12.9% and 13.4%, respectively, was similar to that reported by other researchers in LIS, which ranged from 11.6% to 15.8% for males and between 12.0% and 15.4% for females (Stewart 1972; Briggs and Mushacke 1984; Blake 1994). Although growth increments were lower during 1996, average incremental growth during 3-unit studies was similar to growth values reported for LIS lobsters and appears to be more related to natural variability in seawater temperature than to power plant operation.

Culls

The percentage of culls, lobsters missing one or both claws, was 12.2% of the total catch during 1996, which was within the range of values reported in previous 3-unit (9.8-12.2%) and 2-unit studies (10.8-15.5%; Table 9). Claw loss percentages during 1996 were lowest at Twotree (7.3%), intermediate at Intake (14.9%) and highest at Jordan Cove (15.0%). The 1996 percentages at Twotree and Intake were within the range of previous 2- and 3-unit studies; however, the percentage at Jordan Cove was higher than the range of other 3-unit years (10.9-14.5%), most likely due to increased predation





	MALES			FEMALES		
	N	Growth model	R^2	N	Growth model	R^2
2-Unit	380	y=22.168+0.805(x)	0.70	587	y=12.678+0.942(x)	0.79
3-Unit	970	y=18.816+0.851(x)	0.71	1077	y=16.371+0.883(x)	0.72
1996	70	y=14.662+0.882(x)	0.76	36	y=9.840+0.967(x)	0.72

Fig. 8. Linear regressions and parameter estimates of carapace lengths at tagging and recapture times of male and female lobsters caught during 2-unit studies (1978-85; ——), 3-unit studies (1986-95; - - -), and during 1996 (O). y=size at recapture, x=size at tagging (mm).

TABLE 8. Summary of lobster growth (in mm and as a percentage of initial size) at each station in wire pot catches for the period May through October during 2-Unit (1978-1985), 3-Unit (1986-1995) and 1996 studies.

	2-Unit			3-Unit			1996		
	N	Range of means (mm)	Range of (%)	N	Range of means (mm)	Range of (%)	N	Mean (mm)	Percentage
Jordan Cove	29-85	7.3-9.0	10.6-13.8	48-107	7.3-9.1	10.9-14.3	44	6.2	9.3
Intake	21-55	7.8-9.5	12.0-14.2	22-72	7.0-9.5	10.7-15.2	14	5.6	8.4
Twotree	21-96	8.0-11.1	11.7-16.3	58-113	7.2-10.2	10.6-15.5	48	8.1	12.5

by tautog (see Abundance and Catch-per-Unit-Effort section). Average claw-loss during combined 3-unit studies was lower (11.0%) than during the 2-unit study period (12.1%), and may have been related to the implementation of the escape vent regulation in 1984. This regulation requires that pots contain an opening to allow escape of subiegal-sized lobsters, and thereby reduces injury and mortality associated with overcrowded pots (Landers and Blake 1985). Since 1984, reported claw-loss (11.0%) has been lower than losses reported before the implementation of escape vents (12.7%). Pecci et al. (1978) reported

TABLE 9. Percentage of culls (lobsters missing one or both claws) caught in wire pots during 1978-1996.

	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
1995	14.2	15.5	7.8	11.9
1996	15.0	14.9	7.3	12.2
2-Unit 78-85	14.4	15.2	7.6	12.1
3-Unit 86-96	12.7	13.9	7.3	11.0

that trap-related injuries were associated with water temperature, fishing pressure (i.e., handling by lobstermen), trap soaktime, and shell hardness. Of these factors, Krouse (1976) reported a positive correlation between fishing pressure and the incidence of culls along the coast of Maine. Other benefits of incorporating escape vents in lobster traps have been noted by many researchers (Krouse and Thomas 1975; Fair and Estrella 1976; Krouse 1978; Pecci et al. 1978; Fogarty and Borden 1980; Krouse et al. 1993).

Tagging Program

The total number of lobsters tagged during 1996 (6,221) was within the range reported in previous 3-unit (5,307-9,126) and 2-unit studies when 20 wire traps were used at each site (1981-85, 4,246-7,575; Table 10). The percentage of lobsters recaptured in NUSCO pots during 1996 was 14.4%, lower than the range of values observed in other 3-unit studies (18.1-26.2%) and among the lowest values observed during 2-unit studies (14.4-23.9%; Table 10). Commercial lobstermen also recaptured fewer lobsters during 1996 (6.7%) than in previous 3-unit (13.5-21.6%) and 2-unit studies (21.1-47.6%). In general, the number of tags returned by commercial lobstermen has declined during 3-unit studies due to the implementation of the escape vent regulation in 1984. Because most of the tagged lobsters are sublegal, fewer were retained in commercial traps with escape vents. In contrast, the number of recaptures made in NUSCO traps during 3-unit studies has increased, because they do not have escape vents and retained more tagged sublegal lobsters. The mean CL of lobsters recaptured in NUSCO traps was 72.4 mm during 1996, which was within the range of previous 3-unit studies (72.0-75.0 mm), but lower than the range of 2-unit studies (73.0-75.7 mm). Overall, the lobsters recaptured in NUSCO traps were smaller (73.0 mm) during 3-unit studies than during 2-unit studies (73.9 mm). In contrast, commercial lobstermen recaptured larger lobsters during 3-unit studies (78.9 mm) than during 2-unit studies (77.1 mm). The shifts in the size of lobsters recaptured in NUSCO and commercial traps were also related to the escape vent regulation. Prior to the regulation, commercial lobstermen recaptured many of the sublegal-sized tagged lobsters; currently, with the regulation in force, many of the sublegals escaped from the vented commercial pots, but were 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.

Since the tagging study began, commercial lobstermen have consistently caught a higher percentage of tagged legal-size lobsters than NUSCO. During 1996, 24.8% of the recaptures in commercial pots were legal-size (≥ 82.6 mm) compared to only 4.2% in NUSCO pots (Table 10). The overall percentage of legal-sized lobsters recaptured in both commercial (24.7%) and NUSCO (3.8%) pots was lower during 3-unit studies than during 2-unit studies (27.9 and 11.0%, respectively). The declines in percentage of legal-sized recaptures are attributed to the increase in minimum legal size and to increased fishing effort, which has more than doubled since 1978.

TABLE 10. Lobster tag and recapture statistics in NUSCO pots (May-Oct.) and commercial pots (Jan.-Dec.) from 1978 to 1996.

	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	1776	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	2518	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	1189	21.1	29.3	78.3
1986	5797	1194	20.9	4.7	72.3	1188	20.4	27.5	78.2
1987	5680	1356	23.9	5.5	72.8	1167	20.4	25.3	78.9
1988	6836	1727	25.3	4.3	72.0	1387	20.2	26.7	78.0
1989	6436	1235	19.2	4.4 (9.2)	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	1559	17.1	23.4 (28.6)	79.5
1993	8177	1708	20.9	3.6 (8.8)	73.4	1768	21.6	27.4 (47.4)	79.4
1994	7533	1974	26.2	3.1 (9.3)	73.4	1020	13.5	20.0 (28.6)	77.3
1995	5307	963	18.1	5.4 (13.5)	75.0	1116	21.0	27.1 (34.1)	80.0
1996	6221	897	14.4	4.2 (10.5)	72.4	419	6.7	24.8 (26.4)	79.0
2-Unit 78-85	38716	7310	18.9	11.0	73.9	12770	33.0	27.9	77.1
3-Unit 86-96	72890	15071	20.7	3.8 (8.9)	73.0	13042	17.9	24.7 (32.1)	78.9

[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 in 1990 to 82.6 mm (3 1/4 in). Parenthetical values for percentage legal from 1989 to 1995 and for 3-unit studies (1986-95) corespond to lobsters ≥ 81.0 mm carapace length.

Movement

Results from the lobster tagging program were also used to assess lobster movement. During 1996, most of the lobsters recaptured in NUSCO pots were caught at the same station where they were released (Jordan Cove 96%, Intake 89%, Twotree 94%). Of the lobsters that moved from the release sites, most moved from Intake to Jordan Cove (10%). This pattern of short-range movement was also observed in previous 2- and 3-unit studies and in the recapture information provided by commercial lobstermen. Ninety-five percent of the tagged lobsters recaptured in commercial pots were caught within 5 km of MNPS (Fig. 9). Stewart (1972) demonstrated a strong homing behavior for the nearshore eastern LIS lobster population. Because lobsters are territorial and nocturnal, they have a limited home range; individuals leave their burrows at night and return to the same shelters before dawn. The predominance of localized movement is typical for nearshore coastal lobster populations and agrees with results of other tagging studies conducted in eastern North America (Templeman 1940; Wilder and Murray 1958; Wilder 1963; Cooper 1970; Cooper et al. 1975; Fogarty et al. 1980; Krouse 1980, 1981; Campbell 1982; Ennis 1984).

Although the tag and recapture studies indicated that most lobsters were nonmigratory and remained in the local area, some lobsters made significant migrations. Most of the lobsters that moved farther than 5 km from MNPS during 2- and 3-unit studies traveled to the southeast. Since 1978, only 18 tagged lobsters were caught by commercial lobstermen in western LIS, compared to 807 caught in The Race and 106 caught in Block Island Sound (10 km and 25 km southeast of MNPS, respectively; Fig. 10). The large number of tagged lobsters caught by commercial lobstermen in The Race suggests that

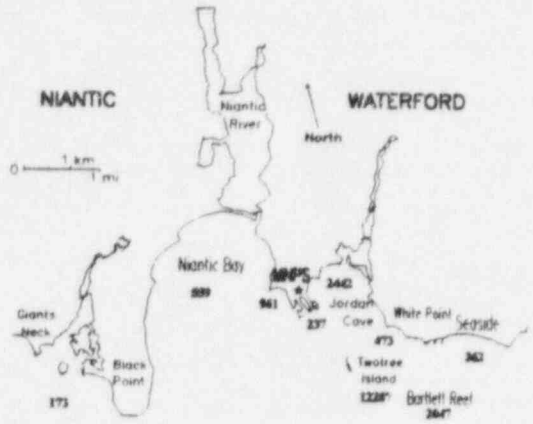


Fig. 9. Location and number of lobsters recaptured in the MNPS area by commercial lobstermen from 1978 to 1996.

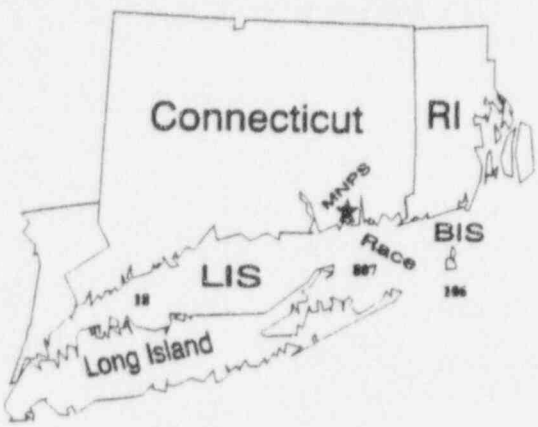


Fig. 10. Location and number of lobsters caught by commercial lobstermen >5 km from MNPS in the vicinity of Long Island and Block Island Sounds (1978-1996).

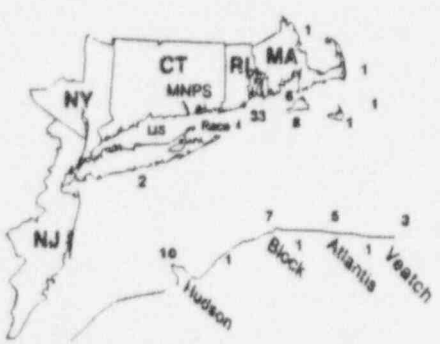


Fig. 11. Number of tag returns at locations >50 km from MNPS (1978-1996).

this deep water channel between Long Island and Block Island is a migration route for lobsters that exit LIS. Once out of the sound, lobsters moved easterly and were recaptured along the Rhode Island coast (33), Buzzards Bay, Massachusetts (6), and in waters near Martha's Vinyard and Nantucket Island (Fig. 11). Only two individuals were reported being caught in nearshore waters along the south shore of Long Island, compared to 28 lobsters which moved farther than 150 km to deeper offshore waters on the edge of the continental shelf, where they were caught in submarine canyons (Hudson, Block, Atlantis, and Veatch). Other researchers have observed similar exchanges between inshore and offshore lobster populations (Saila and Flowers 1968; Uzmann et al. 1977; Cooper and Uzmann 1980; Campbell and Stasko 1985, 1986).

Entrainment

Lobster larvae samples were collected at the MNPS discharges from mid-May through July during 1996. Because of the MNPS shutdown and limited operation of cooling water pumps, only 44 lobster larvae samples were collected during 1996; by contract, between 66 and 104 samples were collected annually in previous 2- and 3-unit studies. In addition to fewer samples collected in 1996, the average volume of entrainment samples was also lower (2,822 m^3), when compared to the range of previous lobster larvae entrainment studies (3,513-4,556 m^3).

As a result of the MNPS shutdown only 19 larvae were collected during 1996, which was well below the range reported in previous 2- and 3-unit studies (102-625). Only 10 Stage I and 9 Stage II larvae were collected during 1996 (Fig. 12). Stage I larvae predominated in previous 2-unit (86 and 87%) and 3-unit studies (38-90%). In general, with the exception of 1988 and 1992, Stage II and III larvae were rarely collected; Stage IV larvae were the second-most abundant larval stage collected in previous 3-unit studies. The fact that no Stage IV larvae were collected during 1996 may also be related to the limited operation of cooling-water pumps due to the MNPS shutdown. Stage IV larvae are strong swimmers; Cobb et al. (1989) reported an average swimming speed of 18 cm/s for wild free-ranging Stage IV lobster larvae. In an early study of

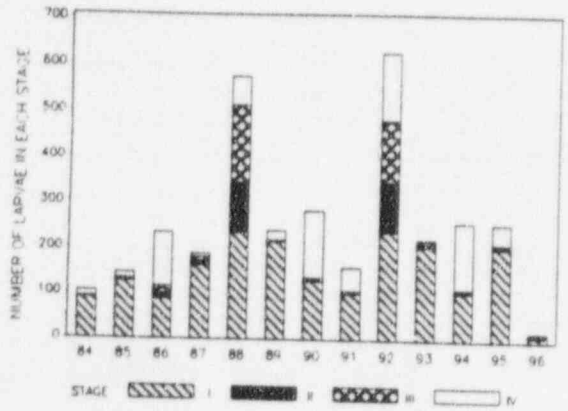


Fig. 12 Annual number of lobster larvae and their stage composition (Stage I-IV) collected in samples taken at the MNPS discharges from 1984 through 1996.

water currents at the intakes of Unit 1, velocities of about 25 cm/s were measured immediately outside the Unit 1 intake during full power operations (4 cooling water pumps; VAST 1972). In previous entrainment studies, sampling occurred at one of the three MNPS discharges during full power operation. However, during 1996, lobster larvae sampling took place when only one or two cooling water pumps were operating. With fewer pumps operating, intake velocities were presumably lower and Stage IV larvae could have avoided entrainment through the cooling-water system.

Day and night entrainment samples have been collected since 1984 to examine diel variation in lobster larvae abundance. During 1996, the density of lobster larvae in day and night samples was similar (0.329 and 0.242 per 1000 m^3, respectively; Table 11). Significantly higher larval densities were observed in night samples in six years of 3-unit operation (1986, 1989, 1990, 1991, 1994, 1995). This observed diel variability was similar to results found elsewhere. Early laboratory studies on lobster larvae behavior demonstrated that Stage I larvae exhibit positive phototaxis and disperse from surface waters during darkness (Templeman 1937, 1939). In contrast, field surveys conducted by Harding et al. (1987) in Canadian waters indicated that most Stage I larvae were collected at depths between 15 and 30 m during the day and were rarely found below 10 m at night. Boudreau et al. (1991) concluded that thermal gradients influenced the vertical migration of lobster larvae; all four stages were found to seek warm water above a thermocline regardless of time of day. More recently, DiBacco and Pringle (1992) found significantly more larvae (Stage I) during nighttime than during daytime neuston tows made within a protected coastal embayment along the Nova Scotian Shelf.

TABLE 11. Annual Δ-mean density (number per 1000 m^3 ± 95% C.I.) of lobster larvae collected in day and night entrainment samples from 1984 through 1996.

Year	Time of day	Δ-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	0.290-0.951
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
1995	Day	0.594	-0.122-1.310
	Night	2.189[b]	0.369-4.009
1996	Day	0.329	-1.399-2.057
	Night	0.242	-0.731-1.215

[a] Number per 1000 m^3.
[b] Significant difference between day and night densities based on 2-sample t-tests ($p<0.05$).

The annual Δ-mean density of lobster larvae collected in entrainment samples during 1996 (0.364 per 1000 m^3) was the lowest value reported since the entrainment studies began in 1984 (previous 3-unit range of 0.525-1.385; 2-unit range of 0.409 and 0.504; Table 12). An estimated 20,106 lobster larvae were entrained through the MNPS cooling-

TABLE 12. Annual Δ-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 through 1996.

Year	Time period included	Number larvae	Mean density[a]	95% C.I.	Cooling Vol. (m^3 x 10^6)	Entrainment 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
1995	30May-21Jul	254	1.385	0.470-2.300	495.9	686,826	233,075-1,140,578
1996	6Jun-1Jul	19	0.364	0.194-0.535	55.2	20,106	10,716-29,551

[a] Mean densities are based on the Δ-mean (NUSCO 1988b and Pennington 1983).
[b] Unit 3 began commercial operation.

water system in 1996, which was the lowest estimate reported during both 3-unit (296,173-686,826) and 2-unit studies (77,458 and 128,550). Lobster larvae entrainment is related to both the annual larval density and the performance of the MNPS units during the hatching season. When all units operate at full capacity, cooling-water demands are at a maximum and resulting entrainment estimates are higher. Conversely, entrainment estimates are low when one or more units are shutdown for maintenance or refueling. The low entrainment estimate during 1996 was primarily due to MNPS shutdown; combined 3-unit cooling water volume of 55.2 m^3x10^6 was well below the range of values reported previously (189.4-837.6 m^3x10^6). Nonetheless, since Unit 3 began operating in 1986, entrainment estimates have been substantially higher because the cooling-water demand of Unit 3 alone is approximately the volume required by Unit 1 and 2 combined.

The impacts on the adult lobster population due to entrainment of lobster larvae are difficult to assess due to the high variability in lobster larvae abundance and stage composition (Bibb et al. 1983; Fogarty 1983; Lux et al. 1983; Blake 1984, 1988) and the lack of reliable estimates of larval and post-larval survival rates (Phillips and Sastry 1980; Caddy and Campbell 1986; Cobb 1986; Blake 1991). Disagreement among researchers on the source and dispersion mechanism of lobster larvae, and on egg and larval mortality, has led to a range of survival estimates during the larval life history phase from less than 1% in Canadian waters (Scarratt 1964, 1973; Harding et al. 1982) to more than 50% in LIS (Lund and Stewart 1970; Blake 1991). Mechanisms of lobster larvae dispersal in coastal waters may be related to surface water circulation patterns (Fogarty 1983); surface currents, regulated by the wind and tide, converge and are visible on the surface waters as "slick" or "scum" lines. These convergence areas delineate zones of upwelling and downwelling and were reported to contain high densities of planktonic organisms, including lobster larvae (Cobb et al. 1983; Blake 1988). "Slick" lines were often seen in the MNPS area stretching from near Twotree Island into Niantic Bay and could explain the patchy distribution of lobster larvae in entrainment samples. Furthermore, based on the short duration of the first larval stage (3-5 days), the source of Stage I larvae collected in the MNPS cooling-water was probably from the local spawners (e.g., from Twotree, where over 13% of females have been berried during 3-unit operation). Stage IV larvae, however, are in the water column between 4 and 6 weeks. Therefore, based on water circulation patterns in LIS, it is unlikely that Stage IV larvae originate locally. 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. More recent studies have shown that Stage IV larvae exhibit directional swimming behavior and moved tens of kilometers from the spawning grounds (Cobb et al. 1989; Rooney and Cobb 1991; Katz et al. 1994).

Conclusions

The American lobster fishery is an extremely important fishery for New England and mid-Atlantic states. Scientists and fishery managers are concerned about the health of the lobster resource, and are focusing on the intense fishing pressure applied to the resource and potential long-term effects on recruitment. Several management measures have been implemented to improve lobster recruitment and survival, including requirements for escape vents in lobster pots and increases in minimum legal size. Results from our studies indicated that more than 90% of the lobsters larger than the minimum legal size are removed by fishing each year and the trend in legal lobster abundance has significantly declined since lobster studies began in 1978. This decline is due, in part, to increased fishing rates, which have more than doubled since 1978, and, more recently, to the increases in minimum legal size in 1989 and 1990. While the total number of lobsters caught (all sizes) and total CPUE in 1996 were within the range of previous years, the CPUE of legal lobsters in 1996 was the lowest observed in 10 years of 3-unit operational studies. The magnitude of legal catches in any year is highly dependent upon the abundance of lobsters one molt smaller than legal size the year before. Legal catches were expected to decline in 1996, based on low total CPUE during 1995, which was the lowest reported in almost 20 years, with fewer sublegal-size lobsters available to molt to legal size in 1996.

One of the most important physiological factors regulating lobster activity is water temperature, and May to October water temperatures during 1996 were among the coldest observed in our lobster studies. As a result of the below-normal water temperatures, lobster catches and molting peaked later during 1996. Overall water temperatures during the period of 3-unit operation were, on average, warmer than those observed during 2-unit operation and subsequent peaks in lobster catch and molting occurred earlier. Other changes were observed in the population characteristics of local lobsters during 3-unit studies, but were related to increased fishing rates and to implementation of new fishery regulations rather than to power plant impacts. The most notable change in lobster population characteristics during 3-unit studies was an increase in the proportion of berried females collected. The objective of increasing the minimum legal size of lobsters was to enhance recruitment and to sustain the lobster resource; larval production should increase as a larger proportion of berried females will be able to spawn before reaching legal size.

Due to the MNPS shutdown during 1996, the density of lobster larvae and estimated total number of larvae entrained through the cooling-water systems were the lowest values reported since the entrainment studies began in 1984. Overall entrainment levels have been substantially higher during 3-unit operation than during 2-unit operation, due to the higher cooling-water demand of Unit 3. The potential effect of higher larval entrainment on subsequent legal lobster abundance is difficult to assess due to the uncertainty concerning the source of entrained larvae, their survival rate, and the relatively long period of time between larval settlement and recruitment to the fishery. The impact of lobster larvae entrainment is not immediately observable and its effect on the fishery may not be seen for several years after the impact occurs. Continued monitoring of lobsters will demonstrate the relationship of MNPS operations on the local lobster population and its development.

References Cited

Aiken, D.E. 1973. Proecdysis, setal development, and molt prediction in the American lobster, (*Homarus americanus*). J. Fish. Res. Board Can. 30:1337-1344.

Aiken, D.E. 1980. Molting and Growth. Pages 91-163 *in* J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol. I, Academic Press, Inc., New York.

Aiken, D.E., and S.L. Waddy. 1980. Reproductive biology. Pages 215-276 *in* J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol. I, Academic Press, Inc., New York.

Aiken, D.E., and S.L. Waddy. 1982. Cement gland development, ovary maturation, and reproductive cycles in the American lobster *Homarus americanus*. J. Crust. Biol. 2:315-327.

Anthony, V.C., and J.F. Caddy. 1980. Proceedings of the Canada-U.S. workshop on status of assessment science for N.W. Atlantic lobster

(*Homarus americanus*) stocks (St. Andrews, N.B., Oct 24-26, 1978). Can. Tech. Rep. Fish. Aquat. Sci. 932. 186 pp.

ASMFC (Atlantic States Marine Fisheries Commission). 1996. A review of the population dynamics of American lobster in the Northeast. Special Report No. 61. 48 pp.

Auster, P.J. 1985. Factors affecting catch of American lobster, *Homarus americanus* in baited traps. Univ. Conn. Sea Grant Prog. Tech. Bull. Ser. CT-SG-85-1. 46 pp.

Bibb, B.G., R.L. Hersey, and R.A. Marcello, Jr. 1983. Distribution and abundance of lobster larvae (*Homarus americanus*) in Block Island Sound. NOAA Tech. Rep. NMFS SSRF-775:15-22.

Bigelow, H.B., and W.C. Schroeder. 1953. Fishes of the Gulf of Maine. U.S. Fish Wildl. Serv. Bull. 53:1-577.

Blake, M.M. 1984. Annual progress report Connecticut lobster investigations, January-December 1983. NOAA-NMFS Project No. 3-374-R. 47 pp.

Blake, M.M. 1988. Final Report Connecticut lobster investigations January 1, 1983-December 31, 1987. NOAA-NMFS Project No. 3-374-R. 103 pp.

Blake, M.M. 1991. Connecticut lobster (*Homarus americanus*) population recruitment studies January 1, 1988-December 31, 1990. NOAA-NMFS Project No. 3IJ4. 87 pp.

Blake, M.M. 1994. Connecticut lobster (*Homarus americanus*) population recruitment studies April 1, 1991-March 31, 1994. NOAA-NMFS Project No. 3IJ4. 174 pp.

Blake, M.M., and E.M. Smith. 1984. A marine resources plan for the state of Connecticut. Connecticut Dept. of Environ. Protection, Mar. Fish. 244 pp.

Boudreau, B., Y. Simard, and E. Bourget. 1991. Behavioural responses of the planktonic stages of American lobster *Homarus americanus* to thermal gradients, and ecological implications. Mar. Ecol. Prog. Ser. 76:13-23.

Briggs, P.T., and F.M. Mushacke. 1979. The American lobster in western Long Island Sound. NY Fish Game J. 26:59-86.

Briggs, P.T., and F.M. Mushacke. 1980. The American lobster and the pot fishery in the inshore waters off the south shore of Long Island, New York. NY Fish Game J. 27:156-178.

Briggs, P.T., and F.M. Mushacke. 1984. The American lobster in western Long Island Sound: Movement, growth and mortality. NY Fish Game J. 31:21-37.

Caddy, J.F., and A. Campbell. 1986. Summary of session 9: summary of research recommendations. Can. J. Fish. Aquat. Sci. 43:2394-2396.

Campbell, A. 1982. Movements of tagged lobsters released off Port Maitland, Nova Scotia, 1944-80. Can. Tech. Rep. Fish. Aquat. Sci. No. 1136. 41 pp.

Campbell, A., and A.B. Stasko. 1985. Movements of tagged American lobsters, *Homarus americanus*, off southwestern Nova Scotia. Can. J. Fish. Aquat. Sci. 42:229-238.

Campbell, A., and A.B. Stasko. 1986. Movements of lobsters (*Homarus americanus*) tagged in the Bay of Fundy, Canada. Mar. Biol. 92:393-404.

Cobb, J.S. 1986. Summary of session 6: ecology of population structures. Can. J. Fish. Aquat. Sci. 43:2389-2390.

Cobb, J.S., T. Gulbransen, B.F. Phillips, D. Wang, and M. Syslo. 1983. Behavior and distribution of larval and early juvenile *Homarus americanus*. Can. J. Fish. Aquat. Sci. 40:2184-2188.

Cobb, J.S., D. Wang, R.A. Richards, and M.J. Fogarty. 1986. Competition among lobsters and crabs and its possible effects in Narragansett Bay, Rhode Island. Pages 282-290 *in* G.S. Jamieson and N. Bourne, eds. North Pacific Workshop on stock assessment and management of invertebrates. Can. Spec. Publ. Fish. Aquat. Sci. 92.

Cobb, J.S., D. Wang, D.B. Campbell, and P. Rooney. 1989. Speed and direction of swimming by postlarvae of the American lobster. Trans. Am. Fish. Soc. 118:82-86.

Cooper, R.A. 1970. Retention of marks and their effects on growth, behavior and migrations of the American lobster, *Homarus americanus*. Trans. Amer. Fish. Soc. 99:409-417.

Cooper, R.A., R.A. Clifford, and C.D. Newell. 1975. Seasonal abundance of the American lobster, *Homarus americanus*, in the Boothbay Region of Maine. Trans. Amer. Fish. Soc. 104:669-674.

Cooper, R.A., and J.R. Uzmann. 1980. Ecology of juvenile and adult *Homarus americanus*. Pages 97-142 *in* J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol II, Academic Press, Inc., New York.

DiBacco, C., and J.D. Pringle. 1992. Larval lobster (*Homarus americanus*, H. Milne Edwards, 1837) distribution in a protected Scotian Shelf bay. J. Shellfish Res. 11:81-84.

Dow, R.L. 1966. The use of biological, environmental and economic data to predict supply and to manage a selected marine resource. The Amer. Biol. Teacher 28:26-30.

Dow, R.L. 1969. Cyclic and geographic trends in seawater temperature and abundance of American lobster. Science 164:1060-1063.

Dow, R.L. 1976. Yield trends of the American lobster resource with increased fishing effort. Mar. Technol. Soc. 10:17-25.

Ennis, G.P. 1971. Lobster (*Homarus americanus*) fishery and biology in Bonavista Bay, Newfoundland. 1966-70. Fish. Mar. Serv. Tech. Rep. 289. 46 pp.

Ennis, G.P. 1974. Observations on the lobster fishery in Newfoundland. Fish. Mar. Serv. Tech. Rep. 479. 21 pp.

Ennis, G.P. 1984. Small-scale seasonal movements of the American lobster, *Homarus americanus*. Trans. Am. Fish. Soc. 113:336-338.

Fair, J.J., and B. Estrella. 1976. A study on the effects of sublegal escape vents on the catch of lobster traps in five coastal areas of Massachusetts. Unpublished manuscript, Mass. Div. Mar. Fish. 9pp.

Flowers, J.M., and S.B. Saila. 1972. An analysis of temperature effects on the inshore lobster fishery. J. Fish. Res. Board Can. 29:1221-1225.

Fogarty, M.J. 1983. Distribution and relative abundance of American lobster, *Homarus americanus* larvae: New England investigations during 1974-79. NOAA Tech. Rep. NMFS SSRF-775. 64 pp.

Fogarty, M.J., and D.V.D. Borden. 1980. Effects of trap venting on gear selectivity in the inshore Rhode Island American lobster, *Homarus americanus*, fishery. Fish. Bull., U.S. 77:925-933.

Fogarty, M.J., D.V.D. Borden, and H.J. Russell. 1980. Movements of tagged American lobster, *Homarus americanus*, off Rhode Island. Fish. Bull., U.S. 78:771-780.

Graulich, K. 1996. A model to assess egg production and the impacts of fishing mortality on total egg production in Long Island Sound lobsters. Division of Marine Resources - New York State Department of Environmental Conservation. 22 pp.

Harding, G.C., W.P. Vass, and K.F. Drinkwater. 1982. Aspects of larval American lobster (*Homarus americanus*) ecology in St. Georges Bay, Nova Scotia. Can. J. Fish. Aquat. Sci. 39:1117-1129.

Harding, G.C., J.D. Pringle, W.P. Vass, S. Pearre Jr., and S.J. Smith. 1987. Vertical distribution and daily movement of larval lobsters *Homarus americanus* over Browns Bank, Nova Scotia. Mar. Ecol. Prog. Ser. 49:29-41.

Herrick, F.H. 1911. Natural history of the American lobster. Bull. U.S. Bureau Fish. 29:149-408.

Hollander, M., and D.A. Wolfe. 1973. Nonparametric statistical methods. John Wiley and Sons. New York. 503 pp.

Katz, C.H., J.S. Cobb, and M. Spaulding. 1994. Larval behavior, hydrodynamic transport, and potential offshore recruitment in the American lobster, *Homarus americanus*. Mar. Eco. Prog. Ser. 103:265-273.

Keser, M., D.F. Landers, Jr., and J.D. Morris. 1983. Population characteristics of the American lobster, *Homarus americanus*, in eastern Long Island Sound, Connecticut. NOAA Tech. Rep. NMFS SSRF-770. 7 pp.

Krouse, J.S. 1973. Maturity, sex ratio, and size composition of the natural population of American lobster, *Homarus americanus*, along the Maine coast. Fish. Bull., U.S. 71:165-173.

Krouse, J.S. 1976. Incidence of cull lobsters, *Homarus americanus*, in commercial and research catches off the Maine coast. Fish. Bull., U.S. 74:719-724.

Krouse, J.S. 1978. Effectiveness of escape vent shape in traps for catching legal-sized lobster, *Homarus americanus*, and harvestable-sized crabs, *Cancer borealis* and *Cancer irroratus*. Fish. Bull., U.S. 76:425-432.

Krouse, J.S. 1980. Summary of lobster, *Homarus americanus*, tagging studies in American waters (1898-1978). Can. Tech. Rep. Fish. Aquat. Sci. 932:135-140.

Krouse, J.S. 1981. Movement, growth, and mortality of American lobsters, *Homarus americanus*, tagged along the coast of Maine. NOAA Tech. Rep. NMFS SSRF-747. 12 pp.

Krouse, J.S., and J.C. Thomas. 1975. Effects of trap selectivity and some population parameters on the

size composition of the American lobster, *Homarus americanus*, catch along the Maine coast. Fish. Bull., U.S. 73:862-871.

Krouse, J.S., K.H. Kelly, D.B. Parkhurst Jr., G.A. Robinson, B.C. Scully, and P.E. Thayer. 1993. Maine Department of Marine Resources Lobster Stock Assessment Project 3-IJ-61-1. Annual report April 1, 1992 through January 31, 1993. 61 pp.

Kurata, H. 1962. Studies on the age and growth of Crustacea. Bull. Hokkaido Reg. Fish. Res. Lab. 24:1-115.

Landers, D.F., Jr., and M.M. Blake. 1985. The effect of escape vent regulation on the American lobster, *Homarus americanus*, catch in eastern Long Island Sound, Connecticut. Trans. 41st Annual Northeast Fish Wild. Conf. 9 pp.

Lund, W.A., Jr., and L.L. Stewart. 1970. Abundance and distribution of larval lobsters, *Homarus americanus*, off southern New England. Proc. Natl. Shellfish. Assoc. 60:40-49.

Lux, F.E., G.F. Kelly, and C.L. Wheeler. 1983. Distribution and abundance of larval lobsters (*Homarus americanus*) in Buzzards Bay, Massachusetts, in 1976-79. NOAA Tech. Rep. NMFS SSRF-775:29-33.

Mauchline, J. 1976. The Hiatt growth diagram for Crustacea. Mar. Biol. 35:79-84.

McConnaughey, R.A., and L.L. Conquest. 1993. Trawl survey estimation using a comparative approach based on lognormal theory. Fish. Bull., U.S. 91:107-118.

McLeese, D.W., and D.G. Wilder. 1958. The activity and catchability of the lobster (*Homarus americanus*) in relation to temperature. J. Fish. Res. Board Can. 15:1345-1354.

NMFS (National Marine Fisheries Service). 1993. Report of the 16th Northeast Regional Stock Assessment Workshop. Northeast Fish. Sci. Cen. Ref. Doc. 93-18. NOAA/NMFS Northeast Fisheries Science Center, Woods Hole, MA. 118 pp.

NMFS. 1996. Report of the 22nd Northeast Regional Stock Assessment Workshop (22nd SAW), Stock Assessment Review Committee (SARC) Consensus Summary of Assessments. NOAA/NMFS Northeast Fisheries Science Center, Woods Hole, MA. 135 pp.

NUSCO (Northeast Utilities Service Company). 1982. Lobster Population Dynamics-A Review and Evaluation. Pages 1-32 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Resume 1968-1981.

NUSCO. 1986. The effectiveness of the Millstone Unit 1 sluiceway in returning impinged organisms to Long Island Sound. Enclosure to letter D01185 dated May 27, 1986 from R.A. Reckert, NUSCO, to S.J. Pac, Commissioner, CTDEP. 18 pp.

NUSCO. 1987a. Lobster population dynamics. Pages 1-42 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of studies prior to Unit 3 operation 1987.

NUSCO. 1987b. The effectiveness of the Unit 3 fish return system 1987. 20 pp.

NUSCO. 1988a. The usage and estimation of DELTA means. Pages 311-320 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.

NUSCO. 1988b. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CTDEP. 21 pp.

NUSCO. 1988c. Hydrothermal Studies. Pages 323-355 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.

NUSCO. 1996. Lobster studies. Pages 9-32 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1995.

Pecci, K.J., R.A. Cooper, C.D. Newell, R.A. Clifford, and R.J. Smolowitz. 1978. Ghost fishing of vented and unvented lobster, *Homarus americanus*, traps. Mar. Fish. Rev. 40:9-43.

Pennington, M. 1983. Efficient estimators of abundance, for fish plankton surveys. Biometrics 39:281-286.

Phillips, B.F., and A.N. Sastry. 1980. Larval ecology. Pages 11-57 *in* J.S. Cobb, and B.F. Phillips, eds. The biology and management of Lobsters, Vol II, Academic Press, Inc., New York.

Richards, R.A., J.S. Cobb, and M.J. Fogarty. 1983. Effects of behavioral interactions on the

catchability of American lobster, *Homarus americanus*, and two species of *Cancer* crab. Fish. Bull., U.S. 81:51-60.

Richards, R.A., and J.S. Cobb. 1987. Use of avoidance responses to keep spider crabs out of traps for American lobsters. Trans. Amer. Fish. Soc. 116:282-285.

Rooney, P., and J.S. Cobb. 1991. Effects of time of day, water temperature, and water velocity on swimming by postlarvae of the American Lobster, *Homarus americanus*. Can. J. Fish. Aquat. Sci. 48:1944-1950.

Saila, S.B., and J.M. Flowers. 1968. Movements and behavior of berried female lobsters displaced from offshore areas to Narragansett Bay, Rhode Island. J. Cons. Int. Explor. Mer. 31:342-351.

Scarratt, D.J. 1964. Abundance and distribution of lobster larvae (*Homarus americanus*) in Northumberland Strait. J. Fish. Res. Board Can. 21:661-680.

Scarratt, D.J. 1970. Laboratory and field tests of modified sphyrion tags on lobsters (*Homarus americanus*). J. Fish. Res. Board Can. 27:257-264.

Scarratt, D.J. 1973. Abundance, survival, and vertical and diurnal distribution of lobster larvae in Northumberland Strait 1962-63, and their relationships with commercial stocks. J. Fish. Res. Board Can. 30:1819-1824.

Scarratt, D.J., and P.F. Elson. 1965. Preliminary trials of a tag for salmon and lobsters. J. Fish. Res. Board Can. 22:421-423.

Sen, P.K. 1968. Estimates of the regression coefficient based on Kendall's tau. J. Am. Stat. Assoc. 63:1379-1389.

Skud, B.E., and H.C. Perkins. 1969. Size composition, sex ratio and size at maturity of offshore northern lobsters. U.S. Fish Wildl. Spec. Sci. Rep. Fish. 598. 10 pp.

Smith, E.M. 1977. Some aspects of catch/effort, biology, and the economics of the Long Island lobster fishery during 1976. NOAA-NMFS, Commer. Fish. Res. Dev. Act, Project No. 3-253-R-1. 97 pp.

Smith, E.M., E.C. Mariani, A.P. Petrillo, L.A. Gunn, and M.S. Alexander. 1989. Principal fisheries of Long Island Sound, 1961-1985. Connecticut Dept. of Environ. Protection, Div. of Conservation and Preservation, Bureau of Fisheries, Mar. Fish. Pro.

Stewart, L.L. 1972. The seasonal movements, population dynamics and ecology of the lobster, *Homarus americanus* (Milne-Edwards), off Ram Island, Connecticut. Ph.D. Thesis, University of Connecticut, Storrs, CT. 112 pp.

Templeman, W. 1935. Local differences in the body proportions of the lobster, *Homarus americanus*. J. Biol. Board Can. 1:213-226.

Templeman, W. 1936. Local differences in the life history of the lobster (*Homarus americanus*) on the coast of the maritime provinces of Canada. J. Biol. Board Can. 2:41-88.

Templeman, W. 1937. Habits and distribution of larval lobsters (*Homarus americanus*). J. Biol. Board Can. 3:343-347.

Templeman, W. 1939. Investigations into the life history of the lobster (*Homarus americanus*) on the west coast of Newfoundland, 1938. Newfoundland Dep. Nat. Resour. Res. Bull. (Fish) 7. 52 pp.

Templeman, W. 1940. Lobster tagging on the west coast of Newfoundland, 1938. Newfoundland Dep. Nat. Resour. Res. Bull. (Fish) 8. 16 pp.

Thomas, J.C. 1973. An analysis of the commercial lobster (*Homarus americanus*) fishery along the coast of Maine August 1966 through December 1970. NOAA-NMFS Tech. Rept. SSRF-667. 57 pp.

Uzmann, J.R., R.A. Cooper, and K.J. Pecci. 1977. Migrations and dispersion of tagged American lobsters, *Homarus americanus*, on the southern New England Continental Shelf. NOAA Tech. Rep. NMFS SSRF-705. 92 pp.

VAST. 1972. A study of water currents at intakes of Unit No. 1. Prepared for Millstone Point Co. Rept. No. 67-83-06-72A. 11pp.

Wilder, D.G. 1953. The growth rate of the American lobster (*Homarus americanus*). J. Fish. Res. Board Can. 10:371-412.

Wilder, D.G. 1963. Movements, growth and survival of marked and tagged lobsters liberated in Egmont Bay, Prince Edward Island. J. Fish. Res. Board Can. 20:305-318.

Wilder, D.G., and R.C. Murray. 1958. Do lobsters move offshore and onshore in the fall and spring? Fish. Res. Board Can. Atl. Prog. Rep. 69:12-15.

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Rocky Intertidal Studies

Introduction

Rocky intertidal habitat is extensive along most of New England's coastline, and supports a rich and diverse community of attached algae and animals. Owing to its location and relative immobility, this community is exposed to a number of environmental stresses, including the heated effluent from coastal power plants. Studies of rocky shore communities are commonly included in ecological monitoring programs designed to assess the impacts of those power plants (Vadas et al. 1976, 1978; Wilce et al. 1978; NAESCO 1994; NAI 1996; NUSCO 1996).

Rocky intertidal studies at Millstone Nuclear Power Station (MNPS) are part of an extensive environmental monitoring program whose primary objective is to determine whether differences (*e.g.*, in abundance, distribution or species composition) 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 algae and animal species found on nearby rocky shores, to describe temporal and spatial patterns of occurrence and abundance of these organisms, and to identify physical and biological factors that induce variability in the 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 most recent 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 to present). The program has undergone considerable modification over the past 17 years, with the most recent change involving the reduction in number of sampling sites from nine to four; this report emphasizes results from the currently sampled stations, but refers to previous studies (*e.g.*, NUSCO 1996) where appropriate.

Materials and Methods

Qualitative Sampling

Qualitative algal collections were made during odd-numbered months at four rocky intertidal stations (Fig. 1). These stations are, in order of most to least exposed to prevailing winds and storm forces: Fox Island-Exposed (FE), Millstone Point (MP), White Point (WP), and Giants Neck (GN). The MP station was added in September 1981; FE, WP and GN have been sampled since March 1979. Prior to 1994, qualitative collections were made monthly, but as current procedures call for sampling only in odd-numbered months, only those months from historical data are included. A year of qualitative sampling is determined to be from March to the following January, *i.e.*, the latest year of qualitative algal data (1996) comprises collections from March 1996 to January 1997. The 1985 sample year (3/85 - 1/86) terminated the 2-unit operational period; the 1986 sample year (3/86 - 1/87) 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); MP and WP are 300 and 1700 m from the discharges, respectively, and potentially impacted by the plume. The GN station is about 6.5 km west of Millstone Point, and unaffected by MNPS operation.

Qualitative collections were used to characterize the attached flora at each site during each sampling period. 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 recorded from MNPS sampling stations. Excluded from these lists are diverse diatom taxa, cyanobacteria and some crustose, endophytic or endozooic algal species. These elements of the microbiota are present but too

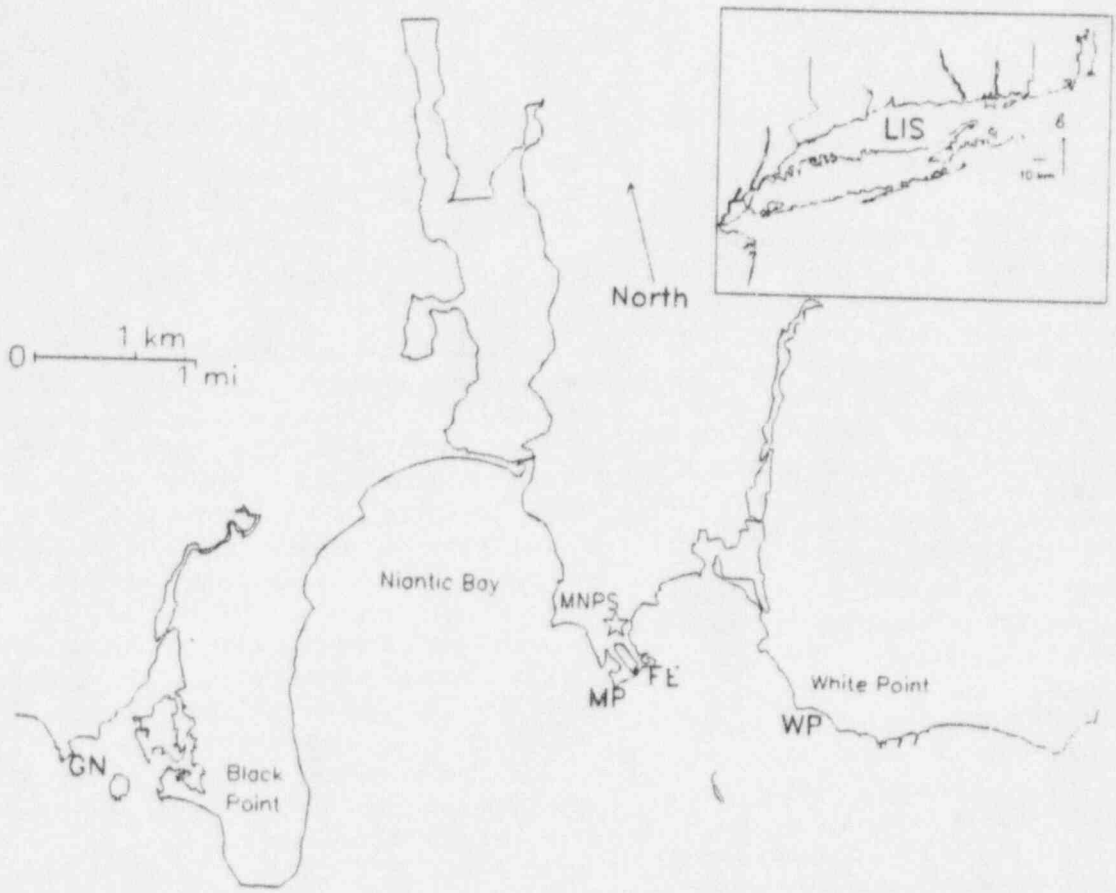


Fig. 1. Location of the MNPS rocky intertidal sampling sites: GN=Giants Neck, MP=Millstone Point, FE=Fox Island-Exposed, WP=White Point.

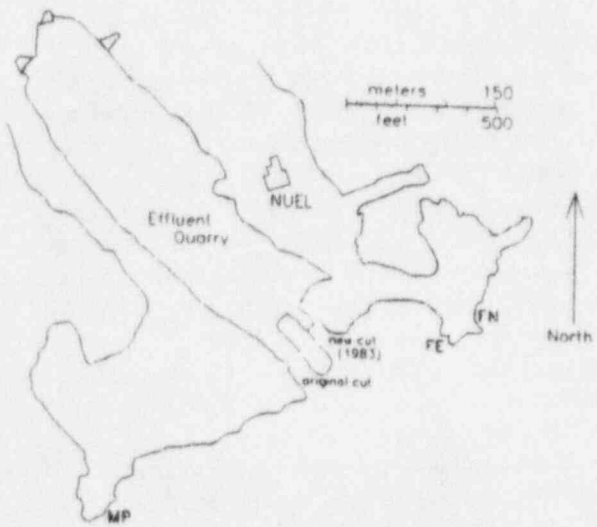


Fig. 2. Detail map of the MNPS vicinity: FN=new experimental *Ascophyllum* site (1985-present); MP and FE as in Fig. 1.

difficult to consistently collect, and, for many species, to identify as components of a large-scale environmental program. Also included in our lists are taxa that 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), as updated by Villalard-Bohnsack (1995).

Abundance Measurement

Abundance of rocky intertidal organisms was expressed as percentage of substratum cover. At each qualitative collection station, 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 non-destructively sampled six times per year, in odd numbered months. 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 *Ascophyllum nodosum*, a perennial brown alga, were studied at two reference stations (GN, 6.5 km west of the discharge and WP, 1.7 km east of the discharge, Fig. 1) and a potentially impacted 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* had been monitored earlier, at a site *ca.* 75 m east of the original Millstone quarry cut (FO), from 1979 to 1984. This *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).

Upright shoots, or fronds, of *Ascophyllum* were measured monthly, after onset of new vesicle formation, from April to the following April. At each station, fifty fronds were marked at their bases with a numbered plastic tag, and five apices on each individual 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 individuals. Tags lost to thallus breakage were not replaced, and the pattern of loss was used as a measure of mortality. Loss of the entire frond 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 annual 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 (α= -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. Substrata not occupied by macrobiota were classed as free space. Cover values of selected species were plotted against time. Similarities of communities (represented as annual collections at each station) 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/year groupings.

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 (estimate of length at the end of 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). The α parameter was 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 (1995-1996) were plotted for all stations together and for each station separately, with summaries of 2-unit (1979-1986) and 3-unit (1986-1996) 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

Macroalgal communities in the vicinity of MNPS may be exposed to elevated water temperatures resulting from the thermal effluent discharge. Because temperature is important in determining macroalgal species occurrence and distribution (Lüning 1990), alterations of patterns of spatial and temporal species occurrence are likely. The current qualitative algal sampling program is used to monitor these patterns by applying various floristic analyses to data compiled from periodic algal collections.

Qualitative algal sampling results are presented in Table 1 as percent frequency of species occurrence by month, and during 3-unit and 2-unit operational periods by station. The total number of species collected and identified in 1996 was 101. This total was within the range of annual totals for 2-unit (81-111; period total of 136) and 3-unit (88-105; period total of 141) periods. No new species were noted in 1996. By restricting our analyses to only the four currently sampled stations, 150 algal species have been collected since 1979; this total is only slightly lower than the 160 species reported last year (NUSCO 1996), when all nine stations were sampled. Of the ten 'lost' species, only *Laminaria digitata* had occurred as anything other than a trace component of our flora; *L. digitata* had been relatively common, but only at the Twotree Island (TT) sampling site.

One effect of a warm-water discharge on the local macroalgal community may be seen as a shift in seasonal occurrence of species characteristically found in 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, then use this information as a baseline from which power plant-induced changes can be assessed. In the Millstone area, a characteristic suite of species, typical of cold-water period (January-May) collections, includes *Dumontia contorta, Polysiphonia stricta, Spongonema tomentosum, Desmarestia viridis, Chorda tomentosa, Ulothrix flacca, Urospora penicilliformis, Monostroma grevillei, Protomonostroma undulatum* and *Spongomorpha arcta* (Table 1). An equally distinctive group of species characteristic of warm-water (July-November) collections includes *Champia parvula, Lomentaria baileyana, Callithamnion roseum, Ceramium diaphanum, Grinnellia americana, Dasya baillouviana, Hincksia mitchelliae, Enteromorpha clathrata, Bryopsis plumosa* and *B. hypnoides*.

Shifts in natural occurrence patterns related to thermal plume exposure 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) can 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 study site nearest the discharge, FE. For example, two cold-water red algae, *Dumontia contorta* and *Polysiphonia stricta*, were common components of the winter/spring flora at FE during 2-unit operation, occurring in 26% of collections there prior to Unit 3 start-up. However, *D. contorta* has not yet been collected at FE during 3-unit operation (Table 1), and *P. stricta* has occurred in only 2% of the 3-unit collections. Other cold-water species (*Ulothrix flacca, Monostroma grevillei, Protomonostroma undulatum* and *Spongomorpha arcta*) occur occasionally at FE during the 3-unit period, but much less frequently than during the 2-unit period. 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 americana, Dasya baillouviana, Hincksia mitchelliae*, and *Bryopsis hypnoides*.

Occurrence patterns of perennial species at FE have also changed during 3-unit operation. Specifically, we have documented the establishment of populations of species with geographical distributions which extend into warm temperate and

TABLE 1. Qualitative algal collections (Mar. 1979 - Jan. 1997) by month, and by station during 2-unit (3/79-1/86) and 3-unit (3/86-1/97) operating periods. Values represent number of times found, as a percentage of possible times found. A dash before a species indicates that it 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 detail. The last three columns represent 2-unit, 3-unit, and overall study summaries ('T' =present, but <1%).

	by month						2-unit				3-unit				Summaries		
Rhodophyta	J	M	M	J	S	N	FE	GN	MP	WP	FE	GN	MP	WP	2-U	3-U	tot
-Stylonema alsidii	4	3	0	7	23	7	12	12	0	10	9	8	0	9	9	6	7
Porphyrostromium ciliare	29	14	14	10	36	38	31	29	17	29	35	27	3	21	27	22	24
Erythrotrichia carnea	3	3	3	1	12	6	0	7	0	2	11	5	5	3	3	6	5
Erythrocladia subintegra	1	0	0	0	1	4	2	0	0	0	3	0	3	0	1	2	1
Erythropeltis discigera	3	1	0	1	6	7	10	5	0	0	6	0	2	3	4	3	3
-Bangia atropurpurea	70	86	36	12	30	49	33	43	50	50	48	50	55	44	43	49	47
-Porphyra leucosticta	71	75	58	19	10	30	31	36	46	29	50	52	56	41	34	50	44
-Porphyra umbilicalis	55	80	88	51	30	41	52	69	46	62	76	33	74	44	59	57	57
-Porphyra linearis	6	6	1	0	0	0	0	0	0	0	0	0	11	3	0	3	2
Audouinella purpurea	3	0	3	1	1	3	14	0	0	2	0	0	2	0	5	T	2
-Audouinella secundata	28	23	25	20	16	14	24	33	33	24	23	14	21	11	28	17	21
Audouinella daviesii	4	3	3	4	1	3	2	7	4	5	3	2	2	3	5	2	3
-Audouinella saviana	10	16	19	4	13	17	12	17	21	10	18	17	3	14	14	13	13
Audouinella sp.	0	0	0	1	0	0	0	2	0	0	0	0	0	0	1	0	T
-Gelidium pusillum	30	20	16	20	23	32	0	7	0	0	45	83	0	15	2	36	24
Nemalion helminthoides	0	0	0	6	0	0	0	0	0	0	0	0	6	0	0	2	1
Bonnemaisonia hamifera	1	4	9	13	0	1	0	0	0	19	0	2	2	15	5	5	5
-Agardhiella subulata	22	17	12	20	26	23	26	5	8	21	82	2	0	6	16	22	20
Polyides rotundus	3	3	6	10	9	7	5	2	13	21	5	2	5	6	10	4	6
-Cystoclonium purpureum	71	58	71	49	17	49	57	62	67	64	5	62	55	68	62	47	53
Gracilaria tikvahiae	13	6	3	1	10	9	2	0	0	0	41	0	0	2	1	11	7
-Ahnfeltia plicata	35	35	33	36	23	30	71	24	50	52	3	3	42	41	49	22	32
-Phyllophora pseudoceranoides	25	9	4	10	4	14	10	14	8	40	3	3	5	15	19	6	11
-Coccotylus truncatus	7	13	10	6	6	9	5	10	13	21	2	2	5	18	12	6	8
-Chondrus crispus	96	96	96	97	96	96	79	100	100	100	88	100	100	100	94	97	96
-Mastocarpus stellatus	57	51	49	52	51	65	19	36	96	69	0	29	97	100	50	56	54
Rhodophysema georgii	0	0	1	0	0	0	0	0	0	2	0	0	0	0	1	0	T
-Corallina officinalis	74	67	70	70	72	72	98	5	100	83	95	2	95	97	68	72	71
-Dumontia contorta	23	70	75	9	1	1	26	40	38	40	0	44	30	32	36	27	30
Gloisiphonia capillaris	1	1	4	0	0	0	2	0	4	0	3	0	0	2	1	1	1
-Choreocolax polysiphoniae	12	12	6	9	1	0	5	14	0	2	0	20	6	2	6	7	7
Hildenbrandia rubra	0	0	0	1	1	0	0	2	0	0	0	0	0	2	1	T	T
-Palmaria palmata	14	22	22	22	6	14	10	36	13	40	5	11	12	18	26	11	17
-Champia parvula	26	12	4	54	72	59	33	38	8	64	41	23	33	52	39	37	38
-Lomentaria baileyana	3	0	0	9	52	7	19	21	4	19	15	9	0	11	17	9	12
-Lomentaria clavellosa	6	7	6	1	3	4	2	7	0	7	0	2	2	15	5	5	5
-Lomentaria orcadensis	1	1	0	0	4	0	0	2	0	5	2	0	0	2	2	1	1
-Antithamnion cruciatum	30	1	10	49	43	45	29	48	33	64	20	17	17	33	45	22	30
-Antithamnion pectinatum	54	29	23	41	54	55	0	0	0	0	73	45	85	64	0	67	43
Callithamnion corymbosum	0	0	0	0	3	3	0	0	0	7	2	0	0	0	2	T	1
-Callithamnion roseum	4	3	1	9	32	19	21	10	17	10	30	3	2	5	14	10	11
-Callithamnion tetragonum	36	19	13	12	23	33	48	40	54	48	2	2	20	14	47	9	23
Callithamnion byssoides	0	0	0	4	0	0	0	2	0	0	0	3	0	0	1	1	1
-'Callithamnion baileyi'	26	12	3	26	23	38	0	2	0	2	5	39	50	36	1	33	21
-Ceramium deslongchampii	3	0	1	0	4	3	0	5	0	0	0	5	0	5	1	2	2
-Ceramium diaphanum	1	0	0	25	38	9	5	17	4	24	6	9	5	26	13	11	12
-Ceramium nodulosum	86	81	87	93	86	86	88	88	88	95	71	91	80	94	90	84	86
Ceramium cimbricum	0	0	0	0	1	0	0	0	4	0	0	0	0	0	1	0	T

TABLE 1. (cont.)

Rhodophyta	by month						2-unit				3-unit				Summaries		
	J	M	M	J	S	N	FE	GN	MP	WP	FE	GN	MP	WP	2-U	3-U	tot
-Spermothamnion repens	52	26	20	38	43	57	17	50	33	69	8	52	26	64	43	37	39
-Spyridia filamentosa	1	0	0	3	19	9	0	14	0	0	0	20	0	5	4	6	5
-Scagelia pylaisaei	1	0	3	1	1	3	0	2	4	7	2	0	0	2	3	1	2
Griffithsia globulifera	0	0	0	0	4	0	0	0	0	0	0	0	0	5	0	1	1
-Grinnellia americanum	3	1	0	7	13	19	2	0	0	12	26	0	0	11	4	9	7
-Phycodrys rubens	0	4	7	3	4	3	2	0	0	19	0	0	0	9	6	2	4
-Dasya baillouviana	12	0	0	17	46	36	12	10	4	17	42	23	8	18	11	23	19
-Chondria sedifolia	0	0	0	3	7	0	0	5	0	0	0	6	0	2	1	2	2
-Chondria baileyana	1	1	1	1	19	1	0	7	17	5	2	8	5	0	6	3	4
Chondria capillaris	0	0	0	1	6	0	0	7	0	0	0	3	0	0	2	1	1
-Polysiphonia denudata	4	0	1	3	3	3	2	0	0	5	8	0	2	2	2	3	2
-Polysiphonia harveyi	36	16	14	46	51	39	64	52	58	57	32	12	20	17	58	20	34
-Polysiphonia lanosa	78	72	74	70	70	71	45	88	100	79	0	85	100	95	75	71	72
-Polysiphonia nigra	6	14	20	0	0	1	0	0	0	14	11	9	0	15	4	9	7
-Polysiphonia fucoides	26	14	16	28	30	23	14	19	4	55	11	24	2	50	25	22	23
-Polysiphonia stricta	28	42	58	13	6	3	26	43	29	38	2	18	23	35	35	19	25
Polysiphonia elongata	1	0	0	0	0	0	0	0	0	0	0	0	0	2	0	T	T
Polysiphonia fibrillosa	3	0	0	1	1	0	2	5	0	0	0	0	2	0	2	T	1
Polysiphonia flexicaulis	7	0	0	0	0	1	0	0	0	0	2	2	2	5	0	2	1
-Polysiphonia novae-angliae	91	61	45	91	88	94	69	64	71	71	100	77	77	82	69	84	79
-Rhodomela confervoides	3	13	6	0	0	0	7	10	0	10	0	0	3	3	7	2	4

Phaeophyta	by month						2-unit				3-unit				Summaries		
	J	M	M	J	S	N	FE	GN	MP	WP	FE	GN	MP	WP	2-U	3-U	tot
-Ectocarpus fasciculatus	4	14	33	12	22	22	26	12	38	24	3	20	15	21	23	15	18
-Ectocarpus siliculosis	17	46	52	48	29	32	40	50	25	43	21	45	29	45	41	35	37
Ectocarpus sp.	3	4	0	3	3	0	5	2	13	2	0	2	2	0	5	1	2
Hincksia granulosa	3	6	13	3	1	4	2	5	8	5	14	5	2	2	5	5	5
-Hincksia mitchelliae	16	7	20	22	43	26	36	12	8	19	47	21	9	18	20	24	22
-Pilayella littoralis	12	20	42	10	9	7	7	57	8	21	2	33	0	12	25	12	17
-Spongonema tomentosum	12	39	25	0	0	0	14	19	17	7	11	12	9	15	14	12	13
Acinetospora sp.	0	0	1	0	1	0	0	0	0	0	0	0	0	3	0	1	T
-Ralfsia verrucosa	55	52	49	68	71	67	64	71	50	69	11	77	67	76	65	58	60
-Elachista fucicola	57	70	81	83	68	57	60	71	63	64	59	76	73	79	65	72	69
Halothrix lumbricalis	0	4	4	0	1	0	0	2	0	5	0	3	2	2	2	2	2
-Leathesia difformis	0	0	49	57	0	0	14	10	33	14	8	27	21	18	16	19	18
-Chordaria flagelliformis	0	1	23	29	16	1	5	14	29	21	3	6	18	11	16	9	12
Sphaerotrichia divaricata	0	0	3	0	0	0	0	0	0	2	0	0	2	0	1	T	T
Eudesme virescens	0	0	1	0	0	0	0	0	0	0	0	0	0	2	0	T	T
Pogotrichum filiforme	0	4	3	0	1	0	0	5	0	7	0	0	0	2	3	T	1
Phaeosaccion collinsii	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0	T	T
-Punctaria tenuissima	0	12	3	1	0	0	0	5	0	2	3	5	0	5	2	3	3
-Punctaria latifolia	6	10	6	1	0	1	2	0	4	7	8	3	2	6	3	5	4
Punctaria plantaginea	1	3	4	7	3	3	0	7	8	7	0	9	0	2	5	3	4
-Petalonia fascia	75	86	93	48	3	41	55	55	54	76	44	59	58	62	61	56	57
-Scytosiphon lomentaria	33	97	96	71	0	13	45	74	50	62	42	48	52	48	59	48	52
-Desmarestia aculeata	3	6	9	10	3	12	2	0	4	17	0	9	2	20	6	8	7
-Desmarestia viridis	0	25	48	3	0	0	5	10	21	14	3	12	15	23	11	13	13
Chorda filum	0	0	9	14	0	0	0	2	0	14	2	0	0	12	5	3	4
-Chorda tomentosa	0	4	10	3	0	0	5	0	0	10	0	0	2	8	4	2	3
-Laminaria longicruris	14	6	13	9	7	7	0	10	17	12	2	8	8	23	9	10	9
-Laminaria saccharina	49	52	84	83	58	52	60	74	63	64	39	59	74	74	65	62	63
-Sphacelaria cirrosa	54	29	39	39	45	52	57	48	13	17	83	67	9	29	36	47	43
Sphacelaria rigidula	0	0	0	0	1	3	0	0	0	2	0	3	0	0	1	1	1

TABLE 1. (cont.)

	by month						2-unit				3-unit				Summaries		
Phaeophyta	J	M	M	J	S	N	FE	GN	MP	WP	FE	GN	MP	WP	2-U	3-U	tot
-Ascophyllum nodosum	91	94	94	94	91	91	79	100	100	100	68	100	100	100	94	92	93
-Fucus distichus s edentatus	7	13	6	1	4	0	12	5	13	0	2	2	12	3	7	5	5
-Fucus distichus s evanescens	3	12	9	1	0	3	10	14	0	10	0	2	5	2	9	2	5
-Fucus spiralis	9	4	9	9	10	6	0	7	4	2	0	5	36	0	3	10	8
-Fucus vesiculosus	97	99	99	100	97	97	81	100	100	100	100	100	100	100	95	100	98
-Sargassum filipendula	13	12	12	12	13	13	0	0	0	0	77	0	0	0	0	19	12
	by month						2-unit				3-unit				Summaries		
Chlorophyta	J	M	M	J	S	N	FE	GN	MP	WP	FE	GN	MP	WP	2-U	3-U	tot
-Ulothrix flacca	41	68	36	6	7	20	38	40	29	45	14	32	30	21	39	24	30
-Urospora penicilliformis	68	75	25	3	6	32	31	36	42	38	30	33	33	39	36	34	35
-Urospora wormskjoldii	14	19	10	9	1	3	10	2	4	7	23	8	5	11	6	11	9
'Urospora collabens'	4	3	3	3	0	0	5	5	4	2	3	0	0	2	4	1	2
Acrochaete viridis	1	0	0	1	0	0	0	0	4	0	0	0	0	2	1	T	T
-Protomonostroma undulatum	9	74	74	1	0	1	26	40	25	36	8	29	27	29	33	23	27
-Monostroma grevillei	7	48	41	0	1	3	24	29	21	19	2	15	18	17	23	13	17
-Monostroma oxyspermа	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0	T	T
-Spongomorpha arcta	7	38	64	9	0	1	19	26	33	19	8	17	27	20	23	18	20
-Spongomorpha aeruginosa	3	6	46	3	1	0	7	12	17	7	2	12	17	9	10	10	10
'Codiolum gregarium'	0	0	0	1	0	0	0	2	0	0	0	0	0	0	1	0	T
Capsosiphon fulvescens	0	0	1	1	1	0	0	2	0	5	0	0	0	0	2	0	1
Capsosiphon groenlandicum	0	4	0	1	0	1	2	2	8	2	0	0	0	0	3	0	1
-Blidingia minima	70	72	84	67	72	61	71	62	67	57	85	56	85	74	64	75	71
Blidingia marginata	1	0	0	6	0	1	5	2	0	2	0	3	0	0	3	1	1
-Enteromorpha clathrata	4	1	10	35	41	6	29	24	4	29	11	15	3	20	23	12	16
-Enteromorpha flexuosa	54	52	55	58	68	72	57	36	33	60	88	58	58	64	48	67	60
-Enteromorpha intestinalis	17	30	42	49	38	16	38	52	38	60	17	27	26	23	48	23	32
-Enteromorpha linza	68	62	87	83	86	75	64	50	75	71	89	76	88	83	64	84	77
-Enteromorpha prolifera	22	20	25	20	29	23	31	48	38	48	8	17	5	23	41	13	23
Enteromorpha torta	1	0	4	6	0	0	0	5	0	2	0	3	0	5	2	2	2
Enteromorpha ralfsii	0	0	0	10	4	0	2	5	0	5	2	5	0	2	3	2	2
Percursaria percursa	0	0	1	1	3	0	0	2	0	5	0	0	0	2	2	T	1
-Ulva lactuca	96	81	91	97	96	97	98	98	96	95	83	92	94	94	97	91	93
-Prasiola stipitata	16	14	14	19	19	17	0	48	4	0	2	71	0	0	14	18	17
-Chaetomorpha linum	46	22	38	84	80	67	71	50	88	74	14	64	61	58	69	49	56
Chaetomorpha melagonium	0	0	1	0	0	0	0	0	0	0	0	2	0	0	0	T	T
-Chaetomorpha aerea	65	45	49	57	58	61	71	24	42	50	71	48	62	61	47	61	56
-Cladophora albida	0	1	7	9	10	0	7	5	4	17	2	5	0	3	9	2	5
-'Cladophora flexuosa'	12	1	22	49	33	7	14	2	29	24	17	21	27	29	16	23	21
'Cladophora glaucescens'	0	0	1	1	0	0	0	0	4	2	0	0	0	0	1	0	T
Cladophora laetevirens	0	0	0	1	1	0	0	0	4	2	0	0	0	0	1	0	T
-'Cladophora refracta'	9	4	7	13	16	4	17	7	25	21	6	2	3	8	17	5	9
-Cladophora sericea	13	12	51	43	29	25	48	24	17	40	26	26	24	27	34	26	29
'Cladophora crystallina'	0	0	0	0	3	0	0	0	0	0	0	2	0	2	0	1	T
-Cladophora hutchinsiae	6	4	7	7	7	12	12	5	8	12	3	8	11	3	9	6	7
-Cladophora rupestris	1	1	6	9	4	4	2	5	0	2	9	2	6	5	3	5	4
-Cladophora ruchingeri	1	1	4	16	14	6	0	0	0	0	23	2	6	15	0	11	7
-Rhizoclonium riparium	9	16	17	36	22	4	19	31	4	26	6	32	5	17	22	15	17
'Rhizoclonium kerneri'	0	3	1	1	0	0	0	2	0	0	0	3	0	2	1	1	1
'Rhizoclonium tortuosum'	0	0	0	0	0	1	2	0	0	0	0	0	0	0	1	0	T
-'Bryopsis plumosa'	12	0	3	7	14	12	21	10	0	5	20	2	2	5	10	7	8
-Bryopsis hypnoides	4	0	9	19	6	9	5	5	4	0	18	9	2	12	3	10	8
Derbesia marina	6	3	1	1	1	6	24	5	0	0	2	0	0	0	8	T	3
-Codium fragile	94	81	86	93	97	93	100	83	92	81	98	82	88	98	89	92	91

tropical regions, and are therefore tolerant of 3-unit temperature regimes at FE (*e.g.*, *Gracilaria tikvahiae*, *Agardhiella subulata* and *Sargassum filipendula*; Taylor 1957; Lüning 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.

Other changes to the overall flora that occurred during the 3-unit period appear unrelated to power plant operation. Increased occurrence of *Gelidium pusillum* has been observed at FE (absent from collections during 2-unit operation, but present in 45% of the 3-unit collections). This increase of *G. pusillum* also occurred at sites farther from the discharge, including a pronounced increase (from 7% to 83%) at our control site GN. The area-wide introduction of an exotic species, *Antithamnion pectinatum*, also occurred during 3-unit operation; it was not collected prior to 1986, but has been found in 67% of the 3-unit collections.

Community analyses, based on annual collections at each station, also reflect both site-specific and area-wide changes to the algal flora. For instance, groupings of collections at GN, MP and WP (Figs. 3a, b and c, respectively) separate into early and later sampling years, with the point of separation around 1986-1988. This separation was certainly influenced by the increasing contribution in recent years of species like *Antithamnion pectinatum*, mentioned above. However, even with such floristic separation, all annual collections at these three stations (GN, MP, WP) clustered at greater than 50 - 60% similarity, indicating a high degree of consistency in the year-to-year floral assemblages at these sites.

In contrast, the overall similarity of annual collections at FE (Fig. 3d) was only 35%; four groupings are apparent at about the 60% similarity level. The first group represents collections made during 2-unit 1-cut operational years (1979-82), when the unimpacted flora at FE was similar to that observed at other exposed stations. Temperature conditions were severely altered when the second quarry cut was opened in 1983, and account for the characteristic disturbed, or early successional stage, flora (*e.g.*, opportunistic *Enteromorpha* and *Polysiphonia* spp.) collected at FE in the second group (1983 and 1984). By 1985 and 1986 (the third group), populations of *Chondrus crispus*, *Fucus vesiculosus*, *Ascophyllum nodosum* and *Ahnfeltia plicata* and their associated epiphytes (*e.g.*, *Ceramium nodulosum*, *Protomonostroma undulatum*, *Elachista fucicola* *Cystoclonium purpureum* and *Polysiphonia lanosa*) were eliminated, while opportunistic species had become dominant. Elevated temperature conditions at FE were more consistent in subsequent years comprising collections in the fourth group (1987-96). 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). It is particularly interesting to note that the algal community that has developed at FE in response to elevated water temperatures has persisted, at least qualitatively, through 1996, despite the extended outages of all three units.

Abundance Measurement

Along with the changes of patterns in species occurrence discussed above, exposure to the MNPS thermal plume can change patterns of species abundance and community dominance hierarchy in rocky shore communities. Changes in species abundance and altered patterns of zonation at impacted sites may not be evident when descriptors of qualitative community characteristics (as presented in the previous section) are used. Therefore, quantification of species abundance and distribution patterns, through determination of percent substratum coverage, complements qualitative algal studies.

Abundance measurement studies were designed to sample species abundance over an area sufficiently large as to accurately describe large scale patterns of abundance in each intertidal zone (high, mid and low) at each sampling site. Among-station differences in abundance patterns are then related to site-specific physical and biological controlling mechanisms including, for stations near the MNPS discharge, exposure to elevated temperature regimes. Following subsections describe abundance patterns of important intertidal organisms, *i.e.*, barnacles, *Fucus*, *Chondrus* and common epiphytes, along with analyses of overall community structure.

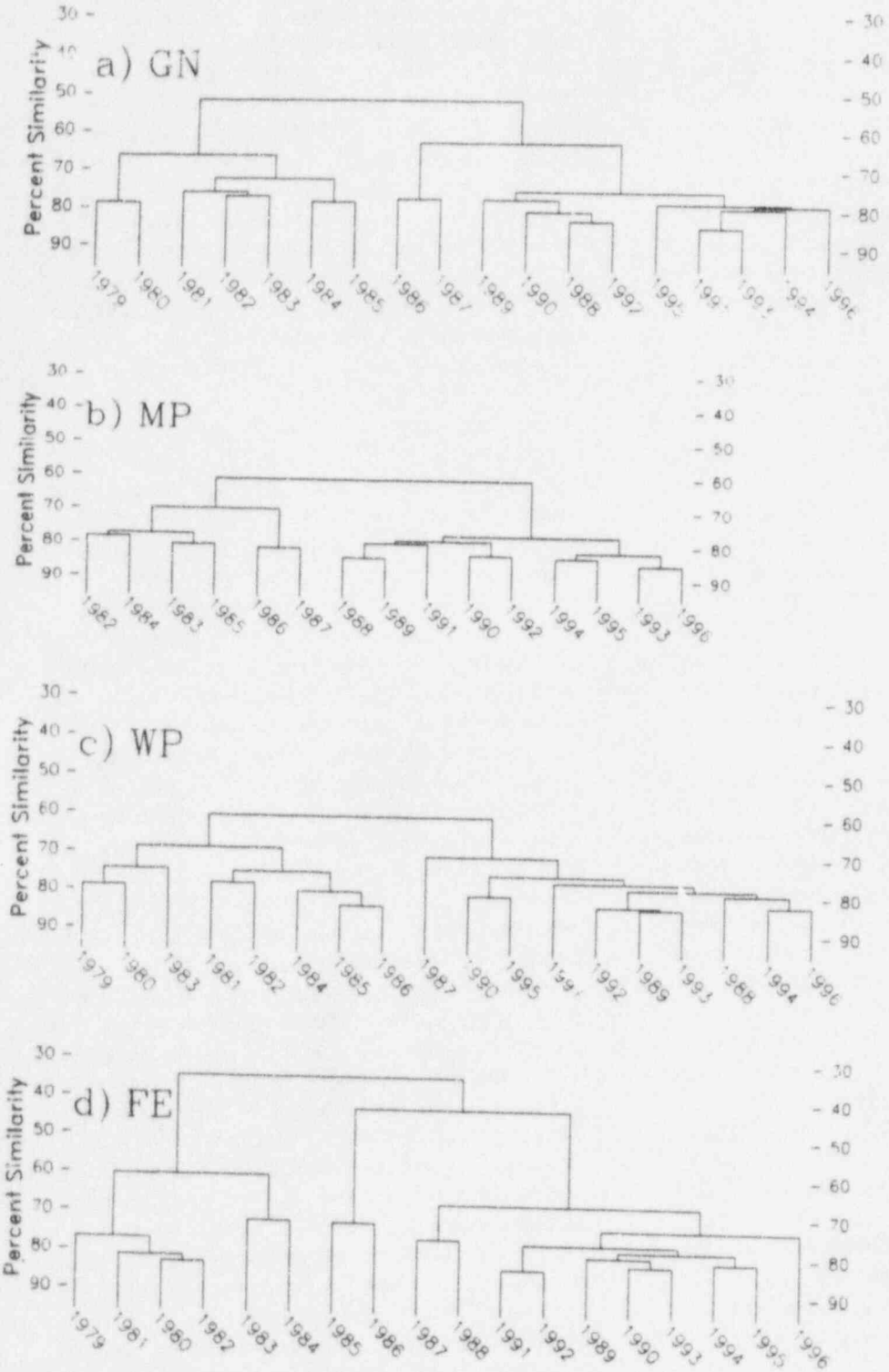


Fig. 3. Clustering dendrogram of percent similarity of qualitative algal collections, by year, at rocky intertidal sampling stations.

Barnacles

Barnacles (primarily *Semibalanus balanoides*) are the dominant invertebrate on local rocky shores, and are most abundant in the mid intertidal zone (Zone 2). Barnacles exhibit an annual pattern of abundance marked by reproduction and settlement in early spring, rapid growth and surface cover increases in summer, and decreased abundance through autumn and winter due to competition for space, predation and physical disturbance (Connell 1961; Menge 1976; Bertness 1989; NUSCO 1993).

The barnacle annual abundance cycle described above was observed at all Millstone study sites, in all three intertidal zones in 1995-96 (Fig. 4). Maximum barnacle cover in the high intertidal (Zone 1) during 1995-96 ranged from 11% (GN) to 49% (FE). Minimum coverage in Zone 1 ranged from 1% at GN to 10% at FE. In the mid intertidal (Zone 2), maximum barnacle cover was lowest at WP (43%) and highest at MP (80%); minimum cover was lowest at FE (4%) and highest at GN (38%). Low intertidal (Zone 3) maxima during 1995-96 ranged from 5% (FE) to 64% (MP). The annual minimum in Zone 3 was lowest at FE and MP (0%), and highest at GN (5%).

Seasonal barnacle abundance patterns at all study sites, except FE, have been relatively consistent, and are the result of the temporal stability of environmental conditions at these sites. Variability in these conditions among stations is considerable, however; natural site-specific factors such as degree of site exposure to wind and waves and slope of available substratum appear to be the most important mechanisms controlling barnacle cycles and patterns of zonation (NUSCO 1993).

In addition to natural factors, thermal plume effects accounted for temporal and spatial changes in barnacle abundance. Due to the influence of tides on the thermal plume, effects of thermal increase 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. Previous reports (e.g. NUSCO 1996) suggested that elevated temperatures directly impacted low intertidal barnacles by causing complete population mortality in late summer every year since Unit 3 start-up. However, absence of barnacles from Zone 3 at FE in September 1996 was not the result of thermal stress, as all three units were shut down. Rather, this loss, as well as the reduced recruitment in Zone 3, particularly since 1989, is related to long-term community development during MNPS operation. Specifically, environmental condition at FE since Unit 3 start-up have allowed for establishment of an extensive low intertidal *Codium fragile* population at FE. This population, which persisted through the 1995-96 sampling year, competitively excludes barnacles through preemption of habitat space (Underwood and Denley 1984; NUSCO 1993).

Fucus

The dominant alga on local shores, the rockweed *Fucus vesiculosus*, forms an extensive canopy over barnacles in the mid intertidal zone, and also occurs in high and low 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 species include *F. distichus* subsp. *edentatus*, *F. distichus* subsp. *evanescens* (both occur mostly subtidally) and *F. spiralis*, which occurs in the high intertidal.

Intertidal *Fucus* distribution patterns and seasonal abundance cycles in the MNPS area are similar to those reported elsewhere in New England (Lubchenco 1980, 1983; Topinka et al. 1981). At most study sites, *Fucus* abundance typically peaks annually in late summer or autumn, reflecting high recruitment and growth rates prior to and during this period (Fig. 5). Maximum abundance during 1995-96 in Zone 1 was greatest at FE (57%), with maxima at the other stations ranging from 1 to 20% (Fig. 5). Highest Zone 2 cover during 1995-96 occurred at MP (74%); maximum cover was least at WP (39%). In Zone 3, maximum *Fucus* cover during 1995-96 was greatest at GN and WP (17%); the lowest abundance peak occurred at FE (9%).

Fucus abundance patterns varied among study populations, reflecting environmental conditions unique to each site. In general, *Fucus* is most abundant on moderately exposed shores, common environments at 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

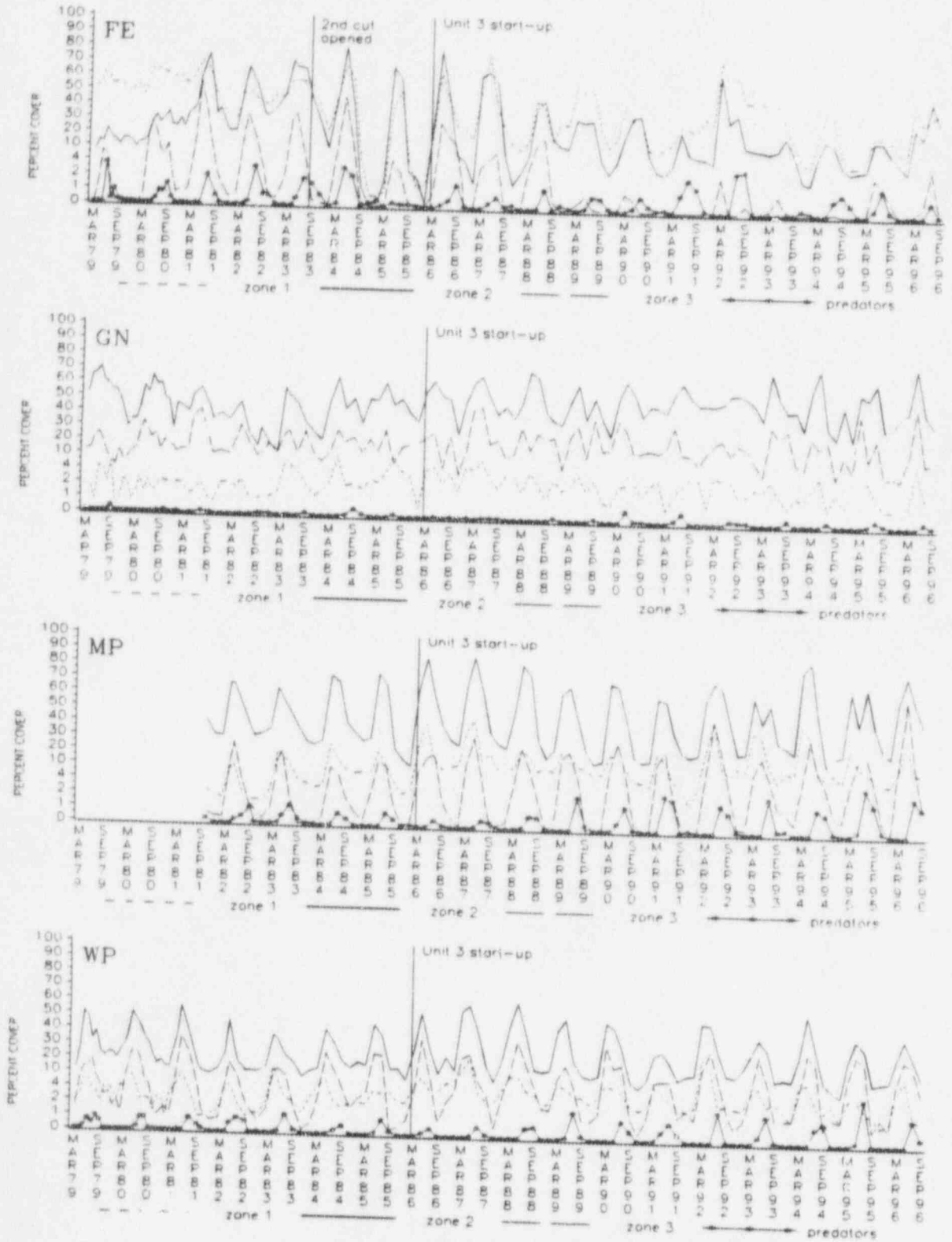


Fig. 4. Abundance of barnacles in each zone, and of predatory snails in Zone 3, of undisturbed transects, from 3/79 to 9/96.

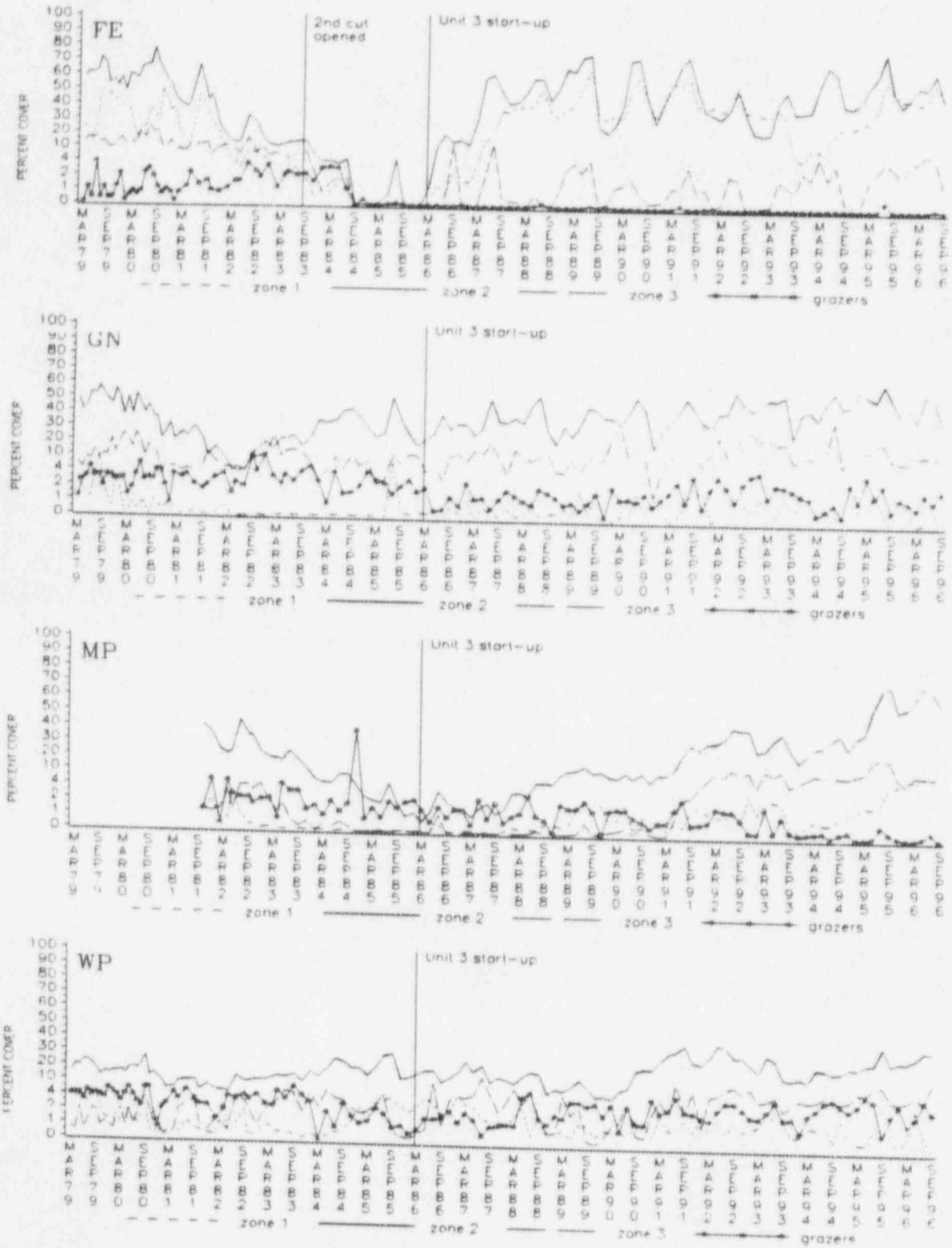


Fig. 5. Abundance of *Fucus* in each zone, and of grazing snails in Zone 3, of undisturbed transects, from 3/79 to 9/96.

and Norton 1978, 1980; Keser and Larson 1984). At our sampling sites, *Ascophyllum* is only abundant in transects at WP and GN. 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 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), except for 1996 (the extended outage of all three units permitted near-ambient conditions for much of the most recent sampling year). In other years since the opening of the second cut, 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.

Fucus has exhibited long-term abundance cycles at other study sites more distant from the discharge than FE that are likely unrelated to MNPS operation. Most notable of these is the protracted decline/recovery cycle at MP discussed in previous reports (*e.g.*, NUSCO 1994, 1995, 1996). Increases in *Fucus* abundance continued during 1995-96 to historic highs in Zones 1 and 2 at MP (Fig. 5); *Fucus* coverage in Zone 3, although down slightly from last year, remained high relative to coverage prior to 1991. Due to proximity to the MNPS discharge (*ca.* 250 m to the east) and the moderate temperature increases measured at MP (2-3°C above ambient during slack tides; NUSCO 1994) the possibility of a power plant impact has been investigated, but no direct evidence exists, linking the pattern of *Fucus* abundance at MP to power plant operation. The present *Fucus* population at FE recovered relatively rapidly after Unit 3 start-up, even under much greater temperature extremes than those at MP. *Fucus* now appears to be well established at MP under 3-unit operating conditions, occupying almost 75% of available substrata in Zone 2.

Chondrus and common epiphytes

Perennial populations of the red alga, *Chondrus crispus*, form an extensive turf on most low intertidal rock surfaces in the MNPS area. Several seasonally abundant algal taxa coexist as epiphytes on *Chondrus* (*e.g.*, *Monostroma* spp. (including *Protomonostroma*) and *Polysiphonia* spp.) instead of competing directly for primary space. Because low intertidal habitat is more susceptible to power plant impacts (as mentioned in previous sections), documentation of abundance patterns of *Chondrus* and its associated epiphytes is critical to our ecological monitoring program.

Stable *Chondrus* abundance has been documented at three of the four study sites (all but FE) during the study period. Abundance maxima at these sites during 1995-96 ranged from approximately 65 to 75%. *Chondrus* abundances observed during 1995-96 at these three sites were within historic ranges (Fig. 6).

The *Chondrus* population at FE now consists of scattered individual plants, with abundance estimates never exceeding 3% during 1995-96. 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 *Chondrus* 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 *Chondrus* thalli have been observed in upper Zone 3 study quadrats. These thalli are present 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 re-establishment of the *Chondrus* population at FE during the MNPS operational period is unlikely. Even with near-ambient water temperatures during the 3-unit shutdown, *Chondrus* did not establish or maintain appreciable substratum coverage in 1996. The low intertidal community at FE is now composed primarily of an extensive *Codium fragile* population, persistent populations of *Sargassum filipendula*, *Gracilaria tikvahiae* and ephemeral algae including *Ulva lactuca*, *Enteromorpha* spp.

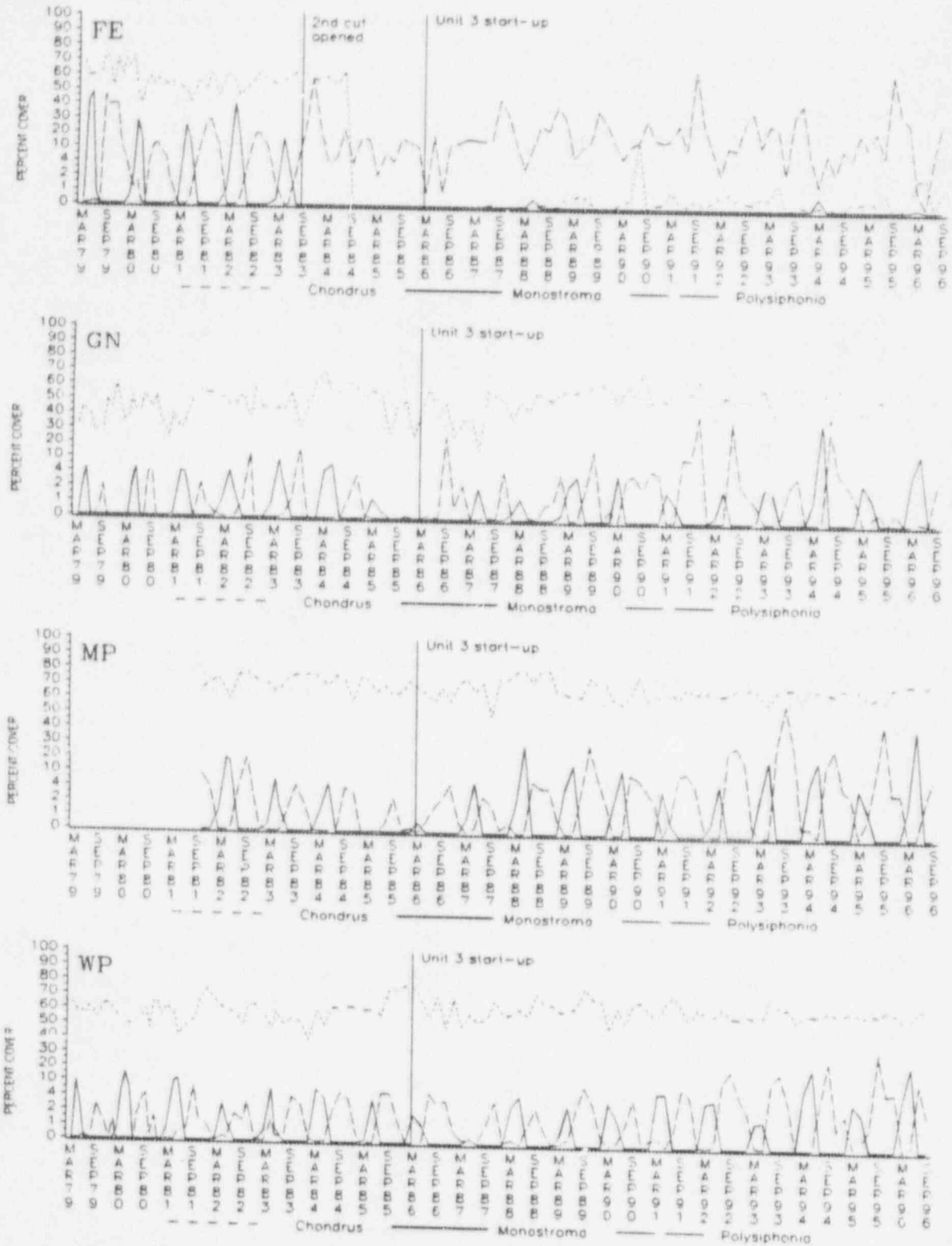


Fig. 6. Abundance of *Chondrus* and major epiphytes in Zone 3 of undisturbed transects, from 3/79 to 9/96.

and *Polysiphonia* spp. and occasionally (*e.g.*, 1995), 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 opportunity to document potential temporal shifts in abundance of these species in response to altered temperature regimes, typical of those at FE. *Polysiphonia* spp. (mostly *P. novae-angliae* and *P. harveyi*) are common warm-water epiphytes on *Chondrus*, *Ascophyllum* and *Codium*; they may also grow attached to rock. The annual abundance cycle of *Polysiphonia* spp. is characterized by a late summer peak, with cover declining to near 0% by winter at most study sites (Fig. 6). Peak abundances during 1995-96 varied considerably from station to station (3% at GN, 11% at MP, 15% at WP, and 36% at FE). The annual cycle in *Polysiphonia* spp. abundance has been consistent at all stations except at FE throughout the study period. Elevated temperature regimes at FE since the opening of the second quarry cut (1983) 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 such species are typically absent from other sites, including FE prior to 1983. Interestingly, during the 3-unit shut-down in the past year, *Polysiphonia* abundance in March and May was the lowest recorded at this site since the opening of the second cut, and was similar to this species' abundance at the other rocky shore stations.

The annual abundance cycle of *Monostroma* spp. (*M. grevillei* and *Protomonostroma undulatum*) can be described as being out of phase with 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). This annual abundance cycle occurred consistently over the study period at all study sites except FE. *Monostroma* was virtually absent from FE in 1995-96 (only 0.2% in March); peak abundance elsewhere ranged from 17% (GN) to 28% (WP) to 44% (MP). Since 1984, *Monostroma* has been observed in FE study transects only rarely, and its cover has never exceeded 1%. Prior to 1984, peak annual *Monostroma* cover at FE was similar to other exposed sites, ranging from 17% to 48%. Prolonged low water temperatures, typical of late winter/early spring at most stations, and suitable for *Monostroma*, have rarely occurred at FE under 2-cut operating conditions. However, it is likely these condition will occur at FE in spring of 1997, as all three units are expected to be shut down.

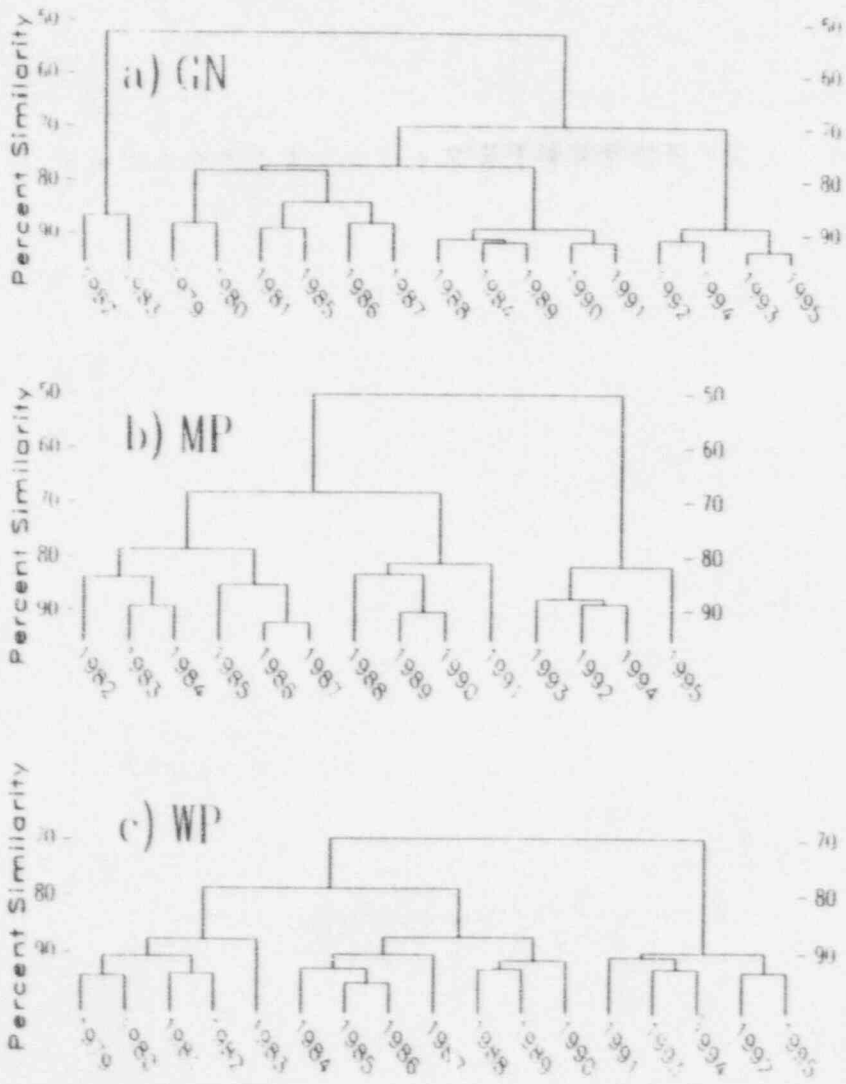
Community Analysis

Local rocky shore communities are composed of over one hundred species of attached macroalgae and sessile or slow-moving animals. The abundance and distribution of these species are influenced by complex interactions between physical processes (*e.g.*, tidal height, exposure to waves, water temperature) and biological processes (*e.g.*, inter- and intraspecific competition for light, space and nutrients, grazing and predation (including that by species not normally considered intertidal organisms, such as fish and shorebirds), growth and reproductive cycles). Characterization of these communities may be descriptive; *e.g.*, abundance of species whose populations are stable or predictably variable may be represented as time-series of percentage of substratum coverage, as in previous sections. However, comparisons among stations, or among years at a given station, may also be made using multivariate techniques, similar to those described in the Qualitative Algal section, using the abundance of all species found in the transects, even those that are rare or unpredictable in their occurrence.

Previous analyses (*e.g.*, NUSCO 1996) have shown that of the rocky shore stations, only at Fox Island-Exposed was the community sampled during the 2-unit operational period appreciably different from that sampled since Unit 3 start-up. At each of the other sites, similarities were highest between operational periods at the same station, indicating a relatively consistent species composition throughout the study period.

These findings are supported by the most recent sampling data as well, where Bray-Curtis similarity matrices, using annual average abundances of all taxa found in mid and low intertidal zones at each station, are illustrated as clustering dendrograms (Fig. 7). At all sites but FE (*i.e.*, GN, MP and WP; Figs. 7a, b and c), all years grouped together at a 50-70% similarity level, with most of the differences attributable to changes in *Fucus* cover.

Annual samples at FE (Fig. 7d) were much more dissimilar than those at other stations; at the 50% similarity level, years grouped into three



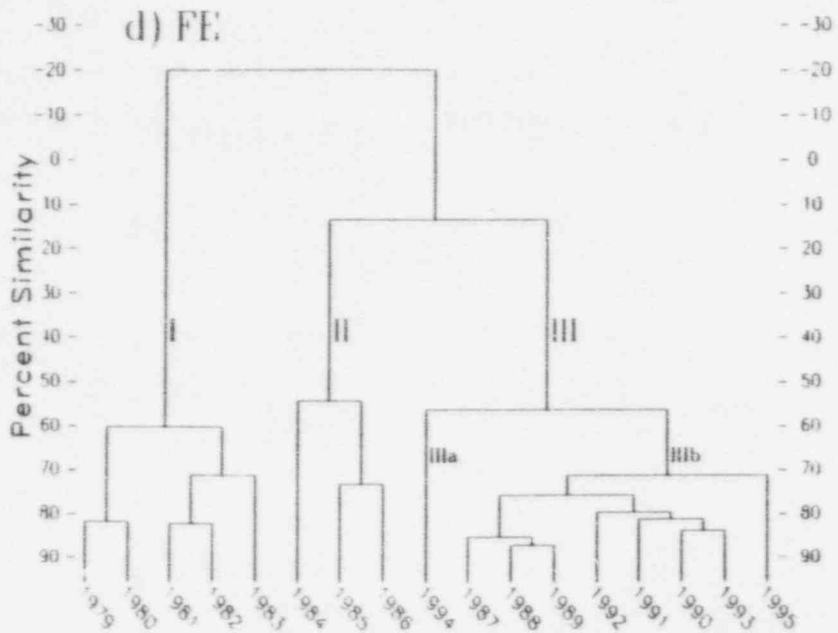


TABLE 2. Average percent substratum coverage of taxa with mean overall (Fox Island - Exposed only, all years) abundance >2%, in groupings determined by community analysis; group numbers correspond to those in Fig. 7d.

taxon	Group I	Group II	Group III	Group IIIa	Group IIIb
Codium fragile	0.95	24.98	33.41	12.21	36.06
Fucus vesiculosus	24.33	2.76	25.15	27.50	24.86
Semibalanus balanoides	18.94	15.29	9.29	5.27	9.80
rock	16.91	16.04	3.82	5.25	3.64
Chondrus crispus	30.28	4.88	0.39	0.18	0.42
Polysiphonia harveyi	5.90	6.03	11.83	4.77	12.72
Enteromorpha flexuosa	0.92	14.54	5.34	5.68	5.29
Enteromorpha linza	1.23	6.31	9.00	6.16	9.35
Mytilus edulis	0.34	2.55	11.43	34.86	8.51
Ulva lactuca	3.09	3.00	5.72	3.54	5.99

Fig. 7. Clustering dendrogram of percent similarity of undisturbed communities, by year, at rocky intertidal sampling stations: a) Giants Neck, b) Millstone Point, c) White Point, d) Fox Island - Exposed

distinct clusters. Group I, comprising 1979 to 1983, represents the mid and low intertidal community prior to the opening of the second quarry cut. This community (Table 2) was characterized by high coverage by *Chondrus*, *Fucus* and barnacles, with an appreciable amount of available free space (rock); it was typical of the communities found at nearby unimpacted sites. After the opening of the second cut (August 1983), elevated water temperatures resulting from 2-cut 2-unit operating conditions drastically altered the community at FE (NUSCO 1985). This altered community is represented by Group II (Fig. 7d, Table 2), characterized by sharply reduced populations of *Chondrus* and *Fucus*, and increases of *Codium* and *Enteromorpha* spp.

Following start-up of Unit 3 in 1986, further changes to the rocky shore community were noted at FE. Conditions existing during 2-cut 3-unit operation (Group III; Fig. 7d, Table 2) permitted re-establishment of an extensive *Fucus* population in the mid intertidal, although *Chondrus* in the low intertidal remained scarce, due to competitive exclusion by *Codium*, *Polysiphonia* and *Enteromorpha* spp. (*cf.* previous sections). Most years since Unit 3 began operation (Group IIIb) show a high degree of within-group similarity (>70%), indicating a relatively consistent species composition. The exception is 1994 (Group IIIa), distinguished by the anomalously large, although temporary, settlement of *Mytilus* noted last year (NUSCO 1996).

As the sample year for these quantitative community analyses begins in March, the most recent year (3/95 - 1/96) does not include data from the period since April 1996, when all three units were shut-down. Future reports will document the effects of this extended outage on the rocky shore communities of nearby sites.

Ascophyllum nodosum Studies

The status of three local populations of the brown alga *Ascophyllum nodosum* has been assessed since 1979 through monthly monitoring of plant growth and mortality. *Ascophyllum* is a key species within the MNPS ecological monitoring program and these studies, as elsewhere, document the value of this species as a sensitive indicator of local environmental conditions. In particular, *Ascophyllum* exhibits easily quantifiable responses to even slight changes in temperature, which makes this species a critical biomonitoring tool for studies of the ecological effects of thermal effluents. An extensive review of phenological, ecological and applied monitoring studies of *Ascophyllum* was presented in NUSCO (1993). The results of 1995-96 growth and mortality studies are compared with results from overall 2-unit and 3-unit operational periods, and are presented below.

Growth

The Gompertz growth model (Gendron 1989), when fitted to monthly *Ascophyllum* tip length data (Fig. 8), provides useful indicators of *Ascophyllum* population growth characteristics. Annual growth in 1995-96 (Fig. 8a), as indicated by α parameter of the model, was significantly lower ($P<0.05$) at FN during 1995-96 (78.0 mm) than growth at both GN (99.4 mm) and WP (100.2 mm). The difference between *Ascophyllum* growth at GN and WP in 1995-96 was not significant. The inflection point, a parameter of the model which identifies the time of maximum growth rate, was earlier at FN in 1995-96 (6 July) than at GN and WP (1 and 2 August, respectively).

Annual growth at GN during 1995-96 was significantly higher than growth over 2-unit operational periods (90.1 mm), but not significantly different from the 3-unit period growth (96.9 mm). The difference between growth estimates during 3-unit and 2-unit operation at GN was significant. Inflection points for 2-unit and 3-unit periods were within a day of each other (27 and 26 July, respectively). Growth during 1995-96 at WP was significantly higher than growth during both operational periods (Fig. 8c); 90.2 mm (2-unit) and 87.8 mm (3-unit). The difference between operational period estimates at WP was not significant, and operational period inflection points were within three days of each other (31 July and 28 July for the 2-unit and 3-unit periods, respectively). At FN, growth during 1995-96 was significantly lower than during the 1985-86 2-unit year (90.5 mm) and the 3-unit operational period (116.5 mm; Fig. 8d). The 3-unit mean was also significantly higher than growth during the 2-unit year. The inflection point for the 2-unit year was 18 July, and 22 July for the 3-unit period.

The among-station relationships for *Ascophyllum* growth were atypical in 1995-96,

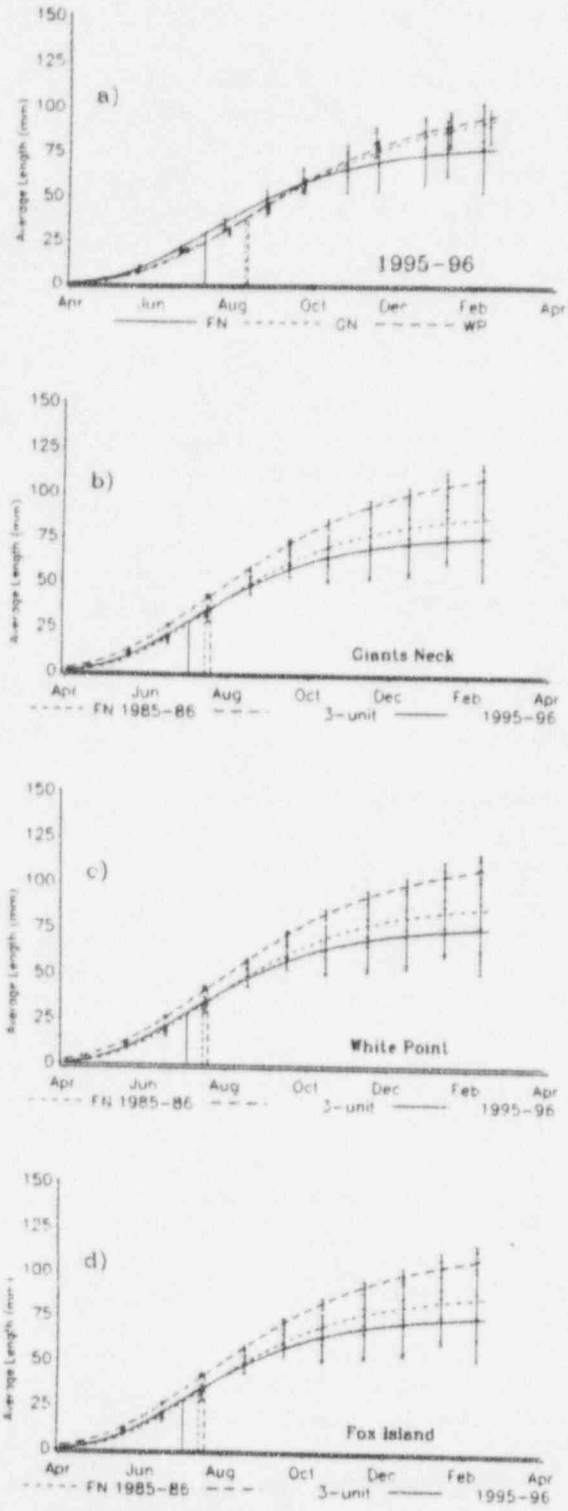


Fig. 8. *Ascophyllum* growth: a) during 1995-1996, b-d) present year, 3-unit and 2-unit operational periods at each station. Curves are the Gompertz growth model fitted to tip length data, including inflection points. Error bars represent monthly mean lengths ± 2 SE.

relative to previous 3-unit study years. For the first time in the 3-unit period, growth at FN was significantly lower than at the reference sites WP and GN. In most previous 3-unit years, growth at FN had been significantly higher than growth at WP and GN. This growth enhancement was attributed to incursion of the MNPS thermal plume to FN, which elevated water temperatures up to 3-4°C for 3-4 hours each tidal cycle. These conditions in previous years created favorable conditions for *Ascophyllum* growth by: 1) extending the period of "normal" or "ambient" peak growing conditions for local 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).

Conditions for *Ascophyllum* growth at FN during 1995-96 were not favorable, even though power plant operational status was similar to previous 3-unit years. Characteristics of the annual pattern of growth at FN observed in 1995-96 (early rapid growth in early summer followed by a decline in growth rate in autumn) were similar to those observed at the original experimental station (FO) during 1983-84 (NUSCO 1992). The FO population that year was stressed by temperature increases of 7-9°C following the opening of the second quarry cut. It is unlikely that the thermal plume alone caused stress to the FN population in 1995-96. The summer of 1995 was one of the warmest observed during this study. Ambient monthly average surface water temperatures in late summer/early autumn 1995 were 1-3°C above operational period averages (see Lobster Section in NUSCO (1995)). These unusually high summer temperatures (up to 22°C) probably contributed to higher growth observed at references sites in 1995-96, and coupled with 2-4°C thermal incursion, may have created stressful temperature conditions that reduced growth rate of the FN population in late summer. Kanwisher (1966) and Vadas et al. (1978) both reported gradual demise of *Ascophyllum* once temperatures exceeded 26°C.

Mortality

Environmental stress to local *Ascophyllum* populations can result in breakage and loss of the

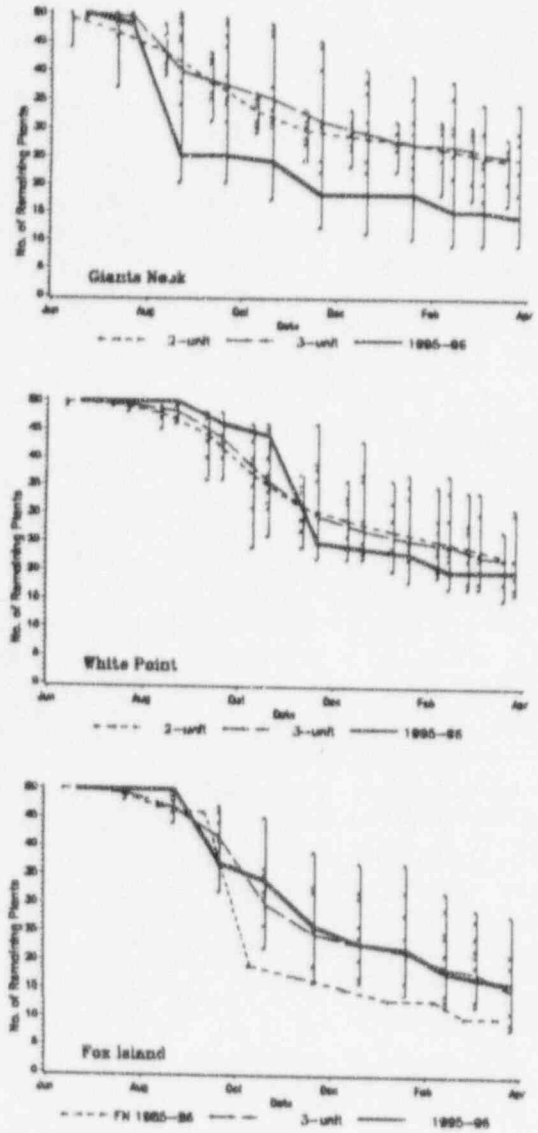


Fig. 9. *Ascophyllum* mortality, as number of remaining tagged plants, at each station.

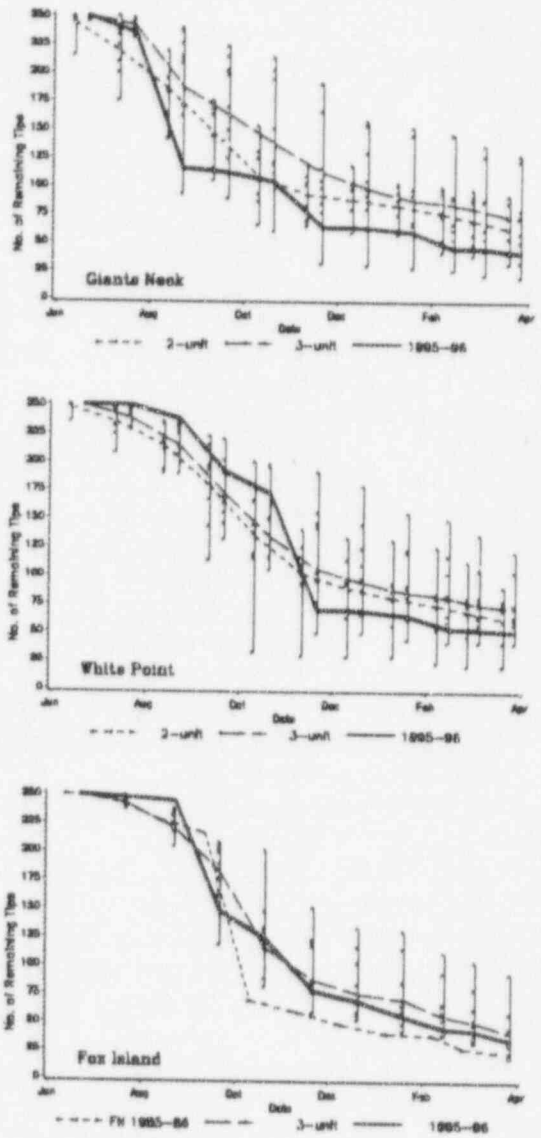


Fig. 10. *Ascophyllum* mortality, as number of remaining tagged tips, at each station.

upright shoots or fronds, commonly referred to as mortality. Population mortality is monitored by examining patterns of frond base tag loss (referred to as plant loss; Fig. 9) and apical tag loss (tip loss; Fig. 10). Plant loss at GN during 1995-96 (72%) was higher than both 2-unit (52%) and 3-unit (51%) operational means. Plant loss at WP during 1995-96 (60%) was also higher than operational period means of 55% and 56% for 2-unit and 3-unit periods, respectively. Plant loss at FN (68%) was lower than the 1985-86 2-unit year (80%) and the 3-unit mean (70%). During 1995-96, the greatest loss of plants at GN occurred between July and August, at WP between October and November, and at FE between August and September. None of these losses was associated with a major storm event. Temporal tip loss relationships at each station were similar to those described above for plant loss. Tip loss at GN during 1995-96 was 84%, which was higher than the 2-unit and 3-unit means of 75% and 72%, respectively. Similarly, 1995-96 tip loss at WP (80%) was high compared to the 2-unit (75%) and 3-unit (72%) means. At FN, tip loss for 1995-96

(86%) was intermediate, relative to the 1985-86 2-unit year (90%) and the 3-unit period (81%).

Ascophyllum mortality study in 1995-96 revealed no evidence of power plant impact. Our sampling site nearest the discharge (FN), has generally had higher mortality rates than at reference sites. However, these higher mortality rates do not appear to be related to proximity to the discharge, but rather to the higher degree of population exposure to wind- and wave-induced stress at FE, compared to the more sheltered reference sites. Furthermore, while population stress was indicated by growth studies in 1995-96, mortality estimates were well within the range of previous 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 the 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 *Ascophyllum* population recovery at FO, our original potentially impacted site, following power plant-induced elimination of the entire population from the site in 1984, has not changed from that reported in recent previous years (NUSCO 1995, 1996). 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, are outside the extremely limited range of conditions required for successful widespread *Ascophyllum* recruitment.

Conclusions

Rocky shores in the vicinity of MNPS continue to support a rich and diverse community of intertidal plants and animals. Impacts of the power plant to these local communities are limited to those areas exposed to the thermal effluent for at least part of the tidal cycle. Effects noted since Unit 3 began operation include shifts in occurrence and abundance of many intertidal species at FE, *e.g.*, presence or extended season of occurrence for species with warm-water affinity (*Codium, Sargassum, Gracilaria, Mytilus*) and absence or abbreviated season for species with cold-water affinity (*Chondrus, Monostroma, Dumontia, Littorina*). This altered community at FE has exhibited such consistency and resilience to change that many components appear able to persist, even in the absence of a thermal addition, as during an extended power plant shut-down.

Incursions of water with temperatures elevated 2-4°C above ambient impacted the *Ascophyllum* population nearest the discharge (FN). When such addition to ambient temperature did not exceed the tolerance of *Ascophyllum*, this resulted in increased growth relative to that at more distant stations; this pattern was seen in most 3-unit years. However, in the 1995-96 growing season, ambient water temperatures were unusually warm, and the thermal addition contributed by the MNPS discharge may have been sufficient to produce unfavorable conditions for growth at FN.

In summary, impacts to the rocky shore associated with operation of MNPS are restricted to within 150 m of the discharge. Current rocky intertidal studies have been sufficient to detect and document these ecologically significant changes to the local shore communities, and will allow assessment of any further changes, should they occur.

References Cited

Baardseth, E. 1955. Regrowth of *Ascophyllum nodosum* After Harvesting. Inst. Ind. Res. Stand., Dublin. 63 pp.

Baardseth, E. 1970. Seasonal variation in *Ascophyllum nodosum* (L.) Le Jol. in the Trondheimsfjord with respect to the absolute live and dry weight and the relative contents of dry matter, ash and fruit bodies. Bot. Mar. 13:13-22.

Bertness, M.D. 1989. Intraspecific competition and facilitation in a northern acorn barnacle population. Ecology 70:257-268.

Brinkhuis, B.H., N.R. Tempel and R.F. Jones. 1976. Photosynthesis and respiration of exposed salt marsh fucoids. Mar. Biol. 34:349-359.

Chock, J.S., and A.C. Mathieson. 1979. Physiological ecology of *Ascophyllum nodosum*

(L.) Le Jolis and its detached ecad *scorpioides* (Hornemann) Hauck (Fucales, Phaeophyta). Bot. Mar. 22:21-26.

Clifford, H.T., and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press, New York. 229 pp.

Connell, J.H. 1961. Effects of competition, predation, by *Thais lapillus* and other factors on natural populations of the barnacle, *Balanus balanoides*. Ecol. Monogr. 31:61-104.

Cousens, R. 1982. The effect of exposure to wave action on the morphology and pigmentation of *Ascophyllum nodosum* (L.) Le Jolis in south-eastern Canada. Bot. Mar. 25:191-195.

Cousens, R. 1986. Quantitative reproduction and reproductive effort by stands of the brown alga *Ascophyllum nodosum* (L.) Le Jolis in south-eastern Canada. Est. Coast. Shelf Sci. 22:495-507.

Draper, N., and H. Smith. 1981. Applied Regression Analysis. John Wiley and Sons, New York. 709 pp.

Gendron, L. 1989. Seasonal growth of the kelp *Laminaria longicruris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. Bot. Mar. 32:345-354.

Jones, J.E., and A. Demetropoulos. 1968. Exposure to wave action: Measurements of an important ecological parameter on rocky shores on Anglesey. J. Exp. Mar. Biol. Ecol. 2:46-63.

Kanwisher, G.W. 1966. Photosynthesis and respiration in some seaweeds. Pages 407-420 *in* H. Barnes (ed.) Some Contemporary Studies in Marine Science. George Allen Unwin Ltd., London.

Keser M., and B.R. Larson. 1984. Colonization and growth dynamics of three species of *Fucus*. Mar. Ecol. Prog. Ser. 15:125-134.

Lance, G.N., and W.R. Williams. 1967. A general theory of classificatory sorting strategies, I. Hierarchical systems. Comput. J. 9:373-380.

Lubchenco, J. 1980. Algal zonation in the New England rocky intertidal community: an experimental analysis. Ecology 61:333-244.

Lubchenco, J. 1983. *Littorina* and *Fucus*: effects of herbivores, substratum heterogeneity, and plant escapes during succession. Ecology. 64:1116-1123.

Lüning, K. 1990. Seaweeds: Their Enviromnent, Biogeography, and Ecophysiology. John Wiley and Sons, Inc. New York. 527 pp.

Menge, B.A. 1976. Organization of the New England rocky intertidal community: role of predation, competition and environmental heterogeneity. Ecol. Monogr. 46:355-393.

NAESCO (North Atlantic Energy Service Co.). 1994. Seabrook environmental studies, 1993. A characterization of environmental conditions in the Hampton-Seabrook area during the operation of Seabrook Station.

NAI (Normandeau Associates, Incorporated). 1996. Seabrook Station 1995 environmental studies in the Hampton-Seabrook area. A characterization of environmental conditions during the operation of Seabrook Station.

NUSCO (Northeast Utilities Service Company). 1985. Rocky Shore. Pages 1-41 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1984.

NUSCO. 1987. Rocky Intertidal Studies. Pages 1-66 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Summary of studies prior to Unit 3 operation.

NUSCO. 1992. Rocky Intertidal Studies. Pages 237-292 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1991.

NUSCO. 1993. Rocky Intertidal Studies. Pages 49-92 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1992.

NUSCO. 1994. Rocky Intertidal Studies. Pages 51-79 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1993.

NUSCO. 1995. Rocky Intertidal Studies. Pages 171-201 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1994.

NUSCO. 1996. Rocky Intertidal Studies. Pages 41-66 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1995.

Schonbeck, M.W., and T.A. Norton. 1978. Factors controlling the upper limits of fucoid algae on the shore. J. Exp. Mar. Biol. Ecol. 31:303-313.

Schonbeck, M.W., and T.A. Norton. 1980. Factors controlling the lower limits of fucoid algae on the shore. J. Exp. Mar. Biol. Ecol. 43:131-150.

Schneider, C.W. 1981. The effect of elevated temperature and reactor shutdown on the benthic marine flora of the Millstone thermal quarry, Connecticut. J. Therm. Biol. 6:1-6.

South, G.R., and I. Tittley. 1986. A checklist and distributional index of the benthic marine algae of the North Atlantic Ocean. Huntsman Marine Laboratory and British Museum (Nat. Hist.), St. Andrews and London. 76 pp.

Stromgren, T. 1977. Short-term effects of temperature upon the growth of intertidal fucales. J. Exp. Mar. Biol. Ecol. 29:181-195.

Stromgren, T. 1981. Individual variation in apical growth rate in *Ascophyllum nodosum* (L.) Le Jolis. Aquat. Bot. 10:377-382.

Taylor, W.R. 1957. Marine Algae of the Northeast Coast of North America. Univ. Mich. Press, Ann Arbor. 870 pp.

Topinka, J., L. Tucker, and W. Korjeff. 1981. The distribution of fucoid macroalgal biomass along central coastal Maine. Bot. Mar. 24:311-319.

Underwood, A.J. and E.J. Denley. 1984. Paradigms, explanations and generalizations in models for the structure of intertidal communities of rocky shores. pp. 151-180 *in* D.R. Strong, Jr., D. Simberloff, L.G. Abele and A.B. Thistle, eds., Ecological Communities: Conceptual Issues and the Evidence. Princeton University Press, Princeton N.J. 611 pp.

Vadas, R.L., M. Keser, and P.C. Rusanowski. 1976. Influence of thermal loading on the ecology of intertidal algae. Pages 202-251 *in* G.W. Esch and R.W. MacFarlane (eds.) Thermal Ecology II. ERDA Symposium Series, Augusta, GA.

Vadas, R.L., M. Keser, and P.C. Rusanowski. 1978. Effect of reduced temperature on previously stressed populations of an intertidal alga. Pages 434-451 *in* J.H. Thorp and G.W. Gibbons (eds.) DOE Symposium Series, Springfield, VA. (CONF-771114, NTIS).

Vadas, R.L., and W.A. Wright. 1986. Recruitment, growth and management of *Ascophyllum nodosum*. Actas II Congr. Algas Mar. Chilenas:101-113.

Villalard-Bohnsack, M. 1995. Illustrated Key to the Seaweeds of New England. Rhode Island Nat. Hist. Surv., Kingston R.I. 144 pp.

Wilce, R.T., J. Foertch, W. Grocki, J. Kilar, H. Levine, and J. Wilce. 1978. Flora: Marine Algal Studies. Pages 307-656 *in* Benthic Studies in the Vicinity of Pilgrim Nuclear Power Station, 1969-1977. Summary Rpt. Boston Edison Co.

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Eelgrass

Introduction

Eelgrass (*Zostera marina* L.) occurs widely in estuaries and lagoons of temperate and warm boreal coasts in the Atlantic and Pacific Oceans (Setchell 1935). Such a geographically broad distribution is the result of this species' wide tolerances to temperature, salinity and water depth (Osterhout 1917; Setchell 1929; Uphof 1941; Burkholder and Doheny 1968; Dillon 1971; Thayer et al. 1984). After the disappearance of most eastern North American and European populations in the 1930s (Tutin 1942; Rasmussen 1973, 1977), the importance of eelgrass to coastal ecosystems (described in previous reports, e.g., NUSCO 1994) has become widely recognized. Following the destruction of *Zostera* populations in the 1930s, shoreline erosion resulted from increased wave scour and changes in current patterns. Habitat alteration also occurred within the subtidal zone, evidenced by declines in abundance of many commercially and recreationally valuable species (Stauffer 1937; Dexter 1947; Milne and Milne 1951; Orth 1973, 1977; Rasmussen 1973, 1977; Thayer et al. 1975; Stevenson and Confer 1978; Zieman 1982). Eelgrass beds on the north shore of Long Island Sound (LIS) are concentrated in shallow protected areas east of the Connecticut River (Koch and Beers 1996). Extensive meadows of eelgrass are common in the vicinity of Millstone Nuclear Power Station (MNPS). Temperature changes have been demonstrated to affect eelgrass 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; Evans et al. 1986; Zimmerman et al. 1989; Taylor et al. 1995; Vergeer et al. 1995). Because of the ecological importance of eelgrass and the prediction that the 3-unit thermal plume could reach to the nearest population in Jordan Cove (ENDECO 1977; NUSCO 1988), the present study was initiated in 1985 to monitor this population and others nearby. Objectives of the present study are to identify temporal patterns of eelgrass distribution, abundance and reproduction in the vicinity of MNPS and to determine the extent to which changes in these patterns are the result of natural variability or MNPS operation.

Materials and Methods

Three eelgrass study sites in the vicinity of MNPS were sampled during 1996 (White Point-WP, Jordan Cove-JC, Niantic River-NR) (Fig. 1). The WP and JC stations, located 1.6 km and 0.5 km east of the power plants discharge, respectively, 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 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 changes in distributional patterns of eelgrass in the river. The original sampling site (#1), located midway between Camp Rowland and the navigation channel (Fig. 1), was sampled in the summer of 1985 and June 1986. A substantial population decline at site #1 occurred in July 1986, resulting in the establishment of another NR sampling site (#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. However, the recovery of the eelgrass population at the old NR site (#1), noted in 1993, permitted NR samples to be taken again at this station during the 1993 and 1994 sampling periods (June-September). Again, no plants were observed at site #1 in September 1994 or at the beginning of the 1995 sampling year (June), and a new sampling site had to be established on the east side of the channel, opposite Smith Cove (site #4). Monthly observations of NR#1, NR#2 and NR#3 have continued since their population disappeared, however, no eelgrass recolonization was observed at any of these sites through the 1996 sampling year.

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

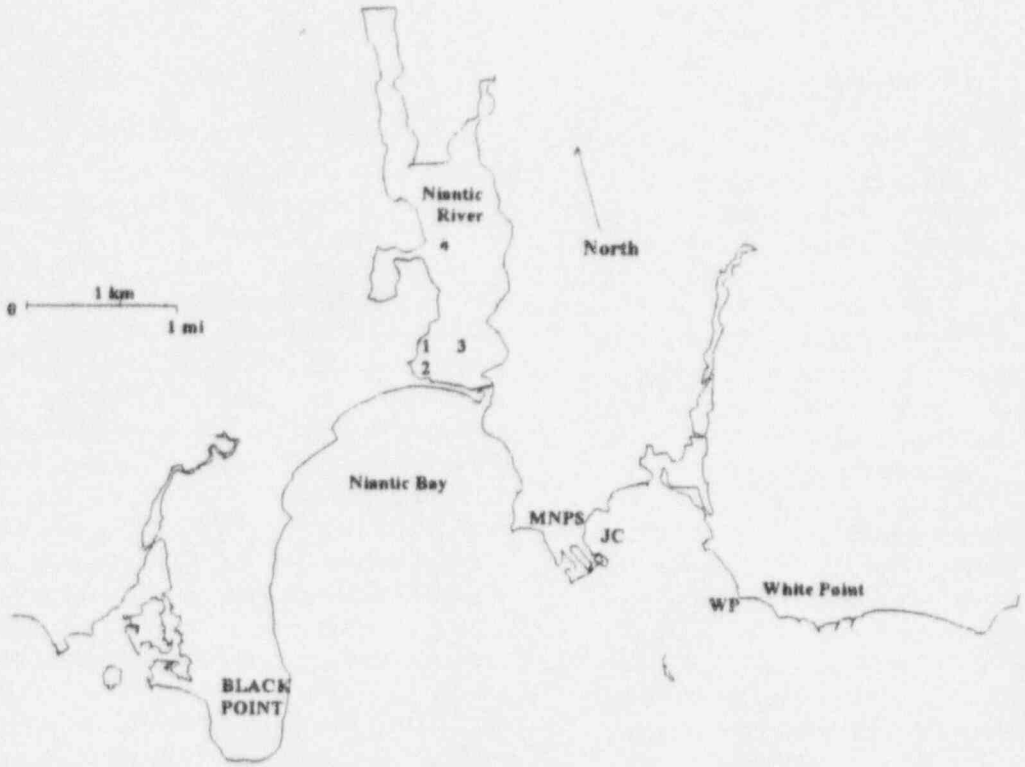

Fig. 1. Map of the Millstone Point area showing the location of eelgrass sampling stations, JC=Jordan Cove, NR=Niantic River(1=sampled 1985-June 1986 and 1993-1994, 2=sampled July 1986, 3=sampled 1987-1992, 4=sampled 1995-1996), WP= White Point.

taken to the laboratory for processing. A 3.5 cm diameter x 5 cm deep core was taken concurrently with eelgrass samples for analysis of sedimentary characteristics at each station. Temperature in Jordan Cove was measured by submerging an encased thermistor-recorder. Temperature measurements have been recorded in Jordan Cove since 1991. All Millstone units were shutdown during the 1996 sampling period, so there was no possibility of thermal addition to Jordan Cove. This provided an opportunity to more closely examine natural factors that affect temperature at JC, such as insolation and tides. To assess daily cycles, temperatures are reported this year (1996) as hourly averages (rather than daily averages) for 2 discrete one-week periods in June (when air and water temperatures are coolest, but daily solar irradiance is at a maximum) and August (when warmest water temperatures typically occur; NUSCO 1996).

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 used to estimate the percentage of reproductive shoots in the population. Shoots were rinsed in freshwater to remove invertebrates and epiphytes. 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 constant weight. Dry weights from 1988 to 1996 were estimated from the wet-weight/dry-weight relationship obtained above.

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 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.

Results

Temperature

Average daily water temperatures at the MNPS intakes and discharge for the period June through September 1996 are presented in Figure 2a. Because MNPS was not discharging heated effluent, intake and discharge temperatures were always within 1°C of each other. Temperatures during this period ranged from 12°C in early June to 21°C in late August/early September. Water temperatures at the JC eelgrass station and at the MNPS intake and discharge are presented as hourly averages over two six-day periods in 1996 (June 16-21 and August 4-9; Figs. 2b and c, respectively) to better illustrate natural daily temperature cycles in Jordan Cove. As with daily average temperatures, intake and discharge hourly temperatures were generally within 1°C of each other, and fluctuation over the daily cycle was minimal at these two sites. Conversely, considerable temperature fluctuation was observed at JC over the daily cycle, with increases of up to 4-5°C observed on some days. Lowest temperatures were similar to those measured at the MNPS intake and discharge. The highest peaks observed (e.g., June 16 and 18, August 8 and 9) were related to solar insolation, as all occurred in mid-afternoon of sunny days; lower peaks were noted on days with more cloud cover.

Some of the lower peaks occurred at other times of the day such as night or early morning. These secondary peaks were the result of flushing of more protected (and presumably warmer) waters of upper Jordan Cove out to the JC eelgrass site, as they typically followed ebbing tide. Additionally, heat transfer from sediments may have caused additional warming, particularly during slack tide. Similar daily temperature patterns observed in Monisweag Bay, Maine were also attributed to the effects of direct solar heating, indirect solar heating via mud flats, and tidal effects (Thompson 1978).

Sediments

Sediments at eelgrass sampling stations have been characterized since 1985 through monthly (June-September) measurements of mean grain size, and silt/clay and organic contents (Fig. 3). In general, sediments at stations nearest MNPS (JC and WP) have been less variable than those at NR. Variability at NR has possibly been caused by frequent relocation of the sampling site within the Niantic River. Sediments collected during 1996 were coarser at JC (mean grain size, 0.22-0.25 mm), than those at WP (0.10-0.14 mm) and NR (0.09 mm in all months). Silt/clay content in 1996 was highest at NR (monthly range: 34.8-40.3%) relative to WP (8.6-25.5%) and JC (1.1-3.6%). Similarly, sediment organic content was higher at NR (6.7-7.6%) than at WP (1.4-5.4%) or JC (0.9-1.6%). All sediment parameters measured at JC and WP in 1996 were within the ranges for previous years. Monthly mean grain size and organic content estimates at NR during the 1996 sampling period were also within historical ranges, although mean grain size was smaller relative to most previous samples. Silt/clay content at NR in 1996 ranged from 34.8 (August) to 40.3% (September). Monthly silt/clay content estimates at NR in 1996 were high compared to most previous years, with the September 1996 value being the highest observed at any site in the Niantic River since the beginning of this study in 1985.

Shoot Density

Annual mean shoot density in 1996 was highest at JC (440 shoots/m^2), intermediate at NR (310 shoots/m^2) and lowest at WP (171 shoots/m^2; Table 1). Annual mean densities at all stations were within historical ranges. Monthly shoot densities (shoots/m^2) in 1996 ranged from 313 (August) to 537 (June) at JC, from 253 (July) to 476 (June) at NR, and from 115 (July) to 215 (September) at WP. Monthly mean densities were also within the ranges of previous years (Fig. 4).

Trend analysis applied to time-series of monthly shoot densities indicated significant decreasing trends at JC (slope=-3.912 shoots/m^2/sample period; $p<0.01$) and WP (slope=-2.985 shoots/m^2/sample period; $p<0.01$). Trend analysis was not performed on NR data due to lack of a consistent time-series for any given site resulting from frequent sampling station relocations following localized population disappearances.

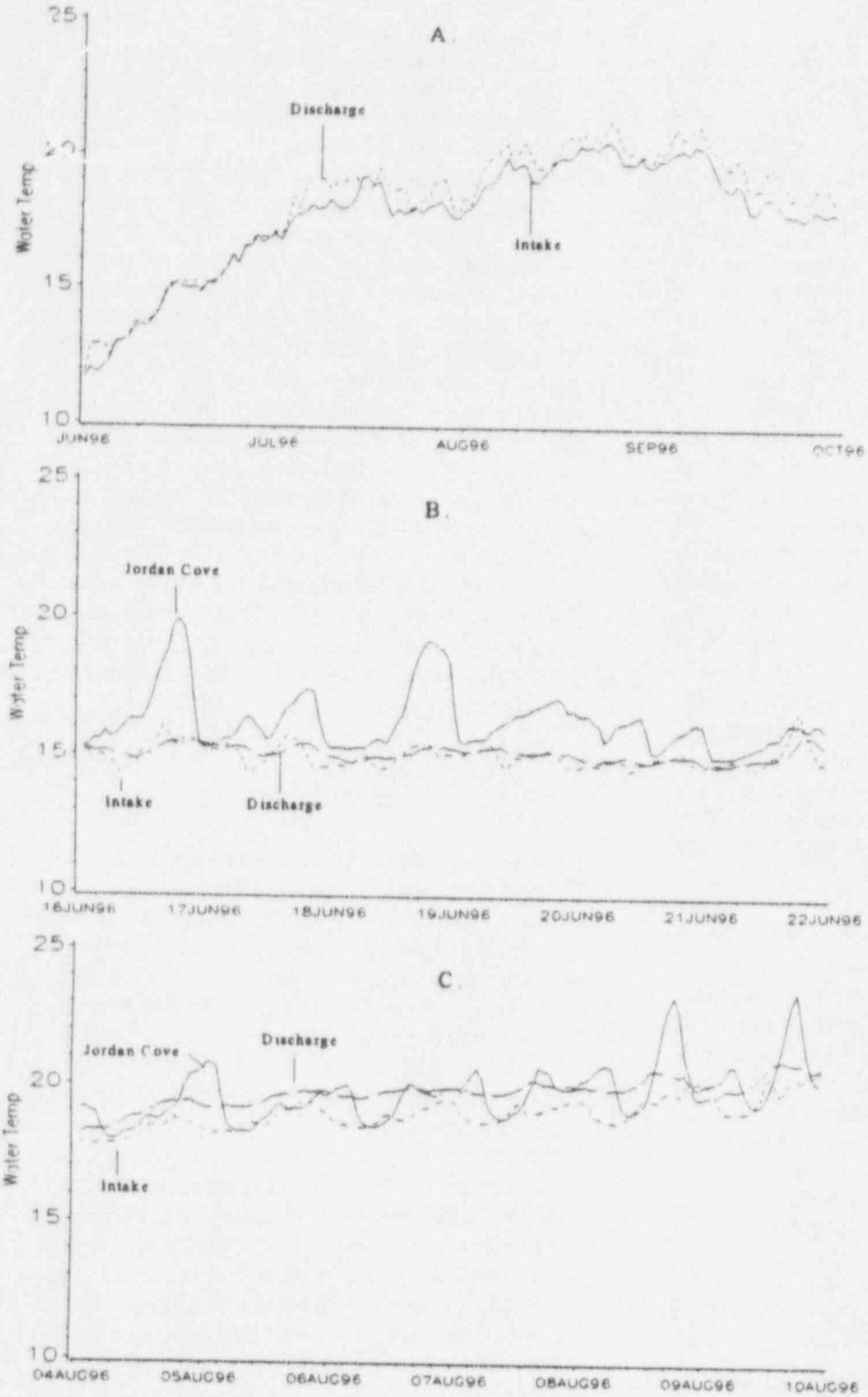

Fig. 2. Mean daily water temperatures at the MNPS intake and discharge from June through September (A.) 1996. hourly average water emperatures at the Jordan Cove eelgrass station and MNPS intake and discharge temperatures from June 16-23 (B.) and August 4-10 (C.).

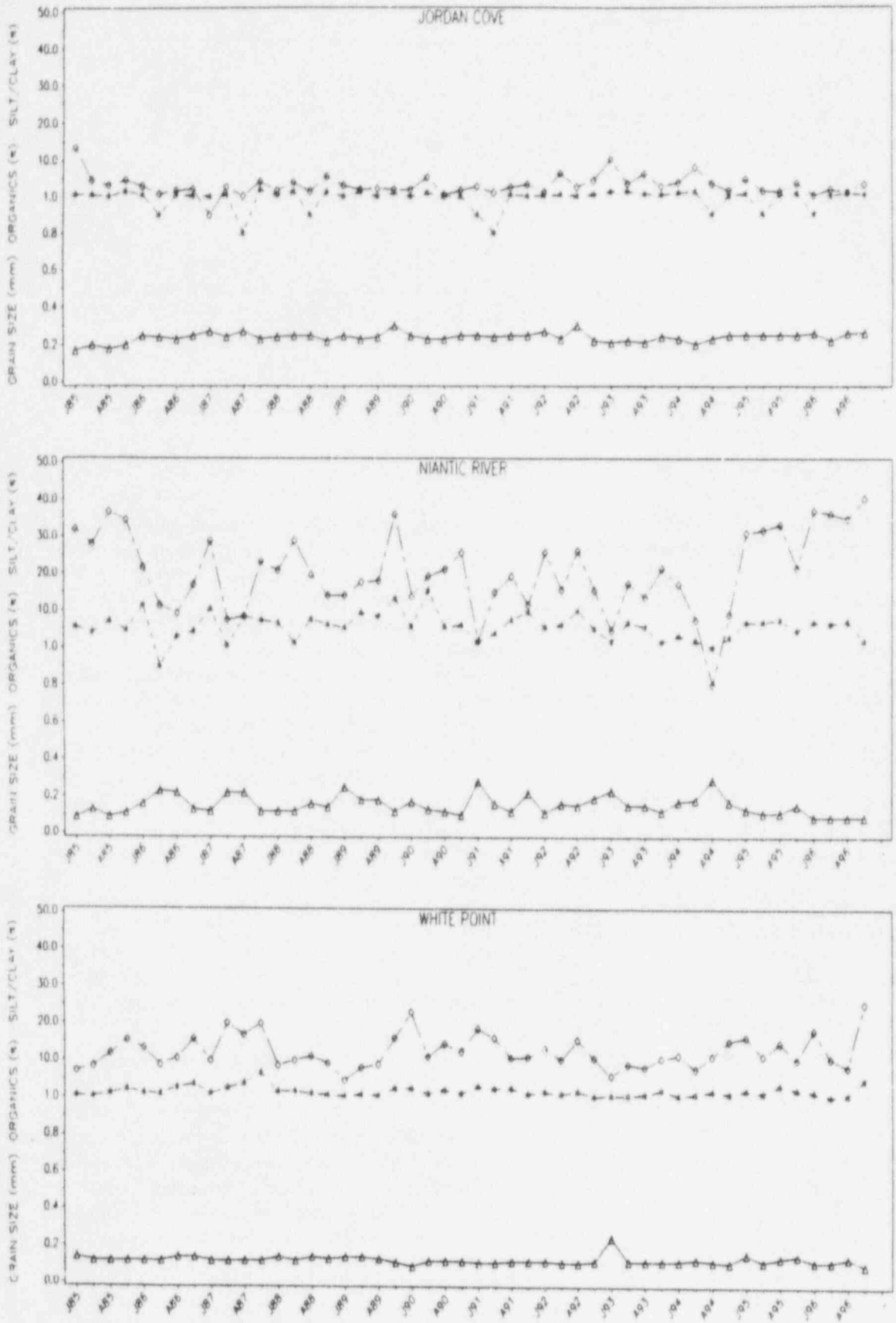

Fig. 3. Mean grain size (△), organic content (*) and silt/clay content (◇) of sediments at MNPS eelgrass stations . Jordan Cove. Niantic River and White Point sampled during the period June-September from 1985 through 1996.

TABLE 1. Annual and monthly average shoot density (no./m^2), length (cm) and dry weight standing stock (gm/m^2) of eelgrass sampled near MNPS during the period June to September since 1985.

	ANNUAL MEANS												1996			
	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	Jun	Jul	Aug	Sept
Shoot Density																
Jordan Cove	572	713	542	468	411	338	603	630	484	282	450	440	537	414	313	414
Niantic River	413	72	294	307	240	225	249	233	385	132	239	310	476	253	269	246
White Point	286	218	227	161	335	185	242	204	310	141	237	171	178	115	167	215
Shoot Length																
Jordan Cove	57	57	77	75	74	38	48	53	54	35	37	51	58	45	57	44
Niantic River	50	39	81	88	94	73	51	48	58	28	23	66	82	70	58	48
White Point	107	116	126	86	110	106	87	72	107	95	92	79	98	110	64	58
Standing Stock																
Jordan Cove	243	276	258	238	202	105	169	210	160	60	104	121	127	110	139	110
Niantic River	156	32	184	181	183	143	81	79	125	18	29	90	135	103	74	48
White Point	265	260	201	90	236	180	148	110	275	100	180	89	97	84	81	95

Shoot Length

Average shoot lengths during 1996 were longest at WP (79 cm), shortest at JC (51 cm) and intermediate at NR (66 cm); annual means were within overall ranges observed since 1985. Monthly shoot lengths in 1996 were highest in June or July and lowest in September. Shoot lengths in 1996 were 44-58 cm at JC, 48-82 cm at NR, and 58-110 cm at WP, and fell within historical ranges at each station (Fig. 4).

Standing Stock

Average eelgrass standing stock during 1996 was higher at JC (121 g/m^2) than at NR (90 g/m^2) and WP (89 g/m^2; Table 1). The annual standing stock estimates at JC and NR during 1996 were within the historic range; however, standing stock at WP in 1996 was the lowest recorded over the entire study period.

Monthly standing stock estimates in 1996 ranged from 110 g/m^2 (July and September) to 139 g/m^2 (August) at JC, from 48 g/m^2 (September) to 135 g/m^2 (June) at NR, and from 81 g/m^2 (August) to 97 g/m^2 (June) at WP. Monthly mean standing stock estimates for 1996 were within historic ranges (Fig. 6). Trend analysis, performed on monthly dry-weight estimates over the entire time-series, indicated that standing stocks have significantly declined since 1985 at JC (slope=-4.079 g/m^2/sample period, $p<0.05$). There was no significant trend in standing stock over the study period at WP. Standing stock estimates have declined at NR over the study period; however, trend analysis was not performed on the NR standing stock because of sampling station relocations.

Reproductive Shoots

Annual and monthly percentages of reproductive shoots are presented in Table 2. The highest annual percentage of reproductive shoots at any station since 1985 occurred at NR in 1996 (15.0%); in previous years, annual percentages of reproductive shoots at NR have ranged from 0 to 8.7%. At JC the 1996 annual percentage (2.0%) was within the range of previous years. At WP, 4.1% of the shoots collected in 1996 were reproductive; this was also within the range of annual percentages in previous years (0.4-10.4%). Monthly percentage of reproductive eelgrass shoots in 1996 was highest in June at NR and WP (29.6 and 9.9%, respectively) and July at JC (3.7%). The July 1996 sample at NR also contained a high percentage of reproductive shoots (17.8%). No reproductive shoots were collected at NR after the July 1996 sample period. In August 1996, 0.3% and 3.6% of the plants sampled at JC and WP, respectively, were reproductive; no reproductive plants were collected at any site in September.

Discussion

Considerable fluctuations in population parameters monitored for eelgrass have been noted over the past 12 years. Fluctuations have been most pronounced in the Niantic River, where a general declining trend has been evident since early study years. While a patchy eelgrass population continues to inhabit the Niantic River, health of the overall Niantic River population remains questionable when compared to early study years (1985-1990) and to historic observations (Marshall 1994). Since 1985, we have observed only patchy transient populations within

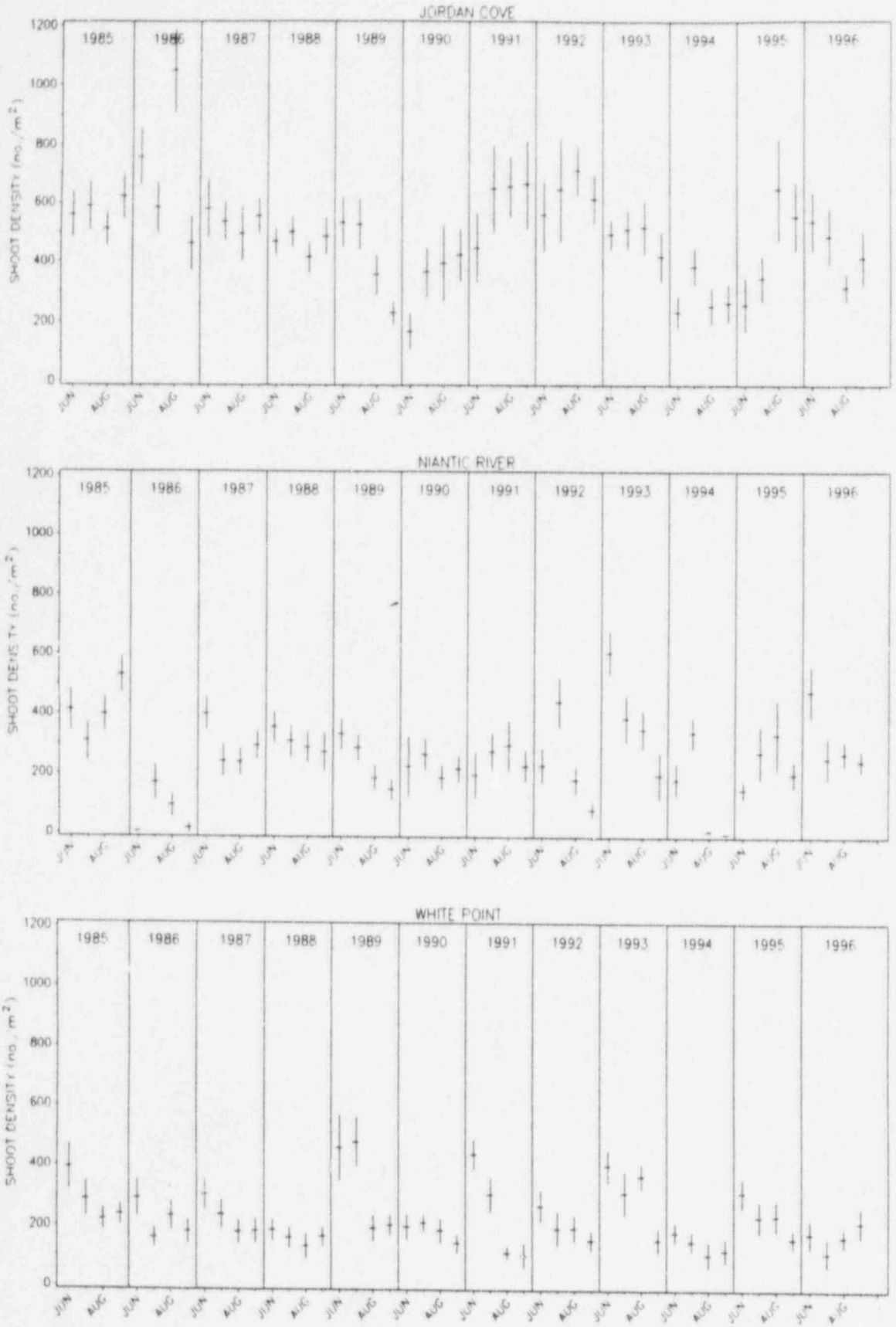

Fig. 4. Mean number of eelgrass shoots per m^2 ±95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985-1996.

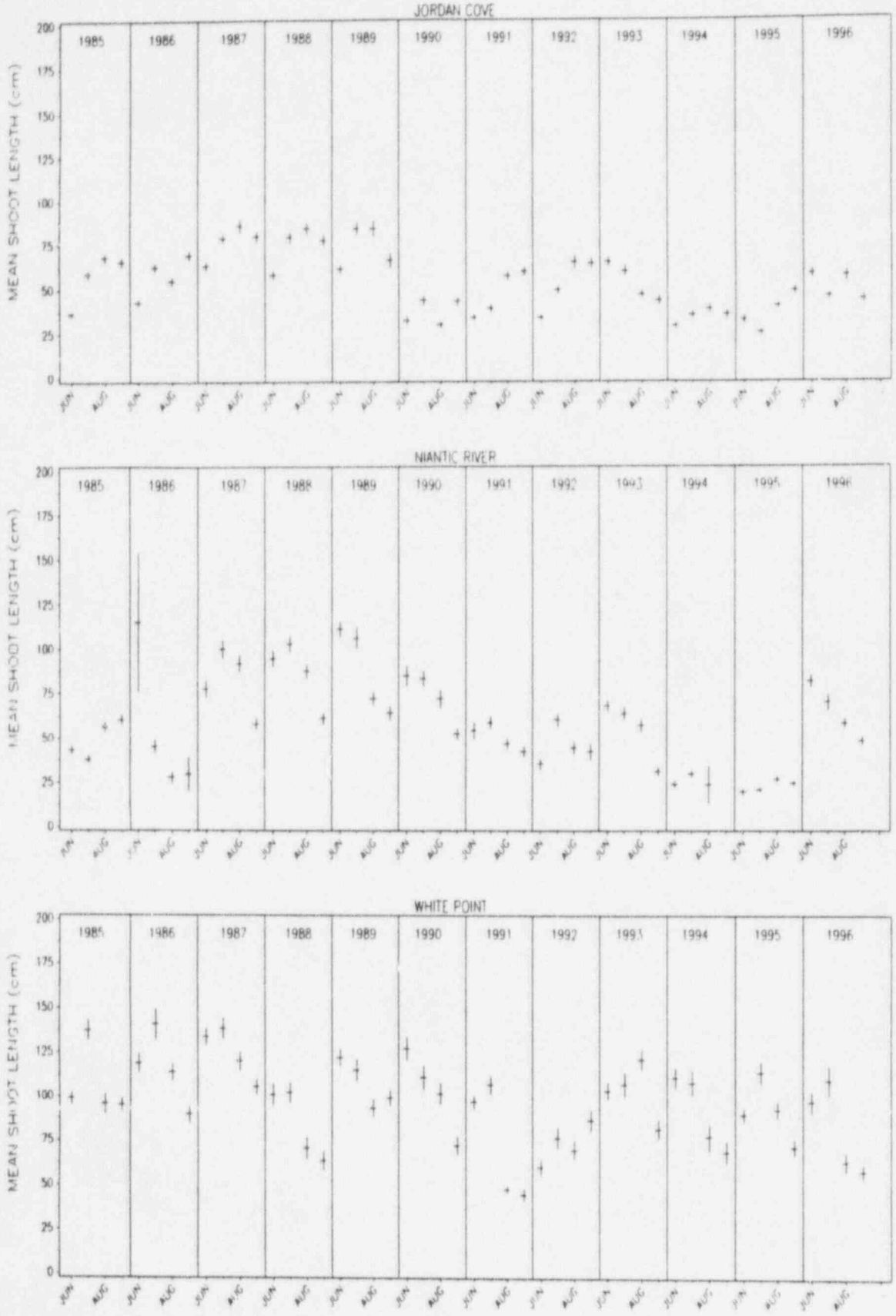

Fig. 5. Mean shoot length (cm) ±95% C.I. at Millstone eelgrass stations during the period June-September from 1985 through 1996.

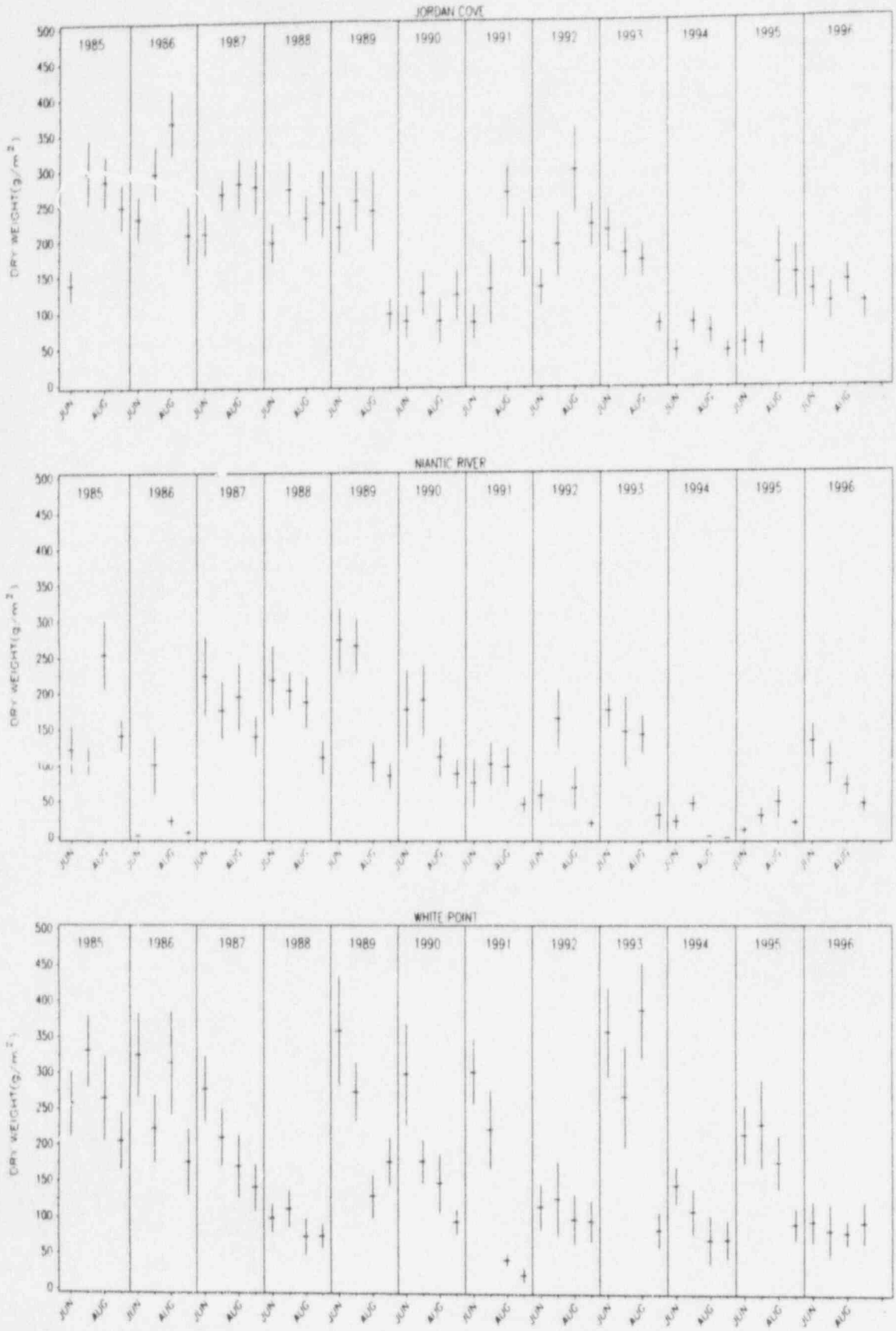

Fig 6. Mean dry weight (grams per m^2) ±95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1996.

TABLE 2. Number of reproductive shoots, total number of shoots and percentage of reproductive shoots at eelgrass sampling stations from June 1985 through September 1996.

Year	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	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
1995	8	0.4	6	256	2.3	2	343	0.6	0	646	0.0	0	554	0.0
1996	35	2.0	16	537	3.0	18	488	3.7	1	323	0.3	0	414	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
1995	0	0.0	0	149	0.0	0	373	0.0	0	334	0.0	0	200	0.0
1996	186	15.0	141	476	29.6	45	253	17.8	0	269	0.0	0	244	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
1995	35	3.7	24	314	7.6	11	234	4.7	0	237	0.0	0	161	0.0
1996	28	4.1	16	176	9.9	6	125	4.8	6	167	3.6	0	215	0.0

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

the Niantic River. Localized short-term eelgrass recolonization has been reported in the Niantic River previously (at NR #1 from 1989-1993; NUSCO 1994). However, while all previously sampled areas in the river (NR #1, NR #2 and NR #3) continued to show no signs of population recovery in 1996, no indications of population decline have been noted at the current sampling station in the Niantic River (NR #4), sampled in 1995 and 1996.

Long-term decline/recovery cycles have been reported throughout the distribution of eelgrass since the 1930s. 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 grazing and uprooting by swans (Marshall 1994) and other waterfowl (Vermaat and Verhagen 1996) to human activities, e.g., eutrophication (Bulthuis 1983; Orth and Moore 1983; Cambridge and McComb 1984; Neverauskas 1985; Burkholder 1993; Taylor et al. 1995), decreased light penetration (Fletcher and Fletcher 1995; Koch and Beer 1996) and changes in near-shore land use (Kemp et al. 1983). Short (1988) suggested that a decline in water quality and the presence of *Labyrinthula* contributed to declines of eelgrass in the Niantic River in the late 1980s. It is likely that these factors, perhaps further exacerbated by waterfowl grazing, were largely responsible for more recent population losses observed during this study. The high number of reproductive shoots in the 1996 samples from NR make it reasonable to expect small transient populations to reappear in the river in the future through seed production and dispersal. Regardless of the nature of environmental conditions, none of the factors affecting this eelgrass population were related to the operation of MNPS because thermal effluent does not reach the Niantic River.

Some indication of moderate population decline was also noted at the other eelgrass study sites. Given the proximity of the JC study population to MNPS, and because modeling predictions indicated possible exposure of this population to the MNPS 3-unit thermal plume (ENDECO 1977; NUSCO 1988), population characteristics and temperature conditions at this site have been closely monitored. Following decline in 1994, the population at JC improved in 1995 and 1996. However, analyses of the eleven-year time-series still showed declining trends in two important population parameters: shoot density and standing stock biomass. This trend is likely due to relatively high shoot density and standing stock levels in early study years and fluctuating levels in more recent years.

It is difficult to associate the general decline of the Jordan Cove eelgrass population with a power plant impact, owing 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, to freezing in winter. A continuous temperature recorder deployed on the sediment surface within the JC study population measured higher water temperatures, relative to ambient temperatures at the MNPS intakes in 1996 (Fig. 2). Temperatures were elevated up to 5°C above intake temperatures during afternoons on sunny days, and smaller temperature increases appeared related to tidal flushing of warmer water from upper Jordan Cove. Regardless of the exact mechanism, since MNPS was not operating and not discharging heated effluent during the summer of 1996, the elevated temperature was a natural phenomenon. Therefore, it is possible that the disparity between JC and ambient temperatures observed in previous years was primarily related to these natural factors.

Setchell (1929) first stressed the importance of temperature in regulating eelgrass growth and development. It was later shown that eelgrass is sensitive to even small temperature variations (Thayer et al. 1984). Eelgrass fails to produce seeds at temperatures above 15-20^{0}C (Burkholder and Doheny 1968; Orth and Moore 1983). Higher water temperatures, e.g., from heated power plant effluents, could eliminate eelgrass from nearby areas (Phillips 1974; Thayer et al. 1984). For example, abundance of another seagrass (*Thalassia testudinum*) 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), declined significantly in the vicinity of power plant effluents. These studies indicated that elevated water temperatures increased respiration beyond levels that could be supported by plant photosynthesis. Increased water temperatures in Jordan Cove, regardless of the cause, could stress this population for reasons described above. However, the apparent decline of the *Zostera* population in JC may be related to factors other than temperature; similar to the Niantic River, shallow water in Jordan Cove allows for high rates of eelgrass grazing by brant, geese and swans. A general decline in shoot length has also been observed at JC, possibly indicating that water depths are decreasing at JC, perhaps due to movement of sand into the eelgrass bed from nearby sand bars. The dynamic nature of nearshore sand flats in Jordan Cove have been documented historically (OMNI 1995). If water is becoming shallower at JC, depth related stress mechanisms may have also become

more pronounced, accounting for population declines in recent years.

A decline in shoot density, but not standing stock, was noted at WP. The reasons for this apparent decline are unclear. The population at WP is in deeper water, and therefore less susceptible to the natural environmental stress mechanisms potentially affecting the JC population. The MNPS thermal plume most likely does not affect the population at WP, based on hydrodynamic modeling and thermal plume studies. Furthermore, MNPS did not produce a thermal plume during the eelgrass growing season in 1996, when shoot density and standing stock were low relative to many past sampling years when the plant was operating. Therefore, this apparent short-term decline of the eelgrass population at WP in 1996 was due to natural variability, and not to MNPS operation.

Conclusions

Shallow water habitats near MNPS at JC and WP continued to support relatively stable eelgrass populations, while similar habitats in the Niantic River have, for the most part, become unsuitable for maintenance of populations. Since 1985, the Niantic River eelgrass population has declined, and has been characterized by small transient patches. Patch expansion through rhizome spreading, typical of a healthy population, has not been noted in the Niantic River over the study period, and the only observed recolonization event through seed germination was short-lived (<2 years). Because the Niantic River is located well away for any influence of the MNPS thermal plume, declines there were related to other environmental factors such as water quality, disease or waterfowl grazing.

Data collected in 1996, when MNPS was not operating, demonstrated that natural environmental variability factored strongly in less extreme population fluctuations observed historically at WP and JC. While the WP population is on the fringe of the predicted areal extent of the thermal plume, temperature monitoring there has never indicated thermal incursion. Therefore, recent declines were attributed to natural variablity rather than power plant operation. Elevated temperatures at JC, relative to ambient MNPS intake temperatures, have been measured directly in the past, and were possibly responsible for periodic population declines observed at that site. Elevated summer temperatures of up to 4-5°C were observed at JC in 1996, and were attributed solar warming. These temperature increases in 1996 were similar to increases in previous years when MNPS was operating, indicating that the MNPS thermal plume has, at most, only a minor influence on water temperatures at the JC study site. Thermal plume modeling and field studies indicated temperatures increases of up to 1-2°C at JC. Increased natural environmental stress related to sand shoaling within the eelgrass bed may have caused declines in shoot density and standing stock at JC. Reduced water depth over the bed may cause this population to be more susceptible to solar heating and grazing by waterfowl.

References Cited

Bulthuis, D.A. 1983. Effects of *in situ* light reduction on density and growth of the seagrass *Heterozostera tasmanica* (Martens *ex* Aschers.) den Hartog in Western Port, Victoria, Australia. J. Exp. Mar. Biol. Ecol. 67:91-103.

Burkholder, P.R., and T.E. Doheny. 1968. The biology of eelgrass. Contribution No. 3. Dept. Conservation and Waterways, Town of Hempstead,Long Island. Contribution No. 1227 Lamont Geological Observatory, Palisades, New York. 120 pp.

Burkholder, J. 1993. Botanist Investigates Impact of Nitrate on Seagrasses. Coastlines 3:6.

Cambridge, M.L., and A.J. McComb. 1984. The loss of seagrass from Cockburn Sound, Western Australia. I. The time course and magnitude of seagrass decline in relation to industrial development. Aquat. Bot. 20:229-243.

den Hartog, C. 1987. "Wasting Disease" and other dynamic phenomena in *Zostera* beds. Aquat. Bot. 27:3-14.

Dexter, R.W. 1947. The marine communities of a tidal inlet at Cape Ann, Massachusetts: A study in bio-ecology. Ecol. Monogr. 17:261-294.

Dillon, C.R. 1971. A comparative study of the primary productivity of estuarine phytoplankton and macrobenthic plants. Ph.D. Dissertation. Univ. North Carolina, Chapel Hill. 112 pp.

ENDECO (Environmental Devices Corporation). 1977. Postoperational Units 1 and 2, preoperational Unit 3 hydrothermal survey of the Millstone Power Station. Rpt. to Northeast Utilities Service Co.

Evans, A.S., K.L. Webb, and P.A. Penhale. 1986. Photosynthetic temperature acclimation in two coexisting seagrasses, *Zostera marina* L. and *Ruppia maritima* L. Aquat. Bot. 24:185-197.

Fletcher, S.W. and W.W. Fletcher. 1995. Factors affecting changes in seagrass distribution and diversity patterns in the Indian River Lagoon

complex between 1940 and 1992. Bull. Mar. Sci. 57:49-58.

Folk, D. 1974. Petrology of Sedimentary Rocks. Hempshill Pub. Co., Austin, Texas. 192 pp.

Hollander, M., and D.A. Wolfe. 1973. Non-parametric statistical methods. John Wiley and Sons, New York. 503 pp.

Kemp, W.M., W.R. Boynton, R.R. Twilley, J.C. Stevenson and J.C. Means. 1983. The decline of submerged vascular plants in Upper Chesapeake Bay: summary of results concerning possible causes. Mar. Tech. Soc. J. 17:78-89.

Keser, M., B.R. Larson, R.L. Vadas, and W. McCarthy. 1978. Growth and ecology of *Spartina alterniflora* in Maine after a reduction in thermal stress. pp. 420-433 *in* J.H. Thorpe and J.W. Gibbons (eds). Energy and Environmental Stress in Aquatic Systems. DOE Symposium Series (CONF-771114). Nat. Tech. Infor. Ser., Springfield, VA.

Koch, E.W. and S. Beers. 1996. Tides, light and the distribution of *Zostera marina* in Long Island Sound, USA. Aquat. Bot.. 53:97-107.

Marshall, N. 1994. The Scallop Estuary: The Natural Features of the Niantic River. The Anchorage Publisher, St. Michaels, MD. 152 pp.

Milne, L.J., and M.J. Milne. 1951. The eelgrass catastrophe. Sci. Am. 184:52-55.

Neverauskas, V.P. 1985. Port Adelaide sewage treatment works sludge outfall. Effect of discharge on the adjacent marine environment. Progress report, July 1982-May 1984. EWS Rpt. 85/6.

NUSCO (Northeast Utilities Service Company). 1988. Hydrothermal Studies. pp. 323-355 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-Unit Operational Studies 1986-1987.

NUSCO. 1993. Eelgrass. pp. 33-48 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. 1992 Ann. Rpt.

NUSCO. 1994. Eelgrass. pp. 35-49 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. 1993 Ann. Rpt.

NUSCO. 1996. Eelgrass. pp. 69-79 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. 1995 Ann. Rpt.

OMNI (DDL OMNI Engineering LLC). 1995. Jordan Cove Study.

Orth, R.J. 1973. Benthic infauna of eelgrass, *Zostera marina*, beds. Chesapeake Sci. 14:258-269.

Orth, R.J. 1977. The importance of sediment stability in seagrass communities. pp. 281-300 *in* B.C. Coull (ed). Ecology of Marine Benthos. Univ. South Carolina Press, Columbia, SC.

Orth, R.J., and K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. Science 222:51-52.

Osterhout, W.J.V. 1917. Tolerance of fresh water by marine plants and its relation to adaptations. Bot. Gaz. 63:146-149.

Patriquin, D.G. 1975. 'Migration' of blowouts in seagrass beds at Barbados and Curacao, West Indies, and its ecological and geographical implications. Aquat. Bot. 1:163-189.

Phillips, R.C. 1974. Transplantation of seagrasses with special emphasis on eelgrass, *Zostera marina* L. Aquaculture 4:161-176.

Phillips, R.C. 1980. Responses of transplanted and indigenous *Thalassia testudinum* Banks *ex* König and *Halodule wrightii* Aschers. to sediment loading and cold stress. Contrib. Mar. Sci. 23:79-87.

Rasmussen, E. 1973. Systematics and ecology of the Isefjord marine fauna (Denmark). Ophelia 11:1-495.

Rasmussen, E. 1977. The wasting disease of eelgrass (*Zostera marina*) and its effect on environmental factors and fauna. pp. 1-51 *in* C.P. McRoy and C. Helfferich (eds). Seagrass Ecosystems: A Scientific Perspective. Marcel Dekker Inc., New York. 314 pp.

Roessler, M.A. 1971. Environmental change associated with a Florida power plant. Mar. Poll. Bull. 2:87-90.

Roessler, M.A., and J.C. Zieman, Jr. 1969. The effects of thermal additions on the biota of southern Biscayne Bay, Florida. pp. 136-145 *in* Proceed. Gulf and Caribbean Fish. Inst. Contrib. No. 1165, 22nd Ann. Sess.

Roman, C.T., and K.W. Able. 1988. Production ecology of eelgrass (*Zostera marina* L.) in a Cape Cod salt marsh-estuarine system, Massachusetts. Aquat. Bot. 32:353-363.

Sen, P.K. 1968. Estimates of regression coefficients based on the Kendall's tau. J. Am. Stat. Assoc. 63:1379-1389.

Setchell, W.A. 1929. Morphological and phenological notes on *Zostera marina* L. Univ. Calif. Publ. Bot. 14:389-452.

Setchell, W.A. 1935. Geographic elements of the marine flora of the North Pacific Ocean. Am. Nat. 69:560-577.

Short, F.T. 1988. Eelgrass-scallop research in the Niantic River: Final report to the Waterford-East Lyme Shellfish Commission. November 15, 1988. 12 pp.

Stauffer, R.C. 1937. Changes in the invertebrate community of a lagoon after disappearance of the eelgrass. Ecology 18:427-431.

Stevenson, J.C. and N.M. Confer. 1978. Summary of available information on Chesapeake Bay submerged vegetation. US Fish Widl. Serv FWS/OBS-78/66/

Taylor, D.I., S.W. Nixon, S.L. Granger, B.A. Buckley, J.P. McMahon, and H.-J. Lin. 1995. Responses of coastal lagoon plant communities to different forms of nutrient enrichment - a mesocosm experiment. Aquat. Bot. 52:19-34.

Thayer, G.W., S.M. Adams, and M.W. LaCroix. 1975. Structural and functional aspects of a recently established *Zostera marina* community. pp. 518-540 *in* L.E. Cronin (ed). Recent Advances in Estuarine Research. Acad. Press, New York.

Thayer, G.W., W.J. Kenworthy, and M.S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic coast: A community profile. FWS/OBS-84-02. 147 pp.

Thompson, V.S. 1978. Final Report. Environmental Surveillance and Studies at the Maine Yankee Nuclear Generating Station 1969-1977. Section 5. Estuary Water Temperatures.

Tutin, T.G. 1942. *Zostera*. J. Ecol. 30:217-226.

Uphof, J.C.T. 1941. Halophytes. Bot. Rev. 7:1-58.

Vermaat, J.E., and F.C.A. Verhagen. 1996. Seasonal variation in the intertidal seagrass *Zostera noltii* Hornem.: coupling demographic and physiological patterns. Aquat. Bot. 52:259-281.

Vergeer, L.H.T., T.L. Aarts, and J.D. deGroot. 1995. The 'wasting disease' and the effect of abiotic factors (light intenstiy, temperature, salinty) ant infection with *Labyrinthula zosterae* on the phenolic content of *Zostera marina* shoots. Aquat. Bot. 52:35-44.

Wood, E.J.F., W.E. Odum, and J.C. Zieman. 1969. Influence of seagrasses on the productivity of coastal lagoons. Pages 495-502 *in* A. Ayala Castanares and F.B. Phleger (eds). Coastal Lagoons. Univ. Nac. Autonoma de Mexico, Ciudad Univ., Mexico, D.F.

Zieman, J.C. Jr. 1970. The effects of a thermal effluent stress on the seagrasses and macro-algae in the vicinity of Turkey Point, Biscayne Bay, Florida. Ph.D. Dissertation, Univ. Miami. Coral Gables, Fla. 129 pp.

Zieman, J.C. Jr. 1972. Origin of circular beds of *Thalassia* (Spermatophyta: Hydrocharitaceae) in Southern Biscayne Bay, Florida, and their relationship to mangrove hammocks. Bull. Mar. Sci. 22:559-574.

Zieman, J.C. Jr. 1982. The ecology of the seagrasses of South Florida: A community profile. U.S. Fish and Wild. Service. FWS/OBS-82/25.124. 26 pp.

Zimmerman, R.C., R.D. Smith and R.A. Alberte. 1989. Thermal acclimation and whole-plant carbon balance in *Zostera marina* (eelgrass). J. Exp. Mar. Biol. Ecol. 130:93-109.

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Benthic Infauna

Introduction

Long Island Sound benthic habitats in the vicinity of Millstone Nuclear Power Station (MNPS) support rich and diverse infaunal invertebrate communities. These communities are important as a source of food for numerous invertebrate and vertebrate species, including lobsters and 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). The burrowing and tube-building activities of infauna also promote nutrient recycling from sediments to the water column (Goldhaber et al. 1977; Aller 1978; Gaston and Nasci 1988), and promote the passage of oxygenated water deeper into the sediments.

The close association of benthic communities with the sediments, where most pollutants ultimately accumulate, also makes them effective indicators of acute and chronic environmental conditions (Diaz and Schaffner 1990; Warwick et al. 1990; Somerfield et al. 1995). 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; Prena 1995; Somerfield et al. 1995) provide a framework of baseline studies that may be used to evaluate impacts on benthic marine systems.

Environmental variability is inherent to coastal benthic systems (Holland 1985; Nichols 1985; Holland et al. 1987; Warwick 1988; Rees and Eleftheriou 1989; Boero 1994). Natural variability, together with an incomplete knowledge of how physical and biological factors influence the structural and functional ecology of benthic communities (Diaz and Schaffner 1990), hinders the ability to describe those communities. Thus, long-term monitoring studies are necessary to assess changes in marine environments (Thrush et al. 1994; Prena 1995). Such studies are the principal means of characterizing changes in species composition and fluctuations in abundance, which occur in response to acute or chronic climatic conditions (Boesch et al. 1976; Flint 1985; Jordan and Sutton 1985), variations in biological factors, such as competition and predation (e.g., Levinton and Stewart 1982; Woodin 1982; Kneib 1988), and human activities.

The MNPS ecological monitoring program has included studies of soft-bottom subtidal infauna in the vicinity of the power plant 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 to regional shifts in species composition and abundance that apparently were the result of natural events. This report presents results from the 1996 sampling year, and compares them to results summarized from 2-unit (1980-85) and 3-unit (1986-1996) operational periods at MNPS.

Materials and Methods

Subtidal infaunal communities in the vicinity of MNPS were sampled twice per year (June and September) from 1980 through 1996 at four stations (Fig. 1). A sampling year is comprised of collections made in June and September of the calendar year. 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 occurring independently of power plant operations. The intake station (IN), located 100 m seaward of MNPS Unit 2 and Unit 3 intake structures, 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 and 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 potentially impacted by 3-unit operations. The area encompassing this station experiences increases in surface water temperatures of 0.8 to 2.2°C above ambient during certain tidal stages (primarily ebb tide) due to the 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 brought to the surface. Samples were taken to the laboratory, where they were fixed with 10% buffered formalin. After a minimum of 48 hours, organisms were floated from the sediments onto a 0.5 mm mesh sieve and preserved in a 70% ethanol solution with Rose Bengal added to facilitate sample processing. Samples were examined using dissecting microscopes (10x); 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 too small to be quantitatively sampled by our methods (meiofauna; e.g., nematodes, ostracods, copepods, and foraminifera) were not sorted. Grain size and 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

Sediments

Sediment sieve fractional weights were used to construct cumulative curves for 2-unit (1980-85) and 3-unit (1986-96) operational periods by pooling the June and September 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 describes 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:

$$C_\phi = 100 \exp(-\beta e^{-K\phi})$$

where C_ϕ is the cumulative sediment weight at a given particle size (ϕ), and β and K 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 and for the current year using non-linear regression methods. Two-sample t-tests were used to test for differences (α=0.05) between the β and K parameters of curves, based on data collected during each operational period.

Trend Analysis

The nonparametric (i.e., distribution-free) Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether the 2-unit and 3-unit time 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 monthly data (June and September), and a graphical representation of the linear trend are provided for community abundance, numbers of species, and for selected taxa.

Community Analyses

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 taxon i in year j; X_{ik} is the abundance in year k; and n is the number of taxa in common, for which, on average, at least two individuals were found per year. A flexible-sorting (β = -0.25), clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967).

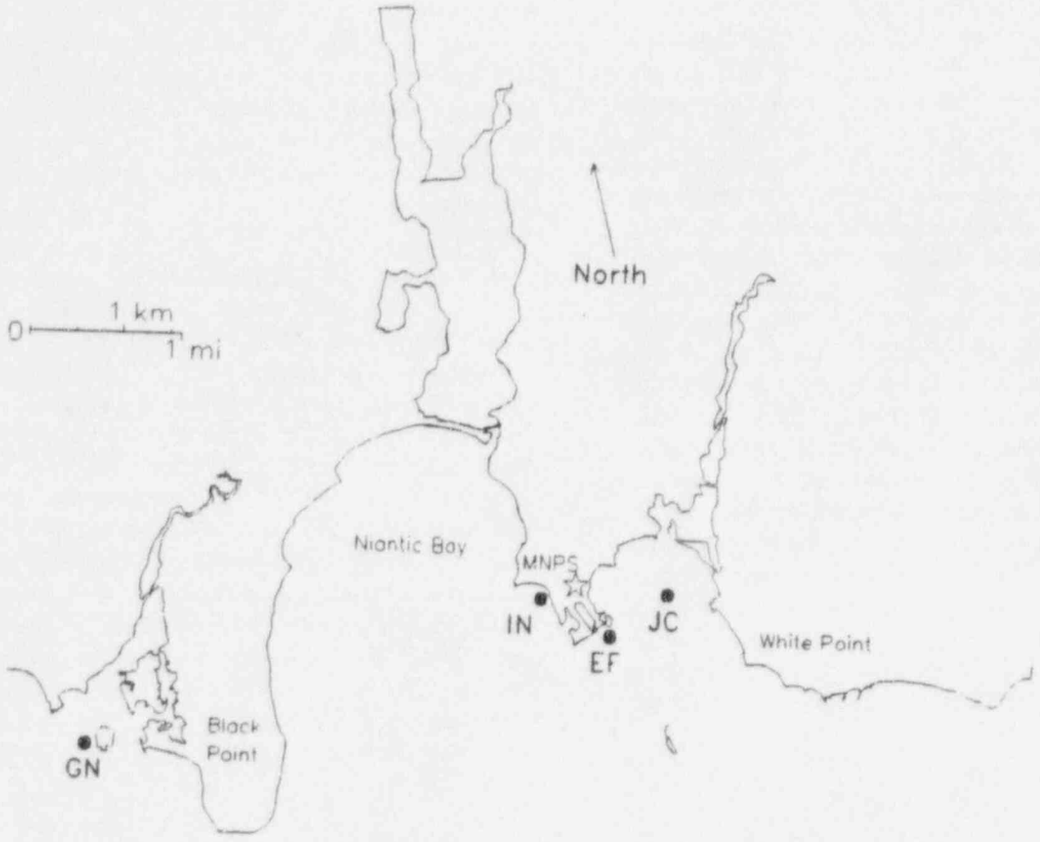

Fig. 1. Map of the Millstone Point area showing the location of subtidal infaunal sampling stations (EF=Effluent, GN=Giants Neck, IN=Intake, JC=Jordan Cove) established as part of the long-term monitoring program for Millstone Nuclear Power Station.

Results

Sedimentary Environment

Parameters used to characterize the sedimentary environment at infaunal sampling stations in the vicinity of MNPS included analysis of mean grain size and silt/clay content (Fig. 2). Based on mean grain size, sediments in 1996 were finest at JC (0.17 mm and 0.26 mm in June and September, respectively) and coarsest at EF (0.52 mm and 0.54 mm). Mean grain sizes were intermediate at IN (0.29 mm and 0.31 mm) and GN (0.40 mm and 0.44 mm). Silt/clay contents of sediments collected in 1996 were highest at JC (16.1% and 24.9% in June and September, respectively), lowest at EF (1.3% and 1.8%), and intermediate at GN (12.2% and 12.8%) and IN (9.3% and 9.6%). Mean grain sizes and silt clay contents observed at all stations during 1996 were within the ranges of previous years of 3-unit operation. Mean grain sizes at EF in 1996 were above the 2-unit period range, and silt/clay content was below the 2-unit range. Conversely, at JC in 1996, mean grain sizes were below the 2-unit range, and silt/clay was above the 2-unit range.

Cumulative curves based on sediment sieve fraction weights (Fig. 3) were used to characterize sedimentary environments, and allowed statistical comparison of sediments collected during 1996, and 2-unit (1980-85) and 3-unit (1986-96) operational periods. Based on t-tests of Gompertz parameters estimated by fitting Gompertz curves to the data, significant differences between 2-unit and 3-unit periods were noted at the EF and JC stations. The shift at EF reflected the declining silt/clay fraction and the slightly larger grain size since Unit 3 began operation (Fig. 2). Conversely, at JC, increased

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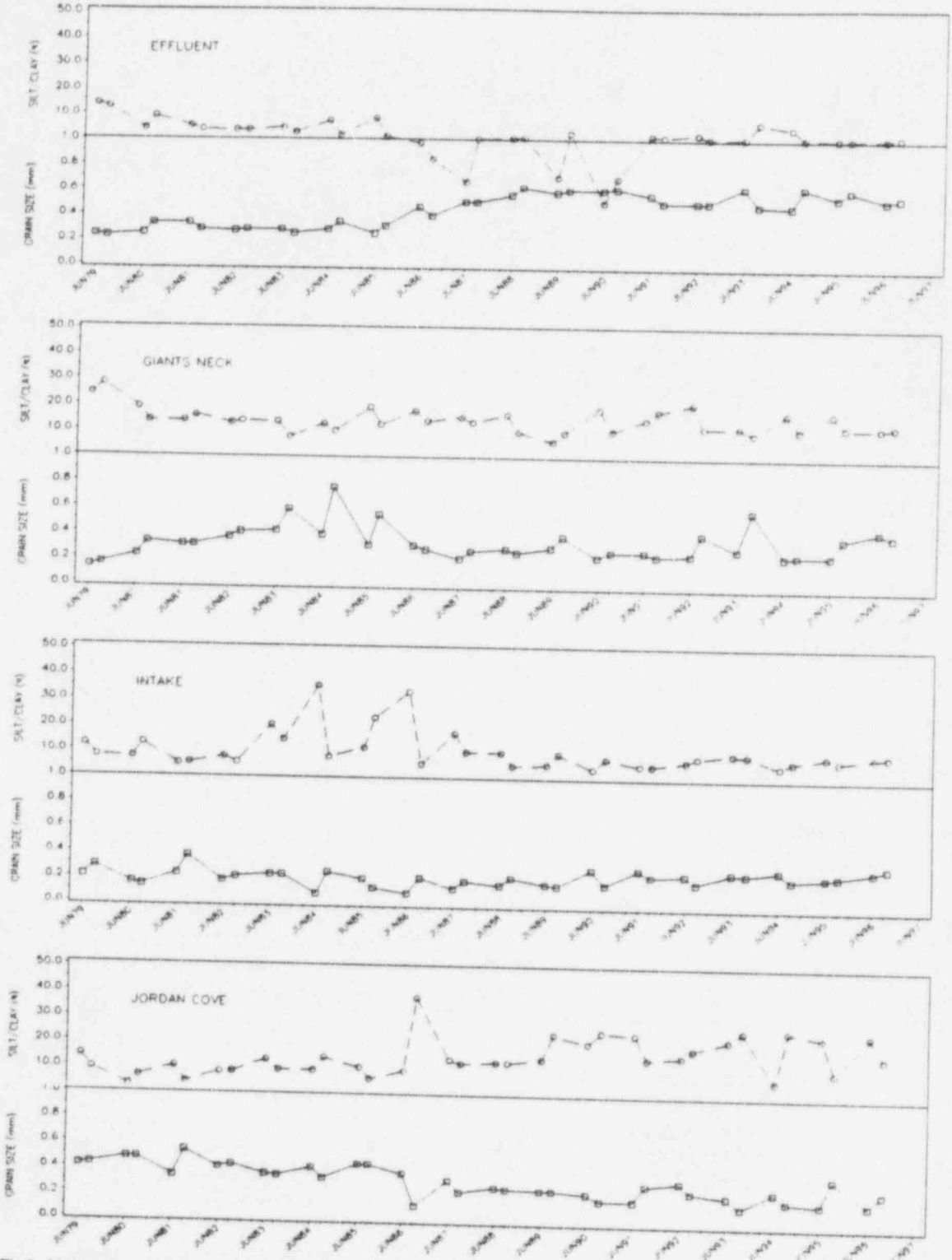

Fig 2. Mean grain size (mm) and silt/clay content (%) of sediments at Millstone subtidal sand stations for June and September from 1979 to 1996.

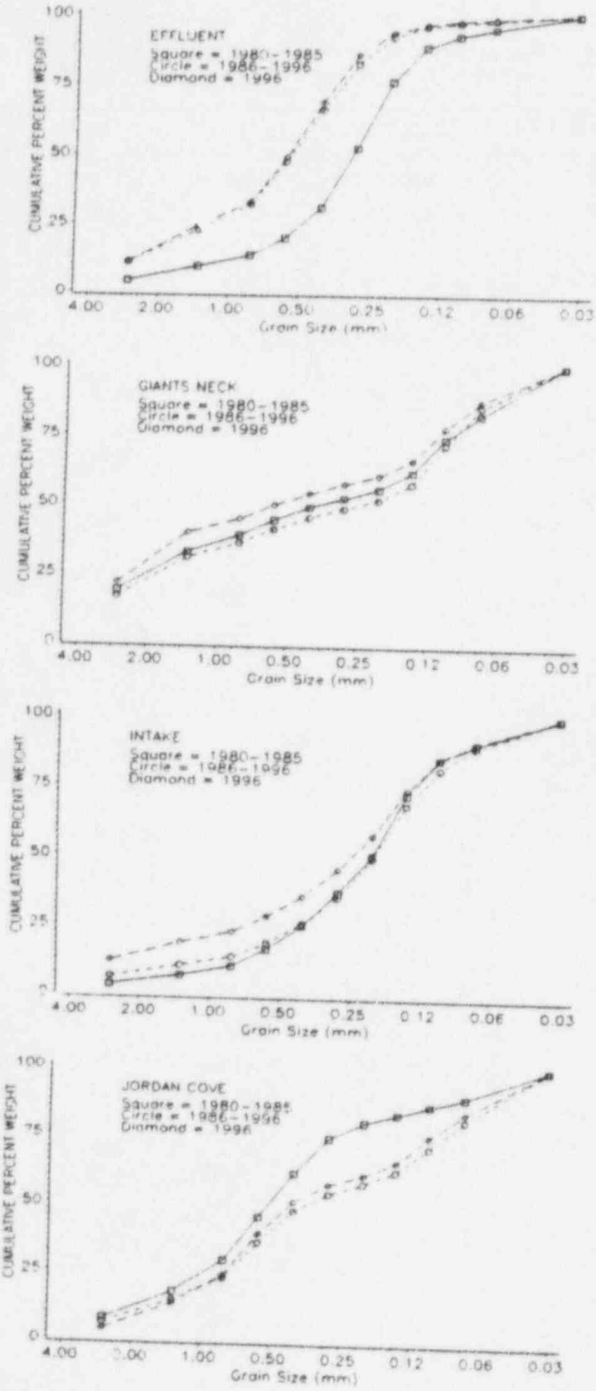

Fig. 3. Cumulative curves based on fractional weights of sediment collected during the 2-unit (1980-1985) and 3-unit (1986-1996) operational periods, and during 1996, at Millstone subtidal stations.

silt/clay content and decreased average grain size over the same time period distinguished 3-unit sediments from 2-unit sediments. This trend of finer sediments at JC during 3-unit operation continued through 1996, as these sediments (based on cumulative curves) were also significantly different from those collected during 2-unit operation.

General Community Composition

Parameters used to monitor general benthic community composition included numbers of species and of individuals in major invertebrate groups collected during 1996, and means of these parameters for 2-unit and 3-unit operational periods (Table 1). Number of species in 1996 was highest at EF (94), lowest at JC and IN (80) and intermediate at GN (86). At both GN and JC, the numbers of species in 1996 were lower than 2-unit and 3-unit operational period means (86 vs. 107 and 92, respectively at GN; and 80 vs. 100 and 93, respectively at JC). At EF, the number of species in 1996 was the same as the 3-unit mean (94), but less than the 2-unit mean (110). The 1996 number of species at IN was greater than the 2-unit mean (68), but less than the 3-unit mean (81). Among-station relationships were similar between operational periods; mean number of species was highest at EF, lowest at IN, and the intermediate numbers of species at JC and GN were similar to each other.

Total number of organisms collected in 1996 was highest at JC (6,311), lowest at IN (1,948), and intermediate at GN (4,581) and EF (2,889). The relationships among sampling sites over both operational periods were similar to those in 1996: highest abundance at JC (7,115 and 7,578, during 2-unit and 3-unit periods, respectively), and lowest at IN (1,565 and 3,048, respectively). Total abundance in 1996 was low at EF, GN and JC relative to operational period means. At IN, 1996 total abundance was lower than the 3-unit mean, but higher than the 2-unit mean.

Most of the invertebrate species identified in 1996 were polychaetes; the numbers of polychaete species ranged from 46 to 50 (Table 1). Polychaetes had also been the dominant taxon in the 2-unit and 3-unit operating periods (species number ranges of 35-58 and 44-53, respectively). Polychaetes were also most abundant in terms of numbers of individuals, accounting for more than half of the total individuals at all stations but EF. Mollusc and

TABLE 1. Total number of species (S) and number of individuals (N) for each major taxon collected at the MNPS infaunal stations during 1996, and annual means and Coefficient of Variation (CV) during 2-unit (1980-1985) and 3-unit (1986-1995) operational years.

	1996		2-Unit Period (1980-1985)				3-Unit Period (1986-1996)			
	(S)	(N)	MEAN (S)	CV*	MEAN (N)	CV	MEAN (S)	CV	MEAN (N)	CV
Effluent										
Polychaeta	50	624	58	3.7	2431	16.9	51	2.8	1184	13.0
Oligochaeta	-	1816	-	-	1809	8.9	-	-	2120	8.1
Mollusca	21	230	21	3.1	303	22.6	20	5.7	253	16.0
Arthropoda	23	176	31	5.7	520	19.7	23	6.5	251	14.2
Rhynchocoela	-	43	-	-	104	31.5	-	-	92	34.4
Total	94	2889	110		5167		94		3900	
Giants Neck										
Polychaeta	50	3034	58	4.4	3654	1.8	51	2.9	3745	10.3
Oligochaeta	-	1170	-	-	1000	14.0	-	-	1134	8.7
Mollusca	18	113	20	9.5	188	15.0	19	4.9	153	13.3
Arthropoda	18	221	29	4.7	517	32.4	22	8.7	615	40.8
Rhynchocoela	-	43	-	-	39	26.4	-	-	37	15.4
Total	86	4581	107		5398		92		5684	
Intake										
Polychaeta	46	1060	35	4.1	838	19.3	44	4.2	1761	19.6
Oligochaeta	-	435	-	-	131	18.7	-	-	308	18.9
Mollusca	14	180	12	18.0	156	31.9	15	6.2	235	14.4
Arthropoda	20	256	21	9.9	432	14.1	22	5.5	732	50.2
Rhynchocoela	-	17	-	-	8	26.2	-	-	12	14.9
Total	80	1948	68		1565		81		3048	
Jordan Cove										
Polychaeta	47	4407	55	4.1	3972	24.1	53	1.9	5076	10.6
Oligochaeta	-	1405	-	-	2540	16.1	-	-	1413	7.2
Mollusca	18	386	22	14.1	338	25.2	21	4.1	441	7.4
Arthropoda	15	102	23	12.4	219	19.7	19	5.6	605	38.2
Rhynchocoela	-	11	-	-	46	27.4	-	-	43	16.7
Total	80	6311	100		7115		93		7578	

* C.V. of the mean estimate = (Standard Error/Mean) x 100

arthropod species were not as numerous as polychaetes; ranges for numbers of species in 1996 were 15-23 (Arthropoda) and 14-21 (Mollusca). Numbers of mollusc and arthropod species in 1996 were generally consistent with operational period means. At EF during 3-unit operation and during 1996, oligochaetes were dominant. At GN and JC, oligochaetes were second in abundance (after polychaetes) during 1996 and over both operational periods, followed by either arthropods or molluscs (Table 1). Oligochaetes also ranked second in 1996 at IN, but ranked fourth and third during the 2-unit and 3-unit periods, respectively. Arthropods ranked third in abundance at IN during 1996, but ranked second during both 2-unit and 3-unit operational periods. Rhynchocoels contributed little to total invertebrate abundance at any station during either operational period.

Changes in community composition between the 3-unit and 2-unit operational periods occurred at three of four sampling stations. The polychaete-dominated community at EF, present during 2-unit operation, was replaced during the 3-unit period by one dominated by oligochaetes. Abundances of polychaetes approached those of oligochaetes at times during recent years (*e.g.*, NUSCO 1993, 1994), but during most 3-unit years, including 1996, oligochaetes remained dominant. Also, the number of arthropod species at EF was lower during 3-unit operation and 1996 than during the 2-unit period. The opposite trend was observed at JC; i.e., more polychaetes and arthropods and fewer oligochaetes were collected in the 3-unit period than in the 2-unit period. At IN, the numbers of all taxonomic groups was greater than during 3-unit than 2-unit operation. At GN, there was little difference in community composition between operational periods.

Faunal Abundance

Mean faunal densities for the June and September collections during 1996 were 176 and 149 individuals per core at EF, 249-236 at GN, 112-68 at IN, and 420-436 at JC (Fig. 4). During 1996, densities were highest in the June collections at all stations except JC. Faunal densities at each station were within their respective historic ranges established since 1980. In general, infaunal abundance at all stations during the 3-unit period (1986-96) was similar to that observed during the 2-unit operational years. However, analyses of long-term trends in community abundance indicate a significant ($p>0.01$) decreasing trend in number of individuals collected at EF during 3-unit operation. There were no significant trends in community abundance evident at GN, IN or JC during either operational period.

Numbers of Species

Mean numbers of species in 1996 June and September (respectively) collections were 22 and 21 at EF, 26-23 at GN, 21-15 at IN, and 26-27 at JC (Fig. 5). These means were within the range of values observed at each station over the previous sampling periods. A significant increasing trend in species number was evident at EF, GN and JC during 2-unit operation, that resulted from high numbers of species collected at these stations just prior to the 3-unit period. During the 3-unit operating period, there were significant decreasing trends at EF, GN and JC. However, at IN, there were no significant trends during either operational period.

Community Dominance

Infaunal community structure, based on relative abundance of common taxa, is summarized for 1996 and 3-unit and 2-unit operational periods (Table 2). In 1996, infaunal communities were characterized by only a few clearly dominant (>25% of total abundance) taxa and included representatives of the class Oligochaeta, and the polychaete species *Aricidea catherinae* and *Tharyx* spp. Other dominant taxa in 1996 included the mollusc *Tellina agilis*, the polychaete species *Parapionosyllis longicirrata*, *Protodorvillea gaspeensis*, *Polycirrus eximius*, *Exogone dispar*, and *Mediomastus ambiseta*, and representatives of the polychaete family Maldanidae. The five most abundant taxa at each station in 1996 accounted for more than 60% of all individuals, and included: Oligochaeta, *Tellina agilis*, *Parapionosyllis longicirrata*, *Protodorvillea gaspeensis*, and *Aricidea catherinae* at EF; *Tharyx* spp., Oligochaeta, *Polycirrus eximius*, *Exogone dispar*, and *Aricidea catherinae* at GN; Oligochaeta, *Aricidea catherinae*, *Tharyx* spp., *Protodorvillea gaspeensis*, and *Mediomastus ambiseta* at IN; and

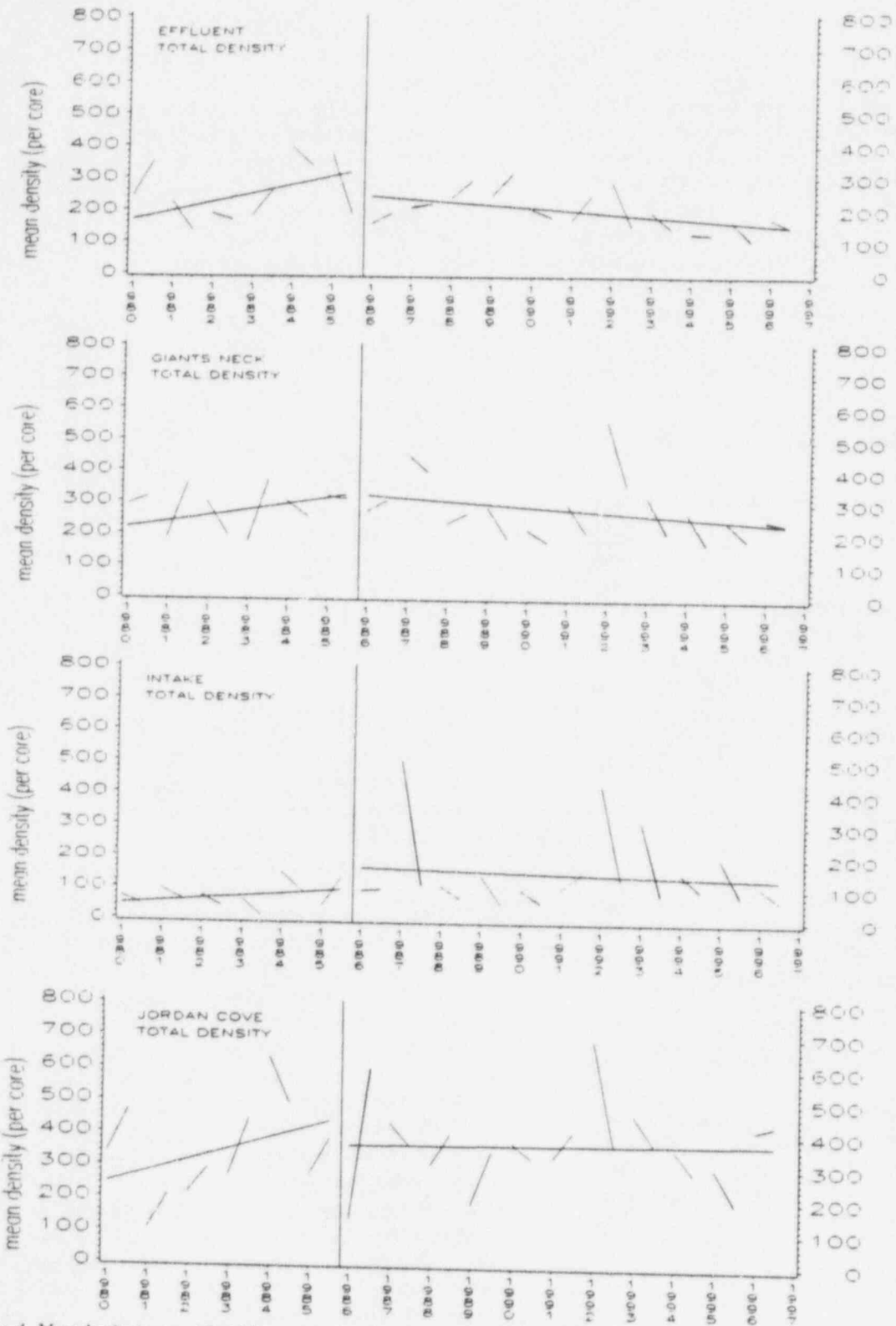

Fig. 4. Mean density (per core) for Millstone subtidal stations in June and September for every year from 1980 through 1996.

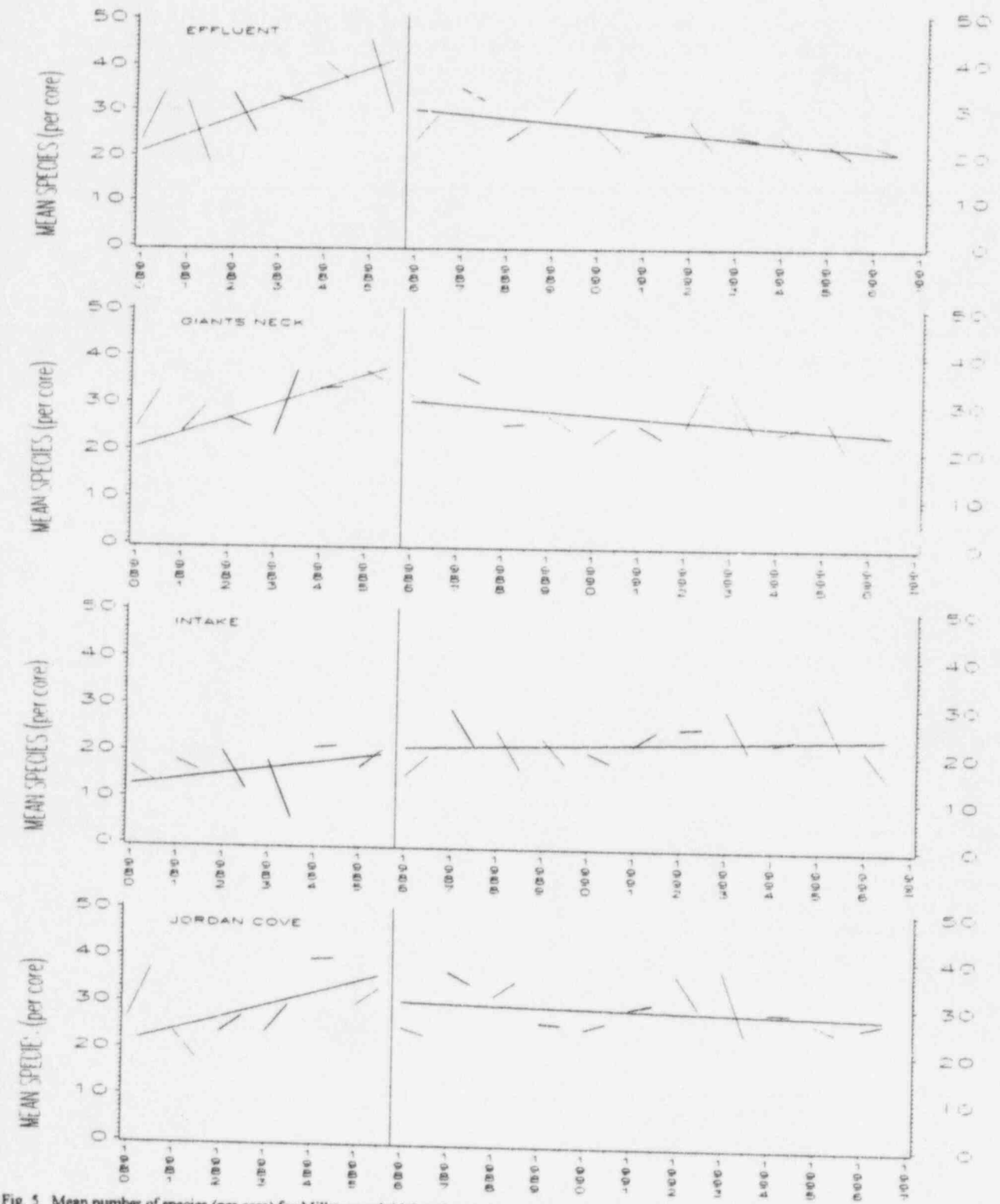

Fig. 5. Mean number of species (per core) for Millstone subtidal stations in June and September samples for every year from 1980 through 1996.

Table 2. Relative abundance (%) and coefficient of variability (CV) for each of the ten most abundant taxa collected in the June and September periods at the Millstone subtidal stations during 3-Unit operational years (1986-1996) and 2-Unit operational years (1980-1985).

	1996	3-Unit 1986-1996		2-Unit 1980-1985	
Effluent	%	%	CV*	%	CV
Oligochaeta	64.3	53.4	8.1	35.6	8.9
Tellina agilis	4.7	2.5	19.1	2.7	27.4
Parapionosyllis longicirrata	4.1	1.6	19.4	0.7	27.6
Protodorvillea gaspeensis	3.9	6.1	14.2	4.1	11.8
Aricidea catherinae	3.7	1.1	26.3	2.6	26.1
Tharyx spp.	2.4	1.7	21.1	8.5	64.9
Rhynchocoela	1.4	2.4	34.4	1.8	31.5
Ampelisca vadorum	1.2	1.2	29.0	0.6	52.2
Pagurus acadianus	1.1	1.5	14.7	0.9	52.2
Polydora caulleryi	0.7	0.5	23.9	1.2	53.9
Polycirrus eximius	0.1	3.1	45.8	10.6	41.2
Mediomastus ambiseta	0.5	1.3	51.1	2.4	60.3
Eumida sanguinea	0.0	0.3	51.3	1.7	27.7
Exogone hebes	0.2	0.2	14.6	1.6	15.5
Prionospio steenstrupi	0.7	2.6	43.9	1.0	29.0
Haliplanella luciae	0.0	1.8	46.4	0.6	63.4
Giants Neck					
Tharyx spp.	45.7	22.6	14.8	18.3	10.6
Oligochaeta	28.7	21.6	8.8	17.3	14.0
Polycirrus eximius	2.9	2.2	17.5	2.3	20.4
Exogone dispar	2.5	2.2	12.1	1.9	24.1
Aricidea catherinae	2.3	10.8	18.4	20.3	16.3
Protodorvillea gaspeensis	1.8	2.8	12.2	3.1	13.9
Harmothoe imbricata	1.4	0.7	19.2	0.6	24.6
Phoxocephlus holbolli	1.1	1.2	13.7	2.3	21.9
Polydora caulleryi	1.0	0.7	28.9	1.9	34.5
Mediomastus ambiseta	0.9	6.3	38.6	6.4	38.6
Prionospio steenstrupi	0.4	7.3	62.3	1.9	29.8
Ampelisca vadorum	0.2	2.3	52.4	0.2	64.6
Scoletema tenuis	0.5	2.0	17.6	2.5	14.7

Table 2. Con't.

	1996	3-Unit 1986-1996		2-Unit 1980-1985	
Intake	%	%	CV*	%	CV
Oligochaeta	27.1	11.7	18.9	10.1	18.7
Aricidea catherinae	13.8	9.5	22.0	8.4	37.9
Tharyx spp.	7.7	3.9	24.5	4.1	34.4
Protodorvillea gaspeensis	6.9	2.8	21.4	1.2	56.6
Mediomastus ambiseta	4.7	5.9	39.8	7.8	58.3
Tellina agilis	3.6	3.3	22.6	3.3	32.7
Gammarus lawrencianus	3.4	1.2	37.2	2.9	65.8
Exogone hebes	3.0	6.1	18.0	4.6	35.9
Ampelisca vadorum	2.2	2.9	30.0	1.6	49.5
Maldanidae	2.1	0.6	28.8	5.2	97.8
Ampelisca verrilli	0.0	2.5	35.7	6.1	42.0
Capitella spp.	0.2	3.9	26.9	4.2	38.9
Ampelisca abdita	0.0	1.6	85.5	3.5	55.4
Leptocheirus pinguis	0.3	5.8	90.1	3.4	73.1
Nucula proxima	0.7	3.7	32.0	2.8	47.0
Jordan Cove					
Aricidea catherinae	35.7	18.5	18.8	14.3	6.6
Oligochaeta	17.4	18.6	7.2	38.0	16.1
Tharyx spp.	10.2	4.0	21.6	3.8	24.9
Maldanidae	9.3	1.2	79.2	0.0	22.0
Mediomastus ambiseta	5.0	14.0	39.2	13.1	51.7
Scoletema tenuis	4.4	5.0	10.0	4.9	30.0
Polycirrus eximius	2.3	4.4	28.9	4.0	31.3
Prionospio steenstrupi	2.0	9.1	52.1	0.7	38.8
Leptocheirus pinguis	1.5	6.2	45.0	0.8	29.5
Cossura longocirrata	1.3	0.7	21.8	0.0	-
Nucula proxima	1.3	2.3	16.8	0.4	24.5
Capitella spp.	0.1	1.5	27.6	1.3	19.3
Polydora caulieryi	0.0	0.2	41.2	2.2	68.4
Tellina agilis	1.1	1.2	18.2	1.6	30.5
Eumida sanguinea	0.0	0.2	43.9	0.9	42.7

*C.V. = (Standard Error/Mean) x 100

Aricidea catherinae, Oligochaeta, *Tharyx* spp., Maldanidae, and *Mediomastus ambiseta* at JC. In most instances, these organisms were also dominant taxa during both 2-unit and 3-unit operational periods.

Year-to-year changes in rankings of dominant taxa have been observed since 1980, with 1996 being no exception. Some shifts in relative abundance in 1996 were noted at several stations. For example, oligochaete relative abundance in 1996 was higher at EF (64.3%), GN (28.7%) and IN (27.1%) compared to 2-unit (35.6%, 17.3% and 10.1%, respectively) and 3-unit (53.4%, 21.6% and 11.7%, respectively) operational periods. Oligochaete relative abundance at JC in 1996 (17.4%) was comparable to that during the 3-unit period (18.6%), but lower than during the 3-unit period (38.0%). High relative abundance of *Tharyx* spp. was also noted at three stations in 1996: GN (45.7% in 1996 vs. 18.3% and 22.6% during 2-unit and 3-unit periods, respectively); IN (7.7% vs. 4.1% and 3.9%); and JC (10.2% vs. 3.8% and 4.0%). Relative abundance of *Tharyx* spp. at EF in 1996 (2.4%) was consistent with low levels observed over the 3-unit period (1.7%), when compared to the 2-unit period (8.5%). Decreases in relative abundance of *Mediomastus ambiseta* in 1996 were observed at GN (0.9%) and JC (5.0%), as higher percentages had been noted over 2-unit and 3-unit periods (6.4% and 6.3%, respectively at GN; 13.1% and 14.0%, respectively at JC).

Some of the shifts in community dominance hierarchy in 1996 were unique to one station. *A. catherinae* was the top-ranked taxon at JC in 1996; relative abundance in 1996 (35.7%) was approximately two times higher than during both operational periods when this species ranked second. Conversely, relative abundance of *A. catherinae* was reduced considerably in 1996 at GN (2.3%; fifth ranked), while this polychaete comprised 20.3% (first ranked) and 10.8% (third ranked) of the total individuals collected during 2-unit and 3-unit periods, respectively. *Polycirrus eximius* was not among the top ten dominants at EF in 1996, accounting for only 0.1% of the total individuals collected. This species had ranked second (10.6%) and third (3.1%) at EF during 2-unit and 3-unit periods, respectively. Similarly, three amphipod species (*Ampelisca verrilli*, *A. abdita*, and *Leptocheirus pinguis*) and one mollusc (*Nucula proxima*) that had been common components of 2-unit and 3-unit infaunal communities at IN were rare or absent in collections made in 1996. Other notable shifts in 1996, relative to overall operational periods, include increases in relative abundance of *Protodorvillea gaspeensis* at IN and of Maldanidae at JC, and a decrease in relative abundance of *Prionospio steenstrupi* at GN.

More consistency in community dominance hierarchy was apparent between 2-unit and 3-unit operational periods than among years. For most stations, infaunal communities during both periods were characterized by three or fewer clearly dominant taxa, i.e., oligochaetes at EF, *Tharyx* spp., oligochaetes, and *Aricidea catherinae* at GN, and oligochaetes, *A. catherinae*, and *Mediomastus ambiseta* at JC. There were, however, some notable shifts in relative abundance of some dominant taxa between 2-unit and 3-unit periods. Oligochaetes were more dominant at EF during 3-unit operation (53.4%) than during 2-unit operation (35.6%). Concomitant with this increased dominance of oligochaetes were decreases in abundance of polychaetes species at EF during 3-unit operation (e.g., *Tharyx* spp. and *Polycirrus eximius*), as discussed previously. Conversely, at JC, oligochaetes decreased from 38.0% relative abundance during 2-unit operation to 18.6% during 3-unit operation; increases were noted during the 3-unit period at JC for *Polycirrus eximius* and *Leptocheirus pinguis*. While relative abundance of community dominants between periods were more consistent at GN than at EF or JC, a substantial 3-unit decrease occurred in relative abundance of *A. catherinae* (from 20.3% to 10.8%) and an increase for *Prionospio steenstrupi*. At IN, there was no clearly dominant taxon during either period, as relative abundances for any taxon rarely exceeded 10% and were generally comparable between periods.

Dominant Taxa

Of the many species of benthic infauna that comprise the local soft-bottom community, eight infaunal taxa have been identified as being affected or potentially affected by construction and operation of MNPS, as well as indicators of natural environmental events (e.g., storms), and are therefore used as biomonitoring tools. 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.

Oligochaetes - Oligochaeta was the most numerous taxon at EF, IN and JC during both 2-unit and 3-unit operating periods, accounting for 35.6-53.4%, 10.1-11.7% and 38.0-18.6% of total individuals, respectively (Table 2). Oligochaetes were also abundant at GN, representing 17.3 and 21.6% of the individuals during 2-unit and 3-unit periods, respectively. Oligochaete abundances during 1996 were highest at EF (97-107/core), lowest at IN (23-27/core), and intermediate at JC (72-74/core) and GN (61-66/core). These densities in 1996 were within the ranges of densities for previous study sample periods (Fig. 6a-d).

Trend analysis of oligochaete abundance during 2-unit operation revealed an apparent but not significant increase at EF (Fig. 6a). Analysis for the 3-unit period indicated a significant increasing trend in oligochaete abundance at IN (Fig. 6c). At GN, densities remained relatively constant during the 2-unit and 3-unit operating periods (Fig. 6b). At JC (Fig. 6d), oligochaete abundance was generally lower and less variable during 3-unit operation, compared to 2-unit operation; no significant trends were noted for either operational period.

Aricidea catherinae - The polychaete *A. catherinae* was among the top ten dominant taxa at all stations during 2-unit operation, and at all stations except EF during the 3-unit operating period (Table 2). Average densities for June and September 1996 were highest at JC at 113-178/core, respectively. Average densities were lowest at GN (5/core) followed by EF (3-10/core), and IN (13-2/core). Monthly average densities at EF and IN in 1996 were within historical ranges, although higher than any previous 3-unit years at EF and lower than recent years at IN. Densities at JC were among the highest recorded, while those at GN were among the lowest (Fig. 6e-h).

Despite the relatively low abundance of *Aricidea catherinae* at IN during 1996, a significant increasing trend was exhibited at this site during the 3-unit operating period (Fig. 6g); a significant increase also occurred at JC (Fig. 6h). In contrast, abundances have significantly declined at GN during 3-unit operation, largely due to low densities in each of the previous 5 years. There was no trend in abundance at any station during the 2-unit operational period.

Mediomastus ambiseta - *M. ambiseta* was the fifth most abundant taxon at IN and JC in 1996 (Table 2), tenth at GN, and twelfth at EF. Highest densities in 1996 were recorded at JC (19-25/core), followed by IN (5/core in both June and September) and GN (2-3/core; Fig. 6i-k). *M. ambiseta* at EF averaged <1/core. These values were within the ranges of 2-unit and previous 3-unit years, but were much lower than the maximum abundances exhibited during area-wide pulses of this opportunistic polychaete (*e.g.*, 1983-84, 1986, 1994-94). This high year-to-year variability contributed to the lack of significant trends in abundance during both operational periods.

***Tharyx* spp.** - *Tharyx* spp. was the numerically dominant taxon collected at GN in 1996 (45.7% of the individuals), third most abundant at IN and JC (7.7 and 10.2% of the individuals, respectively), and fifth at EF (2.4%; Table 2). Densities were highest at GN (104-117/core), lowest at EF (1-7/core), and intermediate at IN and JC (4-10/core and 40-49/core, respectively; Fig. 6l-o). The high values at GN and JC contributed to significant increasing trends at these stations during the 3-unit operating period; at EF and IN, 1996 values were within historical ranges and no trends occurred at these stations during the 3-unit period. In the 2-unit operating period, *Tharyx* spp. at EF exhibited an apparent but not significant increase; however, abundances at GN and JC did increase significantly. No trends in abundance were apparent at IN during either operating period.

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 fifth and seventh, respectively; Table 2). This species was among the dominant taxa at JC in 1996 (ranking seventh); monthly density range was 4-16/core (Fig. 6q). It was not among the dominant taxa collected at EF in 1996, with mean densities of 1/core for each sampling period (Fig. 6p). Like *Mediomastus ambiseta*, *P. eximius* has exhibited region-wide pulses of abundance (e.g., 1983-84, 1991), and considerable year-to-year variability. Trend analysis indicated no significant trends at either site during 2-unit or 3-unit operation.

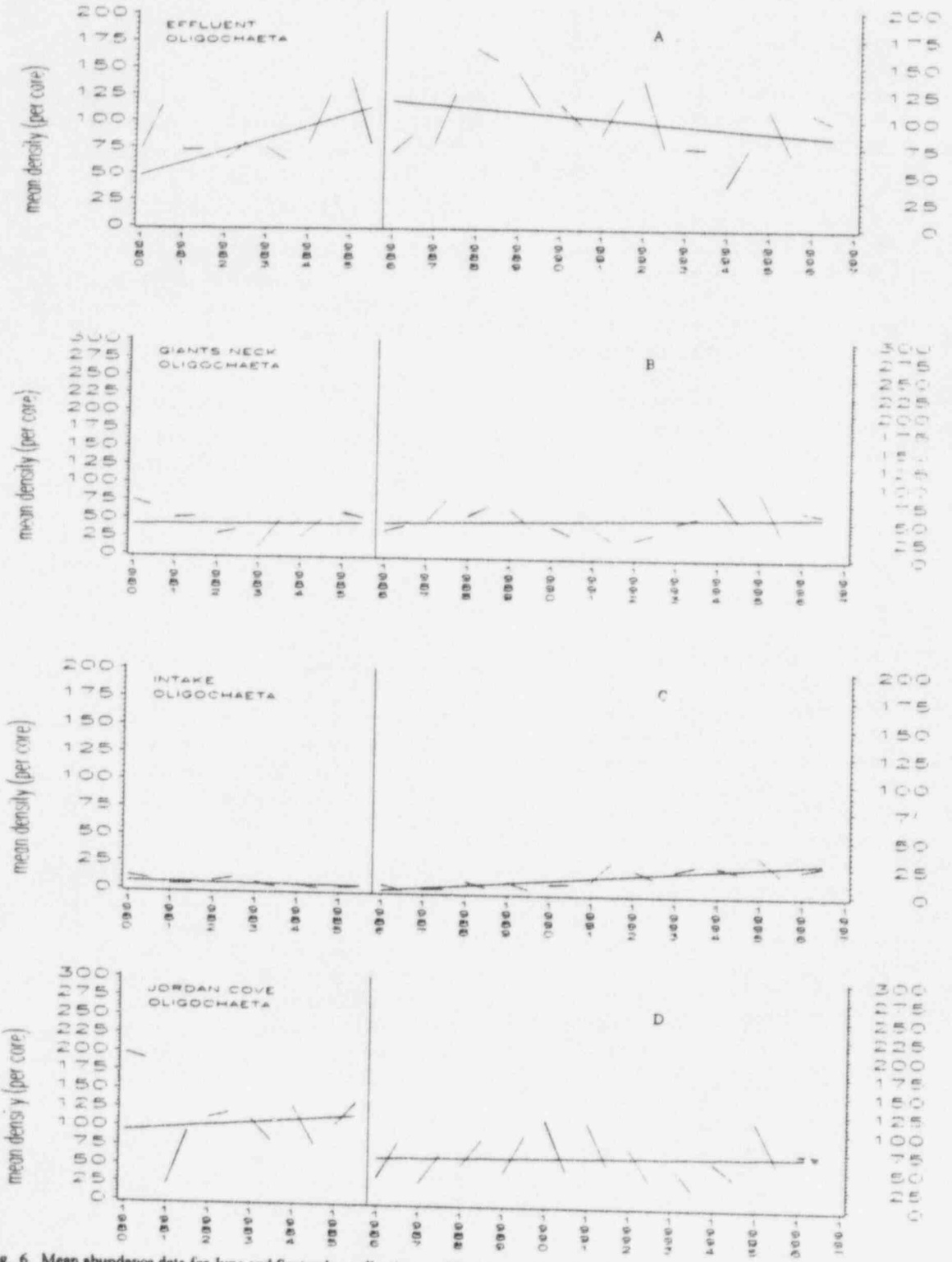

Fig. 6. Mean abundance data for June and September collections and linear regressions for subtidal communities collected during 2-unit (1980-1985) and 3-unit (1986-1996) operating periods.

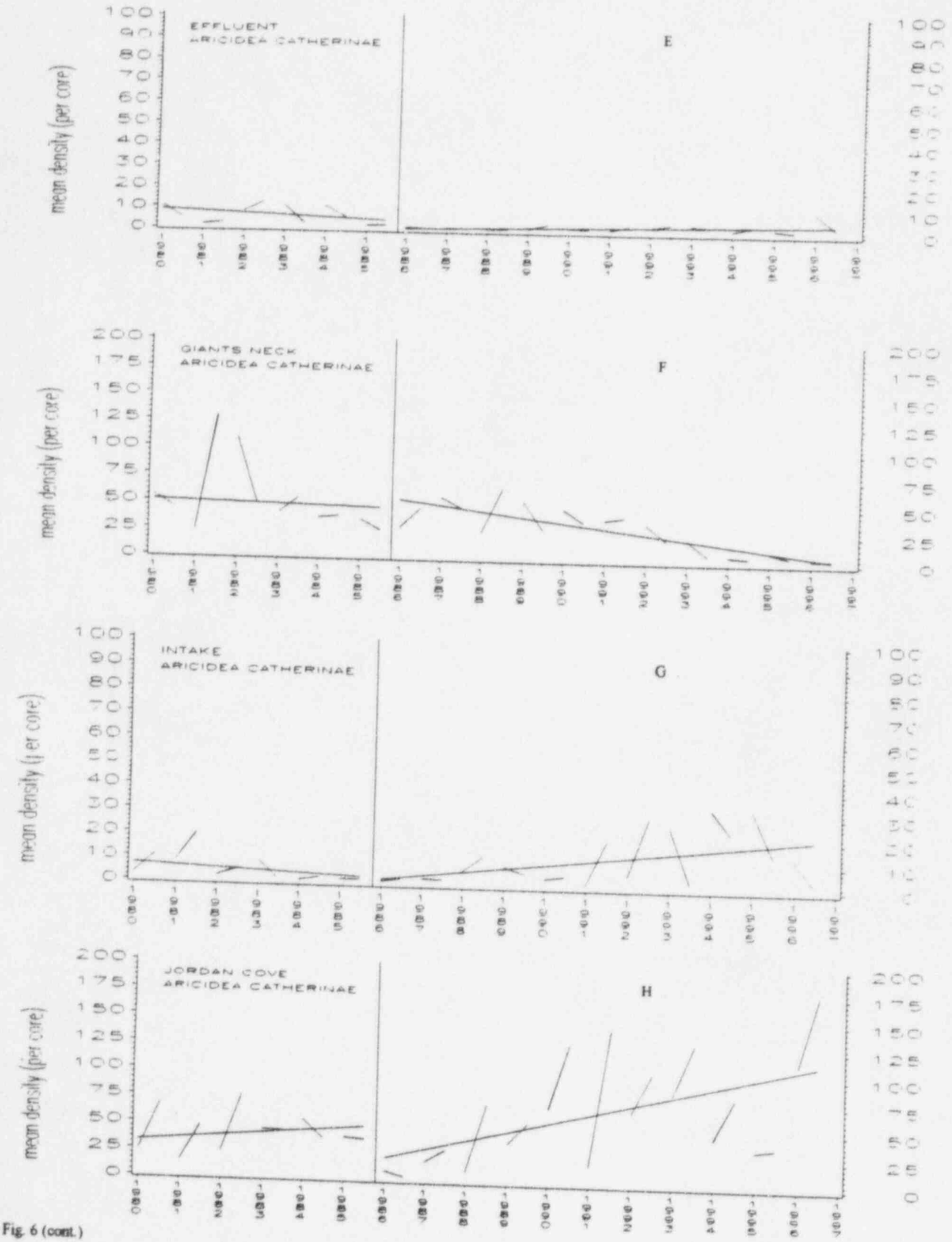
EFFLUENT
ARICIDEA CATHERINAE
E
GIANTS NECK
ARICIDEA CATHERINAE
F
INTAKE
ARICIDEA CATHERINAE
G
JORDAN COVE
ARICIDEA CATHERINAE
H
mean density (per core)

Fig. 6 (cont.)

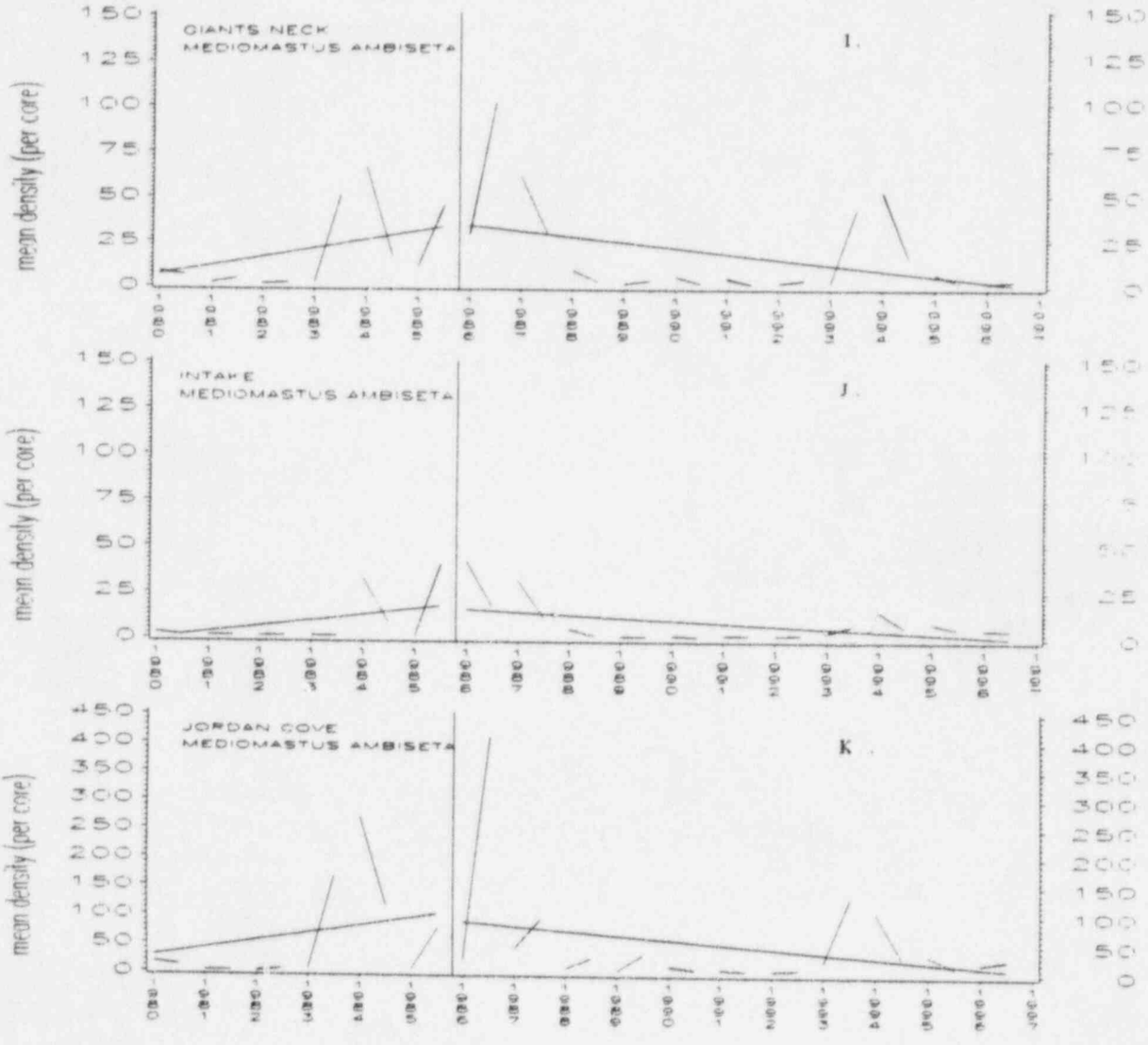
GIANTS NECK
MEDIOMASTUS AMBISETA
I.
INTAKE
MEDIOMASTUS AMBISETA
J.
JORDAN COVE
MEDIOMASTUS AMBISETA
K.
mean density (per core)

Fig. 6 (cont.)

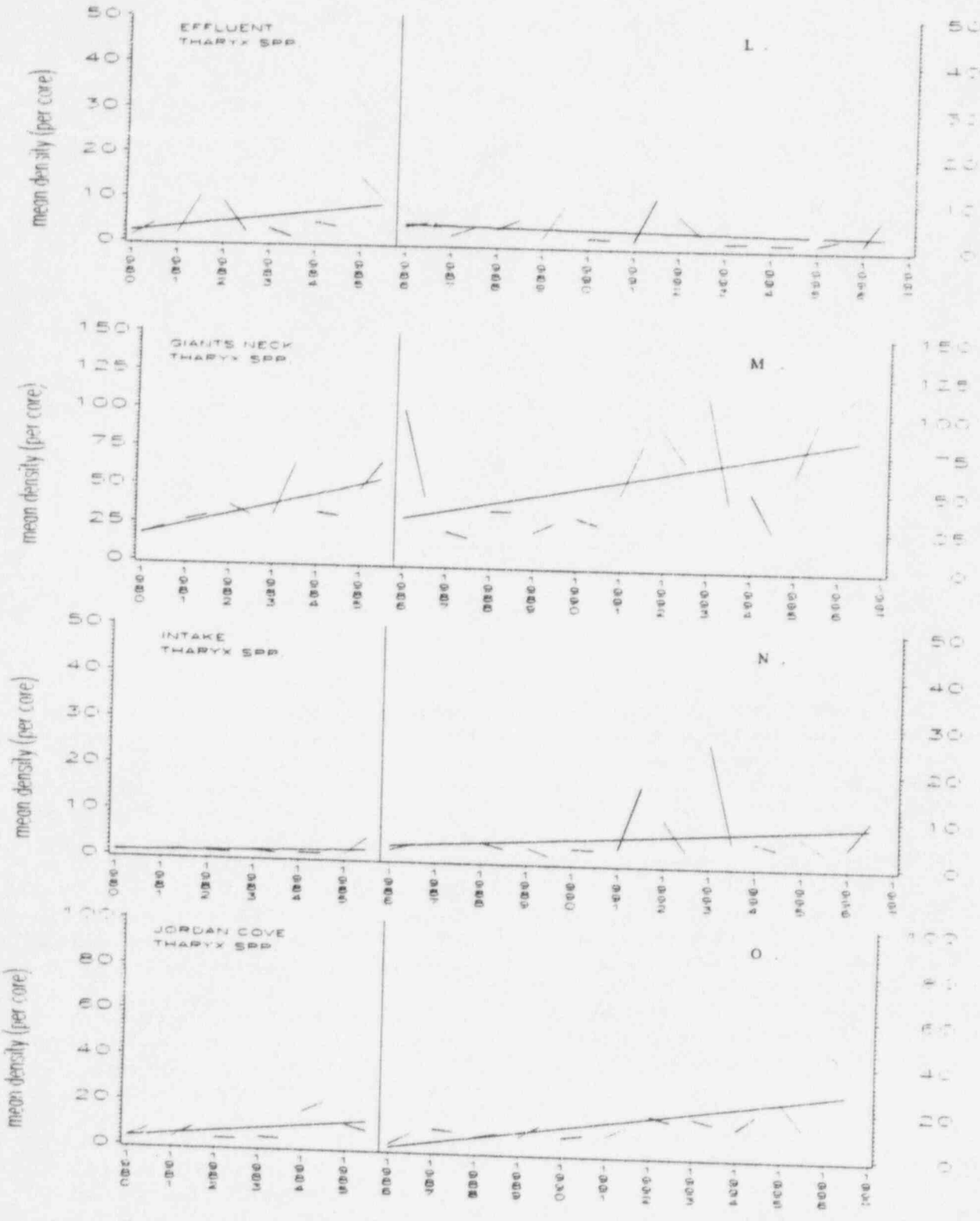
EFFLUENT
THARYX SPP.
L.
mean density (per core)
GIANTS NECK
THARYX SPP.
M.
mean density (per core)
INTAKE
THARYX SPP.
N.
mean density (per core)
JORDAN COVE
THARYX SPP.
O.
mean density (per core)

Fig. 6. (cont.)

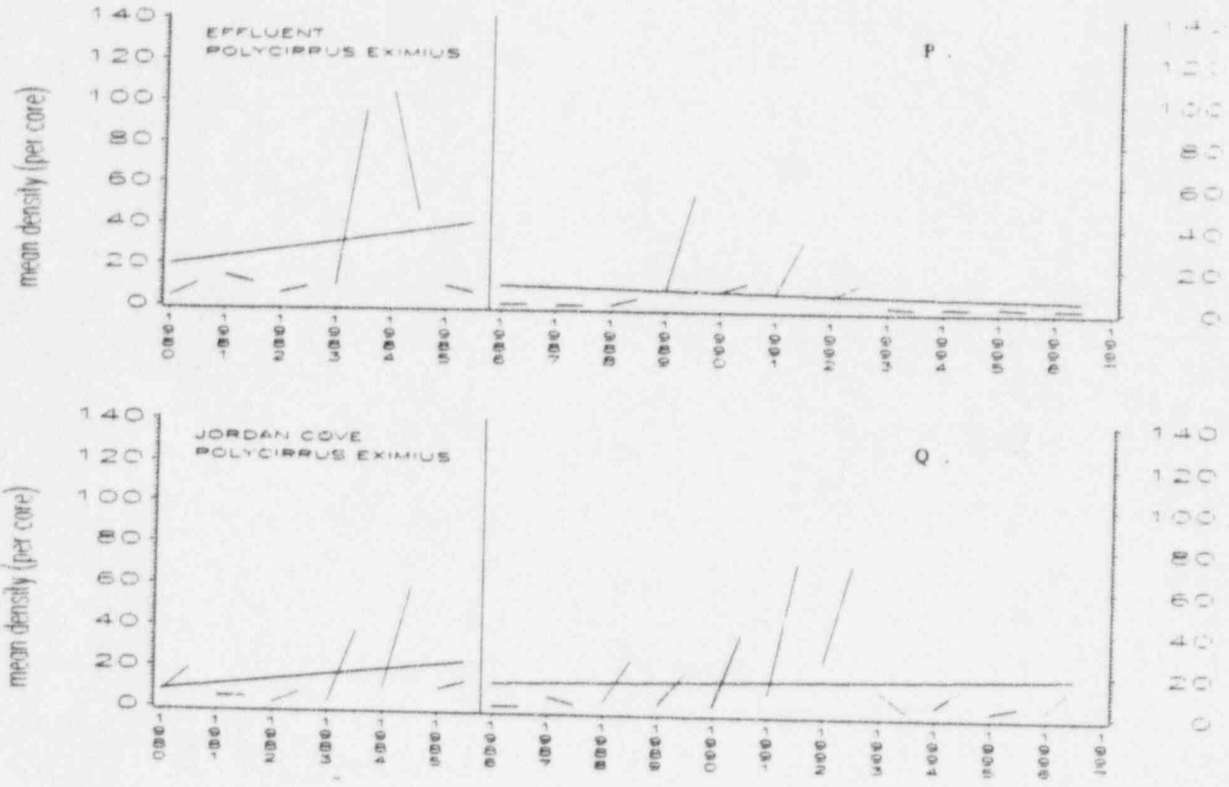

Fig. 6. (cont.)

Scoletema tenuis - *S. tenuis* was among the top ten dominant species in infaunal communities during the 2-unit and 3-unit operating period only at JC (ranking fourth and sixth, respectively, and representing 4.9 and 5.0% of the individuals collected) and GN (ranking fifth [2.5%] and tenth [2.0%], respectively; Table 2). Average densities during 1996 were 1-3/core at GN (ranked eleventh, 0.5% of the individuals) and 10-28/core at JC (ranked sixth, 4.4%). These densities were within the range of density values from previous years. At GN, a significant increasing trend was observed during 2-unit operation, followed by a significant decrease during 3-unit operation. At JC, no trends were detected during either period.

Protodorvillea gaspeensis - *P. gaspeensis* has been among the dominant species at EF during the 2-unit (ranked second, representing 6.1% of the individuals found) and 3-unit (ranked fourth, 4.1% of the individuals) operating periods (Table 2). *P. gaspeensis* was also a dominant at GN, ranking fifth during the 2-unit period and sixth during the 3-unit period, and representing 2.8 and 3.1% of the individuals, respectively. *P. gaspeensis* rankings during 1996 (fourth at EF and sixth at GN) were comparable to these operational period rankings. Average densities in 1996 were within the range of 2-unit and 3-unit studies reported previously, averaging 4-6/core at EF and 3-6/core at GN (Fig. 6t-u). A significant increasing trend in *P. gaspeensis* was noted at GN during the 2-unit operating period, followed by a significant decrease during the 3-unit period; no significant trend was detected at EF during either operational period.

Nucula proxima - This bivalve has been only a minor component of the infaunal communities at IN and JC, but is included here because it exhibited an apparent increase in abundance during the time of MNPS construction activities. For example, an increase occured at IN after the removal of the cofferdam at the Unit 3 intake (1983), and at JC after deposition of sediments scoured from the discharge area after Unit 3 start-up (1986). During 1996, *N. proxima* was not among the top ten dominants at IN or JC; mean density ranges were 1-2/core and 6/core/month, respectively (Fig. 6v-w). Average

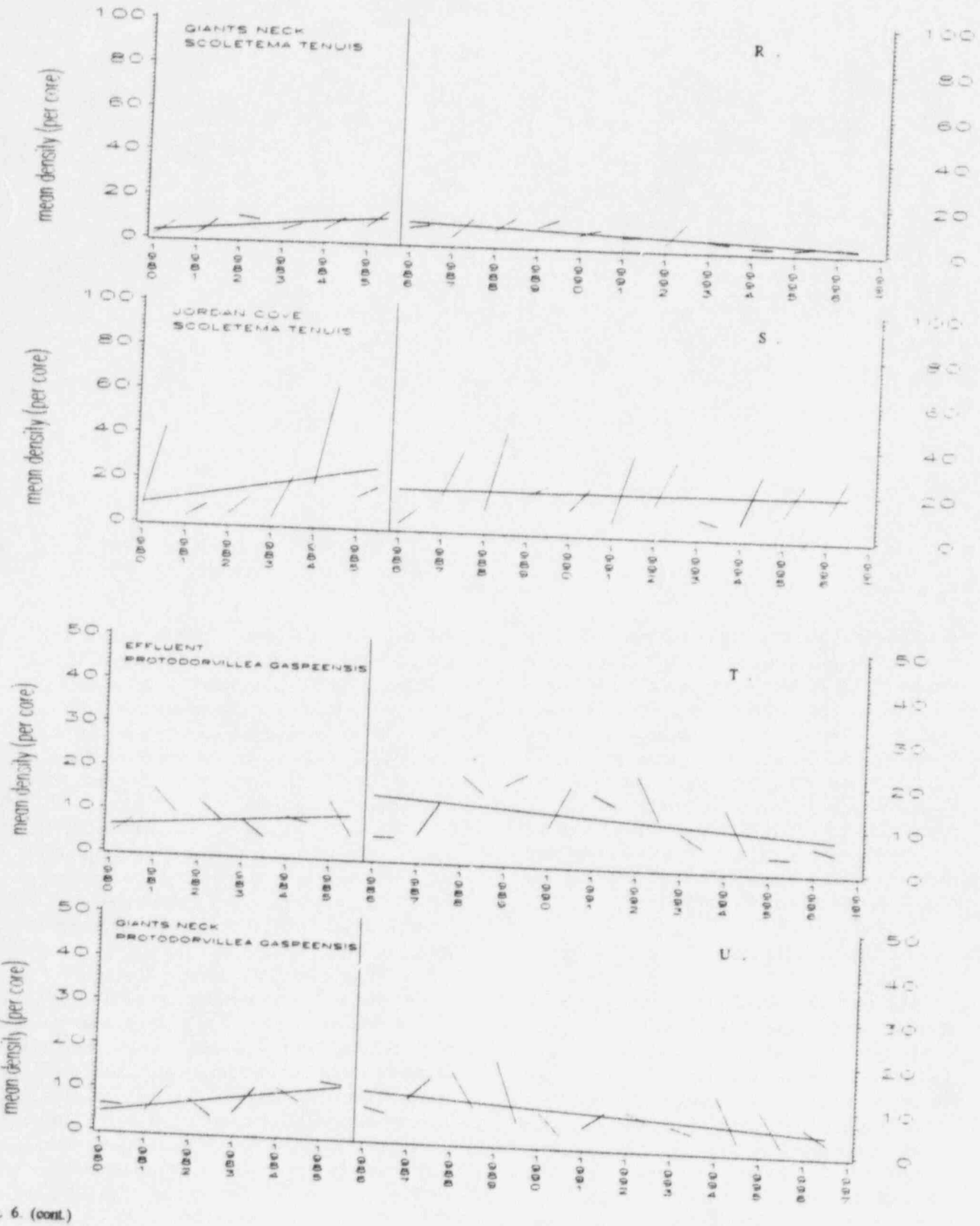
GIANTS NECK
SCOLETEMA TENUIS
R
mean density (per core)
JORDAN COVE
SCOLETEMA TENUIS
S
mean density (per core)
EFFLUENT
PROTODORVILLEA GASPEENSIS
T
mean density (per core)
GIANTS NECK
PROTODORVILLEA GASPEENSIS
U
mean density (per core)

Fig. 6. (cont.)

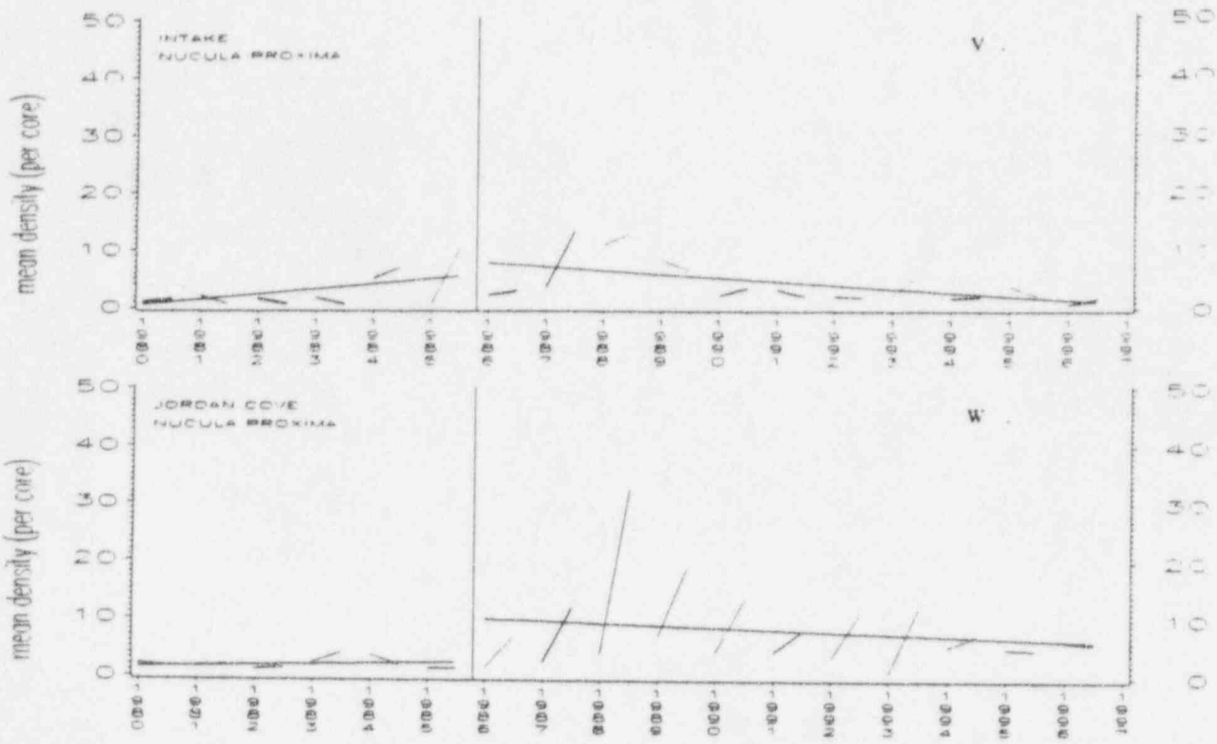

Fig. 6. (cont.)

densities in 1996 were within the range of those recorded in previous years, but lower than those at the beginning of the 3-unit period, indicating a return to levels seen prior to 3-unit construction and operation. Trend analysis revealed a significant increasing trend during 2-unit operation at IN, and a significant decrease in the 3-unit period (Fig. 6v); there were no significant trends evident at JC during either operating period (Fig. 6w). However, recent *N. proxima* abundance at both stations has been more similar to levels noted in early years, prior to cofferdam removal and start-up of Unit 3.

Classification and Cluster Analysis

As illustrated above, annual abundances of several benthic infaunal taxa exhibited temporal trends that have been associated with construction and operation activities at MNPS. Other trends were attributed to area-wide changes in abundance unrelated to the power plant. Each of these species represents one component of a complex community, affected by competition, predation, and other biological and physical structuring factors. To provide a more complete characterization of this community, multivariate Bray-Curtis similarity indices were calculated for each pair of annual collections, using all species whose abundance averaged at least two individuals per year, and illustrated as a clustering dendrogram for each station (Fig. 7a-d). A higher degree of similarity indicates more consistency in the community composition among years.

The overall among-year similarity at EF was approximately 50%, and three groups of annual collections were apparent at a minimum similarity of 55%. Group I (Fig. 7a, Table 3a) consisted of collections from 1980 through 1985, characterized by high densities of *Polycirrus eximius*, *Tharyx* spp. and *A. catherinae*, and relatively low numbers of oligochaetes and *Protodorvillea gaspeensis*. Group II (1986-88) was characterized by lesser abundances of *P. eximius*, *Tharyx* spp. and *A. catheriniae*, and increased abundance of oligochaetes; other taxa that became more common at EF in this period were *Haliplanella luciae* (anemone) and *Mytilus edulis* (blue mussel). The 1986-88 period was a time of maximum sediment scour associated with Unit 3 start-up. All annual collections made at EF since 1989 (Group III), including 1996, exhibited

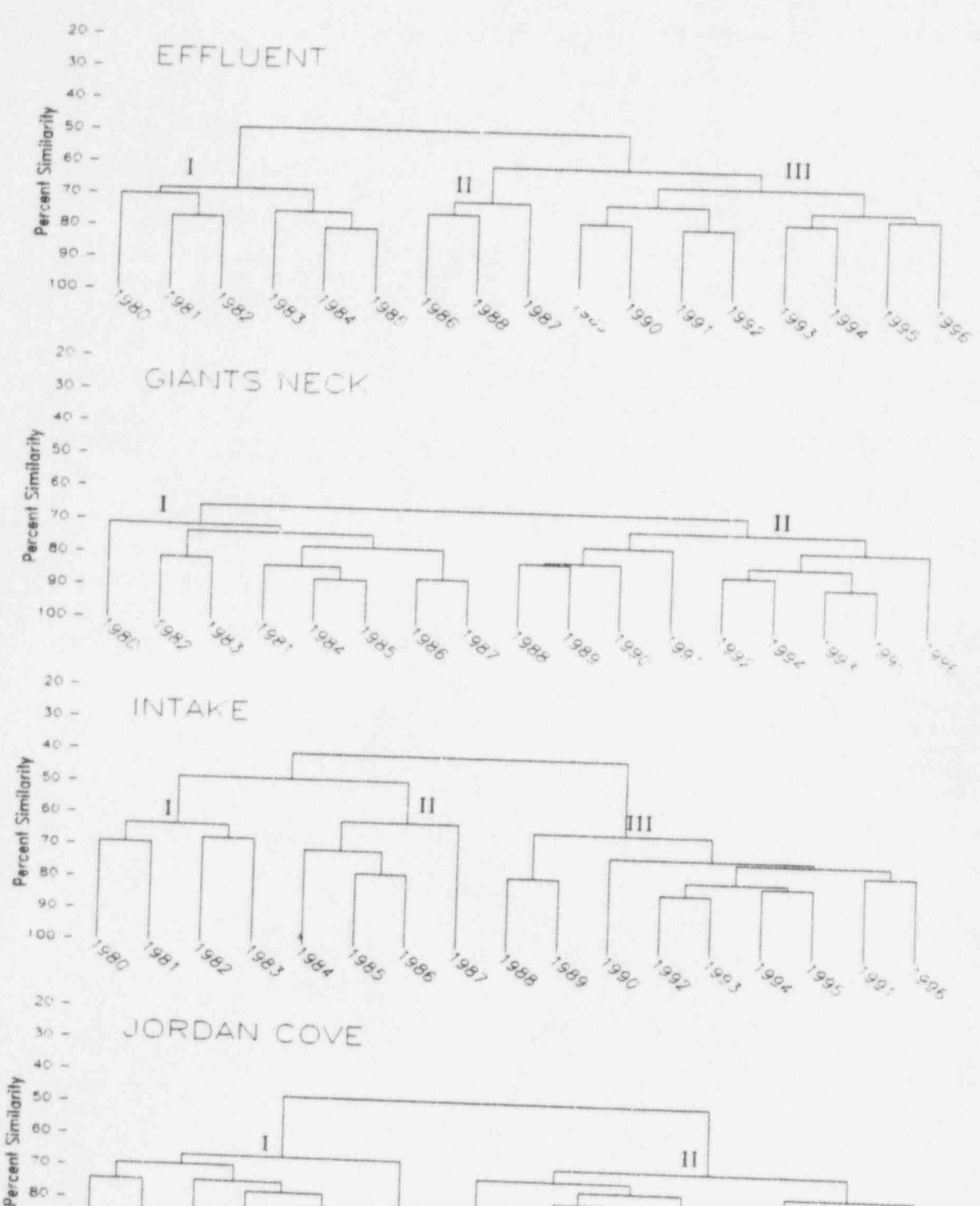

Fig. 7. Clustering dendrograms of natural log-transformed density data for annual infaunal communities at subtidal sand stations near MNPS.

Table 3. List of taxa (mean density/year) associated with major group divisions in classification and cluster analyses.

Effluent (A.)

taxon	Group I	Group II	Group III
Oligochaeta	1809	2392	2018
Polycirrus eximius	574	23	191
Protodorvillea gaspeensis	193	267	249
Tharyx spp.	491	69	74
Tellina agilis	146	72	95
Mediomastus ambiseta	154	146	12
Haliplanella luciae	36	227	32
Rhynchocoela	104	85	94
Prionospio steenstrupi	49	33	127
Aricidea catherinae	128	7	50
Ampelisca verrilli	69	83	11
Capitella spp.	66	66	9
Pagurus acadianus	40	39	62
Cauleriella spp.	42	50	46
Mytilus edulis	13	101	19
Parapionosyllis longicirrata	34	18	74

Giants Neck (B.)

taxon	Group I	Group II
Tharyx spp.	1029	1271
Oligochaeta	1035	1133
Aricidea catherinae	1080	526
Mediomastus ambiseta	582	182
Prionospio steenstrupi	98	613
Protodorvillea gaspeensis	183	152
Scoletema tenuis	140	102
Ampelisca vadorum	142	99
Polycirrus eximius	111	125
Exogone dispar	113	122
Leptocheirus pinguis	206	11
Phoxocephalus holbolli	116	71
Polydora cauleryi	93	36
Cirratulidae	103	0
Capitella spp.	52	43
Polydora quadrilobata	36	52
Microphthalmus aberrans	49	34

Intake (C.)

taxon	Group I	Group II	Group III
Leptocheirus pinguis	13	894	32
Mediomastus ambiseta	13	509	72
Oligochaeta	158	72	361
Prionospio steenstrupi	27	42	466
Aricidea catherinae	157	29	302
Exogone hebes	71	19	186
Ampelisca verrilli	61	170	38
Tharyx spp.	64	52	129
Capitella spp.	73	61	97
Nucula proxima	14	126	83
Ampelisca vadorum	8	145	64
Ampelisca abdita	9	185	2
Tellina agilis	56	48	84
Pygospio elegans	14	16	138
Maldanidae	146	1	16
Owenia fusiformis	4	112	16
Polydora ligni	4	112	15
Unciola serrata	8	90	30

Table 3. Con't.

Jordan Cove (D.)		
taxon	Group I	Group II
Oligochaeta	2365	1423
Mediomastus ambiseta	1707	684
Aricidea catherinae	795	1529
Prionospio steenstrupi	65	746
Scoletema tenuis	349	400
Polycirrus eximius	273	363
Tharyx spp.	255	318
Leptocheirus pinguis	53	515
Nucula proxima	38	169
Tellina agilis	117	84
Capitella spp.	102	98
Polydora caulleryi	140	10
Microphthalmus aberrans	44	82
Maldanidae	2	101
Mitrella lunata	52	41
Rhynchocoela	46	42
Parapionosyllis longicirrata	48	39
Exogone hebes	36	50
Eumida sanguinea	66	19

relatively high within-group similarity, especially since 1993 (>70%); the relatively stable environment in recent years has allowed development of a benthic infaunal community characterized by relatively high densities of *Prionospio steenstrupi* and *Parapionosyllis longicirrata*, and low abundance of *Tharyx* spp., *M. ambiseta*, and *Capitella* spp.

At the GN reference site, all annual collections clustered at >65% similarity (Fig. 7b). At a >70% level, two groups were apparent; Group I (1980-87) was distinguished by high abundance of *A. catherinae*, *Leptocheirus pinguis* and *Mediomastus ambiseta* (Table 3b), and Group II (1986-1995), by high densities of *Prionospio steenstrupi*. Both periods were characterized by relatively high densities of oligochaetes and *Tharyx* spp.. This consistency and the overall congruent community composition in all sample years contributed to the high similarity among groups.

In contrast to EF and GN, annual collections at IN exhibited low among-group similarity (40%). This dissimilarity was influenced by the consistent absence of a dominant suite of species. At a 50% level, three groups were apparent (Fig. 7c); the first group represented annual collections from 1980 to 1983, and was characterized by relatively high abundance of *Aricidea catherinae*, oligochaetes, and the polychaete family Maldanidae (Table 3c). Group II represented 1984-87; the infaunal community in this period was strongly affected by the Unit 3 coffer dam removal and intake dredging, and was characterized by high abundance of the opportunistic polychaete *Mediomastus ambiseta*, the amphipod *Leptocheirus pinguis*, low densities of *Aricidea catherinae* and oligochaetes. The maldanids that had been highly abundant in Group I virtually disappeared during this time period. Also abundant in this period were the gammaridean amphipods *Ampelisca verrilli*, *A. vadorum*, and *A. abdita*, the polychaetes *Owenia fusiformis* and *Polydora ligni*, and the bivalve *Nucula proxima*. Collections from recent years (1988-96; Group III) were characterized by high densities of *A. catherinae* and oligochaetes (as were the earlier Group I years), but several other polychaetes (the syllid *Exogone hebes*, the cirratulid *Tharyx* spp., and the spionids *Prionospio steenstrupi* and *Pygospio elegans*) were relatively abundant as well.

Collections from JC clustered into two groups at a 60% similarity level (Fig. 7d). Group I (1980-1986) was characterized by high abundance of *M. ambiseta* and oligochaetes, and low abundance of *P. steenstrupi*, *L. pinguis*, *N. proxima* and Maldanidae (Table 3d). In contrast, Group II (1987-1996) was distinguished by relatively low densities of *M. ambiseta* and oligochaetes; *A. catherinae* was more abundant in this period than in earlier years, as were *P. steenstrupi*, *L. pinguis*, *N. proxima* and Maldanidae. Several species (*e.g.*, *Scoletema tenuis*, *Polycirrus eximius*, *Tharyx* spp., *Tellina agilis* and

Capitella spp.) were abundant throughout the study period, contributing to the relatively high within-group similarity.

Discussion

Three of the four infaunal communities monitored in the vicinity of MNPS during 1996 continued to reflect the effects of past and ongoing disturbances brought on by power plant construction and operation activities. Community changes related to short-term disturbance events, such as dredging at IN and silt deposition at JC, were still evident, even 10-12 years after the events. Indications of ongoing, but still incomplete, community recovery processes were also noted at IN and JC in recent study years. Rather than recovery, stabilization of the altered community at EF has occurred under chronic disturbance conditions created by the MNPS discharge. Highest community stability over the study period were observed at GN, which is not influenced by MNPS, that reflected only variability in natural environmental conditions.

As the infaunal community is largely affected by its sedimentological environment, stability of sediments at GN was reflected by consistency of the GN population abundances. Sediments collected at GN in 1996 were similar to previous years, and overall community composition at GN has been consistent over the study period; this site was numerically dominated by the same four taxa (oligochaetes, *Tharyx* spp., *Aricidea catherinae* and *Mediomastus ambiseta*) at similar relative abundance during both 2-unit and 3-unit operational periods.

Monitoring data from GN also proved useful in substantiating area-wide shifts in species abundance and community structure. These shifts in populations included large increases in abundance of the amphipods *Leptocheirus pinguis* and *Ampelisca* spp., and of the opportunistic polychaete, *M. ambiseta*, which occurred over several years (1983-1988; NUSCO 1989), as well as increased abundance 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 concluded to be independent of power plant construction or operation. A similar pulse in abundance of *M. ambiseta* occurred in 1994 at GN, JC, and IN, indicating a region-wide phenomenon (NUSCO 1995).

Discrete physical disturbances of relatively short duration (months or a few years) initiated community changes at IN and JC. 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 then, sediments (primarily silt/clay content levels) have stabilized and become more similar to pre-impact years; ongoing community recovery is evident. In particular, species richness at IN has increased in recent years, as have abundances of organisms that were common prior to 1983 (e.g., *Aricidea catherinae*). These trends, along with concomitant decreases in abundance of *N. proxima* and other opportunistic species (NUSCO 1993), all indicate that a recovery process is ongoing at IN. This process is clearly not complete, however, as other taxa which have established post-impact community dominance, such as oligochaetes and *Tharyx* spp., maintained or increased their degree of dominance in 1996 and may persist indefinitely. Long-term recovery following this type of disturbance is typical of marine benthic communities (Kaplan et al. 1974; Swartz et al. 1980; Nichols 1985; Berge 1990).

Silt deposition at JC related to sediment scouring in the MNPS discharge area following Unit 3 start-up resulted in increased sediment silt/clay content and rapid infaunal community change. Abundances of the previously dominant oligochaetes and the polychaetes, *Aricidea catherinae* and *Polycirrus eximius*, all decreased (NUSCO 1988a). This depositional event likely occurred over a short period (i.e., months) at the beginning of the 3-unit operational period, and its impact has evidently lessened since 1986. For example, the *A. catherinae* population rebounded to levels observed during 2-unit years within a few years of Unit 3 start-up, and has steadily increased to record high abundance in 1996. However, other signs of recovery noted in previous years (e.g., reduced silt/clay content during 1994, and rebounding abundances of oligochaetes and *P. eximius* through 1993; NUSCO 1994, 1995) were less evident in recent years. Lower abundances of these two taxa in recent years were probably due to the long-term persistence of some of the deposited silt/clay at JC still evident after 1996 sampling. Another indication of slow community recovery is persistence of the opportunistic mollusc, *Nucula*

proxima, which became a dominant taxon after Unit 3 start-up. 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; Maurer et al. 1986; Emerson 1989; Brey 1991). Further monitoring is necessary to document the long-term effects of this depositional event on future infaunal community development and recovery at JC, particularly in light of the anticipated restart of Unit 3 in the summer of 1997.

Sediment scour from the MNPS discharge directly impacted both the sediments and the infaunal community at EF. Relatively coarse sediment with low silt/clay levels are now characteristic of 3-unit benthic habitat at EF. While sediment characteristics remain distinctly different from those observed prior to 1986, the altered sedimentary environment at EF, and the infaunal community it supports, has stabilized under the new environmental conditions created by the 3-unit discharge. Oligochaete and *Protodorvillea gaspeensis* abundances in recent years, including 1996, have generally decreased to levels observed during the 2-unit period. The relative stabilization of the sediments at EF has also allowed for the return of *Aricidea catherinae*, a species common during 2-unit operation. It appears that effects of the MNPS discharge are a dominant structuring mechanism for the infaunal community at EF. An example of this is the absence of the opportunistic polychaete *Mediomastus ambiseta* and the decline of *Tharyx* spp at EF. *M. ambiseta* essentially exhibited an area-wide increase in 1994-95, related to regional natural environmental cues that were apparently overridden by plant discharge effects at EF. The previous increase of *M. ambiseta* was initiated in 1984 (prior to scouring related to the 3-unit discharge), and was apparent at all stations, including EF. Similarly, *Tharyx* spp. has generally exhibited an area-wide increase at all stations except EF, where reduced abundances observed after Unit 3 start-up have continued through 1996. As benthic infaunal species are closely linked to the sedimentological characteristics of their surroundings, recovery of the EF community to a pre-impact condition is not expected to begin until power plant-induced scour ceases.

Conclusions

Benthic habitats, and their associated infaunal communities, at IN, JC and EF continued to reflect impacts related to construction and operation of MNPS during 1996. The sedimentary environment and infaunal community structure at the unimpacted control site, GN, exhibited the most stability over the study period, relative to the other sites. Some indications of increased stability and community recovery have been observed in recent years at stations impacted by short-term episodic disturbance events (e.g., dredging and construction activities at IN, and siltation at JC). This recovery is ongoing; community parameters of species composition and abundance show similarities to the pre-disturbance period, but clearly show differences and indications of disturbance at both stations. Continuous effluent scour at EF, first noted after Unit 3 start-up, remains a dominant structuring factor on both the sedimentary environment and infaunal community in the immediate vicinity of the discharge, and is expected to maintain this role throughout MNPS 3-unit operation.

References Cited

Aller, R.C. 1978. Experimental studies of changes produced by deposit-feeders on pore water, sediment, and overlying water chemistry. Am. J. Sci. 278:1185-1234.

Berge, J.A. 1990. Macrofaunal recolonization of subtidal sediments. Experimental studies on defaunated sediment contaminated with crude oil in two Norwegian fjords with unequal eutrophication status. I. community responses. Mar. Ecol. Prog. Ser. 66:103-115.

Boesch, D.F., R.J. Diaz, and R.W. Virnstein. 1976. Effects of tropical storm Agnes on soft-bottom macrobenthic communities of the James and York estuaries and the lower Chesapeake Bay. Chesapeake Sci. 17:246-259.

Boesch, D.F., and R. Rosenburg. 1982. Response to stress in marine benthic communities. Pages 179-200 *in* G.W. Barrett and R. Rosenburg, eds. Stress Effects on Natural Ecosystems. John Wiley, New York.

Boero, F. 1994. Fluctuations and variations in coastal marine environments. Mar. Ecology 1:3-25.

Brey, T. 1991. The relative significance of biological and physical disturbance: an example from intertidal and subtidal sandy bottom communities. Estuar. Coast. Shelf Sci. 33:339-360.

Commito, J.A., and M. Boncavage. 1989. Suspension-feeders and coexisting infauna: an enhancement counterexample. J. Exp. Mar. Biol. Ecol. 125:33-42.

Commito, J.A., C.A. Currier, L.A. Kane, K.A. Reinsel, and I.M. Ulm. 1995. Dispersal dynamics of the bivalve *Gemma gemma* in a patchy environment. Ecol. Monogr. 65:1-20.

Diaz, R.J., and L.S. Schaffner. 1990. The functional role of estuarine benthos. Pages 25-56 in Contrib. 1595. College of William and Mary, Virginia Inst. of Mar. Sci.

Draper, N., and H. Smith. 1981. Applied regression analysis. John Wiley and Sons, New York. 709 pp.

Emerson, C.W. 1989. Wind stress limitation of benthic secondary production in shallow, soft-sediment communities. Mar. Ecol. Prog. Ser. 55:65-77.

Flint, R.W. 1985. Long-term estuarine variability and associated biological response. Estuaries 8:159-169.

Folk, D. 1974. Petrology of Sedimentary Rocks. Hempshill Publishing Company, Austin, Texas. 182 pp.

Franz, D.R. and J.T. Tanacredi. 1992. Secondary production of the amphipod *Ampelisca abdita* Mills and its importance in the diet of juvenile winter flounder (*Pleuronectes americanus*) in Jamaica Bay, New York. Estuaries 15:193-203.

Gaston, G.R., and J.C. Nasci. 1988. Trophic structure of macrobenthic communities in the Calcasieu Estuary, Louisiana. Estuaries 11:201-211.

Gilbert, R.O. 1989. Statistical methods for environmental pollution monitoring. Van Nostrand Reinhold Company, New York. 320 pp.

Goldhaber, M.B., R.C. Aller, J.K. Cochran, J.K. Rosenfield, C.S. Martens, and R.A. Berner. 1977. Sulfate reduction, diffusion bioturbation Long Island Sound sediments: Report of the FOAM Group. Am. J. Sci. 277:193-237.

Holland, A.F. 1985. Long-term variation of macrobenthos in a mesohaline region of Chesapeake Bay. Estuaries 8:93-113.

Holland, A.F., A.T. Shaughnessy, and M.H. Hiegel. 1987. Long-term variation in mesohaline Chesapeake Bay macrobenthos: Spatial and temporal patterns. Estuaries 10:227-245.

Hollander, M., and D.A. Wolfe. 1973. Non-parametric statistical methods. John Wiley and Sons, New York. 503 pp.

Horn, M.H., and R.N. Gibson. 1988. Intertidal fishes. Sci. Am. 256:64-70.

Jordan, R.A., and C.E. Sutton. 1985. Oligohaline benthic invertebrate communities at two Chesapeake Bay power plants. Estuaries 7:192-212.

Jumars, P.A., and K. Fauchald. 1977. Between-community contrasts in successful polychaete feeding strategies. Pages 1-20 *in* B.C. Coull, ed. Ecology of Marine Benthos. Univ. of South Carolina Press, Columbia, S.C. 467 pp.

Kaplan, E.H., J.R. Welker, and M.G. Kraus. 1974. Some effects of dredging on populations of macrobenthic organisms. Fish. Bull. 72:445-480.

Kneib, R.T. 1988. Testing for indirect effects of predation in an intertidal soft-bottom community. Ecology 69:1795-1805.

Lance, G.N., and W.R. Williams. 1967. A general theory of classificatory sorting strategies, I. Hierarchical systems. Comput J. 9:373-380.

Levinton, J.S., and S. Stewart. 1982. Marine succession: The effect of two deposit-feeding gastropod species on the population growth of *Paranais littoralis* Muller 1784 (Oligochaeta). J. Exp. Mar. Biol. Ecol. 59:231-241.

Maurer, D., R.T. Keck, J.C. Tinsman, W.A. Leathem, C. Wethe, C. Lord and T.M. Church. 1986. Vertical migration and mortality in marine benthos in dredged material: a synthesis. Int. Rev. ges. Hydrobiol. 71:49-63.

Moeller, P., L. Pihl, and R. Rosenberg. 1985. Benthic faunal energy flow and biological interaction in some shallow marine soft bottom habitats. Mar. Ecol. Prog. Ser. 27:109-121.

NAESCO (North Atlantic Energy Service Corporation). 1994. Seabrook environmental studies, 1993. A characterization of environmental conditions in the Hampton-Seabrook area during the operation of Seabrook Station.

Nichols, F.H. 1985. Abundance fluctuations among benthic invertebrates in two Pacific estuaries. Estuaries 8:136-144.

NUSCO. (Northeast Utilities Service Company). 1987. Benthic Infauna. Pages 1-59 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of Studies Prior to Unit 3 Operation.

NUSCO. 1988a. Benthic Infauna. Pages 59-117 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.

NUSCO. 1988b. Hydrothermal Studies. Pages 323-354 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.

NUSCO. 1989. Benthic Infauna. Pages 38-98 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1988.

NUSCO. 1992. Benthic Infauna. Pages 187-222 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1991.

NUSCO. 1993. Benthic Infauna. Pages 115-150 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1992.

NUSCO. 1994. Benthic Infauna. Pages 81-102 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1993.

NUSCO. 1995. Benthic Infauna. Pages 203-227 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1994.

Prena, J. 1995. Temporal irregularities in the macrobenthic community and deep-water advection in Wismar Bay (Western Baltic Sea). Estuarine, Coastal and Shelf Sci. 41:705-717.

Rees, H.L. and A. Eleftheriou. 1989. North Sea benthos: A review of field investigations into the biological effects of man's activities. J. Cons. int. Explor. Mer. 45:284-305.

Regnault, M., R. Boucher-Rodoni, G. Boucher, and P. Lasserre. 1988. Effects of macrofauna excretion and turbulence on inorganic nitrogenous exchanges at the water-sediment interface. Cah. Biol. Mar. 29:427-444.

Rhoads, D.C., and D.K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. J. Mar. Res. 28:150-178.

Richards, S.W. 1963. The demersal fish population of Long Island Sound. Bull. Bingham Oceangr. Coll. 8:1-101.

Sen, P.K. 1968. Estimates of regression coefficients based on the Kendall's tau. J. Am. Stat. Assoc. 63:1379-1389.

Somerfield, P.J., H.L. Rees, and R.M. Warwick. 1995. Interrelationships in community structure between shallow-water marine meiofauna and macrofauna in relation to dredging disposal. Mar. Ecol. Prog. Ser. 127:103-112.

Swartz, R.C., W.A. DeBen, F.A. Cole, and L.C. Bentsen. 1980. Recovery of the macrobenthos at a dredge site in Yaquina Bay, Oregon. Pages 391-408 *in* R.A. Baker, ed. Contaminants and Sediments, Vol. 2. Ann Arbor Science Publisher, Inc., Ann Arbor, Mich.

Thrush, S.F., R.D. Pridmore, and J.E. Hewitt. 1994. Impacts on soft-sediment macrofauna: the effects of spatial variation on temporal trends. Ecol. Appl. 4:31-41.

Turk, T.R., and M.J. Risk. 1981. Effects of sedimentation on infaunal invertebrate populations of Cobequid Bay, Bay of Fundy. Can. J. Fish. Aquat. Sci. 38:642-648.

Warwick, R.M. 1986. A new method for detecting pollution effects on marine macrobenthic communities. Mar. Biol. 92:557-562.

Warwick, R.M. 1988. Effects on community structure of a pollutant gradient-introduction. Mar. Ecol. Prog. Ser. 46:149.

Warwick, R.M., T.H. Pearson and Ruswahyuni. 1987. Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance biomass method. Mar. Biol. 95:193-200.

Warwick, R.M., H.M. Platt, K.R. Clark, J. Agard and J. Gobin. 1990. Analysis of macrobenthic and meiobenthic community structure in relation to pollution and disturbance in Hamilton Harbour, Bermuda. J. Exp. Mar. Biol. Ecol. 138:119-142.

Watling, L. 1975. Analysis of structural variations in a shallow estuarine deposit-feeding community. J. Exp. Mar. Biol. Ecol. 19:275-313.

Watzin, M. C. 1986. Larval settlement into marine soft-sediment systems: Interactions with the meiofauna. J. Exp. Mar. Biol. Ecol. 98:65-113.

Woodin, S.A. 1982. Browsing: important in marine sedimentary environments? Spionid polychaete examples. J. Exp. Mar. Biol. Ecol. 60:35-45.

Young, M.W., and D.K. Young. 1982. Marine macrobenthos as indicators of environmental stress. Pages 527-539 *in* G.F. Mayer, ed. Ecological Stress and the New York Bight: Science and Management. Proceedings of the symposium; 1979 June 10-15; New York, New York. Estuarine Research Federation, Columbia, S.C. 715 pp.