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

Millstone Environmental Laboratory

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Dominion

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

2002 Annual Report

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Executive Summary

Winter Flounder Studies

The local Niantic River population of winter flounder (*Pseudopleuronectes americanus*) is potentially affected by the operation of Millstone Power Station (MPS), particularly by entrainment of larvae through the cooling-water systems of the operating units. As a result, extensive studies of the life history and population dynamics of this important sport and commercial species have been undertaken since 1976. Each year, surveys of adult spawners are carried out in the Niantic River during late winter and early spring. Larval sampling is conducted at the plant discharges, in Niantic Bay, and at three stations in the Niantic River. Age-0 juveniles are collected at two sites in the river. Winter flounder are also commonly found in year-round trawl monitoring program (TMP) samples.

Seawater temperature, which can have potentially important effects during spawning, egg incubation, the larval period, and the remainder of the first year of life, attained record or near-record highs during most months of 2002. During the 2002 adult winter flounder spawning season, Δ-mean trawl catch-per-unit-effort (CPUE) of fish larger than 15 cm in the Niantic River was 2.2. Although relatively low, this CPUE represented about a 50% increase over 2001. Abundance of winter flounder spawners peaked in the early 1980s and decreased thereafter, most likely from stock and recruitment effects (i.e., decreased per-capita recruitment at high stock sizes), a generally warming winter trend, and increased rates of exploitation. Niantic River CPUE was significantly correlated with several other winter flounder abundance indices in other Southern New England waters and numbers currently remain low throughout the region.

The Jolly stochastic model was applied to mark and recapture data to estimate the absolute abundance of the Niantic River adult spawning population. The abundance estimate for 2001 of winter flounder larger than 20 cm was 4.1 thousand, the lowest estimate of the series. Other recent estimates of about 6-10 thousand fish were considerably less than estimated population sizes during 1984-91, which ranged between about 33 and 80 thousand. Annual female spawner abundance estimates since 1976 ranged from 2 thousand in 2001 to 75 thousand in 1982, with corresponding total egg production estimates from about 1.7 to 44.8 billion for the same years.

Abundance of newly-hatched (Stage 1) winter flounder larvae was about average this year in comparison to the long-term (1983-2002) time-series in both Niantic River and Bay. However, more Stage 1 larvae were found than expected from the relatively low spawner abundance. This was attributed to higher egg survival, a potentially important compensatory mechanism. Overall, larval winter flounder abundance in 2002 was below average in the Niantic River and was the third lowest in 20 years in Niantic Bay.

Rates of larval growth and development were positively correlated with water temperature, but other factors such as density and prey abundance likely also affected growth. Despite relatively high water temperatures, growth rate of larvae this year in both the river and bay were the lowest of their respective time-series. This may have affected survival as the larval mortality rate in 2002 increased from record low levels during the past 2 years to just below the long-term average. Density-dependence was examined by comparing mortality and egg production estimates (a measure of yolk-sac larval stage abundance) at various monthly and seasonal water temperatures. Results of this analysis suggested that larval mortality decreased with decreasing egg production (i.e., density-dependency) and increasing April water temperatures (i.e., faster development).

Following larval metamorphosis and settlement, demersal age-0 young were sampled by beam trawl at two sites in the Niantic River. Following a record high abundance of newly settled young in 2001, densities were lower this year, although better than those typically observed since 1983. Mortality rate in 2002 was about the same as the long-term average at one station, but was the third highest at the other station. Although sampling variability was relatively high in 2002, juvenile abundance in late summer appeared to be lower than during the past 2 years.

The 2001-02 Δ-mean CPUE calculated for young winter flounder collected during late fall and early winter at TMP stations was the lowest recorded in 26 years and did not reflect the high abundance of the 2001 year-class as indicated by beam trawl sampling during the previous summer. Nevertheless, over the long term these two age-0 abundance indices were significantly correlated and helped to identify emerging strong or weak year-classes. In any case, however, fewer age-1 juveniles have been taken during each year of the Niantic River adult spawning population surveys since the early 1980s, with a CPUE for fish in the lower river navigational channel particularly low. This abundance decrease was much less when a CPUE was calculated for fish found only in the upper portion of the river. The distribution of

age-1 fish in Niantic River and Bay appeared to have changed over the years, which was attributed to environmental factors (e.g., water temperature, bottom habitat) unrelated to MPS operation.

Abundance indices of age-0 young were either not significantly correlated or were negatively correlated with the abundance of female adult spawners 3 to 5 years later. Conversely, positive correlations were found between age-1 abundance indices and these older fish. However, the forms of these significant relationships were unclear. Thus, none of the early life stages were considered to be a reliable predictor of potential future year-class strength. Unknown natural processes (e.g., predation) that occur after winter flounder become age-1 appear to be operating so as to produce fewer adult recruits from more abundant year-classes of juveniles.

A stock and recruitment relationship (SRR) was determined for the Niantic River winter flounder population using adult CPUE data. A February water temperature parameter significantly improved the model fit. A depensatory SRR (per capita recruitment rate decreasing with decreasing parental stock size) was examined, but the model was not significant.

The number of larvae entrained through the condenser cooling-water system at MPS is a measure of potential impact to winter flounder. Annual estimates of entrainment were related to both larval densities in Niantic Bay and plant operation. The 2002 entrainment estimate of about 119 million was considerably less than the totals of more than 300 million found in 2000 and 2001. This total reflected the relatively low larval densities found in Niantic Bay this year and less than full cooling-water demand by MPS. As in previous years, Stage 3 larvae predominated (82%) in entrainment collections. The entrainment rate (annual abundance index divided by total flow) has varied since 1976 without trend, indicating that larval production and availability in Niantic Bay remained stable despite increased water use during the 1986-95 period of three-unit operation. Correlations between entrainment estimates and abundance indices of age-0 juveniles were positive, implying no entrainment effect and that the more larvae available for entrainment, the more that metamorphosed and settled in Niantic River and Bay. This was also illustrated by a comparison of annual entrainment and juvenile year-class strength, which indicated that entrainment estimates served as an index of emerging year-class strength rather than being the most important factor in setting abundance.

The impact of larval entrainment on the Niantic River stock depends upon the fraction of the annual winter flounder production entrained each year, which is determined in this study as equivalent eggs.

Empirical mass-balance calculations showed that a large number of entrained larvae likely come from a number of sources in Long Island Sound (LIS), including stocks associated with the Connecticut and Thames Rivers, as well as from the Niantic River. For the second year, estimates of the fraction of entrained larvae in 2002 from the Niantic River were similar when determined by either the mass-balance model (14%) or an independently conducted study of genetic stock identification (12%). Remarkably similar estimates in both 2001 and 2002 suggested that the mass-balance model provided reasonable estimates of Niantic River larval stock entrainment. The Niantic River production loss estimate of 19.7% in 2002 was above the long-term average of 13.9%. However, based on the apparent increase in egg survival noted in recent years, a factor which was not incorporated into the model, production loss estimates since 1995 may have been conservatively high.

A stochastic computer simulation model (SPDM) was used for long-term assessments of MPS impact. Annual female spawner biomass (lbs) was determined in simulations over a period extending from 1960, a decade before the operation of Unit 1, to 2060, 15 years after the projected shutdown of Unit 3 in 2045, which assumes that a 20-year operating license renewal will be attained by MPS. Conditional larval entrainment mortality rates (termed ENT) from the mass-balance model were input to the model. The long-term geometric mean value of ENT was used in SPDM forward projections, and, to account for model uncertainties, both high and low rates were calculated by scaling the mean value by a factor of 1.5 and 1/1.5, respectively. Values of fishing mortality (F) provided by the Connecticut Department of Environmental Protection in 2001 were used with the model, with the most recent mean rate of 0.74 applied going forward. In the SPDM simulations, five stochastic time-series were generated: a theoretical unfished stock, the size of which depended only upon the dynamics of winter flounder reproduction, natural rate of increase, environmental variability, and natural mortality; a baseline stock affected by both fishing and natural mortality and other factors relating to natural population dynamics, but without any effects of MPS; and three impacted stocks, which further added effects of MPS (primarily larval entrainment with postulated low, mean, and high ENT rates going forward) to those of fishing and natural variation. Used initially in all simulations, an initial unfished stock size of 121,348 lbs represented the maximum spawning potential for the Niantic River stock and the critical stock size (25% of the calculated Maximum Spawning Potential) was determined as 30,337 lbs.

For the baseline population projection, the exploited biomass quickly became about 30 thousand lbs by 1982 and declined further under relatively high F to a low of 5.7 thousand lbs in 1994, which was only 19% of the critical stock size. With some reduction in F beginning in 1999, the stock recovered somewhat, but remained between 11 and 12 thousand lbs through 2060. Under the mean ENT rate, the impacted stock also attained its lowest biomass of 4.3 thousand lbs in the mid-1990s. Absolute differences between the baseline and impacted stocks increased to about 5.4 thousand lbs in 2017. However, following the projected retirements of Units 2 and 3, by 2060 impacted stock sizes began to approach those of the modeled baseline, which was affected by fishing and other natural factors only without any power plant effects. The simulations under the high and low ENT rates generally paralleled those of the mean rate, only differing in magnitude. Of note, the F rate used in the SPDM is critical and actual spawner abundance could depart greatly in the simulations if this or other simulated conditions were not matched in reality. Fishing reduces stock biomass at a much greater rate than larval entrainment by removing individuals, particularly larger ones, from the same year-class every year for as long as any fish remain. In contrast, entrainment removes fish only once in the lifetime of a year-class, early in life before many compensatory processes have occurred.

To provide some perspective on Niantic River winter flounder stock size estimates, annual exploitation rates were determined and used with annual recreational and commercial landings data of winter flounder for LIS. The calculated exploitable biomass of Niantic River winter flounder likely represented only 2% or less of the entire winter flounder resource in LIS during the past two decades.

To date, efforts of regulatory agencies to control fishing mortality have not resulted in large increases in abundance for winter flounder stocks across the region, including the Niantic River. Even so, the remaining small adult spawning stock in the river continues to produce relatively large numbers of larvae and young fish, which are a likely result of population compensatory mechanisms. The effective retirement of Unit 1 in late 1995 was followed by an immediate reduction of about one-quarter of the MPS cooling-water flow, which has permanently lessened plant impact, but has not resulted in stronger year-classes and subsequent enhanced recruitment to the spawning stock. Despite relatively good abundance of post-entrainment immature winter flounder, significant recruitment to the adult spawning population has not occurred in recent years, due to as yet unknown factors removing these fish from the popu-

lation. Environmental effects, including changes to Niantic River habitat conditions and interactions with other species, especially during early life history, also are important factors likely affecting winter flounder recruitment processes.

Fish Ecology Studies

The objective of the fish ecology monitoring program at MPS is to determine whether operation of the electrical generating units has adversely affected the occurrence, distribution, and abundance of local fishes. Potential MPS impacts include entrainment of fish eggs and larvae through the condenser cooling-water system, impingement of juvenile and adult fish on intake screens (which has been mitigated by the installation of fish return sluiceways), and changes in distribution or abundance attributable to the thermal discharge. Trawl, seine, and ichthyoplankton monitoring programs were established in 1976 to provide the basis for identifying taxa potentially affected, as well as information on long-term abundance trends used to measure changes in the local populations.

This report summarizes data collected in the monitoring programs from June 2001 through May 2002 (report year 2001-02). During the 2001-02 report period, MPS Units 2 and 3 were both operating most of the time and Unit 1 was shut down.

The potential effects of MPS were assessed by conducting detailed analyses on seven taxa most susceptible to MPS operational impact from entrainment or thermal effects. Analyses of these species generally focused on comparing temporal trends over the past 26 years. No significant long-term trends were detected for juvenile or adult silversides (*Menidia menidia* and *M. beryllina*) collected by seine in Jordan Cove (JC), while adult abundance showed a significant decreasing trend at the Intake (IN) as revealed by catches in trawls. No significant long-term trends were detected for all life stages of grubby (*Myoxocephalus aeneus*), cunner (*Tautogolabrus adspersus*) eggs and larvae, and tautog (*Tautoga onitis*) and sand lance (*Ammodytes americanus*) larvae. Atlantic menhaden (*Brevoortia tyrannus*) larvae showed a significant increasing trend in abundance during the past 26-years. A significant negative trend was observed for silversides at the IN trawl station. Since the mid-1980s, cunner have become less abundant at IN, exhibiting a significant negative trend, which was probably related to removal of the Unit 3 rock cofferdam, a preferred habitat for this species. Despite the negative trend in abundance for cunner at trawl station IN during 2001-02, tautog abundance

was at a historic high at station IN. Tautog eggs exhibited a negative trend in abundance, yet the abundance of larvae was the highest observed. The large numbers of tautog and cunner eggs entrained at MPS did not appear to affect future recruitment and subsequent spawning stock biomass of these two fishes because the proportion of juvenile recruits relative to adults has increased and there has been an increasing trend in numbers of tautog captured in lobster pots at JC. Egg survival experiments revealed that approximately 20% of eggs collected at the power plant discharge were viable and overall survival of these was 11%, about half that observed for eggs collected at the intake. Dominion-sponsored research conducted by University of Connecticut scientists showed that annual fecundity of tautog is nearly double previously reported values.

This year the Δ-mean density for bay anchovy (*Anchoa mitchilli*) eggs increased substantially over the 1999-2000 estimates, which were the lowest recorded. However, the Δ-mean density of larvae was lower than average and both eggs and larvae exhibited a significant declining trend. The bay anchovy appears to be experiencing a regional decline in abundance. A sharp drop in abundance was measured over the past decade in Narragansett Bay and populations declined dramatically in Chesapeake Bay after 1993. The bay anchovy is an important forage species for striped bass and recent increases in striped bass abundance along the Atlantic Coast may have contributed to the reduced numbers. None of the long-term abundance trends determined for various life stages of selected species could be directly related to the operation of MPS.

Lobster Studies

The American lobster (*Homarus americanus*) occurs in the Northwest Atlantic from Canada to Cape Hatteras and supports one of the most valuable commercial fisheries in New England. In Connecticut, the economic value of the lobster fishery has been between $12 and $15 million annually. Lobsters are also one of the largest mobile benthic invertebrates in the western North Atlantic and are important inhabitants of coastal ecosystems. Several aspects of MPS operation could potentially impact the local population of American lobster. In early summer, after hatching from eggs, lobster larvae swim to the surface to begin the 6- to 8-week planktonic phase of their life cycle and are susceptible to entrainment through cooling water systems. Juvenile and adult lobsters can be impinged on intake traveling screens or be exposed to the heated effluent in the discharge area. Because of

the economic and ecological importance of this species, lobsters have been monitored from May through October since 1978, using wire lobster traps set at three stations around MPS. Since 1984, entrainment studies have been conducted during the hatching season, to estimate the number of lobster larvae drawn through the cooling water system. The objective of the lobster monitoring program is to determine if operation of MPS has caused changes in local lobster abundance and population characteristics beyond those expected from changes in the fishery and natural variability.

Fishing pressure on the species is intense throughout its range. The number of traps used in the U.S. fishery has increased eightfold since the early 1980s. Coincidental with the increase in fishing effort, landings made in the commercial fishery throughout New England have increased markedly over the last three decades. Commercial landings of lobsters in Connecticut waters of LIS increased from 0.8 million pounds in 1979 to a record 3.7 million pounds in 1998. However, landings declined 65% between 1998 and 2001 to only 1.3 million pounds. Landings declined even further by 2002 when only 1.1 million pounds were harvested.

The recent decline in lobster abundance has been attributed to a significant lobster die-off in western LIS. Beginning in the fall of 1998 and throughout the summer of 1999, an unexplained highly virulent illness emerged in lobsters harvested from western LIS. Lobstermen reported that more than half of the lobsters caught in traps were dead or died before arriving at port. Concurrent with the lobster die-off in western LIS, a widespread outbreak of shell disease syndrome began in the fall of 1998 in our area of eastern LIS and other coastal areas of southern New England. Naturally occurring bacteria and fungi have been implicated as causative agents of shell disease. These microorganisms consume the exoskeletons of crustaceans, resulting in a shell that is pitted and marred with necrotic lesions. The unsightly appearance of the lobster shell can greatly affect marketability and although shell disease is not immediately fatal, death may ultimately occur. In summer 2002, lobster fishers in central and western LIS described new reports of a significant number of moribund and dead lobsters. Affected lobsters were characterized by an orange discoloration of the abdomen, lethargy, and poor post-capture survival.

In response to the die-off in western LIS and shell disease outbreak in eastern LIS, the U.S. Secretary of Commerce declared a failure of the LIS commercial lobster fishery in January 2000. In 2001, the U.S. Congress allocated $13.9 million in federal disaster funds for the LIS lobster industry. Half of this Federal funding was earmarked for research investigations on

causes and economic impacts of the die-off and shell disease in LIS lobsters. At present, researchers suspect that the 1998-1999 die-offs in western LIS were caused by multiple disease pathogens exacerbated by environmental stressors, such as increased seawater temperature, anoxia and contaminants. Preliminary results link the latest mortalities during the summer 2002 to an accumulation of calcium carbonate in lobster tissue (calcinosis) mediated by prevailing high bottom water temperature in LIS.

The catastrophic die-off observed in western LIS was not observed in our study area in eastern LIS during 2002. However, the abundance of lobsters (CPUE) in the MPS area declined during 2002 to the lowest levels observed in our 25-year monitoring program. If lobster recruitment in our area depends on larval production in western LIS, the lobster mortality event in western LIS could be responsible for the declining catch and may affect catches in the future. This source-sink relationship between western and eastern LIS and the long-term sustainability of the Connecticut fishery may be further threatened by the widespread outbreak of shell disease in our area over the past four years.

The most notable changes in the population characteristics of local lobsters in recent years, including 2002, were observed in the proportion of egg-bearing females and their size structure, which may be related to changes in lobster growth and the size at which females become sexually mature. The benefits of earlier maturation may be significant to lobster population dynamics. Small size at maturity and subsequent egg production from sublegal-sized females may explain why LIS lobsters are so resilient in the face of intense exploitation. Currently, the smaller size at sexual maturity allows females to spawn once or twice before reaching the legal size and could maintain current population levels into the future. However, these benefits may now be offset by the recent outbreak of shell disease. We found that egg-bearing females had the highest prevalence and severity of shell disease, although the effects of the disease on egg-development, hatching success and larval survival are unknown. If berried females suffer higher natural mortality due to shell disease, the effect on egg production and subsequent recruitment in our area could be significant.

The die-off in western LIS, the outbreak of shell disease in eastern LIS and changes noted in lobster growth and size at maturity coincided with natural increases in seawater temperature over the past two decades. Since LIS is the southern extreme of the American lobsters range in nearshore waters, a gradual rise in seawater temperature may have profound effects on the biology and physiology of this species. Results from our lobster monitoring program over the past 25

years suggest that operation of the MPS cooling water system since the early 1970s has not caused a decrease in the local lobster population. However, it is unclear what impacts the power plant may have (if any) on the local lobsters when abundance levels are low. Recent changes in population characteristics and long-term fluctuations in abundance were related more to changes in environmental conditions, most notably seawater temperature, or to other factors mediated by temperature such as increased susceptibility and transmission of diseases.

Rocky Intertidal Studies

The MPS thermal plume has altered the rocky intertidal community along approximately 150 m of shoreline on the east side of the MPS discharge to LIS. This altered shore community at Fox Island differs from those at other sampling sites in species abundance, distribution, seasonal occurrence and overall community composition. Thermal addition to this site has varied over the study period due to the addition of a second quarry cut in 1983, the start-up of Unit 3 in 1986, the extended shutdown of MPS during 1996-1998, and two-unit operation beginning in 1999 with the permanent retirement of Unit 1.

Seasonal shifts in species occurrence were noted during qualitative algal sampling at Fox Island-Exposed (FE), and were attributed to thermal input from the MPS discharge. These shifts included abbreviated season for cold-water species (e.g., *Monostroma grevillei*, *Spongomorpha arcta*, and *Dumontia contorta*) and extended season for warm-water species (e.g., *Grinnellia americana*, *Dasya baillouviana*, and *Bryopsis hypnoides*). Establishment of perennial populations of *Graciliaria tikvahiae* and *Sargassum filipendula* at FE were also detected through qualitative studies during early three-unit years. These populations were not observed during many collections at FE during the shutdown period, but returned after Unit 3 restart in 1998. In addition, a new species to our study, *Hypnea musciformis*, was collected at FE in 2001, and is now a common component of the FE flora

Thermal impacts on species abundance and distribution patterns at FE were most apparent in the low intertidal zone. During three-unit operation, an expansive population of the opportunistic green alga *Codium fragile*, a perennial *Polysiphonia* spp. population, and periodically heavy sets of the blue mussel *Mytilus edulis* replaced the low intertidal *Chondrus* population its associated seasonal epiphytes (e.g., *Monostroma*, *Spongomorpha*) at FE. Following an extended shutdown of MPS from 1996

to 1998, further community changes were noted at FE. *Codium* steadily declined following Millstone restart and is no longer a low intertidal dominant at FE. Seasonally high abundance of *Hypnea musciformis* and expanded populations of *Sargassum filipendula, Corallina officinalis*, and *Gelidium pusillum* now characterize the low intertidal at FE. The low intertidal communities at all other sites continue to resemble typical rocky shore communities throughout New England.

High *Ascophyllum* growth at FN compared to other sites has been noted during many previous study years, and has been attributed to elevated temperatures from the MPS discharge. However, a similar relationship was observed during two growing seasons (1996-97 and 1997-98) while all three units were shutdown, but was not evident during the first year following Unit 3 restart (1998-99) or during 2001-2002, when two units were operating. Natural influences of other factors such as ambient temperature conditions, nutrients and light may play a more important role in determining *Ascophyllum* growing conditions at FN than does thermal plume incursion from the MPS discharge.

In addition to the localized changes noted for rocky intertidal communities in close proximity to the MPS discharge, the rocky intertidal monitoring program has documented regional patterns and modifications unrelated to MPS operation. These include the introduction and spread of an exotic red alga, *Antithamnion pectinatum*, decreases in barnacle abundance in recent years, and a long-term increase in abundance of the common brown rockweed, *Fucus vesiculosus*.

Benthic Infauna

Benthic infaunal monitoring studies in the vicinity of MPS have documented long-term and short-term trends in sedimentary characteristics, community composition and individual species abundances. Analysis of long-term data has provided evidence of acute and chronic impacts that are either related to power plant construction and operation or other, presumably natural, disturbances. Infaunal communities exhibiting evidence of MPS impacts were observed at stations in the immediate vicinity of MPS (IN, EF and JC). Community changes at these sites were closely associated with changes in sediment composition related to power plant-induced disturbance. The infaunal community at the GN reference site was unaffected by plant operation and provided a baseline from which variability associated with natural environmental cycles and disturbances could be assessed.

The reference site (GN) has exhibited changes in sediments and community structure over the entire period. Sediment mean grain size was highly variable but silt/clay was relatively consistent, with no long-term trends in either parameter. Infaunal community composition at GN has generally been dominated by three taxa, (*Tharyx* spp., oligochaetes and *Aricidea catherinae*); however, some long-term trends in abundance of these taxa were noted. *Aricidea catherinae*, common at GN in the early monitoring years, has declined in abundance, while *Tharyx* spp. abundance has increased over the entire study period. The GN station has been valuable in validating periodic area-wide shifts in species abundance and community structure in response to natural factors unrelated to MPS operation. Most recently, there was a pulse in the abundance of the polychaete *Polydora socialis* at all stations in 2001 indicating a regional increase. Similarly, population shifts occurred for the opportunistic polychaete *Mediomastus ambiseta*, at GN (and at stations near MPS) during 1983-88, 1994, and 1998-99. These increases could not be explained by changes in sedimentary or regional abiotic factors and because they occurred at the reference station, their cause was determined as independent of power plant operation or construction.

Physical disturbances initiated community changes at IN and JC in the period between 1980-1987. Impacts were first noted at IN, were related to dredging and cofferdam removal during Unit 3 construction from 1983 to 1985. Sedimentary characteristics, (primarily silt/clay content levels) have become more consistent over time. Some infaunal community stabilization and recovery was also apparent. Numbers of individuals and species richness at IN have increased, as have abundances of organisms more common in early study years, particularly, *Aricidea catherinae* and *Tharyx* spp. These trends, along with concomitant decreases in abundance of *Nucula annulata* and other opportunistic species (*e.g.*, the amphipods *Leptocheirus pinguis* and *Ampelisca* spp.), indicate some degree of recovery at IN. Recovery at IN is not complete because other organisms which established post-impact community dominance, such as oligochaetes and *Protodorvillea gaspeensis*, have maintained or increased their degree of dominance in recent years and may persist indefinitely

Silt deposition at JC, attributed to sediment scouring from MPS discharge area following Unit 3 start-up in 1986 resulted in increased sediment silt/clay content and abrupt changes in infaunal community structure. The altered infaunal community has exhibited further changes in recent years, including 2002 studies. For example, *A.*

catherinae and *Tharyx* spp. abundances have exceeded levels observed prior to 1986. Other trends toward recovery were noted in the 2002 data (e.g., slightly lower silt/clay content and lower abundances of oligochaetes). However, the opportunistic mollusc *Nucula annulata* that established a population at JC following sediment changes has maintained a population through 2002 that is still well above pre-Unit 3 levels. This biological evidence of disturbance is consistent with trends in sediment characteristics: elevated silt/clay levels in sediments at JC persisted through 2002.

The benthic infaunal community at EF still experiences active MPS operational impact processes. Current scour from the MPS discharge directly impacted both the sediments and the infaunal community 100 m from the discharge at EF, particularly after Unit 3 startup in 1986 and again following the re-start in 1998. Relatively coarse sediments with low silt/clay levels are characteristic of the 3-unit benthic habitat at EF. The altered sedimentary environment at EF, and the infaunal community it supports, continues to show changes under the new environmental conditions created by the 3-unit discharge. The relative stabilization of the sediments and brief plant shutdown allowed for rebounds of *Tharyx* spp. and *Aricidea catherinae*, taxa common at EF during two-unit operation. Oligochaete abundances have generally decreased from high abundances during the early three-unit period and in recent years (2000-2002) remain among the lowest observed during the study period, possibly related to the heavy mussel set observed in the discharge area in 2000. The EF station was the only site in 2002 to have a pulse in abundance of the amphipod *Microdeutopus gryllotalpa*, a species uncommon to any station prior 2002. There was no apparent cause for this pulse.

Eelgrass

Ecologically important eelgrass (*Zostera marina*) populations have been monitored since 1985 at three locations; two near the fringes of the MPS thermal plume in Jordan Cove, and one farther away in the Niantic River. Analysis of long-term trends indicated some degree of decline in eelgrass standing stock biomass at all three areas monitored. The spatial relationship of the long-term eelgrass declines suggests primary causal factors other than the power plant discharge or regional climate change.

Eelgrass beds at the two monitoring sites nearest MPS (JC and WP) have exhibited slight, but statistically significant, long-term declines in two population parameters, standing stock biomass and shoot length. However, compared to the Niantic River, these sites generally supported healthy populations, based on parameters monitored, throughout the 17-year study. These two populations are exposed to variability in water temperature more indicative of natural solar warming and hydro-dynamic conditions in Jordan Cove than the MPS discharge. These natural factors are particularly influential at the JC sampling site, which is the most shallow of the study sites, and is immediately adjacent to extensive shallow sand flats vulnerable to solar warming. In addition to temperature, there does not appear to be any relationship between WP and JC population fluctuations and power plant discharge flow and heat output. With Unit 1 permanently retired, the likelihood of thermal plume incursion at JC and WP in the future is further reduced.

Eelgrass shoot density and standing stock biomass estimates have been lowest and most variable during the study period in the Niantic River, where reference stations have been monitored. A new sampling site was established in the Niantic River in 2000 within the only remaining viable eelgrass bed found in this estuary. Site relocation was necessitated by the 1999 die-off of the eelgrass bed where the previous site had been located. This was the fifth time since 1985 that we have documented loss of an eelgrass bed in the Niantic River. While standing stock and shoot density estimates have remained low at this site since 2000, 2002 data indicate slight improvement. Because the Niantic River is located well away from any influence of the MPS thermal plume, declines at NR sites cannot be attributed to MPS operation. While the causes were not determined, anthropogenic influences such as nutrient loading from surface run-off and groundwater sources may have contributed to observed declines. The Niantic River has a more restricted tidal inlet and is closer to sources of nutrient enrichment than Jordan Cove.

Introduction

Reporting Requirements

This report summarizes results of ongoing environmental monitoring programs conducted in relation to the operation of Millstone Power Station (MPS). MPS 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), last issued by the Connecticut Department of Environmental Protection on December 14, 1992 to Northeast Nuclear Energy Company. The permit was transferred to Dominion Nuclear Connecticut, Inc. (DNC), effective on March 31, 2001. The regulations in the permit allow the MPS 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 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 the Council be informed of results of MPS 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 MPS that were presented most recently in DNC (2002).

Study Area

MPS 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 MPS 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·sec^{-1} and at maximum is about 8,500 m^3·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 typically ranges between 26 and 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 the shoreline. Strong winds, particularly from the southwest, can at times result in locally heavy seas (to 1.5 m or greater) near Millstone Point. Additional information on local hydrography and meteorology can be found in NUSCO (1983).
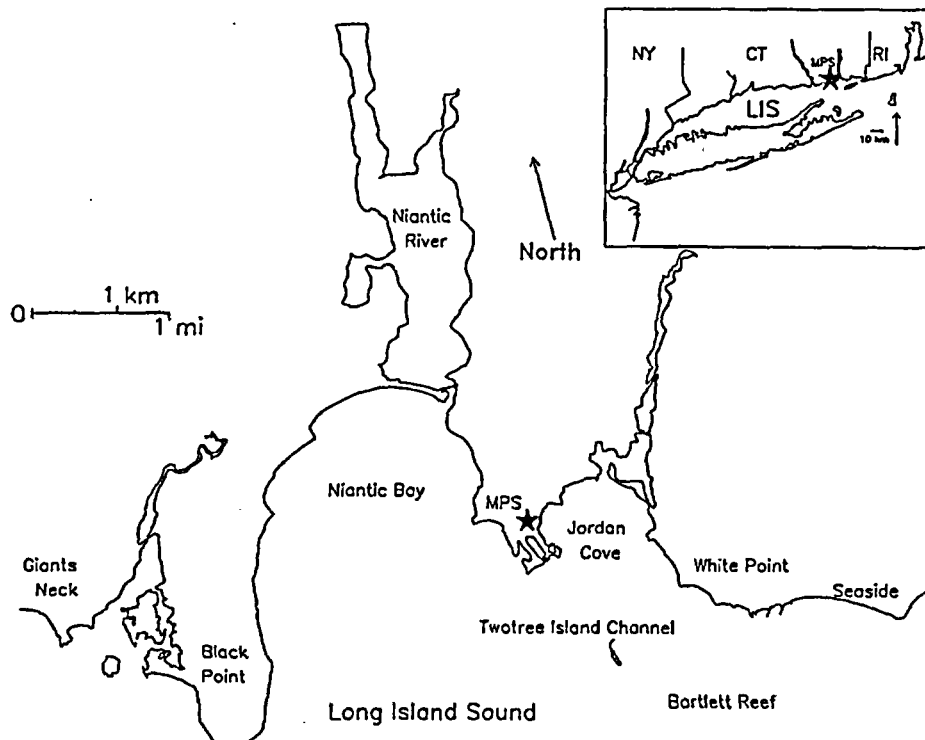
Fig. 1. The study area for biological monitoring conducted to assess the effects of MPS operation.

Millstone Power Station

The MPS complex consists of three 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 and on July 17, 1998 Northeast Utilities announced its retirement. Unit 2 is an 870-MWe pressurized water reactor that began commercial operation in December 1975 and Unit 3 (1,150-MWe pressurized water reactor) commenced commercial operation on April 23, 1986. All units use once-through cooling water systems with rated flows of 28.0, 36.1, and 59.4 $m^3 \cdot sec^{-1}$ for Units 1 through 3, respectively, including condenser and service water flow. Cooling water is drawn by pumps from depths of about 4.6-7.6 m below mean sea level within 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, at Unit 2 in May 2000, and at Unit 3

during its construction to return aquatic organisms washed off the traveling screens back to LIS. Installation and operation of sluiceways have minimized the impact of impingement at MPS (NUSCO 1986, 1988a, 1994, DNC 2001). A chronology of significant events associated with MPS construction and operation, including installation of devices designed to mitigate environmental effects and unit operational shutdowns exceeding 2 weeks, are found in Table 1. For reasons beyond the scope of this report, extended shutdowns occurred between late 1995 and early 1999, with Unit 3 returning to service in July 1998 and Unit 2 in May 1999 (Table 1). Some service water and condenser cooling water pumps were operated at each of these units during the period of extended shutdown, but only limited amounts of service water were withdrawn at Unit 1 after late 1995. Cooling-water use varied throughout the life of the station, depending upon plant operation, and total monthly flow (in millions of m³) from January 1976 through December 2002 is shown in Figure 3. During 2002, Unit 2 utilized 89.3% of its nominal maximum flow and Unit 3 totaled 92.9%, which reflected the high capacity and sustained operation of both units this year, even with the refueling outages at both units that occurred during 2002 (Table 1). Using presently permitted capacity for all three MPS units, the station utilized 70.8% of permitted cooling-water volume in 2002.

TABLE 1. Chronology of major construction and operation events at MPS through 2002.

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	EL
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	EL
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	EL
March 19, 1976	Unit 3 coffer dam construction finished	EL
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	EL
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	EL
May 28, 1983 to January 12, 1984	Unit 2 shutdown	DNGL
December 1983	Fish return system installed at the Unit 1 intake	EL
August 1983	Second quarry cut opened	EL
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	EL
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
September 20 to December 18, 1986	Unit 2 shutdown	DNGL
December 1 to 15, 1986	Unit 1 shutdown	DNGL

TABLE 1. (cont.).

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; EL
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; EL
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; EL
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; EL
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	EL
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 15 to May 23, 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	Unit 1 shutdown; retirement of unit announced July 17, 1998	MOSR; EL
February 20, 1996 to May 11, 1999	Unit 2 shutdown	MOSR
March 30, 1996 to July 5, 1998	Unit 3 shutdown	MOSR
December 11-29, 1998	Unit 3 shutdown	MOSR
June 4, 1999	Original Quarry Cut Fish Barriers replaced	EL
April 30 to June 29, 1999	Unit 3 shutdown	MOSR
April 22 to June 1, 2000	Unit 2 shutdown	MOSR
May 23, 2000	Commenced operation of Unit 2 fish return sluiceway	EL
February 3 to March 30, 2001	Unit 3 shutdown	MOSR
February 16 to April 2, 2002	Unit 2 shutdown	MOSR
September 7 to October 7, 2002	Unit 3 shutdown	MOSR

[a] DNGL refers to the daily net generation log, EL to Millstone Environmental Laboratory records, EDAN to the environmental data acquisition network, and MOSR to the monthly nuclear plant operating status report.
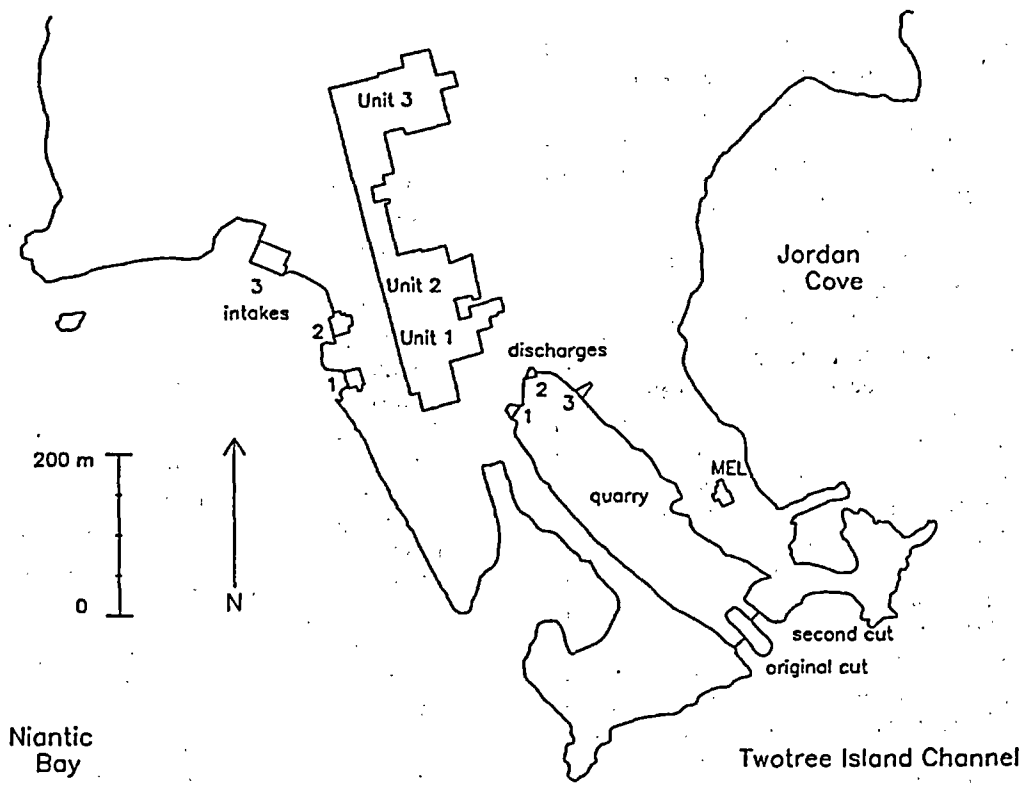
Fig. 2. The MPS site, showing the intake and discharge of each unit, the quarry, and the two quarry discharge cuts.
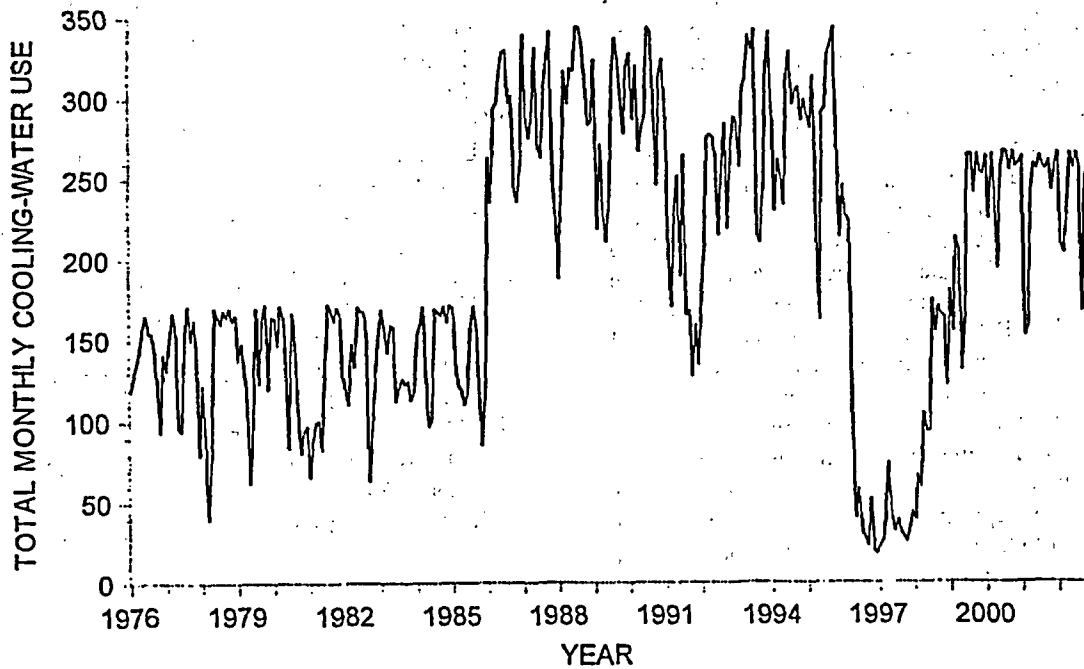

Fig. 3. Total monthly cooling-water use in millions of m³ at MPS from January 1976 through December 2002.

MPS cooling water can be heated in Units 1 (formerly), 2, and 3 from ambient temperature to a maximum of 13.9 (formerly), 12.7, and 9.5°C, respectively. Each unit has a separate discharge structure that releases the heated effluent into an abandoned granite quarry (ca. 3.5 ha surface area, maximum depth of approximately 30 m). 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 discharge, which is typically about 10.5°C warmer than ambient under present station operation, exits the quarry through two channels (cuts), whereupon it mixes with LIS water (Fig. 2). All previous hydrothermal surveys conducted at MPS were described in detail in NUSCO (1988b). The thermal plume is warmest in the immediate vicinity of the cuts and the surface-oriented plume from three-unit operation was shown to cool to less than 2.2°C above ambient within about 1,100 m of the quarry (Fig. 4). Beyond this distance the plume is highly dynamic and varies mostly with tidal currents. Having a smaller volume, the plume from two-unit operation covers less area than did the former full three-unit operation shown in Figure 4 and calculations of its present extent are given in Adams (2001)

Monitoring Programs

This report contains a separate section for each major monitoring program, some of which have been conducted without interruption or significant change since 1976. These long-term studies have provided the representative data and scientific bases necessary to assess potential biological impacts as a result of MPS 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.

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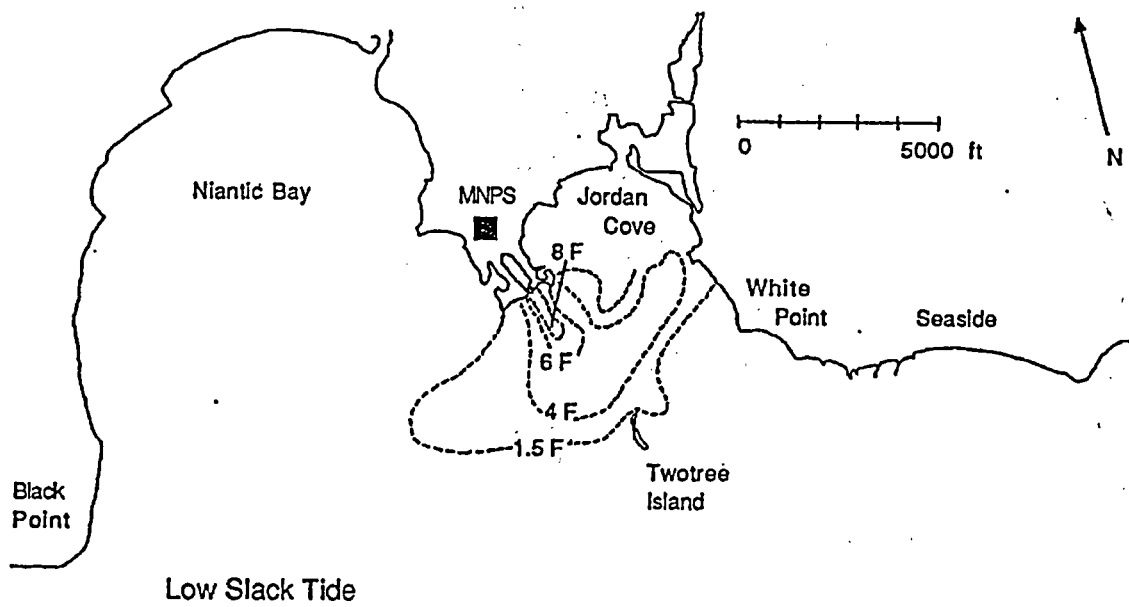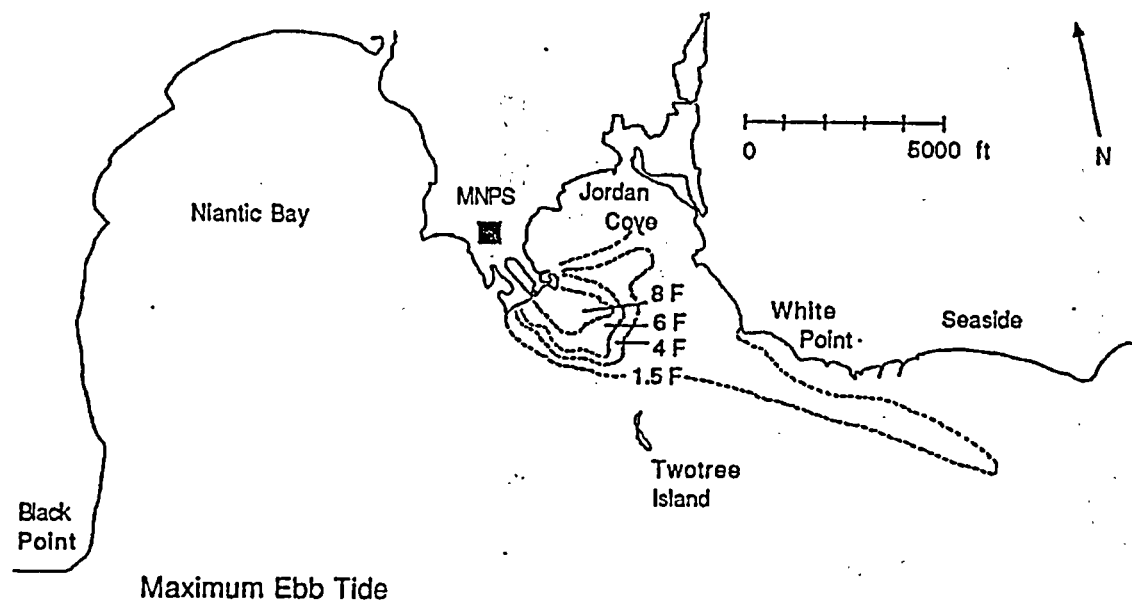
Fig. 4. Locations of selected three-unit thermal plume isotherms (1.5°F, 4°F, 6°F, and 8°F) under various tidal conditions.
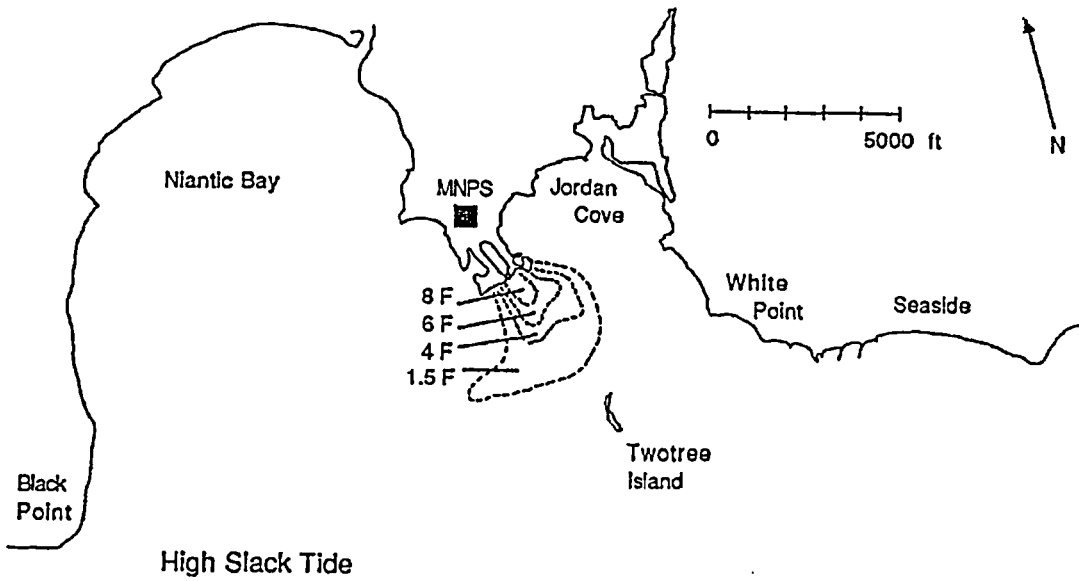
Maximum Flood Tide

High Slack Tide
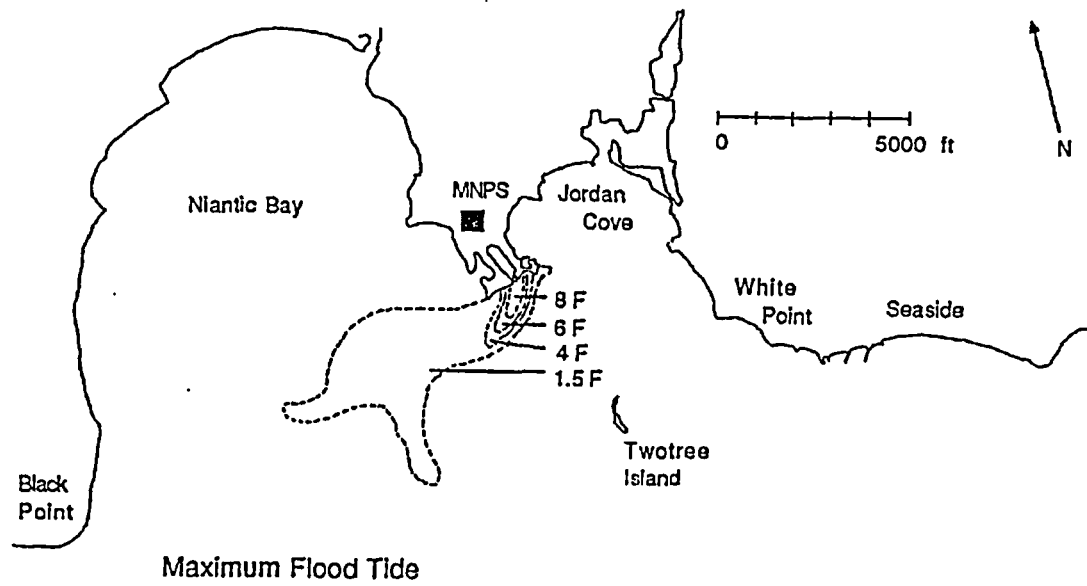
Fig. 4. (continued).

Winter Flounder Studies

Winter Flounder Studies

Introduction

Since 1973, the winter flounder (*Pseudopleuronectes americanus*; classification of Pleuronectidae revised by Cooper and Chapleau 1998) has been the subject of environmental impact studies at Millstone Power Station (MPS). Results of these studies have been provided to the Connecticut Department of Environmental Protection (DEP) in a series of annual reports and special submissions. This species is an important sport and commercial fish in Connecticut (Smith et al. 1989) and a dominant member of the local demersal fish community. Its abundance tends to be cyclical and population sizes fluctuate considerably (Jeffries and Johnson 1974; Jeffries and Terceiro 1985). Of particular concern have been the effects of the station on the local Niantic River spawning population.

The winter flounder occurs 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). Seasonal movements and reproductive activities of winter flounder are well documented (e.g., Klein-MacPhee 1978). Details of winter flounder, abundance, distribution, and size information in LIS were provided by Gottschall et al. (2000) and spawning information by Howell and Molnar (1996, 1998). Most adult fish enter inshore waters in late fall and early winter and spawn nocturnally 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; Stoner et al. 1999). Females require 3 years for oocyte maturation and one batch of eggs matures each year (Dunn and Tyler 1969; Dunn 1970; Burton and Idler 1984). In eastern LIS, females begin to mature at age-3 and most are mature at age-4 with most males maturing at age-2 (NUSCO 1987), although Johnson et al. (1998) reported a small percentage of both age-2 females and age-1 males in LIS as mature. The current estimated mean fecundity of Niantic River females is approximately 585,167 eggs per fish. Stoner et al. (1999) described spawning behavior in a large experimental laboratory aquarium. Crawford (1990) reported that winter flounder deposit the demersal and adhesive eggs on gravel bars, algal mats, eelgrass (*Zostera marina*) beds, and near freshwater springs. Viable hatch of winter flounder eggs is greatest at 3°C in salinities of 15 to 35‰ and decreases with increasing temperature (Rogers 1976). Egg incubation ranges from 5 to 31 days (Rogers 1976) with Keller and Klein-MacPhee (2000) reporting hatching on 20 days at 4.1°C and 30 days at 1.6°C. Larval development through metamorphosis takes about 6 to 8 weeks (Laurence et al. 1979) and is also 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 in which their survival may be reduced. Older, larger larvae maintain some control over their position by vertical movements and may spend considerable time on the bottom. Following metamorphosis, most young-of-the-year winter flounder settle or move into shallow inshore waters. Densities can be highest in unvegetated areas adjacent to eelgrass beds, although habitat use by young varies (Goldberg et al. 2002). Yearlings (age-1 fish) become photonegative and are typically found in deeper waters (Pearcy 1962; McCracken 1963; Casterlin and Reynolds 1982). 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). Klein-MacPhee (1978) and Pereira et al. (1999) summarized other aspects of winter flounder life history. Because the early life history of the European plaice (*Pleuronectes platessus*) 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.

MPS operation can result in the impingement of juvenile and adult winter flounder on the traveling screens of the cooling-water intakes and the entrainment of larvae through condenser cooling-water systems. The impact of impingement at MPS was largely mitigated by the installation and operation of fish return sluiceways at all units, with winter flounder survival exceeding 90% at Unit 3 (NUSCO 1986c, 1988a, 1994b). A fish return at Unit 2 began operation in May 2000 and its effectiveness during

the first year of operation was evaluated in DNC (2001c). Although only 16 winter flounder were collected in this latest study, all survived impingement and a subsequent 72-h holding period. Also in regards to overall station impact, note that although Unit 1 was officially retired in July 1998, the unit was effectively shut down in November 1995. Thus, numbers of fish impinged or entrained at MPS relative to previous three-unit operation (1986-95) have been reduced accordingly as Unit 1 utilized approximately 23% of former total water volume entrained at the station.

Unlike many other marine fishes with large coast-wide populations, the mortality of entrained winter flounder larvae potentially has greater significance as larvae are products of local spawning from 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 MPS cooling-water system. Presently, a combination of sampling programs and analytical methods are used to examine current abundance and trends of the Niantic River population. This report updates results given most recently in DNC (2002) and includes abundance estimates of various life history stages of winter flounder and inferences that can be drawn from the relatively long time-series of available information. Due to the large amount of material presented on MPS winter flounder studies, except for some general information on methodology immediately following this section, detailed materials and methods (e.g., field sampling, data analyses) are presented within appropriate subsections of Results and Discussion.

A mass-balance model, discussed below in the section entitled MPS Impact Assessment, provides an annual estimate of the fraction of the Niantic River winter flounder reproductive output removed by larval entrainment at MPS. This model was re-evaluated in DNC (2001b) and both DNC and two independent reviewers analyzed the sensitivity of input parameters. Studies of more direct methods (i.e., larval stock identification) to assign entrained winter flounder larvae to source populations were undertaken in 2000-2, including the use of genetic (DNA) and chemical composition (microelements) techniques, reports of which were previously submitted to DEP (Crivello 2002; Moran 2002) and a final report for the 2002 DNA work is forthcoming (Crivello in preparation). The goal of these studies was to provide direct quantitative estimates of entrainment impacts to the Niantic River and other nearby winter flounder populations. The fractions of

entrained winter flounder larvae attributed to the Niantic River from this work are compared to results of the mass-balance model later in this report.

A computer simulation model, the winter flounder stochastic population dynamics model (SPDM), first described in NUSCO (1990), has been used to examine long-term effects of MPS operation. The SPDM simulates the long-term effects of historical and projected rates of fishing mortality and simultaneous plant operation. Plant operation results in annual losses from impingement of juveniles and adults and, using estimates of production loss to the Niantic River stock provided by the mass-balance model, the entrainment of larvae through the MPS cooling-water system (Lorda et al. 2000). This model was also modified for use in evaluating various cooling-water system alternatives in DNC (2001b). This work was an extensive evaluation of the MPS cooling-water system and included investigations of the feasibility of reducing either once-through condenser cooling water flow or larval fish entrainment. The effectiveness of various alternatives in reducing entrainment mortality on winter flounder and other fishes was determined. In both DNC (2001b) and in the present application, an assumption was made that the period of MPS operation would be extended beyond the existing U.S. Nuclear Regulatory Commission (NRC) license period for both Units 2 and 3 as DNC is currently investigating the possibility of a 20-year license renewal for both units, which would extend their operational lifetimes to 2035 and 2045, respectively. However, no formal submission has yet been made to the NRC.

General Materials and Methods

Data needed to assess MPS impact on the winter flounder come from several biological sampling programs (Fig. 1). Specific timing and seasonal duration of sampling was designed to reflect the annual life cycle of Niantic River winter flounder. Some programs (e.g., Niantic River adult and larval surveys, age-0 survey) investigate specific life history stages of winter flounder. Information on juvenile (age-0 and age-1) winter flounder was obtained from three sources: a special sampling program specifically targeted post-larval young-of-the-year; catches of age-0 juveniles from specific stations and months of the trawl monitoring program (TMP); and the Niantic River adult spawning abundance surveys during which age-1 juveniles are incidentally collected. Thus, data on juvenile fish abundance were available from about May of their birth year into April of the following year. Other programs sample an entire

5	Trawl Monitoring														
4				Age - 0 Juvenile Survey											
3			Ichthyoplankton Entrainment Monitoring												
2		Winter Flounder Larval Survey													
1	Niantic River Adult Winter Flounder Survey														

F | M | A | M | J | J | A | S | O | N | D | J | F | M | A

Year - 0   Year - 1
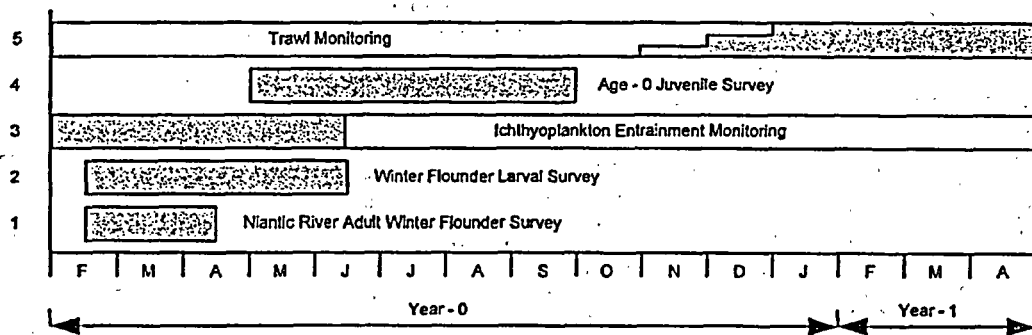
1. February-April sampling (spawning season) of adults and juveniles throughout the Niantic River.
2. February-June larval sampling at three stations in the Niantic River and one in Niantic Bay.
3. Year-round monitoring of all ichthyoplankton at the MNPS discharges.
4. Late May-September sampling of 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 and later) stations near MPS. Juvenile catch data come from two stations in November (all years), four (1976-95) or three (1996 and later) stations in December, and six (1976-95) or three (1996 and later) stations in January and February that allow the year-0 cohort to be followed into a second calendar year.

Fig. 1. Current sampling programs contributing data for computation of winter flounder abundance indices. Darkened areas show months from which data were used in this report for the two year-round sampling programs.

local fish community year-round. These include the TMP, in which catches of winter flounder constitute a major component, and the entrainment ichthyoplankton monitoring program at MPS. 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.

Data from various field sampling programs were used in the calculation of annual and seasonal indices of relative abundance, often with a 95% confidence interval (CI) or ±2 standard errors (SE) indicating precision of the estimate. Indices of abundance were computed for various life-stages of winter flounder, such as various developmental stages of larvae, age-0 and 1 juveniles, and adults, which include spawning females and estimates of egg production. For some indices a long-term mean was calculated with variability described by the coefficient of variation (CV = 100 × [SD / mean], where SD = standard deviation). Spearman's rank-order correlation (Snedecor and Cochran 1967; Hollander and Wolfe 1973) was used to examine the correlation among various time-series of abundance indices. In some instances, a nonparametric, distribution-free Mann-Kendall test (Hollander and Wolfe 1973) was used to determine the direction of change if an annual abundance time-series represented a significant ($p \leq$ 0.05) trend. Most data analyses were carried out using the Statistical Analysis System (SAS) computer programs (SAS Institute Inc. 1990a-d). The type of

abundance index selected in each case depended upon the particular stage of life, sampling effort, and suitability of the data. A detailed description of each index is given below in various subsections of the Results and Discussion.

Results and Discussion

Seawater Temperature

Water temperature is important because it affects many biological processes such as adult distribution and spawning; larval growth, development, and mortality; and the settlement, distribution, growth, and mortality of demersal young. Information on water temperature was obtained from continuous temperature recorders at the intakes of MPS. Daily mean temperatures (°C) were determined from available records of 15-min average temperatures, from which monthly, seasonal, or annual means were calculated. During annual Niantic River adult winter flounder surveys, representative water temperature measurements were taken at the surface and bottom with a YSI Model 30 Salinity/Temperature/Conductivity meter.

Seawater temperatures recorded at the MPS intakes in 2002 set records for warmest mean monthly temperatures in February, March, April, and August and were among the top three in January, May, July, and September (Table 1). Means for June and

TABLE 1. Monthly and annual mean seawater temperature (°C) from January 1976 through December 2002 as calculated from mean daily water temperatures recorded continuously at the intakes of MPS Units 1 and 2 through July 2001 and at Units 2 and 3 thereafter.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual mean
1976	3.65	3.31	4.81	7.55	10.75	15.11	18.29	19.60	18.93	15.04	9.28	4.73	10.90
1977	0.67	0.36	2.85	5.66	10.72	14.92	19.08	20.33	19.41	15.58	12.18	6.72	10.84
1978	3.01	1.09	1.67	4.85	9.10	14.24	17.68	19.82	19.24	16.14	12.47	7.74	10.64
1979	4.53	1.48	3.35	5.93	10.50	15.57	18.84	20.91	20.05	15.99	12.41	8.60	11.57
1980	5.16	2.38	2.80	6.38	10.44	14.76	18.44	20.23	20.16	16.07	10.25	5.73	11.10
1981	1.06	2.63	3.36	6.40	10.19	15.48	19.51	20.86	19.94	14.75	11.07	6.29	11.01
1982	2.20	1.56	3.04	5.41	10.06	14.16	17.98	21.10	20.01	15.95	12.47	8.97	11.13
1983	5.58	3.74	4.55	7.07	10.50	15.05	19.10	19.17	20.57	17.37	12.57	7.90	11.98
1984	4.84	4.02	3.98	6.58	10.84	15.53	18.90	20.60	19.52	16.41	13.04	9.07	11.97
1985	4.36	2.36	4.17	7.02	10.95	14.99	18.98	21.24	20.44	17.46	13.14	7.95	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	19.44	18.86	15.35	10.34	6.99	10.58
1997	3.72	3.20	4.16	6.50	9.38	13.66	17.94	19.47	19.06	15.83	10.99	6.31	10.90
1998	4.89	4.09	4.94	7.58	11.85	15.16	18.79	20.85	20.34	16.40	11.51	8.68	12.14
1999	4.97	4.59	5.30	8.27	12.01	17.38	20.05	21.46	21.07	17.37	13.08	9.00	12.92
2000	4.62	2.73	5.12	7.48	11.43	16.40	19.81	21.05	20.47	17.06	12.72	7.38	12.22
2001	4.08	4.03	4.64	6.74	11.35	16.52	19.53	20.91	20.93	17.08	13.17	10.37	12.50
2002	6.03	5.16	5.77	8.56	11.47	15.87	19.96	21.82	20.71	17.35	12.05	6.86	12.68
Overall mean	4.20	3.07	4.08	6.82	10.77	15.35	18.94	20.52	19.82	16.30	11.98	7.61	11.67
CV (%)	34	39	25	13	7	5	4	3	3	6	9	18	6

October were also above average. At the end of the year, November was slightly warmer than the 27-year mean and only December had a below average mean monthly temperature.

The seasonal mean winter temperature of 5.67°C in 2002 was a new record, exceeding the mean for 1991 by nearly 0.3°C (Table 2). The mean of 11.96°C during spring was the third highest recorded and followed a trend of warmer than average springs observed since 1998. The mean summer temperature of 20.83°C was nearly the same value as the 1999 record high of 20.85°C. Only the fall mean water temperature of 12.09°C was not exceptionally warm in comparison to other years since 1976. Because of the generally warm water temperatures in 2002, the annual mean was 12.68°C, the second highest of the series after 1999 (Table 1). Other annual mean temperatures exceeding 12°C also occurred in 1990, 1991, 1995, and 1998-2001.

Monthly mean temperatures were most variable during January through March (monthly CV = 25-39%; Table 1), when winter flounder spawning and early larval development occur. Temperatures were most stable (CV = 3-7%) from May through October,

a period during which juveniles dominated collections of winter flounder and when most energy is allocated to growth rather than to reproductive activities.

The mean water temperatures given above reflected conditions in Niantic Bay, where the MPS intakes are located. Water temperature in the Niantic River usually has a larger annual range with colder temperatures found during winter and warmer water in summer. This most likely occurs because the relatively smaller water volume of the river responds more quickly than LIS to changes in air temperature and solar radiation. During March, when considerable spawning, egg incubation, and larval development take place, mean water temperature in the Niantic River was determined from spot readings taken on several days each week during the annual adult winter flounder surveys. From 1976 through 2002, these means differed from those recorded at MPS by 0.6°C or less. In about one-half of the years, the water temperature in March was slightly warmer in the river than in Niantic Bay and in the remainder, the river was slightly cooler. Data were insufficient to calculate comparative means during three surveys.

TABLE 2. Seasonal[a] mean seawater temperature (°C) from 1976 through 2002 as calculated from mean daily water temperatures recorded continuously at the intakes of MPS Units 1 and 2 through July 2001 and at Units 2 and 3 thereafter.

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
1997	3.71	9.84	18.82	11.04
1998	4.65	11.53	19.99	12.21
1999	4.97	12.55	20.85	13.15
2000	4.19	11.76	20.45	12.38
2001	4.26	11.54	20.45	13.55
2002	5.67	11.96	20.83	12.09
Overall mean	3.80	10.98	19.77	11.97
CV (%)	28	7	3	8

[a] Winter defined as January-March, spring as April-June, summer as July-September, and fall as October-December.

In 2002, most water temperatures in the Niantic River were similar to those recorded at the MPS intakes in Niantic Bay. However, particularly warm (7.3-8.8°C) water temperatures were recorded at station 54 in the upper river (Fig. 2) during and after mid-March, several degrees warmer than found in the basin (sta. 51) and in the lower river.

Adult Winter Flounder

Field Sampling

Sampling methodology of the adult winter flounder spawning surveys in the Niantic River has been consistent since 1983 (Danila 2000). Surveys usually begin between mid-February and mid-March after most ice cover disappears from the river and continue into April. Sampling ceases when the proportion of reproductively active females decreased to less than 10% of all females examined for 2 consecutive weeks, indicating that most spawning is completed.

In these surveys, the Niantic River was divided into a number of sampling areas, which are referred to as stations (Fig. 2). No samples were taken outside of the navigational channel in the lower portion of the river during 1979-2000 because of an agreement made with the East Lyme-Waterford Shellfish Commission to protect bay-scallop (Argopecten irradians) habitat. With the present lack of both bay scallops and extensive eelgrass beds in the river, permission was obtained to sample shallow areas and in 2001 a few tows were randomly taken in the area immediately south of station 51, west of station 6, and east of the navigational channel, extending as far south as the juvenile winter flounder station WA shown on Figure 2. Due to a lack of winter flounder along with heavier seaweed loads found there in 2001, this area was not sampled in 2002.

Winter flounder were sampled on at least 2 days of each survey week using two survey vessels. A 9.1-m otter trawl with a 6.4-mm bar mesh codend liner was used to capture winter flounder, which were held in water-filled containers aboard the boat before they
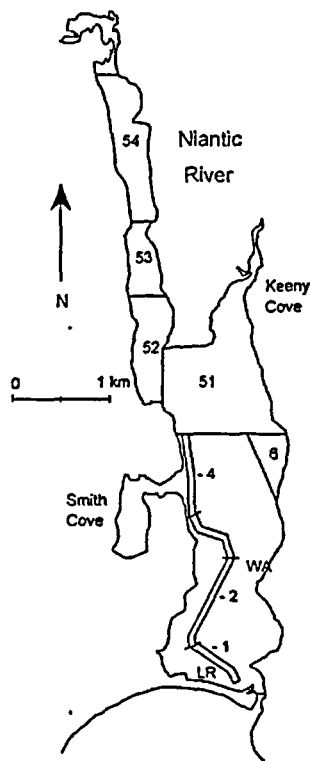
Fig. 2. Location of stations sampled in the Niantic River during 2001 for adult winter flounder from February 19 through April 3 (numbers) and age-0 winter flounder from May 28 through September 30 (letters).

were processed. Since 1983, all fish larger than 20 cm were measured to the nearest mm in total length and had 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 and, at minimum, all winter flounder examined were classified as either smaller or larger than 15 cm. Gender and reproductive condition of larger winter flounder were determined by either observing eggs or milt, or as suggested in Smigielski (1975), by noting the presence (males) or absence (females) of ctenii on left-side caudal peduncle scales. Before release, healthy fish larger than 15 cm (1977-82) or 20 cm (1983 and after) were marked in a specific location with a number or letter made by a brass brand cooled in liquid nitrogen.

Since 1983, the brand mark and location have been varied in a unique manner such that the year of marking was apparent for recaptured fish. During 1999-2001, a mark (termed photonic tag) was also applied by the injection of fluorescent-pigmented particles into the dorsal, anal, or caudal fin of most

freeze-branded winter flounder using a special injector (NEW WEST Technologies BIOMETRIX System 1000 MICRO-Ject™ or SuperMICRO-Ject™ portable injectors). Because of problems with the MICRO-Ject™ applicator in 1999, not all freeze-branded fish received a photonic tag that year, but this was corrected prior to the 2000 work. The efficacy of this system versus freeze branding was examined by marking winter flounder using each technique and comparing marks observed on recaptured fish, including fish marked during the 1999-2001 spawning surveys and in the 2000 special post-spawning sampling survey (for details of the latter work, see the section entitled Post-Spawning Survey of Adult Winter Flounder in DNC 2001a). This sampling was also expected to provide additional information on the utility of the annual CPUE index as a measure of spawning winter flounder abundance because a comparison could be made between the proportions of fish recaptured in 2002 marked in either the regular and post-spawning surveys.

Relative Annual Abundance

Data Analyses. Trawl catch-per-unit-effort (CPUE) is used to describe the relative annual abundance of winter flounder ≥15 cm taken in the Niantic River during the late February-early April spawning season. Following standardization of collection methods and catch data, a Δ-mean CPUE was established as the annual relative abundance index of Niantic River winter flounder spawners (NUSCO 1988c). Components of standardization for CPUE calculation included tow length, tow duration, weekly effort, minimum fish length, and, in some instances, gender (see Spawning Stock Size and Egg Production, below). Tow distance was initially measured using radar or LORAN, but more recently with differential Global Positioning System. Tow distance was fixed in 1983 (with exceptions noted below) because using the same tow length at all stations was expected to reduce this component of variability. 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. However, in 1990, tow distance at station 1 was reduced to 0.46 km because of a new bridge that was constructed at the river mouth. Particularly during 1987, 1989, 1990, and 2000, tows one-half to two-thirds of the standard length were commonly taken, mostly in the upper river at stations 6 and 51, to avoid overloading the trawl with macroalgae and

detritus. Because catch data from station 2 were also used in the TMP, tows there were made over 0.69 km, the standard for that particular sampling program, although shorter tows were often made at this location as well due to high detrital loads.

Because a standard tow distance was not set prior to 1983, catches of winter flounder made during each spawning survey were standardized to a 15-min tow duration at stations 1 and 2 and the shallow flats area and to a 12-min tow time at the other locations. In general, tow duration was usually greater in the lower river than in the upper river due to differences in tidal currents and amounts of 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, most of which occurred 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.

Finally, 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 that year.

The 2002 Spawning Survey. The 2002 adult winter flounder spawning survey began on February 19 and the survey was completed on April 3 (Table 3). Effort was relatively consistent, with 52 to 57 trawl tows taken weekly. The season was determined by the rate of female spawning, which was found by observing weekly changes in the percentage of gravid fish larger than 26 cm, the size at which about half of all observed females were mature (NUSCO 1988b). Most spawning in 2002 was completed by late March, as illustrated by the proportion of gravid females observed (Fig. 3). Although the weekly fractions of observed females that were gravid at the start of the survey varied between about 30 to 60% during 1999-2002, they decreased relatively similarly through late March. Even though 2002 was a record warm winter, proportionately more gravid females were observed in late March and early April than during the previous 3 years. Because a majority of females were spent at the start of surveys in most years, much of the spawning apparently occurs earlier in winter. Exceptions include extremely cold years (e.g., 1978, 1996, 2001), when many gravid females were present even after relatively late starting dates of the surveys because of heavy ice cover (Fig. 4). In contrast, the proportions of gravid females in warmer years (e.g., 1989-91) were not only lower at the start of sampling, spawning was also completed earlier, even though the surveys commenced earlier in the season.

TABLE 3. Time period and duration of annual surveys in the Niantic River during the winter flounder spawning season from 1976 through 2002.

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[a]
1990	February 20 - April 4	7
1991	February 13 - March 20	6
1992	February 18 - March 31	7
1993	February 16 - April 7	8[b]
1994	March 22 - April 13	4
1995	February 28 - April 6	6
1996	February 27 - April 3	6
1997	February 18 - March 25	6
1998	February 9 - April 1	8
1999	February 23 - April 6	7
2000	March 1 - April 5	6[c]
2001	February 26 - April 11	7
2002	February 19 - April 3	7

[a] Limited sampling during week 2 because of ice formation.
[b] Almost no sampling during week 3 and limited sampling during weeks 2 and 5 because of ice and weather conditions.
[c] An additional 6 weeks of post-spawning sampling conducted from April 10 through May 17.

The large concentrations of macroalgae that hindered sampling in many areas of the river during the late 1980s and early 1990s decreased as of the late 1990s, with notable exceptions. Similar to the
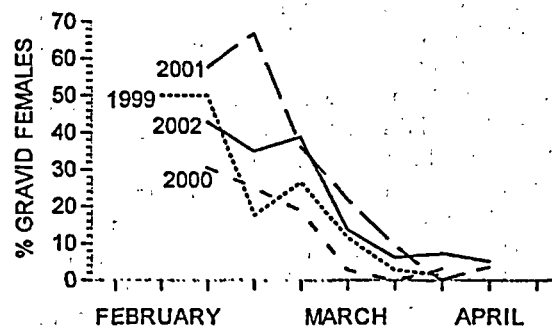
Fig. 3. Weekly percentage of gravid Niantic River female winter flounder larger than 26 cm taken during the 1999-2002 adult population abundance surveys.

WARM YEARS

COLD YEARS
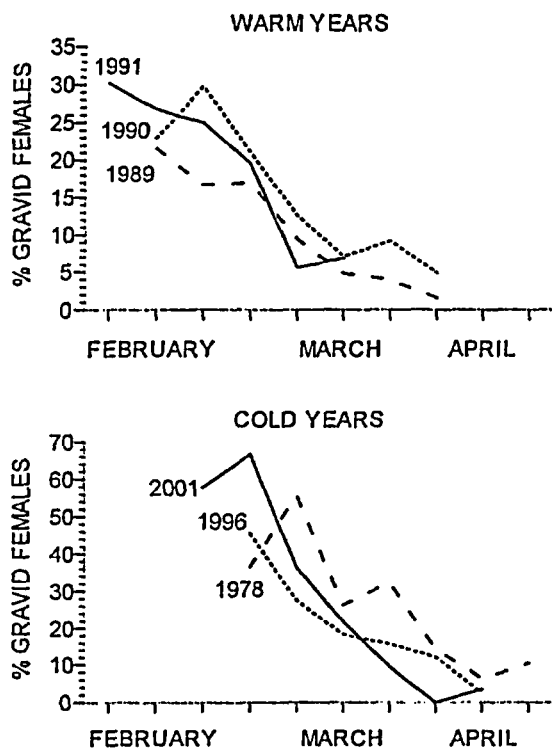
Fig. 4. Weekly percentage of gravid Niantic River female winter flounder larger than 26 cm taken during representative warm (1989-91) and cold (1978, 1996, 2001) years of adult population abundance surveys. (Note that the vertical scales differ between the graphs).

past few years, less material was taken in most areas of the northwestern river arm (stas. 52-54) and in the basin (stas. 6 and 51). However, very close to shore and at station 4, seaweeds were more common and precluded sampling in these areas. The large masses of blue mussel (*Mytilus edulis*) found covering the bottom throughout station 1 in the lower river navigational channel during 2001 were gone, although dead shell material was quite common in the trawl collections.

As winter flounder abundance decreased through the 1990s, adults appeared to concentrate into relatively small areas of the Niantic River, such as the upper river arm (stas. 52-54) and particularly at station 54. Few fish were taken at stations 1, 2, or within large portions of 51. Along the eastern shoreline of the river (stas. 6 and adjacent 51), ripe males were more abundant than females, indicating that these areas were likely spawning sites. Because large adults were also common in the shallow area of northernmost station 54, winter flounder probably complete most spawning in shallower river areas.

Relative Abundance Index. Because of an increasing frequency of zero catches in the mid-1990s, the relative abundance index was changed from a median to a Δ-mean CPUE (described in NUSCO 1988c), which is the best estimator of the population mean when the data come from a distribution that contains numerous zero values and the distribution of the non-zero values is approximately lognormal (Pennington 1983, 1986). Prior to 1993, 97-100% of tows taken each year had at least one winter flounder ≥15 cm, but the fraction each year since then varied from as high as 95% to a low of 69% (2001); 76% of the tows in 2002 had at least one winter flounder ≥15 cm. Even though catches of winter flounder were sparse in 2002, the Δ-mean CPUE of 2.2 fish per standardized tow represented a 50% increase over 2001 (Table 4; Fig. 5). Nevertheless, the abundance of winter flounder spawners in the Niantic River remains at a low level with CPUE since 1995 fluctuating around 2 fish per tow.

Low levels of winter flounder abundance were also indicated by the DEP trawl survey, which takes place throughout LIS. Gottschall et al. (2002) reported that the 2001 spring survey abundance index for winter flounder (fish of all ages taken in April and May) was the third lowest of the time-series after those for 1999 and 2000 and was about one-half of the long-term (1984-2001) mean. Age-2 and 3 fish had particularly low abundances. The age-4+ index (adults) in 2001 was just over the long-term average. In the same DEP annual report, MacLeod (2002) noted that the recreational fishing total catch and harvest of winter flounder in 2000 were the lowest ever recorded for Connecticut. A conclusion of this DEP report was that low abundance of winter flounder in inshore waters was a result of low recruitment to the spawning stock and that biomass is expected to decline further over the next few years as smaller year-classes advance to the adult spawning stock. Preliminary indices from the 2002 DEP trawl survey (V. Crecco, CT DEP, Old Lyme, CT, pers. comm.) indicated another decrease in the winter flounder abundance index for fish of all ages to the second lowest after 1999. Abundance of age-4+ fish was the third lowest after 1993 and 1994, declining by nearly half from 2001.

In comparison to 2000 and 2001, the fractions of female winter flounder of many size-classes were intermediate in abundance during 2002 (Fig. 6). Relatively similar abundances of both smaller and larger fish, however, were found during the past several years. Abundance in 2002 peaked at 31 cm with various peaks occurring between 30 and 36 cm

TABLE 4. 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 2002 adult population abundance surveys.

Survey year	Weeks used for CPUE computation[c]	Tows acceptable for CPUE[d]	Adjusted number of tows used[e]	Non-zero observations	Δ-mean CPUE estimate	Standard error	95% confidence interval for Δ-mean CPUE
1976	7	169	224	222	48.4	2.9	42.7 - 54.1
1977	6	223	264	261	27.5	1.7	24.3 - 30.8
1978	6	150	174	174	31.2	2.3	26.6 - 35.7
1979	5	127	140	140	41.0	4.0	33.0 - 48.9
1980	5	117	150	149	41.5	2.9	35.9 - 47.1
1981	7	181	232	232	50.8	2.5	45.9 - 55.7
1982	5	118	149	149	47.8	3.5	41.0 - 54.6
1983	7	232	238	237	31.3	1.3	28.8 - 33.9
1984	7	246	287	286	18.4	0.7	17.1 - 19.7
1985	7	268	280	277	17.1	0.7	15.7 - 18.5
1986	7	313	343	341	12.2	0.5	11.2 - 13.3
1987	5	234	270	267	16.9	0.9	15.2 - 18.6
1988	6	292	312	310	17.9	0.7	16.5 - 19.3
1989	6	272	306	302	13.9	0.7	12.6 - 15.1
1990	7	307	343	342	11.2	0.5	10.3 - 12.2
1991	6	301	330	324	16.7	0.9	14.9 - 18.5
1992	7	380	406	395	7.7	0.3	7.0 - 8.3
1993	7	288	392	344	3.4	0.2	3.0 - 3.7
1994	4	185	212	201	6.4	0.5	5.5 - 7.3
1995	6	320	342	283	2.6	0.1	2.4 - 2.9
1996	6	310	342	242	1.6	0.1	1.4 - 1.8
1997	6	302	348	288	2.4	0.1	2.2 - 2.6
1998	7	363	385	306	2.1	0.1	1.9 - 2.3
1999	7	336	364	297	2.7	0.2	2.4 - 3.1
2000	6	305	324	248	3.0	0.2	2.5 - 3.4
2001	7	357	371	255	1.4	0.1	1.3 - 1.6
2002	7	382	399	308	2.2	0.1	2.0 - 2.5

[a] Catch per standardized tow (see text for details).
[b] Mostly age-2 and older fish.
[c] Effort equalized among weeks; during several years weeks with very low effort were not used to compute CPUE.
[d] Only tows of standard time or distance were considered.

(probably ages-4 through 6; NUSCO 1987) likely representing variation in year-class strength of spawning females. The decline in female winter flounder abundance over the entire period of sampling since 1976 was evident when catches from 1981 (largest CPUE since 1976; Table 4), 1985,
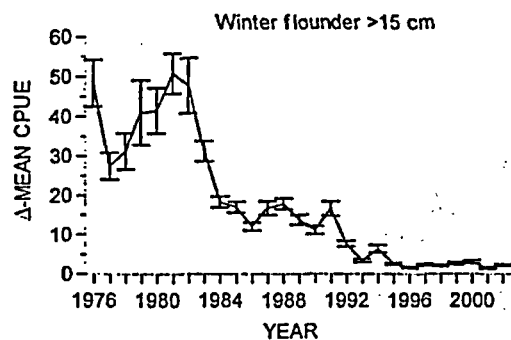
Fig. 5. Annual Δ-mean CPUE and 95% confidence interval of Niantic River winter flounder larger than 15 cm from 1976 through 2002.

1990, and 2002 were compared (Fig. 7). The high abundance of females from 23 to 34 cm in 1981 were likely fish from the strong year-classes produced during the mid to late 1970s. Conversely, low abundance of mid-sized spawners in 2002 was likely related to relatively low production of juveniles in the mid-1990s (see Juvenile Winter Flounder, below).

Since the early 1980s, large decreases in abundance have occurred for all size-classes of female winter flounder, with the exception of the very largest (≥40 cm) females, which were never very numerous. Larger fish have made up a relatively greater proportion of adult female winter flounder in the Niantic River since the mid-1990s. Shake (2001) also noted that large winter flounder were more common in the DEP LIS spring trawl survey in 1999 and 2000 than in previous years. However, despite relatively high abundance of age-0 fish produced in 1988, 1992 and 1994, female winter flounder from 20 to 30 cm in length did not become particularly abundant in subsequent years. Simpson et al. (1996)
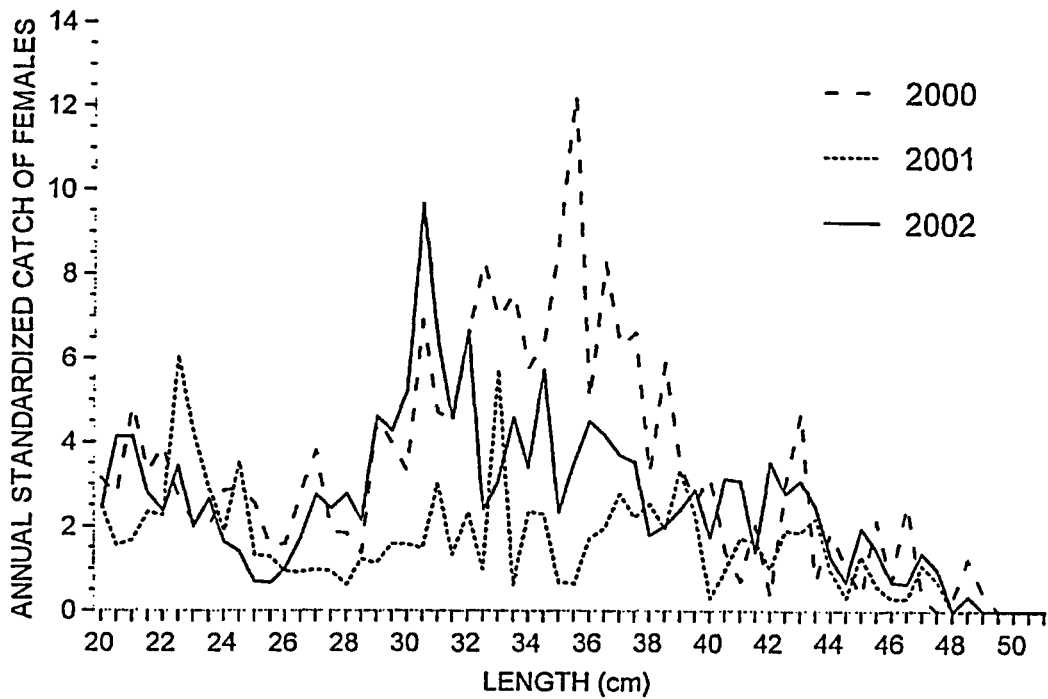
Winter Flounder 19

Fig. 6. 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 2000 through 2002.
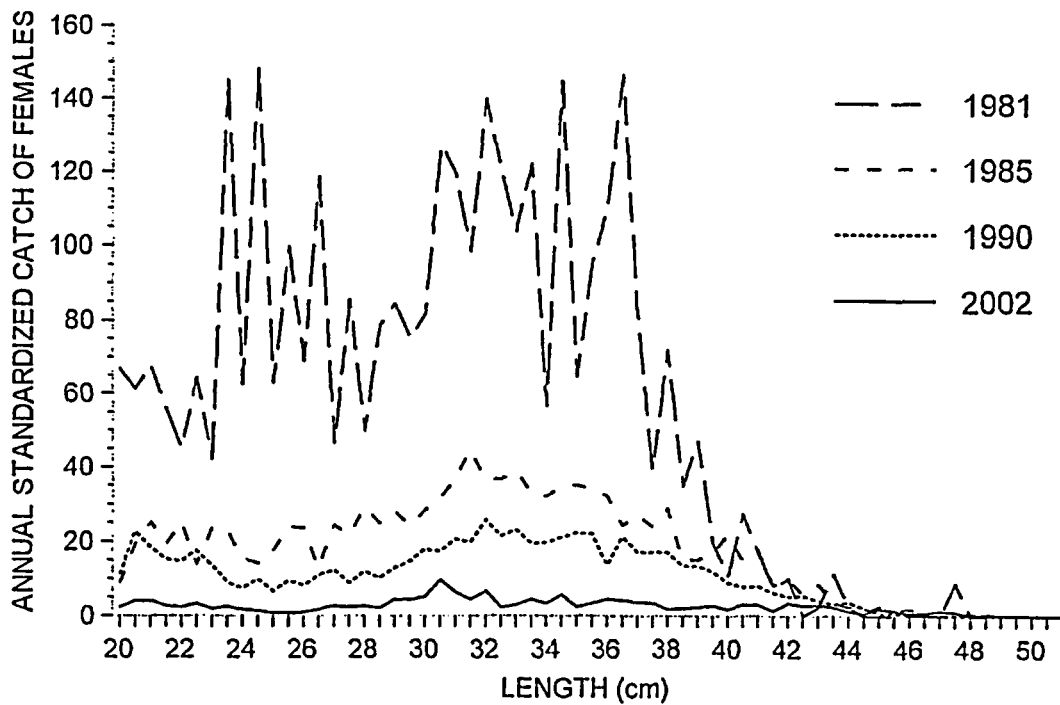
Fig. 7. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season in 1981, 1985, 1990, and 2002.

and Desfosse et al. (1998) noted similar trends for the 1988 and 1992 year-classes of winter flounder in LIS and in Massachusetts, respectively. They reported that apparently strong year-classes observed at ages-1 and 2 did not result in expected higher abundances at age-3 and older. Among the explanations suggested for this phenomenon were variable discard mortality of juveniles in the commercial fishery, high rates of fishing mortality once the winter flounder recruited to the fisheries, and non-random fishing effort that occurred in overfished stocks (Simpson et al. 1996; Desfosse et al. 1998). Factors critical to the formation of winter flounder year-class strength are discussed in a section appearing later in this report.

Another factor affecting winter flounder spawner abundance in the Niantic River during recent years has been seal predation. Particularly in years when abundance is low, this source of mortality may be of importance. Harbor seals (*Phoca vitulina concolor*) were first noted in the Niantic River during the 1993 winter flounder spawning season. This year, fewer seals were observed either inside or outside the Niantic River mouth in comparison to previous years. Only two winter flounder were captured in 2002 that had wounds consistent with seal attacks. Since passage of the Marine Mammal Protection Act in 1972, harbor seals have increased in number (Blaylock et al. 1995) and have expanded their range southward from northern New England (Payne and Schneider 1984). Although some studies show harbor seals prefer to feed on small, schooling fishes, they will also opportunistically prey on seasonally dominant species and have the capacity to shift prey selection rapidly in response to shifts in availability (Payne and Selzer 1989). Sampling also indicated that suitable alternative prey species were not common in the Niantic River during the winter flounder spawning season.

Recaptures from the 2000 Regular and Special Post-spawning Surveys. In 2000, a Δ-mean of 11.8 winter flounder larger than 20 cm was recorded during the special post-spawning survey, a value larger than all annual spawning survey CPUE estimates made since 1991 (DNC 2001a). A total of 784 fish was uniquely marked and released during this mid-April through mid-May sampling, considerably more than the 593 marked during the March-early April spawning survey (both surveys were 6 weeks in duration). Although the post-spawning survey CPUE was considered somewhat biased because effort was not allocated to all areas of the river and was concentrated in the relatively deeper waters of the upper river basin (sta. 51; Fig. 2), the relatively high value nevertheless indicated that many

more winter flounder were available for capture in a large portion of the Niantic River than were found in that area during the regular spawning survey. In addition, larger-sized fish and females were caught in significantly greater proportions in the post-spawning than in the regular spawning survey. However, percentages of fish marked during the 1998 and 1999 spawning surveys (combined totals) and recaptured were nearly the same in each of the 2000 surveys (4.4, 5.4%). These results suggested that winter flounder were perhaps more common in the Niantic River than indicated by the lower CPUE of 3.0 determined in the regular spawning survey. It was also possible that some fish from other spawning stocks could have entered the river to feed as part of normal post-spawning movements in spring. However, recaptures of the fish marked in both 2000 surveys during 2001 and 2002 provided additional insights regarding the status of these winter flounder. The fractions of fish that were marked in the 2000 surveys and subsequently recaptured were practically identical: 2.9% and 2.2% of the spawning survey fish and 2.8% and 2.9% of the post-spawning survey specimens in 2001 and 2002, respectively. The similar recapture rates of previously marked fish indicated that the winter flounder found in the Niantic River from winter into spring likely represented the same stock, with numbers not appreciably increased by immigration from other populations.

These results further illustrated that the magnitude of annual CPUE can be affected by when and where effort is expended. In particular, surveys from 1976 through 1982, when annual winter flounder CPUE values were generally high, were less structured in terms of areal effort allocation. Sampling effort often occurred where fish were most common, and the surveys also went later into spring when winter flounder were apparently more available. As spring progresses, winter flounder withdraw into the upper river basin from other areas. Increases in weekly catches seen during the final weeks of sampling in many surveys likely reflected a similar movement of winter flounder into deeper areas of the river from the shallow flats that were not sampled following increases in water temperature. However, in 2001, a few tows taken on the shallow flat south of station 51 did not produce any more winter flounder than were found in deeper portions of the upper river. Therefore, additional winter flounder present in the river must have been located in other shallow-water areas not surveyed (e.g., close to the shoreline) or in Smith Cove, which cannot be sampled due to numerous moorings and soft sediment.

Absolute Abundance Estimates

Data Analyses. 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 appropriate for open populations as long as basic assumptions are approximately met (see Cormack 1968; Southwood 1978; Begon 1979; Pollock et al. 1990 for details). 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 in years prior to 1984 because of uncertainty in data records and ambiguity caused by re-use of brands from earlier 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).

Comparison of Marking Methodology. During 1999-2001, most winter flounder ≥20 cm in length were tagged with both a freeze brand and a photonic pigment mark (see Adult Winter Flounder – Field Sampling, above). Observations of recaptured fish were expected to provide information on the efficacy of both marking methodologies (i.e., whether marks were successfully retained for 1 or more years). The rationale for this work was that each year a few winter flounder have been caught with an apparent freeze brand, but the exact year of marking could not be ascertained because the mark was indistinct. This loss of mark and recapture information warranted an examination of another tagging methodology.

Observations of both within-year and following-year recaptures showed that no winter flounder were caught that had a photonic brand, but lacked a freeze brand. Thus, initially the freeze brand appeared to be the more stable mark. Only 82% of 55 within-year recaptures of freeze-banded fish in 1999 had an observable photonic mark, although this result was confounded because about one-third of the fish were not tagged or were poorly marked due to equipment problems with this marking system. In addition, even though the photonic mark fluoresces under UV light, difficulties were encountered in the field while using these lights and the light boxes constructed for this purpose. Subsequently, this practice was abandoned. As a result of increased expertise in using the photonic marking equipment, 100% and 96% of within-year freeze-brand recaptures seen during the 2000 regular (total of 69 observed) and special post-

spawning (178) surveys, respectively, also had a photonic mark. In 2001, 93% (41) of fish had both marks, indicating that these brands were relatively comparable in the short term, as long as there were no photonic injector problems. Only 31% (36) of the 1999 photonic marks were observed on freeze-band recaptures in 2000 (both surveys combined) and 20% (10) in 2001, percentages that were expected to be low because not all fish were released with a photonic mark. Use of an estimate of fish not having a photonic mark but released in 1999 resulted in corrected recapture rates of 56% and 36%, respectively, for these 2 years. Only one fish marked in 1999 (also tagged in 2000) was recaptured in 2002 and it lacked the photonic mark. Nearly all fish received a photonic mark in 2000, but only 72% (18) and 85% (26) of fish recaptured in 2001 during the regular and special post-spawning surveys, respectively, had both brand types. Combining data from the regular and special surveys in 2000, only about one-half of the winter flounder had retained the photonic mark in 2002. A similar fraction was found for fish released in 2001. The photonic marks were more difficult to apply, particularly on larger winter flounder, which have a tough integument resistant to retention of the pigment particles. Observations of recaptures were also more difficult, with the pigment mark often very small or faint. Because of occasional equipment problems along with less than complete photonic mark retention in the long-term, the freeze brand remains as the tagging method of choice for the Niantic River winter flounder surveys. Photonic marks will therefore no longer be applied to winter flounder.

The 2002 Spawning Survey. During 2002, the number of winter flounder ≥20 cm in length that were caught and freeze-branded in the Niantic River varied from 50 in the first week of sampling to 141 during the final week. The total of 649 winter flounder marked and released during 2002 was a large increase over the 371 tagged in 2001 and was the largest number branded since 1995 (Table 5). Thirty-nine previously marked fish were recaptured in 2002, also representing the largest number of recaptures since 1996-98, years each having 37 recaptures recorded. The proportion of recaptured fish to the total observed within a year was particularly low (4.1-6.0%) during 1997-2000 in comparison to 1992-96 and 2001 (7.1-14.8%). The recapture rate of 6.0% in 2002 falls within the lower group. The fate of marked adults between spawning seasons is unknown. A previous summary of movements and exploitation of Niantic River winter flounder given in NUSCO (1986a, 1987) indicated that nearly all fish moved from the river to offshore waters by late spring

TABLE 5. Mark and recapture data from 1983 through 2002 used to estimate abundance of winter flounder larger than 20 cm in the Niantic River during the spawning season.

Survey year	Number observed	Number not previously marked	Number marked and released	Number recaptured
1983	5,615	5,615	5,615	0
1984	4,103	3,973	4,083	130
1985	3,501	3,360	3,417	141
1986	3,031	2,887	3,024	144
1987	2,578	2,463	2,573	115
1988	4,333	4,107	4,309	226
1989	2,821	2,589	2,752	232
1990	2,297	2,142	2,275	155
1991	4,333	4,081	4,324	252
1992	2,346	2,136	2,336	210
1993	984	838	972	146
1994	1,035	961	1,033	74
1995	682	604	681	78
1996	379	342	376	37
1997	642	605	642	37
1998	637	600	632	37
1999	617	592	614	25
2000	594	568	593	26
2001	371	342	370	29
2002	649	610	649	39

Number of fish marked in a given year that were recaptured during subsequent annual surveys

Survey year	83	84	85	86	87	88	89	90	91	92	93	94	95	96	97	98	99	00	01
1984	130																		
1985	47	94																	
1986	23	45	76																
1987	2	13	27	73															
1988	7	22	31	62	104														
1989	2	11	9	33	32	145													
1990	1	7	4	15	13	37	78												
1991	1	4	4	12	25	30	50	126											
1992	0	0	1	1	3	21	20	50	114										
1993	0	0	0	1	0	4	3	15	20	103									
1994	0	0	0	1	0	0	4	5	12	25	27								
1995	0	0	0	0	0	1	1	2	8	8	17	41							
1996	0	0	0	0	0	0	0	2	2	4	5	4	20						
1997	0	0	0	0	0	0	0	2	1	6	3	5	7	13					
1998	0	0	0	0	0	0	0	0	1	2	0	4	5	2	23				
1999	0	0	0	0	0	0	0	0	0	0	0	1	0	3	4	17			
2000	0	0	0	0	0	0	0	0	0	2	0	0	0	0	1	5	18		
2001	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	3	8	17	
2002	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1	3	0	12	21

following spawning. Most fish moved to the east, with many fish taken by the sport and commercial fisheries in Connecticut and New York waters of LIS, still others off Rhode Island and southern Massachusetts, and finally, some individuals moved as far as Nantucket Shoals or were taken off southern Cape Cod. These fish return to Connecticut and Niantic Bay waters in fall and early winter prior to entering the river for spawning and are subjected to a local trawl fishery during that period.

Most (21 = 54%) fish recaptured in 2002 were marked in 2001, followed by fish branded in 2000 (12 = 31%). A few fish were taken from 1996-98 and one from 1994 was at large for 7 years; the longest interval between marking and recapture was 8 years. Excluding 1984 and 1985, years at the start of the mark-recapture data series, and 1994 (36%) and 1997 (33%), fish marked during the preceding year accounted for about one-half to two-thirds of all recaptures. During both 1994 and 1997, however,

fewer recaptured fish were marked during the immediately preceding year in comparison to other years. It appears that in recent years fewer fish that had been marked more than 3 years previous to the survey were recaptured in comparison to previous years, but whether this was due to fewer fish being branded and available for recapture, higher rates of mortality following release, or a combination of the two is unknown.

Absolute Abundance Estimates. The mark-recapture data from 2002 provided an initial abundance estimate of 4,124 winter flounder larger than 20 cm and present in the Niantic River during the 2001 spawning season, by far the lowest absolute abundance estimate (Table 6). Because standard errors of N appear to be correlated with N due to the particular form of Jolly's variance formula, the 95% CIs computed are generally considered unreliable as a

TABLE 6. Estimated abundance[a] of winter flounder larger than 20 cm taken during the spawning season in the Niantic River from 1984 through 2001 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				0.330	0.040	0.252 - 0.407
1984	57,978	8,417	41,480 - 74,476	0.555	0.065	0.428 - 0.682
1985	79,508	10,850	58,241 - 100,775	0.363	0.042	0.281 - 0.444
1986	49,418	6,260	37,149 - 61,687	0.652	0.068	0.519 - 0.785
1987	76,180	9,861	56,852 - 95,508	0.589	0.062	0.467 - 0.710
1988	66,290	7,261	52,058 - 80,521	0.462	0.050	0.364 - 0.559
1989	42,312	4,901	32,706 - 51,918	0.373	0.040	0.294 - 0.452
1990	33,168	3,891	25,542 - 40,794	0.908	0.107	0.700 - 1.117
1991	68,155	8,363	51,763 - 84,546	0.189	0.024	0.142 - 0.236
1992	16,969	2,189	12,678 - 21,259	0.486	0.080	0.330 - 0.643
1993	11,927	2,060	7,888 - 15,965	0.489	0.097	0.299 - 0.680
1994	17,811	3,543	10,867 - 24,754	0.444	0.103	0.243 - 0.646
1995	8,715	2,032	4,731 - 12,698	0.461	0.133	0.200 - 0.721
1996	7,551	2,244	3,153 - 11,949	0.470	0.147	0.182 - 0.758
1997	8,968	2,677	3,721 - 14,214	0.317	0.099	0.123 - 0.510
1998	6,118	1,967	2,263 - 9,973	0.448	0.138	0.178 - 0.718
1999	10,340	3,545	3,391 - 17,289	0.368	0.112	0.149 - 0.587
2000	8,427	2,793	2,953 - 13,901	0.346	0.108	0.134 - 0.558
2001	4,124	1,379	1,378 - 6,826			
Mean	31,887	1,304	29,330 - 34,443	0.458	0.012	0.434 - 0.482

Year	Sampling intensity (p)	Standard error of p	95% CI for p	Annual recruitment (B)	Standard error of B	95% CI for B
1984	0.070	0.0102	0.050 - 0.090	47,359	9,077	29,569 - 65,149
1985	0.044	0.0060	0.032 - 0.056	20,626	5,245	10,345 - 30,907
1986	0.061	0.0078	0.046 - 0.076	43,955	8,547	27,202 - 60,707
1987	0.034	0.0044	0.025 - 0.042	21,477	6,351	9,028 - 33,925
1988	0.065	0.0072	0.051 - 0.079	11,722	3.735	4,402 - 19,043
1989	0.066	0.0078	0.051 - 0.082	17,403	3,168	11,194 - 23,611
1990	0.069	0.0082	0.053 - 0.085	38,054	6,311	25,685 - 50,423
1991	0.063	0.0078	0.048 - 0.079	4,092	1,562	1,032 - 7,153
1992	0.138	0.0179	0.103 - 0.173	3,679	1.288	1,154 - 6,203
1993	0.082	0.0143	0.054 - 0.110	11,982	2,830	6,436 - 17,528
1994	0.057	0.0115	0.035 - 0.079	803	1.381	-1,904 - 3,510
1995	0.077	0.0182	0.042 - 0.116	3,536	1,552	493 - 6,578
1996	0.049	0.0147	0.020 - 0.077	5,421	2,051	1,401 - 9,441
1997	0.070	0.0209	0.029 - 0.118	3,280	1,440	457 - 6,102
1998	0.102	0.0326	0.038 - 0.189	7,600	2,997	1,725 - 13,475
1999	0.058	0.0197	0.028 - 0.144	4,623	2,231	249 - 8,997
2000	0.068	0.0226	0.024 - 0.110	1,208	1,071	-890 - 3,307
2001	0.087	0.0292	0.030 - 0.145			
Mean	0.070	0.0039	0.062 - 0.078	14,519	600	13,344 - 15,694

[a] Annual estimates of these parameters may vary from those reported in DNC (2002) because of mark and recapture data added from the 2002 adult winter flounder population survey (Table 5).

measure of sampling error except at very high sampling intensities (Manly 1971; Roff 1973; Pollock et al. 1990). Abundance estimates for the most recent years are subject to change as additional recaptures are made during future surveys. For example, because of the recaptures made in 2002, the initial abundance estimate given in DNC (2002) of 9,300 fish for 2000 decreased by 873 fish to 8,427. However, the 1997 estimate increased by 46% (+3,249) and increases also occurred for the 1994-98 abundance estimates relative to results presented last year.

Sampling intensity (p), or the probability that a fish will be captured, was estimated as 0.087 for 2002, which was greater than the series average of 0.070 for this parameter. Sampling intensities of at least 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 (0.05-0.09), regardless of population size or number of fish marked. Estimates of p only approximated or exceeded 0.10 in 1992 (0.138) and 1998 (0.102), so N has likely been underestimated. Loss of information because brands were missed or from mortality of fish handled also requires increased sampling effort. Other sampling errors, model assumptions, and biases inherent in the Jolly model that could have affected these estimates were discussed in NUSCO (1989) and Pollock et al. (1990).

Although the Jolly estimates are subject to some error, annual Δ-mean CPUE and Jolly abundance estimates were significantly correlated (Spearman's rank-order correlation; r = 0.937; p = 0.0001; n = 18; Fig. 8). Thus, based on a Δ-mean CPUE of 2.2 for winter flounder 20 cm and larger in 2002, absolute abundance of winter flounder this year likely was from 8 to 10 thousand fish. By extrapolation from the CPUE in 1981, total abundance of winter flounder larger than 20 cm in that year of extreme abundance may have ranged from 150 to 200 thousand.

Estimates of both Φ (survival) and B (recruitment) are considered to be much less reliable than those of abundance when using the Jolly model (Manly 1971; Cormack 1972; Bishop and Sheppard 1973; Buckland 1980, 1982; Arnason and Mills 1981; Hightower and Gilbert 1984; NUSCO 1986b) and have not been used as quantitative estimators of survival or recruit-
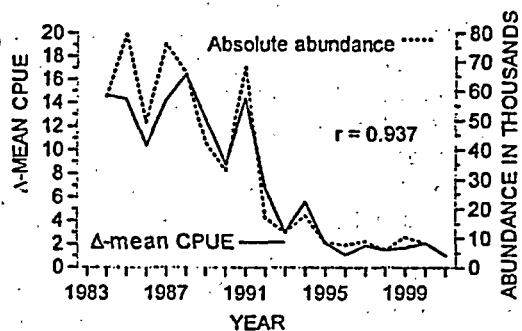
Fig. 8. Comparison between estimates of absolute abundance of winter flounder larger than 20 cm in the Niantic River during the spawning season and the corresponding Δ-mean CPUE from 1984 through 2001.

ment in the Niantic River winter flounder studies. Survival estimates varied considerably from year to year (0.189-0.908; Table 6). The initial value of 0.346 for 2000 was less than the long-term average of 0.458 and survival estimates for 1997-99 were also lower than average, reflecting low rates of recapture in those years. Although not a reliable quantitative measure, these values still likely reflected continuing high rates of mortality to adult winter flounder. Except for 1993 (11,982 fish), estimated recruitment values were relatively low following 1990. Estimates of B are also relatively imprecise, having 95% CIs with wide ranges. As for other parameter estimates based on only 1 year of recapture information, estimates of Φ and B based on only 1 or 2 years of recapture data may change considerably with the addition of data from subsequent annual surveys.

Spawning Stock Size and Egg Production

A second index of relative abundance was based on the size distribution of female fish from adult spawning survey catches standardized by weekly and yearly effort (i.e., number of tows). Catches were adjusted by effort to insure that each size group of fish was given equal weight within each week of sampling, 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 Niantic River female winter flounder spawning stock was the basis for the calculation of annual recruitment and egg production.

The annual standardized catch of spawning females and the production of eggs by this stock were determined from available data on abundance, sex

ratios, sexual maturity, and size-frequencies. The sex ratio of winter flounder larger than 20 cm during the 2002 spawning season in the Niantic River was 1.88 females to each male (Table 7; Fig. 9). Although like most years, when considerably more females were taken than males, this was the lowest ratio observed since 1996. In general, more highly skewed (≥ 1.70) sex ratios have been found since 1994 as winter flounder abundance decreased. Over all years no significant trend in sex ratio was found. However, since 1986 a highly significant increasing trend in sex ratio has occurred (Mann-Kendall test, $Z = 3.30$; n = 17; p < 0.001). The reasons for the preponderance in females are unknown. Mature females are larger than males, so greater exploitation of females might be expected, which would tend to disproportionately decrease their numbers in comparison to males. However, this seems to be contrary to the above findings and may indicate susceptibilities that differ by gender for fishing (e.g., availability), natural mortality (e.g., predation, disease, or longevity), or their availability to trawl sampling during the winter flounder survey (e.g., males remaining to spawn in

Fig. 9. Female to male sex ratio of Niantic River winter flounder larger than 20 cm from 1976 through 2002.

shallow areas less frequently sampled). Stoner et al. (1999) reported ratios of 1.50 to 2.33 in favor of females for the Navesink River, New Jersey, and Saila (1962a, 1962b) and Howe and Coates (1975) for other winter flounder populations in southern New England. Witherell and Burnett (1993) found greater proportions of female winter flounder in Massachusetts waters, particularly in older age-classes. Based on evidence of greater senescent mortality reported for males by Burton and Idler (1984), they believed that males likely have a higher natural mortality rate, which would tend to skew sex ratios in favor of females.

The gender of winter flounder larger than 20 cm during the Niantic River surveys was ascertained by external examination of collected specimens. Correct classification to gender was readily obvious in fish that were ripe or gravid, as it was for recently spawned females having distinctly thin, flaccid ovaries. However, fish not in spawning condition were sexed by palpation of their scales for ctenii according to the method of Smigielski (1975). Bejda and Phelan (1998) noted that the latter methodology may be subject to error, but, as discussed in NUSCO (1999), a systematic misclassification of fish by gender was unlikely during the spawning survey. Although errors in sexing fish can and probably do occur during external examinations, the skewed sex ratio found for Niantic River winter flounder during the spawning season appears to be real and likely occurs for the reasons given above.

Stoner et al. (1999) observed that female winter flounder held in a large laboratory aquarium spawned on average 40 times during the season. Most individuals spawned over a duration of about 1 week. Similarly, in laboratory spawning of winter flounder by NUSCO in the early 1980s, some females were observed to spawn more than once over a period of several days. However, throughout the many years of study in the Niantic River only a few females were

TABLE 7. Female to male sex ratios of winter flounder taken during the spawning season in the Niantic River from 1977 through 2002.

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
1997	2.40	2.40
1998	2.18	2.18
1999	2.36	2.36
2000	2.15	2.15
2001	2.39	2.39
2002	1.88	1.88
Geometric mean	1.57	1.57

observed in a condition intermediate between gravid and spent. These observations suggest that perhaps some additional spawning by individuals may occur or that a spawning event, which takes place at night (Stoner et al. 1999), may have been interrupted. However, most females examined appeared to have been either fully gravid, completely spent after recent spawning, or in a post-spawning (i.e., feeding) state. Because Burton and Idler (1984) reported that winter flounder only produce one batch of mature eggs each year, fecundity estimates are likely unaffected even if eggs are released during more than one spawning event, unlike a serially spawning species, such as the tautog (*Tautoga onitis*), which continuously produces batches of eggs over an extended period.

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-cm size-classes) made from 1981 through the present. About half of all observed females were observed to be mature at 26 cm (NUSCO 1988b), which is comparable to L_{50} estimates of size-at-maturity of 25.8 cm reported for LIS by Johnson et al. (1998) and 28.3 and 27.6 cm for Massachusetts waters by Witherell and Burnett (1993) and O'Brien et al. (1993), respectively. Pooled estimates were adjusted to give continuously increasing fractions of mature fish through 34 cm. All females 34 cm or larger were considered to be mature (Fig. 10). 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):

$$\text{fecundity} = 0.0824(\text{length in cm})^{4.506}. \quad (1)$$
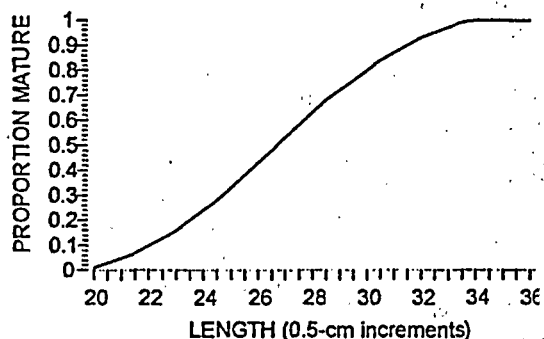
Fig. 10. Empirically derived proportion of mature Niantic River female winter flounder by length used in the calculation of stock and recruitment. All females smaller than 20 cm were immature and larger than 34 cm were mature.

This relationship was used with the annual standardized catch of mature females and their length composition to calculate egg production. Although some density-dependent regulatory mechanisms have been proposed for adult flatfishes, size-specific fecundity appears to be stable over a wide range of abundances and may decrease only at the very highest densities (Rijnsdorp 1994). Thus, no adjustments were considered for the Niantic River winter flounder length-fecundity relationship. Annual mean fecundity was determined by dividing the sum of all individual egg production estimates by the standardized catch of females spawning per year.

Since an excellent correlation (r = 0.937) exists between Δ-mean trawl CPUE and Jolly abundance estimates, the annual standardized catches of all fish ≥20 cm for 1984-2000 (the 2001 Jolly estimate was considered to be preliminary and not used) were divided by the corresponding annual total abundance estimate from the Jolly model. Annual ratios ranged between 2.2 and 5.4% and had a geometric mean of 3.3%. Thus, the relative numbers of females and eggs produced each year, as determined from the standardized catches, were conservatively assumed to represent about 3.3% of the absolute values and a multiplier of 30.3 was used to scale up standardized catch indices to absolute numbers of female winter flounder spawning in the Niantic River. 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 of later years had estimates of absolute abundance been available for the earlier period. Annual estimates of the number of spawning females were also used to determine egg production and for the derivation of a relationship between stock and recruitment for Niantic River winter flounder. As this scaling factor differed from that used in DNC (2002) and previous reports, revised estimates of female stock size, recruitment, and egg production resulted.

Estimates of female stock size ranged from a maximum of about 75 thousand fish in 1982 to a minimum of 2 thousand in 2001 (Table 8). Female abundance in 2002 was nearly twice the 2001 estimate. Despite the preponderance of larger mature females in the Niantic River, the number of spawning females has been very low since 1995 because of decreased abundance of winter flounder overall. Mature females usually comprised approximately one-third to one-half of all winter flounder 20 cm and larger. Highest fractions of mature females were most often found in recent years, including 1995 (63%) and 1997 (60%). The high fractions were related to highly skewed sex ratios and larger fish present during these years. Including only females

·TABLE 8. Relative and absolute standardized catch of female winter flounder spawners and corresponding egg production in the Niantic River from 1977 through 2002.

Survey year	Relative index of spawning females[a]	% mature females (of all fish)[b]	% mature females (of all females)[c]	Average fecundity[d]	Relative index of total egg production[e]	Total female stock size[f]	Total egg production (X 10⁹)[f]
1977	1,069	36	61	450,470	481.4	31,526	14.586
1978	1,644	51	76	503,750	828.3	48,511	25.099
1979	1,197	37	64	472,405	565.7	35,328	17.141
1980	1,004	34	51	466,919	468.9	29,626	14.208
1981	2,411	44	70	519,461	1,252.5	71,140	37.956
1982	2,551	49	77	580,082	1,479.8	75,265	44.843
1983	1,705	47	78	577,885	985.5	50,313	29.863
1984	822	40	77	577,774	474.9	24,253	14.392
1985	846	43	75	608,374	514.9	24,969	15.602
1986	604	42	86	666,132	402.5	17,826	12.196
1987	780	39	85	622,995	485.9	23,010	14.724
1988	1,172	53	89	676,469	792.8	34,579	24.025
1989	891	52	90	726,652	647.1	26,274	19.609
1990	498	41	73	631,150	314.4	14,695	9.526
1991	963	47	85	600,832	578.7	28,418	17.537
1992	485	53	94	730,945	354.2	14,296	10.733
1993	247	54	93	815,957	201.6	7,288	6.108
1994	458	56	87	648,741	297.3	13,523	9.010
1995	192	63	87	772,187	148.1	5,659	4.488
1996	88	52	82	843,825	74.3	2,598	2.251
1997	166	60	83	796,337	131.8	4,884	3.995
1998	118	55	79	755,317	89.1	3,479	2.699
1999	107	45	65	698,084	75.0	3,168	2.271
2000	167	55	80	737,566	123.0	4,919	3.727
2001	68	49	69	823,565	56.1	2,009	1.700
2002	129	51	79	804,157	103.5	3,797	3.136

[a] Based on proportion of the relative annual standardized catches of winter flounder that were mature females.
[b] As a proportion of all winter flounder 20 cm or larger.
[c] As a proportion of all female 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.
[f] Calculated on the assumption that the relative annual standardized catches were approximately 3.3% of absolute values.

larger than 20 cm, mature fish made up from 51 to 94% of annual female totals. Higher values indicated that, in many years, smaller and younger non-spawning females were not present in the Niantic River or that incoming year-classes were very weak. For some years, including 1999 (65%) and 2001 (69%), smaller fractions of mature females reflected the presence of larger numbers of immature winter flounder, individuals that were expected to recruit into the spawning population in subsequent years, which was reflected by an increase in spawning females in the year that followed.

Annual egg production estimates were determined using annual catch, age, maturity, and length information with the length-fecundity relationship for Niantic River winter flounder. Yearly egg production during 1977-2002 ranged from about 1.7 to 44.8 billion (Table 8). Differences in percent maturity resulting from variable annual length-frequency distributions affected mean fecundity, which was low

during the late 1970s when smaller fish were more abundant. However, mean fecundity has been relatively high since 1992 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 annual fecundity estimates. Estimates were also relatively high in 1988, 1989, and 1991 as older and larger females dominated moderately-sized reproductive stocks. Total egg production decreased to relatively low values in recent years because of very low winter flounder abundance. Estimated long-term mean fecundity of Niantic River female winter flounder, determined by dividing the total number of eggs by the total number of mature females since 1977, was 585,167. Each total annual egg production estimate was divided by the present value of mean fecundity to produce an estimate of scaled parental stock size used in a Stock-Recruitment Relationship (SRR), which is discussed later in this report.

Comparisons Among Regional Abundance Indices

Comparisons were made among the abundance of Niantic River winter flounder and regional abundance indices, including the spring CPUE of winter flounder from the DEP LIS-wide stratified random trawl survey (Gottschall et al. 2002; V. Crecco, CT DEP, Old Lyme, CT, pers. comm.), the mean CPUE from a spring and fall stratified random trawl survey conducted by the Rhode Island Department of Environmental Management, Division of Fish and Wildlife (RIFW) in Narragansett Bay, RI and nearby waters (Lynch 2000; T. Lynch. RIFW, pers. comm.), an annual CPUE developed from weekly trawl sampling conducted by the University of Rhode Island (URI) near Fox Island in upper Narragansett Bay (sampling program described in Jeffries and Johnson 1974; Jeffries and Tereceiro 1985; Jeffries et al. 1989; D. Taylor, University of Rhode Island Graduate School of Oceanography, Narragansett, RI, pers. comm.), and a stratified random trawl survey conducted during spring by the Massachusetts Division of Marine Fisheries (MDMF) in state waters extending from the tip of Cape Cod to the Rhode Island border (Howe et al. 1999; T. Currier, MDMF, Pocasset, MA, pers. comm.).

Most regional abundance indices of winter flounder were strongly correlated (Spearman's rank-order correlation) with one another (Table 9). These correlations were positive, indicating no inverse trends in abundance among areas. The Δ-mean CPUE of adult winter flounder spawning in the

Niantic River was significantly correlated with two Rhode Island winter flounder CPUE indices, the RIFW Narragansett Bay trawl index (r = 0.8009) and the URI Fox Island series (0.8010), and with the MDMF Southern Massachusetts trawl index having catches expressed as biomass per tow (0.6898). A weaker, but still significant correlation was found with the DEP LIS-wide CPUE (0.5667). This result may be due to a shorter time-series for comparison and because the DEP trawl survey had limited sampling in eastern LIS (Gottschall et al. 2000).

Comparisons of normalized indices of abundance for these regional data sets showed good correspondence among them (Figs. 11 and 12). The two Rhode Island CPUE indices each had a peak in 1979, 2-3 years before the peak in abundance was observed in the Niantic River, perhaps because the Rhode Island catches had a larger component of younger fish than the mostly older evident by the mid-1980s in each of these winter flounder abundance indices. Gibson (1998) noted that winter flounder abundance in Narragansett Bay began declining in the late 1970s and remained low for more than a decade, not rebounding to levels seen previously. Although the Rhode Island catches increased from the early 1990s to a small peak in 1995, perhaps reflecting catch of fish from the relatively strong 1992 year-class, CPUE subsequently decreased in years following to present all-time lows. In comparison, a larger increase in abundance was seen for the DEP LIS CPUE through 1996, but this index has also decreased subsequently to the lowest values of this series in 1999, 2000, and 2002. In contrast, the Niantic River population size

TABLE 9. Matrix of Spearman's rank-order correlations among various regional winter flounder abundance indices.

Abundance Index	RIFW - Narragansett Bay	URI - Fox Island	CT DEP - LIS	MDMF - Southern Mass.
Niantic River adult winter flounder survey (≥15 cm) Δ-mean CPUE	0.8009[a] 0.0001 ** 24	0.8010 0.0001 ** 27	0.5667 0.0114 * 19	0.6898 0.0001 ** 25
RIFW Narragansett Bay trawl survey spring and fall mean CPUE		0.8965 0.0001 ** 24	0.5158 0.0238 * 19	0.6971 0.0002 ** 224
URI Fox Island annual geometric mean CPUE			0.5404 0.0169 * 19	0.8229 0.0001 ** 25
CT DEP LIS spring geometric mean CPUE				0.2953 0.2197 NS 19

[a] The three statistics shown for each correlation matrix element are:
  correlation coefficient (r),
  probability of a larger r (NS - not significant [p > 0.05], * - significant at p ≤ 0.05, ** - significant at p ≤ 0.01), and
  number of annual observations (sample size).
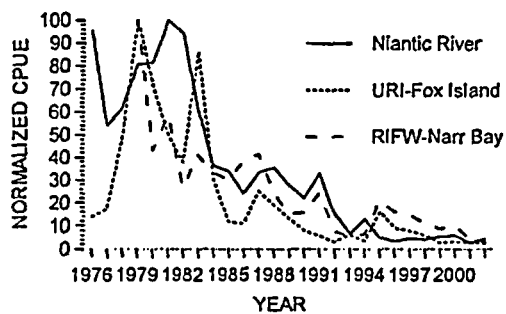
Fig. 11. Comparison of regional winter flounder abundances from 1976 through 2002. Included are the Δ-mean CPUE of fish larger than 15 cm taken in the Niantic River during the spawning season, the mean CPUE from Rhode Island Fish and Wildlife (RIFW) spring and fall trawl surveys in Narragansett Bay, and the annual geometric mean CPUE of winter flounder taken at the University of Rhode Island (URI) Fox Island trawl station in Narragansett Bay. Dividing all values by the corresponding largest estimate and multiplying by 100 normalized each CPUE series.

Fig. 12. Comparison of regional winter flounder abundances from 1984 through 2002. Included are the Δ-mean CPUE of fish larger than 15 cm taken in the Niantic River during the spawning season, the annual geometric mean CPUE of winter flounder taken at the University of Rhode Island (URI) Fox Island trawl station in Narragansett Bay, and the annual geometric mean CPUE of winter flounder taken during April-June by Connecticut Department of Environmental Protection Marine Fisheries in Long Island Sound (DEP LIS). Dividing all values by the corresponding largest estimate and multiplying by 100 normalized each CPUE series.

remains small, although except for 2001 with modest increasing abundance since 1996 (Fig. 12).

Coherence among abundance indices of winter flounder within a relatively small geographical region should be expected. Fox et al. (2000) reported synchrony in the recruitment of European plaice throughout the entire waters of the United Kingdom, although abundances in adjacent areas tended to be most similar. According to NEFSC (1999), which is a recent stock assessment of winter flounder on the northeastern coast of the United States, each winter flounder abundance survey discussed above sampled a distinct geographical area and was likely providing measures of different components of the same aggregated stock. Further, older winter flounder dominated catches in the Niantic River surveys, but younger fish were prevalent in the DEP LIS sampling. Based on catch-at-age data and length-frequency information presented in NEFSC (1999) and Lynch (2000), respectively, younger winter flounder made up a large component of the catches in these surveys as well. Thus, direct comparisons of these indices may need some qualification, even though temporal trends appeared to have some similarities. Also noted in NEFSC (1999) was that regional abundance surveys tended to present a continuum of optimistic to pessimistic trends in abundance and illustrated variable tracking of year-class strength when they were compared among one another. This was attributed to possible differences in the availability of winter flounder to sampling; this was also noted for

the Niantic River studies. A further indication was given by the Atlantic States Marine Fisheries Commission (ASMFC) Winter Flounder Management Board, who recently reported that stock biomass of the Southern New England and Mid-Atlantic winter flounder remained at below target levels, even though F was believed to be at 0.3-0.4 (ASMFC 2001). Some state surveys indicated that stock biomass has actually decreased. With improvement in stock status not observed in some member state waters, some ASMFC members also questioned whether winter flounder distribution patterns have changed over time. These findings were re-iterated in NEFSC (2002), which stated that fishing mortality for the Southern New England-Middle Atlantic stock of winter flounder increased since 1994 and needed to be reduced by 37% to achieve F_{MSY}. This stock was considered overfished. Further, NEFSC (2002) noted that the Virtual Population Analysis exhibited a severe retrospective pattern of underestimation of fishing mortality and an overestimation of spawning stock biomass during the late 1990s. This may have contributed to some of the uncertainty observed by the ASMFC regarding winter flounder abundance.

Finally, in a recent study on long-term recruitment trends of New England groundfish, Brodziak et al. (2001) concluded that nine of eleven stocks examined had a significantly declining trend in spawning stock biomass. Among these stocks, the 1963-96 time-series of Southern New England winter flounder showed significant declining trends for both spawning stock biomass and recruits-per-spawner. This result adds to the mounting empirical evidence supporting the notion that the decline in winter flounder abundance during the past 20 years or so has taken place over a wide regional scale and well beyond any possible influence of MPS.

Larval Winter Flounder

Field Sampling, Laboratory Processing and Data Analyses

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. 13). To collect larvae, a 60-cm bongo plankton sampler weighted with a 22.7-kg oceanographic

depressor and fitted with 3.3-m long nets with mesh size of 202 μm was deployed during February and March. Net size was changed to 333 μm in April to increase catchability of larger larvae. Volume of water filtered was determined from a single General Oceanics 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 towline necessary to sample the mid-water and bottom strata was determined by water depth and towline angle measured with an inclinometer. Nets were towed for 6 minutes and each net typically filtered about 120 m^3 of water. Only one of the duplicate samples was retained for laboratory processing, typically the bongo that sampled the greatest volume. When present, jellyfish medusae at the three 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
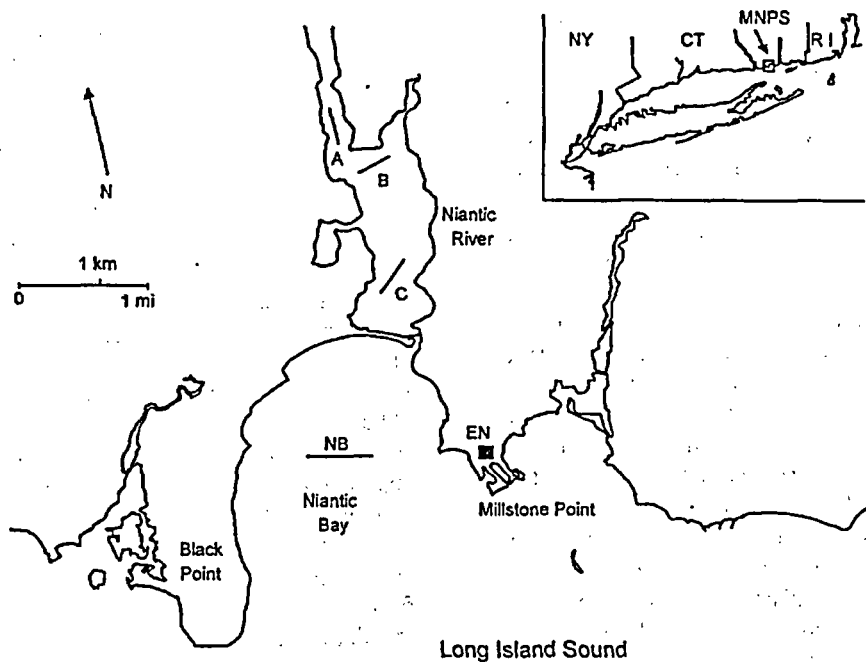
Fig. 13. Location of stations (denoted by letters) sampled for larval winter flounder during 2002.

increase data collection efficiency (NUSCO 1987). Larval sampling at the Niantic River stations usually begins in early to mid-February. In 2002, sampling in the river began on February 4 at all stations and sampling continued at each station until March 20, after which station A was dropped due to lack of larvae in samples. Sampling ceased at Station B on May 28 due to lack of larvae in samples. Stations C and NB were last sampled on June 11. 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, collections were taken on 2 days each 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 through the remainder of the larval season. Water temperature and salinity measurements (surface, midwater, and bottom) were recorded at each sampling station using a YSI Model 30 Salinity/Temperature/Conductivity meter. All ichthyoplankton samples, including the entrainment collections described below, were preserved with 10% formalin.

Winter flounder larvae entrained through the MPS cooling-water system have been sampled at the plant discharges (station EN, Fig. 13) since 1976. These collections provide information not only for the estimation of entrainment impact, but also on the abundance and temporal distribution of winter flounder larvae in Niantic Bay. In the earliest years of study, collections usually alternated between the discharges of Units 1 and 2, depending upon plant operation and the resulting water flow from the condenser cooling-water pumps. In more recent years, collections were also taken at the Unit 3 discharge. Collections were eliminated at the Unit 1 discharge in 1995 when this unit was retired from operation. Larvae were collected with a 1.0 x 3.6-m conical plankton net of 333-μm mesh deployed from a gantry system into the discharge water from each unit. Four General Oceanic (GO) Model 2030 flow-meters 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. Under normal operations, the net is usually deployed for 3 to 4 minutes (filtering about 200 m^3), with variation in sampling time dependent upon the number of

circulating water pumps in operation and tidal stage. Samples of longer duration (6-10 min) were taken when less than the normal complement of circulating water pumps was in operation. Sampling frequencies and volume filtered have varied since 1976 (NUSCO 1987, 1994a). In recent years, sampling was conducted during both day and night once per week in February and 3 days and nights per week during March through June. Entrainment samples were processed similarly to field collections of larvae, which are described below.

During laboratory processing and in accordance with laboratory sorting protocols, preserved ichthyoplankton samples were split at minimum to one-half volume and to lesser volumes if larvae were relatively abundant. 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 begun, but left eye not 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 caudal fin base.

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 MPS.

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:

$$C_t = A \times \exp(-\exp[-k \times \{t - p\}]) \qquad (2)$$

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

The time of peak abundance was estimated by the parameter p. The origin of the time scale was set to February 15, which is the approximate date when winter flounder larvae first appear in the Niantic River. Least-squares estimates, standard errors, and asymptotic 95% confidence intervals (CI) for these parameters were obtained by fitting the above equation to the cumulative abundance data using nonlinear regression methods (SAS Institute Inc. 1990d). Cumulative data were obtained as the running sums of the weekly geometric means of the abundance data. The A 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:

$$d_t = A' \times k \times \exp(-\exp[-k \times \{t - p\}] - k \times [t - p]) \qquad (3)$$

where d_t = density at time t and all the other parameters are as described for Equation 2, except for A', which was re-scaled by a factor of 7 (i.e., $A' = 7A$) 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 Ricker (1975) nonlinear relationship:

$$L = a \times E \times \exp(-b \times E) \qquad (4)$$

where L = annual larval abundance of larvae 7-mm and larger at EN as estimated by A (see Equation 2)
  E = annual estimate of egg production in the Niantic River
  a = slope at the origin of the larval recruitment curve
  b = instantaneous mortality dependent upon annual egg abundance

Density-dependent mortality may be assumed when the slope (b) is significantly different from zero. 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 and Distribution

The A 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 A parameter estimates, larval abundance in aggregate during 2002 in the Niantic River (stations A, B, and C combined) was the 12th highest observed during the 20-year period (Table 10). In contrast, abundance in Niantic Bay (stations EN and NB combined) was the third lowest of the series, and was roughly one-fourth the abundance found in 2001. In general, annual abundances in the bay have varied less than in the river. In most years the abundance of larval winter flounder in the river was two to six times greater than the bay. However, in 1988 and 1989, abundances in the river were more than ten times greater than in the bay, but in 1983 and 1999, river abundance was less than two times as much as in the bay. The reason that larval abundance is generally greater in the river than the bay is probably related to preferred winter flounder spawning areas, as discussed below in the comparison of spatial distribution of developmental stages. Biological and physical processes occurring during larval development are variable as no significant relationship was found between the indices of annual abundances (excluding 1983) for the two areas (Spearman's rank-order correlation; r = 0.138; p =

0.561; n = 20). Low abundance in the river during 1983 was attributed, in part, to undersampling because smaller larvae were likely extruded through the 0.333-μm mesh net then used throughout the year (NUSCO 1987).

Annual spatial abundances of the first four larval developmental stages were based on cumulative weekly geometric means at each of the stations sampled (Figs. 14 and 15). The abundance distribution of Stage 5 individuals (i.e., newly transformed to juveniles) was not examined because so few of them were collected by ichthyoplankton sampling gear. Cumulative density data (the annual sum of the weekly geometric means) were used to compare abundances as a surrogate for the A 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 A parameters were highly correlated (Spearman's rank-order correlation; r = 0.999; p < 0.001; n = 120) in a previous comparison (NUSCO 1989), indicating that the cumulative means could be used as another index of larval abundance.

Stage 1 abundances by station during 2002 were about average in both the Niantic River (Fig. 14) and Niantic Bay (Fig. 15). A comparison of annual Stage

TABLE 10. Index of annual abundances and 95% confidence intervals based on the A parameter from the Gompertz function for larval winter flounder taken in Niantic River and Bay from 1983 through 2002.

Year	Niantic River		Niantic Bay	
1983	1,863	(1,798 - 1,929)[a]	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)
1997	23,402	(23,154 - 23,649)	4,146	(4,025 - 4,267)
1998	11,944	(11,666 - 12,223)	2,138	(2,018 - 2,257)
1999	3,892	(3,687 - 4,096)	2,643	(2,513 - 2,772)
2000	1,479	(1,348 - 1,611)	4,114	(3,579 - 4,650)
2001	2,968	(2,901 - 3,035)	5,220	(3,446 - 6,993)
2002	5,144	(5,102 - 5,187)	1,273	(1,235 - 1,311)

[a] Stage 1 larvae undersampled in 1983 because a 202-μm mesh net was not used during the early portion of the larval season.
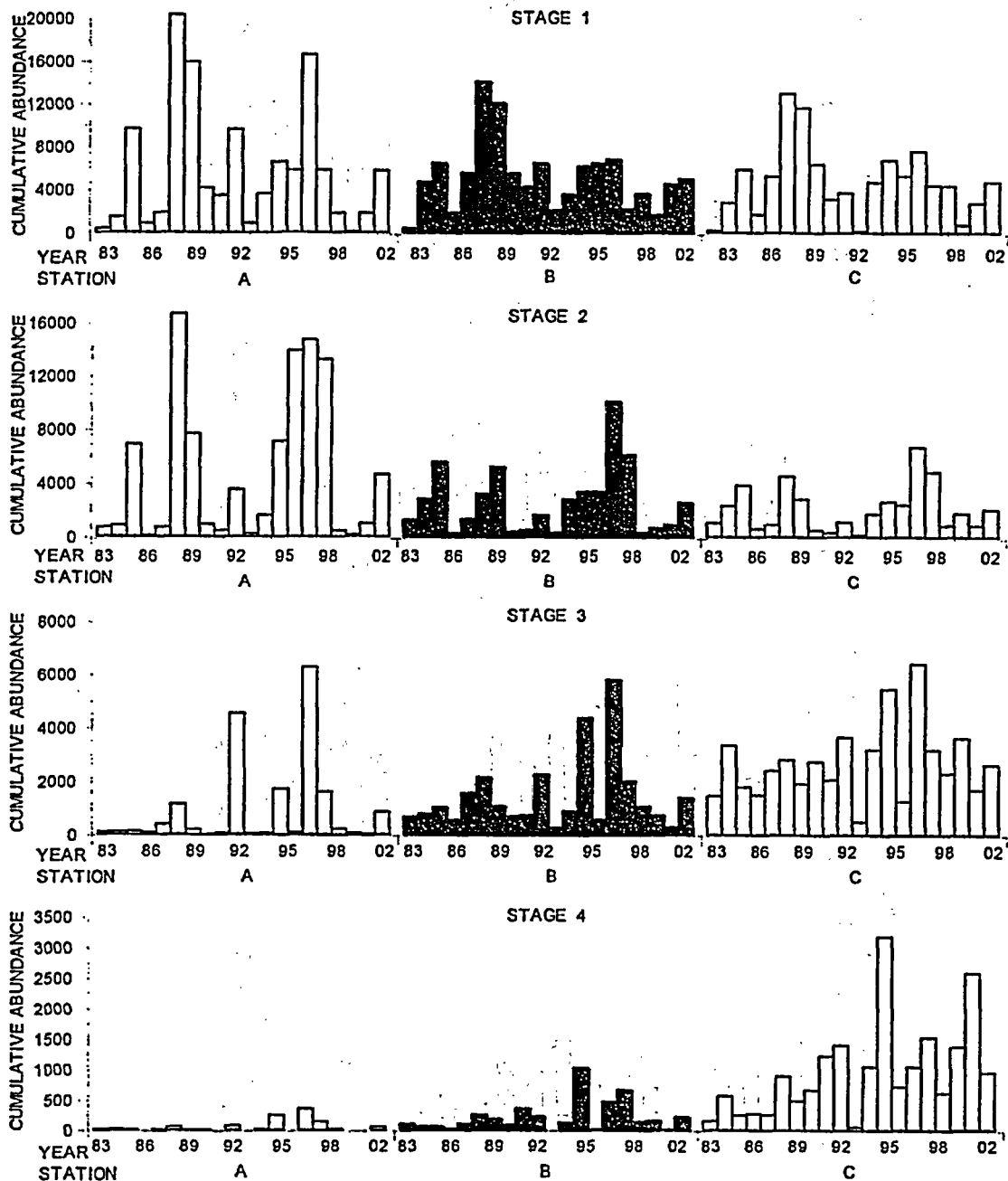
Fig. 14. Index of cumulative density by developmental stage of larval winter flounder at Niantic River stations A, B, and C from 1983 through 2002. (Note that the vertical scales differ among the graphs).

1 abundance among years at the three river stations showed a similar relative ranking at the three stations, with 1988, 1989, and 1997 ranked the highest and 1983, 1986, 1993, and 2000 the lowest. Except for a greater abundance at station A in some years, annual abundances at the three river stations have been similar in most years; significant (p ≤ 0.05) positive correlations were found among Stage 1 annual abundances for all river stations (Table 11). This indicated a somewhat homogeneous distribution of Stage 1 larvae throughout the river. Because winter flounder eggs are demersal and adhesive and the duration of Stage 1 is short (about 10 days), the homogenous distribution suggested either that spawning was not restricted to a specific area of the river or that the river is well-mixed.
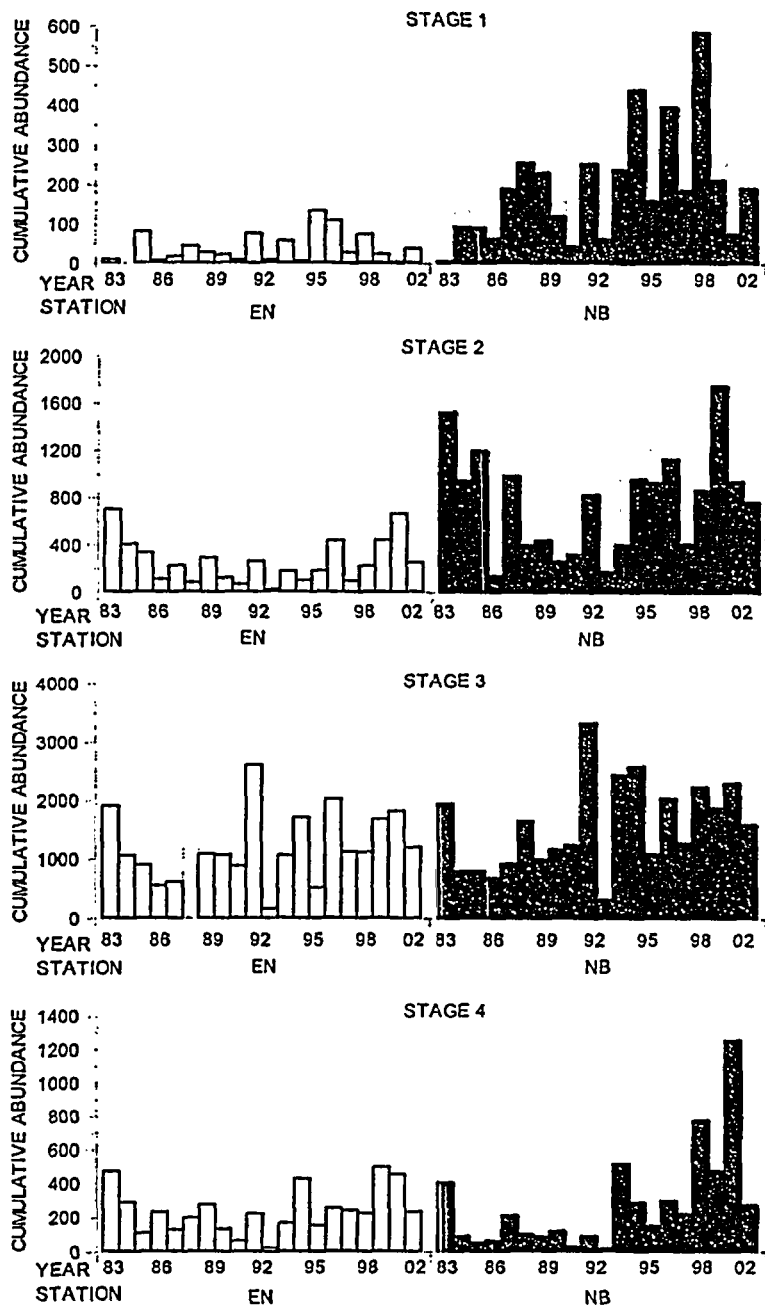
Fig. 15. Index of cumulative density by developmental stage of larval winter flounder at the Niantic Bay stations EN and NB from 1983 through 2002. (Note that the vertical scales differ among the graphs).

On average, abundance of Stage 1 larvae was about 50 times less at the two Niantic Bay stations (Fig. 15) than in the river (Fig. 14), indicating that little, if any, spawning occurred in the bay. Abundance at station NB was consistently greater than at EN, possibly because NB was located closer to the river mouth, the likely source of Stage 1 larvae, and because undersampling occurred at EN as a result of extrusion of Stage 1 larvae through the 333-μm mesh net used

at the MPS discharges. At NB, ranks of annual abundance indices were similar to those of the river stations suggesting that most Stage 1 larvae collected in the bay probably originated from the Niantic River. However, even though abundance at station A was correlated with EN and NB, there were less clear associations between B and C and the two Niantic Bay sites (Table 11).

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TABLE 11. Matrix of Spearman's rank-order correlations among stations of the indices of annual cumulative abundance of each developmental stage of larval winter flounder from 1983 through 2002.

Stage	Station	B	C	EN	NB
1	A	0.8436[a] 0.0001 **	0.8587 0.0001 **	0.4887 0.0288 *	0.4993 0.0250 *
	B		0.8421 0.0001 **	0.4857 0.0299 *	0.4782 0.0330 *
	C			0.4902 0.0282 *	0.6015 0.0050 **
	EN				0.4962 0.0261 *
2	A	0.8827 0.0001 **	0.7910 0.0001 **	-0.0481 0.8403 NS	0.1023 0.6679 NS
	B		0.9233 0.0001 **	0.1669 0.4818 NS	0.3699 0.1084 NS
	C			0.2165 0.3591 NS	0.4015 0.0793 NS
	EN				0.7384 0.0002 **
3	A	0.8947 0.0001 **	0.5939 0.0058 *	0.4812 0.0317 *	0.2421 0.3038 NS
	B		0.7970 0.0001 **	0.5203 0.0187 *	0.3805 0.0980 NS
	C			0.5955 0.0056 *	0.4587 0.0042 **
	EN				0.7368 0.0002 **
4	A	0.5608 0.0101 *	0.2443 0.2993 NS	0.3272 0.1590 NS	0.2382 0.3120 NS
	B		0.5774 0.0077 *	0.1504 0.5269 NS	0.2120 0.3695 NS
	C			0.3173 0.1728 NS	0.4331 0.0565 NS
	EN				0.5459 0.0128 *

[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 $p \leq 0.01$).
    The number of observations (sample size) was 20 for each paired comparison.

Annual abundance of newly hatched winter flounder larvae should be related to adult spawner egg production and the fraction of eggs that hatch. However, from 1995 through 1999 and in 2001-02, abundances of Stage 1 larvae at the three river stations were greater than expected from the low annual egg production estimates determined for the Niantic River spawning stock (Fig. 16). The index of Stage 1 larval abundance was the A parameter from the Gompertz function (Eq. 2) for the Niantic River (stations A, B, and C combined) and the previously discussed egg production estimates were given on
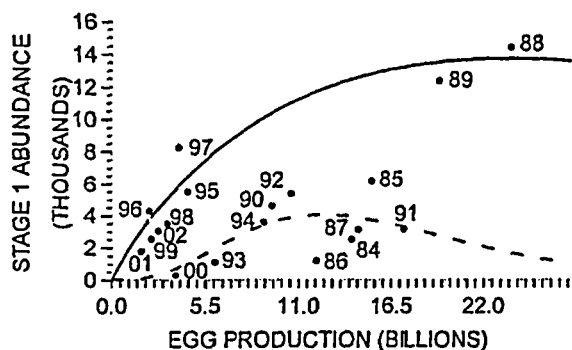
Fig. 16. Comparison between the index of annual Stage 1 abundance in the Niantic River (A parameter of the Gompertz function) and estimated annual egg production from 1984 through 2002. Ricker stock-recruitment functions were fitted to data from 1984-87, 1990-94, and 2000 (dashed line): Stage 1 = $0.0217(Egg^{3.4340} \times e^{-0.2760 \cdot Egg})$; p = 0.0031, and 1988-89, 1995-99, and 2001-02 (solid line): Stage 1 = $1.819(Egg^{0.9015} \times e^{-0.0349 \cdot Egg})$; p = 0.0003.

Table 9. To examine egg survival, a relative annual survival rate index was determined by computing the ratio of the Stage 1 abundance index (in thousands) to annual egg production estimates (in billions), followed by grouping years as either 1984-87, 1990-94 and 2000 or 1988-89, 1995-99, and 2001-02. The mean ratio (relative survival rate index) for the first yearly group was 0.28 (SE = 0.06) and for the latter group was 1.27 (SE = 0.18), indicating that average egg survival appeared to be greater by more than five times during most recent years in comparison to the mostly earlier period. In addition, there was a significant difference between ratios for the two yearly groups (p = 0.001; Wilcoxon two-sample test; Sokal and Rohlf 1969). These relationships were explored further using nonlinear regression techniques fitting the data to the Ricker stock-recruitment equation (Ricker 1975). Statistically significant relationships were found for both groups of years, demonstrating an apparently higher egg survival in 1988-89, 1995-99, and 2001-02 (Fig. 16). The stock-recruitment relationship in these years approached a theoretical asymptote at high levels of egg production, whereas the model for the years 1984-87, 1990-94, and 2000 became depressed at high egg production levels, indicating stronger effects of compensatory processes that increased egg mortality and limited abundance of Stage 1 larvae. The biological significance of these relationships has yet to be determined and warrants closer examination of mechanisms regulating egg survival, such as predation and egg quality.

Stage 2 abundances at the three river stations in 2002 were about average (Fig. 14). On average (1983-2002), the abundance of Stage 2 larvae in the river was about 60% of Stage 1, but ratios for each year were variable. For example, in 1999 Stage 2 abundance was less than 20% of that of Stage 1, in 2000 abundances were similar, and in 2001 Stage 2 abundance was about 30% of that for Stage 1. In general, annual ranks of Stage 2 abundance at the three river stations have been similar to those of Stage 1, which implied a similar annual rate of larval loss (mortality and flushing) during larval development from Stage 1 to 2. In most years, including 2001, Stage 2 larvae occurred predominantly in the river, but were more prevalent in the bay compared to Stage 1. At the two bay stations, Stage 2 abundances in 2002 were also about average, with abundances at Station EN being about half those observed in 2001. Annual abundances were always greater at station NB than at EN (Fig. 15). Significant (p ≤ 0.05) positive correlations of abundance were found among all river stations and between station NB and EN, but not among the river and bay stations (Table 11).

Abundance of Stage 3 larvae in 2002 was the sixth highest observed at station A, seventh highest at station B, and eighth highest at station C, and in the upper third of all years at EN and NB (Figs. 14 and 15). Of the four larval developmental stages, Stage 3 annual abundances were the most similar among all the stations, although numbers tended to be lower at A than elsewhere. The pattern of annual abundances among stations showed positive and significant correlations (p ≤ 0.05) among all stations, except those between stations A and NB and B and NB (Table 11).

Annual abundances of Stage 4 larvae were generally lowest in the upper river (station A) and greatest in the lower river (C), with intermediate abundances found at station B (Fig. 14). This suggests a gradual flushing to the lower river as development proceeds and perhaps an emigration of older, pre-settling larvae transported or emigrating from the bay into the river late in the season. During 2002, few Stage 4 larvae were taken at A, abundance was about average at B, and was eighth highest at C. Annual abundance of Stage 4 larvae collected at station EN approximated the 20-year mean at stations EN and NB (Fig. 15). Including data from 2002, abundance of Stage 4 larvae at station A was only significantly correlated with that at B and B was only similar to C (Table 11). In Niantic Bay, abundances at NB and EN were significantly correlated.

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 (EN and NB combined) for each developmental stage (Table 12). 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 2002, the dates of peak abundance for Stage 1, 2 and 4 larvae in the Niantic River were among the earliest dates of the 20-year period, but it was about average for Stage 3. The date of peak abundance for Stage 2 larvae in

Niantic Bay was one of the later ones, but those for Stage 3 and 4 were among the earliest.

Water temperature in Niantic Bay was warmer than average in February (5.16°C; Table 1) and peak winter flounder spawning probably occurred prior to mid-February (see Adult Winter Flounder, above). Rogers (1976) and Buckley (1982) reported egg incubation times for winter flounder and Keller and Klein-MacPhee (2000) recently noted a 20-day incubation period for eggs at 4.1°C and 30 days at

TABLE 12. Estimated annual dates of peak abundance of larval winter flounder by developmental stage in the Niantic River and Bay and the number of days corresponding to the 95% confidence interval from 1983 through 2002.

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)
1997	March 4 (3)	March 10 (2)	April 2 (4)	April 18 (4)
1998	February 27 (2)	March 5 (2)	April 19 (22)	May 5 (5)
1999	February 28 (3)	March 28 (13)	April 10 (2)	April 24 (6)
2000	March 4 (1)	March 9 (4)	April 17 (5)	April 25 (5)
2001	March 7 (3)	March 4 (2)	April 9 (2)	May 24 (21)
2002	February 16 (1)	February 23 (1)	April 11 (2)	April 28 (2)
		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 (3)	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)
1996	-	April 7 (4)	April 30 (2)	May 24 (9)
1997	-	March 23 (7)	April 13 (1)	April 30 (9)
1998	-	March 25 (6)	April 23 (3)	May 14 (6)
1999	-	April 9 (3)	April 23 (6)	May 9 (16)
2000	-	April 1 (3)	May 3 (11)	May 22 (4)
2001	-	April 9 (3)	-[a]	May 23 (3)
2002	-	April 20 (2)	April 11 (1)	May 4 (1)

[a] Due to low abundance during the 1993 and 2001 sampling, the Gompertz function could not be fit to the data for these developmental stages.

1.6°C. Buckley et al. (1990) observed that egg developmental time was inversely related to water temperature during oocyte maturation and egg incubation. The combination of record warm water temperatures in February and March (Table 1) probably contributed to the earlier than average date of peak abundance for Stage 1 larvae in 2002. A comparison between February water temperatures and the annual dates of Stage 1 peak abundance in the river during 1983 through 2002 showed a significant negative relationship (Spearman's rank-order correlation; $r = -0.690$; $p < 0.001$; $n = 20$). Dates of peak abundance for later developmental stages were likely related to temperature-moderated developmental rates and perhaps variable rates of flushing, which could accelerate the buildup of larval abundance sequentially at stations in the Niantic River and Bay. The relationship between water temperature and larval developmental rate is discussed in more detail in the following section.

Development and Growth

The length-frequency distribution of each larval stage has remained relatively consistent since developmental stage determination began in 1983. Stage-specific length-frequency distributions by 0.5-mm size-classes in 2002 showed some separation in predominant size-classes by developmental stage (Fig. 17). Nearly all (98%) Stage 1 larvae were in the 2.5 to 3.5-mm size-classes, 99% of Stage 2 were 3.0 to 5.0 mm, 96% of Stage 3 were 5.0 to 7.0 mm, and 97% of Stage 4 were 6.0 to 8.0 mm. These predominant size-classes for each developmental stage were generally similar to those reported for previous years (DNC 2002). Consistent stage-specific length-frequency distributions from year to year indicated that developmental stage and length of larval winter flounder were closely related. These data agreed with laboratory studies on larval winter flounder, which showed that there were positive correlations between growth and developmental rates (Chambers and Leggett 1987; Chambers et al. 1988; Bertram et al. 1996).

Length-frequency distributions of larvae (all stages combined) collected in the Niantic River (stations A, B, and C combined) differed from those obtained for Niantic Bay (EN and NB combined) for all years of sampling, including 2002 (Fig. 18). Smaller (≤ 4.0 mm) size-classes usually predominated in the river due to the high abundance of Stage 1 larvae. The cumulative total of 84% of larvae ≤ 4.0 mm in 2002 was slightly more than the historical (1983-2001 combined) length-frequency distribution, which
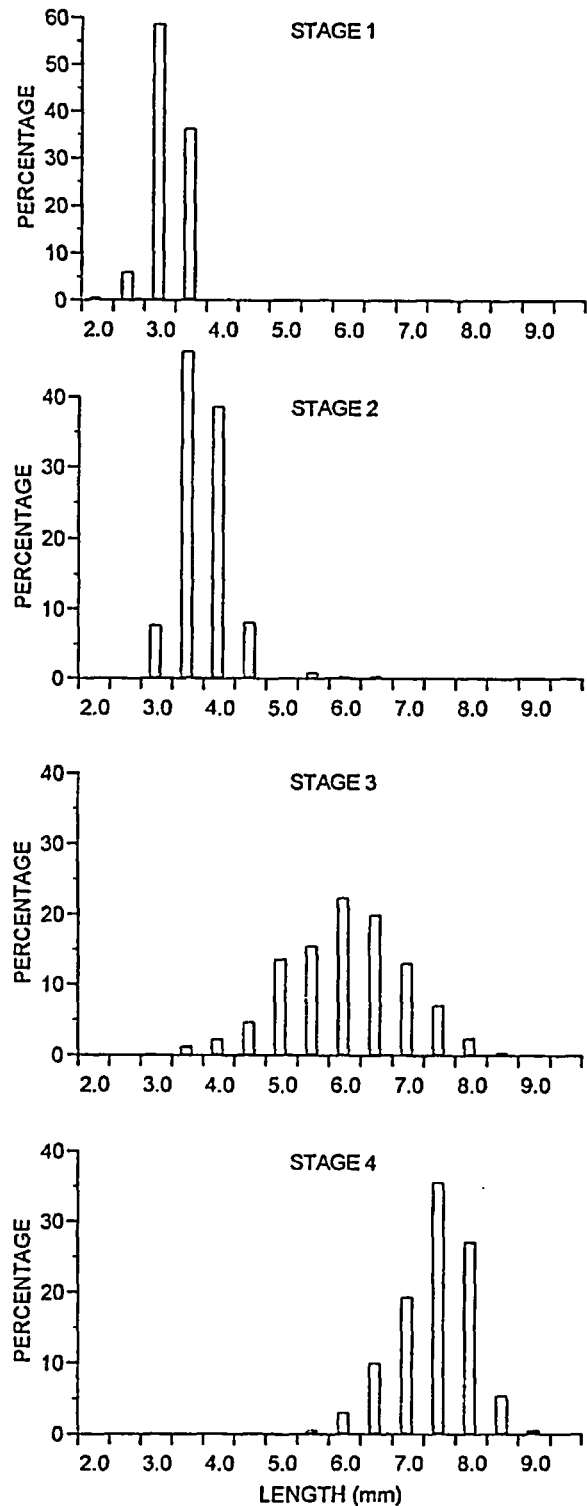
Fig. 17. Combined length-frequency distribution of larval winter flounder by developmental stage at all stations sampled in the Niantic River and Bay during 2002.
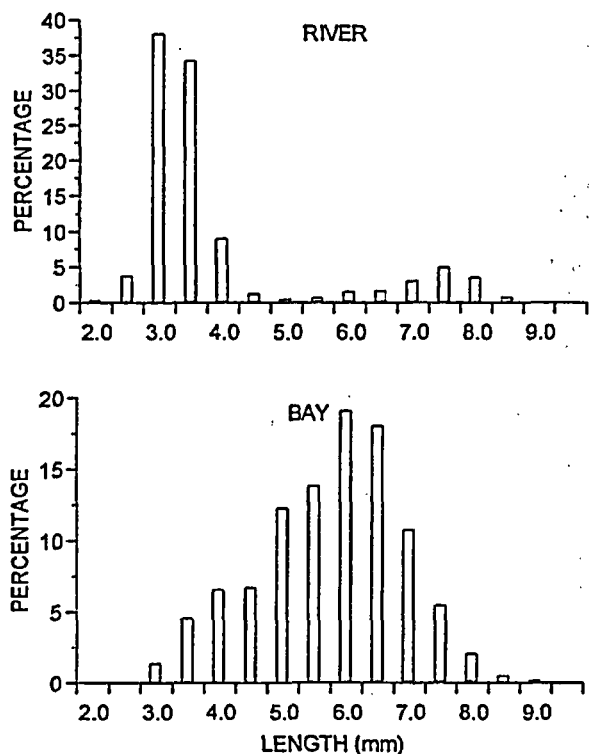
Fig. 18. Length-frequency distribution of larval winter flounder in the Niantic River and Bay during 2002.

showed about 75% of larvae in these smaller size-classes. In addition, length-frequency distributions in the river for some previous years showed a slight increase or near-stable frequencies of larvae in the 7.0 to 8.0-mm size-classes; this was also apparent during 2002. These results suggest that some older larvae were imported into the river. Importation of larger size-classes was also apparent in the length-frequency distribution at a station located at the river mouth, which was sampled in 1991-93 during maximum flood tidal currents (NUSCO 1994a).

The size-class distribution for Niantic Bay in 2002 was also consistent with most previous findings. Generally, a skewed unimodal distribution in length-frequency was found with the larger size-classes predominating, except for 1996 (NUSCO 1997b). Typically, the greatest size frequencies found in the bay were the 5.0- to 7.0-mm size-class (1983-2001 combined, 53%), as they were in 2002 (74%; Fig. 18). The sharp decrease of larvae 8.0 mm and larger occurs as larvae metamorphose to the demersal Stage 5 juvenile.

Length-frequency data from entrainment collections taken from 1976 through 2002 (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 ≤ 0.001) different from zero (Table 13). 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 0.046 to 0.113 $mm \cdot day^{-1}$, with growth in 2002 being the lowest observed in the 20-year monitoring program. To validate this estimation technique, growth rates were estimated from length data collected at station NB from 1979 through 1989 (NUSCO 1990); annual growth rates were highly correlated (r = 0.89; p ≤ 0.001; n = 11) with those from station EN.

In laboratory and mesocosm studies, water temperature affected the growth rate of winter flounder larvae (Laurence 1975; NUSCO 1988b; Casas 1998; Keller and Klein-MacPhee 2000). To examine the effect of temperature on estimated annual growth rates, mean water temperatures in Niantic Bay, determined from data collected by continuous recorders in the MPS intakes, 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 13). The mean temperatures used were not necessarily 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 1999 having the largest growth rate as well as the highest water temperature (Fig. 19). Growth rate in 2002 appeared to be lower than indicated by the 6.8°C mean water temperature. Hovenkamp and Witte (1991) reported a similar exponential relationship between temperature and growth for larval plaice. Keller and Klein-MacPhee (2000) also reported larval winter flounder growth to be significantly greater in warmer than cooler experimental mesocosms. If temperature affects growth rate, then the length of a larva at a specific time during the season should be related to water temperatures to which it has been exposed. Therefore, the mean length of larvae collected at station EN during the period of April 1-15 for each year was compared to the mean March water temperatures (Fig. 20). There was a positive relation-

TABLE 13. 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 2002. 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	6.3
1990	March 4 - May 13	0.071	0.066 - 0.076	5.3
1991	March 10 - April 21	0.059	0.048 - 0.070	6.6
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
1997	March 9 - May 11	0.063	0.056 - 0.070	4.9
1998	March 1 - May 3	0.070	0.064 - 0.075	5.3
1999	March 28 - April 25	0.113	0.100 - 0.126	8.3
2000	March 12 - April 23	0.093	0.085 - 0.102	6.4
2001	March 18 - April 29	0.054	0.046 - 0.062	6.0
2002	March 10 - June 9	0.046	0.037 - 0.058	6.8

[a] Time period of the larval 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.

ship, with larger mean lengths associated with warmer March temperatures. The mean length found in 2002 was even higher than the predicted value at the highest observed March mean water temperature (Table 1).

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
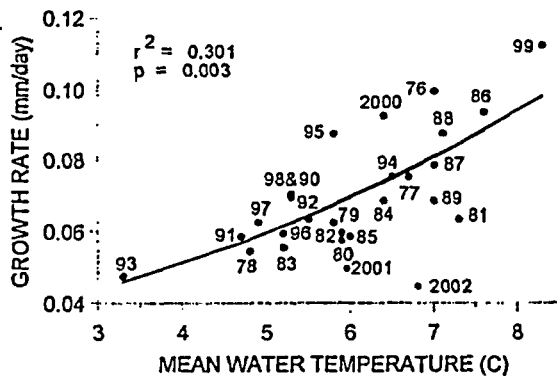
Fig. 19. Exponential relationship between mean water temperature T (°C) and the estimated growth rate G (mm per day) of winter flounder larvae at station EN from 1976 through 2002 (G = 0.033 × $e^{0.116T}$).
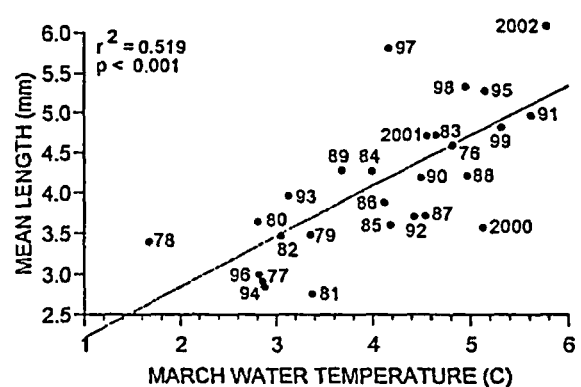
Fig. 20. Relationship between annual mean March water temperature (°C) and the mean length of winter flounder larvae during April 1-15 at station EN for 1976 through 2002.

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. 21). 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 26-year period, possibly because of a 3.6°C difference in the March-April water temperature between one of the earliest (April 13, 1991) and the latest (May 23, 1978) dates of peak abundance. Despite the wide range in annual growth rates, a consistent relationship was found between length-frequency distribution and stage of development (Fig. 17). This was also consistent with laboratory observations of larval winter flounder, as Chambers et al. (1988) found that age at metamorphosis was more variable than length and larval age and length were independent of one another. The date of peak abundance in 2002 was consistent with respect to information on growth as the relatively early date of peak abundance (April 15) was associated with the highest average water temperature of 7.1° recorded for this period.

Growth rates were also estimated for Niantic River larvae using length and temperature 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. 14). A linear model again provided a good fit and slopes (growth rates as mm·day^{-1}) were significantly (p ≤
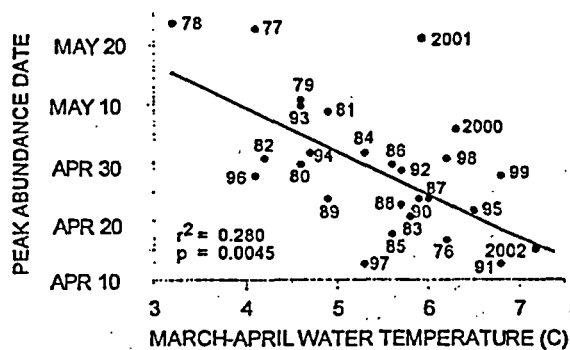
Fig. 21. Relationship between March-April mean water temperature (°C) and the annual date of peak abundance (estimated from the Gompertz function) of winter flounder larvae at station EN from 1976 through 2002.

0.001) different from zero. Estimated growth rates for larvae in the river were generally greater than for larvae from the bay. The 2002 growth rate of 0.064 mm·day^{-1} for the river was below average for the 20-year period, although it was associated with a slightly warmer than average water temperature (Table 14). Growth of larvae in the river in 2002 was lower than growth rates of laboratory-reared fish, which ranged from 0.104 and 0.100 mm·day^{-1} at mean water temperatures of 6.9 and 7.5°C, respectively (NUSCO 1988b).

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 can have a profound effect on recruitment (Houde 1987). Food availability and water temperature appeared to be the two most important factors controlling larval winter flounder growth (Buckley 1982). 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). Although Laurence (1975) demonstrated that the metabolic demands of larval winter flounder increased at higher temperatures, growth rate also increased if sufficient food resources were available, and other laboratory studies showed that larval winter flounder growth rates depend upon prey availability (Laurence 1977; Buckley 1980). The effect of both larval density (i.e., density-dependency) and water temperature on larval growth rates were examined for data collected from the Niantic River by comparing annual growth rates to the abundance index for Stage 2 larvae, the developmental stage during which larvae begin to feed. This method assumed that prey availability was similar from year to year. The annual index of Stage 2 larval abundance was the A parameter (Eq. 2) for all three river stations combined (Table 14). A density-dependent relationship was previously reported (NUSCO 1990, 1991b, 1992a, 1993), but since 1993 this relationship no longer has been significant. Because a strong relationship existed 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. The multiple regression, which included data from 1983 through 1996, was significant (p = 0.035; R^2 = 0.456; df = 11), with a positive slope for temperature and negative slope for Stage 2 abundance. Although the Stage 2 slope was not always significant, this variable was needed for a significant multiple regression model. However, with the addition of 1997 and 1998 data, the relationship

TABLE 14. 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 2002. The 95% confidence intervals of the growth rate, a mean determined for water temperatures recorded at station C during the first 6 weeks of the time period, and the annual abundance indices of Stage 2 larvae in the river are also given.

Year	Time period included[a]	Growth rate (mm·day^{-1})	95% confidence interval	Mean water temperature (°C)[b]	Stage 2 abundance index[c]
1983	March 20 - May 1	0.100	0.096 - 0.104	6.1	749
1984	March 25 - May 6	0.100	0.094 - 0.105	6.4	1,501
1985	March 31 - May 26	0.084	0.080 - 0.088	7.7	4,676
1986	March 23 - May 4	0.109	0.103 - 0.115	8.0	176
1987	March 22 - May 10	0.099	0.095 - 0.103	7.2	829
1988	March 20 - May 21	0.099	0.094 - 0.104	6.8	4,469
1989	March 26 - May 21	0.087	0.082 - 0.092	7.4	3,976
1990	March 25 - May 13	0.106	0.099 - 0.113	7.5	365
1991	March 10 - April 28	0.123	0.114 - 0.132	6.9	252
1992	March 15 - May 17	0.088	0.083 - 0.093	5.7	1,367
1993	March 7 - May 16	0.070	0.065 - 0.075	4.1	133
1994	March 20 - May 29	0.072	0.068 - 0.076	4.7	1,248
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	6.6	4,424
1997	March 2 - April 20	0.136	0.127 - 0.144	5.2	8,449
1998	March 22 - May 3	0.109	0.098 - 0.119	7.4	6,553
1999	March 28 - April 25	0.150	0.129 - 0.171	8.9	288
2000	March 26 - April 30	0.103	0.082 - 0.124	8.1	402
2001	March 25 - April 29	0.106	0.099 - 0.113	5.1	704
2002	March 10 - May 25	0.064	0.060 - 0.068	7.2	1,400

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

was no longer evident (p = 0.280; R^2 = 0.178; df = 13), probably because even though growth rates in those years were among the highest, Stage 2 abundances were also among the highest. With a further addition of data from 1999-2001, a positive relationship was found between growth rate and water temperature (p = 0.008; r^2 = 0.460; df = 16), but there was no apparent density-dependent relationship, even though Stage 2 abundances in these 4 years were among the lowest of the 20-year period. The addition of 2002 data to the model resulted in the loss of the statistically significant relationship (p = 0.241; r^2 = 0.154; df = 17). The varying results from year to year suggested that factors affecting growth may be more complex than just water temperature and abundance, and may be related to factors such as prey abundance and diversity, for which no information was collected. In summary, growth and development of winter flounder larvae in Niantic Bay appeared to be related to water temperature, with a similar, although less certain, relationship possible for larvae found within the Niantic River.

Mortality

The 2002 estimate of instantaneous natural mortality rate (M) of 2.23 from hatching (Stage 1) to 7 mm (Stage 4) was about average across the time-series (Table 15). This instantaneous rate translated

TABLE 15. Estimated larval winter flounder total instantaneous natural mortality rate (M) from hatching to the 7-mm size-class from 1984 through 2002.

Year	Abundance index Newly hatched	7-mm size-class	% mortality rate (A)[a]	Instantaneous mortality rate (M)[a]
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
1997	14,894	1,645	89.0	2.20
1998	9,306	921	90.1	2.31
1999	4,658	791	83.0	1.77
2000	1,725	1,758	-[b]	-[b]
2001	4,252	1,708	59.8	0.92
2002	4,915	530	89.2	2.23
				mean = 2.47

[a] Survival = $1 - A = e^{-M}$.
[b] Could not be calculated as more larger than smaller larvae were collected.

into a total mortality rate (A) of 89.2%, which was also moderate. From 1984 through 2001, total mortality from hatching to just prior to larval metamorphosis ranged from 59.8 to 97.9% with corresponding estimates of M of 0.92 to 3.88.

Based on length-frequency distributions in the river during previous years, most larval winter flounder mortality occurred between the 3.0 and 4.5-mm size-classes, which included yolk-sac (Stage 1) and first-feeding Stage 2 larvae. A decrease in abundance of these size-classes was also apparent in the river during 2002 (Fig. 18, upper). Constructing a catch curve from catch data of these smaller larvae resulted in an estimated M of 2.4. Based on a growth rate of 0.064 mm·day^{-1} determined in 2002 for Niantic River larvae, it would take approximately 23 days for a larva to grow from 3 to 4.5 mm (Table 14). Daily M was therefore 0.104, in contrast to a higher long-term (1983-98) average of 0.174·day^{-1} (NUSCO 2000). This apparently lower than average mortality during the transition to first-feeding larvae in 2002 was illustrated by a smaller than average decline in abundance from Stage 1 to Stage 2 larvae at the three river stations and because Stage 2 larvae were particularly abundant in Niantic Bay this year (Figs. 14 and 15). The decrease in abundance of older larvae was examined using larvae collected in Niantic Bay (Fig. 18, lower). Here the catch curve from 6 to 7.5 mm (Stages 3 to 4) resulted in an estimate of M = 0.86. With a growth rate of 0.064 mm·day^{-1} in Niantic Bay (Table 14), a larva would take approximately 23 days to grow from 6 to 7.5 mm. Thus, an M of 0.037·day^{-1} was indicated.

In other studies, 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. Keller and Klein-MacPhee (2000) reported mortality rates of 5.6 to 8.9%·day^{-1} in their warm mesocosm tanks and 3.6 to 4.0%·day^{-1} in cold tanks, although predators in these experimental tanks were relatively low in comparison to natural systems. Larval mortality rates in the Niantic River appear to be similar.

Gelatinous zooplankton (jellyfish and ctenophores) are potential predators of winter flounder larvae (see Formation of Winter Flounder Year-class Strength, below). During 7 of the 18 years (1983, 1984, 1986, 1989, 1990, 1994, and 1998) sampled, weekly mean larval abundance of larvae at station A was negatively correlated (p ≤ 0.05; Spearman's rank-order correlation coefficients, r_s, ranged between -0.736 and -0.927) with the weekly mean jellyfish volume during the period when both medusae and larvae were collected. Weekly jellyfish volumes in 2002 were

generally higher than the long-term average (1983-2001) from early March through mid-April (Fig. 22), but were not significantly negatively correlated with weekly mean larval abundance (r_s = -0.563, p = 0.322) as was the case in 2001 (DNC 2002). This year, larval abundance at station A was relatively low throughout the sampling period compared to the long-term average, but larvae were consistently captured at the station through mid-April and abundance peaked in late March. Jellyfish abundance at Station A was higher than the long-term average, reaching a peak volume of nearly 4.0 L·500 m^{-3} in late April (Fig. 22). In many years these gelatinous zooplankton generally appeared late in the larval season at station A at a time when winter flounder larval abundance was declining in this area. Thus, the temporal distribution of jellyfish and winter flounder larvae may have overlapped very little. Although for some years there was an apparent negative relationship between larval abundance and gelatinous predators, it may just be coincidental and due to a natural decline in larval abundance by gradual flushing from the upper portion of the river. Thus, the role of jellyfish in cropping winter flounder larvae remains uncertain.

The possibility of density-dependent mortality of winter flounder larvae was examined using a nonlinear 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 8) was used as a measure of spawning stock size. The A parameter from the Gompertz function fitted to the abundance of 7-mm and larger larvae collected from 1976 through 2002 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). Abundance of larvae in the 7-mm and larger size-classes was used as
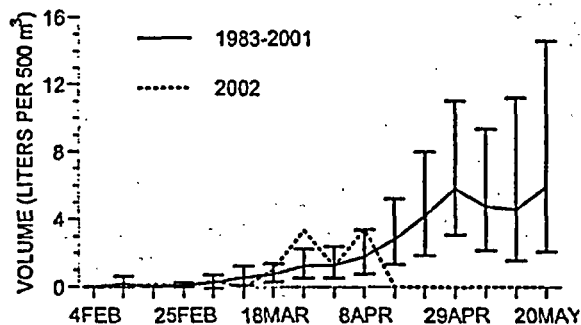
Fig. 22. Comparison of weekly mean volume and 95% confidence interval of gelatinous zooplankton (primarily *Cyanea* sp.) collected at station A in the Niantic River for the period of 1983 through 2001 with the weekly mean volume taken in 2002.

an index of early recruitment because these larvae would soon metamorphose into juveniles. This index of late larval abundance was plotted against egg production estimates and the nonlinear slope (b in Eq. 4) was estimated by nonlinear regression (Fig. 23). A significant slope, $b = 0.196$ ($p < 0.0001$; $n = 26$), suggested the existence of compensatory mortality. As a measure of precision, the standard error for b was only 16% of the estimated value. The 4 years (1995, 1997-99) with the highest abundance of 7-mm and larger larvae were among the years with the lowest egg production estimates (Table 9) and apparently highest egg survival (Fig. 16). Because evidence exists 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 and similar early life history processes. Also, the apparent higher egg survival in the 1995-99 period may have resulted in better recruitment to larger larval size-classes. Although the production of yolk-sac larvae this year in the Niantic River appeared to be more in line with estimated egg production, the high abundance of older larvae appears to support the concept of compensation at low levels of abundance.

Another approach to detect the possible presence of density-dependent larval mortality of the Niantic River stock was a comparison of natural mortality rate (M) of larvae from the river (Table 15) to estimated egg production from 1984 through 2002 (Table 9). In addition, the effect of water temperature on larval mortality was examined. A multiple regression model used egg production estimates with
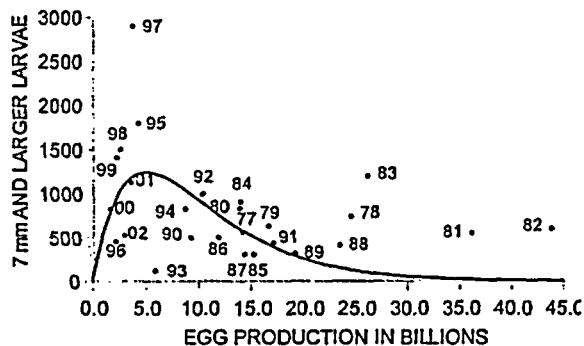
various combinations of seasonal water temperature recorded at MPS (Tables 1 and 2). The combinations included monthly means for March though May, combinations of March and April and April and May, and seasonal means for winter (January-March) and spring (April-June). The best model ($R^2 = 0.592$; $p = 0.004$; $df = 14$) indicated that larval mortality decreased as egg production ($p = 0.010$) decreased and April temperature ($p = 0.011$) increased (Fig. 24). This suggests that density-dependent larval mortality occurred in the Niantic River that was further moderated by April 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 1998 was likely related to one of the lowest annual egg production estimates and the warmest April water temperature (Tables 1, 8, and 14). Conversely, a relatively high mortality rate in 1988 was associated with high egg production and cool temperatures.
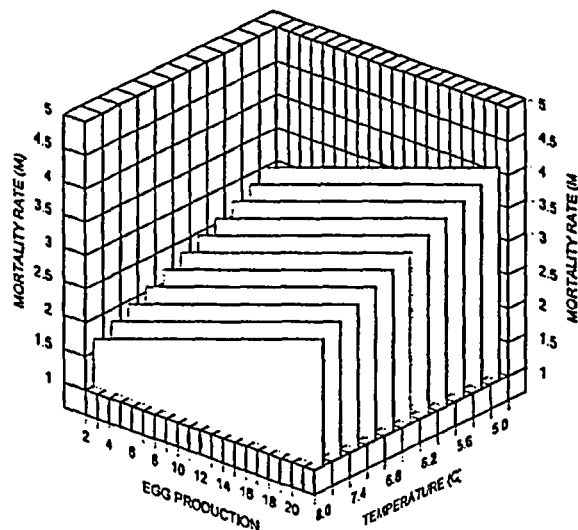
Fig. 24. Relationship between the instantaneous natural mortality rate (M) of winter flounder larvae and annual egg production in the Niantic River and April mean water temperature (°C) at the MPS intakes from 1984 through 1999 and 2001-02: M = 5.297 + 0.0602(egg production in billions) − 0.4252(April water temperature).

Juvenile Winter Flounder

Age-0 Juveniles during Summer

Field Sampling and Data Analyses. The Niantic River serves as a nursery ground for much of the first year of life of post-larval age-0 winter flounder. Abundance of this life stage has been monitored from

Fig. 23. Relationship between the annual winter flounder egg production in the Niantic River and the annual abundance of 7 mm and larger larvae at station EN from 1977 through 2002. The curve represents predicted values from fitting Eq. 4.

late May through the end of September since 1983 (LR) or late 1984 (WA) at two shallow (about 1-2 m in depth), nearshore stations in the Niantic River (Fig. 2). Through 1992, collections were made weekly, but in 1993 sampling frequency was reduced to biweekly, resulting in ten dates sampled during a 19-week season. Sampling occurred during daylight from about 2 hours before to 1 hour after high tide. A 1-m beam trawl was used that had 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 of the four-net sequence was made when fish had grown large enough to become retained by it, as use of larger meshes reduced the amount of detritus, algae, and sevenspine bay shrimp (*Crangon septemspinosa*) 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 net damage. Tow distance was estimated by releasing a measured line attached to a lead weight as the net was hauled at approximately 25 m·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 fish were high, maximum tow length may have been only 60 or 80 m. Surface water temperature was taken with a hand-held mercury stem thermometer.

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 therefore 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:

$$TL = -0.2 + 1.212(SL). \qquad (5)$$

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 Tukey's studentized range test (SAS Institute Inc. 1990c). The relationship between growth and abundance of young and water temperature was examined using multiple linear regression (SAS Institute Inc. 1990d) and functional regression methods described previously in the Larval Winter Flounder section.

To calculate M, all young were assumed to comprise a single cohort with a common birthdate. A catch curve was constructed using data from each station such that the natural logarithm of density was plotted against age (time in weeks) and the slope of the descending portion of the curve provided an estimate of the weekly rate for M (Ricker 1975). Once this rate was determined, the monthly mortality rate (M_{mo}) by station was calculated as M × (30.4 / 7) and monthly survival (S_{mo}) as e^{-Mmo}. For this report, data from both stations (1985-2002) were also pooled and estimates of M and S were made over the entire season (late May through September).

Abundance. 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, Berghahn (1986) caught more young European plaice at night using a beam trawl in comparison to samples taken during the day, while 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 previously in NUSCO (1987, 1990). Although some *Enteromorpha clathrata* algal mats, which hampered sampling efficiency in the Niantic River during much of 1993 and occasionally in other years, was present at LR, no large mats developed that would have affected the catches during 2002. Presence of extensive algal mats may also influence distribution of young winter flounder. Recently metamorphosed (11-15 mm) European plaice avoided filamentous algal mats in comparison to bare sediment in laboratory experiments conducted by Wennhage and Pihl (1994). This preference could potentially affect year-class strength as well as measurements of abundance if large areas of nursery grounds are

vegetated and young are crowded into uncovered patches of bottom.

Following record high abundance of Stage 4 larvae observed in 2001, densities at station C in the Niantic River and at station NB in Niantic Bay were about average this year (Figs. 14 and 15). At the start of sampling in late May young settling in the Niantic River were found at densities of about 50·100 m^2 (Fig. 25). As in 2001, densities increased at both stations through late June, with numbers at LR peaking at about 150·100 m^2 and reaching nearly 100·100 m^2 at WA. Thereafter, abundance at both stations steadily declined until early August, after which it remained relatively stable until late September, when another decline occurred. Increasing abundance during the early part of the season may have reflected a lengthier than usual period for recruitment of settling older larvae that entered the river from Niantic Bay, although the dates for peak abundance of Stage 4 larvae in both Niantic River and Bay this year occurred earlier than in 2001 (Table 12). Also, there may have been a time lag in the shoreward movement of previously settled juveniles from deeper areas of the river. The rate of loss of age-0 winter flounder is discussed below in a subsection discussing their mortality.

Fig. 25. 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 2002.

As measured by early (May-July) and late (August-September) seasonal median CPUE, abundance of age-0 winter flounder during the first half of the season in 2002 was higher (76.3·100 m^2) at LR than at WA (45.0; Table 16). In late summer, fish were almost three times as abundant at LR (26.7) than at WA (9.5). Early summer abundance for both Niantic River stations combined in 2002 was intermediate between record high abundance in 2001 and 4 years of relatively similar abundance observed during 1997-2000 (Fig. 26). The year also had relatively high variation in abundance in contrast to earlier years. Although this summer started with a potential for high production of young, abundance decreases occurring in mid to late summer resulted in the formation of a relatively strong, but not exceptional year-class, as indicated by the combined late summer median CPUE. Again, however, the 95% CI for the late summer CPUE was relatively broad. Nevertheless, abundance of age-0 winter flounder at the end of summer has increased since 1996. Other relatively abundant year-classes since 1984 included those of 1988, 1994, and 1995.

Beginning in 1988, DEP sampled at eight seine stations along the Connecticut coast during September (Gottschall et al. 2002). As was found for the Niantic River, catches throughout LIS showed relatively strong year-classes of winter flounder produced in 1988, 1992, 1994, and 1995. Following 2001, when time-series lows were found at stations in Greenwich, Bridgeport, Milford, New Haven, and Old Lyme, catches at all but the Milford station increased in 2002 (D. Molnar, CT DEP, Old Lyme, CT, pers. comm.). One of the DEP seine stations was at a beach on the Waterford shoreline of the Niantic River adjacent to the beam trawl station WA. The geometric mean seine catch of 9.0 in 2002 was nearly the same value as the CPUE of 9.1 in 2001. Catches of young winter flounder at these two stations were significantly positively correlated (Spearman's rank-order correlation; r =0.576; p = 0.025; n = 15). The proximity of the two sampling sites allowed for comparison of the annual late summer median 1-m beam trawl CPUE index at WA and the DEP seine CPUE using functional regression (Fig. 27). Despite relatively wide confidence intervals associated with each of these two abundance estimates, a significant positive relationship (r = 0.601; p = 0.018; df = 13) was found between these two indices. Some differences in relative abundance found by these two sampling methods appeared to have been explained by summer water temperatures affecting fish distribution (NUSCO 1999), but this effect was less certain when re-examined in NUSCO (2000). In general, however, both sampling methods allowed for

TABLE 16. 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 2002.

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

TABLE 16. (continued).

Survey year[a]	Station	Season[b]	Tows used for CPUE	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness[c]
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
1997	LR	Early	20	19.2	16.7 - 25.0	1.03
	LR	Late	20	7.0	2.0 - 10.0	0.29
	WA	Early	20	53.8	35.0 - 80.0	0.59
	WA	Late	20	9.5	6.3 - 20.0	0.92
1998	LR	Early	20	46.3	32.5 - 60.0	0.18
	LR	Late	20	15.0	6.7 - 20.0	1.08
	WA	Early	20	45.0	25.0 - 65.0	1.42
	WA	Late	20	13.0	8.0 - 25.0	1.00
1999	LR	Early	20	47.5	25.0 - 80.0	1.30
	LR	Late	20	29.2	16.0 - 35.0	0.57
	WA	Early	20	37.5	27.5 - 57.5	2.16
	WA	Late	20	17.1	11.7 - 18.8	1.13
2000	LR	Early	20	38.8	10.0 - 67.5	0.52
	LR	Late	20	27.5	22.5 - 35.0	1.34
	WA	Early	20	40.0	22.5 - 58.3	0.35
	WA	Late	20	47.9	35.0 - 52.5	0.75
2001	LR	Early	20	180.0	107.5 - 247.5	0.27
	LR	Late	20	50.0	36.7 - 82.5	1.73
	WA	Early	20	141.3	45.0 - 227.5	0.10
	WA	Late	20	21.7	10.0 - 32.5	1.71
2002	LR	Early	20	76.3	50.0 - 135.0	0.48
	LR	Late	20	26.7	11.7 - 33.3	0.08
	WA	Early	20	45.0	16.7 - 75.0	0.66
	WA	Late	20	9.5	3.8 - 15.0	0.92

[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.

the identification of both particularly strong and weak year-classes of winter flounder and serve to corroborate one another. Continued sampling of juvenile winter flounder following their movement out of shallow waters during late fall and early winter is discussed in the following section of this report and provides additional evidence of relative year-class strength during the first year of life.

Growth. Growth of young winter flounder is important because size of fish often has important implications for survival, which in turn affects recruitment. Growth of age-0 winter flounder was measured by increases in mean length over time. In some years, increases in weekly or biweekly mean length occurred steadily throughout the season, whereas in other years asymptotes appeared to be reached by mid-summer. During 2002, mean lengths showed only very small increases from week to week before mid-August, after which steady increases were found (Fig. 28). Based on overlapping error bars associated with each biweekly mean length, little difference in size was seen between the two stations. The relatively small standard errors also indicated little variability in size until September, although an occasional individual much larger than average was taken. As in 2001, nearly all fish collected this year

Early summer (late May-July)

Late summer (August-September)
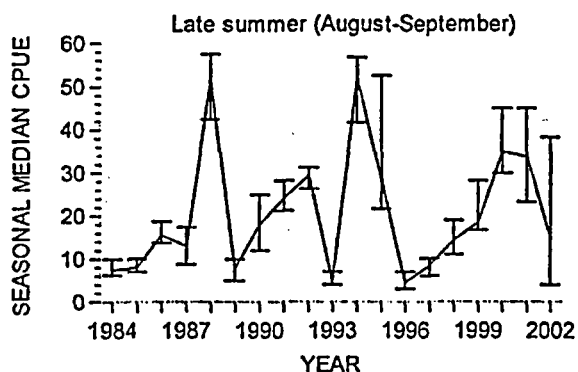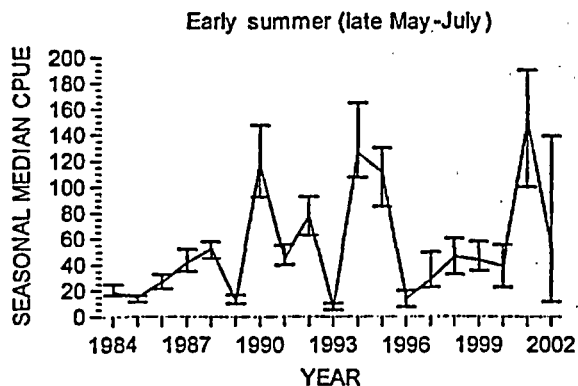
Fig. 26. Early and late summer seasonal 1-m beam trawl median CPUE and 95% CI at Niantic River stations LR and WA combined from 1984 through 2002. (Note that the vertical scales differ between the graphs).

were 60 mm or smaller, but a few individuals had lengths of almost 90 mm.

Fast growth after settlement followed by a rapid decline in growth rate was reported for young winter

r = 0.601

p = 0.018 *
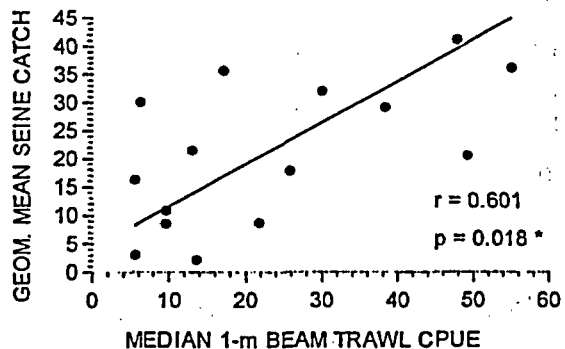
Fig. 27. Relationship (functional regression) between the late summer (August-September) seasonal 1-m beam trawl median CPUE at station WA in the Niantic River and the geometric mean seine catch during September at a nearby shoreline station sampled by CT DEP from 1988 through 2002.

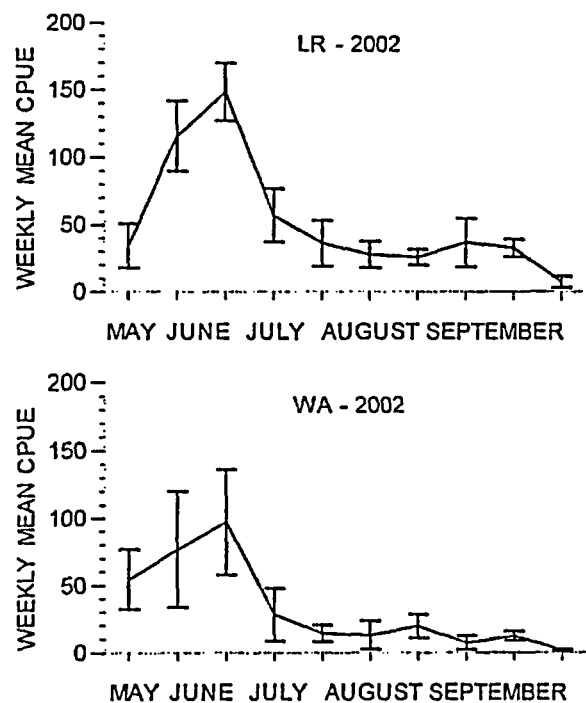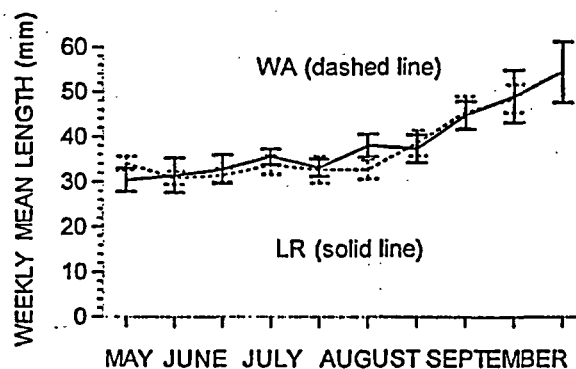
Fig. 28. Weekly mean length (±2 standard errors) of age-0 winter flounder taken at Niantic River stations LR and WA by 1-m beam trawl during 2002.

flounder in New Jersey bays by Sogard and Able (1992), who found nearly imperceptible growth rates by the time young winter flounder 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). The few larger individuals taken this year may have reached a size giving them a competitive advantage over other fish and perhaps were able to utilize a food resource not available to smaller winter flounder.

Mean length of fish at LR during late summer (i.e., July through September) in most years since 1984 was significantly greater than that of fish at WA (Table 17). In some years, most recently in 2000 (12 mm), a relatively large (≥8 mm) difference was found in mean length between the two stations. Differences were moderate (3-4 mm) in a few years (e.g., 1986, 1992, 1998) and mean lengths were similar (−2 to 1 mm) in other years (e.g., 1991, 1994, 2001), including 2002. This year, mean lengths of young were 43.8 mm at LR and 42.8 mm at WA and these means were not significantly different. Although in the lower half of mean lengths observed since 1984, these means were still much greater than those attained in 2001, which were significantly smaller than those from all other years.

Growth of young winter flounder is affected by water temperature, with both positive and negative effects possible. Warmer temperatures result in increased growth rates, but after some point increasing respiratory and other metabolic demands inhibit growth. Sogard and Able (1992) and Meng et al. (2000) noted that optimal growth temperatures for young winter flounder can be exceeded, with the latter finding reduced growth at temperatures

TABLE 17. Comparison of the mean lengths (mm) of age-0 winter flounder taken at stations LR and WA in the Niantic River from mid-July through September of 1983 through 2002. Seasonal mean lengths are ranked from largest to smallest and significant differences in means were tested by one-way analysis of variance and Tukey's studentized range (HSD) test.

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	97	98	99	00	01	02
Difference (LR - WA)	16	15	4	-2	0	8	1	1	4	10	1	8	13	8	3	9	12	1	1

Significant differences among means[a]						Mean length	Number of winter flounder	Station	Year
			A			66.2	281	LR	1983
			B			60.9	112	LR	1993
	C		B			59.7	94	LR	1996
	C		B	D		58.7	231	LR	1984
	C			D		57.8	345	LR	1989
	C	E		D		57.1	258	LR	1985
		E		D		56.0	596	LR	1991
		E				54.6	721	WA	1991
			F			50.9	134	WA	1993
	G		F			50.5	878	LR	1988
	G		F			50.5	902	WA	1988
	G		F	H		49.9	246	WA	1989
	G	I	F	H		48.0	812	LR	1992
	G	I		H		47.7	125	LR	1997
		I		H		47.2	333	LR	1999
		I	J			46.8	147	WA	1996
	K	I	J			46.6	377	WA	1987
	K	I	J			46.5	294	LR	2000
	K	I	J	L		46.2	258	LR	1995
M	K	I	J	L		45.7	563	LR	1986
M	K	I	J	L	N	45.2	632	LR	1987
M	K	O	J	L	N	43.9	810	WA	1992
M	K	O		L	N	43.8	320	LR	2002
M		O		L	N	43.4	690	LR	1990
M		O	P	L	N	43.4	370	WA	1984
M		O	P		N	42.8	155	WA	2002
	Q	O	P		N	42.3	349	WA	1985
	Q	O	P			42.0	533	WA	1990
	Q	O	P			41.9	721	WA	1986
	Q		P	R		40.5	230	LR	1998
	Q		P	R		40.4	392	LR	1994
	Q	S		R		39.6	205	WA	1997
		S		R		38.7	369	WA	1994
		S		R		38.1	415	WA	1995
		S		R		37.8	239	WA	1999
	T	S				37.1	246	WA	1998
	T					34.2	371	WA	2000
			U			31.1	237	WA	2001
			U			29.6	450	LR	2001

[a] Means with the same letter are not significantly (p ≤ 0.05) different from each other.

exceeding 23°C. Manderson et al. (2002) found growth most rapid at 16-17°C and it was relatively poor at above 21°C. With some exceptions, smaller mean lengths tended to occur in years when summer water temperatures (Table 2) were warmest with larger means found during cooler years. In this regard, summer water temperatures from 1999 through 2002 were among the highest in 27 years (Table 2) and likely influenced the relatively small mean lengths observed for fish taken at WA during these summers. Some differences in mean lengths noted each year between LR and WA (Table 17) were also likely due to water temperature, as WA was generally warmer (ca. 0.5-1.0°C) than LR. No numerical relationships have been found between mean length and water temperature (see below) nor have the annual differences in mean length at the two stations sampled been consistent.

Another physicochemical factor potentially affecting growth, distribution, and survival is dissolved oxygen (DO). Growth and survival of young winter flounder were depressed in habitats where DO concentrations were low (ca. <2 mg·L^{-1}) during extended periods (Bejda et al. 1992; Phelan et al. 2000; Meng et al. 2001; Stierhoff and Targett 2002). Data provided in Goldberg et al. (2000) showed that at a mid-Niantic River station (2 m in depth at low tide) in August 1997, DO levels in association with a period of warm water temperatures averaged 3.42 mg·L^{-1} and periodically decreased to below 2 mg·L^{-1}. DO levels at the shallower shoreline stations sampled for age-0 winter flounder are unknown, but low DO in combination with warm water temperatures in late summer could depress growth rates of young winter flounder.

Occurrence of relatively large mean lengths associated with low age-0 winter flounder abundance in some years and small mean lengths in years when fish were abundant suggested that growth was density-dependent. DeLong et al. (2001) reported that age-0 winter flounder growth rate was negatively related to their density. However, numerous accounts of age-0 European plaice were inconclusive with respect to effects of density on 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). They concluded that increases in length corresponded to maximum growth expected from prevailing water temperatures and that growth was not density-dependent. Nash et al. (1994) suspected a complex interaction between density of age-0 European plaice and water temperature upon growth. Bergman et al. (1988) and Van der Veer et al. (1990) noted that growth of young European 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. Pihl and Van der Veer (1992) also believed that growth of young European plaice in Swedish bays was affected by ambient water temperatures and was not food-limited, although Modin and Pihl (1994) later found evidence for density-dependent growth in a year of extremely high densities. In contrast, Berghahn (1987) and Karakiri et al. (1989) suggested that food limitation and not water temperature may have been responsible for differences in growth of European plaice observed among years in the German Wadden Sea. Also, within the Wadden Sea of The Netherlands, growth of European plaice was slower in an area of tidal sand flats than in an area of mixed sediments with a richer benthos (Berghahn et al. 1995). Benthic food

production and its availability also may differ among areas within the Niantic River and likely changes from year to year with variable effects on growth. Juvenile winter flounder growth also appears to be independent of the presence or absence of eelgrass or macroalgae (Meng et al. 2000; Phelan et al. 2000), although Sogard (1992) reported faster growth in unvegetated in comparison to vegetated areas.

The effects of density (median CPUE during July and August) and water temperature (both cumulative degree-days and running average temperature during the periods of both February 1 and May 15 through September 30) on growth (mean lengths achieved during late summer) at each station were examined using a multiple linear regression model. Despite the findings of others regarding the growth of young flatfishes, water temperature generally did not appear to consistently affect age-0 winter flounder growth in the Niantic River, at least using these relatively simple models. Using functional regression, the mean length of young at LR during late summer was significantly negatively correlated (r = −0.734; p = 0.0002; df = 18) with density (Fig. 29). Although several mean lengths at high densities appeared to be greater than would have been expected, mean lengths were also smaller than expected in several years when densities were moderately low. The mean length determined for LR in 2002 (43.8 mm) was greater than predicted (39.2 mm) by this relationship. The relationship between abundance and late summer mean length at station WA was also significant and negative (r = −0.591; p = 0.012; df = 15), but only if data outliers from 1988 and 1991 were excluded. Mean lengths were much larger than expected from the relatively high densities found during these 2 years. However, the mean length achieved in 2002 (42.8 mm) was only 2 mm less than expected from the calculated relationship.

Mortality. Estimates of monthly instantaneous mortality rate (M_{mo}) at each station were obtained annually using catch curves constructed from weekly or biweekly abundance data. This method assumed that young comprised a single-age cohort throughout the season and also did not take into account any off-station movements, which would have positively biased estimates of mortality. With some exceptions, the catch curves generally fit the data well as indicated by relatively high r^2 values (Table 18). Poor fits (e.g., r^2 = 0.47 for LR in 1996) were generally indicative of relatively high variation in weekly abundance estimates. Also, no estimates could be made for WA in 1986, 1993, and 1996 because of considerable variation in weekly abundance. No mortality estimates were determined

Station LR

r = -0.734
p = 0.0002 **

Station WA
x 91

x 88   r = -0.591
        p = 0.012 *
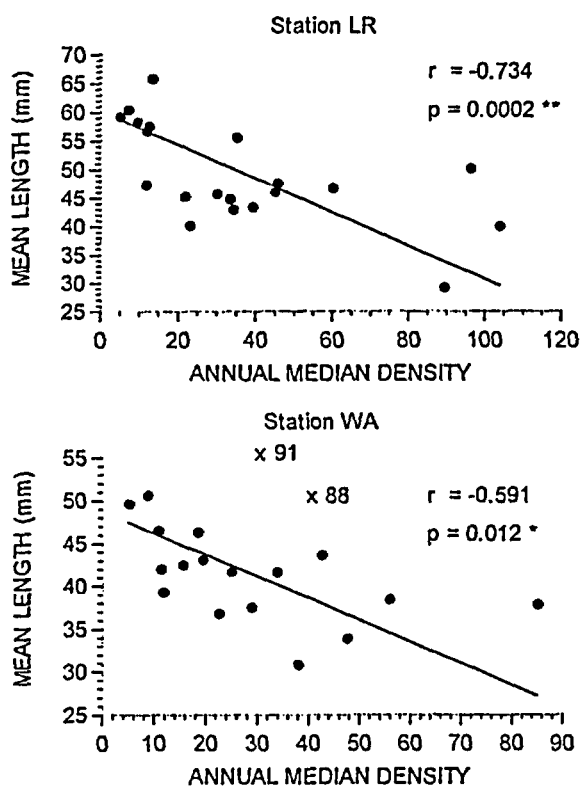
Fig. 29. Relationship (functional regression) between the annual density (median catch per 100 m²) and mean length of age-0 winter flounder during late summer (August-September) at stations LR and WA in the Niantic River from 1983 (LR) or 1984 (WA) through 2002. For WA, data points designated by an 'x' for 1988 and 1991 were not included in the regression. (Note that the vertical and horizontal scales differ between the graphs).

for LR and WA during the high abundance year of 1988 as slopes of these catch curves were not significantly different from zero. This also occurred at WA in 2000, as abundance steadily increased into late summer and remained relatively stable thereafter (DNC 2001a). Moderately high (≥ 0.76) r^2 values were determined for the 2002 catch curves. The M_{mo} estimate for station LR was 0.610 (equivalent to a survival rate S_{mo} of 54.4%), which was similar to the long-term average at this station. However, the M_{mo} of 0.881 (S_{mo} = 41.4%) at WA was the third highest mortality rate observed at that station since 1985 and followed the highest mortality rate found in 2001.

Long-term mean monthly mortality estimates were relatively similar (0.615-0.642) at both Niantic River stations. These values of M were greater than the equivalent value of 0.371 reported by Pearcy (1962) for the Mystic River, CT estuary and 0.336 for post-settlement European plaice in The Netherlands (Beverton and Iles 1992a; Jager et al. 1995), but

within the range of various estimates (0.563-0.693) reported for young European plaice in British coastal embayments (Lockwood 1980; Poxton et al. 1982; Poxton and Nasir 1985; Al-Hossaini et al. 1989). At five different locations in Connecticut, M_{mo} estimates were from 0.24 to 1.21 (Howell 1993; Meise et al. 1999). In Narragansett Bay, RI values ranged between about 0.3 and 1.1 over an 11-year period (DeLong et al. 2001), which was similar to the range of mortality rates found in the Niantic River.

Seasonal mortality rates for age-0 winter flounder in the Niantic River were determined by combining data from both stations (Table 19). This was an attempt to smooth out fluctuations in abundance and poor fits to catch curves that were particularly related to abundance data collected at WA. For the combined catches, significant slopes were found in all years but 1988. Summer seasonal estimates of M ranged from 0.781 in 2000 to 4.320 in 2001 (equivalent to survival of 45.8 and 1.3%, respectively). Particularly high mortality was found during the past 2 years. The long-term mean was 2.335 (S = 9.7%), having a CV of 37%. The magnitude of these estimates illustrates that considerable mortality can occur during the post-entrainment period of settlement of young winter flounder and that ensuing variation in mortality affects resulting year-class strength.

The relationship between annual density of age-0 winter flounder and mortality rate for both sampling stations combined was examined using functional regression (Fig. 30). Although considerable scatter was seen, mortality was significantly positively related to annual median density (r = 0.712; p = 0.0013; df = 15). Highest mortality rates were found associated with highest abundance. Additional discussion on the relationship between fish density and mortality is found below in the section on Formation of Winter Flounder Year-class Strength.

In addition to the Niantic River, Niantic Bay was also sampled for age-0 winter flounder during the summers of 1988 through 1992 (NUSCO 1994a). Mortality of young in the bay was much greater than in the river. Except for a station just outside the mouth of the Niantic River in 1988, no young were found in Niantic Bay from mid-summer through the end of September in each of these years. Even in 1988, however, densities at the bay station in late summer were only 10 to 15% of those in the river at the same time. 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 emigration inshore because none of these smaller, slower growing fish were found in subsequent collections in the river. Because of the

TABLE 18. Monthly instantaneous natural mortality rate (M) estimates and equivalent survival rates (S) as determined from catch curves of age-0 winter flounder taken at two stations (LR and WA) in the Niantic River from 1984 through 2002

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	-	-
1997		8	-0.185 **	0.025	0.90		9	-0.187 **	0.020	0.93
1998		8	-0.101 **	0.013	0.91		8	-0.102 **	0.029	0.68
1999		9	-0.111 **	0.024	0.74		9	-0.128 **	0.029	0.73
2000		8	-0.072 **	0.019	0.71		8	NS	-	-
2001		8	-0.200 **	0.024	0.92		8	-0.276 **	0.037	0.90
2002		9	-0.140 **	0.030	0.76		9	-0.203 **	0.039	0.80

Year	Station	Mortality (M_{mo})^d	Survival (S_{mo})	Station	Mortality (M_{mo})^d	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%		-	-
1997		0.802	44.9%		0.811	44.4%
1998		0.437	64.6%		0.443	64.2%
1999		0.480	61.9%		0.556	57.4%
2000		0.312	73.2%		-	-
2001		0.869	42.0%		1.197	30.2%
2002		0.610	54.4%		0.881	41.4%
	Mean	0.615	54.0%	Mean	0.642	52.6%
	Standard error	0.059		Standard error	0.075	
	CV	41%		CV	42%	

^{a} Weekly sampling during 1984-92 and biweekly sampling thereafter. WA was not sampled in 1984.
^{b} Slope (M) determined by a catch curve (natural logarithm of density plotted against time in weeks) from seasonal sampling during late May through September. The probability level that the slope of the catch curve differs from zero is shown:
      NS - not significant (p > 0.05), * - significant at p ≤ 0.05, ** - significant at p ≤ 0.01.
^{c} Although having a significant slope, the catch curve for 1986 at station WA did not provide a reliable estimate of M because of considerable variation in weekly abundance.
^{d} Monthly mortality rate (M_{mo}) = (-M) × (30.4 / 7) and monthly survival (S_{mo}) = $e^{-M_{mo}}$.

apparent lack of young in Niantic Bay in comparison to the Niantic River, no further sampling was conducted in the bay after 1992. This work also suggested that nearly all winter flounder recruits in this area are likely produced in protected inshore embayments or estuaries rather than in more open areas of LIS. Despite the presence of many winter flounder larvae in coastal Atlantic waters (Smith et al. 1975), only one metamorphosed age-0 winter flounder specimen was captured by Steves et al. (1999) in

TABLE 19. Seasonal (late May through September) and monthly instantaneous mortality rate (M) estimates and equivalent survival rates (S) as determined from catch curves of age-0 winter flounder taken at two stations (LR and WA) combined in the Niantic River from 1985 through 2002.

Year	n[a]	Slope[b]	Standard error	r²	Seasonal M	Seasonal S[c]	Monthly M_mo[d]	Monthly S_mo[d]
1985	15	-0.098 **	0.015	0.76	1.865	15.5%	0.426	65.3%
1986	17	-0.084 **	0.014	0.70	1.599	20.2%	0.365	69.4%
1987	19	-0.103 **	0.013	0.79	1.949	14.2%	0.446	64.0%
1988	19	NS	-	-	-	-	-	-
1989	16	-0.105 **	0.016	0.76	2.003	13.5%	0.458	63.3%
1990	17	-0.208 **	0.025	0.83	3.944	1.9%	0.901	40.6%
1991	18	-0.100 **	0.012	0.82	1.904	14.9%	0.435	64.7%
1992	18	-0.117 **	0.015	0.78	2.227	10.8%	0.509	60.1%
1993	7	-0.086 *	0.022	0.75	1.634	19.5%	0.374	68.8%
1994	9	-0.116 **	0.011	0.94	2.207	11.0%	0.504	60.4%
1995	9	-0.163 **	0.011	0.97	3.101	4.5%	0.709	49.2%
1996	7	-0.117 *	0.033	0.71	2.228	10.8%	0.509	60.1%
1997	10	-0.149 **	0.017	0.90	2.828	5.9%	0.646	52.4%
1998	9	-0.095 **	0.011	0.91	1.813	16.3%	0.414	66.1%
1999	9	-0.121 **	0.017	0.88	2.307	10.0%	0.527	59.0%
2000	8	-0.041 *	0.012	0.64	0.781	45.8%	0.179	83.6%
2001	8	-0.227 **	0.028	0.92	4.320	1.3%	0.988	37.2%
2002	9	-0.157 **	0.031	0.78	2.991	5.0%	0.684	50.5%
Mean					2.335	9.7%	0.534	
Standard error					0.212		0.048	
CV					37%		37%	

[a] Weekly sampling during 1985-92 and biweekly sampling thereafter.

[b] Slope (M) determined from a catch curve (natural logarithm of density plotted against time in weeks) determined by seasonal sampling from late May through September. The probability level that the slope of the catch curve differs from zero is shown:
NS - not significant (p > 0.05), * - significant at p ≤ 0.05, ** - significant at p ≤ 0.01:

[c] $S = e^{-M}$.

[d] Monthly mortality rate $(M_{mo}) = (-M) \times (30.4 / 7)$ and monthly survival $(S_{mo}) = e^{-M_{mo}}$.

sampling on the continental shelf of the New York Bight, indicating that survival of settled winter flounder is extremely low in deeper, offshore areas and that larvae found in these waters are likely surplus production.
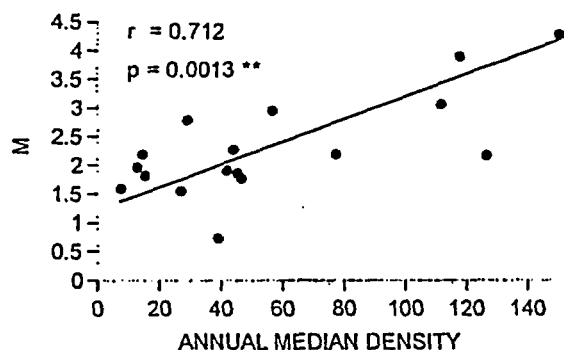
Fig. 30. Relationship (functional regression) between the annual density (median catch per 100 m²) and the instantaneous mortality rate (M) of age-0 winter flounder during summer (late May-September) at stations LR and WA combined in the Niantic River from 1985 through 2002.

Stock Identification. A winter flounder stock identification study performed by Crivello (2002; In preparation) used a DNA-based genetic technique to classify winter flounder larvae entrained at MPS to their source of origin (see MPS Impact Assessment - Larval Entrainment, below). As part of this study, age-0 juveniles collected at stations LR and WA in both early (late June or early July) and late (September) summer from 2000 through 2002 were analyzed using the same techniques as the larvae (except 2000). From 6 to 78 juveniles were processed from these collections by station and date. Fish were assigned to three source populations (Niantic River, Westbrook, Thames River) or as unknown, based on the genetic analysis of yolk-sac larvae collected earlier in the year the three study sites. Most juveniles appeared to have originated from the Niantic River or Westbrook, with fewer coming from the Thames River. Combining data for the two stations and months in 2000, about one-third of the juveniles each came from the Niantic River and Westbrook stocks, 15% were from the Thames River, and the remainder could not be classified specifically to a source area or were unknown. A comparison

could not be made with entrained larvae because the methodology was different in this initial year of work. During 2001, about one-fifth each of the entrained winter flounder larvae and settled juveniles apparently originated from the Niantic River. One-third of entrained larvae in 2001 were assigned to Westbrook, as were 40% of the juveniles; 21% of the larvae and 15% of the juveniles classified to the Thames River. The remaining one-quarter of both larvae and juveniles could not be assigned specifically to a source area or were unknown. In 2002, more than half (59%) of entrained larvae originated from the Westbrook source, as did nearly one-half (48%) of the juveniles. A higher (21%) fraction of juveniles were assigned to the Niantic River population than were entrained larvae (14%), which may indicate that more of the settled winter flounder resulted from larvae that were retained within the Niantic River than were imported. Similar low fractions (4%) of larvae and juveniles were assigned to the Thames River. Once again about one-quarter of the larvae and juveniles in 2002 could not be classified to origin.

Age-0 Juveniles during Late Fall and Early Winter

Field Sampling and Data Analyses. The collection of age-0 fish during late fall and early winter occurs during a transitional period following the 1-m beam trawl sampling of young in summer and preceding the catch of this cohort of fish as age-1 juveniles during the intensive winter-early spring adult winter flounder survey in the Niantic River and elsewhere in the MPS study area. Catches from the year-round trawl monitoring program (TMP; see the Fish Ecology section of this report for methods) were used to determine the abundance of age-0 winter flounder during fall and winter following the movement of these fish from shallow waters near the shoreline to deeper waters as water temperatures decrease. Young were first regularly captured by trawl in November at the two shallower inshore stations (NR and JC) adjacent to inshore nursery grounds and in December at the near-shore Niantic Bay stations (IN and NB). These fish next appeared during January at the deeper-water stations (TT and BR), formerly sampled in LIS as part of the TMP (NUSCO 1996b). A Δ-mean (NUSCO 1988c) index describing relative year-class abundance was developed for age-0 fish using pooled TMP catch data, beginning with the months given above and continuing through the end of February. In January 1996, TMP stations BR, TT, and NB were deleted

(NUSCO 1997a). Therefore, sample size was reduced from 42 during 1976-77 through 1994-95 to 28 in 1995-96 and 25 in 1996-97 and thereafter.

Abundance and Distribution. Based on the availability of data for this report, the most recent Δ-mean CPUE was determined for the 2001 year-class. The Δ-mean CPUE in 2001-02 of 1.5 was the lowest value of the 26-year time-series (Table 20). However, relatively broad 95% CI were computed for nearly all these seasonal Δ-means and differences among years must be viewed with caution. In most years since 1984, when data were first available from Niantic River beam trawl sampling, the late fall-early winter abundance indices reflected densities of age-0 winter flounder found in the river during late summer (Fig. 31). An exception was the 1996-97 Δ-mean of 18.3 for the 1996 year-class, which appeared to be weak from sampling conducted during summer in the Niantic River (NUSCO 1997b). However, the DEP seine catch of young winter flounder showed that the 1996 year-class of winter flounder was likely larger than indicated by the beam trawl catches in late summer of that year. Similarly, the CPUEs of 4.8 in 1995-96, 11.6 in 2000-01, and 1.5 in 2001-02 did not correspond to the relatively high abundance of age-0 fish seen in the Niantic River during the preceding summer months. Greater concordance among abundance indices, however, was found in many other years, including 1994-95, as the Δ-mean CPUE of 31.7 reflected the strong 1994 year-class. Also evident in recent years was good production of young in 1988 and 1992 as well as weaker year-classes in 1993 and 1997, based on Δ-means of 29.6, 31.1, 7.4, and 2.3, respectively. The reason for the disparity found between the abundance age-0 winter flounder in the summers of 2000 and 2001 in the Niantic River and the lack of fish at the TMP stations during the subsequent fall and early winter is not known. These fish could have had different patterns of distribution or dispersal in comparison to other years, were less susceptible to the otter trawl because they were smaller in size (discussed below), or suffered high mortality in late summer or early fall, as suggested by the decline in abundance noted at the Niantic River beam trawl stations in September of 2001 (DNC 2002: Fig. 25) and 2002 (Fig. 25).

Although in many years these two age-0 abundance indices generally tracked one another and were, up until this year, significantly correlated, they are no longer significantly correlated (Spearman's rank-order correlation; r = 0.395; p = 0.105; n = 18). In addition, the fall-winter TMP Δ-mean CPUE and Δ-mean CPUE of winter flounder smaller than 15 cm taken in the Niantic River during the subsequent (late

TABLE 20. The late fall-early winter seasonal[a] Δ-mean CPUE[b] of age-0[c] winter flounder taken at six (1976-95) or three (1996-2002) trawl monitoring stations in the vicinity of MPS.

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
1996-97	25	24	18.3	-4.1 - 40.7
1997-98	25	19	2.3	0.8 - 3.7
1998-99	25	22	10.5	2.1 - 18.9
1999-00	25	23	15.9	-8.3 - 40.2
2000-01	25	21	11.6	-2.7 - 25.8
2001-02	25	18	1.5	0.6 - 2.4

[a] Data restricted during 1976-77 through 1994-95 to November-February at NR and JC, December-February at IN and NB, and January-February at TT and BR and during 1995-96 and thereafter to November-February at NR and JC and December-February at 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.
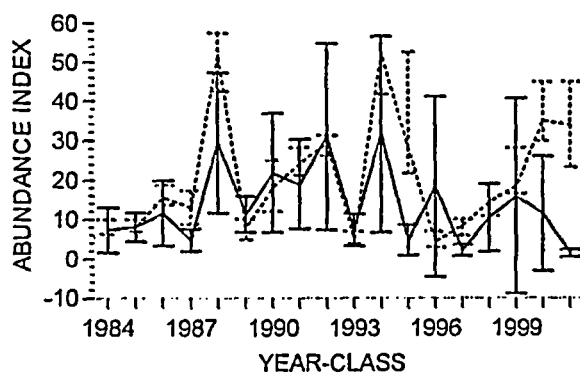
Fig. 31. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line; ±2 standard errors) of age-0 winter flounder (all trawl monitoring program stations) and the late summer Niantic River (stations LR and WA combined) age-0 1-m beam trawl median CPUE (dashed line; ±2 standard errors) for the 1984-2001 year-classes.

February-early April) adult winter flounder survey (see Age-1 Juveniles during Late Winter, below) were not significantly related ($r = -0.128$; $p = 0.532$; $n = 26$; Fig. 32). The relationship between these abundance indices of temporally successive life-stages was unclear. More juvenile winter flounder from the 1984 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, the catch index of young from the TMP in most years has been higher that that of age-1 fish in the river, although the indices were nearly the same magnitude in 1989, 1993, 1997, and 2001. The reason for this areal shift in magnitude of relative abundance that has occurred since the mid-1980s 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 assumes that the relative distribution of fish both inside and outside the river remains constant each year, which likely does not occur.

Fig. 33. Comparison between the mean length (±2 standard errors) of age-0 winter flounder taken during the late summer (August-September) by 1-m beam trawl in the Niantic River and in fall (October-December) by otter trawl at three TMP stations.

Fig. 32. Comparison between the late fall-early winter seasonal Δ-mean CPUE (solid line; ±2 standard errors) 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; ±2 standard errors) of winter flounder smaller than 15 cm for the 1976-2001 year-classes. (Note that the vertical scales differ between the graphs).

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 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). From 1985 through 2002, annual mean lengths of age-0 winter flounder taken by otter trawl from October through December (≤ 115 mm) were from 21 to 46 mm larger than those taken during August and September by 1-m beam trawl (Fig. 33). The annual pattern of mean lengths for the two gear types appeared similar, although they are not significantly related (Spearman's rank-order correlation; $r = 0.139$; $p = 0.581$; $n = 18$). The differences in observed mean lengths could have been achieved entirely by growth, although CPUE indices also may have been biased because smaller individuals were excluded from the TMP catch. Examination of length-frequencies of otter trawl-caught winter flounder from the TMP and the Niantic

River adult winter flounder survey indicated that even though an occasional small fish was taken, small winter flounder did not begin to regularly appear in the catches until they were about 40-50 mm in length. The otter trawl was probably not fully efficient until fish were about 55-60 mm in length. Therefore, lengths achieved by age-0 winter flounder each year (Table 17) may have differentially biased otter trawl CPUE because smaller means occurred more frequently after the mid-1990s than in earlier years. Finally, the fixed locations of the otter trawl sampling stations in relation to the habitat available to juveniles also may have affected the catch comparisons as well as the deletion of three offshore TMP stations.

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 95% CIs around the Δ-mean CPUE values were probably a consequence of this variation. In contrast, variation was usually less in data collected during summer by the relatively efficient 1-m beam trawl, which is also when age-0 winter flounder remain relatively sedentary. 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, which is discussed further below.

Age-1 Juveniles during Late Winter

Field Sampling and Data Analyses. In addition to catches made during the TMP, juvenile winter flounder smaller than 15 cm in length (mostly age-1) were taken along with adults in the annual February-April Niantic River adult spawning surveys, the methods for which were described previously. These fish were processed similarly as the adults, although gender was usually not specified and these fish were not branded. In years when small winter flounder were abundant, a subsample of at least 200 fish was measured each survey week; otherwise, all specimens were measured. For some annual comparisons of abundance, data were restricted to stations 1 and 2 in the lower river navigational channel (Fig. 2) because the distribution of small winter flounder generally varied more than the adult fish, and also because no tows were made in the upper river from 1977 through 1980. Annual Δ-mean CPUE indices of age-1 juveniles were determined as described previ-

ously for fish larger than 15 cm. One CPUE was computed for fish found at stations 1 and 2 combined and, when sufficient data were available, others were calculated for all river stations combined or for the upper river basin and arm (stas. 6 and 51-54). 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 and thereafter, two) TMP stations outside of the Niantic River during January through April, a period which overlapped the adult spawning surveys and also served to increase sample size. Annual effort for this abundance index included 45 trawl collections through 1995 and 18 in subsequent years.

Abundance and Distribution. The Δ-mean CPUE for age-1 juveniles taken in the navigational channel of the lower Niantic River during 2002 was 0.6, which, although twice the CPUE found in 2001, was still the second smallest abundance index found since 1976 (Table 21). Abundance of age-1 fish has been particularly low in the Niantic River navigational

TABLE 21. 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 2002 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	135	182	180	24.1	1.7	20.8 - 27.4
1977	6	206	264	256	25.6	2.5	20.7 - 30.6
1978	6	144	168	164	33.8	4.5	24.9 - 42.6
1979	5	118	135	133	65.1	9.7	46.1 - 84.1
1980	5	115	150	149	56.1	4.9	46.5 - 65.7
1981	7	97	154	154	86.2	7.9	72.8 - 99.5
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	73	77	76	25.3	3.0	19.4 - 31.3
1985	7	83	84	84	33.9	3.7	26.6 - 41.2
1986	7	75	126	115	5.9	0.6	4.8 - 7.0
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	66	91	88	14.5	2.0	10.6 - 18.3
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	42	54	50	6.3	1.1	4.2 - 8.4
1996	6	49	60	38	1.6	0.3	1.0 - 2.2
1997	6	43	48	41	3.2	0.5	2.2 - 4.2
1998	7	56	63	47	2.5	0.3	1.8 - 3.1
1999	7	50	63	39	1.0	0.2	0.7 - 1.3
2000	6	42	48	24	1.0	0.2	0.5 - 1.4
2001	7	41	63	16	0.3	0.1	0.1 - 0.4
2002	7	50	63	26	0.6	0.1	0.4 - 0.8

[a] Catch per standardized tow (see text for details).
[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.

channel during the past 7 years. The presence of a heavy blue mussel set at station 1 in 2001 and the remnant shell deposits in 2002 most likely affected the presence of juveniles in this area as it did adults. This year the median CPUE of juveniles in the upper river was 1.6, which represents the first decrease seen in this abundance index since 1996 (Table 22).

Distribution of juvenile winter flounder during winter largely influences their availability to sampling and apparently differs from year to year, probably as a result of variable environmental conditions, including water temperature, winter storm events, and changes in bottom composition, such as the mussel set. The relative abundance of small winter flounder has also neither been consistent between Niantic Bay and Niantic River from year to year nor within areas of the river. A Δ-mean CPUE computed for winter flounder smaller than 15 cm taken by the TMP from January through April at stations outside of the Niantic River was compared to the Δ-mean CPUE for fish found in the navigational channel of the lower river during the spawning season (Fig. 34). Over the entire time-series, the catch of age-1 winter flounder

in winter and early spring has fluctuated less outside than inside the Niantic River, although the CPUE values in the bay had higher 95% CI because of much smaller sample sizes. However, as noted above, since 1996 the CPUE in the lower river navigational channel has remained between 0.3 and 3.2. As the number of age-1 winter flounder in the lower river declined in recent years, relative abundance of these fish in Niantic Bay increased. The CPUE of fish found in the bay during January through April was greater than that of fish taken in the lower river navigational channel in most years since 1986, including 2002. The catch outside the river during 1995 was the highest of this time-series, indicating that most fish from the relatively strong 1994 year-class did not remain within or re-enter the Niantic River during the adult spawning season. Smaller peaks were seen in the bay catches during 1989, 1993, and 1997, which corresponded to relatively good year-classes produced the previous summer and fall.

Since the mid-1980s, annual effort (as measured by the adjusted number of tows used for calculation of

TABLE 22. 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 (sta. 1 and 2; Fig. 2) with those caught in the upper river arm and basin (sta. 6 and 51-54; Fig. 2) during the 1981 through 2002 adult population abundance surveys.

	Lower river navigational channel			Upper river arm and basin		
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
1981	154	154	86.2	96	89	7.9
1982	70	70	57.4	95	93	7.6
1983	77	76	52.5	140	133	9.9
1984	77	76	25.3	182	169	5.3
1985	84	84	33.9	161	155	10.3
1986	126	115	5.9	245	222	5.9
1987	50	44	6.6	230	185	3.4
1988	54	52	17.0	270	238	4.8
1989	56	50	10.6	264	242	8.9
1990	91	88	14.5	272	197	2.0
1991	60	56	7.4	282	217	1.8
1992	49	44	11.9	378	303	3.3
1993	49	45	6.6	294	230	2.9
1994	24	24	5.6	196	145	2.6
1995	54	50	6.3	300	191	1.3
1996	60	38	1.6	298	113	0.6
1997	48	41	3.2	300	191	1.1
1998	63	47	2.5	322	229	1.6
1999	63	39	1.0	308	201	1.9
2000	48	24	1.0	291	168	2.1
2001	63	16	0.3	326	208	2.2
2002	63	26	0.6	350	232	1.6

[a] Catch per standardized tow (see text for details).
[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.
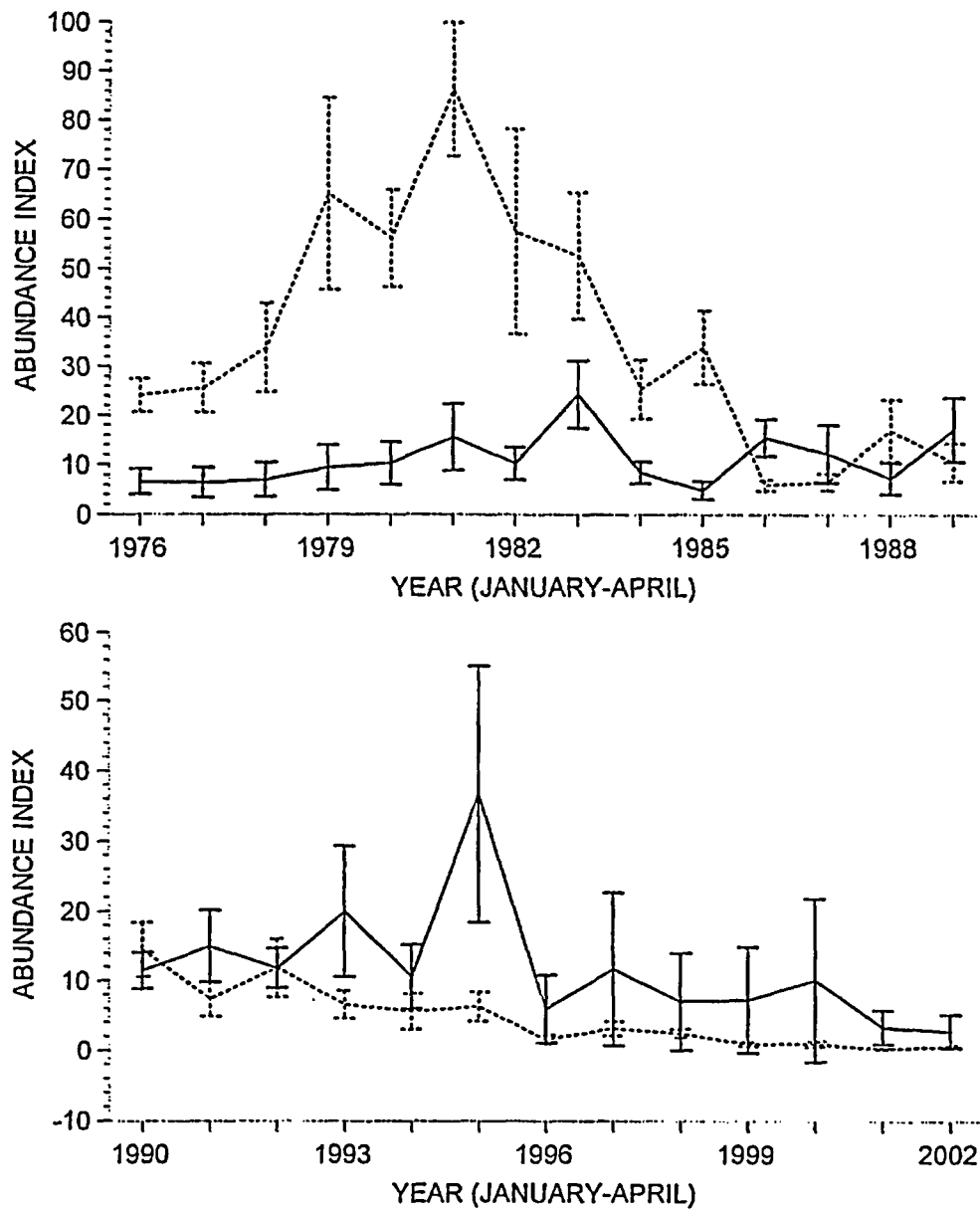
Fig. 34. Comparison between the annual January-April Δ-mean CPUE (solid line; ±2 standard errors) at all trawl monitoring program stations except NR and the Niantic River (stations 1 and 2) spawning survey Δ-mean CPUE (dashed line; ±2 standard errors) of winter flounder smaller than 15 cm from 1976 through 2002. (Note that the vertical scales differ between the graphs).

CPUE) in the upper Niantic River basin and arm (stations 6, 51-54) has increased relative to effort expended in the lower river navigational channel (stations 1 and 2), particularly since abundance in the latter area has decreased during the past decade. There has been some concern that the sharp decrease in abundance following the 1981 peak in Δ-mean CPUE of age-1 winter flounder taken in the lower river navigational channel was an early indication of subsequent poor recruitment to the Niantic River

adult spawning population. Although most of the decrease occurred prior to Unit 3 going online, a question also remained whether or not a plant entrainment effect contributed to this decrease. Since both sampling effort and abundance of age-1 juvenile winter flounder in this small area of the river has decreased from the earliest years of study, another abundance index was calculated for age-1 fish. A Δ-mean CPUE was computed for winter flounder <15 cm that were caught in the upper Niantic River basin

and arm from 1981 through 2002 (Table 22). Insufficient effort was made in this area prior to 1981 for a meaningful calculation. After 1989, a decrease was also observed in the upper river Δ-mean CPUE time-series, but the magnitude of the decline was considerably less than seen in the lower river (Fig. 35). Although remaining lower than it was in the 1980s, CPUE increased steadily from the lowest value found in 1996 through 2001, but decreased again in 2002. As with the lower river CPUE, the upper river abundance may be compared with CPUE from the TMP stations outside of the Niantic River from January through April (same index as shown on Fig. 34). Age-1 winter flounder remained more abundant outside the Niantic River relative to the upper Niantic River for nearly all these years.

A small CPUE for age-1 winter flounder taken within the lower Niantic River may not reflect the abundance of the corresponding year-class, which may be more accurately described by age-0 indices discussed in the two previous sections. A relatively small increase in catch of age-1 fish in the much larger geographical area of Niantic Bay and nearby LIS could have accounted for seemingly low abundance in the river as fish dispersed from a relatively limited and confined area into a much larger geographical space. Similarly, the upper Niantic River basin and arm offers considerably more spatial area for juvenile winter flounder, as do shallow areas within the river not sampled during the spawning

surveys. Because of the differential distribution and abundance of age-1 juveniles, perhaps as a cones-quence of variable environmental conditions influenc-ing 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. Finally, further adjustments to year-class strength occur in the remaining several years of juvenile life. The dispersion of older juvenile winter flounder over large areas of LIS and adjacent estuarine systems that allows for considerable mixing of fish from many spawning stocks makes the task of following a specific cohort from a particular spawning population very difficult. The relationship between juvenile and adult abundance indices is explored further in the following section.

Comparisons among Life-Stages of Winter Flounder Year-Classes

Abundance indices of various life-stages of the 1976 through 2002 year-classes of Niantic River winter flounder found throughout this report are summarized in Table 23. Coefficients of variation (CV) computed for most of these indices were used to compare annual variability in abundance among years (Table 24); most CVs increased from those given in DNC (2002), indicating that variability in abundance
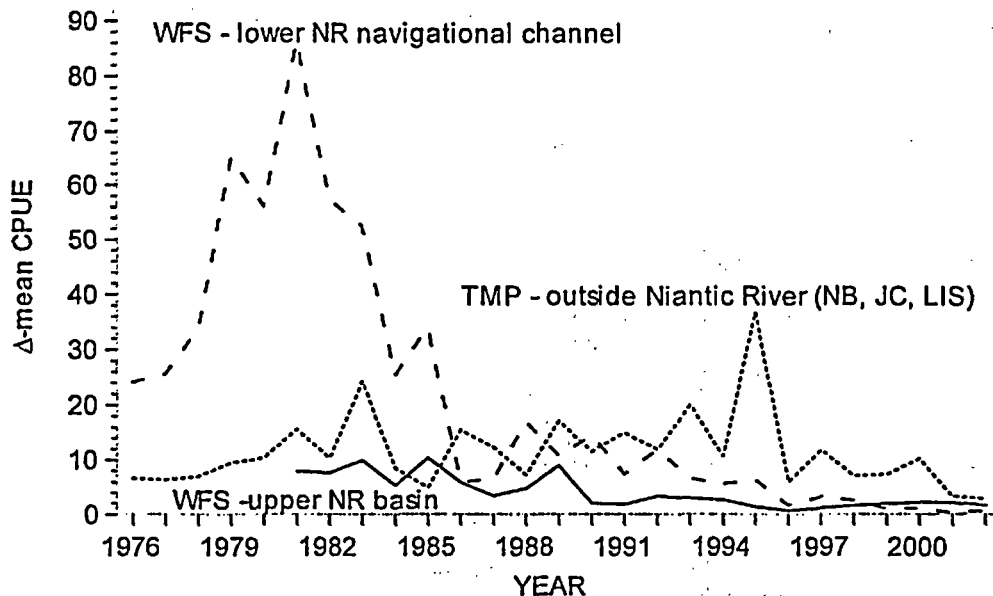
Fig. 35. Comparison of Δ-mean CPUE of age-1 winter flounder taken in the navigational channel (stations 1 and 2) of the lower Niantic River and in the upper river basin (stations 6, 51-54) during the spawning survey (WFS) and the January-April Δ-mean CPUE of age-1 winter flounder taken at all trawl monitoring program (TMP) stations except NR from 1976 through 2002.

TABLE 23. Comparison of abundance indices of various winter flounder life-stages from the 1976 through 2002 year-classes.

	Adult abundance indices		Larval abundance indices					Juvenile abundance indices				
	Niantic River							Age-0	Age-0	Age-0	Age-1	Age-1
	female	annual	Niantic River stations (Feb-Jun)				MPS	Niantic	Niantic	River/bay	Δ-mean	Δ-mean
Year-	spawners	egg	Stage 1	Stage 2	Stage 3	Stage 4	(EN)	R. stas.	R. stas.	Δ-mean	(lower river)	(upper river)
class	(Feb-Apr)	prod.	(3 mm)	(3.5 mm)	(6 mm)	(7.5 mm)	(≥7 mm)	(May-Jul)	(Aug-Sep)	(Nov-Feb)	(Feb-Apr)	(Feb-Apr)
1976	-	-	-	-	-	-	854	-	-	6.1	25.6	-
1977	1,069	481.4	-	-	-	-	567	-	-	5.1	33.8	-
1978	1,644	828.3	-	-	-	-	754	-	-	4.2	65.1	-
1979	1,197	565.7	-	-	-	-	641	-	-	4.2	56.1	-
1980	1,004	468.9	-	-	-	-	845	-	-	10.1	86.2	7.9
1981	2,411	1,252.5	-	-	-	-	561	-	-	7.7	57.4	7.6
1982	2,551	1,479.8	-	-	-	-	610	-	-	19.6	52.5	9.9
1983	1,705	985.5	-	749	408	56	1,215	-	-	6.6	25.3	5.3
1984	822	474.9	2,601	1,501	573	67	917	-	7.5	7.4	33.9	10.3
1985	846	514.9	6,260	4,676	584	35	312	15.0	8.0	8.1	5.9	5.9
1986	604	402.5	1,279	176	301	24	510	26.7	15.6	11.7	6.6	3.4
1987	780	485.9	3,218	829	1,036	48	315	41.7	13.1	4.8	17.0	4.8
1988	1,172	792.8	14,491	4,469	1,531	210	419	52.5	51.7	29.6	10.6	8.9
1989	891	647.1	12,463	3,976	589	73	327	12.5	7.8	11.3	14.5	2.0
1990	498	314.4	4,728	355	258	57	508	117.5	18.0	21.7	7.4	1.8
1991	963	578.7	3,248	252	343	112	439	45.0	24.2	19.0	11.9	3.3
1992	485	354.2	5,476	1,367	2,339	195	1,003	77.1	29.4	31.1	6.6	2.9
1993	247	201.6	1,187	133	111	6[a]	130	7.3	5.0	7.4	5.6	2.6
1994	458	297.3	3,692	1,248	429	90	834	126.3	52.1	31.7	6.3	1.3
1995	192	148.1	5,580	2,023	2,615	787	1,804	111.3	28.3	4.8	1.6	0.6
1996	88	74.3	4,376[a]	4,424	152	31	462	14.2	4.5	18.3	3.2	1.1
1997	166	131.8	8,342	8,449	4,978	202	2,904	28.8	8.4	2.3	2.5	1.6
1998	118	89.1	3,138	6,553	1,550	393	1,509	46.3	14.4	10.5	1.0	1.9
1999	107	75.0	2,632	288	584	73	1,420	43.8	18.5	15.9	1.0	2.1
2000	167	123.0	408	403	444	66	1,134	38.8	35.0	11.6	0.3	2.2
2001	68	56.1	1,934	704	140	66	837	150.0	33.8	1.5	0.6	1.6
2002	129	103.5	3,564	1,400	149	79	229	56.3	15.0	-	-	-

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

TABLE 24. Coefficients of variation (CV) of annual abundance indices[a] of various life stages of Niantic River winter flounder.

Life stage	Abundance index used	Number of observations	CV
Female spawners	Annual standardized catch	26	88%
Age-3 females	Annual standardized catch	24	127%
Age-4 females	Annual standardized catch	23	106%
Age-5 females	Annual standardized catch	22	105%
Eggs	Egg production index	26	80%
Stage 1 larvae	A parameter of Gompertz function	19	78%
Stage 2 larvae	A parameter of Gompertz function	20	109%
Stage 3 larvae	A parameter of Gompertz function	20	125%
Stage 4 larvae	A parameter of Gompertz function	20	133%
Age-0 young	Median CPUE in Niantic River (May-July)	18	76%
Age-0 young	Median CPUE in Niantic River (August-Sept)	19	70%
Age-0 young	Fall-winter Δ-mean at trawl stations	26	73%
Age-1 juveniles	Δ-mean CPUE of fish < 15 cm in lower Niantic River	26	116%
Age-1 juveniles	Δ-mean CPUE of fish < 15 cm in upper Niantic River	22	75%

[a] Indices used correspond to those given on Table 23, except for age-3 through age-5 females.

has increased. Total adult female abundance (CV = 88%) and egg production estimates (80%) were among the least variable of the abundance indices. Considering the first three adult female age-classes, variability decreased from age-3 (127%) to ages-4 (106%) and 5 (105%). This likely reflected not only variation in recruitment of year-classes, but variable numbers of immature fish of age-3 fish that were present in the river each year. This is also consistent with findings of Miller et al. (1991), who noted that interannual variability in the abundance indices of many flatfishes appeared to decrease with age.

Stage 1 larval abundance appeared to be the most stable (CV = 78%) of the early life history stages and had about the same variation as that found for estimates of egg abundance. Abundance was more variable for Stage 2 (109%) and Stage 3 (125%) larvae. Stage 4 larval abundance showed the largest annual variation (133%), as relatively high densities of this developmental stage were seen during several years and an extremely low value was found in 1993 (Table 23). Following metamorphosis, annual variability of newly settled juvenile winter flounder abundance decreased to 76%, a value similar to the CVs for egg production and Stage 1 larvae. Variability decreased slightly further to 70-73% in both late summer and late fall and early winter, after young leave shallow inshore waters. These life stages showed the smallest variability in abundance. If the abundance of age-1 juveniles found only in the upper river during the adult spawning surveys is considered, the CV is an approximately similar 75%. However, a larger CV of 116% computed for age-1 juveniles taken in the navigational channel of the lower Niantic River was probably related to the previously discussed annual differences in distribution related to behavior as much as from actual variation in year-class strength. Considerable variation occurs during the post-settlement juvenile life history stage. In some instances, compensation likely takes place in a density-dependent fashion, whereas other changes in abundance occur independently of density.

Rothschild and DiNardo (1987) reported a median CV of 70% for recruitment indices of various marine fishes, although various flatfishes had CV values mostly less than 75%. This value is consistent with those found for 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 European 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%), variability that is less than found for winter flounder. He attributed the decline in variation of abundance in older juveniles to a density-dependent regulatory mechanism that operated during and shortly after larval settlement.

Relationships among abundance indices of winter flounder for the same year-class are of interest for impact assessment (Tables 25 and 26). 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 spawning females and egg production were highly correlated, which was expected because calculation of the latter included adult female abundance as part of the methodology of estimation. Significant correlations were also found between the abundances of successive larval stages as well as between Niantic River Stage 4 larval abundance and age-0 juveniles collected in the Niantic River during both early and late summer, although considerable scatter was seen in these relationships (Fig. 36). Early and late summer age-0 juvenile abundances were correlated, but the late summer index and late fall-early winter abundance are no longer significantly correlated. Annual density of larger (≥7 mm) larvae collected in entrainment samples at MPS was significantly correlated with Stages 3 and 4 larval abundance, although with considerable scatter (Fig. 37). The abundance of these pre-metamorphosed larvae was not correlated with other larval, juvenile, or adult abundance indices. As discussed previously, the abundance of age-1 winter flounder taken during the Niantic River spawning survey in the navigational channel of the lower river was not significantly correlated with that of young taken during the previous fall and early winter in the TMP (Fig. 32) and was probably mostly related to changes in distribution.

Most Niantic River female winter flounder are not fully recruited to the spawning population until about age-4. Thus, abundance of age-3 spawning females in particular probably was not indicative of year-class strength because only a fraction of these fish were

TABLE 25. 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 MPS discharge.

Index[a]	Adult egg production	Stage 1 larvae	Stage 2 larvae	Stage 3 larvae	Stage 4 larvae	Larvae (≥7 mm) at MPS discharge
Female spawners	0.9699[b] 0.0001 ** 26	0.3544 0.1366 NS 19	-0.0406 0.8650 NS 20	0.1587 0.5039 NS 20	-0.0707 0.7670 NS 20	-0.2041 0.3173 NS 26
Adult egg production		0.4053 0.0852 NS 19	0.0015 0.9950 NS 20	0.2099 0.3745 NS 20	-0.0572 0.8107 NS 20	-0.2766 0.1714 NS 26
Stage 1 larvae			0.6719 0.0016 ** 19	0.5406 0.0169 * 19	0.4662 0.0442 * 19	-0.0614 0.8028 NS 19
Stage 2 larvae				0.6078 0.0045 ** 20	0.4680 0.0374 * 20	0.1684 0.4778 NS 20
Stage 3 larvae					0.6530 0.0018 ** 20	0.4919 0.0276 * 20
Stage 4 larvae						0.4853 0.0301 * 20

[a] Indices used correspond to those given on Tables 23 and 24.
[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 ≤ 0.05, ** - significant at p ≤ 0.01), and
number of annual observations (sample size).

present on the spawning grounds each year. Furthermore, the presence of immature fish in the river may vary from year to year because of environmental conditions. Females of ages-3 through 5 were significantly positively correlated with age-1 juveniles taken in both the lower and upper river during the spawning season, but age-4 and 5 fish were negatively correlated with age-0 fish taken during fall and early winter (Table 27). However, the form of these relationships is unclear (Fig. 38; only lower river shown) and because of the scatter seen in the data plots, these relationships may not be very meaningful. Persistence of some negative correlations among abundance indices perhaps results from unknown processes operating after winter flounder become age-1 that produce fewer adult recruits from more abundant year-classes of juveniles. This was also noted by Simpson et al. (1996) for the 1988 and 1992 year-classes of winter flounder in LIS and by Desfosse et al. (1998) during recent years in Massachusetts as abundant numbers of age-1 and 2 fish did not appear as such at age-3 and older. Explanations suggested include variable discard mortality of juveniles in the commercial fishery; high rates of fishing at recruitment to the fisheries

(Simpson et al. 1996; Desfosse et al. 1998); and non-random fishing effort, which may occur in overfished stocks. Meanwhile, until explanatory factors are identified, none of these life-stage indices can be used as a reliable measure of future year-class strength, a discussion of which is given further below in a section discussing the formation of winter flounder year-class strength.

Stock-Recruitment Relationship (SRR)

An SRR described by Ricker (1954, 1975) provides the basis for calculating survival from egg to immature fish during the first year of the life cycle, which is used with a population dynamics simulation model of Niantic River winter flounder (see Stochastic Simulation of the Niantic River Winter Flounder Stock, below). The stock and recruitment data for determining the SRR form were derived from the catch-at-age of female winter flounder taken 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

TABLE 26. Matrix of Spearman's rank-order correlations among various larval and juvenile winter flounder abundance indices.

Index[a]	Niantic River Stage 4 larvae	Niantic River early summer age-0 juveniles	Niantic River late summer age-0 juveniles	Fall-early winter river-bay age-0 juveniles	Niantic River winter-spring age-1 juveniles (lower river)	Niantic River winter-spring age-1 juveniles (upper river)
Larvae (≥7 mm) at MPS discharge	0.4853[b] 0.0301 * 20	0.3581 0.1445 NS 18	0.3298 0.1679 NS 19	-0.2087 0.3063 NS 26	-0.2397 0.2382 NS 26	-0.2700 0.2243 NS 22
Niantic River Stage 4 larvae		0.5196 0.0271 * 18	0.4609 0.0470 * 19	0.1265 0.6057 NS 19	-0.1472 0.5476 NS 19	-0.2481 0.3057 NS 19
Niantic River early summer age-0 juveniles			0.7606 0.0002 ** 18	0.2428 0.3477 NS 17	-0.1153 0.6594 NS 17	-0.2857 0.2663 NS 17
Niantic River late summer age-0 juveniles				0.3946 0.1051 NS 18	-0.2345 0.3489 NS 18	-0.1115 0.6596 NS 18
Fall-early winter river-bay age-0 juveniles					-0.1283 0.5322 NS 26	0.0435 0.8475 NS 22
Niantic River winter-spring age-1 juveniles (lower river)						0.7228 0.0001 ** 22

[a] Indices used correspond to those given on Tables 23 and 24.
[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 ≤ 0.05, ** - significant at p ≤ 0.01), and
   number of annual observations (sample size).

Horwood 1977). Therefore, the index of annual parental stock size was based on derived egg production and the index of recruits or year-class size was based on calculated egg production accumulated over the life-time of the recruits. This method accounted for variations in year-class strength and in fecundity by size and age. The assumptions and methods used to age Niantic River winter flounder and to calculate a recruitment index expressed as equivalent numbers of spawning females 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, above). A recruitment index was determined by applying an age-length key described in NUSCO (2000) to the annual standardized catches of females partitioned into length categories. Aging females allowed for the determination of abundance by year-class present in each age-class during

successive spawning seasons. 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 during that period.

From observations made of abundance and age over the years, a large fraction of age-3 females and some age-4 fish were apparently immature and not present in the Niantic River during the spawning season; all females age-5 and older were assumed to be mature. 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 (Fig. 10). Because the estimates of age-3 fish present in the Niantic River were thought to be unreliable as a large fraction of them were immature and not present on the spawning grounds, this estimation process was only carried through the 1998 year-class (i.e., age-4 females taken in 2002). The adjusted numbers of mature fish provided an index of the fully recruited year-class expressed as the
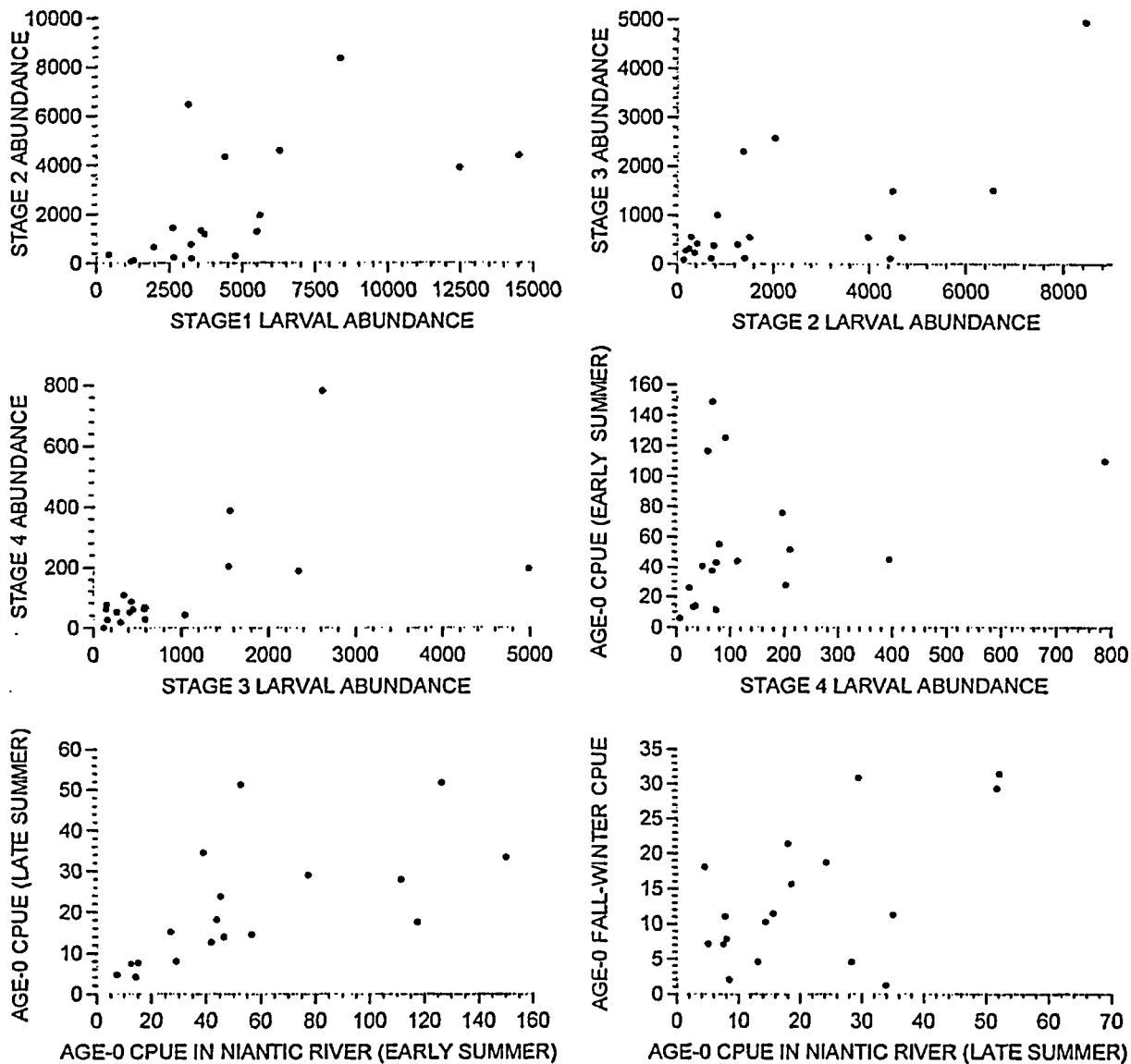
Fig. 36. Comparison between the abundance indices of successive winter flounder early life history stages in the Niantic River (varies among year-classes from 1983 through 2002, depending upon data availability), including larval abundance (A parameter of Gompertz function), age-0 juveniles in the Niantic River (early and late seasonal 1-m beam trawl median CPUE at stations LR and WA combined); and the late fall-early winter seasonal Δ-mean CPUE at all trawl monitoring program (TMP) stations. (Note that the vertical and horizontal scales differ among the graphs).

aggregated number of adult females 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, most of which did not enter the river until fully recruited to the spawning stock. Although this recruitment index could be used together with the annual number of adult females to derive the 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 (Eq. 1). Also, since the recruitment index was based on total lifetime contribution of a year-class, estimates from more recent year-classes needed to be projected into the future. Therefore, an annual survival rate of 0.39, which is equivalent to a total instantaneous total

STAGE 4 LARVAL ABUNDANCE INDEX

STAGE 3 LARVAL ABUNDANCE
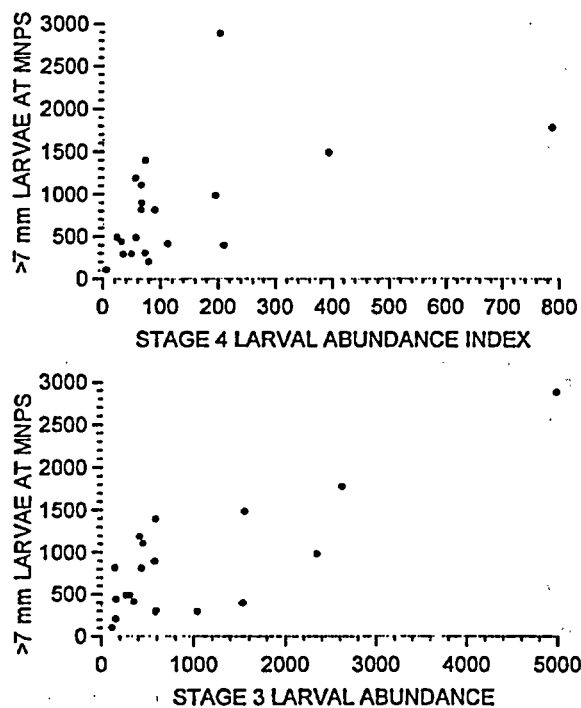
Fig. 37. Comparison between the abundance of Stages 3 and 4 winter flounder larvae taken in the Niantic River (*A* parameter of Gompertz function; 1983-2002) and larvae ≥7 mm taken at station EN (MPS entrainment sampling). (Note that the horizontal scales differ between the graphs).

mortality rate (Z) of 0.94 (the current estimated instantaneous fishing mortality rate, F, for LIS winter flounder of 0.74 plus 0.2 for M), was applied to generate estimates of year-class egg production through 2010 (i.e., age-12 fish from the 1998 year-class. Finally, annual egg production was summed up over the lifetime of each year-class to determine a recruitment index as eggs, which was then converted to equivalent spawning females at the rate of one female for each 585,167 eggs, which is the calculated value of current mean fecundity for the Niantic River stock.

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). This particular form of a SRR has also been applied to other New England flounder stocks (Gibson 1989, 1993). Furthermore, Iles (1994) stated that a dome-shaped stock-recruitment relationship appeared to be generally consistent for a number of flatfish stocks and Brodziak et al. (2001) reported that Southern New England winter flounder demonstrated a density-dependent form of a SRR. The mathematical form of the Ricker SRR is:

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

TABLE 27. Matrix of Spearman's rank-order correlations among various winter flounder larval and Niantic River female spawner abundance indices.

Index[a]	Larvae (≥7 mm) at MPS discharge	Niantic River early summer age-0 juveniles	Niantic River late summer age-0 juveniles	Fall-early winter river-bay age-0 juveniles	Niantic River winter-spring age-1 juveniles (lower river)	Niantic River winter-spring age-1 juveniles (upper river)
Age-3	-0.1226[c]	-0.2321	-0.0735	-0.3498	0.8121	0.8030
female	0.5682 NS	0.4051 NS	0.7867 NS	0.0938 NS	0.0001 **	0.0001 **
spawners[b]	24	15	16	24	24	20
Age-4	-0.1700	-0.2176	-0.2571	-0.4415	0.8120	0.7719
female	0.4382 NS	0.4549 NS	0.3549 NS	0.0349 *	0.0001 **	0.0001 **
spawners[b]	23	14	15	23	23	19
Age-5	0.2106	-0.1374	-0.1033	-0.4261	0.7145	0.7090
female	0.3468 NS	0.6545 NS	0.7253 NS	0.0480 *	0.0002 **	0.0010 **
spawners[b]	22	13	14	22	22	18

[a] Early life history indices used correspond to those given on Tables 23 and 24.
[b] Determined by applying an age-length key (see Figure 4) 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 ≤ 0.05, ** - significant at p ≤ 0.01), and
number of annual observations (sample size).
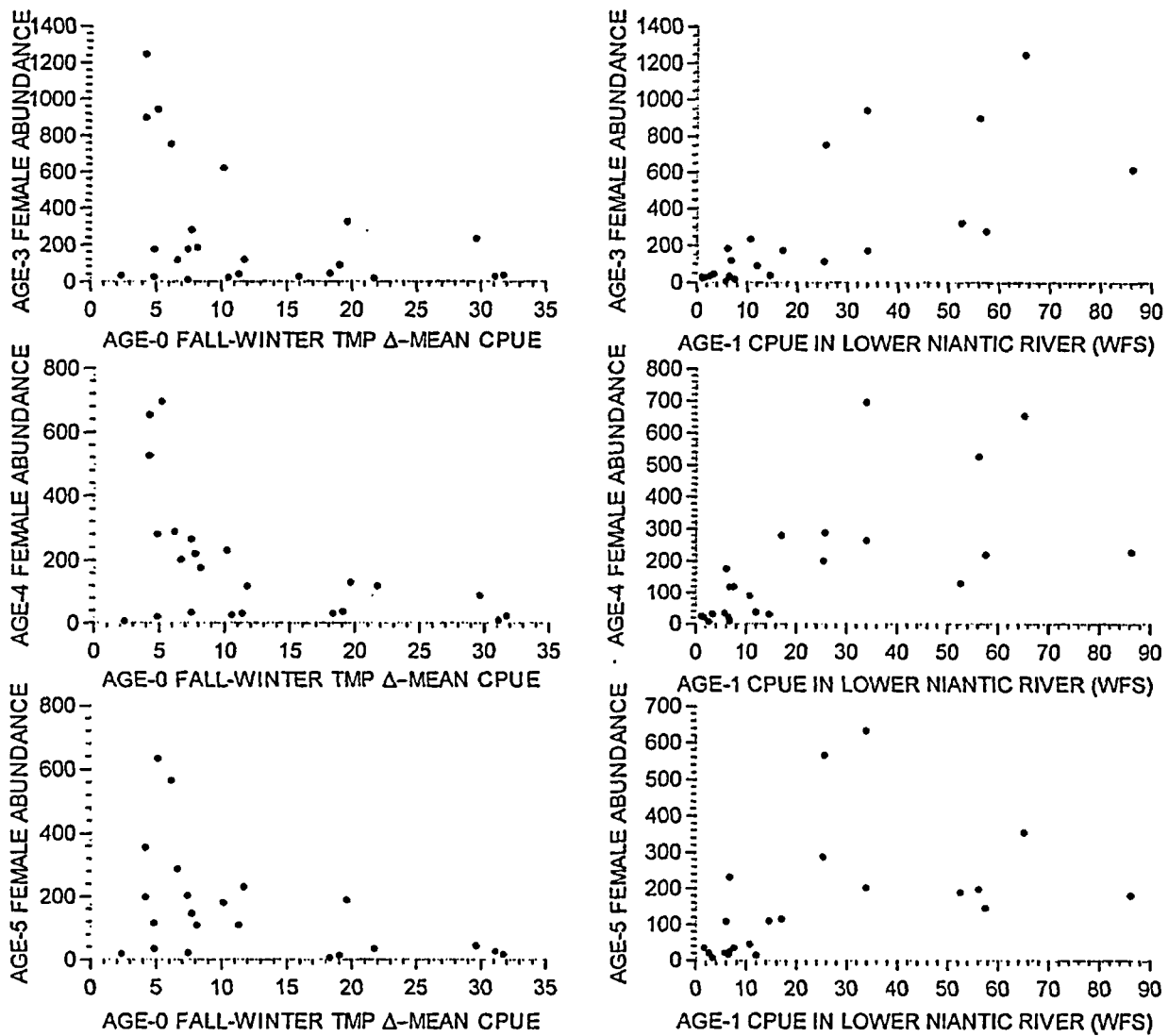
Fig. 38. Comparisons between the late fall-early winter seasonal Δ-mean CPUE of age-0 winter flounder taken at trawl monitoring program stations (TMP) and the Δ-mean CPUE of age-1 winter flounder taken in the lower Niantic River (stations 1 and 2) during the adult winter flounder surveys (WFS), and the relative annual abundance of age-3 (1976-99 year-classes), age-4 (1976-98 year-classes), and age-5 (1976-97 year-classes) Niantic River female winter flounder taken during the spawning season. (Note that the vertical scales differ among the graphs).

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. Using nonlinear regression methods (SAS Institute Inc. 1990d), 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 influencing the SRR. The natural logarithm of winter flounder recruitment is negatively correlated with mean water temperature during February at the intakes of MPS (Spearman's rank-order correlation; $r = -0.510$; $p = 0.015$; $n = 22$), a month during which most spawning and early larval development occurs (NUSCO 1988b, 1989). Following Lorda and Crecco (1987), Gibson (1987, 1993), Iles and Beverton

(1998), Quinn and Deriso (1999: p. 91), and Planque and Frèdou (1999), the February mean water temperature was used as an explanatory variable added to Equation 6 to adjust the two-parameter SRR for temperature effects. This served to reduce recruitment variability and obtain more reliable parameter estimates for the SRR. The resulting temperature-dependent SRR had the form:

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

where the second exponential describes the effect of February water temperature on recruitment and the added 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 (1977-98) 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. Temperature and climatic effects influencing winter flounder reproductive success are also discussed in the following section on the formation of winter flounder year-class strength.

In additional to the standard SRR models, a depensatory form of SRR was investigated. Coincidentally, Walters and Kitchell (2001) recently emphasized that the risk of depensatory effects should be a goal of recruitment research. When depensation processes occur, the per capita recruitment rate decreases with decreasing parental stock size, which may lead to a rapid stock collapse. As suggested by Saila and Lorda (1982), the two-parameter Ricker SRR (Eq. 6) was modified with the addition of another parameter, Ω, and the depensatory version of the Ricker SRR has the form:

$$R_t = \alpha \times (P_t)^{\Omega} \times \exp(-\beta \times P_t) \quad (8)$$

where the parameter Ω must be greater than zero. When $\Omega > 1$, the SRR can describe both depensation below some threshold stock size and compensation for larger stock sizes, as in the two-parameter SRR (Eq. 6). When $\Omega = 1$, the above equation reverts to the standard two-parameter model. When $0 < \Omega < 1$, the SRR curve does not have a domed shape and, thus, it would not be a good description of winter flounder recruitment.

Nonlinear regression methods (SAS Institute Inc. 1990d) were used for estimating the parameters in the above equations. The value of α as estimated from these stock and recruitment data was not used in the SPDM, which uses a value, termed α_0, calculated from life history parameters, the derivation of which is discussed below.

Biological Reference Points. The fishing mortality rate (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 (M = 0.2; total instantaneous mortality rate Z = F + M) 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 u) is obtained as:

$$u = (F / Z) \times (1 - \exp[-Z]) \quad (9)$$

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:

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

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

$$F = \log_e(\alpha) - (\beta \times P_{E(F)}) \quad (11)$$

When F = 0, Equation 10 becomes the equilibrium or replacement level of the unfished stock:

$$P_{rep} = (\log_e[\alpha]) / \beta \quad (12)$$

The fishing rate for "recruitment overfishing" was 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). This concept is discussed in more detail below (see Stochastic Simulation of the Niantic River Winter Flounder Stock).

Although Equations 10 through 12 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 M and F.

Sampling-based Estimates. Egg production estimated from annual spawning surveys was the basis for determining recruitment, because the abundance of other early life-stages have not been reliably correlated with adult winter flounder. Both parental spawning stock size (previously described in Absolute Abundance Estimates) and recruitment indices were scaled to absolute population size using annual estimates of abundance, size frequency, maturity, egg production, and overall population mean fecundity. These scaled annual values were used with the Ricker SRR model as estimates of adult female spawning stock and potential female recruitment (Table 28). The addition of new catch data from the 2002 adult winter flounder survey resulted in some differences among current estimates of spawning females and recruits and those previously reported since an SRR was first described in NUSCO (1990). Trends observed in the SRR time-series of data include a substantial reduction in annual female

recruitment from the 1970-80s and, with a few exceptions, warmer February water temperatures than the long-term average. The trend in February mean water temperature was significantly positive from 1977 through 2002 (Mann-Kendall test, $Z = 3.15$; n = 26; p = 0.002).

A two-parameter SRR model (Eq. 6) was initially fitted to the data. The stock growth potential parameter α (scaled as numbers of fish) for this model was estimated as 1.076 with a standard error of 0.323 (30% of the parameter value). This estimate of α and the estimate of β (the second model parameter) were used as initial values for fitting the three-parameter SRR model with temperature effects (Eq. 7) and the depensatory model (Eq. 8).

Fitting the depensatory model to the same data resulted in a parameter estimate for Ω that was not significantly different from 1. Therefore, a depensatory form of the SRR was not indicated for the Niantic River winter flounder population. Myers et al. (1995) examined spawner-recruit data for 128 fish stocks and reported that evidence for depensation was only indicated in three populations. They concluded

TABLE 28. Annual Niantic River winter flounder stock-recruitment data based on indices of egg production in the 1977 through 1998 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	24,927	52,270	2.09	0.36	-2.51
1978	42,892	43,351	1.01	1.09	-1.78
1979	29,293	36,247	1.23	1.48	-1.39
1980	24,280	31,464	1.29	2.38	-0.49
1981	64,863	28,533	0.44	2.63	-0.24
1982	76,632	28,382	0.37	1.56	-1.31
1983	51,033	27,739	0.54	3.74	0.87
1984	24,595	24,546	1.00	4.02	1.15
1985	26,662	20,438	0.77	2.36	-0.51
1986	20,842	18,637	0.89	3.38	0.51
1987	25,161	16,049	0.64	3.27	0.40
1988	41,057	11,384	0.28	2.67	-0.20
1989	33,510	8,242	0.25	3.24	0.37
1990	16,279	7,158	0.44	4.28	1.41
1991	29,970	5,436	0.18	4.76	1.89
1992	18,341	4,328	0.24	3.68	0.81
1993	10,437	4,047	0.39	3.10	0.23
1994	15,398	3,689	0.24	1.59	-1.28
1995	7,670	3,198	0.42	4.11	1.24
1996	3,847	2,332	0.61	2.12	-0.75
1997	6,829	1,703	0.24	3.20	0.33
1998	4,612	1,262	0.27	4.09	1.22
Mean	27,233	17,293	0.64	2.87	
CV	69%	87%		40%	

[a] Scaled number of female spawners and recruits from expected egg production; scaling factors used were 585,167 eggs per female (mean fecundity) and a multiplier of 30.3 to convert relative abundance to an absolute population size. Indices of female spawners and recruits differ from those reported in DNC (2002) because of data added from the 2001 adult winter flounder population survey and small changes made to the scaling factors.

that most observed fish population collapses could not be reasonably attributed to depensatory fish population dynamics. In another meta-analysis, Liermann and Hilborn (1997) found that no depensation was likely for fishes of four broad taxonomic groups, including pleuronectid flounders, but cautioned that both depensation and hypercompensation were possible for some stocks, given the broad range of their data distribution. In a simulation analysis, Frank and Brickman (2000) found that a SRR can appear to be compensatory even if the stock reproductive dynamics did not exhibit this behavior. They further noted, however, that species exhibiting social behavior, such as group mating or schooling were at most risk for depensation. Winter flounder do not exhibit these traits, which are mostly found in pelagic fishes. Also, spawning in inshore estuaries may protect winter flounder from fisheries directed specifically at spawning aggregations. However, trawlers target fish moving to or from the spawning grounds in fall and winter and spring, including those within Niantic Bay. Furthermore, the Niantic River has had a unique night spear fishery for winter flounder that occurs during the spawning and immediate post-spawning periods, which can be considered as a source of increased fishing mortality

specific to this stock. In recent years, fishing mortality has apparently remained relatively high on LIS winter flounder (Johnson et al. 2000) and the entire South New England-Middle Atlantic complex of winter flounder stocks (NEFSC 2002).

For the three-parameter SRR model with temperature effects, all parameter estimates were significantly different from 0 and the model explained 74% of the variability associated with the recruitment index. Relationships resulting from fitting both the two- and three-parameter Ricker models separately to stock and recruitment data scaled to absolute population sizes are shown as the curved lines in the central portion of Figure 39 as follows: the unadjusted SRR (two-parameter model; Eq. 6) is shown as the more widely-spaced dashed line and the three-parameter model (SRR adjusted for T_{feb}; Eq. 7) is represented by the solid line. The outermost two more finely-spaced dashed lines illustrate low recruitment in the warmest year (Table 28; 1991, T_{Feb} = +1.89) and high recruitment in the coldest year (1977, T_{Feb} = -2.51). A similar plot, showing the effect of a range in water temperatures on the SRR of Irish Sea Atlantic cod (*Gadus morhua*), was shown by Planque and Frèdou (1999).
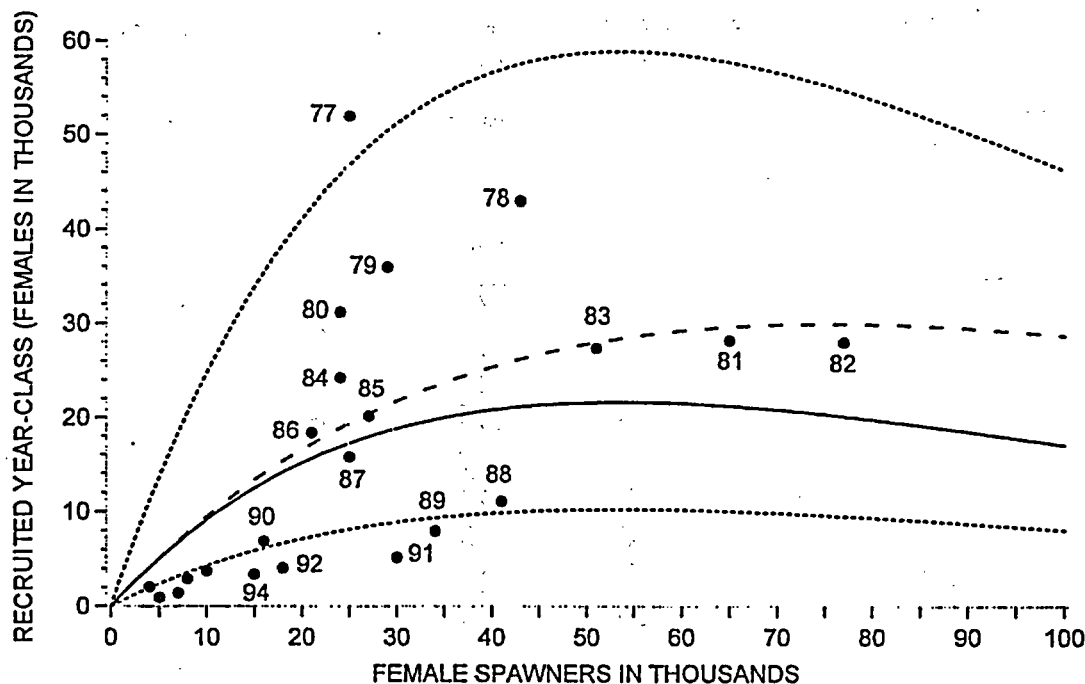
Fig. 39. Ricker SRRs of Niantic River winter flounder (see text for explanation of the four curves plotted). Calculated recruitment indices (see Table 28) of the 1977 through the 1998 year-classes are shown. The five plotted points closest to the origin include (from left to right) those for the 1996, 1998, 1997, 1995, and 1993 year-classes.

When the three-parameter model was used, the current estimate of α was 1.095 with a standard error of 0.228, about 21% of the parameter value (Table 29). Differences among annual values of α seen in the time-series of estimates were likely caused by increased fishing mortality on winter flounder in addition to the inherent instability of parameter estimates fitted to small data sets. In particular, the apparent influence of the 1988-98 data points on the estimate of α were illustrative of higher recent exploitation and poor recruitment. The Niantic River winter flounder population apparently now has a greatly diminished compensatory reserve in response to rates of fishing that increased from 0.71 in 1985 to 1.10-1.33 in the mid-1990s. A recent fishing mortality rate as high as 1.02 was still reported for LIS winter flounder by Johnson et al. (2000). Relatively high abundance of juvenile winter flounder from the 1988 year-class was expected to result in increased numbers of adult fish during 1992-94 that would dominate the spawning population. Unfortunately, winter flounder from these large year-classes were removed quickly by fishing (Simpson et al. 1996). Both the Niantic River studies and Simpson et al. (1996) noted an apparent lack of adult fish in Connecticut waters in recent years, even though juvenile fish of the 1992 year-class appeared to be abundant. Increases in adult abundance resulting from relatively numerous age-0 juveniles seen in the past several years were expected as these fish become mature, but this has not materialized. Other factors may have substantially reduced their numbers prior to adult recruitment, as was seen for the 1988 and 1992 year-classes.

The estimate of Ricker's β parameter, which describes the annual rate of compensatory mortality as a function of the stock size, is an important factor in SPDM simulations. The present value for β is 1.861×10^{-5}, the lowest estimate of the series (Table 29). Previous values ranged between 1.877 and 2.583×10^{-5}. The current long-term (1989-2002) mean estimate of 2.244×10^{-5} for β was less than the average found during the mid-1990s because of five consecutive low estimates, suggesting a possible weakened capability to compensate. However, the present calculation does not include the contribution of year-classes after 1998. These year-classes had increasing numbers of juveniles found in the Niantic River (see Juvenile Winter Flounder), which should result in better recruitment if mortality later in the juvenile stage (ages-1 through 3) is not excessive. Nevertheless, the contribution of these juveniles to future adult stock abundance remains to be seen.

The parameter ϕ, which reflects the effect of February temperature deviations (T_{Feb}) from the 1977-98 mean of 2.87°C (Table 28), has also been used as an SPDM input parameter and was estimated as -0.399 in 2002, a value nearly the same as found in 1999 through 2002 (Table 29). The long-term mean for ϕ was -0.374 (range of -0.418 to -0.259).

TABLE 29. Annual estimates of the modified Ricker stock-recruitment function (Eq. 7) determined for the Niantic River winter flounder population from 1989 through 2002.

Year of estimation	Year-classes included[a]	α[b]	Standard error	β (X 10^{-5})	Standard error (X 10^{-5})	ϕ	Standard error
1989	1977-85	2.646	0.599	2.228	0.456	-0.259	0.095
1990	1977-86	2.502	0.399	2.466	0.372	-0.264	0.064
1991	1977-87	2.226	0.518	2.140	0.461	-0.329	0.098
1992	1977-88	2.149	0.543	2.466	0.567	-0.357	0.010
1993	1977-89	1.977	0.566	2.523	0.642	-0.412	0.108
1994	1977-90	2.071	0.428	2.498	0.478	-0.379	0.077
1995	1977-91	1.710	0.380	2.583	0.516	-0.415	0.078
1996	1977-92	1.473	0.306	2.450	0.488	-0.418	0.075
1997	1977-93	1.442	0.283	2.399	0.463	-0.417	0.072
1998	1977-94	1.186	0.265	1.961	0.513	-0.381	0.082
1999	1977-95	1.082	0.265	2.071	0.570	-0.408	0.088
2000	1977-96	1.125	0.252	1.886	0.478	-0.400	0.082
2001	1977-97	1.095	0.241	1.877	0.472	-0.404	0.081
2002	1977-98	1.095	0.228	1.861	0.437	-0.399	0.076
	Mean	-		2.244		-0.374	

[a] Age-4 considered to be minimum age of recruitment.
[b] The compensatory reserve for an unfished stock (α_0) used in the SPDM is 5.20 (see Table 30 for the derivation of this value).

As noted above, February water temperatures have been generally warmer in recent years and estimates for the ϕ parameter have increased in magnitude. The effect of February temperatures on winter flounder recruitment is discussed in the following section (Formation of Winter Flounder Year-class Strength).

Ricker's stock-at-replacement (P_{rep}; Eq. 12), a derived biological reference point, was estimated using the SRR parameter estimates as 73,486 female spawners. This is the unfished equilibrium spawning stock size, the biomass for which is often referred to as maximum spawning potential (MSP). This reference point was calculated as 121,252 lbs for Niantic River winter flounder (i.e., 1.65 lbs per spawner; see Table 31, below). Stocks with biomass less than the critical size of 25% of MSP (in this case, 30,313 lbs) are considered overfished (Howell et al. 1992).

Estimation of α_0 for SPDM simulations. The stock-recruitment-based estimates of α for the Niantic River winter flounder discussed above underestimated the true slope at the origin of this stock. The method of calculating annual recruitment included the effects of fishing on winter flounder age-2 and older as well as the entrainment of larvae at MPS. Therefore, these direct estimates of α correspond to a compensatory reserve diminished by existing larval entrainment and exploitation rates. Goodyear (1977: Fig. 1) discussed the concept of a 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. 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., α_0 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 (Table 30). Because these methods did 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 geometric mean of $\alpha_0 = 5.20$ calculated from these estimates was used in the SPDM. This parameter describes the inherent potential of a stock to 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). Consequently, the large difference between the derived value of α_0 (5.20) and regression estimates of α based on field data reflects the difference in potential growth between unfished and highly exploited stocks of winter flounder. Use of an unfished stock as a starting point for a population dynamics simulation has a number of advantages, depending upon the particular scenario selected. The data-based estimates of the other two SRR parameters (β and ϕ) used in the population simulations, however, do not depend upon fishing and

TABLE 30. Methods of estimating the compensatory reserve parameter α_0 of the unfished Niantic River winter flounder based on several life history models (modified from Table 4 in Crecco and Howell 1990).

Reference	Equation[a]	Data used[a]	Estimate of α_0
Boudreau and Dickie (1989)	$r_m = 2.88 \times weight^{-0.33}$	weight = 414 Kcal (0.71 lb)	5.90[b]
Hoenig et al. (1987)	$r_m = 425.2 \times t_m^{-0.949}$	t_m = 1,642 days (4.5 years)	5.47[b]
Longhurst (1983)	$r_m = 3K \times ([L_\infty / L_m] - 1)$	L_∞ = 19 inches $K = 0.30$ L_m = 13.55 inches	5.10[b]
Cushing (1971, 1973)	$\alpha_0 = 1.98 + 0.0306(FEC^{0.33})$	FEC = 600,000	4.45
Geometric mean			5.20

[a] r_m = annual intrinsic rate of population increase; weight is weight in Kcal at which 50% of female winter flounder first spawn; t_m = mean generation time in days; K and L_∞ are parameters of the von Bertalanffy growth equation and L_m is mean length; and FEC = mean fecundity.

[b] α_0 = exp($r_m \times t_m$), where t_m = 4.5 years (mean time to maturation for females).

entrainment rates. Since they are not directly related to current winter flounder abundance, their estimates were obtained as long-term averages of their data series in a meta-analysis fashion (Table 29).

Finally, both the mean weight and fecundity of a Niantic River female winter flounder was calculated for a theoretical population of ages 1 through 15 at equilibrium for which only M was assumed (i.e., the unfished population). These calculations used population data previously reported (NUSCO 1990, 2000) and an estimated M of 0.2. The equilibrium calculation for this theoretical unfished stock showed a mean weight of 1.65 lbs per female and a mean fecundity of 1,322,994 eggs per spawner (Table 31).

Formation of Winter Flounder Year-Class Strength

Year-class strengths are likely determined during larval and early juvenile life stages of marine fishes, yet these phases of life history are least understood (Sissenwine 1984; Bailey and Houde 1989; Bradford 1992). Accordingly, formation of winter flounder year-class strength begins at egg deposition, or even before, given some known maternal influences on egg quality. For example, a study by Buckley et al. (1991) noted that female size and time of spawning affected various winter flounder reproductive parameters, including egg size, fecundity, and viability. They observed that eggs deposited earlier

in the season by larger females appeared to have better survival than eggs produced by smaller fish late in the season. Biological consequences of the selective removal of large fish by fishing can be more profound than just reducing egg production because of the aspects of egg size and time of spawning relating to reproductive success (Trippel 1995; Conover 2000).

Less is known about the egg stage than the larval developmental period of winter flounder. Environmental factors have important implications on egg survival and hatching (Bunn et al. 2000). Keller and Klein-MacPhee (2000) reported that in a mesocosm study completed in Rhode Island, winter flounder egg survival, hatching percentage, time to hatch, and initial size were greater in cooler than warmer systems. 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 a diatom, *Skeletonema costatum*. The decomposing material caused a depletion of oxygen and egg death due to anoxia. This diatom was one of the most abundant of the phytoplankton collected at MPS during entrainment sampling from 1977 through 1980 (NUSCO 1981). However, highest densities occurred in summer, after the winter flounder egg incubation period. *Skeletonema costatum* was also a dominant form in the warm (about 1°C above current ambient mean water temperature) treatment in the mesocosm study

TABLE 31. Biomass calculations of the Niantic River female winter flounder spawning stock at equilibrium, based on an instantaneous natural mortality rate of M = 0.2 and an instantaneous fishing mortality rate of F = 0 (i.e., an unfished stock).

Age	Female population size	Fraction mature	Number of mature females	Mean length (cm)	Mean weight of mature females (lbs)	Eggs per mature female	Spawning stock biomass (lbs)	Egg production (millions)
2	1,000.00	0.00	0.00	18.0	-	-	-	0.000
3	818.73	0.25	204.68	27.0	0.497	232,088	101.77	47.504
4	670.32	0.80	536.26	31.0	0.776	432,517	416.36	231.940
5	548.81	1.00	548.81	34.5	1.096	700,390	601.73	384.381
6	449.33	1.00	449.33	37.5	1.435	1,019,793	644.71	458.224
7	367.88	1.00	367.88	39.0	1.628	1,216,926	599.04	447.683
8	301.19	1.00	301.19	40.5	1.839	1,442,512	553.95	434.470
9	246.60	1.00	246.60	42.0	2.068	1,699,372	510.01	419.065
10	201.90	1.00	201.90	43.5	2.316	1,990,489	467.61	401.880
11	165.30	1.00	165.30	45.0	2.584	2,319,011	427.09	383.332
12	135.34	1.00	135.34	46.5	2.872	2,688,253	388.70	363.828
13	110.80	1.00	110.80	48.0	3.182	3,101,703	352.54	343.669
14	90.72	1.00	90.72	49.5	3.514	3,563,020	318.78	323.237
15	74.27	1.00	74.27	51.0	3.869	4,076,040	287.36	302.727
Total	5,181.19		3,433.08				5,669.66	4,541.941

Mean weight per mature female fish = (5,670 lbs ÷ 3,433 mature females) = 1.65 lbs (~39.2 cm fish)
Mean fecundity (unfished stock) = 1,322,994 eggs per female spawner

of Keller and Klein-MacPhee (2000). Macroalgal and detrital mass varied considerably among years of study in the Niantic River since the mid-1970s and the amount of organic material on the river bottom may have had some influence on the survival of demersal winter flounder eggs.

Based on a comparison of estimates of egg production and abundance of Stage 1 larvae (see Larval Winter Flounder, Abundance and Distribution: Fig. 16), egg mortality from unknown causes may be considerable in the Niantic River, but may also vary in a density-dependent fashion in some years. This apparent compensatory effect resulted in egg survival that could have been as much as four times higher in most recent years (1995-2002, except for 2000) than previous. The mechanisms responsible for greater winter flounder egg survival were not studied, but hypotheses concerning this effect include less predation pressure because of fewer cues for predators under low egg abundance or better egg quality because of greater food resources shared among fewer adult females. Also, Buckley et al. (1991) suggested greater relative fecundity and viability of eggs produced by a spawning stock dominated by larger individuals. As noted previously in Adult Winter Flounder - Relative Annual Abundance, larger females have made up an increasing proportion of spawners in recent years.

Regarding predation on eggs, DeBlois and Leggett (1991) and Frank and Leggett (1984) found that both an amphipod (*Calliopius laeviusculus*) and winter flounder both preyed heavily upon demersal capelin (*Mallotus villosus*) eggs, removing up to 39% and 5% of the production, respectively. They suggested that invertebrate predation on demersal fish eggs may be an important regulatory mechanism for population size in marine fishes having demersal eggs. Because demersal fish eggs are immobile and found in relatively dense patches, they may attract many predators (Bunn et al. 2000). In particular, the sevenspine bay shrimp has been observed feeding on winter flounder egg clusters in laboratory tanks (D. Taylor, University of Rhode Island, Graduate School of Oceanography, Narragansett, RI, pers. comm.).

Larval mortality varies considerably from year to year and can have profound effects on the number of young that metamorphose and settle. Larval winter flounder mortality was found to be related to both larval density and April water temperatures (see Larval Winter Flounder, Mortality: Fig. 24). Thus, when egg abundance was particularly low, enhanced egg survival or hatchability was apparent. Relatively high abundance of older larvae, particularly during 1999-2001, that resulted from better larval survival

perhaps reflected a compensatory response related to initially low larval densities.

Intrinsic (e.g., growth) and extrinsic (e.g., predation) factors both affect survival. Several workers (Laurence 1977; Chambers et al. 1988) found that much of larval winter flounder mortality was concentrated in early life and at first feeding, a few weeks after hatching. 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. Therefore, the strength of a year-class could be determined by the availability of sufficient food after completion of yolk absorption. However, Keller and Klein-MacPhee (2000) noted in a mesocosm study that food availability was not the most important factor related to larval mortality as mortality was highest in their warm treatment, which also had the highest food availability. Daily mortality rates increased directly with the abundance of active predators in the mesocosms, but this may have been an effect of the relatively low numbers of predators in their experimental system. They further noted that daily growth and mortality rates were significantly and inversely related.

Predation is often one of the most important causes of larval fish mortality (Bailey and Houde 1989) and this has relevance to winter flounder larval populations. The relationship between winter flounder larvae and their predators was studied by Williams and Brown (1992), who found that escape response increased with increasing larval size, but that it remained slower than that of other larval fishes examined. Larval winter flounder are likely vulnerable to both fish and invertebrate predators. Although susceptible to attacks by planktivorous fishes, the temporal occurrence and abundance of fishes that could potentially prey on larval winter flounder are low, particularly during the early portion of the larval winter flounder season.

Most predation on winter flounder larvae is probably by invertebrate contact predators, such as anemones (Keller and Klein-MacPhee 2000), other cnidarians, and ctenophores, many of which have been previously identified as feeding on flatfish larvae. Van der Veer (1985) reported evidence of a causal predator-prey relationship between larvae of European plaice and European flounder and two such predators, a scyphomedusan (*Aurelia aurita*) and a ctenophore (*Pleurobrachia pileus*). However, most predation by these species occurred at the end of the larval European plaice season and likely did not ultimately affect year-class strength (Van der Veer 1985; Van der Veer et al. 1990). Laboratory studies

showed that successful capture of European 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. aurita*) in Point Judith Pond, RI and believed that they were a significant predator of larval winter flounder. Potential predators of winter flounder larvae in the Niantic River were medusae of the lion's mane jellyfish (*Cyanea* sp.), which can be abundant in the upper river (Marshall and Hicks 1962; Brewer 1989), including station A. This scyphozoan genus selectively preys upon fish and ichthyoplankton (Fancett 1988). Its occurrence in the Niantic River from just after ice-out in late February through early June (Brewer 1989) coincides with the larvae winter flounder season. 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). Also, in 1997 ctenophores, another larval fish predator, were present during the larval winter flounder season (NUSCO 1998). However, the relationship between larval winter flounder and gelatinous zooplankton biomass did not support a cause and effect relationship and the role that these and other planktonic predators have each year on ultimate year-class strength is yet unknown.

Physical, hydrodynamic processes occurring during the larval stage and settlement of juveniles may also affect the success of a year-class (Werner et al. 1997). This can occur by determining their rates of exposure to predation and food resources and location of settlement. Many winter flounder larvae are flushed from natal estuaries and transported by water currents within LIS. Variability may also be imposed on the transport of larvae by winds and work on other flatfishes is illustrative of these effects. Densities of settled juveniles of the brown sole (*Pleuronectes herzensteini*) in Japan were related to the frequency of onshore wind events (Nakata et al. 2000). Variable rates of transport in LIS can affect not only the availability of winter flounder larvae for entrainment at MPS, but likely also where larvae are able to settle as juveniles. Winter flounder, European plaice, and stone flounder (*Platichthys bicoloratus*) larvae each use selective tidal transport to enter preferred coastal nursery areas (Rijnsdorp et al. 1985; Yamashita et al. 1996a; Chant et al. 2000). As illustrated by a secondary peak in abundance of older larvae during the later stages of development, winter flounder larvae also appear to be entering the Niantic River. Nearly all pre-recruits are produced in the river and similar nursery habitats. The survival of metamorphosed juveniles settling in Niantic Bay or other deeper areas outside of inshore nursery grounds is poor, probably because of high rates of predation. Similarly, Yamashita et al. (2000) found that small estuaries produced disproportionately more stone flounder juveniles than geographically much larger nursery areas found in shallow, more exposed areas of Sendai Bay, Japan.

For newly settled young of flatfishes, Van der Veer et al. (1990) speculated that, in general, predation by crustaceans may be a common regulatory process, although this may not be a strong effect in all areas (Nash and Geffen 2000). Predation by various caridean shrimps has been suggested as the cause of high mortality after metamorphosis for several flatfishes, including the winter flounder (Witting and Able 1993, 1995), European plaice (Lockwood 1980; Van der Veer and Bergman 1987; Pihl 1990; Van der Veer et al. 1991; Pihl and Van der Veer 1992; Gibson et al. 1995; Ansell et al. 1999; Van der Veer et al. 2000a), Japanese flounder (*Paralichthys olivaceous*; Seikai et al. 1993), and stone flounder (Yamashita et al. 1996b). Direct evidence now exists for high predation rates on newly settled winter flounder by sevenspine bay shrimp in the Niantic River (Taylor 2002). Brown shrimp (*Crangon crangon*) preferentially preyed upon smaller (10-20 mm) European plaice (Van der Veer and Bergman 1987). Similarly, Witting and Able (1993, 1995) found that the size of age-0 winter flounder significantly affected their probability of predation by sevenspine bay shrimp, 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 shrimp attacks. Predation was also related to shrimp density and steadily increased until reaching an asymptote at densities greater than $10.6 \cdot m^{-2}$ (Witting and Able 1995).

The green crab (*Carcinus maenus*) preys on pleuronectid flounders, including winter flounder (Ansell et al. 1999; Fairchild and Howell 2000). Other predators of juvenile winter flounder include larger fishes (e.g., grubby; *Myoxocephalus aenaeus*) and marine birds (e.g., double-crested cormorants; *Phalacrocorax auritus*). Predation likely continues from settlement throughout summer in shallow nursery habitats preferred by young winter flounder. Pearcy (1962) suggested that several of these species preyed upon juvenile winter flounder in the Mystic River, CT estuary. Winter flounder and other flatfishes occurred frequently in the diets of

cormorants in northern New England and Canada (Pilon et al. 1983; Blackwell et al. 1995; Rail and Chapdelaine 1998) and in Europe (Leopold et al. 1998). Cormorant predation on winter flounder has been frequently observed in the Niantic River. Winter flounder otoliths have been identified in regurgitated pellets of cormorants nesting in areas near MPS and in recent years over 100 active nests were present on Waterford Island alone, just outside the mouth of the Niantic River (M. Male, Old Lyme CT, pers. comm.). Birt et al. (1987) found that close to cormorant nesting colonies, predation depleted fish populations, including those of winter flounder, with higher fish densities observed in areas beyond the foraging range of these birds. Breeding pairs of cormorants have increased steadily since they were first counted as nesting in Connecticut in 1982 with populations increasing by 15% annually between 1986 and 1998 and by 14% in coastal New York (Victoria 2001). In Rhode Island, the annual increase from 1981 to 1990 was even greater (63%).

Manderson et al. (2000) reported that summer flounder (*Paralichthys dentatus*) preyed upon age-0 winter flounder in the Navesink River, NJ with selection increasing as juvenile winter flounder grew from 20 to 90 mm. Striped searobin (*Prionotus evolans*) prey heavily upon young winter flounder when they co-occur (Manderson et al. 1999). Green crab abundance in the Niantic River increased substantially during the 1990s, with a non-parametric Mann-Kendall test ($Z = 4.607$; $n = 27$) showing a significant ($p < 0.0001$) positive trend in the slope of the annual catch at the trawl monitoring program station in the river since 1976. However, abundance dropped off sharply in 2001 and, although an increase from the previous year, the total catch in 2002 was similar to the lower sustained abundance of green crabs found during the 1980s (Fig. 40).
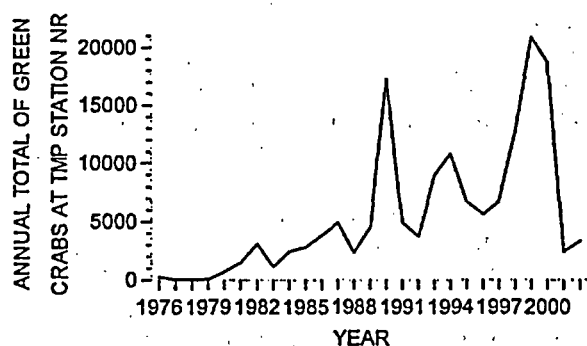
Presence of vegetation (eelgrass, macroalgae) decreased the vulnerability of young winter flounder to predation (Manderson et al. 2000). Eelgrass beds in the Niantic River have also decreased considerably in recent years (Keser et al. 2003), perhaps increasing the vulnerability of winter flounder to predation.

Another cause of mortality during the first year of life of winter flounder is from infection by the microsporidian parasite *Glugea stephani*, the severity of which can vary by fish size, density, and environmental factors such as water temperature (Takvorian and Cali 1981, 1984; Cali et al. 1986; Cali and Takvorian 1991; MacLean 1993). Irregular rates of infection in age-0 winter flounder can also introduce variation in winter flounder recruitment.

As with larvae, the time duration that settled juveniles spend in a vulnerable size range, which is related to growth rate, affects the vulnerability of young to predators such as the sevenspine bay shrimp or green crab. Variation in growth, which depends upon specific location of settling, specific habitat within a location, temperature, food, or physico-chemical parameters (Sogard 1990; Sogard and Able 1992; Bejda et al. 1992; Gibson 1994; Meng et al. 2000, 2001; Phelan et al. 2000; De Long et al. 2001), may have significant implications for young winter flounder survival after metamorphosis. Al-Hossaini et al. (1989) reported greater growth for cohorts of European plaice that settled relatively early in Wales, but these fish also had higher mortality. Conversely, growth was slower in late-settling cohorts, but survival was higher. In contrast to shrimp predation, larger-sized age-0 flatfish may be preferentially selected by birds and certain fishes (Van der Veer et al. 1997). In addition, environmental effects, such as water temperature, may greatly influence the ability of predators to prey upon young winter flounder. In particular, the effects of February water temperature on the recruitment of winter flounder is discussed below.

Temperature also acts indirectly across several early life history stages by affecting growth, development, and mortality. Van de Veer et al. (2000b) proposed that meristic elements, such as the number of vertebrae and fin rays, had effects seen later in the juvenile stage of young European plaice. Temperature-dependent characters, particularly the numbers of vertebrae (established during the egg stage) and fin rays (during the larvae phase), appear to be related to growth and mortality experienced later on by juveniles. This was likely related to variable performance in locomotion and predator avoidance responses. Thus, a non-genetic phenotypic plasticity found during very early life history that was influenced by environmental factors, in combination

Fig. 40. Total number of green crabs taken each year at TMP station NR from January 1976 through December 2002.

with events occurring later in life during the settled juvenile stage, could affect resulting year-class strength.

Variation in annual year-class strength of European plaice was suggested to occur in either the pelagic larval phase (Zijlstra and Witte 1985; Van der Veer 1986) or after settlement during the juvenile stage on nursery grounds (Nash and Geffen 2000). Van der Veer (1986), Van der Veer and Bergman (1987), and Bergman et al. (1988) noted that recruitment variability in European plaice found in The Netherlands was stabilized between years as a result of a density-dependent regulatory process, predation on newly metamorphosed fish by caridean shrimp. In contrast, year-class strength of European plaice in Swedish bays varied to a greater degree (CVs = 67-118%). The latter was thought related to the effects of temperature variation during the larval stage and more variable crustacean predation on newly metamorphosed young European plaice found in more northerly waters (Pihl 1990; Pihl and Van der Veer 1992). However, variable hydrographical effects on settlement of young European plaice may also have occurred in Sweden (Modin and Pihl 1994; Pihl et al. 2000). Thus, population regulation in flatfishes may be coarsely determined during the earliest life history stages by variable survival of eggs and larvae and then fine-tuned by mortality of newly metamorphosed juveniles, which can be density-dependent beyond certain threshold levels of abundance (Van der Veer and Bergman 1987; Iles and Beverton 1991; Beverton and Iles 1992a, 1992b; Rose et al. 1996; Van der Veer et al. 2000a).

High recruitment of winter flounder is associated with cold winters and a significant effect of February water temperature has reduced variability in the Niantic River winter flounder stock and recruitment relationship. The exact mechanism of how February temperatures, in particular, affect winter flounder recruitment remains unknown. Cold February water temperature was also suggested as an environmental influence that increased European plaice recruitment in both The Netherlands (Zijlstra and Witte 1985; Van der Veer 1986; Iles 1994; Van der Veer and Witte 1999; Van der Veer et al. 2000a) and in a small bay on the Irish Sea (Nash et al. 1994). Similarly, recruitment of European plaice throughout much of the waters around the United Kingdom was negatively correlated with water temperatures during February-June, the period encompassing the drift of their eggs and larvae and settlement (Fox et al. 2000). February coincides with most winter flounder spawning, egg incubation, and hatching. These processes and 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. Keller and Klein-MacPhee (2000) reported winter flounder egg survival, percent hatch, and initial size were significantly greater and mortality rates lower in cool as opposed to warm experimental mesocosms. Adult acclimation temperatures and egg and larval incubation temperature affected larval size and biochemical composition. Cold winters and 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 which have to outgrow various predators. Warmer winter water temperatures in Rhode Island have also been correlated with smaller winter-spring phytoplankton blooms (Keller et al. 1999), which may have had consequences to marine food webs. 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. The association of strong year-classes of European plaice with cold winters likely occurred because predatory brown shrimp suffered high mortality during low water temperatures or migrated out of the nursery areas (Zijlstra and Witte 1985; Van der Veer 1986; Pihl 1990; Pihl and Van der Veer 1992; Van der Veer et al. 2000a). Keller and Klein-MacPhee (2000) also observed that the sevenspine bay shrimp remained inactive and buried within the sediments of cool experimental mesocosms, whereas they were active in the warm systems.

Potential effects of temperature and winter climate on winter flounder are further illustrated in Figure 41 by comparing a long-term (1959-2002) CPUE index (annual sum of monthly means) of annual winter flounder (all sizes) developed from weekly trawl sampling conducted by the University of Rhode Island (URI) near Fox Island in upper Narragansett Bay (previously described in Adult Winter Flounder, Comparisons among Regional Abundance Indices)
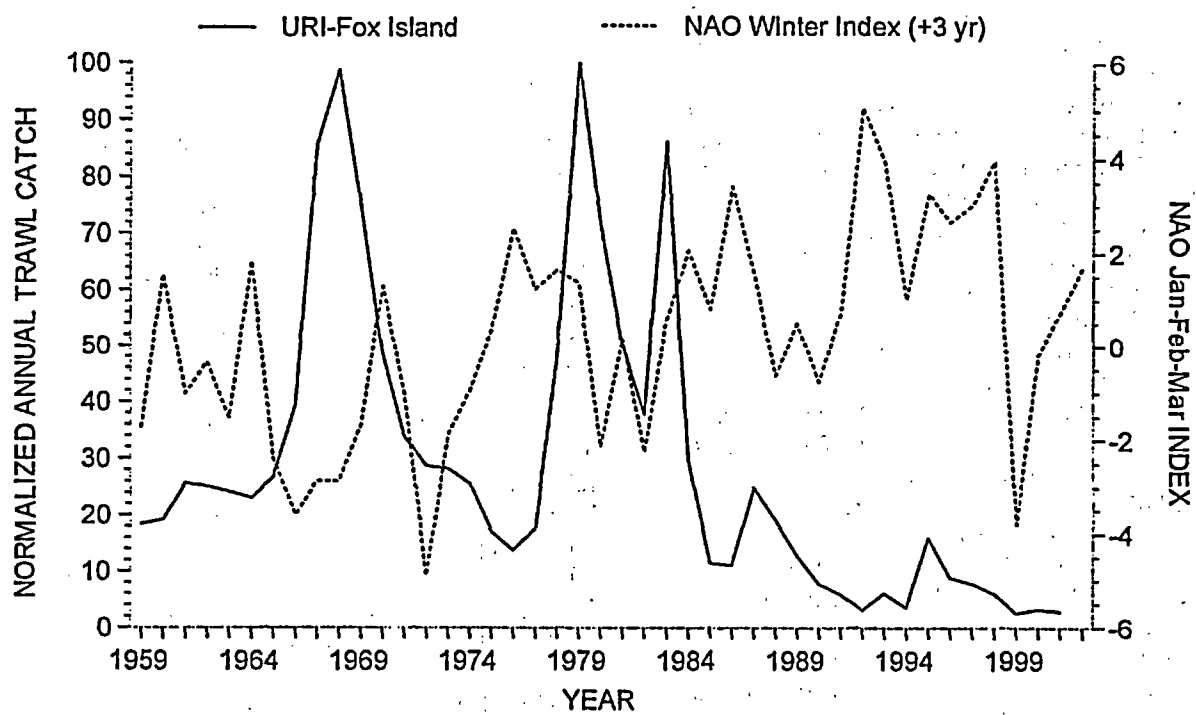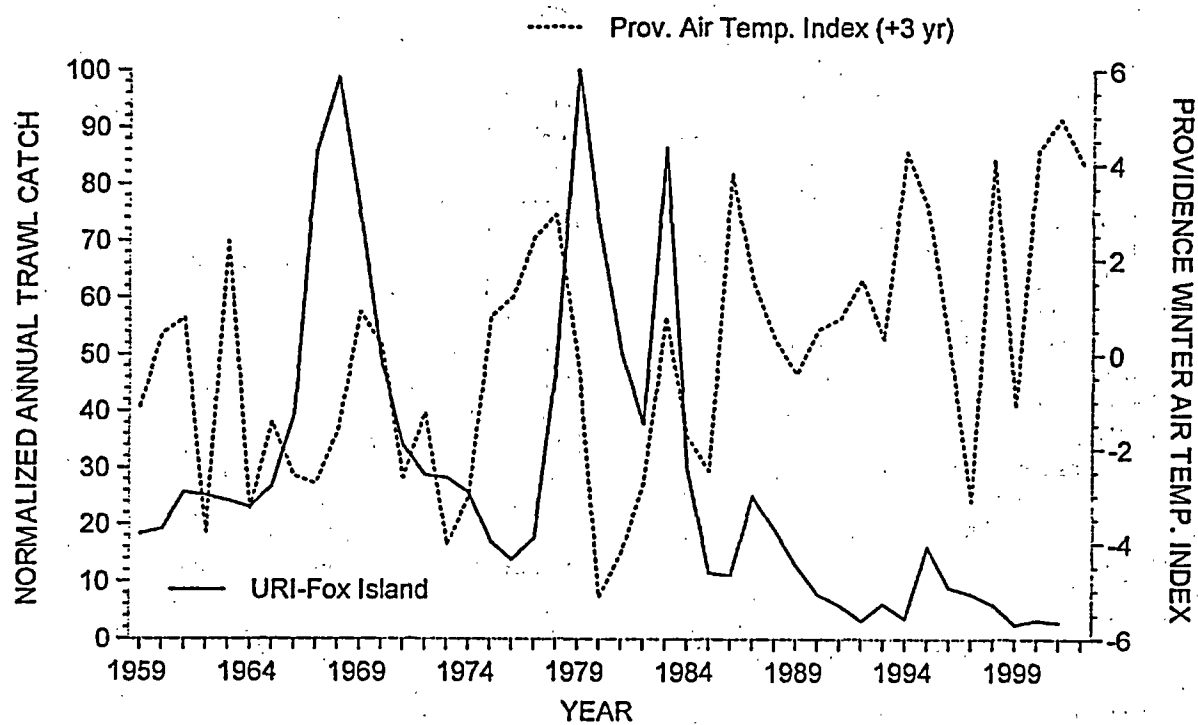
Fig. 41. Comparison from 1959 through 2002 between the annual URI winter flounder trawl CPUE abundance index at Fox Island in Narragansett Bay, RI and the Providence, RI mean winter (December-February) air temperature index (top) and the North Atlantic Oscillation (NAO) winter (January-March) winter index (bottom). Both climatic indices were advanced 3 years relative to the winter flounder abundance index.

with the mean Providence, RI winter (December-February) air temperature (Anonymous 2003a) and the North Atlantic Oscillation (NAO) winter (December-March) index (Anonymous 2003b). The NAO index (Hurrell 1995) is based on the difference of normalized sea level pressures between Lisbon, Portugal (representing the Azores high pressure system) and Stykkisholmur/Reykjavik, Iceland (Icelandic low). Negative phases of the NAO index result in colder, wetter, and windier winters in the northeastern United States and positive phases are related to warmer, milder winters (Greene and Pershing 2000). The NAO is considered to be a driving force in climatic systems in the northern hemisphere (Hurrell 1995; Ottersen et al. 2001). In recent years, considerable interest has developed in examining relationships between climate and biological processes. Climatic oscillations reflected by the NAO may be viewed as a proxy for various regulating forces in marine or other ecosystems, including changes to timing of reproduction, population dynamics, distribution, and in interspecific relationships, including competition and predator-prey (Ottersen et al. 2001). The NAO and Providence winter air temperature indices were positively correlated (Spearman's rank-order correlation; n = 48 years; r = 0.450, p = 0.0013). A Mann-Kendall test also showed that both of these climatic indices had significantly (p =0.003 and p < 0.001, respectively) increasing trends from 1959 through 2002, with the majority of the positive values occurring since the mid-1980s. Conversely, the URI annual trawl CPUE index for winter flounder had a significantly (p < 0.001) declining trend during the same period. When the air temperature and NAO indices were advanced 3 years with respect to the URI trawl index, significant negative correlations (Spearman's rank-order correlation; Providence winter air temperature: r = −0.524, p = 0.0003, n = 44; NAO winter index: r = −0.451, p = 0.0021, n = 44) were found. This occurred even though the trawl catches were comprised of many age-classes of winter flounder and annual winter weather should have affected the reproductive success of only one year-class. There was likely a tendency for relatively large or weak year-classes to dominate trawl catches. This result showed that winter flounder abundance was highest during periods of relatively severe winters, such as occurring in the mid-1960s and late 1970s, when dominant year-classes were produced. Abundance was lowest when winters were milder, years during which fewer fish were produced. The trend of generally warm winters found from the mid-1980s through the present, most likely in combination with relatively high fishing mortality rates, has likely kept winter flounder abundance depressed. The climatic trend may continue as models have predicted that the current positive phase of the NAO may persist through the present and the following decade (Paeth et al. 1999). Sogard et al. (2001) note, however, that local environmental conditions and possibly genetic effects can also control recruitment in addition to large-scale climatic factors.

Recruitment of many fishes may also be affected by density-dependent processes occurring during the first year of life following completion of the larval stage (Bannister et al. 1974; Cushing 1974; Sissenwine 1984; Anderson 1988; Houde 1989; Myers and Cadigan 1993a, 1993b; Bailey 1994). Bannister et al. (1974), Lockwood (1980), Van der Veer (1986), and Pihl et al. (2000) all reported density-dependent natural mortality (M) for young European plaice. Examination of some of these observations, however, indicated that greatest rates of mortality only occurred when extremely large year-classes of European plaice were produced (i.e., three to more than five times larger than average). This was confirmed in analyses by Iles and Beverton (1991) and Beverton and Iles (1992a, 1992b), who reported that although density-dependent mortality was indicated for age-0 European plaice in the North Sea, below a specific density ($1.8 \cdot m^{-2}$) mortality was likely density-independent. Pihl et al. (2000) noted that density-independent mortality was considerable, even at relatively high densities of European plaice. The high production of young Niantic River winter flounder that occurred in 1988 because of very low apparent mortality also showed no sharp peaks in abundance, with densities generally remaining below $1 \cdot m^{-2}$. However, high ($>2 \cdot m^{-2}$) densities of young winter flounder at station LR during some weeks in early summer of 1990, 1994, 1995, and 2001 were followed by the steepest declines in abundance. In contrast, relatively low densities were found at both age-0 sampling stations during 1997, yet M was relatively high (Table 18). When catch data from both stations were combined, a significant positive relationship was found between M and density (Fig. 30). In a study examining 11 years of data from Narragansett Bay, DeLong et al. (2001) reported that the monthly mortality rate of age-0 winter flounder was density-dependent and also positively related to water temperature. They noted that mortality rates were highest in years with temperatures above 18°C. However, the mean summer water temperature at the MPS intakes has exceeded this value for each of the past 26 years (Table 2) and water temperature is even higher at the shallow Niantic River nursery sites. The large difference found in values of M between 2000

and 2001 (Tables 18 and 19) despite the same mean summer water temperature of 20.45°C suggests a less certain temperature-dependent effect on mortality in the Niantic River.

A comparison of early and late season median catch-per-unit-effort (CPUE) of age-0 winter flounder in the Niantic River showed that initially large numbers of young present during late spring and early summer in some years did not necessarily result in high densities of fish at the end of summer (Fig. 42). Differences found between early and late summer were largely related to variation in mortality rates, which affected year-class abundance. Notably, little observed mortality in 1988 and 2000 meant that modest initial sets of young resulted in relatively strong year-classes, whereas high mortality occurring during early summer in 1990, 1995, and 2001 considerably reduced initially high densities by late summer. Above-average survival rates found during both 1999 and 2000 should help these winter flounder year-classes remain abundant. However, success also depends upon mortality rates that affect these fish during the several years remaining before recruitment as seemingly abundant year-classes of young winter flounder found during the 1990s did not result in numerous adults (Simpson et al. 1996; Desfosse et al. 1998). Thus, neither the 3 years of extended shut-downs at MPS nor the appearance of relatively numerous metamorphosed juveniles in recent years have resulted in a sharp rebound in abundance of Niantic River winter flounder. Considerable influence of natural mortality during the first several years of life and fishing mortality reducing adult spawner biomass remain the most important factors in determining recruitment and subsequent adult abundance. These factors also appear to be acting on a larger regional scale than just in the vicinity of MPS, which was discussed previously in Adult Winter Flounder - Comparisons Among Regional Abundance Indices.

MPS Impact Assessment

Larval Entrainment

Data Analyses. Field sampling and laboratory processing of winter flounder larvae entrained through the MPS cooling-water system was previously described (see Larval Winter Flounder - Field Sampling, Laboratory Processing, and Data Analyses). The estimated number of larvae entrained in the MPS condenser cooling water system each year was determined using larval densities at station EN (Fig. 13) and the measured volume of cooling water
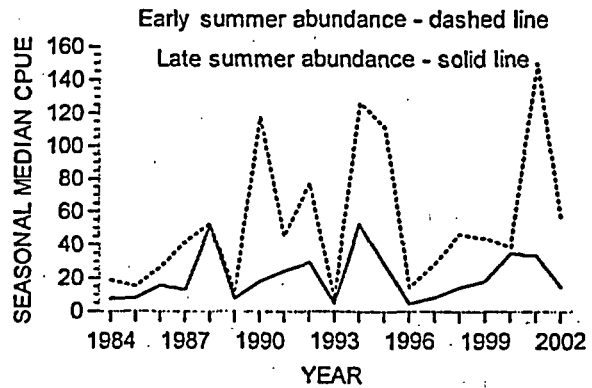
Fig. 42. Comparison between the density (median catch per 100 m^2) of age-0 winter flounder during early and late summer at Niantic River stations LR and WA combined from 1984 through 2002.

used by the three (now two) MPS units in operation (Table 32). The Gompertz density function (Eq. 3) was fitted to larval abundance 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.

Estimates of Larval Entrainment at MPS. The number of winter flounder larvae entrained in the condenser cooling water of MPS is the most direct measure of potential impact on the Niantic River winter flounder stock. Due to the importance of this measure, a review and evaluation of methods to estimate annual entrainment of larval winter flounder by MPS was conducted and provided in NUSCO (1991c). This topic was also summarized in NUSCO (2000).

Nearly all winter flounder larvae collected at station EN were taken from February through June, with most (>90%) during April and May. The entrainment estimate for 2002 of 118.8 million ranked as the sixteenth highest annual estimate of the 27-year period (Table 33). This estimate resulted from the fifth lowest larval abundance (A parameter) found at EN and a moderate (74% of total, considering the rated capacity of all three MPS units) volume of seawater used for condenser cooling (Table 32). During the 2002 larval season, Unit 2 was shut down for refueling from February 16 through April 2, while Unit 3 remained in full operation and Unit 1 did not operate due to decommissioning. If all three MPS units had been operating throughout this season, an estimated 160.5 million larvae would have been entrained, an estimate 1.35 times as large as the actual entrainment total. The fraction of entrainment

TABLE 32. For comparative purposes, the annual average cooling-water flow and percent of nominal maximum flow at MPS Units 1 through 3 and station total during a nominal April 1-June 15 larval winter flounder season from 1971 through 2002. Entrainment estimates given in Table 33 are based on seawater volumes utilized by MPS during the annual larval period of occurrence.

	Unit 1[a]		Unit 2		Unit 3		MPS three-unit total	
Nominal flow at 100% capacity:	28.0 m³·sec⁻¹		36.1 m³·sec⁻¹		59.5 m³·sec⁻¹		123.6 m³·sec⁻¹	
Fraction of total MPS flow:	0.227		0.292		0.481		1.000	
Year[b]	Average flow	% of maximum	Average flow	% of maximum	Average flow	% of maximum	Average flow	% of maximum
1971	-	68.0	-	-	-	-	-	15.4
1972	-	99.4	-	-	-	-	-	22.6
1973	-	23.7	-	-	-	-	-	5.4
1974	-	97.0	-	-	-	-	-	22.0
1975	-	84.4	-	-	-	-	-	19.2
1976	25.8	92.1	29.9	82.9	-	-	55.7	45.1
1977	26.4	94.1	15.0	41.5	-	-	41.3	33.4
1978	21.3	76.0	27.1	75.2	-	-	48.4	39.2
1979	10.0	35.7	21.8	60.4	-	-	31.8	25.7
1980	22.7	81.1	26.4	73.3	-	-	49.2	39.8
1981	1.8	6.5	32.2	89.2	-	-	34.0	27.5
1982	26.6	94.9	34.0	94.3	-	-	60.6	49.1
1983	26.2	93.7	28.5	78.9	-	-	54.7	44.3
1984	5.3	19.0	34.4	95.4	-	-	39.8	32.2
1985	26.7	95.4	15.3	42.3	-	-	42.0	34.0
1986	25.7	91.9	35.2	97.4	52.5	88.2	113.4	91.7
1987	24.9	88.9	36.0	99.6	50.9	85.7	111.8	90.5
1988	27.9	99.8	29.3	81.1	55.0	92.4	112.2	90.8
1989	9.4	33.6	29.4	81.6	41.7	70.2	80.6	65.2
1990	26.2	93.5	28.6	79.4	47.1	79.2	101.9	82.5
1991	2.8	10.1	27.7	76.7	52.3	87.9	82.8	67.0
1992	26.3	93.8	25.7	71.1	48.3	81.2	99.9	80.9
1993	26.5	94.5	35.6	98.5	57.3	96.3	119.3	96.5
1994	10.5	37.4	26.9	74.6	55.7	93.7	93.1	75.4
1995	27.7	98.9	18.5	51.2	31.3	52.7	77.5	62.7
1996	0.7	2.5	9.1	25.2	15.0	25.1	24.7	20.0
1997	0.0	0.0	8.3	22.9	8.5	14.3	16.8	13.6
1998	0.7	2.6	4.3	11.8	30.8	51.7	35.8	28.9
1999	0.7	2.7	25.0	69.3	32.7	54.9	58.4	47.3
2000	0.8	2.9	22.9	63.5	59.0	99.2	82.7	66.9
2001	0	0	33.0	91.4	59.0	99.2	92.0	74.4
2002	0	0	32.3	89.5	59.1	99.4	91.5	74.0

[a] Unit 1 shut down on November 4, 1995 and its retirement was announced in July 17, 1998. Cooling-water flow at Unit 1 included only the service water system during 1996-2000 and beginning in 2001 no cooling water was used.

[b] No records of cooling-water flow were available for 1971-75; net electrical generation records were used to estimate flow.

foregone was the same as the proportion of cooling water not used by MPS in 2002.

The percentage of available total permitted flow used by MPS from March through May during the three-unit (1986-2002) period ranged between 13.6% in 1997 to 96.5% in 1993 (Table 32). During the larval winter flounder seasons of 1996-98, all three units were shut down, resulting in the smallest circulating water flows since Unit 2 went online in 1976 (see the Introduction section of this report for a summary of plant operational history). However, variable larval abundance also influenced the entrainment estimates. During 1993, the estimate was

the second lowest of the time-series because larval abundance at EN that year was by far the lowest recorded. In contrast, the 1997 entrainment estimate was higher than expected because larval abundance that year was quite high. Similarly, relatively high abundance of winter flounder larvae in both 1992 and 2001 in combination with moderately high water use resulted in the two largest entrainment estimates.

As in previous years, larvae in Stage 3 of development predominated in 2002 entrainment collections. This year, the fractions of total entrainment by developmental stage were 6.3% for Stage 2, 83.0% for Stage 3, 10.7% for Stage 4, and

TABLE 33. Annual abundance index (A parameter of the Gompertz function) with 95% confidence interval of winter flounder larvae in entrainment samples and total annual entrainment estimates and the volume of seawater entrained at MPS each year from 1976 through 2002 during the larval period of occurrence.

Year	A parameter	Standard error	95% confidence interval	Number entrained $(X\ 10^6)^a$	Seawater volume entrained $(m^3\ X\ 10^6)^a$
1976	1,656	32	1,588 - 1,724	121.4	628.8
1977	751	47	650 - 852	29.3	443.7
1978	1,947	352	1,186 - 2,706	80.0	390.3
1979	1,296	81	1,121 - 1,470	44.4	342.7
1980	2,553	37	2,475 - 2,632	167.6	562.2
1981	1,163	23	1,113 - 1,213	45.3	373.0
1982	2,259	36	2,184 - 2,334	163.5	637.9
1983	2,966	21	2,921 - 3,012	210.5	540.6
1984	1,840	47	1,741 - 1,939	84.3	508.1
1985	1,585	48	1,483 - 1,686	79.5	469.1
1986	903	31	837 - 968	122.6	1,063.7
1987	1,194	23	1,145 - 1,242	165.0	1,193.4
1988	1,404	42	1,315 - 1,493	183.7	1,173.1
1989	1,677	13	1,650 - 1,704	167.3	888.6
1990	1,073	25	1,021 - 1,125	132.6	1,173.5
1991	1,149	18	1,110 - 1,189	116.4	749.8
1992	3,974	76	3,812 - 4,136	492.3	1,075.7
1993	328	23	280 - 377	41.9	1,387.1
1994	1,709	38	1,626 - 1,790	173.2	919.7
1995	2,571	47	2,470 - 2,671	213.6	1,005.5
1996	1,388	78	1,222 - 1,554	51.2	471.6
1997	3,241	61	3,112 - 3,371	75.6	173.1
1998	2,039	47	1,939 - 2,139	84.1	358.3
1999	1,928	40	1,844 - 2,011	145.9	747.7
2000	3,688	170	3,330 - 4,046	331.1	1,002.7
2001	4,033	336	3,327 - 4,739	376.2	963.3
2002	1,142	18	1,105 - 1,179	118.8	990.8

0.1% for Stage 5; no Stage 1 larvae were found. For the period of 1983-2001, estimated composition was 3.3% of the combined total for Stage 1, 19.5% for Stage 2, 63.0% for Stage 3, 13.8% for Stage 4, and 0.5% for Stage 5. In 2002, fewer Stage 1, 2 and 4 larvae and more Stage 3 larvae were entrained in comparison to the long-term averages.

Effect of Entrainment on a Year-class. As noted previously in this section, both flow and larval densities vary over the larval season and among years (Fig. 43). The time-series of annual entrainment abundance (an index described by the A parameter of the Gompertz function) and the annual volume of seawater used at MPS have relatively similar CVs (51 and 44%, respectively). The entrainment estimates, however, have a CV of 72% because both larval densities and plant operation affect this estimate and they vary independently from year to year. A high larval abundance for a particular year is not functionally associated with high flow and vice versa. In fact, these two measures are not significantly correlated (Spearman's rank-order correlation; $r = -0.137$; $p = 0.496$; $n = 27$). Thus, cooling-water flow alone cannot be used as a measure of plant effect, as the fraction of Niantic River larval production available to entrainment and its relation to total larval
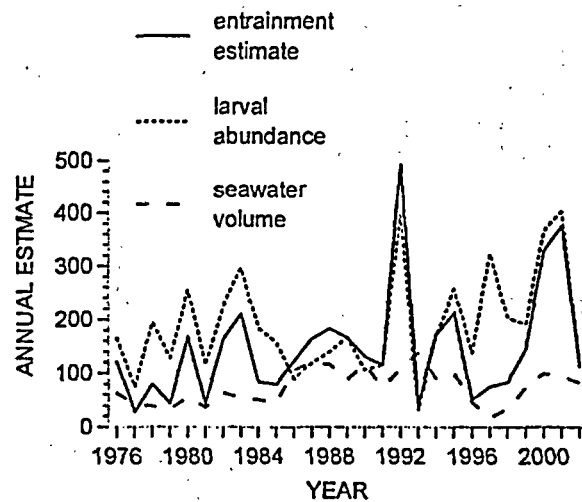
Fig. 43. Comparison between the annual estimates of larval winter flounder entrainment in millions, larval abundance at EN given as the annual mean density of larvae (A parameter of the Gompertz distribution), and seawater volume entrained in tens of millions of m^3 at MPS from 1976 through 2002. The annual mean larval density estimates were divided by 10 to fit the same scale as the other two measures.

entrainment changes from year to year. Variable annual rates of entrainment appear to be real and may be related to physical processes (e.g., precipitation, winds) that vary both intra- and interannually during the larval developmental period, when the abundance, distribution, natural mortality, and behavior of larvae are also variable.

The annual entrainment abundance index (A parameter of the Gompertz function) divided by the total volume of seawater entrained each year results in an index of the annual rate of entrainment, which is expressed as the number of larvae per unit volume of cooling water (Fig. 44). Based on the non-parametric Mann-Kendall test, the entrainment rate since 1976 has varied with no significant trend in slope (Z = −0.188; p = 0.851; n = 27). The entrainment rate time-series has a CV of 100%, which is greater than the CVs for density, flow, and numerical entrainment estimates. Larval production and availability in Niantic Bay remained stable, despite increases in MPS cooling-water use during 1986-95, the effective period of full three-unit operation. Also, reduced recruitment has not been the result of reduced larval abundance, because the latter has exhibited no trend after many years of MPS operation.

Relationships between larval entrainment estimates and various indices of juvenile abundance were examined to determine the effect of entrainment on a year-class. Annual entrainment estimates were significantly positively correlated with two abundance indices of age-0 juvenile winter flounder, including the median CPUE of age-0 fish taken in both early and late summer in the Niantic River (Table 34). Entrainment estimates were not significantly correlated with age-0 juveniles taken at TMP stations in late fall-early winter and age-1 fish collected in
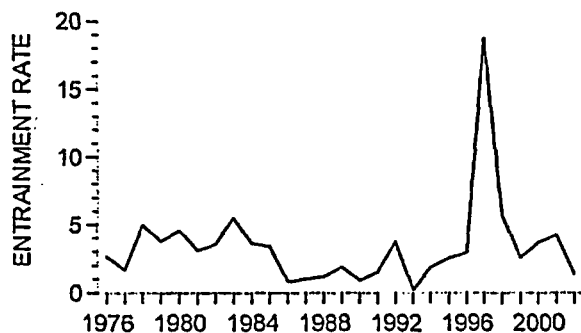
Fig. 44. Annual rate of entrainment of winter flounder larvae at MPS, defined as the annual mean density of larvae (A parameter of the Gompertz distribution) divided by the volume of seawater entrained each year from 1976 through 2002. The data were scaled to result in a numerical rate as shown.

either the lower or upper Niantic River during the adult winter flounder surveys. Although statistically significant, the form of the relationships between the entrainment estimates and the age-0 abundance indices was not obvious, however (Fig. 45).

Densities of winter flounder larvae 7 mm and larger taken at EN (an index of abundance rather than one of estimated entrainment impact) were significantly correlated with abundance of both Stage 3 and 4 larvae in the Niantic River (Table 25; Fig. 37). The significant correlation coefficients found between entrainment estimates and age-0 abundances were positive, implying no apparent entrainment effect and indicating that the more larvae available for entrainment, the more larvae that settled as demersal young in the Niantic River. 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 27).

Entrainment estimates appear to only reflect emerging year-class strength rather than being the most important factor affecting numerical abundance. Based on beam trawl catches in the Niantic River since 1983, most age-0 fish were produced in 1988, 1992, 1994, 1995, 2000, and 2001, with particularly weak year-classes formed during the mid-1980s, 1989, 1993, 1996 (although fish from this year-class appeared to be abundant during fall and winter), and 1997. The highest larval winter flounder entrainment estimates at MPS during the three-unit period were in 1992 and 1995 and lowest estimates were found in 1993 and 1996-98, years following the retirement of Unit 1 and during the extended shutdowns of Units 2 and 3. Subsequently, the second and third highest entrainment estimates occurred in 2000 and 2001. This suggests that in some years, year-class strength was strongly influenced by events occurring during early life history stages that resulted in abundant larvae and early settling juveniles. The examples of 1988 and 2000 (low mortality) and 1990 and 2001 (high mortality) also illustrated that mortality during the first summer on the nursery ground, which is totally unaffected by MPS operation, profoundly influences year-class strength and, ultimately, recruitment to adult stocks.

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 as well as larval densities in Niantic Bay. Hydrodynamic modeling (NUSCO 1976) and tidal-current drogue studies (NUSCO 1992b) showed that much of the

TABLE 34. Spearman's rank-order correlations between the annual estimates of larval winter flounder entrainment at MPS and the abundance indices of several post-entrainment early life history stages and a calculated apparent larval survival rate.

Index[a]	Niantic River early summer age-0 juveniles	Niantic River late summer age-0 juveniles	Fall-early winter river-bay age-0 juveniles	Niantic River winter-spring age-1 juveniles (lower river)	Niantic River winter-spring age-1 juveniles (upper river)
Annual estimate of entrainment at MPS	0.6037[b] 0.0080 ** 18	0.7947 0.0001 ** 19	0.2860 0.1567 NS 26	-0.2627 0.1949 NS 26	-0.0802 0.7227 NS 22

[a] Indices used correspond to those given on Tables 23, 24, and 33.
[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 ≤ 0.05, ** - significant at p ≤ 0.01), and
   number of annual observations (sample size).

condenser cooling-water used by MPS enters Niantic Bay from LIS. Other stocks are known to spawn both to the east and west of the bay (Howell and Molnar 1996, 1998) and results from tidal studies also indicated that a large number of winter flounder larvae entered Niantic Bay from LIS (NUSCO 1992a, 1992b). Mass-balance calculations were used to
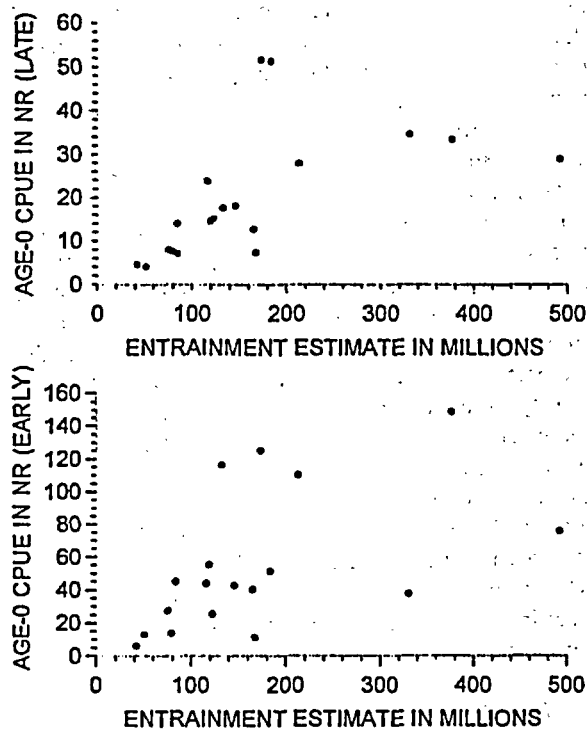
Fig. 45. Comparison between the 1-m beam trawl median CPUE of age-0 winter flounder taken at Niantic River stations LR and WA combined during both early (1985-2002) and late summer (1984-2002) with annual entrainment estimates of winter flounder larvae at MPS. (Note that the vertical scales differ among the graphs).

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 the present; 10 of these years (1986-95) occurred during full three-unit operation and 4 years (1996-99) with limited plant operation. The mass-balance model was not completed during 2000 because a reliable estimate of larval winter flounder mortality could not be obtained (DNC 2001a). Staff and two independent consultants for DNC conducted recent assessments of the mass-balance model, including sensitivity analyses. Output of this model, at worst, should have biases well within ±50% (DNC 2001c) and was far more sensitive to errors in larval density estimates than to errors in larval mortality rates.

Three potential inputs of larvae 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. 13). The few yolk-sac larvae collected annually in Niantic Bay suggested that minimal spawning and 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 MPS, 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 MPS, 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:

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

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)
  $NumEnt$ = 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 from 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:

$$Source\ or\ Sink = NB_{t+5} - NB_t + NumEnt + Mort - FromNR + ToNR \qquad (14)$$

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

$$5\text{-}day\ change = NB_{t+5} - NB_t \qquad (15)$$

Thus:
$$Source\ or\ Sink = 5\text{-}day\ change + NumEnt + Mort - FromNR + ToNR \qquad (16)$$

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

that the same conclusions were reached with either 5- or 10-day periods.

Daily entrainment estimates were based on data collected at station EN and the actual daily volume of condenser cooling water used at MPS. The daily entrainment estimates were summed over each 5-day period (*NumEnt*). Crecco and Howell (1990) determined annual stage-specific mortality rates for 1984-89, V. Crecco (CT DEP, Old Lyme, CT, pers. comm.) for 1990, and MPS Environmental Laboratory staff for the following years. 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); the same 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 16 above. These annual rates were modified to daily stage-specific mortality rates by assuming a duration of 10 days each 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 lower river after adjusting for the rate of flushing between that station and the river mouth. 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:

$$FromNR = 9.751 + 0.473(\text{daily density at station C}) \qquad (17)$$

The 95% CI for the slope ($r = 0.969$; $p = 0.001$; df = 8) 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 collections were taken by mooring the research vessel to the railroad bridge and making continuous oblique tows (NUSCO 1994a). Comparison of densities from the paired stations of NB and RM showed a poor relationship. Therefore, the Gompertz density curve (Eq. 3) was used to estimate daily densities at the two stations. 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:

$$ToNR = 128.149 + 2.073(NB_i) \qquad (18)$$

The 95% CI for the slope ($r^2 = 0.705$; $p = 0.001$; $df = 406$) 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 ($NB_i = 0$), the term *ToNR* was conservatively set to zero. The term *Source or Sink* in Equation 16 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 MPS each year from the Niantic River. If *FromNR* can support the number of larvae entrained by MPS, 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 MPS three-unit operation, the results of which will be provided below.

The results for each 5-day period of the 2002 larval winter flounder season are provided as an example of these calculations (Table 35). Computations for other years were given in NUSCO (1993, 1994a, 1995, 1996a, 1997b, 1998, 1999, 2000) and DNC (2002).

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 in the 2002 larval season from positive to negative when the abundance of larvae in the bay began to decline between April 6 and April 11 (Table 35). A negative *Source or Sink* term indicated a net loss of larvae from Niantic Bay during the first part of the larval season, but beginning with the 5-day period starting on March 17, the *Source or Sink* term became positive. This indicated that larvae from other sources in LIS were required to support the change in larval abundance and balance the equation. Timing of this change in the *Source or Sink* term was similar to previous years (NUSCO 1993, 1995, 1996a, 1997b, 1998, 1999, 2000; DNC 2002), except for an earlier date of February 25 in 1993 (NUSCO 1994a). Considerably fewer larvae were entrained (*NumEnt*) than were imported from LIS (i.e., positive *Source or Sink*), starting in mid to late March. Also in 2002, the weekly estimates of *NumEnt* were considerably smaller than the number entering the river from the bay (*ToNR*).

The proportion of entrained larvae from the Niantic River during each 5-day period of the season 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 MPS 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 2002, an estimated 23.1 million larvae were entrained

TABLE 35. Results of mass-balance calculations during each 5-day period in 2002.

Start of 5-day period	5-day change (X 10⁶)	Number entrained (NumEnt) (X 10⁶)	Loss due to mortality (Mort) (X 10⁶)	Number from the Niantic River (FromNR) (X 10⁶)	Number to the Niantic River (ToNR) (X 10⁶)	Source or Sink (X 10⁶)
2-15	0.0[a]	0.0	0.0	26.1	0.0	-26.1
2-20	0.0	0.0	0.0	27.9	0.0	-27.9
2-25	0.1	0.0	0.0	27.2	6.7	-20.4
3-02	0.3	0.1	0.1	24.7	6.9	-17.4
3-07	1.0	0.3	0.3	21.3	7.5	-12.3
3-12	2.1	0.9	0.2	17.8	9.0	-5.6
3-17	3.2	2.5	0.5	14.4	11.8	3.7
3-22	3.8	4.6	0.9	11.4	15.6	13.3
3-27	3.4	7.2	1.3	9.0	19.6	22.5
4-1	2.3	9.6	1.6	7.0	22.8	29.4
4-06	1.0	10.0	1.8	5.4	24.8	32.2
4-11	-0.3	10.9	1.9	4.2	25.3	33.6
4-16	-1.2	11.2	1.8	3.3	24.6	33.0
4-21	-1.7	10.3	1.6	2.6	23.0	30.6
4-26	-1.9	9.9	1.4	2.1	21.0	28.3
5-01	-1.9	8.6	1.2	1.7	18.9	25.1
5-06	-1.8	7.2	1.0	1.4	16.9	21.9
5-11	-1.6	5.9	0.8	1.1	15.0	19.0
5-16	-1.3	4.8	0.7	1.0	13.4	16.5
5-21	-1.1	3.8	0.5	0.9	12.0	14.4
5-26	-0.9	3.0	0.4	0.8	10.9	12.7
5-31	-0.7	2.4	0.3	0.7	10.0	11.3
6-05	-0.6	1.9	0.3	0.7	9.3	10.2
6-10	-0.5	1.5	0.2	0.6	8.7	9.3
6-15	-0.4	1.1	0.2	0.6	8.3	8.6
6-20	-0.3	0.9	0.1	0.6	7.9	8.1
6-25	-0.2	0.7	0.1	0.6	7.6	7.6

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

from the river, which represented 14.2% of the annual total entrained (Table 36).

The potential impact of larval entrainment on the winter flounder population depends upon the age of each larva at the time it is entrained, as 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 37). In most years, including 2002, larvae in Stage 3 of development dominated the Niantic River larvae entrained. However, Stage 2 larvae in 1984 and 1985 and Stages 1 and 2 larvae in 1996 predominated as the larvae attributable to the Niantic River. Consistently, however, most entrained Stage 3 and 4

larvae were determined as originating from sources other than the Niantic River.

The interpretation of mass-balance calculation results has been substantiated by results from several specially designed studies. Some of the larger larvae from other areas enter the Niantic River during a flood tide, forming a stable or increased frequency of larger (≥6 mm) size-classes during some years (Fig. 18). In special bay-wide sampling in April and May of 1991 (NUSCO 1992a), months during which about 75% of Stage 3 larvae are entrained, more larvae entered Niantic Bay from LIS east of Millstone Point and passed by the MPS 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. To determine how consistently more Stage 3 larvae were entrained during flood tides than during ebb tides, Stage 3 larval abundances at EN were examined by tidal stage (combined data, 1983-99). Data were restricted to collections when Stage 3 larvae were present and were tested with the Wilcoxon two-sample test. Stage 3 larval densities

TABLE 36. 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 2002.

Year	Total entrainment (X 10⁶)	Niantic River larval entrainment (X 10⁶)	% entrainment attributed to the Niantic River
1984	84.3	32.0	37.9
1985	79.5	27.8	34.8
1986	122.6	28.1	22.4
1987	165.0	41.7	25.3
1988	183.7	39.5	21.3
1989	167.3	33.5	20.0
1990	132.6	38.5	28.9
1991	116.4	35.2	30.2
1992	492.3	80.3	16.3
1993	41.9	6.0	13.9
1994	173.2	50.4	28.8
1995	213.6	77.9	36.4
1996	51.2	30.4	58.8
1997	75.6	9.3	12.3
1998	84.1	25.9	30.9
1999	145.9	32.9	22.5
2000	331.1	87.6	26.3
2001	376.2	80.6	21.4
2002	119.3[a]	23.1	14.2

[a] Total entrainment value differs from that reported on Table 33 due to averaging of values during calculations.

were significantly (p < 0.001) higher during a flood tide (570 samples) than an ebb tide (583 samples), indicating that most entrained Stage 3 larvae likely entered Niantic Bay from LIS.

Comparison of the Mass-balance Model and DNA-based Entrainment Fraction Estimates. Estimates of the weekly fraction of entrained winter flounder originating from the Niantic River were calculated using results from both the mass-balance model and the larval DNA analysis reported by Crivello (2002; In preparation) for 2001 and 2002. He examined 536 mostly Stage 1 and 2 larvae collected from three source populations (Niantic River, 164; Thames River, 174; Westbrook, 198) in 2001. In 2002, 635 stage 1 and 2 larvae were examined from the same source populations (Niantic River, 189; Thames River, 222; Westbrook, 224). Based on their genetic characteristics, he then assigned 1,067 Stage 2 through 4 entrained larvae to a source population in 2001 and 969 Stage 2 through 4 larvae in 2002. The entrained larvae were collected based on a stratified random sampling model, with larger sample sizes taken during weeks of peak entrainment. Overall in 2001, 24% of the larvae examined were assigned to the Niantic River source population, 21% were from the Thames River, 34% were from Westbrook, and the remainder could not be classified specifically or were from unknown sources. As in the mass-balance model, Niantic River larvae were more predominant in March and April,

but less so in May and June of 2001. In 2002, 12% of the entrained larvae examined were assigned to the Niantic River, 8% to the Thames River, 59% to Westbrook, and 21% were unclassified or from unknown sources.

A more precise estimate of loss to the Niantic River population was possible by applying the fraction of larvae assigned to the river by Crivello (2002; In preparation) for each sampling date to the estimated entrainment during a comparable time period. This also allowed a comparison with the mass-balance model results of total Niantic River entrainment in 2001 (DNC 2002) and this year. Estimated numbers of winter flounder larvae entrained during each 5-day interval (*NumEnt* in Table 35) from March 7 through June 25 were multiplied by the calculated fraction of larvae originating from the Niantic River based on the DNA analysis of Crivello (2002; In preparation) or the mass-balance model. Because DNA sampling intervals did not match exactly with time intervals used to estimate total entrainment of larvae, DNA fraction estimates were interpolated to match 5-day entrainment estimates from Table 35. Also, the first and last estimates of Niantic River entrainment based on DNA analysis results are the summed totals for all dates prior to and after the first and last DNA sample dates, respectively. Multiplying the DNA fraction estimates by the entrainment estimates produced estimates of the 5-day totals of larvae entrained that originated from the Niantic River for 2001 (DNC

TABLE 37. Estimated number of winter flounder larvae entrained at MPS by developmental stage from the Niantic River and other sources, based on mass-balance calculations for 1984 through 2002.

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	14.9	13.9	3.1
	Other	0.1	23.9	24.7	3.5
1985	Niantic River	3.4	17.4	6.9	0.4
	Other	0.8	10.4	34.1	6.4
1986	Niantic River	0.7	7.3	15.2	4.3
	Other	1.5	24.0	59.1	10.7
1987	Niantic River	0.8	15.2	23.9	1.9
	Other	0.6	30.1	85.0	7.5
1988	Niantic River	3.9	9.4	24.4	1.5
	Other	1.1	7.6	113.2	22.7
1989	Niantic River	2.8	11.2	19.0	0.5
	Other	4.1	40.4	81.1	8.4
1990	Niantic River	0.9	6.2	27.6	3.7
	Other	0.8	12.1	72.4	9.0
1991	Niantic River	0.2	3.6	26.7	4.7
	Other	0.6	8.8	65.4	6.4
1992	Niantic River	5.6	10.1	55.7	8.8
	Other	30.0	54.0	295.0	33.0
1993	Niantic River	0.3	1.2	3.8	0.5
	Other	1.2	4.9	22.4	7.5
1994	Niantic River	2.8	12.3	28.7	6.1
	Other	2.6	24.2	80.1	16.5
1995	Niantic River	0.6	6.8	55.6	14.8
	Other	1.1	13.5	104.0	17.4
1996	Niantic River	13.2	12.6	4.1	0.3
	Other	1.6	4.2	11.5	3.9
1997	Niantic River	0.7	1.6	6.1	0.9
	Other	5.2	12.1	43.0	6.1
1998	Niantic River	0.9	5.0	16.3	3.8
	Other	0.3	2.8	37.9	17.1
1999	Niantic River	2.8	9.0	17.2	3.9
	Other	7.7	21.4	64.8	19.1
2000	Niantic River	3.5	20.8	46.3	16.7
	Other	6.0	55.7	139.3	42.9
2001	Niantic River	11.1	25.7	33.7	11.4
	Other	0.0	65.8	160.8	67.9
2002	Niantic River	0.3	0.9	14.7	1.2
	Other	0.0	6.0	82.6	13.7

2002) or 2002 (Table 38). These totals were then summed to estimate the total number of Niantic River larvae entrained. The mass-balance estimated fraction of entrained winter flounder larvae originating from the Niantic River was derived by calculating the fraction of entrained larvae originating from the Niantic River over each 5-day period beginning on March 7, when larvae first appeared in samples, and extending through June 25. The 5-day fractions were then multiplied by the number of larvae entrained during each 5-day period. These products were then summed to produce a total estimate of entrained larvae attributable to the Niantic River stock. Correlation analysis was used to examine the strength of the relationship between DNA analysis estimates and mass-balance modeling results and a t-test for two population means (method for paired comparisons using weekly estimates; SAS Institute Inc. 1990b) was used to investigate the significance of the difference between the means of

the two independent estimates. Both analyses indicated that weekly fractions of entrained winter flounder larvae originating from the Niantic River were higher early in the larval period and declined over the course of the spring in 2001 (Fig. 46). Weekly estimates of entrained winter flounder larvae originating from the Niantic River determined by the DNA analysis were significantly positively correlated with estimates from the mass-balance model (r = 0.74; p = 0.003) in 2001. Further, in 2001 the weekly estimates were not significantly different (paired t-test; p = 0.33). On average across the entire winter flounder larval season in 2001, the fraction of entrained larvae attributable to the Niantic River was 0.22 and 0.21 as estimated by the DNA analysis and mass-balance modeling, respectively (DNC 2002). These proportions translated into 82.45 million Niantic River larvae entrained as determined by the DNA analysis and 80.65 million Niantic River larvae entrained estimated by the mass-balance modeling

TABLE 38. Five-day estimates of the total number of winter flounder larvae entrained in 2002 (see Table 35), fraction attributable to the Niantic River based on the genetic DNA analysis of Crivello (In preparation), the number of Niantic River larvae entrained based on this analysis, fraction attributable to the Niantic River based on mass-balance modeling results, and the number of Niantic River larvae entrained based on mass-balance modeling results.

5-day period	5-day entrainment estimate (10^6)	DNA-based fraction[a]	Total (10^6) attributed to the Niantic River from genetic study[b]	Mass-balance model fraction	Total (10^6) attributed to the Niantic River from the mass-balance model
2 March	0.062			1.000	0.062
7 March	0.341			1.000	0.341
12 March	0.885			1.000	0.885
17 March	2.529			0.797	2.016
22 March	4.555			0.462	2.104
27 March	7.202			0.285	2.056
1 April	9.624	0.054	2.052*	0.192	1.850
6 April	10.045	0.074	1.157	0.144	1.451
11 April	10.902	0.078	1.273	0.111	1.218
16 April	11.173	0.082	1.303	0.091	1.013
21 April	10.314	0.153	2.244	0.078	0.805
26 April	9.884	0.134	1.741	0.068	0.671
1 May	8.570	0.115	1.282	0.062	0.534
6 May	7.208	0.291	1.054	0.059	0.424
11 May	5.911	0.110	0.843	0.057	0.337
16 May	4.787	0.104	0.645	0.056	0.269
21 May	3.829	0.097	0.482	0.056	0.216
26 May	3.046	0.250	0.985	0.057	0.174
31 May	2.380	0.125	0.388	0.059	0.140
5 June	1.742	0.000	0.000	0.060	0.112
10 June	1.471	0.533	2.417*	0.062	0.091
15 June	1.143			0.064	0.073
20 June	0.888			0.066	0.058
25 June	0.690			0.067	0.046
Mean fraction		0.123		0.142[c]	
Total	119.3		14.669		16.945

[a] Bolded values are actual fraction determined from DNA analysis (Crivello In preparation), while non-bold values are interpolated.
[b] * indicates values are products of the summed tails of the entrainment estimates and the DNA fraction.
[c] Geometric mean.
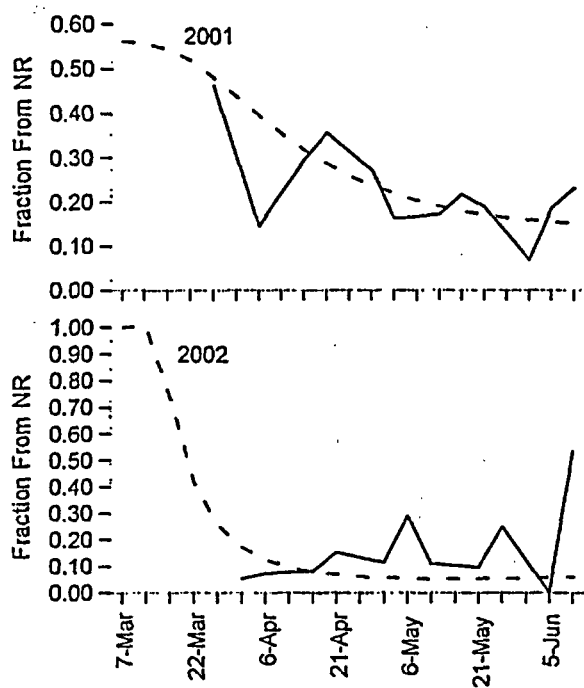
Fig. 46. Fraction of entrained winter flounder larvae attributed to the Niantic River as determined by DNA analysis (solid line) and mass-balance modeling (dashed line) in 2001 and 2002.
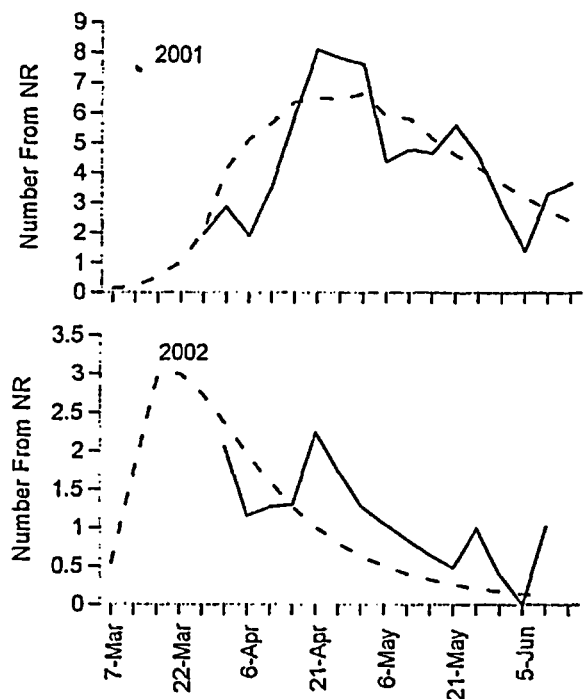
Fig. 47. Numbers of entrained winter flounder larvae attributed to the Niantic River as determined by DNA analysis (solid line) and mass-balance modeling (dashed line) in 2001 and 2002.

approach. While the DNA analysis and mass-balance modeling techniques provided similar estimates of the proportion and number of entrained larvae originating from the Niantic River in 2002, the relationships between these estimates were only apparent for numbers of Niantic River larvae entrained (Figs. 46 and 47). Estimates of weekly numbers of Niantic River larvae entrained derived from the DNA analysis and mass-balance model were significantly positively correlated (r = 0.62; p = 0.0135). No significant correlation existed between estimates of weekly fractions of Niantic River larvae entrained (r = −0.38; p = 0.160). Mass-balance estimates of the proportion of Niantic River flounder entrained were highest during the early part the 2002 larval season, while the DNA analysis showed higher proportions later in the season (Fig. 46). In fact, the mass-balance model attributed 100% of winter flounder entrained as originating from the Niantic River in early March 2002. Unfortunately, sampling for the DNA analysis did not begin until early April, precluding any comparison between the two techniques during the early part of the larval season. The mass-balance model showed that numbers of Niantic River larvae entrained peaked in late March and early April, agreeing with results of DNA analysis (Fig. 47). On

average across the entire winter flounder larval season in 2002, the fractions of entrained larvae attributable to the Niantic River were 0.12 and 0.14 as estimated by the DNA analysis and mass-balance modeling, respectively. These proportions translated into 14.67 million Niantic River larvae entrained as determined by the DNA analysis and 16.95 million Niantic River larvae entrained estimated by the mass-balance modeling approach (Table 38).

In a separate study having the same objective of identifying the stock of origin for entrained larvae, Moran (2002), using a technique analyzing individual larvae for a number of microelements, estimated that from about 10 to 20% of entrained larvae in 2001 originated from the Niantic River. However, due to the demands of this technique, fewer larvae were analyzed than in the Crivello (2002) study, a total of 105 larvae from the same source sites and 142 entrained larvae. Also, the entrained larvae were only collected during the period of peak entrainment (mid-April through mid-May). For these same weeks, 26% of the entrained larvae examined by Crivello (2002) were assigned to the Niantic River population.

Estimated Production Loss from the Niantic River Stock. Estimates of Niantic River larvae entrained by developmental stage were compared to

annual abundance estimates for each larval stage in the river. The latter were computed by applying specific mortality rates to each stage of early life history, beginning with total annual egg production estimates (Table 8), which have declined in recent years (see Spawning Stock Size and Egg Production, above). This allowed for the determination of percent production loss (i.e., conditional entrainment mortality, termed ENT) 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 and was discussed in NUSCO (1997b). As noted previously, 20% of the Niantic River discharge passed through MPS during full three-unit operation; with the retirement of Unit 1, this fraction should now be approximately 15%. For theoretical full three-unit operation, 20% of daily estimates of Stage 1 larval densities at station C were used to calculate the entrainment of these larvae from the Niantic River in the estimation of production loss, but during periods of reduced plant operation the estimates were proportionally decreased, 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 actual entrainment sampling densities. The equivalent percentage of the Niantic River winter flounder egg production entrained in 2001 and 2002 were 35.4% and 19.7%, respectively, which were the second and sixth largest conditional mortality rates (ENT) of the 18-year time-series (1995 being the highest at 38.5%). Note that the value for 2000 is not considered reliable due to the lack of a larval mortality estimate for that year (DNC 2001a).

The largest equivalent egg production losses calculated for most recent years may be inflated due to greater egg survival to hatching since 1995 (Fig. 16). The relationship between egg production and Stage 1 larval abundance indicated that survival during recent years may have been as much as four times that prior to 1995. This makes estimates of ENT for the past 6 years conservatively high. In the calculation of annual production loss, the same egg survival rate was used for the entire series. However, if egg survival since 1995 was much greater than occurred previously (1984-94), this process would proportionally reduce the estimated conditional mortality (Fig. 48). This example was provided by arbitrarily increasing egg survival by from 20 to 50%; annual larval mortality rates remained the same for the initial and revised estimates. The revised estimates were more similar to those of previous years and bracketed the geometric mean of the ENT time-series.

The total entrainment estimate in 2002 was the lowest since 1998, representing a 68% decrease from the 2001 estimate (Table 36). Nevertheless, entrainment estimates from 1996 through 1998 were the lowest since Unit 3 went online, except for 1993, when larvae were very low in abundance. Since 1977, annual egg production estimates in the river were the lowest from 1995 through 2002 (Table 8), but near-average (1996, 2001 and 2002) or high (1995 and 1997-99) larval abundance in the bay (Table 10) suggested increasing survival during early life history, which probably resulted from several factors. As noted above, egg survival for these years was apparently better than usual, as many more Stage 1 larvae were found than were expected (Fig. 16). Larval mortality rates in the river during 1995, 1997, 1998, and 1999 were among the lowest estimates found, although the mortality rate in 1996 was greater than the long-term average (Table 15). The larval recruitment indices for 1995-99 and 2001 at station EN (Fig. 23) were the highest calculated while the 2002 value was also higher than expected. Larval development is directly related to growth rate and shorter larval periods likely result in better survival (Houde 1987). For example, during 1995 and 1997-99, larval growth and development were among the fastest in comparison to previous years, as indicated by greater annual growth rates in the river (Table 14), had the largest mean length during the first 2 weeks of April (Fig. 20), and some of the earliest dates of peak abundance (Fig. 21). Higher larval survival in these years, which appeared to be related to faster growth and development, apparently also resulted in more Stage 3 larvae available for entrainment during the same years.

Stochastic Simulation of the Niantic River Winter Flounder Stock

Modeling Strategy and Background. The stochastic population dynamics model (SPDM) developed to simulate the Niantic River winter flounder stock includes a life-cycle module designed to generate new year-classes of flounder based on the three-parameter Ricker form of the stock-recruitment relationship (SRR) given as Equation 7. Although the SRR equation does not appear explicitly in the model formulation, the SPDM equation, which computes mortality through the first year of winter flounder life, was derived from the SRR and assumes a Ricker-type form of recruitment. Beyond that point (i.e., age-1) in

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 (ENT) by developmental stage from 1984 through 2002. Numbers of larvae entrained from the Niantic River were based on mass-balance calculations (see text).

Year	Stage of development	Niantic River abundance[a] ($\times 10^6$)	Entrainment from the Niantic River[b] ($\times 10^6$)	% production entrained (ENT)
1984	Stage 1	2,877	9.8	0.3
	Stage 2	688	14.9	2.2
	Stage 3	338	13.9	4.2
	Stage 4	236	3.1	1.3
	Total		41.7	7.9
1985	Stage 1	3,119	15.1	0.5
	Stage 2	746	17.4	2.4
	Stage 3	367	6.9	1.9
	Stage 4	256	0.4	0.2
	Total		39.8	4.9
1986	Stage 1	2,438	10.9	0.4
	Stage 2	685	7.3	1.1
	Stage 3	357	15.2	4.2
	Stage 4	249	4.3	1.7
	Total		37.7	7.5
1987	Stage 1	2,943	33.0	1.1
	Stage 2	827	15.2	1.8
	Stage 3	431	23.9	5.5
	Stage 4	301	1.9	0.6
	Total		74.0	9.1
1988	Stage 1	4,802	79.6	1.7
	Stage 2	718	9.4	1.3
	Stage 3	259	24.4	9.4
	Stage 4	186	1.5	0.8
	Total		114.9	13.2
1989	Stage 1	3,920	63.6	1.6
	Stage 2	547	11.2	2.0
	Stage 3	180	19.0	10.6
	Stage 4	121	0.5	0.4
	Total		94.3	14.6
1990	Stage 1	1,904	31.7	1.7
	Stage 2	782	6.2	0.8
	Stage 3	215	27.6	12.8
	Stage 4	185	3.7	2.0
	Total		69.1	17.3
1991	Stage 1	3,505	12.6	0.4
	Stage 2	2,446	3.6	0.1
	Stage 3	744	26.7	3.6
	Stage 4	603	4.7	0.8
	Total		47.6	4.9
1992	Stage 1	2,145	22.0	1.0
	Stage 2	899	10.1	1.1
	Stage 3	331	55.7	16.8
	Stage 4	265	8.8	3.3
	Total		96.6	22.3
1993	Stage 1	1,221	10.9	0.9
	Stage 2	631	1.2	0.2
	Stage 3	114	3.8	3.3
	Stage 4	80	0.5	0.6
	Total		16.4	5.0

TABLE 39. (cont.).

Year	Stage of development	Niantic River abundance (X 10^6)	Entrainment from the Niantic River[a] (X 10^6)	% production entrained (ENT)
1994	Stage 1	1,801	26.1	1.4
	Stage 2	988	12.3	1.2
	Stage 3	496	28.7	5.8
	Stage 4	431	6.1	1.4
	Total		73.2	9.9
1995	Stage 1	897	37.9	4.2
	Stage 2	566	6.8	1.2
	Stage 3	235	55.6	23.7
	Stage 4	157	14.8	9.4
	Total		115.1	38.5
1996	Stage 1	450	23.1	5.1
	Stage 2	159	12.6	7.9
	Stage 3	49	4.1	8.3
	Stage 4	33	0.3	0.9
	Total		40.0	22.2
1997	Stage 1	799	5.6	0.7
	Stage 2	525	1.6	0.3
	Stage 3	274	6.1	2.2
	Stage 4	155	0.9	0.6
	Total		14.2	3.8
1998	Stage 1	539	5.4	1.0
	Stage 2	321	5.0	1.6
	Stage 3	150	16.3	10.8
	Stage 4	87	3.8	4.4
	Total		30.5	17.8
1999	Stage 1	454	16.8	3.7
	Stage 2	174	9.0	5.2
	Stage 3	109	17.2	15.8
	Stage 4	92	3.9	4.3
	Total		46.9	29.0
2000[c]	Stage 1	745	5.8	0.8
	Stage 2	274	20.8	7.6
	Stage 3	150	46.3	30.8
	Stage 4	123	16.7	13.6
	Total		89.6	52.7
2001	Stage 1	340	11.4	3.4
	Stage 2	276	25.7	9.3
	Stage 3	212	33.7	15.9
	Stage 4	169	11.4	6.8
	Total		82.2	35.4
2002	Stage 1	627	25.7	4.1
	Stage 2	155	0.7	0.5
	Stage 3	106	14.6	13.8
	Stage 4	84	1.1	1.3
	Total		42.1	19.7
Geometric mean				13.9

[a] Values differ from those given in DNC (2002) because of differences in annual egg production estimates.
[b] Entrainment estimates attributed to the Niantic River are higher than those in Tables 36 and 37 due to adjustments made for Stage 1 entrainment.
[c] Values listed for 2000 are based on estimated stage-specific mortality rates due to lack of actual mortality estimates for that year.
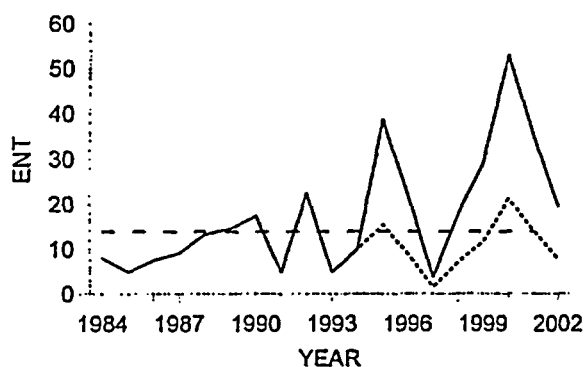
Fig. 48. Comparison of annual estimates of ENT (MPS entrainment conditional mortality rate) of Niantic River winter flounder larvae as given in Table 39 (solid line) and with a revised calculation based on increasing egg survival from 20% to 50% during 1995-2002 (dashed line). The horizontal line indicates the geometric mean of the values found in Table 39.

the life-cycle simulation, the population model simply keeps track of the annual changes in abundance of each year-class resulting from natural mortality and fishing, together with growth in size and increased fecundity. Population updates take place 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; Goodyear and Christensen 1984; Reed et al. 1984). In the SPDM, 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, which helps to conceptualize the model, and simplifies the computer code when describing the fish population either as numbers of fish or as biomass (allowing for size variation within each age-class). 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*). The Niantic River winter flounder SPDM and its application to impact assessment work at MPS was described in Lorda et al. (2000).

Model Components. The solid-line boxes in Figure 49 describe the SPDM as presently used, while the box with dashed lines corresponds to the previously described mass-balance calculations dealing with spatial larval distribution and entrainment loss estimates, which are not an integral

part of the SPDM. Model components are briefly described in Figure 49, with the more important components, such as the one labeled age-0 cohort (box 3) and the two random inputs (boxes 3A and 3B), described in more detail below.

The most critical aspects in the formulation of a stock-recruitment based population model are the specific equation and parameter estimates used to calculate total mortality during the first year of life of the fish (i.e., from egg to 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 to maturation age). This involved the extension of stock-recruitment theory, which was developed for fish that spawn only once, to iteroparous fish with multi-age spawning stocks. The form of the equation as used in the present model is:

$$Z_{0,t} = \log_e(FEC) + \log_e(ASF) - \log_e(\alpha) + n_t - (\phi \times WT_t) - Z_{1,2} + (\beta \times P_t) \quad (19)$$

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 $\alpha = \alpha_0$, the theoretical rate of increase in the absence of fishing, estimated from winter flounder life history parameters independently of the Niantic River stock and recruitment data (the derivation of α_0 was discussed in the Stock and Recruitment section); 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 rate through the immature age-classes; and the last term ($\beta \times P_t$) is a feed-back mechanism that simulates 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).

When simulating plant effects such as entrainment, the parameter α in Equation 19 is reduced by a factor

1 INPUT DATA and Simulation Parameters

Random Seawater Temperature 3A

2 Fraction of Larval PRODUCTION LOST (Larval Distribution and Mass-balance Estimates)

3 AGE-0 COHORT Natural & Compens. Mort. Entrainment Losses

Random Noise 3B

4 ADULT COHORT CALCULATIONS Natural & Fishing Mortality Impingement Losses

6 ADULT FISH Annual Popul. Projections

7 Probabilistic Risk Assessment

5 Annual SPAWNING STOCK and Projected Egg Production

OUTPUT

Fig. 49. Diagram of the stochastic fish population dynamics simulation model (SPDM) used to assess long-term effects of larval winter flounder entrainment at MPS. The computer implementation of the SPDM is in Fortran77. Brief descriptions of the computer program components referenced in the diagram follow:

1. Subprogram that process the input data files (see Table 40 for a sample of the main input file). The subprogram also verifies data ranges and stores parameters and data used in each simulation within common memory.

2. Auxiliary programs (not an integral part of SPDM) that estimate the annual fraction of Niantic River flounder production lost to larval entrainment. These estimates are based annual larval production, cooling water flow at the plant intakes, tidal exchange rates between Niantic Bay and River, and mass-balance calculations of weekly larval densities near the plant intakes in Niantic Bay.

3. Subprogram that calculates the number of young fish surviving to the end of their first year of life. Natural and compensatory mortality are described using a Ricker-type recruitment equation incorporating a temperature-dependent term (Eq. 7). Survival reduction due to larval entrainment is explicitly described as additional mortality (Eq. 19).

3A. Subroutine that generates random normal deviates of mean water temperature with given mean and variance derived from water temperature data during critical larval development (input to Eq. 19 in box 3).

3B. Subroutine that generates random standard normal deviates to simulate random variability in the natural mortality rates of early life stages of winter flounder (input to Eq. 19 in box 3).

4. Subprogram that updates the numbers of adult fish in each age group at the end of each time-step or year. This process is implemented with a Leslie matrix, which accepts random variation in selected parameters. Natural and fishing mortality rates in addition to fish losses caused by impingement are used in the calculations.

5. Subroutine that calculates annual egg production from fecundity-at-age and the annually updated age structure calculated in box 4. This annual egg production is the population feedback that starts each new cycle in box 3.

6. Subroutine that summarizes the adult population numbers and annual catch as biomass by age-class.

7. Subprogram that conducts a probabilistic risk analysis when the population dynamics is simulated as an stochastic process. Reference biological points "at risk" of being exceeded are provided with the initial input.

equivalent to the projected reduction in annual production. Myers et al. (1999) described a similar scaling of α to deal with the estimation of α under low spawner abundance. Stochasticity in the winter flounder model (Fig. 49) has two annual components: a random term (n_t) that represents uncertainties associated with the estimate of Ricker's α_0 parameter and environmental variability in the form of random deviations ($\phi \times WT_t$) from the long-term mean February water temperature. These two components of annual variability are incorporated into the calculation of each new year-class via the mortality from egg to age-1 (Eq. 19). The random noise term n_t is simulated as independent random variates from a normal distribution with zero mean and variance equal to σ^2. The value of σ was estimated during the model calibration runs as the amount of variance required to generate short time-series of projected spawning stocks with a similar CV as that observed in field data. Similarly, the term ($\phi \times WT_t$) represents the effect of annual variability of February water temperatures on larval survival. This effect becomes random when the input February water temperatures are generated as independent random variates from a normal distribution with mean and variance equal to the mean and variance of February water temperatures at the MPS intakes from 1977 through the current year.

. A stochastic simulation of winter flounder population dynamics also provides the proper framework for applying probabilistic risk assessment methodology (box 7 of Fig. 49). The 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. This simulation is a relatively simple application of Monte Carlo methods because only one population parameter, the age-0 survival rate, is stochastic. For this application of the SPDM, 100 replicate simulations per scenario were used, which was judged to be sufficient, given the amount of variability found (NUSCO 1990). Also, as noted in DNC (2001b), increasing the number of replicates to 500 or 1,000 produced very little additional variation in the SPDM output.

Monte Carlo replications of the projected stock size for any given year can also be used to derive the empirical cumulative frequency distribution (cdf) for that year without assuming any particular statistical distribution. This methodology was used to assess the risk of postulated stock reductions resulting from entrainment and impingement at MPS. The probabilities of stock reductions below some specified reference size were empirically derived from the cdf

of stock size replications in selected years. Additionally, approximate 95% CIs associated with point estimates of annual stock sizes could also be calculated.

Model 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 19) applied reasonably well to the Niantic River winter flounder stock. A second assumption was that the β and ϕ parameters of the SRR could be estimated from annual time-series of field data and that, in particular the value of α_0, which was based on life history parameters only, was a reasonable estimate. Although the population was not assumed to be at steady state, the average fecundity and annual 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 for winter flounder have been 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.

Another assumption is that no temperature trend or large-scale environmental changes (e.g., global warming) has or will occur during the years simulated in each population projection. However, this assumption may not be entirely accurate as there has been a consistent pattern of warmer than average water temperatures in LIS during late winter and early spring in recent years (Foertch 2000). Because these periods coincide with winter flounder spawning, egg incubation, larval development, metamorphosis and settling, and early demersal life, temperature-dependent effects could affect the reproductive success of winter flounder and its ultimate population size (see Formation of Winter Flounder Year-class Strength for a discussion of temperature effects). Finally, no trends in fishing rate were assumed to occur beyond 1999 in these simulations. However, this last assumption is very unrealistic, given the

recent 20-year history of dramatic changes in fishing rates that is presented below.

Model Input Data. The dynamics of the Niantic River winter flounder stock were simulated using the SPDM under a real-time scenario running from 1960, a decade before the operation of Unit 1, to 2060, which encompasses a recovery period 15 years after the projected shutdown of Unit 3 in 2045. This time period assumes that a 20-year license renewal for Units 2 and 3 will have been received from the NRC. The scenarios used power plant effects that were based on actual or projected operation in each year concurrently with estimates of F based on historic or projected rates of commercial exploitation and sport fishing for winter flounder in Connecticut. Parameters used in the SPDM include: F, with an additional instantaneous mortality of 0.01 that accounts 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; discussed further below) estimated from the mass-balance calculations given previously in Tables 35-37 and 39; 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 (1971-2045) that corresponded to MPS operation. Based on improvements to the fish return at Unit 3 and the installation of a fish return at Unit 2, even this small value may be an overestimate of current impingement impact. Other data, rates, and inputs to the SPDM are summarized in Table 40 and included the number of age-classes, age-specific rates of maturation, natural mortality, average weight and fecundity at age, the estimated value of α_0, long-term average estimates of β and ϕ, February water temperature statistics, and specific factors set for each simulation.

Because the ability of a fish stock to withstand additional stress is reduced by fishing mortality (Goodyear 1980), the actual fishing exploitation rates on the spawning stock are critically important in simulations of long-term effects of larval entrainment, particularly as F has been 3 to 7 times as much as natural mortality (0.2) of adult winter flounder in recent decades and remains the dominant force driving the dynamics of the spawning stock. The annual schedule of nominal F values was based on DEP-generated estimates (D. Simpson, CT DEP, Old Lyme, CT, pers. comm.), for which the time-series begins in 1984. This schedule of F rates was also used in DNC (2001b), a report evaluating cooling water system alternatives at MPS. These exploitation rates take into account both fishing effort and the effects of length-limits and other regulations implemented by the DEP to reduce fishing mortality in Connecticut waters. Values of F used in the simulations were stepped up from 0.40 in the 1960s and 1970s, prior to MPS operation, and reached a peak of 1.517 in 1990 (Table 41; Fig. 50), reflecting an historical period of high exploitation of winter flounder in the 1980s and 1990s (NEFSC 2002). Subsequently, estimates of F decreased during the late 1990s. Based on discussions held with DEP (D. Simpson, CT DEP, Old Lyme, CT, pers. comm.) prior to preparing DNC (2001b), the most recent estimate of fishing mortality (0.74) was based on the mean of the last available annual point estimate of F provided by DEP (1999-2000) and the last available three-point moving average of F determined for age-4+ winter flounder in LIS. This value was assigned to 1999 and subsequent years in simulating winter flounder stock size projections into the future. As a result of protective regulations, age-classes 1 through 3 are not fully vulnerable to fishing mortality, particularly in more recent years, so correction factors (Table 42) were used to reduce the annual values of F for these ages that are found on Table 41.

The potential impact of entrainment depends upon the age of each larva at the time it is entrained, because an older individual has a greater probability of surviving to reproductive age than a younger one. The number of individuals entrained from each developmental stage was estimated from the proportion of each stage collected at the MPS discharge. Since the proportion of entrainment attributed to the Niantic River was estimated from mass-balance calculations, the number of larvae for each stage could be allocated to either the Niantic River or other sources (Table 37). Most Stage 3 larvae entrained (the predominant stage collected at the MPS discharges) apparently originated from sources other than the Niantic River. Conditional mortality rates (i.e., production loss estimates) for larval entrainment (ENT) from 1984 through 2001 used in SPDM simulations (Table 41) were estimated directly using the mass-balance calculations under actual MPS operating conditions during these years (Table 39). The conservative nature of the mass-balance calculations was previously discussed as were the larval stock identification studies (Crivello 2002; Moran 2002), which independently verified that fractions of entrained larvae attributed to the Niantic River winter flounder stock in 2001 and 2002 (Crivello in preparation) were similar among the two studies and the mass-balance calculation. Larval losses for early years of plant operation (1971-83) were simulated in NUSCO (2000) by modifying a

TABLE 40. Data, rates, and other inputs used with the Niantic River winter flounder population dynamics simulation model (SPDM).

Model input	Value used or available		
Number of age-classes in population	15		
Earliest age at which all females are mature	5		
Fraction mature, mean wt (lbs), and mean fecundity by age[a]:			
Age-1 females	0	0.011	0
Age-2 females	0	0.125	0
Age-3 females	0.25	0.497	232,088
Age-4 females	0.80	0.776	432,517
Age-5 females	1.00	1.096	700,390
Age-6 females	1.00	1.435	1,019,793
Age-7 females	1.00	1.628	1,216,926
Age-8 females	1.00	1.839	1,442,512
Age-9 females	1.00	2.068	1,699,372
Age-10 females	1.00	2.316	1,990,489
Age-11 females	1.00	2.584	2,319,011
Age-12 females	1.00	2.872	2,688,253
Age-13 females	1.00	3.182	3,101,703
Age-14 females	1.00	3.514	3,563,020
Age-15 females	1.00	3.869	4,076,040
Age after which total annual mortality is constant	4		
Instantaneous mortality rates M and F at age-1	0.8425	$F \times 0.02$[b]	
Instantaneous mortality rates M and F at age-2	0.20	$F \times 0.25$	
Instantaneous mortality rates M and F at age-3	0.20	$F \times 0.60$	
Instantaneous mortality rates M and F at age-4+	0.20	$F \times 1$	
Initial number of female spawners	73,486[c]		
Initial biomass (lbs) of female spawners	121,252[c]		
Mean fecundity of the stock (eggs per female spawner)	1,322,994[d]		
α_n for the unfished ($F = 0$) stock	5.20[e]		
β from the three-parameter SRR (mean value from 1989-2002)[f]	2.244×10^{-5}		
ϕ from the three-parameter SRR (mean value from 1989-2002)[f]	-0.374		
Mean February (1977-2002) water temperature (°C)	3.07		
standard deviation	1.19		
minimum temperature	0.36		
maximum temperature	5.16		

[a] See Table 31 for the derivation of weight and fecundity at age.
[b] Values of M remain constant during all spawning cycles or years simulated. Fish at ages-1-3 are partially recruited and the multipliers shown (≥1987; see Table 42 for previous years) are used to reduce F accordingly (see Table 41 for annual values of F used in the SPDM).
[c] Corresponds to the unfished stock at equilibrium, $P_{n,p}$ (see Eq. 12) and a mean weight of 1.65 lbs.
[d] Calculated for the Niantic River winter flounder female spawning stock at equilibrium in the absence of fishing (see Table 31).
[e] Indirectly calculated from life history parameters (see Stock-recruitment Relationship and Table 30).
[f] See Table 29.

randomly chosen value of ENT from 1984 through 1995 proportional to condenser cooling-water flows at MPS actually used during the annual April 1-June 15 larval winter flounder season in those years and these values are given in Table 41. For years going forward in the simulations, a geometric mean ENT of 14.20% was calculated for the time-series of production loss estimates using the annual values for 1986-95 (years of MPS three-unit operation), 1999, 2001, and 2002. Values for the extended shutdown years of 1996-98 were excluded, as was the estimate

for 2000, which although given in Table 39, was computed using a different methodology due to a lack of a larval mortality estimate that year. Because of this, the geometric mean value was also used for 2000 in the simulations. This mean value was further reduced (based on unit cooling water demand) to 8.84% in 2035 following the projected retirement of Unit 2 and to 0 in 2045 after the projected retirement of Unit 3. Because there is still some uncertainty regarding annual estimates of ENT, separate model

TABLE 41. Schedule of conditional entrainment (ENT values) and fishing mortality rates (F) adjusted for impingement (IMP), including values of F for less than fully vulnerable age-classes, as implemented in the current SPDM simulations.

Time step	Simulation years	% of year-class reduction based on calculated or simulated levels of ENT that were used[a]	Nominal F (plus IMP of 0.01)	F at age-1[b]	F at age-2[b]	F at age-3[b]
0-10	1960-70	0	0.40	0.016	0.300	0.400
11	1971	3.4	0.51	0.020	0.375	0.500
12	1972	4.9	0.51	0.020	0.375	0.500
13	1973	0.3	0.51	0.020	0.375	0.500
14	1974	3.2	0.51	0.020	0.375	0.500
15	1975	2.5	0.61	0.024	0.450	0.600
16	1976	9.3	0.61	0.024	0.450	0.600
17	1977	7.2	0.61	0.024	0.450	0.600
18	1978	3.6	0.61	0.024	0.450	0.600
19	1979	13.8	0.61	0.024	0.450	0.600
20	1980	6.4	0.61	0.024	0.450	0.600
21	1981	3.1	0.71	0.028	0.525	0.700
22	1982	10.9	0.71	0.014	0.350	0.700
23	1983	9.6	0.71	0.014	0.350	0.700
24	1984	7.9	1.290	0.026	0.640	1.280
25	1985	4.9	1.155	0.023	0.573	1.145
26	1986	7.5	1.124	0.022	0.557	1.114
27	1987	9.1	1.065	0.021	0.264	0.633
28	1988	13.2	1.108	0.022	0.275	0.659
29	1989	14.6	1.173	0.023	0.291	0.698
30	1990	17.3	1.527	0.030	0.379	0.910
31	1991	4.9	1.442	0.029	0.358	0.859
32	1992	22.3	1.409	0.028	0.350	0.839
33	1993	5.0	1.166	0.023	0.289	0.694
34	1994	9.9	1.091	0.022	0.270	0.649
35	1995	38.5	0.934	0.018	0.231	0.554
36	1996	22.2	0.762	0.015	0.188	0.451
37	1997	3.8	0.917	0.018	0.227	0.544
38	1998	17.8	0.825	0.015	0.204	0.489
39	1999	29.0	0.75	0.015	0.185	0.444
40	2000	14.2	0.75	0.015	0.185	0.444
41	2001	35.4	0.75	0.015	0.185	0.444
42	2002	19.7	0.75	0.015	0.185	0.444
43-74	2003-2034	Low ENT = 9.47 Mean ENT = 14.20 High ENT = 21.31	0.75	0.015	0.185	0.444
75-84	2035-2044	Low ENT = 5.89 Mean ENT = 8.84 High ENT = 13.25	0.75	0.015	0.185	0.444
85-100	2045-2060	0	0.74	0.015	0.185	0.444

[a] See text for explanation of values used.
[b] See Table 42 for adjustment factors.

projections were also made using both "high" and "low" mean values of ENT for 2003-2060. These values were derived by scaling the corresponding mean value of ENT by a factor of 1.5 and 1/1.5, respectively (Table 41). These ranges were somewhat greater than values within two standard errors of the mean ENT, providing a very conservative confidence band around simulated output based on mean ENT values.

Model Output and Calibration. All stock projections were 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 that 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. Annual
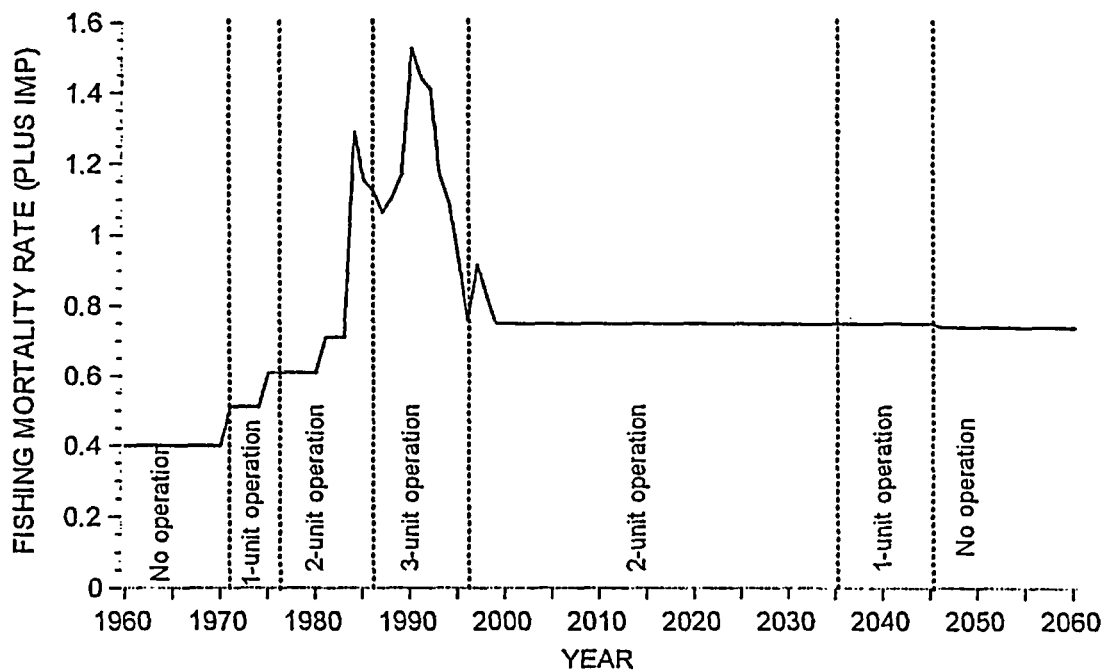
Fig. 50. Historic and projected annual instantaneous mortality rate due to fishing (F), determined in consultation with CT DEP, plus a small (0.01) component accounting for impingement mortality (IMP) at MPS as implemented in the SPDM simulations for this report (see Table 41). The vertical dashed lines illustrate periods of actual or projected one-, two-, and three-unit operation at MPS with an assumed 20-year license renewal for Units 2 and 3, which would extend their operation to 2035 and 2045, respectively.

TABLE 42. Factors used to adjust fishing mortality of less than fully vulnerable age-classes (ages-1-3) as used in the Niantic River winter flounder SPDM.

Period	Age-1	Age-2	Age-3
≤1981	0.04	0.75	1.00
1982-86	0.02	0.50	1.00
≥1987	0.02	0.25	0.60

abundance of females by 0.5-cm size-class was converted to weight (lbs) using a length-weight relationship determined for Niantic River winter flounder:

$$\text{weight} = (1.20173 \times 10^{-5}) \times (\text{length})^{3.226} \qquad (20)$$

and an annual mean weight of 1.08 lbs was computed from 1977 through 2002 (Fig. 51). Thus, during this period both number and weight are nearly synonymous. Although annual mean weight of female spawners from 1995 through 2002 averaged about 1.3 lbs, this value would most likely decrease towards 1 lb if population size increased as mean weight was determined to be almost exactly 1 lb during 1977-85, a period when winter flounder were more abundant than at present.
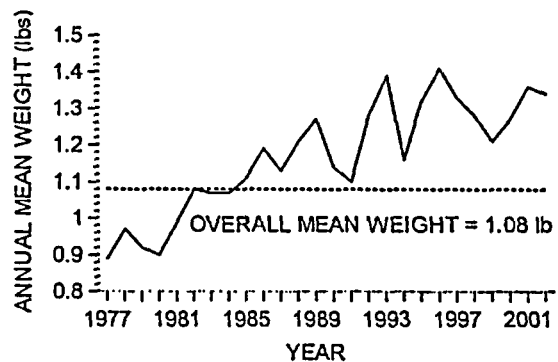
Fig. 51. Annual mean weight (lbs) of Niantic River female winter flounder spawners from 1977 through 2002. The overall grand mean weight (dashed line) was 1.08 lbs.

A complete simulation of MPS impact consisted of three stochastic time-series of female spawning stock sizes generated during separate and independent SPDM simulation runs completed under a specified set of population parameters and plant operation conditions, including random variability. There were three basic scenarios: 1) a theoretical unfished stock, whose size was dependent only upon natural mortality and the dynamics of winter flounder reproduction and

environmental variability (i.e., with no fishing or plant operational effects); 2) an exploited stock with reduced biomass as a result of fishing along with natural variation, but without any power plant effects (i.e., the baseline time-series without MPS effects); and 3) a stock with a biomass reduced by two types of anthropogenic mortality (fishing: F, and power plant effects: IMP and ENT) occurring along with natural variation (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 any MPS operation. The third time-series provided a basis for quantitatively assessing MPS impact on the Niantic River winter flounder population. As noted, three subsets of the impacted stock were determined using the mean, low, and high values of ENT for the projections.

Prior to carrying out SPDM simulations presented in DNC (2001b), several calibration runs were conducted to insure that reasonable agreement existed between the population projections and recent abundance estimates of the Niantic River winter flounder female spawning stock. Population parameter estimates used by the model were reviewed and updated such that model output would match more closely the observed abundance time-series. The instantaneous natural annual mortality rate (M) of 0.50 for age-1 flounder used in previous applications of the SPDM (e.g., NUSCO 2000) was perhaps too low since it amounts to a monthly attrition rate of only 4.1% (annual survival $S = e^{-M}$). Field sampling showed consistently that numbers of age-0 juvenile winter flounder decline rapidly during the 4.5 months following metamorphosis and settlement (see Age-0 Juveniles during Summer), with a 18-year (1985-2002) mean survival rate of only 9.7% through this period, or an average monthly attrition rate of about 41%. Specific information is lacking to calculate a mortality rate during late fall and winter and when this group of fish becomes age-1 in the following spring. Although M most likely decreases as these fish grow older and larger, mortality of these older immature fish nevertheless remains higher than that of adult fish. Data from a study of Mystic River, CT winter flounder (Pearcy 1962) that were discussed by Klein-MacPhee (1978) in a synopsis of biological data for winter flounder suggested a total survival rate of 0.41 for winter flounder between the ages 12.4 and 22.4 months. Since this survival rate is equivalent to an annual natural mortality rate of M = 1.07, or twice the rate

formerly used in SPDM applications, model output was re-calibrated by progressively increasing the initial value of M = 0.50 for age-1 winter flounder. After a few trial runs of the SPDM using the schedule of fishing rates (Table 41) and the same annual production losses (ENT) given in NUSCO (2000), model output converged to current stock levels. At the point in the calibration process at which model output matched field data-based adult stock sizes during the last 4 years, the simulated M for age-1 flounder was 0.8425. Since this value appeared reasonable, although still less than the estimate of Pearcy (1962), it was adopted as a better estimate for model runs presented in DNC (2001b) and subsequent applications. Rates of M for older winter flounder remained unchanged from previous SPDM applications (Table 40).

Simulation Results. Plots of the stochastic variability for both baseline and impacted (mean ENT rate) stocks show the 95% CI relatively close to the mean, but the minimum and maximum replicates, which were a measure of the random variation in the simulations, had a relatively wide range during the initial (approximately 24-42 thousand lbs) years of the scenarios (Fig. 52). The smallest differences found between the minimum and maximum stock sizes were about 7.4 thousand lbs for the baseline and 5.9 thousand lbs for the impacted stock, each occurring in 1994, 1 year prior to the lowest biomass of each simulation.

Based on the age and size structure of an unfished female winter flounder stock at equilibrium (Table 31), the unfished stock size used initially in all simulations was 121,252 lbs (value of P_{rep}), which was equivalent to 73,486 female spawners (Table 40; see also Stock-Recruitment Relationship, above). This initial stock size represented the maximum spawning potential (MSP) for the unfished Niantic River female spawning stock, a biological reference point discussed in greater detail below. The geometric mean estimate of MSP from the SPDM simulations was 121,348 lbs, which was remarkably similar to the deterministic estimate of P_{rep} used to initiate the model runs. A critical stock size (25% of the MSP) was determined as 30,337 lbs for these simulations, which is shown as the horizontal line in Figure 52. Allowing for natural variation in the simulation, all replicates, even the largest values, were less than 25% of MSP for the baseline stock beginning in 1985 for both baseline and impacted stocks. Nearly all replicates remained smaller than this critical value for the duration of the simulated time-series, with the exception of 2048 in the baseline stock.
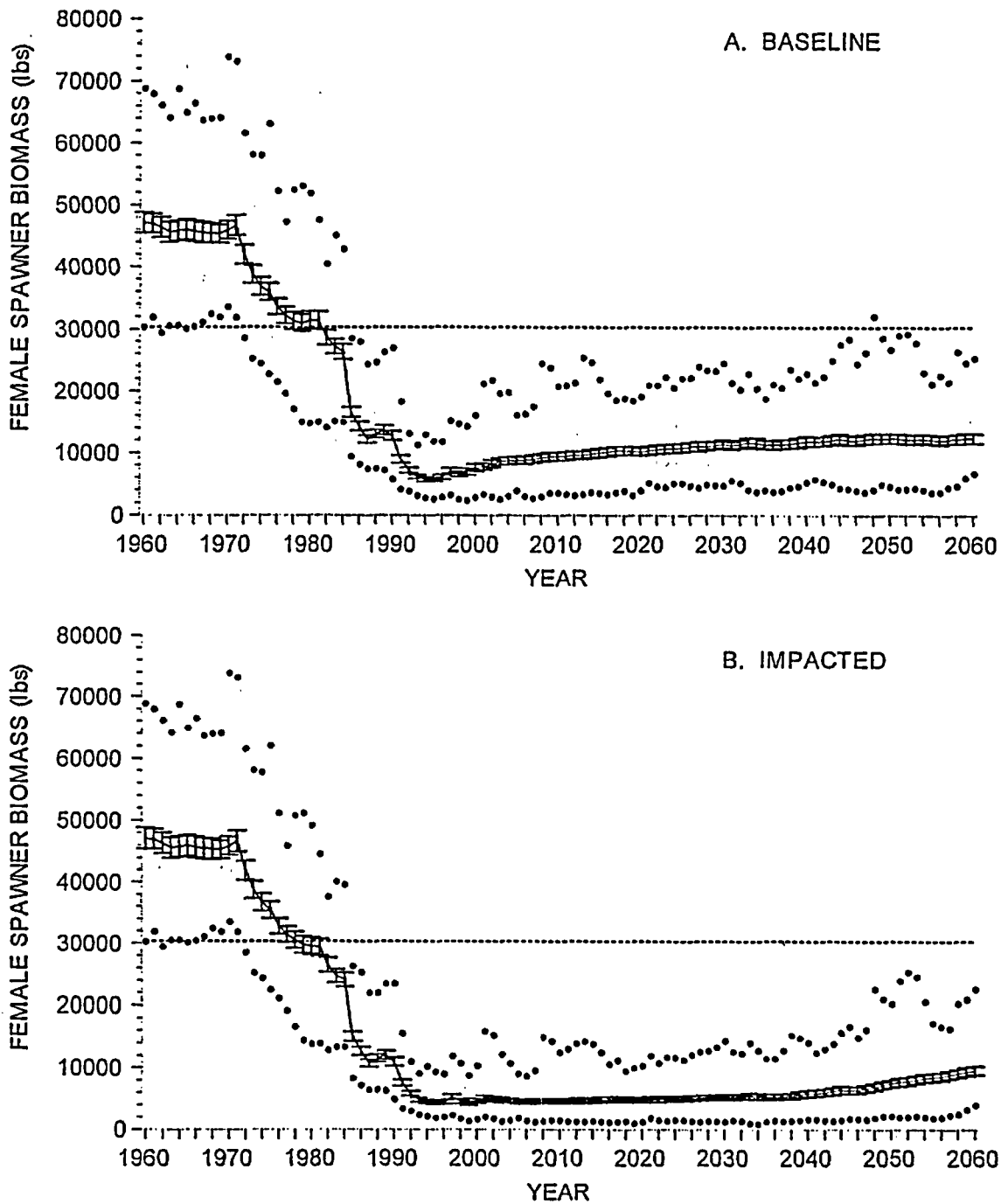
Fig. 52. Stochastic variability associated with projected Niantic River female winter flounder stocks expressed as biomass in lbs for: A. The baseline stock with simulated fishing rates (F) shown on Table 41 and Figure 49 with F in future years set at 0.74, but with no effects from MPS operation, and B. The impacted stock with both fishing effects (same as A) and MPS impact (ENT + IMP). 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 (mean ENT rate going forward) stocks illustrated in Figure 53. 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, 30,337 lbs), defined in Howell et al. (1992) as stock biomass equal to 25% of the maximum spawning potential, which was determined as the geometric mean of the unfished stock shown on Figure 55.

For the simulated baseline projection (shown as the solid line in Fig. 53), the stochastic mean size of the exploited stock under the starting nominal fishing rate of F = 0.40 was quickly reduced to less than 30 thousand lbs in 1982. The baseline responded as expected to the high rates of fishing through the mid-1990s and the stock steadily declined to its lowest point of 5,679 lbs in 1994, only about 19% of the critical stock size. With some reduction in F beginning in 1999 and holding this value at 0.74 from 1999 until the end of the simulation time-series allowed the stock to recover to some extent, but mean stock sizes mostly remained between 11 and 12 thousand lbs through 2060.

To determine the effect of MPS on the Niantic River female spawning stock, the baseline time-series is compared to the impacted (in this case, the mean ENT rate) time-series, shown as the dashed line in Figure 53. The impacted series corresponds to projections of the baseline stock, but with additional annual losses due to MPS operation (i.e., ENT + IMP), the effects of which first begin to be seen in

1974. Similar to the baseline, the lowest projected stock biomass levels (about 4.4 thousand lbs) were attained in the mid-1990s. Absolute differences between the baseline and impacted stocks were initially small (<1.2 thousand lbs) through 1979, but, in general, steadily increased to about 2.8 thousand lbs in 2000. However, the effects of the shutdown of Unit 1 and decreased fishing rates that were simulated beginning in the late 1990s began to propagate through the spawning population projections thereafter and stock sizes increased accordingly. The increase was greater in the baseline than the impacted stock, with differences again steadily increasing from about 4 thousand lbs in 2004 to about 5.4 thousand lbs in 2017. Using a fixed level of ENT, the impacted stock projection shown here exhibited less variation than in previous applications of the SPDM (e.g., NUSCO 2000), where randomly selected entrainment production loss estimates (ENT) and unit cooling-water flows resulted in fluctuating biomass decreases and increases.
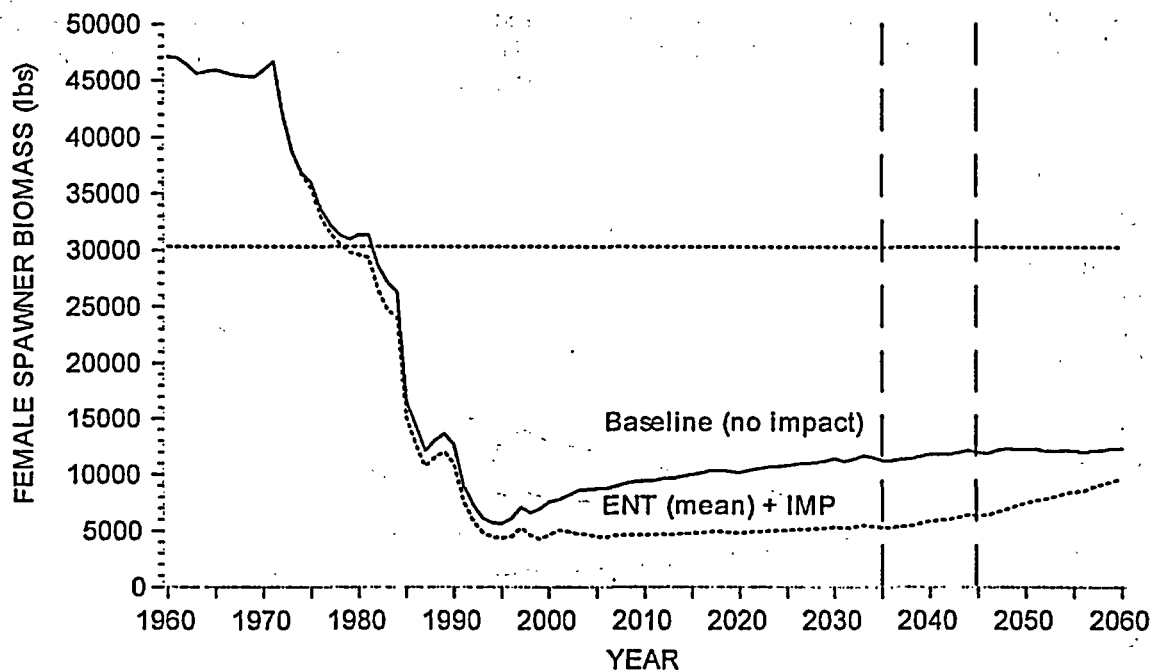
Fig. 53. Results of the SPDM simulation showing the combined effects of larval entrainment rate and fishing (with impingement) given as the dashed line labeled "ENT (mean) + IMP" on the biomass in lbs of Niantic River female winter flounder spawning stock. Entrainment (ENT) and fishing (F) rates changed annually during years of MPS units in operation (see text and Table 41 for details). A geometric mean ENT rate and an F of 0.74 were used going forward. The solid line labeled "Baseline (no impact)" represents the baseline with variable fishing effects only and no plant impact. All stock sizes are averages of 100 Monte Carlo replicates. The horizontal dashed line represents the critical stock size (here, 30,337 lbs), defined in Howell et al. (1992) as stock biomass equal to 25% of the maximum spawning potential, which was determined as the geometric mean of the unfished stock shown on Figure 55. The vertical dashed lines indicate proposed retirement dates of 2035 for Unit 2 and 2045 for Unit 3.

As the simulated retirement of Units 2 and 3 occurred in 2035 and 2045, respectively, impacted stock sizes began to approach those of the baseline. Differences decreased to about 5.8 thousand lbs 7 years after the retirement of Unit 2 and to around 4.3 thousand lbs 7 years after the retirement of Unit 3. The impacted stock biomass remained less than the baseline, with a difference of about 2.8 thousand lbs in 2060 at the conclusion of the simulation timeline. However, at the rate of F used in the simulation, the impacted line would likely become equivalent to the baseline within another decade.

To examine the extent that uncertainties in production loss estimates for Niantic River winter flounder might have in stock projections of female spawner biomass, as noted above, three levels of ENT were used in going forward: a mean rate, a high rate (mean × 1.5), and a low rate (mean ÷ 1.5); F remained fixed at 0.74. These three ENT rates resulted in stock trajectories that were parallel to some extent, but differed in magnitude (Fig. 54). Biomass levels in these three projections first began to diverge beginning in 2006. From this point on, biomass under a continuously applied high ENT rate generally continued to decrease, reaching a minimum level of

3.4 thousand lbs in 2036, just after the projected shutdown of Unit 2. Following the projected retirement of Unit 3 in 2045, a relatively steep increase in biomass followed, attaining about 7.6 thousand lbs at the end of the simulation time-series in 2060. The biomass projections under the mean ENT rate showed a somewhat steady increase from about 4.5 to 6 thousand lbs. The biomass increase accelerated after the retirement dates for Units 2 and 3 and terminated at nearly 10 thousand lbs in 2060. Under the low ENT rate, biomass immediately began to increase, although some negative perturbations were seen as a result of model stochasticity. Female spawner biomass exceeded 9 thousand lbs by 2050 and eventually reached 10.7 thousand lbs in 2060. Proportional differences between the mean and low ENT projections were greatest in 2036-39, just after the projected retirement of Unit 2, when biomass under the mean ENT rate was 78% of that under the low rate and biomass under the high rate was about two-thirds of the mean rate and half that under the high rate. At the conclusion of the simulated time-series, biomass under the mean rate was within 10% of the low rate and biomass under the high ENT rate was within about 21 and 29% of the mean and low rates, respectively.
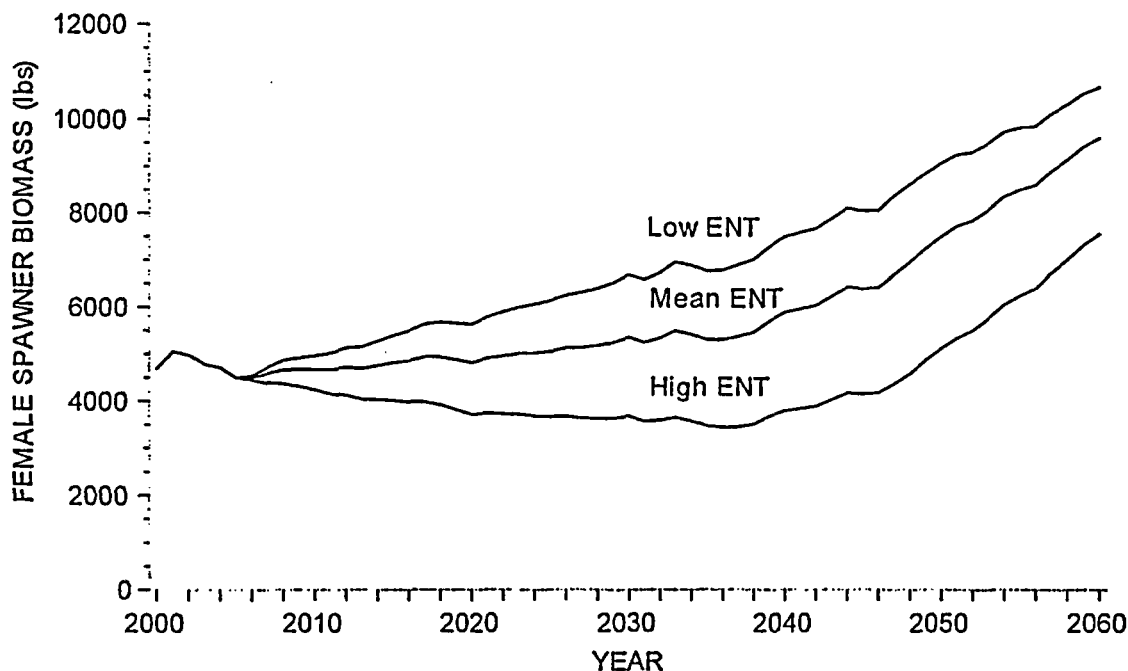
Fig. 54. Results of SPDM simulations for the years 2000-2060 showing the combined effects of larval entrainment rate and fishing (with impingement) on the biomass in lbs of Niantic River female winter flounder spawning stock. A fishing mortality rate of 0.74 was used going forward. Three rates of ENT are compared going forward: the mean, a high rate (mean × 1.5), and a low rate (mean ÷ 1.5). All stock sizes are averages of 100 Monte Carlo replicates.

The factors used in creating the high and low ENT rates modeled encompassed the variation seen in the historical production loss rates (Table 41). Another consideration is that production loss in recent years may have been overestimated if egg survival and hatching rates were greater than in the past (Fig. 48). Unless physical conditions (e.g., current patterns, Niantic River discharge) permanently change, it is likely that annual variation will continue to be seen in production loss estimates due to natural physical and biological factors. Thus, effects to Niantic River female winter flounder biomass would most likely be more variable than seen in each of these three population projections. However, the high and low rates should be indicative of the minimum and maximum biomass levels, given that other conditions (F in particular) remain as modeled.

The rate of fishing on winter flounder stock size is of great importance and population projections are only realistic for the particular fishing rates simulated. Actual spawner abundance could depart considerably from predictions if fishing rates or other simulated conditions are not matched by actual conditions. For example, if fishing rates were to become high (e.g., >0.8) again at any point in the future, differences between the baseline and impacted stock series would become wider and recovery would take longer, assuming that fishing would eventually decrease once again. The different nature of stock reductions caused directly by fishing and impingement, which affect mostly adult fish, and those resulting from larval losses through entrainment at MPS 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 larger fish and, thus, reduces the average weight of the spawners remaining in the stock. However, the most important difference between fishing 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 winter flounder life history. The relative effects of stock reductions due to fishing and MPS impact can be assessed by comparing the unfished stock projection line to those for the fished stock with and without plant effects (Fig. 55). Most biomass reductions can clearly be attributed to fishing. However, as fishing mortality was reduced and stock biomass increased, winter flounder population size of the impacted stock was increasingly smaller in comparison to the baseline until MPS units ceased operation, when the two stocks began to converge.

Probabilistic Assessment of MPS Effects. The stochastic variability associated with stock projections of the baseline and impacted stocks as shown in Figure 52 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 cumulative distribution frequency (cdf) of 100 replicate predictions for each year, including stock sizes both smaller and larger than the mean. Stock sizes projected for each simulation scenario at ten decadal years from 1970 through 2060 are given in Table 43. The theoretical unfished stock in each of the years varied little, fluctuating between about 127 and 129 thousand lbs. Prior to MPS operation in 1970, each of the baseline and the impacted stocks were identical (geometric mean of 45,938 lbs) and represented about 36% of the unfished stock. By 1990, winter flounder spawning stocks under full MPS three-unit operation declined to about one-quarter of the 1970 size, which was mostly the result of increased fishing, as the impacted stock was only about 1.7 thousand lbs less than the baseline. As noted previously, smallest stock sizes were predicted for the mid-to late 1990s as a result of high rates of exploitation during the early 1990s. In 2000, the baseline and impacted stocks were only about 6% and 4% of the unfished stock, respectively. In following years, the baseline stock responded more rapidly to decreased fishing than the impacted stock, with the latter only finally approximating the former in 2060, 15 years after MPS ceased operation.

To assess effects of MPS operation, the probability that the Niantic River female winter flounder spawning stock would fall below three selected threshold sizes was determined directly from the cdf of selected annual stock sizes for each of the nine selected years. The threshold 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 (30,337 lbs for the current SPDM simulations) is considered overfished and its continued maintenance is questionable; spawner abundance may decline to even lower levels. Alternatively, conservative fishing rates that preserve 40% or more 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 upon various combinations of length (10, 11, or 12 inches)
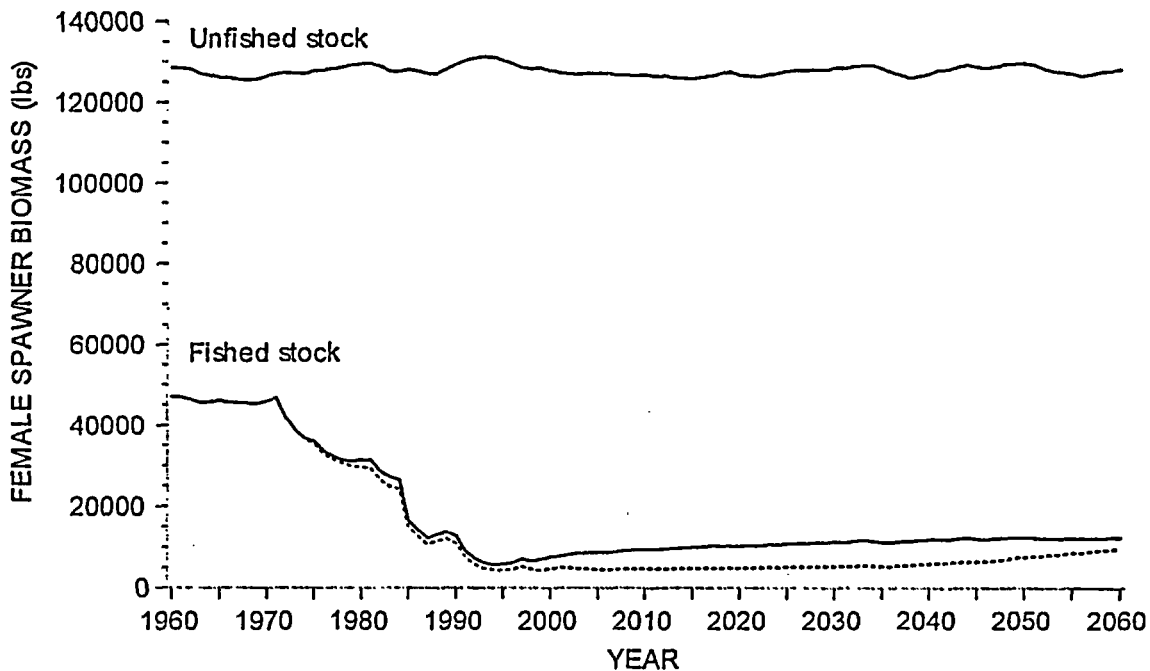
Fig. 55. Comparison of Niantic River winter flounder female stock biomass in lbs after the effects of fishing (the baseline stock, shown as a the lower solid line) and MPS operation under calculated entrainment and impingement rates (shown as the lower dashed line) as shown on Table 41 and in Figures 51-53 with the theoretical (SRR-based estimate) unfished stock (shown as the upper solid line). The baseline stock shown here was determined using simulated fishing rates (F) found on Table 41, with F going forward set at 0.74. All stock sizes are averages of 100 Monte Carlo replicates.

and trawl codend mesh (3.5, 4.5, 5.0, or 5.5 inches) restrictions imposed on the commercial fishery. The difference between the probabilities found for the baseline and impacted stocks was an estimate of risk for MPS effects.

Using the above criteria for years of actual or projected MPS operation (i.e., during the decadal years shown in Table 43), only in 1980 did biomass of both baseline and impacted (mean ENT) stocks have any probability of exceeding some of the threshold sizes. In 1980, the baseline stock was in all likelihood (p = 0.98) smaller than 40% of MSP (48,392 lbs), had high probability (p = 0.86) of being less than 30% of MSP (36,404 lbs), but had about an equal chance (p = 0.45) of being less than 25% of MSP. In 1980, the impacted stock was almost certainly (p = 0.99) less than 40% of MSP and was most likely (p = 0.86) less than 30% of MSP, and a somewhat greater than even (p = 0.54) chance of being less than 25% of MSP. In all other years shown in Table 43, the probability that the biomass of both the baseline and impacted stocks were less than all three levels of MSP was certain (p = 1.00), including in 2060, 15 years after the retirement of Unit 3 and at the end of the simulation time-series.

The effect of a constant fishing mortality rate of 0.74 in all future years was to keep the Niantic River winter flounder stock moderately depressed to levels as shown on Figures 52 through 55.

In summary, SPDM output shows Niantic River female winter flounder spawner biomass stabilizing and slowly increasing following the shutdown of MPS Units 2 and 3 in 2035 and 2045, but at levels constrained by the prevailing fishing mortality of 0.74 used in the projections. For a winter flounder stock to reach a desirable size, which according to Howell et al. (1992) is greater than 40% of MSP, it appears that fishing mortality would have to be reduced to much lower levels. However, given the uncertainty of the fishing rate currently operating on the Niantic River winter flounder stock and the almost impossible task of keeping it stable over many years, projected biomass gains become less reliable with elapsed time.

As shown in other SPDM simulations of the Niantic River winter flounder population, even modest reductions in F results in immediate and greater increases in stock biomass than from much larger fractional reductions in larval entrainment (DNC 2001b).

TABLE 43. Expected biomass in pounds of female winter flounder spawners at ten selected points in time for the theoretical unfished[a], baseline[b], and impacted[c] (baseline plus MPS impact for the low, mean, and high time-series of ENT values; see Table 42 and Figures 52-54) stocks of the Niantic River population. The baseline and impacted stock projections were determined using one common historical set of fishing mortality rates (1960-2002) and a projected (2003-2060) future fishing rate of 0.74 during SPDM simulations. Expected mean stock sizes are geometric means of 100 Monte Carlo replicates and 95% confidence intervals (CI) are given.

Type of population simulated	1970	1980	1990	2000	2010	2020	2030	2040	2050	2060
Theoretical unfished stock:										
Geometric mean	126,510	129,470	129,244	127,714	126,783	126,689	128,432	127,035	129,689	128,138
95% lower CI	124,163	126,511	126,411	124,979	124,160	124,015	126,009	124,062	126,816	125,272
95% upper CI	128,900	132,499	132,140	130,509	129,461	129,421	130,900	130,080	132,627	131,069
Baseline stock:										
Geometric mean	45,938	31,407	12,710	7,576	9,461	10,183	11,446	11,860	12,315	12,374
95% lower CI	44,529	30,001	11,973	7,053	8,850	9,569	10,824	11,111	11,601	11,641
95% upper CI	47,392	32,880	13,491	8,138	10,115	10,836	12,103	12,660	13,073	13,153
Impacted stock - low ENT										
Geometric mean	45,938	29,601	11,007	4,691	4,967	5,640	6,682	7,497	9,078	10,694
95% lower CI	44,529	28,278	10,360	4,355	4,597	5,219	6,221	6,927	8,457	10,003
95% upper CI	47,392	30,985	11,693	5,053	5,367	6,094	7,175	8,114	9,743	11,432
Impacted stock - mean ENT										
Geometric mean	45,938	29,601	11,007	4,691	4,676	4,820	5,361	5,891	7,504	9,621
95% lower CI	44,529	28,278	10,360	4,355	4,327	4,454	4,974	5,414	6,947	8,956
95% upper CI	47,392	30,985	11,693	5,053	5,052	5,217	5,778	6,411	8,105	10,335
Impacted stock - high ENT										
Geometric mean	45,938	29,601	11,007	4,691	4,246	3,726	3,675	3,802	5,136	7,567
95% lower CI	44,529	28,278	10,360	4,355	3,930	3,435	3,392	3,464	4,702	6,969
95% upper CI	47,392	30,985	11,693	5,053	4,588	4,042	3,982	4,172	5,609	8,216

[a] No fishing (F = 0) or MPS effects.

[b] Fishing effects (F = 0.74 going forward), but no MPS impact of entrainment or impingement.

[c] Combined effects of entrainment and impingement (low, mean, or high ENT + IMP) at MPS in addition to fishing rates used for the baseline stock.

Biomass Calculations. To provide some perspective on the Niantic River winter flounder stock size estimates found above, from both actual sampling and in the SPDM projections, recreational and commercial landings of winter flounder in Connecticut waters were obtained (NMFS 2003) for 1981 through 2001 (Table 44). Using the F rates used in the SPDM simulations (Table 41), annual exploitation rates (u) were calculated as follows:

$$u = (F / [F + M]) \times (1 - exp[-F - M] \quad (21)$$

where M is the instantaneous natural mortality rate for adult winter flounder (0.2). The annual exploitation rates were then used to estimate the annual winter flounder stock sizes (N) of winter flounder in Connecticut waters, which was presumed to be the aggregate of all stocks in LIS:

$$N = C / u \quad (22)$$

where C is total annual Connecticut landings of winter flounder in lbs. Because of variable F rates, LIS calculated biomass estimates also varied. Most stock sizes from 1981 through 1990 exceeded 2 million lbs and were as high as 4.4 million lbs in 1982 (Table 44). However, biomass decreased to less than 2 million lbs in 1991 and to less than 1 million lbs in 1999 and 2000 before increasing to about 1 million lbs in 2001. Data from annual winter flounder abundance surveys were used to determine the size of the exploitable biomass of Niantic River winter flounder for the same years. The annual standardized catches of both male and female winter flounder present during the spawning season in the Niantic River were determined using the minimum legal size for retention (see Table 2 of NUSCO 2000), which was lower in some years for the recreational fishery than the commercial fishery. Since 1981, minimum size has increased from 20.3 to 30.5 cm. Weights were determined using Equation 20 with length and abundance information and

TABLE 44. Annual Connecticut recreational and commercial harvest (lbs) of winter flounder, total landings (C), instantaneous fishing mortality rate (F), calculated exploitation rate (u), calculated stock size in lbs of Long Island Sound (LIS) winter flounder, estimated exploitable biomass (lbs) of the Niantic River winter flounder spawning stock, and the fraction that the Niantic River stock made up of the LIS winter flounder resource from 1981 through 2001.

Year	Recreational landings (lbs)[a]	Commercial landings (lbs)[a]	Total landings in lbs (C)	F[b]	Exploitation rate (u)[c]	LIS stock biomass in lbs (N)[d]	Niantic River exploitable[e] stock biomass (lbs)	% Niantic River of LIS stock biomass
1981	626,781	1,153,200	1,779,981	0.700	0.462	3,856,471	124,351	3.22
1982	905,542	1,134,500	2,040,042	0.700	0.462	4,419,914	130,028	2.94
1983	306,170	1,171,500	1,477,670	0.700	0.462	3,201,490	92,504	2.89
1984	1,184,009	1,308,900	2,492,909	1.280	0.668	3,731,961	51,381	1.38
1985	946,150	1,193,900	2,140,050	1.145	0.630	3,399,585	45,378	1.33
1986	597,398	569,400	1,166,798	1.114	0.620	1,882,070	39,400	2.09
1987	1,001,087	1,424,300	2,425,387	1.055	0.601	4,035,639	52,392	1.30
1988	891,997	749,900	1,641,897	1.098	0.615	2,670,116	69,813	2.61
1989	721,890	553,300	1,275,190	1.163	0.635	2,008,422	54,258	2.70
1990	434,690	1,063,090	1,497,780	1.517	0.725	2,066,376	31,076	1.50
1991	360,717	844,700	1,205,417	1.432	0.706	1,707,690	54,852	3.21
1992	151,410	704,300	855,710	1.399	0.698	1,225,767	30,965	2.53
1993	84,176	552,113	636,289	1.156	0.633	1,004,474	16,127	1.60
1994	99,463	307,000	406,463	1.081	0.609	669,903	22,104	3.31
1995	257,070	356,133	613,203	0.924	0.555	1,105,046	10,237	0.93
1996	116,961	[f]	-	0.752	0.485	-	4,449	-
1997	237,116	426,474	663,590	0.907	0.549	1,209,822	7,246	0.60
1998	275,467	[f]	-	0.815	0.512	-	4,617	-
1999	69,090	377,403	446,493	0.740	0.480	930,740	4,395	0.47
2000	13,953	445,239	458,416	0.740	0.480	955,594	6,944	0.73
2001	23,256	489,595	512,851	0.740	0.480	1,069,067	3,170	0.30
							Geometric mean	1.54

[a] From NMFS (2003).
[b] As given on Table 41.
[c] $u = (F / [F + M]) \times [1 - exp(-F - M)]$, where $M = 0.2$, the instantaneous natural mortality rate of adult winter flounder.
[d] $N = C / u$.
[e] To be conservative, based on the minimum legal size for retention by the recreational fishery, which in some years had lower size limits than the commercial fishery. Estimates include the calculated biomass of both males and females found in the Niantic River during the annual spawning survey.
[f] No data available from NMFS (2003).

summed over all size-classes. Annual totals represented exploitable biomass of the Niantic River stock (Table 44). Exploitable stock size decreased from more than 120 thousand lbs in 1981-82 to 3-7 thousand lbs in 1996-2001 as abundance decreased and minimum size for exploitation increased. Through 1995, these values represented from 0.9 to 3.2% of the LIS aggregated stock biomass, were only about 0.5-0.7% during 1997-2000, and decreased to an estimated 0.3% in 2001. The geometric mean of the time-series indicated that the Niantic River winter flounder population probably made up less than 2% of the exploitable stock biomass in LIS during the past two decades, an indication of the size of this population relative to the total resource in Connecticut waters.

Conclusions

Abundance of adult winter flounder spawners in the Niantic River peaked in the early 1980s as a result of extraordinarily large year-classes produced during abnormally cold winters occurring during the late 1970s. Thereafter, abundance decreased because of stock and recruitment effects (i.e., decreasing per capita recruitment at high adult stock sizes for several years) coupled with increased exploitation rates as well as a succession of increasingly warmer winters, which appears to have negatively affected winter flounder reproductive success. Abundance was further depressed beginning in 1992, primarily as a result of historically high fishing mortality rates that prevailed from the late 1980s through the late 1990s (NEFSC 2002). Based on a previous tagging study, Niantic River winter flounder range widely

throughout Southern New England during most of the year and are subject to a variety of fisheries. At present, the Niantic River stock most likely makes up less than 2% of the exploitable winter flounder biomass in LIS. Similar declines in abundance of winter flounder stocks found throughout Southern New England were also attributed to high rates of fishing (Desfosse et al. 1998; Brown and Gabriel 1998). Niantic River spawning stock abundance is significantly and positively correlated with several abundance indices from Connecticut, Rhode Island, and Massachusetts. These region-wide trends indicated that similar factors were influencing winter flounder population dynamics across a broad geographical scale.

In 2002, fewer winter flounder larvae were found in Niantic Bay than during the past several years and their abundance was at about the long-term average. Dates of peak abundance for several larval developmental stages in both the river and the bay were among the earliest observed. This was likely a result of record warm winter and spring water temperatures occurring this year. Despite current low abundance of spawning females and resulting low egg production, abundance of newly hatched larvae was greater than expected in Niantic River and Bay during 6 of the last 7 years. Relatively numerous yolk-sac larvae during these years indicated that egg survival within the Niantic River could have been as much as four times higher than in previous years. The cause of this apparent density-dependent effect is unknown, but increased egg survival could have been related to decreased predation on eggs or from possible effects of adult female size. For example, Buckley et al. (1991) demonstrated that a spawning stock dominated by larger individuals had greatest egg viability and hatching success. Low larval mortality found in several recent years also likely reflected a compensatory response as the 4 years with the highest abundance of winter flounder larvae ≥7 mm occurred in 1995, 1997-99, and 2001, years that coincided with some of the lowest egg production estimates.

A relatively low entrainment density at MPS during 2002 also reflected the decreased larval abundance in Niantic Bay. Further, with Unit 2 shut down for refueling during late winter and early spring, the 2002 entrainment estimate was moderate after 2 years of relatively high entrainment. Entrained larvae are a mixture from several sources, including the Niantic River, winter flounder stocks associated with the Thames and Connecticut Rivers, and perhaps others from more distant locations. Results of the mass-balance model and the genetic stock identification stock analysis (Crivello 2002; In preparation) showed good agreement in estimating the proportion of winter

flounder larvae entrained at MPS that originated in the Niantic River during both 2001 (21 and 22%, respectively) and 2002 (14 and 12%). Further, both analyses also showed that the fraction of entrained larvae originating in the Niantic River decreased over the course of the larval period. The high level of agreement between these approaches gives credence to the reliability of the results.

Because larval winter flounder densities and entrainment estimates vary independently from year to year, high larval abundance was not functionally associated with high cooling-water flow. Thus, the fraction of Niantic River larval production entrained changes each year. Reduced recruitment has not been the result of reduced larval abundance, because the latter has exhibited no trend after many years of MPS operation. Because annual entrainment estimates were positively correlated with several juvenile winter flounder abundance indices, no entrainment effect was implied. Further, the greater the number of larvae that were available for entrainment, the more that settled as juveniles in Niantic River or Bay. However, based on previous sampling, those young settling in Niantic Bay probably do not contribute appreciably to year-class abundance, likely due to higher rates of mortality in areas outside of the inshore nursery grounds (e.g., Niantic River) known to produce most of the recruits.

Adult recruitment is largely influenced by natural biological and physical processes occurring in early life during larval and demersal juvenile stages, from the first summer through the next several years of life. Brodziak et al. (2001) noted a long-term declining trend for Southern New England winter flounder biomass and recruitment and that higher recruitment was more likely at higher spawner abundance, even though it was likely that this stock complex was best represented by a density-dependent SRR. As noted above, however, very small adult spawning stocks in recent years nevertheless produced large numbers of late stage larvae, likely as a result of low egg and larval mortality rates. There is no indication of depensatory stock dynamics in the Niantic River population. However, in some years initially high densities of settled age-0 juveniles observed in the Niantic River were greatly reduced by the end of summer through high mortality rates. Also, growth of juveniles during most recent summers has been relatively poor. This has unknown, but likely negative consequences for survival during fall and winter and to subsequent recruitment.

As seen in the studies at MPS and elsewhere, long-term abundance indices indicated that apparent losses during later juvenile life stages appeared to have disproportionately reduced expected good recruit-

ment at age-4 from larger year-classes of young winter flounder. Greater than average predation on pre-recruit sizes of winter flounder may be occurring, which could account for the disappearance of relatively abundant young before they become spawning adults. In recent years, highly abundant populations of green crabs, cormorants, and striped bass may be preying on juvenile winter flounder and increasing numbers of harbor seals may be feeding on larger juveniles and adult spawners. Unfortunately, little quantitative information is available to explain how either fishing or predation has affected juvenile winter flounder mortality. These processes occurred independently of MPS operation as demersal age-0 winter flounder, in particular, are unaffected by plant operation.

Simulations conducted this year using the SPDM continued to clarify the issue of the different scales by which winter flounder spawning biomass responds to changes in mortality due to larval entrainment and to exploitation of adult fish. For commercially exploited and long-lived species like the winter flounder, the long-term effect of larval entrainment on adult fish biomass is relatively minor when compared to the effects of commercial exploitation. This occurs because larval entrainment impacts each year-class not only once, but early in life when natural mortality is high, while commercial fishing impacts the year-class after the fish become vulnerable to the fishery year after year for as long as the year-class persists. Additionally, annual exploitation rates can be as high as 65% or more, such as for winter flounder in the early 1990s, while entrainment rates are generally much less. This was recently demonstrated by O'Connor (2001), who conducted a generic Leslie matrix-based comparative analysis and concluded that "...fishing has a more severe effect on future populations than chronic impacts that decrease only first-year survival or fecundity." However, it is difficult to quantify the difference in how stock biomass changes under these two types of mortality because the response depends on complex factors, such as density-dependent processes early in the life of the fish and the age structure of the spawners, which changes with variable annual recruitment and fishing pressure. Population dynamics models like the SPDM, with explicit representation of the age structure and integration of compensatory effects, are well-suited to simulate and help quantify the stock response to losses from both entrainment and the fisheries.

A substantial recovery of the Niantic River and some other regional winter flounder stocks has not occurred despite regulatory efforts to reduce fishing mortality, which remains higher than desirable

(NEFSC 2002). Even though some year-classes demonstrated high juvenile winter flounder abundance throughout the region, their numbers declined rapidly when they reached a fishable size (Simpson et al. 1996; Desfosse et al. 1998). Fishing, high rates of predation, or other natural mortality may also be removing juvenile winter flounder before they recruit to the adult spawning population. Sinclair and Murawski (1997) noted that persistent overfishing of pre-recruitment fish was a major factor in the decline of a number of Northwest Atlantic groundfish stocks, although environmental conditions probably have also affected recruitment and will influence how rapidly a stock can rebuild (Brodziak et al. 2001).

Compensatory responses during early life history helped mitigate low abundance of spawning adults and maintained the viability of the Niantic River winter flounder population. The retirement of Unit 1 in July 1998 resulted in a 23% reduction in MPS cooling-water demand and entrainment. The installation of a fish return sluiceway at Unit 2 in 2000 further reduced an already low impact of impingement due to high survival of winter flounder in these systems. The substantial and permanent reduction in station operation and the extended shutdowns of Units 2 and 3 during 1997-99, however, did not result in markedly stronger year-classes or enhanced recruitment of adult winter flounder. Continued efforts to reduce fishing mortality could ensure a quicker recovery of the Niantic River winter flounder population. Similarly, variable environment effects (e.g., water temperature: Keller and Klein-MacPhee 2000; hydrodynamic processes: Werner et al. 1997) and interactions with other species, especially during larval and juvenile life history, appear also to be important factors in the recruitment processes of winter flounder.

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Fish Ecology Studies

Fish Ecology Studies

Introduction

Long Island Sound (LIS) supports a diverse assemblage of fish species, including year-round residents, seasonally migratory fishes, and rarely seen transient species. Several of these species support important commercial and sport fisheries worth millions of dollars each year in Connecticut (Sampson 1981; Blake and Smith 1984).

The objective of the fish ecology monitoring program at Millstone Power Station (MPS) is to determine whether operation of the electrical generating units has adversely affected the occurrence, distribution, and abundance of local fishes. Potential MPS impacts include entrainment of early life history stages through the condenser cooling-water system, impingement of juvenile and adult fish on intake screens, and changes in distribution or abundance attributable to the thermal discharge. Numbers of fish eggs and larvae entrained have been reliably estimated each year since 1976. However, effects of entrainment mortality are more difficult to assess as fish population dynamics are influenced by many biological processes, such as compensatory mortality, density-dependent growth, age at maturity, fecundity, population age composition and size structure, and stock size and range.

Estimates of larger fish impinged on traveling screens can be readily made, but in the case of eggs and larvae, the implications of fish removal are difficult to assess as adult populations are concurrently affected by natural and fishing mortality rates. At MPS, however, the impact of fish impingement has largely been mitigated by the installation and operation of fish return sluiceways (NUSCO 1986, 1988c, 1994b; NNECO 1999; DNC 2001).

Changes in the thermal regime of local waters due to MPS operation have been documented (NUSCO 1988b; see the Introduction section to this report). Should water temperatures exceed species tolerance level, individuals may be forced to move from the area, abandoning potentially important spawning or nursery grounds. Chronic exposure to increased temperature regimes may also affect the seasonality of spawning, growth and development of individuals, and possibly even local stock population dynamics.

Several monitoring programs were established to provide information for the assessment of impacts from MPS operation on local fish assemblages. The programs include sampling of ichthyoplankton (fish eggs and larvae) at the power station and in nearby waters, demersal fishes by trawl, and shore-zone species by seine. These monitoring programs have been modified several times as a result of periodic evaluations (NUSCO 1987, 1994a, 1995). Data collected provided a basis for identifying taxa potentially affected by plant operation, as well as information on long-term abundance trends used to measure changes in local populations.

Fishes potentially impacted by MPS were selected either because of their susceptibility to entrainment or because of their distribution in relation to the thermal plume. The selected taxa include the American sand lance (*Ammodytes americanus*), anchovies (*Anchoa* spp.), Atlantic menhaden (*Brevoortia tyrannus*), silversides (*Menidia* spp.), grubby (*Myoxocephalus aenaeus*), cunner (*Tautogolabrus adspersus*), and tautog (*Tautoga onitis*). During the past several years, increased emphasis has been placed on the tautog. This species supports one of the principal sport and commercial fisheries of LIS (Smith et al. 1989; ASMFC 1996) and its numbers have been declining in many areas since 1984, likely from overfishing of this slow-growing and long-lived fish (Simpson et al. 1995).

Impingement impacts at the MPS were substantially abated due to the shutdown of MPS Unit 1 (23% cooling-water reduction) and the installation of aquatic organism returns at Units 2 and 3 (NUSCO 1988c, 1994b; DNC 2001). A return system was included as part of the intake structure during the construction of Unit 3; in early 2000 a sluiceway was backfitted at Unit 2. A 1-year study to evaluate its operation and effectiveness was completed in July 2001. A report of the results submitted to the CTDEP concluded that the system worked as designed and successfully returned impinged marine organisms back to Long Island Sound (DNC 2001).

This report summarizes data collected from the monitoring programs during June 2001 through May 2002. These data are compared to findings presented in DNC (2002). MPS Units 2 and 3 were both operating during most of this report period, although Unit 2 was shut down from February 16 through April 2 for refueling while Unit 3 was shut down from September 7 through October 7. Unit 1 was shut down permanently on November 4, 1995, which resulted in less impact on all life stages of fish because entrainment and impingement rates are directly related to the amount of cooling-water used. The potential effects of MPS are assessed below for each selected fish taxon.

Materials and Methods

Results of year-round sampling are presented using 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 the season of occurrence for a species crossed a calendar year, the annual period was termed a report year (e.g., 2001-02). When a species was primarily collected only within a calendar year, the annual period was presented as a specific year (e.g., 2002). The materials and methods that follow correspond to the most recent sampling periods that have been completed.

Ichthyoplankton Program

Sampling frequency of ichthyoplankton entrained through the MPS cooling-water system varied seasonally during 2001-02. 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 daytime sample per week was collected during October through January. Samples were collected at either the Unit 2 or 3 discharge (station EN, Fig. 1).

To collect samples from the plant discharge, a 1.0 x 3.6-m conical plankton net with 335-μm mesh was deployed using 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. Under normal plant operation
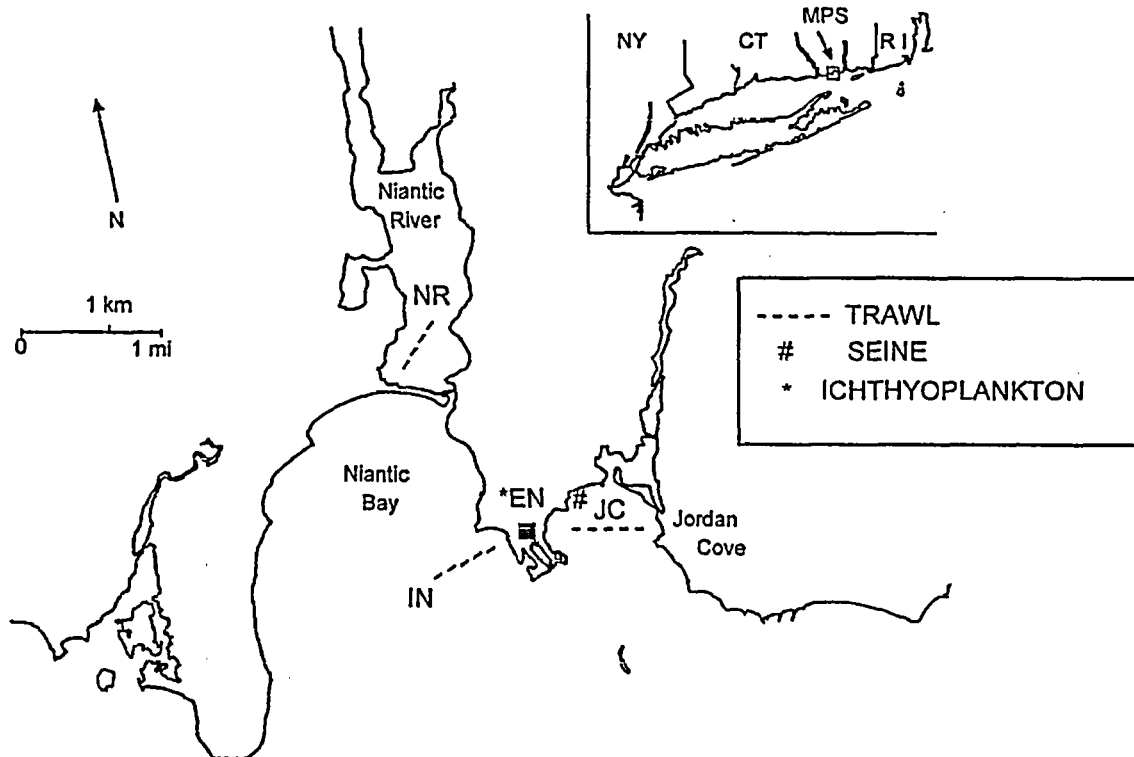
Fig. 1. Location of current trawl, seine and ichthyoplankton sampling stations.

the net was usually deployed for 3 to 4 minutes. All ichthyoplankton collections were preserved in 10% formalin.

Ichthyoplankton samples were split in the laboratory 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, when only two (one day and one night) samples per week were typically examined. Tautog, cunner, and anchovy (bay anchovy, *A. mitchilli* and striped anchovy, *A. hepsetus)* eggs were identified and enumerated in all samples collected from April through August. Tautog and cunner (Family Labridae) have eggs of similar appearance and 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 m³ of water filtered.

Trawl Program

Triplicate bottom tows were made to collect demersal fish using a 9.1-m otter trawl with a 0.6-cm codend liner every other week throughout the year at three stations: Niantic River (NR), Jordan Cove (JC), and Intake (IN) (Fig. 1). A typical standard tow was 0.69 km in length, 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 only occurred at the NR or JC sites. 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) was 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 May 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

A 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. The Δ-mean is the best estimator of the mean for abundance data that approximates 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, data to calculate each Δ-mean were restricted by month to reduce the number of zero values in the distribution tails, which extend beyond the temporal boundaries of the seasonal occurrence. A nonparametric, distribution-free Mann-Kendall test (Hollander and Wolfe 1973) was used to determine the direction of change if an annual Δ-mean time-series represented a significant (p ≤ 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 being particularly well-suited for analysis of environmental monitoring data, because no distributional assumptions are required and small sample sizes are acceptable. Graphical methods are useful aids to formal tests for trend analysis (Velleman and Hoaglin 1981). In particular, methods for smoothing time series of abundance data help visualize short-term patterns often obscured by high variability. Moving averages were used to smooth large abundance fluctuations and plotted against time to provide a visual aid to the statistical trend analyses. The time interval or "window" for the moving average was about 20% of the time series length as suggested by Gilbert (1989). Longer intervals provide a greater smoothing effect, but they may also obscure short-term patterns, which can be of interest in our relatively short time series of abundance data.

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 abundances 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:

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

where C_t = cumulative density at time t

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

A = total or asymptotic cumulative density

p = inflection point scaled in days since first occurrence daye

k = shape parameter.

The origin of the time scale was set to the date when the eggs or larvae generally first appeared in the waters off MPS. Least-squares estimates, standard errors, and asymptotic 95% confidence intervals of the A, 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. 1990). 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 lognormal 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:

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

where A' equals $7 \cdot A$ 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 MPS. Annual entrainment estimates were determined by summing all daily estimates during the period of occurrence.

Results and Discussion

Species Composition and Relative Annual Abundance

A total of 123 fish taxa was recorded as eggs, larvae, juveniles, or adults from collections made in the trawl, seine, and ichthyoplankton programs as part of the Fish Ecology monitoring studies at MPS from June 1976 through May 2002. This total includes fishes collected at present and former sampling stations during this 26-year period, with 112 taxa taken by trawl, 49 by seine, and 58 enumerated in ichthyoplankton samples (Appendix I). Anchovies (mostly bay anchovy) accounted for almost half (41.1%) of the larvae collected at the MPS discharges (station EN) from June 1976 through May 2001 (Table 1), although only 58 were collected this year (Table 2). Overall, winter flounder (*Pseudopleuronectes americanus*) comprised another 13.6% and thirteen other taxa made up most of the remainder (Table 1). Although Atlantic menhaden (*Brevoortia tyrannus*) accounted for almost 16% of all larvae entrained since June 1976, only 80 were collected in entrainment samples during 2001-02. Cunner, tautog, and anchovies accounted for nearly 85% of the eggs collected (Table 1). Silversides (Atlantic silverside, *M. menidia*, and inland silverside, *M. beryllina*)

TABLE 1. Taxonomic composition of ichthyoplankton collected at EN (as a percentage of the total) from June 1976 through May 2002 for larvae and May 1979 through September 2001 for eggs.

Taxon	Larvae	Eggs
Anchoa spp.	41.1	4.6
Brevoortia tyrannus	15.8	
Pseudopleuronectes americanus	13.6	
Ammodytes americanus	6.6	
Myoxocephalus aenaeus	5.3	
Pholis gunnellus	2.5	
Tautoga onitis	2.3	27.3
Tautogolabrus adspersus	2.2	53.0
Enchelyopus cimbrius	1.4	
Ulvaria subbifurcata	1.0	
Liparis spp.	1.0	
Clupea harengus	1.0	
Syngnathus fuscus	0.9	
Scophthalmus aquosus	0.7	
Peprilus triacanthus	0.6	

dominated (80%) the seine catch at station JC (Appendix II); another 10% were killifishes (striped killifish, *Fundulus majalis*, and mummichog, *F. heteroclitus*), with fourspine stickleback (*Apeltes quadracus*) and Atlantic menhaden each making up 4% and 3% of the catch, respectively. The dramatic increase in the numbers of Atlantic menhaden collected in seines over the past few years was not seen during 2001-02 with only 24 fish caught. Eight taxa accounted for over 80% of the total catch at the

TABLE 2. The annual Δ-mean density (no./500 m³) of the most abundant fish eggs and larvae collected at EN for each year from June 1976 through May 2002.

Taxon	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989
EGGS[a]														
T. adspersus	-	-	-	8,348	8,379	7,326	7,874	7,580	6,707	12,842	2,579	5,017	5,388	6,255
T. onitis	-	-	-	1,648	3,741	2,501	3,561	2,372	1,817	4,027	2,833	2,972	2,211	3,373
Anchoa spp.	-	-	-	1,558	999	769	499	2,415	3,631	118	586	64	32	32
LARVAE[b]														
Anchoa spp.	1,152	931	483	2,168	2,430	5,768	816	1,421	302	1,102	1,244	126	359	619
P. americanus	-	106	143	114	285	129	233	297	210	180	87	109	116	203
A. americanus[c]	-	94	318	119	111	136	21	27	18	9	3	13	41	31
B. tyrannus	5	3	3	1	2	1	9	18	2	38	2	2	5	47
M. aenaeus	-	41	38	36	38	107	72	68	50	68	34	29	95	63
P. gunnellus	-	13	13	16	58	58	27	13	14	14	22	4	26	9
T. adspersus	29	58	1	13	58	78	31	49	4	12	4	5	9	14
T. onitis	37	36	1	11	46	83	44	33	3	15	3	7	17	15
E. cimbrius	2	8	6	8	6	1	6	13	5	8	8	12	45	31
Liparis spp.	27	30	10	16	22	5	13	8	36	1	4	42	18	12
U. subbifurcata	-	5	9	14	14	16	17	6	4	60	7	9	23	41
S. fuscus	3	7	4	9	8	13	7	9	9	5	4	6	7	5
S. aquosus	9	11	1	5	5	5	2	13	3	1	4	3	5	3
P. triacanthus	14	3	1	2	11	17	9	9	1	2	3	<1	9	5
C. harengus	-	1	1	1	<1	6	1	0	1	<1	2	1	14	1

Taxon	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
EGGS[a]													
T. adspersus	7,269	6,987	2,776	4,535	8,722	4,266	8,801	3,610	1,459	3,530	8,245	6,495	-
T. onitis	1,942	2,040	1,189	1,394	1,350	1,807	2,323	587	897	1,373	2,322	2,759	-
Anchoa spp.	89	317	62	329	234	118	36	5	114	1	<1	29	-
LARVAE[b]													
Anchoa spp.	1,122	799	178	203	475	181	175	131	106	129	65	58	-
P. americanus	106	99	381	21	142	224	81	255	137	182	328	296	88
A. americanus[c]	24	7	18	28	43	63	18	11	28	13	53	12	8
B. tyrannus	16	81	37	8	44	56	145	23	28	58	293	80	-
M. aenaeus	30	24	58	34	48	43	85	140	55	39	45	188	27
P. gunnellus	6	3	15	8	28	17	41	22	42	12	22	21	5
T. adspersus	68	209	8	10	25	12	12	7	21	15	17	79	-
T. onitis	33	99	13	6	12	8	18	2	14	59	12	113	-
E. cimbrius	37	98	5	18	9	8	2	10	9	4	2	12	-
Liparis spp.	3	23	14	12	5	<1	46	2	2	0	4	0.3	-
U. subbifurcata	51	34	28	2	18	8	<1	18	1	10	1	1	6
S. fuscus	3	5	3	6	4	6	2	2	2	4	3	2	-
S. aquosus	4	12	2	2	3	1	4	2	1	2	3	2	-
P. triacanthus	29	10	2	2	5	3	1	2	1	2	4	7	-
C. harengus	1	2	9	7	4	4	33	11	7	13	0.5	1	0

[a] Fish eggs not identified prior to 1979. Data were seasonally restricted to May-August for *T. adspersus*, May-August for *T. onitis*, and June-August for *Anchoa* spp.

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

[c] Period of occurrence (December-May) extends across calendar year; year given for Δ-means is that of the latter 5 months.

three trawl stations (Appendices III-V). These were the winter flounder (42%), scup (*Stenotomus chrysops*, 17%), silversides (mostly Atlantic silverside; 7%), windowpane (*Scophthalmus aquosus*; 5%), grubby (4%), skates (mostly little skate, *Raja erinacea*; also the winter skate, *R. ocellata*, and clearnose skate, *R. eglanteria*; 3%), anchovies (mostly bay anchovy; 3%), and cunner (3%). The total catch of fish over the 26-year period was similar between IN (124,260) and NR (121,583), with the catch at each of these stations roughly twice the total of 79,944 fish at JC.

Temporal changes in the composition of the dominant taxa (cited above) collected in the trawl and ichthyoplankton programs during the 26-year period were compared using Δ-means. Changes in the composition of seine catches were not examined because silversides have always dominated the catch and these changes will be discussed below. In trawl sampling, winter flounder and scup had the largest annual Δ-mean CPUE during each report year, with silversides, grubby, windowpane, and skates also relatively numerous (Table 2). The 2001-02 Δ-mean CPUE for winter flounder (3.1) was the lowest recorded, slightly less than half that observed in 2000-01. In comparison, the Δ-mean CPUE for scup (143.9) was

the second highest in the 26-year period. The 2000-01 Δ-mean CPUE for silversides (3.7) and grubby (0.8) were slightly lower than average. The Δ-mean of 1.2 for skates was in the range of historic catches while 0.5 for windowpane was the lowest observed in the time-series (Table 3).

Each species and life stage of ichthyoplankton collected at EN occurred during specific time periods. Therefore, Δ-mean densities (no./500 m^3) were computed from data taken during standardized periods of occurrence for each taxon. Because of seasonal occurrence and sample processing, some species have means determined through 2002 and others through 2001 (Table 3). Cunner eggs were always the most abundant of the fish eggs collected followed by tautog. A large decline in anchovy egg abundances occurred after 1984 with the 2001 Δ-mean density (29.0) showing a modest increase over 2000 and 1999 (Table 2).

Abundance of most of the 15 dominant larval taxa was within the range of values found in previous years. Notable exceptions include tautog, whose larvae were the most abundant observed in the time series, and cunner, with the second highest larval Δ-mean in the time series with only 1991 being higher. Larval grubby Δ-mean densities were much

TABLE 3. 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 2002.

Year	Winter flounder	Scup	Silversides	Windowpane	Grubby	Skates
1976-77	23.9	14.8	18.2	1.7	0.6	0.7
1977-78	15.6	13.0	8.5	1.8	0.9	0.6
1978-79	16.7	5.6	10.1	0.9	0.9	0.4
1979-80	26.6	6.2	7.1	1.8	1.9	0.4
1980-81	34.8	9.2	3.3	1.6	1.8	0.8
1981-82	28.9	7.9	2.5	1.5	2.5	0.6
1982-83	49.4	25.1	3.3	2.2	3.3	1.0
1983-84	30.6	25.9	2.8	3.0	2.1	2.6
1984-85	31.3	14.3	2.0	2.4	1.8	0.7
1985-86	23.5	8.3	3.8	2.5	1.2	1.8
1986-87	27.3	24.1	23.1	3.0	2.3	1.8
1987-88	27.3	17.4	4.1	4.3	1.6	2.2
1988-89	41.0	11.4	5.0	3.6	3.5	2.6
1989-90	23.1	11.0	2.4	4.9	1.7	2.4
1990-91	28.4	25.8	2.9	3.3	2.2	3.4
1991-92	26.7	176.0	8.6	2.4	1.4	3.2
1992-93	25.3	56.3	18.4	3.7	2.7	2.2
1993-94	16.9	2.7	2.5	6.0	1.3	3.2
1994-95	22.6	26.4	2.2	3.4	2.3	2.1
1995-96	11.4	5.1	1.6	3.4	0.9	3.5
1996-97	11.6	0.3	0.1	3.0	1.9	1.4
1997-98	5.2	8.1	1.9	1.8	2.0	2.1
1998-99	9.9	15.4	3.7	2.2	2.0	1.8
1999-00	10.1	6.5	3.0	1.8	0.8	2.1
2000-01	7.7	34.3	0.8	2.8	0.5	1.4
2001-02	3.1	143.9	3.7	0.5	0.8	1.2

[a] Data were seasonally restricted to June-October for scup, October-February for silversides, but unrestricted (June-May) for the remaining taxa.

lower than in 2001 at only 27 (Table 2). On occasion, larvae of other species, particularly those of cunner, tautog, rock gunnel (*Pholis gunnellus*), fourbeard rockling (*Enchelyopus cimbrius*), and radiated shanny (*Ulvaria subbifurcata*) 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 MPS. Most fish eggs collected at station EN were from cunner, tautog, and anchovies and the most numerous larvae were anchovies, winter flounder, American sand lance, grubby, and Atlantic menhaden (Tables 4 and 5). The annual numbers of eggs and larvae entrained were related to their abundance at station EN and plant operations (i.e., cooling-water usage). MPS Units 2 and 3 were both operating during most of this report period. The 2001 entrainment estimates of both cunner (4,339 million) and tautog (3,016 million) eggs were within the historic ranges (Table 5). Anchovy egg entrainment was relatively low at about 8 million but higher than observed in the previous 2 years (Table 5).

The 2001 entrainment estimate of 24 million larval anchovies was less than the long-term average of 436 million (Table 5). This was due to the low density of anchovy larvae this year (58; Table 2). The 2002 entrainment estimate of 6 million American sand lance larvae was within the range of previous estimates. Entrainment estimates of grubby larvae in 2002 (33 million) and Atlantic menhaden larvae in 2001 (147 million) were also within the range of previously observed values. Entrainment of winter flounder larvae totaled 119 million in 2002 and this loss is evaluated in the Winter Flounder Studies section of this report.

TABLE 4. Estimated number of anchovy, winter flounder, American sand lance and grubby larvae entrained each year from 1976 through 2002 at MPS and the volume of cooling water on which the entrainment estimates were based.

Year	Anchovies No. entrained (x10^6)	Anchovies Volume (m^3)^b (x 10^6)	Winter Flounder No. entrained (x10^6)	Winter Flounder Volume (m^3)^b (x 10^6)	American sand lance[a] No. entrained (x10^6)	American sand lance[a] Volume (m^3)^b (x 10^6)	Grubby No. entrained (x10^6)	Grubby Volume (m^3)^b (x10^6)	Atlantic menhaden No. entrained (x10^6)	Atlantic menhaden Volume (m^3)^b (x10^6)
1976	378	591	121	629	16	800	12	644	3	890
1977	414	549	29	443	80	888	31	667	2	857
1978	161	631	80	390	176	710	11	477	3	1,035
1979	805	533	44	342	111	810	20	564	<1	895
1980	877	486	168	562	112	974	32	742	2	700
1981	1,448	610	45	373	75	627	42	418	2	999
1982	449	525	164	638	27	927	48	650	14	859
1983	613	463	211	541	30	996	55	745	20	753
1984	167	581	84	508	18	877	39	673	4	989
1985	690	579	80	469	8	900	35	627	44	816
1986	1,093	1,206	123	1,064	4	1,766	54	1,341	5	1,776
1987	117	1,114	165	1,193	35	1,962	52	1,453	2	1,785
1988	383	1,284	184	1,173	86	1,813	112	1,299	7	2,009
1989	530	1,158	167	889	44	1,708	68	1,198	208	1,885
1990	978	1,221	133	1,174	45	2,004	47	1,415	37	1,939
1991	451	754	116	750	7	1,572	31	1,090	56	1,085
1992	151	979	492	1,075	22	1,526	73	1,157	52	1,601
1993	197	1,050	42	1,387	48	2,011	52	1,429	28	1,800
1994	509	1,193	173	920	65	1,819	56	1,203	70	1,923
1995	175	1,198	214	1,006	89	1,783	58	1,221	91	1,860
1996	24	133	51	472	18	1,056	41	797	23	211
1997	17	125	76	173	3	227	28	180	5	213
1998	63	571	84	358	11	474	22	348	35	944
1999	136	905	146	748	14	1,112	49	844	140	1,532
2000	68	1,008	331	1,003	88	1,613	47	1,131	474	1,647
2001	24	992	370	963	14	1,513	178	1,024	147	1,610
2002	-c	-	119	983	6	1,612	33	1,123	-c	-

[a] Includes data from December of the previous calendar year.
[b] Volume was determined from the condenser cooling water flow at MPS during the season of occurrence for each taxa.
[c] Not calculated because larvae occur after the end of the report period (May 2001).

TABLE 5. Estimated number of cunner, tautog, and anchovy eggs entrained each year from 1979 through 2001 at MPS and the volume of cooling water on which the entrainment estimates were based.

Year	Cunner No. entrained (x10⁶)	Cunner Volume (m³)[a] (x 10⁶)	Tautog No. entrained (x10⁶)	Tautog Volume (m³)[a] (x 10⁶)	Anchovies No. entrained (x10⁶)	Anchovies Volume (m³)[a] (x 10⁶)
1979	1,053	684	448	684	324	578
1980	1,660	762	969	762	87	604
1981	1,547	769	1,398	769	287	673
1982	2,078	808	1,253	808	210	680
1983	1,899	753	1,019	753	371	601
1984	2,135	779	1,323	779	883	659
1985	2,814	790	1,720	790	27	675
1986	2,855	1,772	3,750	1,772	522	1,484
1987	4,090	1,687	3,597	1,687	31	1,402
1988	4,294	1,843	2,693	1,843	15	1,558
1989	4,307	1,547	3,002	1,547	5	1,327
1990	3,634	1,724	2,101	1,724	27	1,465
1991	4,117	1,198	1,521	1,198	105	978
1992	2,648	1,484	1,338	1,484	18	1,216
1993	5,421	1,655	2,062	1,655	228	1,357
1994	6,146	1,627	2,069	1,627	177	1,389
1995	5,527	1,536	2,562	1,536	30	1,327
1996	872	264	313	264	4	168
1997	569	212	111	212	<1	161
1998	581	745	496	745	53	644
1999	1,959	1,200	1,168	1,200	<1	1,004
2000	4,802	1,402	2,149	1,402	<1	1,173
2001	4,339	1,448	3,016	1,448	8	1,217

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

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, anchovies, Atlantic menhaden, grubby, cunner, tautog, and winter flounder. Silversides were also selected for further analysis because their distribution in Jordan Cove may be affected by the MPS thermal discharge. The Δ-mean densities (no./500m³) of eggs and larvae, Δ-mean CPUE in trawl (no./0.69 km), and seine CPUE (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 and in coastal 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 typically 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 collected in all three Fish Ecology sampling programs, although relatively few juveniles and adults were taken by seine or trawl (Appendices II-V). Most sand lance were collected as larvae in winter and spring at station EN. Abundance appeared to be considerably less in February than during January or March through May. Larval abundance peaked in the late 1970s and early 1980s, followed by a rapid decline beginning in 1982 (Table 6; Fig. 2). The Δ-mean density of larvae at EN during 2002 was 8, which was within the range of the historic data, although among the lowest of the time-series. There was a significant declining trend (p = 0.027) detected over the 26-year time series based on the Mann-Kendall test.

Sand lance larvae are the fourth most abundant larval taxon collected at station EN (Table 1). Annual entrainment estimates, which were based on

TABLE 6. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of American sand lance larvae collected at EN from June 1976 through May 2002.

Year	Δ-mean density
1977	94 ± 17
1978	318 ± 117
1979	119 ± 25
1980	111 ± 26
1981	136 ± 32
1982	21 ± 4
1983	27 ± 8
1984	18 ± 4
1985	9 ± 2
1986	3 ± 1
1987	13 ± 4
1988	41 ± 13
1989	31 ± 13
1990	24 ± 7
1991	7 ± 2
1992	18 ± 6
1993	28 ± 10
1994	43 ± 13
1995	63 ± 29
1996	18 ± 7
1997	11 ± 5
1998	28 ± 15
1999	13 ± 11
2000	53 ± 29
2001	12 ± 4
2002	8 ± 3

[a] Data seasonally restricted to December-May and year given for Δ-means is that of the latter 5 months.

abundance and related to cooling-water flow during the larval season, ranged from 3 to 176 million (Table 5). The 2002 entrainment estimate of 6 million larvae and the MPS cooling-water flow during the period of occurrence were comparable to the historic range of values.

Declines in sand lance abundance during the 1980s were also apparent in other areas of the Northwest

Fig. 2. Annual Δ-mean densities (data points) with a 5-year moving average (line) of American sand lance larvae at station EN from 1977 through 2002.

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 (*Clupea harengus*) and Atlantic mackerel (*Scomber scombrus*) (Nizinski et al. 1990). These two fishes prey heavily upon sand lance, and following their decreases from overfishing in the 1970s sand lance increased in abundance (Sherman et al. 1981; Monteleone et al. 1987). In more recent years, both of these predators have again become more abundant and sand lance abundance has decreased (Stephenson and Kornfield 1990; Smith and Morse 1993; NMFS 1995). Given the large changes in abundance of American sand lance along the Atlantic coast, effects of MPS operation are difficult to ascertain, but are likely small in comparison to the large-scale natural fluctuations typically associated with this species. The sand lance is a short-lived species and, as such, any changes in abundance from the MPS outages should have been readily detected. No changes were evident in 2002 and larval abundance was within its historic range.

Anchovies

The bay anchovy, one of the most abundant fishes found along the Atlantic Coast (McHugh 1967), is 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 anchovies are common in nearshore and estuarine waters during warmer months, but move offshore in winter (Vouglitois et al. 1987). Chesapeake Bay bay anchovies were found to have little genetic variation, indicating a lack of stock structure, likely due to enormous population size and considerable movements and mixing of stocks (Morgan et al. 1995). This homogeneity serves to lessen the effects of any localized impacts. Although the striped anchovy also occurs from Uruguay to Nova Scotia, its occurrence north of the Chesapeake Bay is variable and the striped anchovy is usually found farther 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 in MPS samples, about 96% of the anchovy eggs collected at station EN were determined to be 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 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 hatch in about 24 hours at water temperatures of 27°C (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 II-V). Juvenile anchovies resulting from the summer spawn were typically captured by trawl from August through October. Anchovy abundance in some years varied by two orders of magnitude and the majority were collected in only a few years of sampling at the three trawl stations (Appendices III-V).

Anchovies dominated larval collections and their eggs ranked third in abundance (Table 1). Eggs and larvae occurred in collections made from June through September. Annual egg and larval abundances were significantly correlated (Spearman's rank-order correlation coefficient $r = 0.71$; $p = 0.001$). In 2001, the Δ-mean density of 29.0 for anchovy eggs was among the lowest recorded but was higher than the all-time low abundances observed in 1999 and 2000 (Table 7; Fig. 3). Despite the increase in egg abundance, the Δ-mean density of 58 for larvae was the lowest observed in the 26-year period (Table 7; Fig. 3). Other incidences of large annual changes in bay anchovy egg abundance were observed in LIS during 1952-55 (Richards 1959) and in Barnegat Bay, NJ during 1976-81 (Vouglitois et al. 1987). At MPS, egg and larval densities decreased by an order of magnitude following the early to mid-1980s and have remained relatively low, except larvae in 1990. Results of a Mann-Kendall test indicated that densities of both eggs and larvae had significant negative trends (eggs $p = 0.001$, slope $= -38.8$; larvae $p = 0.001$, slope $= -48.9$) for the entire time-series.

TABLE 7. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of anchovy eggs and larvae collected at EN from June 1976 through May 2001.

Year	Δ-mean density of eggs	Δ-mean density of larvae
1976		1,152 ± 419
1977		931 ± 408
1978		483 ± 206
1979	1,558 ± 807	2,168 ± 908
1980	999 ± 689	2,430 ± 1,249
1981	769 ± 273	5,768 ± 3,326
1982	499 ± 202	816 ± 240
1983	2,415 ± 2,130	1,421 ± 530
1984	3,631 ± 3,528	302 ± 165
1985	118 ± 72	1,102 ± 453
1986	586 ± 366	1,244 ± 893
1987	64 ± 19	126 ± 69
1988	32 ± 26	359 ± 216
1989	32 ± 29	619 ± 416
1990	89 ± 88	1,122 ± 853
1991	317 ± 306	799 ± 801
1992	62 ± 70	178 ± 80
1993	329 ± 260	203 ± 103
1994	234 ± 204	475 ± 410
1995	118 ± 154	181 ± 117
1996	36 ± 29	175 ± 196
1997	5 ± 5	131 ± 90
1998	114 ± 123	106 ± 80
1999	1 ± 1	129 ± 135
2000	<1 ± 1	65 ± 39
2001	29 ± 61	58 ± 90

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

The entrainment of eggs and larvae by MPS probably represents the greatest direct operational impact on anchovies that spawn in the Millstone area. In 2001, the entrainment estimate of eggs (8 million) was higher than the 1999 and 2000 estimates (Table 4). The larval entrainment estimate (24 million) was similar to the 1996, 1997, 1998, and 2000 estimates, but was much lower than in the past (Table 5). Densities of both anchovy eggs and larvae had significant negative trends. Bay anchovies appear to be experiencing a regional decline in abundance. A sharp drop in abundance was measured over the past decade in Narragansett Bay, Rhode Island (Tim Lynch, RI DEM, pers. comm.). The Maryland DNR Juvenile Finfish Seining Survey data also indicated that the bay anchovy population declined dramatically in Chesapeake Bay after 1993 (Price 1999). The decrease in anchovy eggs and larvae seen in recent years is probably reflective of regional decreases in abundance. The fact that there is little genetic variation among the abundant anchovy stocks along the Atlantic Coast serves to lessen any localized impacts of fluctuations in abundance.
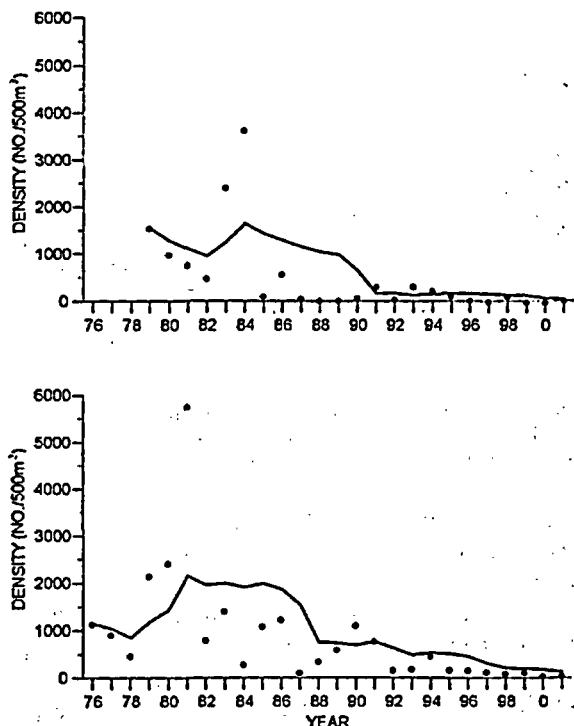
Fig. 3. Annual Δ-mean densities (data points) with moving averages (line) of 4 years for anchovy eggs (1979-2001) and 5 years for anchovy larvae (1976-2001) at station EN.

Atlantic menhaden

The Atlantic menhaden is a pelagic species that filter-feeds on phytoplankton, zooplankton, and detritus (Lewis and Peters 1994). Atlantic menhaden have been reported from Nova Scotia to southeastern Florida, but are most common from the Gulf of Maine to northern Florida (Reintjes 1969; Ahrenholz 1991). It is a coastal migratory species considered to be estuarine-dependent, with young-of-the-year found in coastal bays, rivers, and sounds, often penetrating to upstream limits of saline water (Reintjes and Pacheco 1966). The Atlantic menhaden supports the largest commercial fishery along the Atlantic coast with landings primarily used for fish meal, oil, and solubles (Vaughan and Smith 1988; Ahrenholtz et al. 1987). This species also serves as forage for many marine fishes, birds, and mammals. Although it has a life span of up to 10-12 years, the current fishery is highly dependent upon pre-spawning fish; most females mature during their second year of life at about 180 mm in length (Lewis et al. 1987). Based on its movements and distribution, the Atlantic menhaden is treated as a single stock for manage-

ment, although Epperly (1989) found some evidence for northern and southern subpopulations that were based on meristic and biochemical differences.

Atlantic menhaden exhibit a distinct annual cycle of movements (Nicholson 1971, 1978; Dryfoos et al. 1973; Kroger and Guthrie 1973). More localized movements have been attributed to responses to gradients in the densities of their phytoplankon prey (Friedland et al. 1989, 1996). Fish are found throughout most of their range during summer, although the population is generally stratified by age and size, with the oldest and largest individuals found farthest north. Adult fish begin to move southwards in September, followed by juveniles later in fall and early winter. Atlantic menhaden overwinter in waters off the Carolinas and southward and begin relatively rapid movements northward in March and early April and re-occupy their summer range by about June. Spawning takes place at night (Ferraro 1980b), and can occur during any month of the year, but the presence of larvae and juveniles in specific waters is largely associated with the annual movement/ reproductive cycle. Most spawning takes place in the ocean where early larval growth and development occurs. Some spawning also takes place in major estuaries (Ahrenholz 1991). Spawning was reported to occur within LIS and adjacent waters from late spring through early fall (Dietrich 1979; Ferraro 1981; Powell and Phonlor 1986). The bimodal distribution of larvae found in the MPS samples, however, suggested two distinct spawning periods in LIS, a larger one in the summer and a smaller event during fall. Ferraro (1981) also reported similar bimodal spawning periods for Atlantic menhaden in the Peconic Bays, NY. Atlantic menhaden found offshore of New England and New York waters begin spawning in September and spawning activity increases as fish move progressively south in October and November, peaking off North Carolina in winter. Eggs and larvae are tolerant of a relatively wide range of salinities and temperatures (Ferraro 1980a). Small larvae are moved landward by Ekman transport (Nelson et al. 1977), and further development occurs in nearshore coastal and estuarine waters, which are sought out by juveniles. Larval and juvenile movements (including diel vertical migration) have been found to be affected by environmental cues, such as light (Forward et al. 1993, 1996) and temperature (Friedland and Haas 1988; De Vries et al. 1995) with increased swimming activity and movement towards surface waters occurring at night.

Atlantic menhaden were taken in all three monitoring programs, but were collected irregularly by trawl or seine. Most (90%) Atlantic menhaden were collected in 6 out of 26 years of sampling at the

trawl stations (Appendices III-V). The majority (53%) of the Atlantic menhaden collected by trawl were found at NR. The Atlantic menhaden collected by seine at JC were mostly (>70%) taken in the last 4 years, although only 24 were collected this year (Appendix II). The seine catch of Atlantic menhaden was composed entirely of young-of-the year, reflecting the landward migration of juveniles seeking coastal and estuarine nursery areas as suggested by Nelson et al. (1977).

Some eggs of Atlantic menhaden were identified in entrainment collections taken from April 1979 through September 1992, but they were never abundant and made up less than 0.1% of all fish eggs taken (NUSCO 1994a). This suggested that most spawning in LIS did not occur near MPS, even though the Atlantic menhaden was the second most abundant larval taxon collected (Table 1). The Δ-mean density of 80 in 2001 for Atlantic menhaden was within the range of values reported and larvae were less than one-third as abundant as in 2000 when abundance was the highest recorded (Table 8). Larval densities have increased since 1988, with many density estimates during recent years above the moving average (Fig. 4). Based on a Mann-Kendall
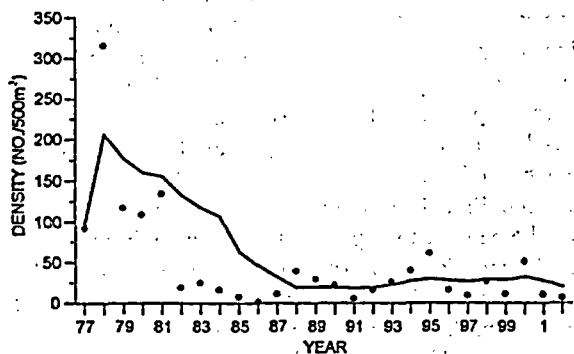

Fig. 4. Annual Δ-mean densities (data points) and a 5-year moving average (line) of Atlantic menhaden larvae at station EN from 1976 through 2001.

test, a significant (p = 0.001; slope = 2.8) increasing trend was detected during the period from 1976 to 2001.

The foremost operational impact by MPS to Atlantic menhaden is the entrainment of larvae. In 2001, the entrainment estimate of 147 million larvae was less than one-third that observed in 2000 when entrainment was the highest recorded (Table 5). The large increase in Atlantic menhaden abundance over the past decade indicated that MPS has had minimal or no impact on their numbers, with abundance in northern waters likely increasing because of reduced fishing mortality along the Atlantic coast. Variation in entrainment and catch of young likely reflected variation in the spawning and transport of eggs and larvae within LIS.

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). Both species provide important trophic links between zooplankton and larger piscivorous fishes and birds. 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

TABLE 8. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of Atlantic menhaden larvae collected at EN from June 1976 through May 2001.

Year	Δ-mean density
1976	5 ± 1
1977	3 ± 1
1978	3 ± 1
1979	1 ± 1
1980	2 ± 1
1981	1 ± 0.4
1982	9 ± 3
1983	18 ± 10
1984	2 ± 1
1985	38 ± 22
1986	2 ± 1
1987	2 ± 1
1988	5 ± 4
1989	47 ± 64
1990	16 ± 12
1991	81 ± 140
1992	37 ± 28
1993	8 ± 4
1994	44 ± 76
1995	56 ± 60
1996	145 ± 431
1997	23 ± 18
1998	28 ± 60
1999	58 ± 63
2000	293 ± 373
2001	80 ± 135

[a] Data seasonally restricted to June-December.

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 rapid and mean lengths of Atlantic silverside 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 silversides 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 silversides have small home ranges (Hoff 1972). Silversides typically suffer high (>99%) overwinter mortality (Conover and Ross 1982; Conover 1992).

Nearly all silversides collected in the trawl monitoring program were Atlantic silverside, with less than 0.1% identified as the inland silverside since 1981, when the two species were first distinguished in the MPS sampling. More than 80% of the silversides collected by seine were Atlantic silverside, although relative proportions varied from year to year. Collectively, silversides dominated the seine catch (80%) at JC (Appendix II) and were taken by trawl (7% of the trawl catch) in winter (Appendices III-V), but were rarely (< 0.1% of larvae collected since 1976) found in ichthyoplankton collections.

Silversides were taken at the three trawl stations from October through February after individuals left the shore areas to overwinter in deeper waters. Annual trawl abundances fluctuated considerably (Table 9). Few silversides were taken at IN in 2000-01, but abundance increased in 2001-02. At JC and NR, the 2001-02 Δ-mean CPUE were within the historical ranges. The 2001-02 Δ-mean CPUE was above the moving average at JC and IN and just below the moving average at NR (Fig. 5). A significant negative trend was found at IN (p = 0.002, slope = -3.0) over the 26-year period, but no trend was found at NR or JC based on the Mann-Kendall test. The Δ-mean CPUE for Atlantic silverside taken by seine during 2001 (136) was within the range of observed values while the CPUE for inland silverside was the lowest recorded (0.9; Table 10). In 2001, as in previous years, the abundance of Atlantic silverside greatly exceeded that of the inland silverside (Fig. 6). The 2001 annual Δ-mean CPUE value for Atlantic silverside was greater than the moving average, however, inland silverside abundance was less than its moving average (Fig. 7). Abundance data of both species showed no significant trends for the 21 years examined with the Mann-Kendall test.

To determine if any changes in size distributions

TABLE 9. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of Atlantic silverside collected by trawl at three selected stations from June 1976 through May 2002.

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
1996-97	4 ± 2	0.4 ± 0.1	13 ± 8
1997-98	1 ± 1	0.5 ± 0.5	2 ± 5
1998-99	1 ± 1	21 ± 65	7 ± 11
1999-00	3 ± 3	8 ± 7	7 ± 4
2000-01	<1 ± 0	1 ± 2	3 ± 7
2001-02	3 ± 6	9 ± 40	3 ± 13

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

have occurred, the length-frequencies for seine and trawl catches (expressed as percentages) were examined comparing the current year with historic data (Fig. 8). More 60-80 mm Atlantic silverside and smaller (< 40mm) inland silversides were observed in seine samples this year when compared to the historic length range. The silversides caught by trawl were similar in size as those found in previous years, although there was a somewhat higher proportion of 60-80 mm fish than in the past (Fig. 8).

The primary impact of the operation of MPS on silversides may be the incursion of the thermal plume into the shore-zone area, potentially causing disruption in spawning activity and influencing the distribution of juveniles and adults. The thermal increase is only 0.8°C at the JC station (NUSCO 1988b). 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 MPS thermal plume (see the Eelgrass section of this report and Keser et al. 2002). 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.
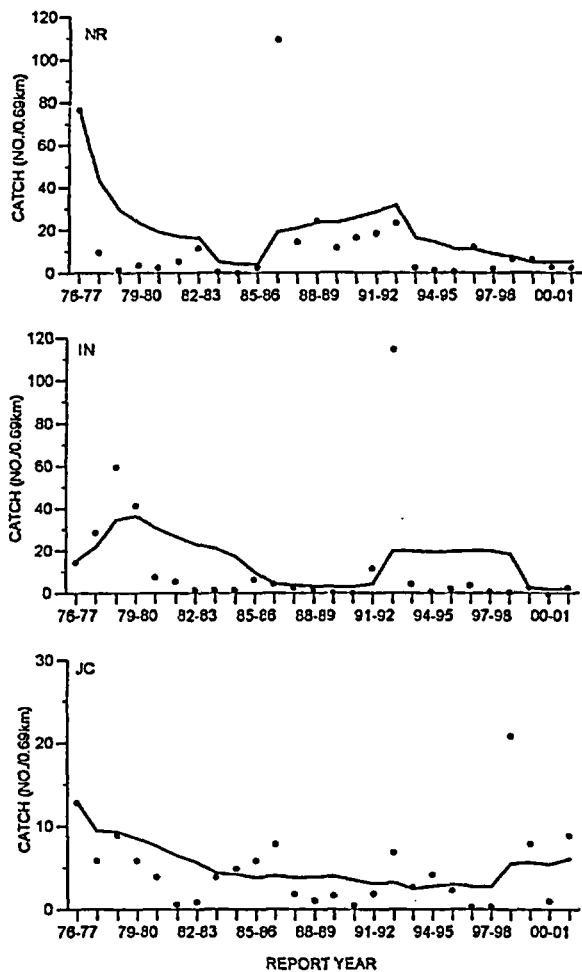
Fig. 5. Annual Δ-mean CPUE (data points) and a 5-year moving average (line) of Atlantic silversides taken by trawl at stations NR, IN, and JC from 1976-77 through 2000-02. Note that vertical scales differ among graphs.

Grubby

The grubby is a demersal fish found in shallow waters along the Atlantic coast from the Gulf of St. Lawrence to New Jersey. It 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 occur throughout the year near MPS. Similar to the winter flounder, grubby produce blood plasma antifreeze proteins and can remain active in very low water temperatures (Reisman et al. 1987). Female grubby reach maturity within one year (Lazzari et al. 1989). Grubby spawn throughout the winter and have a demersal, adhesive egg with an incubation time of 40

TABLE 10. 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 1981 through 2001.

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
1996	78 ± 60	27 ± 23
1997	103 ± 78	52 ± 37
1998	93 ± 94	5 ± 7
1999	108 ± 79	4 ± 3
2000	135 ± 201	9 ± 13
2001	136 ± 550	1 ± 9

[a] Data seasonally restricted to June-November.

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. The grubby is a small fish and has no sport or commercial value. Given its protective spines and cryptic coloration, it probably also has limited forage value. Grubby prey upon many small fishes and benthic invertebrates (Lazzari et al. 1989; Levin 1991).

The grubby was the fifth-most abundant larval fish collected at EN and primarily occurred in samples from February through May. Grubby accounted for
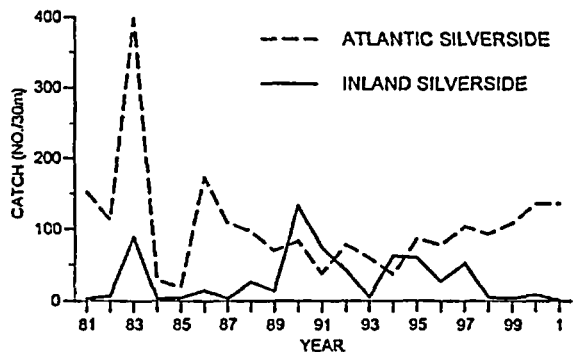
Fig. 6. Annual Δ-mean CPUE (no./30 m) of Atlantic silverside (dashed line) and inland silverside (solid line) taken by seine at station JC from 1981 through 2001.
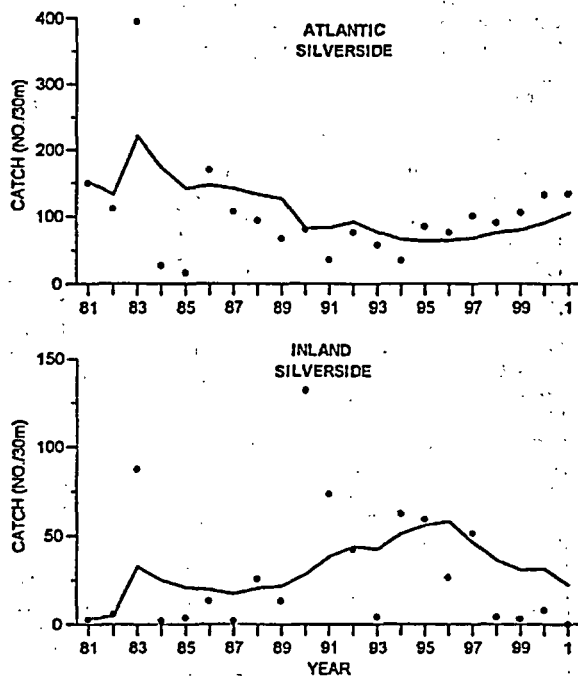
Fig. 7. Annual Δ-mean CPUE (data points) and the 5-year moving average (line) of Atlantic silverside and inland silverside taken by seine at station JC from 1981 through 2001. (Note that the vertical scales differ between the graphs).

Fig. 8. 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 from June 1976 through May 1999 and from June 2000 through May 2001.

5.3% of all larvae collected from June 1976 through May 2002 (Table 1). The Δ-mean density of larvae for 2002 of 27 was the second lowest recorded (Table 11) and well below the 5-year moving average (Fig. 9). Because of fluctuating annual abundance, a Mann-Kendall test revealed no significant temporal trend in larval abundance. The entrainment estimate of 33 million larvae in 2002 was among the lowest observed because of the low density (Table 5).

Predominantly a shallow-water fish, the grubby was the fifth-most abundant fish taken by trawl at the three stations with over 60% of the total collected at NR (Appendices III-V). In 2001-02, catches at NR and JC trawl stations were below the moving average, with the catch at NR being the among the lowest recorded (Table 12; Fig. 10). No significant trends in abundance were found although grubby were found in low abundance at NR for the past 3 years. Smaller grubbies were generally taken by trawl in 2001-02 based on the percent length-frequency distributions (Fig. 11).

Entrainment of grubby larvae is the primary direct plant impact on the resident grubby population. Entrainment of grubby larvae in 2002 was less than the moving average at only 33 million, much lower than the values recorded in 2001 which was the all-

time highest recorded (Table 5). This unexploited species has been among the most stable of the fishes residing near MPS. Because the grubby is a short-lived species that matures in 1 year, any changes in abundance resulting from MPS would have been apparent. Since grubby abundance has been stable, it is likely that the plant has had little or no effect on the population.

Cunner

The cunner occurs from Newfoundland to Chesapeake Bay (Scott and Scott 1988), and prefers structural habitats, such as rocks, pilings, eelgrass or mussel beds, and macroalgae. Cunner are inactive at night (Bradbury et al. 1997) and become torpid when

TABLE 11. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of grubby larvae collected at EN from June 1976 through May 2002.

Year	Δ-mean density
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
1997	140 ± 60
1998	55 ± 19
1999	39 ± 24
2000	45 ± 19
2001	188 ± 89
2002	27 ± 18

[a] Data seasonally restricted to February-May.

TABLE 12. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of grubby collected by trawl at three selected stations from June 1976 through May 2002.

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
1996-97	5.5 ± 1.3	0.7 ± 0.1	2.3 ± 0.5
1997-98	3.7 ± 1.6	0.9 ± 0.3	2.7 ± 1.1
1998-99	3.3 ± 1.0	1.8 ± 0.6	1.3 ± 0.5
1999-00	0.8 ± 0.2	1.2 ± 0.4	0.9 ± 0.4
2000-01	0.7 ± 0.3	0.6 ± 0.3	<0.1 ± 0
2001-02	0.6 ± 0.5	0.9 ± 0.9	1.5 ± 1

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

water temperatures fall below 5-8°C (Green and Farwell 1971; Olla et al. 1975; Dew 1976). Individual cunner 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 do not undertake extensive movements (Green and Farwell 1971; Olla et al. 1979; Lawton et al. 1996). Most cunner live 5 to 6 years, with a maximum age of about 10, which is
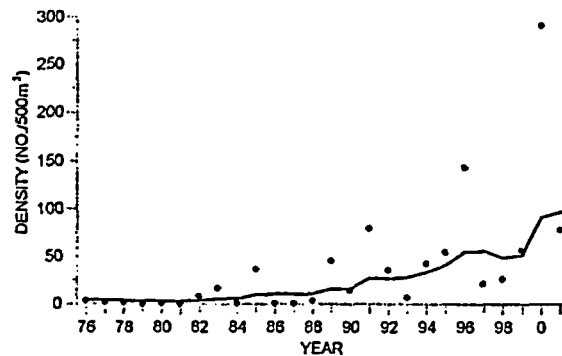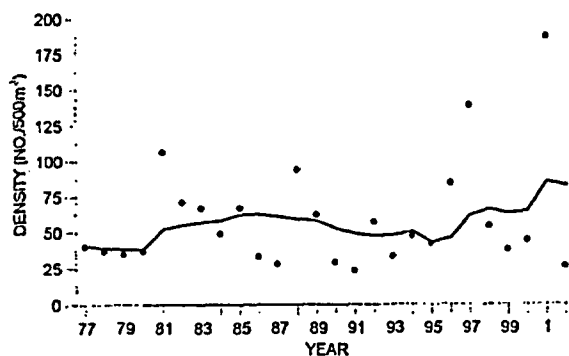

Fig. 9. Annual Δ-mean densities (data points) with a 5-year moving average (line) of grubby larvae at station EN from 1977 through 2002.

less than one-third of the life span of the closely related tautog (Dew 1976; Regan et al. 1982).

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). Larger (>20 cm) males may spawn with individual females, whereas smaller (80-180 mm) males and females spawn in large (30-150+) groups (Pottle et al. 1981). 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 (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). Tupper and Boutilier (1997) reported that settlement was not affected by habitat type or adult density, but post-settlement survival, recruitment success, and adult densities were positively correlated with habitat complexity. They concluded that habitat-mediated post-settlement processes played an important role in cunner population dynamics. Levin et al. (1997) hypothesize that habitat-related differences in post-
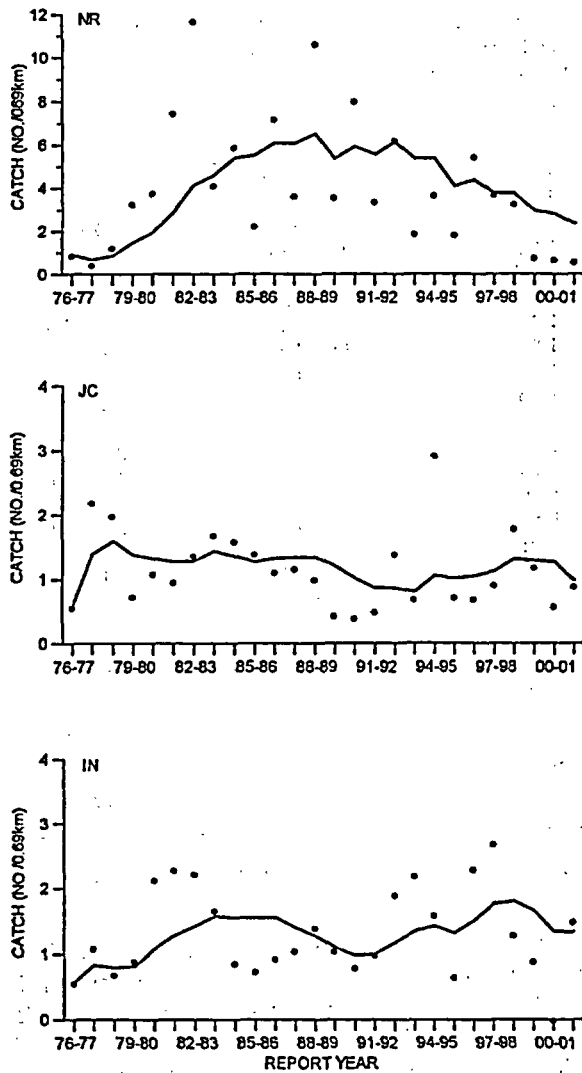
Fig. 10. Annual Δ-mean CPUE (data points) with a 5-year moving average (line) of grubby taken by trawl at stations NR, IN, and JC from 1976-77 through 2001-02. (Note that the vertical scales differ among the graphs).
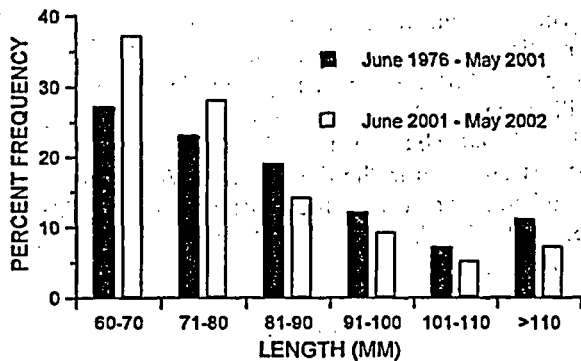
Fig. 11. Length-frequency distribution (10-mm length intervals) of grubby taken by trawl at stations NR, IN, and JC from June 1976 through May 2001 and from June 2000 through May 2002.

settlement processes are the most significant factors affecting cunner populations in the Gulf of Maine.

The cunner has little commercial value and is generally not sought after by sport fisherman, although numerous individuals are often caught while fishing for other species (MacLeod 1995). Regionally, declining trends in adult cunner abundance have been observed in LIS (CTDEP 1998; Smith et al. 1989), Cape Cod Bay (Lawton et al. 1994; Nitschke et al. 2002), and Mount Hope Bay (MRI 1994).

In the MPS area, cunner eggs and larvae are present primarily from May through July. Among the early developmental stages collected at station EN, cunner eggs were the most abundant of all egg taxa, whereas larvae were less common, ranking only eighth overall (Table 1). The 2001 Δ-mean density of eggs was 6,495 and was within the range of historic data (Table 13), as was the egg entrainment estimate of 4,339 million (Table 4). The 2001 annual Δ-mean egg density was above the moving average (Fig. 12), while the corresponding larval Δ-mean density of 79 was the second highest recorded and was higher than the moving average (Table 13; Fig. 12). Based on the Mann-Kendall test, no significant temporal trends were detected for cunner eggs and larvae.

TABLE 13. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of cunner eggs and larvae collected at EN from June 1976 through May 2001.

Year	Δ-mean density of eggs	Δ-mean density of larvae
1976		29 ± 14
1977		58 ± 28
1978		1 ± 0
1979	8,349 ± 4,601	13 ± 5
1980	8,379 ± 3,788	58 ± 19
1981	7,326 ± 3,890	78 ± 36
1982	7,874 ± 2,359	31 ± 14
1983	7,580 ± 6,106	49 ± 26
1984	6,707 ± 4,494	4 ± 2
1985	12,842 ± 6,156	12 ± 10
1986	2,579 ± 1,460	4 ± 1
1987	5,017 ± 3,536	5 ± 3
1988	5,388 ± 3,608	9 ± 4
1989	6,255 ± 5,302	14 ± 12
1990	7,269 ± 7,198	68 ± 61
1991	6,987 ± 5,482	209 ± 157
1992	2,776 ± 1,654	8 ± 4
1993	4,535 ± 2,612	10 ± 6
1994	8,722 ± 9,644	25 ± 18
1995	4,266 ± 3,222	12 ± 9
1996	8,801 ± 6,043	12 ± 6
1997	3,610 ± 3,157	7 ± 4
1998	1,458 ± 6,313	21 ± 18
1999	3,530 ± 3,044	15 ± 11
2000	8,245 ± 8,752	17 ± 14
2001	6,495 ± 13,317	79 ± 144

[a] Data seasonally restricted to May-August for eggs and June-August for larvae.
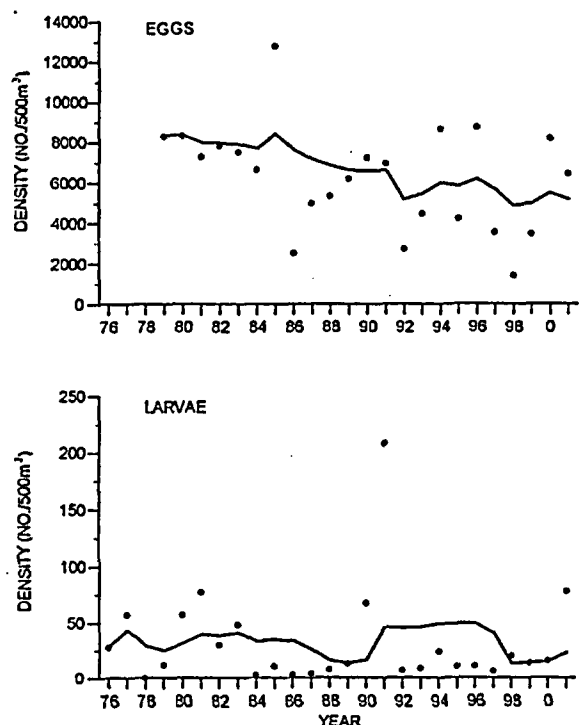
Fig. 12. Annual Δ-mean densities (data points) with moving averages (line) of 4 years for cunner eggs (1979-2001) and 5 years for cunner larvae (1976-2001) at station EN. (Note that the vertical scales differ between the graphs).

TABLE 14. The annual Δ-mean[a] CPUE (no./0.69 km) and 95% confidence interval of cunner collected by trawl at two selected stations from June 1976 through May 2001.

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
1996	0.3 ± 0.2	0.2 ± 0.2
1997	0.5 ± 0.3	0.9 ± 0.6
1998	0.7 ± 0.4	7.2 ± 5.2
1999	0.4 ± 0.2	11.2 ± 10.0
2000	0.9 ± 0.8	13.9 ± 10.0
2001	1.7 ± 12	4.4 ± 5

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

Cunner were rarely taken by seine at JC (Appendix II), but juveniles and adults were caught by trawl, mostly from spring through summer. More cunner were taken at IN (4,343) than at JC (4,106), with relatively few (672) found at NR (Appendices III-V). However, during the past 4 years, many more cunner were collected at JC (2,000) than at IN (185). The 2001-02 annual Δ-mean CPUE values for IN (1.7) remained below catches made in the late 1970s and early 1980s, whereas the 2001 Δ-mean CPUE value for JC (4.4) was within the range of values previously recorded (Table 14). Annual Δ-mean CPUE at JC was below the moving average, while catches at IN were just above the moving average (Fig. 13). Based on the Mann-Kendall test no significant trend in abundance was detected at JC. However, catches exhibited a significant decreasing trend at IN (p < 0.002; slope =-0.3). 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 may have increased their availability to sampling by trawl at the nearby IN station (NUSCO 1997). The Mann-Kendall test was again executed for annual catches (1984-2001) at IN after

the cofferdam was removed and no trend (p=0.8; slope=0) was detected. Ages of cunner were assigned based on an age-length key provided by Serchuk (1972) to determine an age-frequency distribution of fish collected by trawl. Percent length-frequency distributions were determined for past years (1976-2000) and 2000-01 (Fig. 14). The size distributions in 2000-01 differed from the historic data with almost 80% of the cunner caught during 2000-01 being young-of-the-year.

The entrainment of eggs represents the greatest potential impact on the cunner population in the vicinity of MPS. Estimated egg entrainment of 4,339 million in 2000-01 was within the historic range while larval abundance in 2001 was the second highest observed in the time series. With increasing abundance of early life history stages, MPS apparently has had little effect on the local cunner population.

Tautog

The tautog ranges from New Brunswick to South Carolina, but is most common from Cape Cod to the
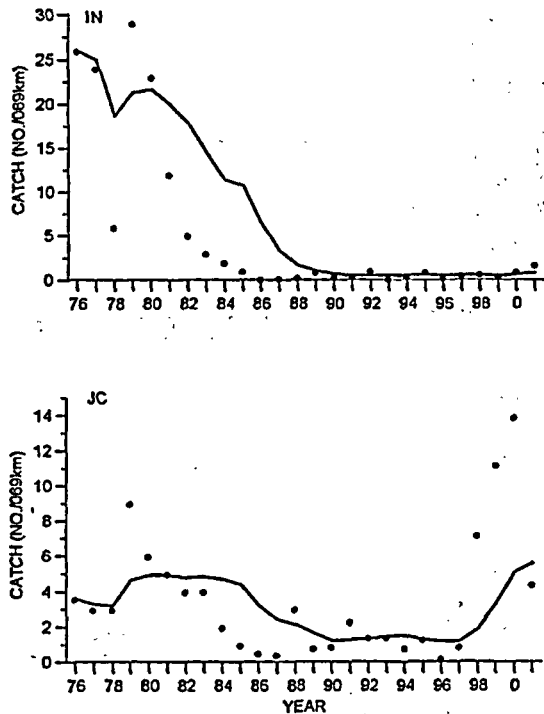
Fig. 13. Annual Δ-mean CPUE (data points) with the 5-year moving average (line) of cunner taken by trawl at stations IN and JC from 1976 through 2001. (Note that the vertical scales differ between the graphs).

Delaware Capes (Bigelow and Schroeder 1953). Tautog are active during the day, but are quiescent during night (Olla et al. 1974). Adult tautog prefer rocky areas and similar reef-like habitats near shore from spring through fall. Juveniles are typically found in eelgrass beds and among macroalgae in coves and estuaries (Tracy 1910; Bigelow and Schroeder 1953; Wheatland 1956; Cooper 1965; Briggs and O'Conner 1971; Sogard and Able 1991;
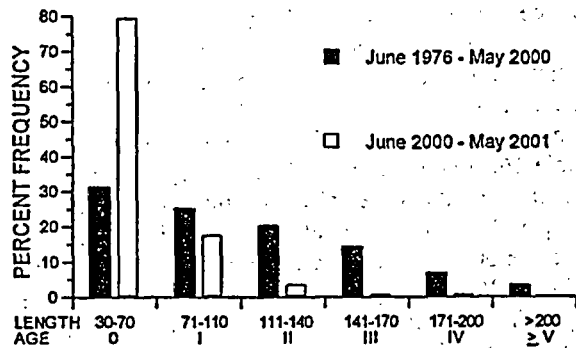
Fig. 14. Length-frequency distribution by length (mm) and age (determined from age-length key of Serchuk 1972) of cunner taken by trawl at stations IN and JC from June 1976 through May 2000 and from June 2000 through May 2001.

Hostetter and Munroe 1993; Dorf and Powell 1997). Young tautog are more restricted in their movements than adults and remain in close association with shelter throughout the year (Olla et al. 1978). 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 1964). Adult growth rates have been estimated for several regions ranging from Narragansett Bay to Virginia (Cooper 1964; 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) and larvae are capable of feeding by 52 hours post-hatching (Schoedinger and Epifanio 1997). 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). In laboratory experiments, Laurence (1973) found that larval tautog may encounter potential physiological stress at higher (>19°C) temperatures and Schoedinger and Epifanio (1997) found that prey densities strongly affected larval growth and development. Size at the end of the first growth season in Narragansett Bay (about 50 mm total length; Dorf 1994) was less than that 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.

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 its sympatric species, cunner, have similar early life history characteristics, and annual Δ-mean densities of their eggs are correlated (Spearman's rank-order correlation coefficient r = 0.47; p = 0.03). The 2001 Δ-mean density for tautog eggs of 2,759 was within the historic range and along with a similar to the high values found in 1996 and 2000, which were among the highest densities seen since 1989 (Table 15). The

TABLE 15. The annual Δ-mean[a] density (no./500 m³) and 95% confidence interval of tautog eggs and larvae collected at EN from June 1976 through May 2000.

Year	Δ-mean density of eggs	Δ-mean density of larvae
1976		37 ± 16
1977		36 ± 17
1978		1 ± 1
1979	1,648 ± 566	11 ± 5
1980	3,741 ± 1,482	46 ± 18
1981	2,501 ± 604	83 ± 36
1982	3,561 ± 1,400	44 ± 21
1983	2,372 ± 994	33 ± 21
1984	1,817 ± 504	3 ± 2
1985	4,027 ± 2,424	15 ± 12
1986	2,833 ± 1,212	3 ± 2
1987	2,972 ± 1,232	7 ± 3
1988	2,211 ± 906	17 ± 10
1989	3,373 ± 2,002	15 ± 7
1990	1,942 ± 978	33 ± 28
1991	2,040 ± 1,052	99 ± 51
1992	1,189 ± 462	13 ± 4
1993	1,394 ± 582	6 ± 3
1994	1,350 ± 658	12 ± 8
1995	1,807 ± 798	8 ± 4
1996	2,323 ± 2,032	18 ± 16
1997	587 ± 235	2 ± 1
1998	897 ± 423	14 ± 13
1999	1,373 ± 636	58 ± 41
2000	2,322 ± 1,336	12 ± <1
2001	2,759 ± 3,012	113 ± 272

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

Fig. 15. Annual Δ-mean densities (data points) with moving averages (line) of 4 years for tautog eggs (1979-2001) and 5 years for tautog larvae (1976-2001) at station EN. (Note that the vertical scales differ between the graphs).

2001 Δ-mean was above the 5-year moving average line (Fig. 15), although a significant negative trend (p = 0.022, slope = -64.5) was found over the 26-year period according to a Mann-Kendall test. The 2001 annual egg entrainment estimate of 3.0 billion also fell within the historic range, but was the highest estimate since 1995 (Table 4).

In contrast to eggs, tautog larvae were not a predominant taxon, ranking seventh in entrainment since 1976 (Table 1). Larval abundance in 2001 was the highest ever observed (Fig. 15). No trends were found in larval abundance. Annual abundances of tautog and cunner larvae were significantly correlated (Spearman's rank-order correlation coefficient r = 0.84; p < 0.001), indicating common processes that affected the abundance of larvae for both of these wrasses.

A preliminary study to examine initial and latent survival of entrained tautog and cunner eggs was conducted on four occasions during the summer of 2001 and again on 5 dates in 2002 using a 0.5-m plankton net having 0.333-mm mesh. To collect eggs, the net was hauled vertically in the Unit 3 discharge and also in Niantic Bay near the MPS intakes to

provide control samples. Net contents were emptied into a bucket with seawater of the same temperature as the collection site and sorted in the laboratory. Sample processing was typically completed within 3 hours from the time of collection. Collected eggs were assessed for viability and live specimens were incubated in chambers with flowing seawater for up to 60 hours to assess latent mortality. Identification to species was determined by examining hatched larvae at the completion of the incubation period. Initial viability of wrasse eggs was consistently higher at the intake site than at the discharge site on all sample dates. The survival data is summarized in Table 16. On average, 53% of the eggs collected at the intake site were alive at time of collection compared to only 20% alive at the discharge site. Latent mortality was higher for entrained eggs with only 11% of eggs surviving to hatch while approximately 29% of the eggs collected at the intake site hatched. Tautog larvae were more prevalent than cunner on all sampling dates at both sites in 2002, comprising between 52% and 80% of hatched larvae but were less than 50% at IN in 2001 (Table 16).

Juvenile and adult tautog were sampled in two sampling programs at MPS, trawl and lobster pot

TABLE 16. Summary of wrasse egg entrainment survival study conducted in 2001 and 2002.

Date sampled	Collection site	Total no. eggs examined	No. viable 0-hours	No. hatched 12-hours	No. hatched 36-hours	No. hatched 60-hours	Initial % viable	% hatched	% tautog
18 June 01	Intake	173	90	0	2	33	52.0	20.2	60.3
	Discharge	438	110	0	8	5	25.1	3.0	39.7
25 June 01	Intake	227	66	15	12	3	29.1	13.2	77.3
	Discharge	437	77	18	7	2	17.6	6.2	22.7
9 July 01	Intake	220	188	1	59	2	85.5	28.2	69.4
	Discharge	583	192	1	7	1	32.9	-1.5	30.6
23 July 01	Intake	15	7	0	0	0	46.7	0	n/a
	Discharge	262	10	0	0	0	3.8	0	n/a
4 June 02	Intake	452	391	2	9	8	86.5	17.9	68.0
	Discharge	223	106	0	0	5	47.5	4.7	80.0
10 June 02	Intake	53	36	0	4	16	67.9	55.6	58.0
	Discharge	264	58	0	0	8	22.0	13.8	52.0
17 June 02	Intake	150	109	1	2	62	72.7	59.6	71.0
	Discharge	508	94	0	0	12	18.5	12.8	66.0
1 July 02	Intake	22	7	0	1	3	31.8	57.1	63.2
	Discharge	154	16	0	0	1	10.4	6.3	65.0
8 July 02	Intake	255	104	0	0	8	40.8	7.7	71.0
	Discharge	31	12	0	0	6	38.7	50.0	66.0

monitoring. Data were analyzed on an annual basis to compare catches from the two programs. Tautog were caught infrequently by trawl because they prefer rocky or reef habitats and are less vulnerable to this sampling gear; annual Δ-mean CPUE could not be calculated because of too many zero values. As an alternative, the annual sum of catches at the trawl stations were used as an index of abundance (Table 17). No significant trends were found for the combined catch of tautog at the three inshore trawl stations during the 26-year period. However, the mostly young tautog taken by trawl had the highest catches during the past 3 years.

Tautog are routinely found in pots used in the lobster population monitoring program (see the Lobster Studies section for details). Since 1984, these fish were counted and measured to provide another index of tautog abundance. Total annual (May-October) catches were examined at each of the three lobster monitoring program stations (Jordan Cove, designated herein as JC; Intake, IN; and Twotree, TT) (Table 18). In the lobster pot program, no significant trends were found for the three stations combined, at IN, or TT. However, there was a significant increasing trend (slope = 6.3; $p < 0.009$) in the catch of tautog at JC.

TABLE 17. Total annual catch of tautog collected by trawl at three selected stations from 1976 through 2001.

Year	NR	JC	IN	Total
1976	46	73	76	195
1977	15	113	70	198
1978	27	59	83	169
1979	47	56	70	173
1980	25	20	46	91
1981	126	24	28	178
1982	80	35	52	167
1983	31	19	40	90
1984	5	16	46	67
1985	25	27	47	99
1986	100	58	25	183
1987	26	33	13	72
1988	50	31	37	118
1989	36	23	25	84
1990	89	34	17	140
1991	67	44	13	124
1992	22	90	8	120
1993	15	51	26	92
1994	12	19	13	44
1995	129	73	28	230
1996	40	47	22	109
1997	36	26	24	86
1998	125	71	12	208
1999	238	138	27	403
2000	324	261	45	430
2001	117	119	105	341

TABLE 18. Total annual catch of tautog collected in the lobster monitoring program at selected stations from 1988 through 2002.

Year	IN	JC	TT	Total
1988	47	15	40	102
1989	25	22	20	67
1990	27	8	11	46
1991	48	7	27	82
1992	32	11	21	64
1993	64	12	26	102
1994	29	8	43	80
1995	14	18	10	42
1996	23	134	63	210
1997	12	27	16	55
1998	91	80	64	235
1999	97	86	28	211
2000	85	80	48	213
2001	23	106	8	137
2002	54	76	200	330

Length-frequency distributions of tautog caught by trawl and those found in the lobster pots were compared (Fig. 16). Ages were assigned to length categories based on age-length information for LIS (Simpson 1989). Young tautog, ages 1 and 2, accounted for a high proportion of the tautog caught in the trawl program, while those found in pots were dominated by fish of ages 3, 4 and 5, during which both males and females become mature. Lobster pots select for certain size-classes of tautog because the 2.5 cm^2 wire mesh do 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 two monitoring programs sample different habitats and life history stages of the tautog population. The trawl stations are located near preferred juvenile habitat containing eelgrass beds and macroalgae bottoms in coves and estuaries (Sogard and Able 1991; Hostetter and Munroe 1993; Dorf and Powell 1997). Adult tautog favor rocky areas and similar reef-like habitats near shore from
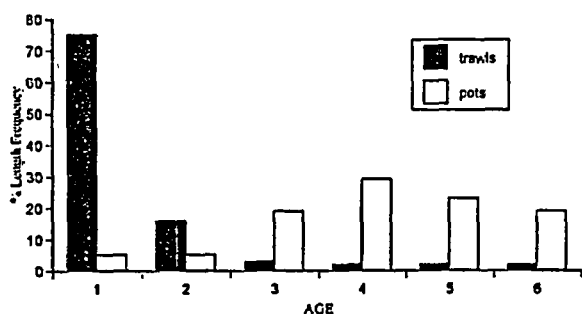
spring through fall (Tracy 1910; Bigelow and Schroeder 1953; Wheatland 1956; Cooper 1964; Briggs and O'Conner 1971) which are sampled by lobster pots. The trawl program provides an index of juvenile tautog in the MPS area, while the lobster pot catches provide a reliable index of newly recruited adults. It is interesting to note that in Figure 17, during some years, peaks in trawl catches preceded those in the lobster pot catches. Most notable was an increase in the 1995 trawl catch followed by a 1996 increase in lobster pot catches. More recently, this pattern did not hold as there were very high catches of juvenile tautog in trawl catches, but lobster pot catches have not yet reflected this increase in abundance.

The greatest direct impact of MPS on tautog stocks is the entrainment of eggs. Tautog eggs are spawned during early evening, the pelagic eggs are dispersed rapidly from spawning sites by tidal transport, and hatch in less than 48 hours. Egg mortality is high immediately following spawning, probably from predation. Since 1993, special studies have been conducted on tautog eggs at MPS to assess the potential entrainment impact on tautog stocks. These tautog egg studies have included diel abundance changes in entrainment samples, spatial distribution in eastern LIS, daily spawning periodicity, and estimation of natural egg mortality. These studies were summarized in NUSCO (1997, 1998, 1999, 2000) and DNC (2001).

Recent work to determine seasonal fecundity estimates based on histological examination of tautog from Virginia waters demonstrated that tautog females are capable of spawning numerous times within a year. White (1996) determined that an individual female tautog spawned, on average, every 1.14 days during their spawning period and, based on this spawning frequency, estimated the total reproductive output for age-3 through 9 female tautog to range between

Fig. 16. Length-frequency distribution by length (mm) and age (determined from age-length key of Simpson 1989) of tautog taken by trawl and lobster pot from 1988 through 2002.
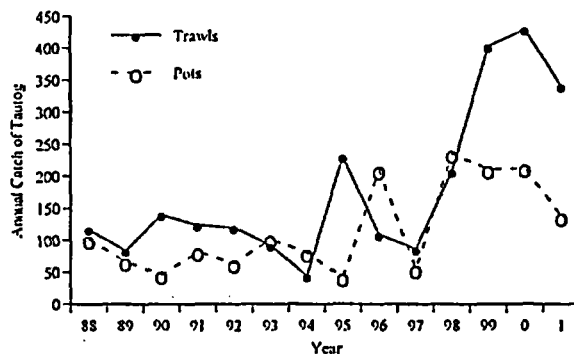
Fig. 17. Total annual catch of tautog in trawl and lobster pot collections from 1988 through 2001.

168,000 and 11,053,00 eggs. Chenoweth (1963) reported maximum egg production per unit of ovary weight for fish 7 to 9 years old with production declining in fish age-16 and older. Seasonal fecundity estimates for tautog in Connecticut waters were verified by a study that was initiated during the summer of 2000 and continued in 2001. This research at the Avery Point campus of the University of Connecticut includes long-term observations of spawning male and female tautog in a laboratory holding facility as well as histological examination of gonads of field-collected fish. A report by LaPlante and Schultz (2002) describing the detailed results of this study was submitted to DEP in July 2002. They concluded that the reproductive performance of captive tautog was not reflective of wild populations. In general, captive fish produced more eggs than field-caught fish. However, captive and wild females were similar in that they demonstrated great plasticity in their reproductive output. Both female size and time of season influenced egg production, and field studies provided evidence for intra- and interseasonal variations in egg output. Predicted annual fecundity in this study was nearly double that observed by White (1996) due to higher batch fecundity. For example, White (1996) predicted that a 400 mm female would produce about 5 million eggs during the course of a season while the results of LaPlante and Schultz (2002) revealed that nearly 10 million eggs could be spawned from a fish the same size. Environmental temperatures did not account for much of the variation observed. They recommend that future research efforts on tautog fecundity examine the influence of food availability and other environmental factors on egg production (Laplante and Schultz 2002).

Recent increases have occurred in tautog egg and larval densities and, based on trawl catches, juvenile tautog abundance has increased in recent years. In addition, the number of newly recruited mature adults has not declined since the late 1980s, as indicated by the selective catch of 3- to 5- year-old tautog in lobster pots in the Millstone area. 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 LIS that began in the mid-1980s and continued through the late 1990s (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 to a decline in the abundance of spawning adults from fishing mortality rather than from the operation of MPS. 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 2002). 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. Recent increases in juvenile abundance may indicate that management measures may be working. However, due to the strong correlation found between the abundance of cunner and tautog larvae, environmental and biotic factors are likely also important in determining year class strength.

Conclusions

Potential MPS impacts on local fish populations include entrainment of eggs and larvae, impingement of juvenile and adults, and distributional changes as a result of the thermal discharge. During the 2001-02 report period, MPS Units 2 and 3 were both operating most of the time and Unit 1 was shut down. Unit 2 was shut down from February 16 through April 2 for refueling. Cooling-water volume was reduced as a result of the decommissioning of Unit 1, which resulted in less entrainment of fish eggs and larvae and impingement of juveniles and adults, as entrainment and impingement rates are directly related to the amount of cooling-water used. Impingement impacts were further reduced at MPS with the installation of an aquatic organism return sluiceway in early 2000 at Unit 2. A 1-year study to evaluate its operation and effectiveness was completed in July 2001 and it was concluded that the system worked as designed and successfully returned impinged marine organisms to Long Island Sound.

Detailed analyses were conducted on seven taxa that were most susceptible to MPS operational impact due to entrainment or effects of the thermal discharge. Analyses of these species generally focused on comparing temporal trends over the past two and one-half decades (Table 19). No significant long-term trends were detected for juvenile and adult silversides by seine sampling at JC, and for all life stages of grubby, cunner eggs and larvae, and tautog and sand lance larvae. Atlantic menhaden larvae showed a significant increasing trend in abundance during the past 26 years. A significant negative trend was observed for silversides at the IN trawl station. Since the mid-1980s, cunner have become less abundant at

TABLE 19. Summary of long-term trends in abundances of various life history stages of selected fish species sampled in the vicinity of MPS from 1976 through 2002.

Species	Program of study and station	Long-term trend
American sand lance	Larval abundance (EN)	Declining
Anchovies	Egg abundance (EN)	Declining
	Larval abundance (EN)	Declining
Atlantic menhaden	Larval abundance (EN)	Increasing
Silversides	Trawl (IN, JC, NR)	Decreasing at IN, none at JC or NR
	Seine (JC)	None
Grubby	Larval abundance (EN)	None
	Trawl (IN, JC, NR)	None
Cunner	Egg abundance (EN)	None
	Larval abundance (EN)	None
	Trawl (IN, JC, NR)	Decreasing at IN, none at JC or NR
Tautog	Egg abundance (EN)	Declining
	Larval abundance (EN)	None
	Trawl (IN, JC, NR)	None
	Lobster pots (IN, JC, TT)	Increasing at JC, none at IN or TT

IN, exhibiting a significant negative trend, which was probably related to removal of the Unit 3 rock coffer-dam, a preferred habitat for this species. Despite the negative trend in abundance for cunner at trawl station IN, tautog abundance was at a historic high at station IN. Tautog eggs exhibited a negative trend in abundance yet the abundance of larvae was the highest observed. The large numbers of tautog and cunner eggs entrained at MPS did not appear to affect future recruitment and subsequent spawning stock biomass of these two fishes because the proportion of juvenile recruits relative to adults has increased and there has been an increasing trend in numbers of tautog captured in lobster pots at JC. Egg survival experiments revealed that approximately 20% of eggs collected at the power plant discharge were viable and overall survival of these was 11%, about half that observed for eggs collected at the intake. Dominion-sponsored research conducted by University of Connecticut scientists showed that annual fecundity of tautog is nearly double previously reported values.

Densities of both anchovy eggs and larvae showed significant negative trends. This year the Δ-mean density for anchovy eggs increased substantially over the 1999-2000 estimate, which was the lowest recorded. However, the Δ-mean density of larvae was lower than average and both eggs and larvae exhibited a significant declining trend. The bay anchovy appears to be experiencing a regional decline in abundance. A sharp drop in abundance was measured over the past decade in Narragansett Bay and populations declined dramatically in Chesapeake Bay after 1993. The bay anchovy is an important forage species for striped bass and recent increases in striped bass abundance along the Atlantic Coast may have contributed to the reduced numbers of bay anchovies.

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APPENDICES

APPENDIX I. List of fishes collected in the Fish Ecology sampling programs in the vicinity of the Millstone Power Station, Waterford, CT from January 1976 through December 2002.

Scientific name	Common name	Trawl[a]	Seine[a]	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	•		•
Chilomycterus schoepfi	striped burrfish	•		
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	•		
Limanda ferruginea	yellowtail flounder	•		•
Liparis spp.	seasnail	•		•
Lophius americanus	goosefish	•		•
Lucania parva	rainwater killifish	•	•	
Lutjanus apodus	schoolmaster		•	
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	•		•

Scientific name	Common name	Trawl	Seine	Ichthyoplankton
Monacanthus hispidus	planehead filefish	*		
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	*	*	*
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	*	*	*
Pseudopleuronectes americanus	winter flounder	*	*	*
Pungitius pungitius	ninespine stickleback	*	*	*
Raja eglanteria	clearnose skate	*		
Raja erinacea	little skate	*		
Raja ocellata	winter skate	*		
Salmo trutta	brown trout	*		
Sciaenidae	drums			*
Scophthalmus aquosus	windowpane	*	*	*
Scomber scombrus	Atlantic mackerel	*		*
Scyliorhinus retifer	chain dogfish	*		
Selar crumenopthalmus	bigeye scad	*		
Selene setapinnis	Atlantic moonfish	*		
Selene vomer	lookdown	*	*	
Synodus foetens	inshore lizardfish	*		
Sphyraena borealis	northern sennet	*		
Sphoeroides maculatus	northern puffer	*	*	*
Squalus acanthias	spiny dogfish	*		
Stenotomus chrysops	scup	*		*
Strongylura marina	Atlantic needlefish		*	
Syngnathus fuscus	northern pipefish	*	*	*
Tautogolabrus adspersus	cunner	*	*	*
Tautoga onitis	tautog	*	*	*
Trachinotus falcatus	permit	*	*	
Trachurus lathami	rough scad	*		
Trachinocephalus myops	snakefish	*		
Trinectes maculatus	hogchoker	*		
Ulvaria subbifurcata	radiated shanny	*		*
Upeneus parvus	dwarf goatfish	*		
Urophycis chuss	red hake	*		
Urophycis regia	spotted hake	*		
Urophycis tenuis	white hake	*		
Urophycis spp.	hake	*	*	*

[a] Includes species collected at other stations near Millstone Power Station that are no longer sampled.

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 2002.

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	24	24	24	24	24	24	33	42	54	36	36	60	60	60	60	63	45	45	42	42	45	42	42	48	48	48	1095
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	6,196	7,999	3,629	3,990	4,191	4,228	139,536
Fundulus spp.	1,634	714	706	472	515	308	640	667	1,312	759	80	364	2,294	655	831	1,152	310	300	1,635	609	117	117	56	139	63	495	16,944
B. tyrannus	0	0	0	0	0	0	1	1	0	0	0	3	2	25	510	171	2	47	14	881	23	381	2,341	1,793	1,193	24	7,417
A. quadracus	463	592	257	264	48	93	88	1,827	167	104	296	92	144	301	125	1,076	55	36	29	9	48	13	23	7	27	17	6,201
C. variegatus	42	284	35	16	7	33	133	27	25	23	1	2	10	2	14	1,169	8	0	55	11	2	3	0	4	1	0	1,907
P. saltatrix	1	0	1	1	0	1	135	0	2	0	0	1	1	3	797	2	0	10	4	3	1	2	1	9	6	5	986
P. pungitius	2	0	6	1	3	2	3	295	7	3	8	2	12	5	4	4	0	0	0	0	0	0	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	1	0	0	1	0	0	280
S. fuscus	1	1	6	1	1	2	12	9	9	5	1	8	12	5	8	8	5	2	13	12	7	39	32	12	34	17	262
Gadidae	0	0	9	0	20	12	11	6	0	2	3	0	1	1	0	0	1	0	1	0	6	14	0	12	10	2	111
M. cephalus	0	0	3	1	41	1	4	4	1	0	0	4	39	0	0	1	0	1	0	0	0	0	1	0	0		100
M. curema	0	0	0	0	0	0	0	1	9	0	0	0	42	1	9	1	0	1	30	0	1	1	1	0	0	1	98
C. harengus	0	0	0	0	0	0	2	0	0	0	30	0	6	1	0	0	0	0	0	0	5	5	22	3	1		75
L. parva	1	0	0	0	0	0	0	2	0	1	0	15	10	2	0	32	3	0	0	1	0	1	1	0	1	1	71
P. americanus	1	1	0	0	6	0	0	2	4	7	4	0	0	3	10	3	2	0	3	0	5	0	10	2	2	6	71
T. onitis	0	0	0	0	0	0	4	0	0	0	0	0	0	7	1	0	0	4	4	0	1	2	11	11	15		60
G. wheatlandi	0	0	0	0	0	0	4	2	6	11	2	6	1	3	0	1	0	2	0	0	3	0	0	0	0	0	41
T. falcatus	0	0	1	0	1	0	0	0	0	0	0	0	22	6	0	0	0	0	2	0	0	1	0	0	0		33
A. rostrata	9	4	12	2	2	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		2	33
M. menidia	0	0	1	0	0	1	0	0	0	2	2	0	1	0	0	0	1	2	3	1	0	2	0	1	9	1	27
C. hippos	0	0	1	0	0	1	0	0	0	0	0	0	3	1	3	0	1	1	2	0	0	9	0	1	0	0	23
Anchoa spp.	0	0	0	0	2	0	7	2	0	0	0	0	0	3	2	0	1	0	2	0	0	1	1	0	0		21
O. mordax	0	0	0	0	0	0	0	0	0	0	0	0	18	0	0	0	0	0	0	2	0	0	0	0	0		20
T. adspersus	0	0	2	0	0	0	3	0	0	0	0	0	0	1	0	0	0	0	0	0	5	0	3	0			14
A. pseudoharengus	0	0	0	0	0	0	0	0	5	0	0	0	3	0	0	0	0	0	2	0	0	0	1	0	0		11
L. apodus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0		7
A. americanus	0	0	0	1	0	0	1	0	0	0	0	1	0	0	1	0	0	1	0	0	0	0	0	1	0		6
S. marina	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	2	0	1	0	0	0	0	0		5
A. aestivalis	0	1	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0	0		4
S. ronner	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0	1	0	0	0	0		4
M-n. saxatilis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	1	0	0		3
C. regalis	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0		2
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0		1
P. gunnellus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Prionotus spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
L. xanthurus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
TOTAL	39,426	17,700	1,559	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	6,414	8,592	6,108	6,013	5,555	4,815	174,734

[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 2002.

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	1,878
Taxon[a]																											
S. chrysops	693	1,465	388	425	903	800	1,436	1,843	1,264	391	1,528	1,315	996	747	1,369	7,903	3,749	277	1,843	282	67	514	2,042	749	1,628	6,251	40,873
P. americanus	924	1,333	1,165	2,078	2,143	1,908	2,332	2,182	1,452	1,807	1,499	1,844	2,410	1,259	1,531	1,008	1,621	1,532	2,160	1,078	995	827	836	801	444	245	37,419
S. aquosus	203	294	107	176	150	161	216	294	199	312	322	761	591	682	357	308	616	1,000	845	558	486	230	407	416	264	125	10,080
Raja spp.	97	90	44	91	144	112	191	358	136	312	344	385	473	448	469	543	299	484	401	556	343	312	312	405	238	207	7,794
Menidia spp.	287	718	805	728	190	146	75	115	80	144	128	53	60	26	24	261	1,915	107	112	82	107	57	705	193	145	92	7,355
T. adspersus	632	666	227	1,022	596	342	207	76	68	27	9	9	12	35	31	17	90	8	31	21	8	24	46	17	49	73	4,343
Anchoa spp.	165	58	806	0	44	354	1	20	13	1,799	95	41	11	1	3	16	2	2	3	12	0	5	183	1	7	15	3,657
C. striata	8	2	0	3	5	39	13	24	25	43	241	8	32	46	49	35	47	5	277	34	7	22	478	248	321	909	2,921
M. oceanicus	45	87	56	72	162	176	208	242	76	59	126	85	111	84	62	47	122	90	147	31	113	141	68	45	137	69	2,661
Gadidae	18	69	63	62	423	315	140	194	94	135	35	373	21	20	17	24	36	23	26	50	23	26	25	19	7	6	2,244
P. dentatus	75	40	16	9	24	49	37	53	80	39	107	121	61	10	63	75	115	108	94	69	150	112	123	195	195	219	2,239
Prionotus spp.	42	30	30	46	66	72	31	67	38	31	104	27	36	215	76	25	19	42	98	136	10	14	46	109	13	57	1,480
E. microstomus	6	0	0	0	1	17	4	15	14	34	107	39	59	12	85	82	86	31	96	36	33	77	178	227	111	118	1,468
P. triacanthus	5	2	12	3	4	9	4	7	0	3	10	5	929	10	328	60	24	16	1	7	0	0	6	1	0	0	1,446
Urophycis spp.	2	5	7	5	21	23	182	45	19	29	11	26	49	25	59	13	43	59	93	44	24	68	58	376	39	117	1,442
M. bilinearis	101	15	2	36	109	48	38	52	26	38	44	4	23	51	47	47	73	5	31	23	7	16	69	34	35	33	1,007
T. onitis	63	70	86	68	47	27	50	41	46	47	23	17	42	18	16	14	9	24	17	28	19	24	15	26	55	97	989
P. gunnellus	54	28	11	7	35	25	36	27	23	12	21	15	24	22	6	7	12	12	27	14	31	29	17	13	61	12	642
H. americanus	7	5	11	19	62	96	115	60	16	7	1	1	0	2	2	1	0	0	0	5	3	3	7	2	0	0	425
O. mordax	1	6	14	0	9	19	29	2	4	16	8	4	4	4	16	6	134	1	12	7	20	2	35	2	1	2	358
S. fuscus	0	14	9	12	24	18	12	12	25	11	7	15	13	7	5	9	10	11	8	7	8	16	5	6	10	6	280
A. pseudoharengus	1	216	8	3	0	4	0	1	0	3	4	1	1	0	0	2	7	17	3	1	0	0	0	0	0	0	272
C. lumpus	8	1	6	16	4	0	4	0	4	0	0	7	2	0	4	0	1	0	0	0	7	1	0	0	117	2	184
S. maculatus	2	1	0	0	1	0	0	2	0	0	1	0	4	3	8	22	16	1	7	5	4	1	9	19	15	5	126
Clupeidae	1	0	0	0	0	0	0	0	0	110	0	0	0	0	0	0	0	0	2	0	0	0	0	0	2	0	115
B. tyrannus	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	5	24	0	3	0	2	0	2	5	0	13	57
C. regalis	1	8	0	1	0	3	1	0	0	1	4	2	10	0	1	1	1	0	1	2	1	0	8	3	0	8	57
P. oblongus	1	0	1	0	3	1	1	2	5	3	7	1	2	0	0	0	1	2	1	1	0	4	5	2	2	0	45
M. americana	2	7	1	0	6	1	0	0	0	0	0	0	1	2	0	2	0	1	1	1	1	0	0	1	8	0	35
H. erectus	0	0	0	0	0	0	0	0	0	2	1	0	1	0	2	6	0	0	0	4	0	4	8	4	0	0	32
Liparis spp.	1	3	4	2	2	2	5	1	3	0	0	0	4	0	0	0	0	0	0	0	2	0	1	0	0	0	30
A. sapidissima	15	2	0	0	4	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	0	25
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	7	1	0	0	0	3	5	0	2	0	0	20
M. hispidus	2	0	2	0	0	0	1	0	3	1	0	0	1	2	2	0	0	3	1	0	0	1	0	0	0	0	19
C. harengus	0	1	0	0	0	0	0	2	1	0	0	0	0	0	1	0	1	0	0	0	0	7	0	2	0	3	18
D. volitans	1	0	0	0	0	0	0	1	0	0	1	4	1	2	3	0	1	0	0	2	0	0	1	0	1		18
G. aculeatus	1	0	1	1	0	2	1	1	1	0	0	0	0	0	0	1	2	1	0	1	3	1	0	1	0	0	18
O. tan	1	0	0	1	4	1	1	0	2	0	1	1	0	0	0	0	0	1	0	0	0	0	1	3	0		17
A. aestivalis	0	0	1	0	0	0	0	9	0	0	2	1	0	0	0	0	1	0	0	0	0	1	1	0	0	0	16
M. octodecemspinosus	0	0	2	2	1	6	3	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16
M. saxatilis	0	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	5	1	2	1	1	3	0	0	0	16
F. tubercularia	1	1	0	0	1	0	0	0	0	0	0	0	0	1	3	1	0	1	0	0	0	1	0	3	0	0	13
M. aeglefinus	0	0	0	0	0	0	2	0	0	0	1	0	3	0	0	0	0	5	0	0	0	0	0	0	0		11
A. schoepfi	0	0	1	0	0	0	0	0	1	1	0	0	0	2	1	0	1	0	0	1	0	1	0	1	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	0	0	0	0	0	0	9

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	2,112
Taxon[a]																											
A. rostrata	1	0	1	0	0	0	1	0	2	1	0	1	0	0	0	1	0	0	1	0	0	0	0	0	0	0	9
M. canis	0	0	1	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	9
P. cruentatus	0	0	0	0	0	0	0	0	3	0	0	0	2	2	1	0	0	0	0	0	0	0	0	0	0	0	8
C. hippos	0	0	0	0	0	0	1	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0	2	0	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	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	0	0	0	0	0	1	6
S. setapinnis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	6
S. foetens	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	2	5
A. quadracus	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	3
C. ocellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	1	0	0	0	3
Gobiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	1	0	0	0	3
L. americanus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	1	0	3
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	1	0	0	0	0	3
P. alta	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	3
C. crysos	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
C. schoepfi	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	2
E. cimbrius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
Lactophrys spp.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	2
M. aureus	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
S. vomer	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	2
U. subbifurcata	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	2
C. oceanicus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	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	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	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	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	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	0	0	0	0	0	0	1
TOTAL	3,522	5,272	3,900	4,896	5,224	4,804	5,416	5,779	3,750	5,437	4,816	5,184	6,026	3,764	4,653	10,558	9,092	3,888	6,377	3,116	2,509	2,578	5,716	3,952	3,970	8,696	124,260

[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 2001.

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	77	78	81	78	78	78	77	78	78	78	2,032
Taxon[a]																											
P. americanus	1,422	991	1,262	1,039	885	385	1,665	1,302	795	1,064	781	617	660	648	551	540	640	656	598	564	432	362	387	432	381	180	19,239
S. chrysops	441	250	162	214	65	12	713	374	26	332	426	196	77	205	608	4,301	628	17	384	132	11	323	445	598	1,205	2,038	14,183
Menidia spp.	479	183	261	446	202	22	73	251	152	161	1,491	232	45	50	23	238	220	91	35	96	14	30	86	191	763	276	6,111
T. adspersus	97	78	90	232	191	263	209	120	73	23	28	15	148	34	55	88	162	50	61	46	6	37	854	512	394	240	4,106
P. gunnellus	20	55	45	37	171	189	137	55	55	116	78	107	278	40	42	78	48	104	220	174	65	160	515	406	410	244	3,849
Anchoa spp.	285	9	32	4	6	6	4	24	25	326	283	164	256	492	15	1,407	4	1	1	16	0	1	62	0	29	29	3,481
Raja spp.	63	64	42	19	45	25	66	239	41	109	84	111	130	108	334	191	196	238	102	209	118	157	104	109	49	88	3,041
Gadidae	38	116	68	75	420	341	171	131	134	186	97	88	16	17	61	58	67	331	52	69	27	36	39	92	82	56	2,868
S. aquosus	135	110	87	108	65	30	95	199	108	155	107	65	74	98	82	64	119	148	77	137	145	78	53	41	23	7	2,410
M. octuxus	33	115	96	46	70	113	98	133	82	76	57	97	159	26	27	45	102	111	137	59	55	90	147	66	153	83	2,276
P. dentatus	80	35	19	16	9	9	65	57	149	53	157	85	35	10	68	75	110	33	33	40	104	58	51	103	124	64	1,642
O. mordax	47	164	51	0	72	11	2	21	216	274	224	227	89	5	8	3	7	1	0	6	1	3	1	8	0	0	1,441
T. onitis	71	106	59	57	22	20	37	18	15	31	57	30	36	20	40	35	91	50	20	74	46	30	79	133	262	166	1,605
S. fuscus	7	13	15	37	27	39	65	84	124	50	57	72	136	12	18	21	67	77	27	92	28	24	58	76	65	32	1,323
Urophycis spp.	8	22	19	17	9	21	81	45	42	37	5	14	23	18	101	15	53	37	87	36	17	19	19	199	35	57	1,036
C. striata	3	0	0	0	0	6	1	6	1	4	45	0	1	3	5	10	9	2	64	4	1	16	443	135	120	149	1,028
B. tyrannus	0	0	0	0	0	0	0	0	0	0	0	1	0	520	3	3	5	4	0	0	2	0	4	3	4	3	552
E. microstomus	11	0	0	1	1	3	5	8	8	22	35	14	4	4	54	23	54	25	29	41	8	32	17	60	16	13	488
H. americanus	2	8	3	36	49	82	145	80	28	8	1	1	0	0	1	0	0	0	0	0	0	0	1	1	0	0	446
A. quadracus	1	3	1	0	3	1	0	5	21	13	85	23	6	0	0	0	19	87	1	2	1	0	0	0	1	0	273
M. bilinearis	65	2	1	4	41	4	9	4	4	5	3	0	0	4	3	9	41	1	3	3	0	5	1	1	10	1	224
C. lumpus	11	10	16	39	4	0	7	0	18	1	1	31	4	1	2	1	15	7	1	2	39	4	1	0	7	0	222
Prionotus spp.	23	7	5	5	4	3	7	1	9	3	15	0	4	7	5	8	12	13	2	3	2	4	7	9	0	3	161
H. erectus	0	0	0	0	0	0	0	0	0	3	1	8	8	3	1	6	20	3	0	0	5	0	1	8	17	2	87
P. triacanthus	1	1	2	0	4	0	0	11	0	0	5	1	26	4	14	3	1	0	0	0	0	0	0	1	3	1	78
C. harengus	0	0	0	0	0	0	0	0	0	63	0	2	0	0	1	3	0	1	0	0	0	0	2	0	2	0	74
F. tabacaria	1	1	0	0	2	0	1	0	8	1	2	0	0	1	11	5	5	2	1	0	0	0	0	8	19	1	69
A. rostrata	1	3	0	1	1	8	9	3	5	4	0	1	1	1	0	5	0	0	0	0	0	0	0	0	1	0	44
C. regalis	1	11	1	0	0	5	0	0	0	0	2	0	1	1	0	0	0	0	1	0	0	0	0	7	11	0	41
S. maculatus	3	2	1	0	0	0	0	1	3	1	0	0	0	2	3	3	0	0	0	5	1	2	2	1	9	1	40
A. americanus	3	2	6	0	6	1	0	0	1	0	0	0	0	0	0	0	1	0	0	1	0	1	0	14	0	0	36
Liparis spp.	1	1	1	2	1	3	3	1	5	0	1	1	2	4	0	0	0	0	1	0	4	1	1	0	0	0	33
M. hispidus	0	1	2	0	0	0	6	0	2	4	1	0	0	1	2	1	0	1	0	0	0	0	0	0	1	0	22
S. scorpinus	0	0	0	0	0	0	0	0	0	0	0	0	16	0	0	0	0	0	0	0	0	0	0	0	1	0	17
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	1	1	0	0	0	0	0	0	1	10	0	16
A. pseudoharengus	1	3	0	0	0	0	0	3	0	0	0	0	0	1	2	0	0	0	0	0	0	0	1	0	0	0	11
A. schoepfi	0	1	0	1	1	0	0	1	0	1	1	0	2	1	1	0	0	0	0	0	0	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	0	2	0	0	0	0	0	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	0	0	0	0	0	1	11
P. pungitius	0	0	0	0	0	0	0	0	5	0	5	0	0	0	0	0	0	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	0	0	0	0	1	0	10
Lactophrys spp.	0	0	0	0	0	0	0	0	2	0	0	0	0	0	1	3	0	1	0	0	0	0	1	1	0	0	9
A. sapidissima	1	0	0	0	0	1	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	77	78	81	78	78	78	77	78	78	78	2,032
Taxon[a]																											
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	4	0	0	1	0	0	0	0	0	0	0	8
G. wheatlandi	0	0	0	0	0	1	1	1	0	1	2	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	8
S. borealis	0	0	0	0	0	0	0	1	1	2	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	6
P. saltatrix	1	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	3	0	0	6
T. maculatus	3	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	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	0	0	1	0	0	0	5
L. xanthurus	1	1	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	4
T. lathami	0	0	0	1	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	4
U. subbifurcata	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	4
M. saxatilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	1	1	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	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	0	0	0	0	0	0	3
D. volitans	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	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	0	0	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	0	0	0	0	0	0	2
P. cruentatus	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	2
S. vomer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	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	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	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	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	0	0	0	0	0	0	1
D. centroura	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	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	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	0	0	0	0	0	0	1
L. americanus	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	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	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	0	0	0	0	0	0	1
M. muratus	0	0	1	0	0	0	0	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	1	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	0	0	0	0	0	0	0	0	1
U. parvus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
P. alta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	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	1,132	1,483	3,387	3,223	4,217	3,736	74,944

[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 2002.

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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	2,034
Taxon[a]																											
P. americana	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	1,467	585	1,030	1,252	1,360	298	77,020
Ammidia spp.	1,203	315	58	121	73	159	315	49	15	93	1,549	318	639	220	317	431	481	75	97	34	352	124	349	359	924	270	8,940
M. aenaeus	57	30	59	161	201	349	587	232	274	174	490	205	631	374	492	273	451	142	299	167	442	375	278	96	122	48	7,009
S. aquosus	76	40	32	134	158	165	230	225	244	136	302	256	173	342	363	234	225	292	94	97	77	110	64	54	39	8	4,150
P. dentatus	41	29	16	14	21	100	82	63	163	75	226	214	191	20	149	173	248	105	165	63	75	228	75	180	235	116	3,072
S. fuscus	20	11	13	31	61	178	132	80	70	117	114	177	128	55	112	90	69	182	87	88	64	29	49	51	20	27	2,055
Prionotus spp.	97	21	2	6	4	63	200	2	2	3	44	1	19	147	98	267	37	12	262	53	6	40	20	465	70	2	1,943
T. onitis	39	16	30	45	25	129	90	16	11	22	110	15	57	28	105	51	24	13	14	128	41	40	126	240	324	126	1,865
G. oculatus	16	12	44	22	186	99	58	7	162	86	8	19	63	8	14	410	135	20	60	3	115	54	79	4	6	1	1,691
A. quadracus	8	3	22	27	190	764	76	6	91	115	22	29	25	11	18	100	50	3	4	4	13	1	46	8	7	6	1,649
Anchoa spp.	10	195	0	2	0	0	11	2	5	16	8	11	168	734	2	12	1	0	12	2	14	3	21	139	0	35	1,403
B. tyrannus	0	13	1	0	1	1	0	1	0	31	10	3	0	788	2	182	23	17	41	0	3	0	2	106	102	19	1,346
Gadidae	5	9	11	8	152	185	71	43	14	209	12	5	32	29	29	12	8	16	20	44	86	66	124	15	11	12	1,228
S. chrysops	58	10	11	5	6	38	27	57	2	3	22	15	3	5	10	175	119	0	83	38	2	12	4	94	263	166	1,228
P. gunnellus	0	1	10	4	14	29	69	32	21	10	18	41	47	58	83	30	48	44	26	26	54	33	142	29	20	25	914
O. tau	96	21	7	17	27	34	24	21	22	31	53	50	56	29	55	17	5	9	18	11	6	8	7	5	8	6	643
T. adspersus	14	4	7	11	7	91	58	60	16	15	38	14	9	15	9	8	10	9	13	8	3	20	113	27	23	70	672
C. striata	15	0	0	1	2	3	2	0	1	13	107	0	3	2	20	39	1	0	9	6	1	2	13	39	58	4	344
O. mordax	55	86	2	0	6	13	30	1	3	37	4	5	34	4	7	3	10	2	1	2	5	0	3	0	0	1	314
A. rostrata	16	11	7	4	8	24	14	21	15	27	26	20	14	4	15	2	2	4	10	7	8	2	5	7	3	6	282
Clupeidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	258	0	0	0	0	0	0	259
S. maculatus	11	5	0	0	7	11	12	7	2	1	1	0	2	4	13	22	3	2	17	15	1	3	1	28	8	1	177
Urophycis spp.	0	0	0	0	1	2	7	4	3	10	1	22	11	6	9	4	2	5	18	16	2	9	17	0	22	6	177
Alosa spp.	0	0	0	0	0	0	0	0	0	0	0	1	0	3	14	11	2	16	111	0	0	0	3	3	0	0	164
Gobiidae	3	0	0	0	4	0	0	2	9	6	2	5	10	2	19	15	1	25	1	3	1	4	3	0	3	11	129
H. erectus	0	0	0	0	0	0	0	1	1	1	10	3	2	2	8	34	1	0	0	0	0	0	6	15	22	0	106
A. americana	1	0	75	1	0	2	14	1	0	0	0	0	1	0	1	0	0	0	0	0	0	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	1	0	1	0	0	0	86
Raja spp.	0	0	0	0	2	1	1	3	0	2	0	0	4	2	4	0	20	2	6	2	9	18	2	4	1	0	83
A. pseudoharengus	0	47	0	2	2	2	3	1	2	1	3	0	1	0	3	0	0	0	1	0	1	0	0	0	0	1	70
E. microstomus	0	0	0	0	1	0	0	3	0	1	4	3	4	1	3	7	8	1	5	1	1	6	3	0	10	1	63
A. xyidissima	2	0	0	4	1	1	0	9	0	0	0	0	0	0	0	0	0	0	0	0	40	0	0	1	0	0	58
A. aestivalis	0	7	1	9	0	0	0	1	2	0	0	0	0	0	9	0	0	1	7	0	0	0	0	0	0	0	37
M. americana	3	0	1	5	1	1	0	0	0	0	0	0	2	9	0	0	1	2	0	0	0	0	0	0	1	0	26
D. volitans	2	0	0	0	0	1	3	0	0	1	1	0	0	0	3	2	2	1	1	4	1	0	0	3	0		25
F. tabacaria	0	1	0	0	0	0	0	0	0	0	0	0	1	1	7	3	0	1	0	0	0	0	3	4	0		21
Morone saxatilis	0	0	2	1	0	1	1	0	0	1	0	0	0	4	1	0	0	2	2	0	3	1	0	1	0	0	20
P. triacanthus	1	0	0	0	0	0	0	2	0	0	0	0	0	8	0	0	2	0	0	0	0	0	1	0	6	0	20
M. bilinearis	0	0	0	0	0	0	3	0	0	0	0	1	1	1	0	0	4	0	0	4	0	1	0	2	0	0	17
Liparis spp.	0	0	0	0	0	1	0	0	0	0	0	0	0	0	9	0	1	0	1	1	0	0	1	0	0	1	16
C. lumpus	0	0	1	1	0	0	1	0	1	0	0	2	0	0	0	2	1	1	0	0	1	1	0	0	2	0	14
Fundulus spp.	0	0	0	0	0	5	2	0	0	1	1	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	12

APPENDIX V. (continued).

Taxon[a]	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	96-97	97-98	98-99	99-00	00-01	01-02	Total
Number of samples	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	78	78	81	78	78	78	78	78	78	78	78	2,034
P. oblongus	0	0	0	2	0	0	0	0	0	0	0	0	2	7	0	1	0	0	0	0	0	0	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	0	0	0	0	0	0	11
O. marginatum	0	0	0	0	0	0	0	0	0	0	0	3	0	0	3	0	0	0	0	0	0	0	0	4	1	0	11
P. saltatrix	0	1	0	0	0	0	2	0	0	0	0	1	0	0	1	0	0	0	1	0	0	0	0	0	0	0	6
S. borealis	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	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	2	0	0	1	0	0	0	0	6
C. harengus	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	8		13
C. oceanicus	0	0	0	0	0	0	0	0	1	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	4
S. foetens	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	4
M. hispidus	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	3
P. pungitius	0	0	0	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
G. wheatlandi	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	1	0	3
M. cephalus	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	1	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	0	0	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	0	0	0	0	0	0	2
P. ferrugineus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
C. regalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0		2
T. maculatus	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	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	0	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	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	0	0	0	0	0	0	1
M. aenis	0	0	1	0	0	0	0	0	0	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	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	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	1	0	0	0	0	0	0	0	1
S. trutta	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
S. retifer	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
X. setapinis	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
TOTAL	4,708	2,159	1,637	3,445	5,111	6,360	9,797	4,609	6,133	3,794	7,375	4,962	9,134	6,633	6,572	7,619	7,288	2,486	3,747	2,044	2,976	1,855	2,666	3,309	3,761	1,353	121,583

[a] Fish identified to lowest practical taxon.

Lobster Studies

Lobster Studies

Introduction

The American lobster, *Homarus americanus*, occurs in the Northwest Atlantic from Canada to Cape Hatteras and supports one of the most valuable commercial fisheries in the northeast United States ($64.8 to $329.5 million from 1978 to 2001; NMFS landings data). For many years, fishery managers have been concerned that the intense exploitation of lobster throughout its range may impact long-term egg production and recruitment to coastal populations (Anthony and Caddy 1980; NFSC 1996; ASMFC 1997). As a result, a number of regulations have been implemented by state and federal agencies to minimize the risk of a population collapse due to recruitment failure by improving larval production and subsequent recruitment. Management measures include prohibition on possession of egg-bearing (or v-notched) females, prohibition on spearing lobster, minimum legal size of 82.6 mm (3 $1/4$ in carapace length), catch limit on lobster taken in trawl gear, and prohibition on possession of lobster parts. Additional measures implemented to control the coastwide escalation of fishing effort by regulating lobster traps used in the fishery include requirements for escape vents and biodegradable fasteners, maximum trap size restrictions, and limits on the number of traps used by fishermen.

The most recent stock assessment for American lobster indicated that instantaneous fishing mortality rates (F) in Long Island Sound (LIS) have exceeded 1.0 for the past decade and the number of traps used in the fishery has increased eightfold since the early 1980s (ASMFC 2000). Coincidental with the increase in fishing pressure, commercial landings of lobsters in Connecticut waters of LIS increased from 0.8 million pounds in 1979 to a record 3.7 million pounds in 1998 (CTDEP fishery statistics). However, landings declined 65% between 1998 and 2001 to only 1.3 million pounds. Landings declined even further by 2002 when 1.1 million pounds were harvested. The decline has been confirmed with fishery independent data collected in the CTDEP trawl survey which indicates that lobster abundance has fallen almost 60% since 1998 (Simpson et al. 2001). The recent decline in lobster abundance has been attributed to a significant lobster die-off in western LIS (LoBue 2001). Beginning in the fall of 1998 and throughout the summer of 1999, an unexplained highly invasive disease emerged in lobsters harvested from western LIS (CTDEP 2000; LoBue and Gottschall 2000).

Lobstermen reported that more than half of the lobsters caught in traps were dead or died before arriving at port. Catch data compiled by CTDEP confirmed these observations, noting that commercial landings for all ports declined dramatically in the fall of 1999 (64-99%) from the 1995-1998 average. Catch per trap haul in the commercial fishery during the fall 1999 declined 83% in western LIS and 39% in central LIS when compared to average CPUE during the period 1995-1998 (CTDEP 2000). At the same time, a widespread outbreak of shell disease syndrome was reported for lobsters in eastern LIS (DNC 2001a; Landers et al. 2001a) and other coastal areas of southern New England (Castro and Angell 2000). In January 2000, the U.S. Secretary of Commerce declared a failure of the commercial lobster fishery in LIS. The Long Island Sound Lobster Initiative (LISLI), an endeavor of Sea Grant programs in New York and Connecticut along with CTDEP and NMFS, was formed after a July 2001 Congressional allocation of $6.6 million in Federal funding for research investigations on causes and economic impacts of the die-off and shell disease in LIS lobsters. Funded researchers are currently investigating many possible factors on an ecosystem-wide basis. These environmental, physiological, and biological stresses include: water quality conditions including elevated temperature and changes in salinity, environmental conditions such as storm events, pollution, lobster crowding, disease-causing organisms, and pesticides. In summer 2002, researchers at the Marine Disease Pathology and Research Consortium Laboratory at Stony Brook University investigated new reports of lobster die-offs in central and western LIS. Preliminary results link the latest mortalities to an accumulation of calcium in lobster tissue (calcinosis) mediated by prevailing high bottom water temperature in LIS (NYSG 2002; Dove et al. 2003). At present, the status of the lobster population in LIS is uncertain and the long-term sustainability of the resource may be threatened.

Because of the regional economic importance of American lobster, the local population in the vicinity of Millstone Power Station (MPS) has been studied extensively since 1978 to determine if power plant operation 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 larvae through the cooling water systems, impingement of juveniles and adults on the intake traveling screens, and effects of the heated effluent on juvenile and adult lobsters in the discharge area. The objectives of the
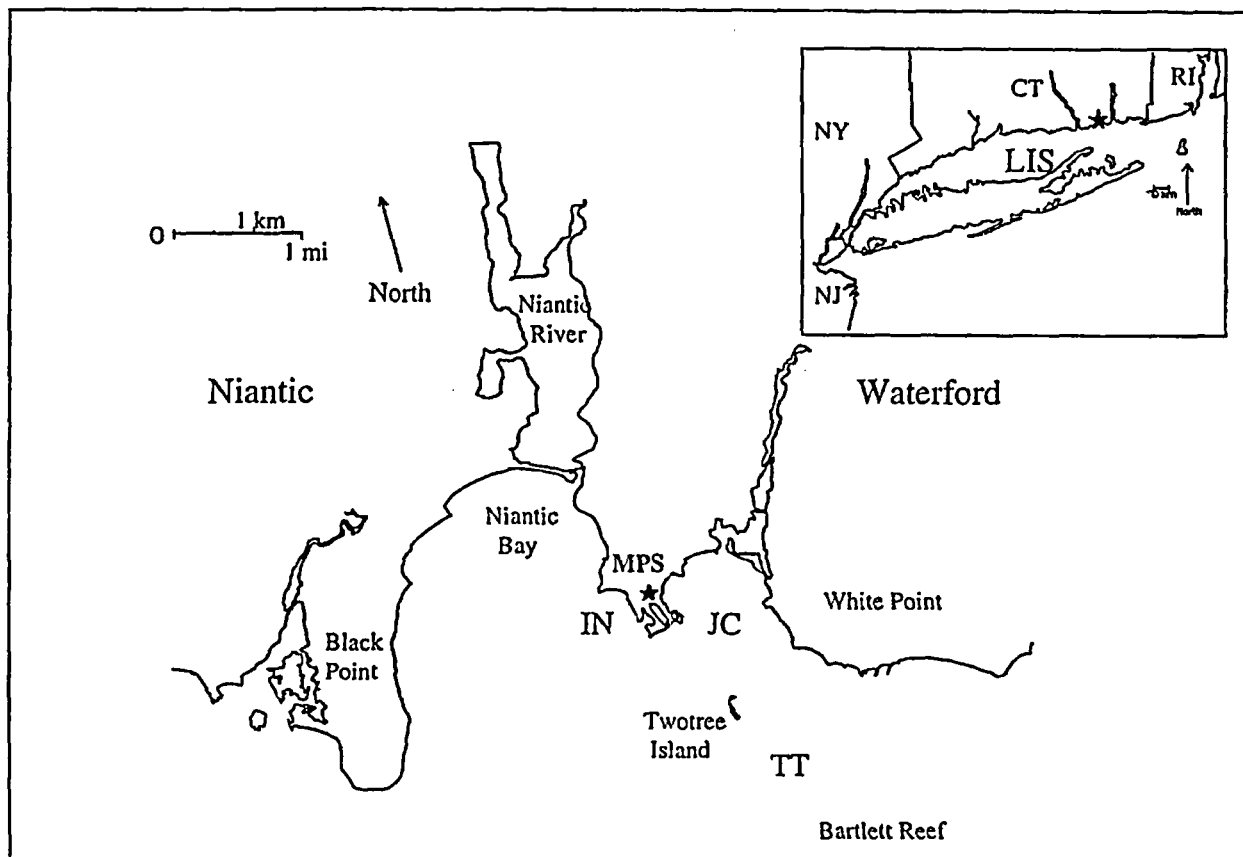
Fig. 1. Location of the Millstone Power Station (MPS), and the three lobster sampling stations, JC=Jordan Cove, IN=Intake, TT=Twotree.

lobster monitoring program at Millstone are to: 1) evaluate year-to-year, seasonal, and among-station changes in catch-per-unit-effort; 2) monitor lobster population demographics including size structure, growth rates, sex ratios, female size at maturity, characteristics of egg-bearing females; and 3) monitor movements of lobsters in the vicinity of MPS. Beginning in 1984, studies were conducted during the hatching season to estimate the number of lobster larvae entrained through the cooling water systems. Impacts associated with recent plant operations on the local lobster population were assessed by comparing results of the 2002 study with those from 1978 to 2001. Emphasis has been placed on assessing long-term trends in the abundance and population characteristics of lobsters collected in the Millstone Point area. These results were compared, when appropriate, to studies conducted by other researchers in LIS and throughout the range of the American lobster.

Materials and Methods

Lobsters were collected from May through October using pot gear. Four pot-trawls, each consisting of five double-entry vinyl-coated wire pots (76 x 51 x 30 cm; 2.5 cm^2 mesh) equally spaced along a 50-75 m line buoyed at both ends, were fished in three rocky areas in the vicinity of MPS (NUSCO 1982, 1987a). Pots set in Jordan Cove (average depth 6 m) were 500 m east of the Millstone discharge (Fig 1). 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 2000 m offshore near Twotree Island. Pots were hauled on Monday, Wednesday, and Friday of each week, weather permitting, and twice per week on holiday weeks. On each sampling trip, surface and bottom water temperatures and salinities were recorded at each station. Lobsters were banded to restrain chelipeds, transported to the laboratory, and kept in a tank supplied with a continuous flow of seawater. Pots were rebaited and reset in the same area. On Fridays, all 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). The size at which females become sexually mature was determined (since 1981) 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). 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 after examination.

Entrainment studies were conducted since 1984 by sampling lobster larvae during the period of their occurrence (May through July) at one of the discharges of Units 1, 2, or 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. The net was typically fished for 45-60 minutes and filtered about 4000 m^3 of cooling water. From 1984 to 1993, eight lobster larvae entrainment samples (four day and four night) were collected each week. Based on a statistical review of the sampling frequency in 1994, the number of samples collected was reduced to six per week (three day and three night). Samples were transported to the laboratory where they were either immediately processed or placed in a 1.0 mm mesh sieve and held in tanks supplied with flowing seawater for later processing. All samples were processed within 24 hours by sorting in a white enamel pan. Lobster larvae were examined for movement and classified as live or dead. All larvae were also classified by developmental stage (I-IV) according to the criteria established by Herrick (1909). The abundance of larvae in entrainment samples was standardized as the number of larvae per unit-volume. The mean density of larvae from May through July was estimated by 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 multiplied 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 the total number of lobsters impinged, as well as their mean size, sex ratio, proportion of culls (missing claws), and survival of impinged lobsters. Possible impacts associated with impingement of lobsters at Units 1, 2, and 3 were mitigated by installing fish return systems in the intakes, which return impinged organisms to LIS (NUSCO 1986a; 1987b; DNC 2001b). 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 MPS area. Since the 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 log-normal 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 MPS 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 the presence of significant trends in the time series of annual CPUE data, and of several other selected population characteristics (size composition, sex ratios, growth rates). 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 bottom water temperature. Molting peaks were derived using the inflection point of the Gompertz growth function (Draper and Smith 1981; Gendron 1989) fitted to data reflecting the cumulative percentage of molting lobsters at weekly intervals during the molting season. The function used to describe annual molting peaks 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 as weeks since May 1st,
   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
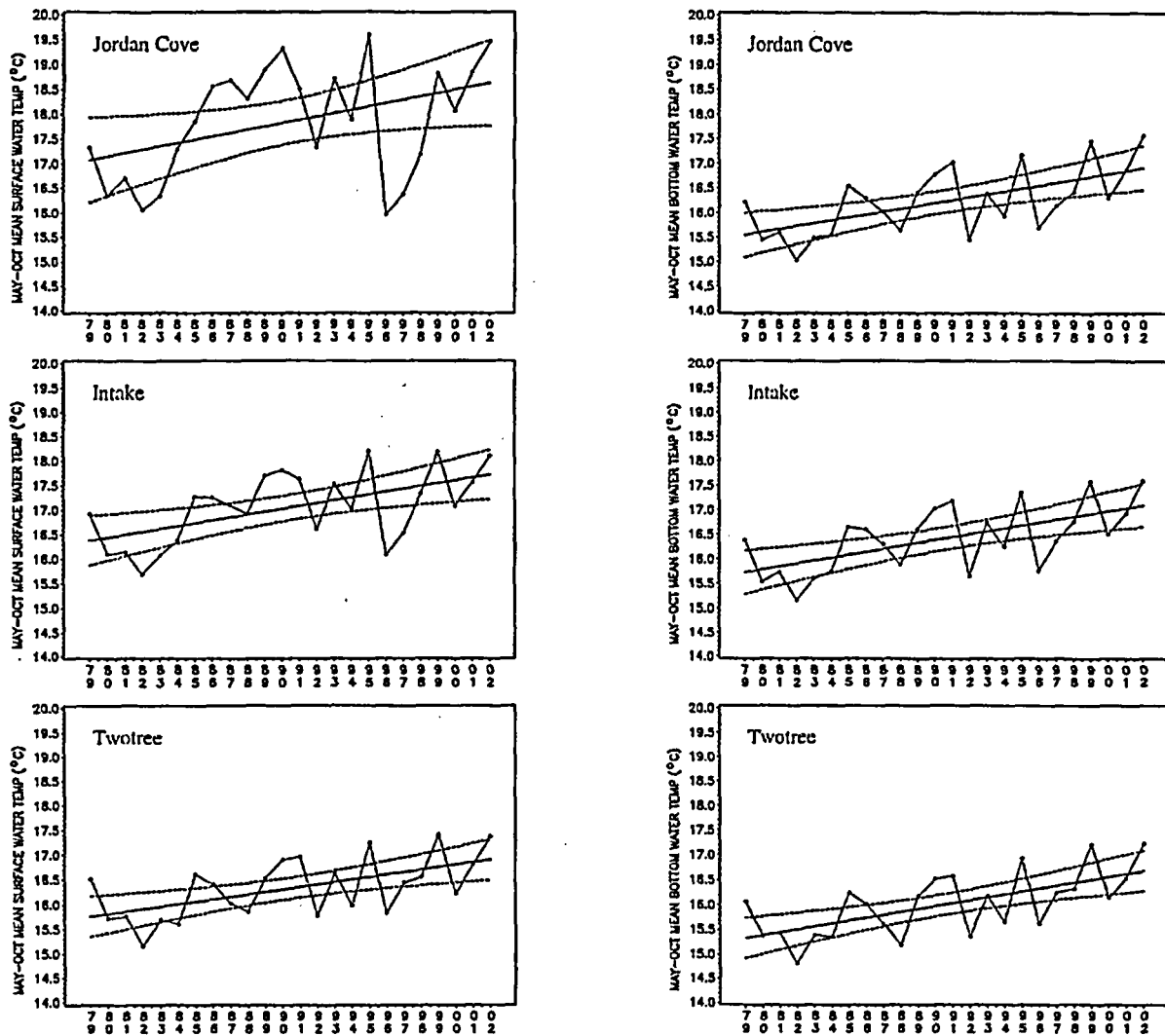
Fig. 2. Mean surface and bottom water temperatures from May through October at each of the lobster sampling stations from 1979 through 2002 and linear trends (± 95% C.I.) of surface and bottom water temperature.

possible relationship between water temperature and molting.

Results and Discussion

Water Temperature

Mean surface and bottom water temperatures measured from May through October at each station from 1979 to 2002 are presented in Fig. 2. Over the past 24 years, average surface water temperatures from May through October ranged between 16.0 and 19.6°C at Jordan Cove, 15.7°C and 18.2°C at Intake, and 15.2°C to 17.4°C at Twotree. Over the same period, the interannual variability in bottom water temperature was generally smaller than surface water temperature,

with bottom water temperatures ranging between 15.0 and 17.6°C at Jordan Cove, 15.2 and 17.6°C at Intake, and 14.8°C and 17.2°C at Twotree. Surface water temperatures were notably higher than bottom water temperatures at the Jordan Cove station, which is located closest to the MPS discharge. Surface temperatures at the Jordan Cove were also slightly warmer during the period 1986-1995 when 3-units were operating. In comparison, surface and bottom water temperatures were most similar at the Twotree station, which is not influenced by the MPS thermal plume.

Long-term trends in the May to October water temperatures were examined using SAS GLM Procedures (SAS 1990). Significant increasing trends in surface water temperatures were found at each station over the past 24 years. The rate of surface

TABLE 1. Catch statistics of lobsters caught in wire pots[a] from 1978 through 2002.

Year	Total number caught	Number pots hauled[b]	Geometric mean total CPUE	95% C.I.	Δ-mean legal CPUE[c]	95% C.I.	Commercial catch[d] (lbs x 10³)
1978	1824	1026	1.600	1.454 - 1.761	0.173	0.144 - 0.202	-
1979	3259	2051	1.404	1.302 - 1.513	0.128	0.107 - 0.148	86.0
1980	2856	2116	1.103	0.997 - 1.221	0.109	0.092 - 0.126	81.1
1981	2236	2187	0.904	0.839 - 0.974	0.098	0.083 - 0.113	131.1
1982	9109	4340	2.006	1.925 - 2.089	0.165	0.144 - 0.186	136.5
1983	6376	4285	1.331	1.250 - 1.418	0.148	0.128 - 0.168	194.7
1984	7587	4550	1.607	1.540 - 1.677	0.159	0.140 - 0.179	251.7
1985	7014	4467	1.352	1.252 - 1.460	0.105	0.090 - 0.120	179.4
1986	7211	4243	1.585	1.501 - 1.673	0.086	0.074 - 0.097	157.3
1987	7280	4233	1.633	1.562 - 1.707	0.079	0.070 - 0.089	194.1
1988	8871	4367	1.929	1.846 - 2.015	0.079	0.068 - 0.091	217.8
1989	7950	4314	1.729	1.645 - 1.817	0.065	0.056 - 0.075	213.7
1990	7106	4350	1.531	1.455 - 1.610	0.076	0.065 - 0.087	327.0
1991	7597	4404	1.542	1.437 - 1.654	0.091	0.078 - 0.104	349.5
1992	11438	4427	2.457	2.352 - 2.565	0.085	0.075 - 0.095	446.9
1993	10195	4194	2.301	2.198 - 2.408	0.080	0.069 - 0.091	358.7
1994	9849	4256	2.199	2.104 - 2.298	0.071	0.061 - 0.093	345.6
1995	6435	4317	1.261	1.152 - 1.380	0.080	0.068 - 0.093	307.2
1996	7531	4249	1.587	1.466 - 1.718	0.067	0.058 - 0.077	266.7
1997	9026	4293	1.960	1.862 - 2.064	0.099	0.086 - 0.112	409.0
1998	10991	4380	2.315	2.189 - 2.448	0.125	0.108 - 0.141	480.2
1999	11216	4057	2.560	2.419 - 2.709	0.151	0.131 - 0.170	402.3
2000	8707	4192	1.849	1.721 - 1.986	0.092	0.078 - 0.106	242.5
2001	7268	4104	1.587	1.476 - 1.705	0.094	0.080 - 0.107	210.0
2002	4829	4370	0.855	0.767 - 0.954	0.061	0.052 - 0.071	170.4

[a] 10 wire pots fished at each station from August through October 1978, and from May through October 1979-81; 20 wire pots fished at each station from May through October 1982-2002.

[b] Represents the total number of pots hauled during the May through October study period, all stations combined.

[c] 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).

[d] Annual commercial catch data from Area 1 eastern LIS (provided by CT DEP).

temperature increase was highest at Jordan Cove (slope=0.067 °C/year, p=0.041), followed by Intake (slope=0.058 °C/year, p=0.004) and Twotree (slope=0.049°C/year, p=0.003). The MPS thermal plume does not influence bottom water temperatures at each of the lobster sampling stations (NUSCO 1988c) and significant (p<0.05) increasing trends were identified in bottom water temperature at each station over the past 24 years (Jordan Cove slope=0.060°C/year; Intake and Twotree slope=0.059°C/year). Similar trends in long-term water temperature records have been reported by other researchers in New England (Keller et al. 1999; Koeller 1999; Keller and Klein-MacPhee 2000). Indications of warming trends are most evident during the winter months and appear to be partially correlated to local air temperature anomalies (Foertch 2000; Manning et al. 2001).

Abundance and Catch-per-Unit-Effort

The total number of lobsters caught each year from May through October since 1978 ranged from 1,824 to 11,438 (Table 1). The total catch was lower from 1978 through 1981 (range=1,824-3,259) because only 10 wire traps were used at each station. Beginning in 1982, 20 wire traps were used at each station and total catch fluctuated between 6,376 and 9,109 lobsters from 1982 to 1991. Catches peaked in 1992 (11,438) and then declined by nearly 50% in 1995 (6,435). For the next four years, catches increased each year and peaked again in 1999 (11,216). Total catch declined each year in 2000 and 2001 (22% and 16%, respectively) and declined to historical lowest levels (since all wire pots were introduced in 1982) in 2002, when 4,829 lobsters were caught in 4,370 pot-hauls. Catch-per-unit-effort of 0.855 lobster/pot in 2002 declined by nearly 50% from 2001 levels (1.587 lobster/pot) and represented the lowest annual mean CPUE value since the study began in 1978 (0.904 and 2.560 lobster/pot). Despite the precipitous and somewhat alarming declines in

lobster abundance over the past three years, no significant trend was identified in the time series of total CPUE since 1978 (slope=0.026, p=0.08). The recent outbreak of shell disease in southern New England waters and the catastrophic die-off and decline in lobster abundance observed in western LIS may be responsible for the lower catches observed in the MPS area during 2002.

The catch of legal-size lobsters is highly dependent on the number of recruit-size lobsters one or two molts below minimum legal size. Since the majority of lobsters caught in our unvented traps are recruit-sized, annual fluctuations in total lobster CPUE correspond closely to changes observed in legal-size lobster abundance. For example, high total CPUEs in 1998 and 1999 corresponded to high legal CPUEs in those years. The 22% decline in total CPUE from 1999 to 2000 corresponded to a 39% decline in legal-size lobster CPUE in 2000 (0.151 in 1999 to 0.092 in 2000; Table 1). Total CPUE declined by nearly 50% from 2001 to 2002 and legal-size CPUE declined 35% to 0.061 in 2002, which represents the lowest legal CPUE value observed since the study began in 1978 (previous range: 0.065-0.173). The time series of annual Δ-mean legal-size lobster CPUEs have exhibited a significant declining trend since 1978 (slope=-0.002, p=0.022).

The trends observed in our indices of lobster abundance (total and legal CPUEs) closely followed trends in abundance of lobsters taken in the eastern LIS commercial fishery. Commercial catches fluctuated between 81,000 lbs and 349,500 lbs from 1979 to 1991 (Table 1). Catches peaked in 1992 (446,900 lbs) and then declined by 40% in 1996 (266,700 lbs). Subsequently, catches increased and reached record levels of 480,200 lbs in 1998. Following the record landings in 1998, commercial catches declined 16% in 1999 (402,300 lbs), 40% in 2000 (242,500 lbs) and 13% again in 2001 (210,000 lbs). Commercial catches continued to decline in 2002 (19%), when only 170,400 lbs were taken in the eastern LIS fishery, the lowest catch observed in our area since 1986.

Annual CPUE values for all sizes of lobster and legal-size lobster are presented for each station in Figure 3. Total CPUE during 2002 was highest at Twotree (1.291), lowest at Jordan Cove (0.649), and intermediate at Intake (0.747); values at Jordan Cove and Intake were the lowest reported in this study (previous range: 0.753-2.701 and 0.839-2.660). Over the past 25 years, total CPUE has significantly increased at Twotree (slope=0.036, p=0.027), but not at Jordan Cove or Intake. The Twotree station also yielded the highest catch of legal-size lobsters during 2002 (0.079), followed by Jordan Cove (0.055) and Intake (0.050; Fig. 3): the legal catches at Jordan Cove were below the range of values reported previously
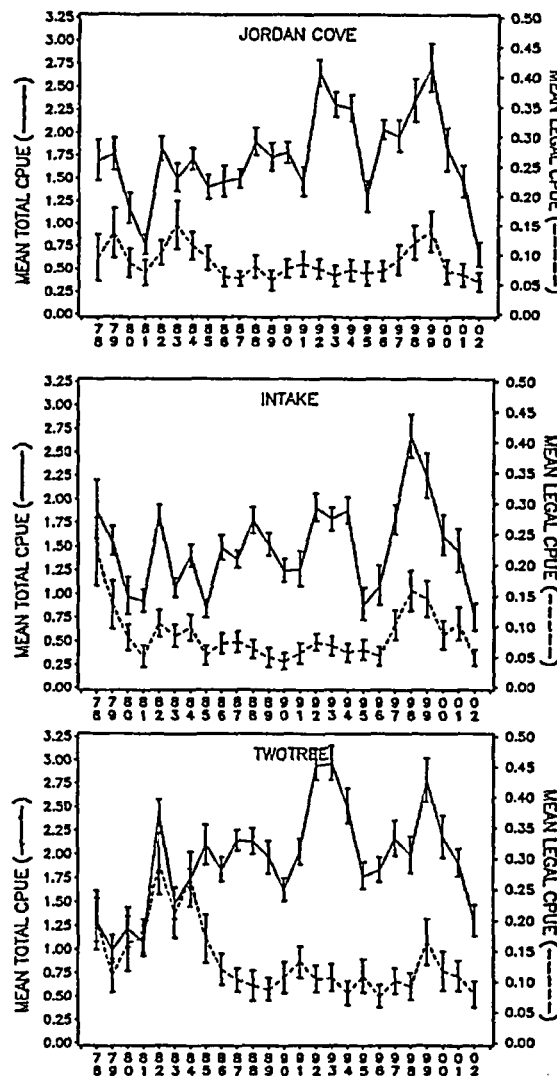
Fig. 3. Mean total CPUE (geometric mean ± 95% C.I.) and mean legal CPUE (Δ-mean ± 95% C.I.) of lobsters caught at each station from 1978 to 2002 (minimum legal size increased from 81.0 mm to 81.8 mm in 1989 and to 82.6 mm in 1990).

(0.057-0.151). In general, when compared with the nearshore stations, Twotree has consistently yielded higher catches of legal-size lobsters. However, long-term trend analysis indicates that catches of legal-size lobster have significantly declined at this station since 1978 (slope=-0.004, p=0.022); whereas no trends in legal-size lobster CPUE were identified at Jordan Cove or Intake.

Comparisons of the long-term (1978-2001) monthly mean total CPUE with the 2002 CPUE for all stations combined indicate similar temporal patterns. Highest total CPUE occurred in May and June and lowest in September and October during both time periods (Fig. 4). All of the monthly total CPUE values during 2002 were lower than the long-term values; total CPUE in September and October 2002 were among the lowest
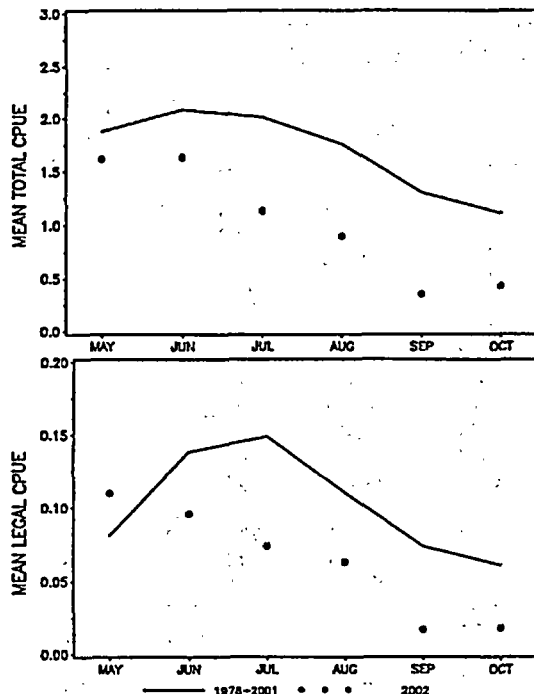
Fig. 4. Monthly mean total CPUE and Δ-mean legal CPUE during the period from 1978 to 2001 and during 2002 studies.

observed in our 25-year lobster studies. Legal-size lobster CPUE peaked in May 2002, which was earlier than the July peak in legal-size CPUE observed during the period 1978-2001. Except for May 2002, all of the monthly mean legal lobster CPUEs were lower than the averages observed for those same months during 1978-2001; September and October legal CPUEs were among the lowest observed. The monthly patterns of our lobster catches and the seasonal trends in LIS landings (highest from July-September; ASMFC 2000) were directly influenced by water temperature. As water temperature increases above 10°C, lobster activity (e.g., feeding, movement, and molting) increases. Similar correlations between temperature and lobster catchability have been reported throughout the range of lobsters (McLeese and Wilder 1958; Dow 1966, 1969, 1976; Flowers and Saila 1972; Koeller 1999).

In addition to lobsters, pot gear often catches other marine organisms including crustaceans, mollusks, and fishes. The presence of these organisms in lobster pots can negatively influence the catch of lobster (DNC 2002). For example, the catchability of lobsters in pots fished near MPS was strongly depressed when spider crabs were abundant in pot gear (Table 2). On several occasions hundreds of spider crabs were caught in a single pot and completely blocked the entry funnels of the trap. Spider crabs can also indirectly influence lobster catchability by consuming bait. During 2002,

TABLE 2. Total number of lobsters and incidental catch of other species caught in traps.

	Range (1984-2001)	2002
Lobster	6435-11438	4829
Rock, Jonah crab	79-2033*	267*
Spider crab	1344-31480*	6581*
Hermit crab	192-721*	433
Blue crab	21-169	107
Winter flounder	5-45*	18
Summer flounder	4-60*	30
Skates	14-94	41
Oyster toadfish	5-76	22
Scup	21-1039*	305
Cunner	41-239*	140
Tautog	39-250*	195
Sea raven	0-20	6
Whelks	21-178*	101*

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

spider crabs continued to be a dominant component of the total by-catch and significantly influenced lobster catches at the Intake station. Rock and Jonah crabs have been dominant components of the by-catch at Twotree and significantly influenced lobster catches at that site during 2002. At Jordan Cove, whelks were found to strongly influence lobster CPUE during 2002; this large gastropod also influenced lobster catches in previous study years. Results of our studies are similar to findings of other researchers in southern New England, which indicated that the incidental catches of rock crabs and spider crabs significantly affect lobster catch (Richards et al. 1983; Cobb et al. 1986; Richards and Cobb 1987). Behavioral interactions between crabs and European lobsters (H. gammarus) were demonstrated to have substantial effects on the catch of both species in pot gear and operated in a density-dependent manner (Addison and Bannister 1998).

Population Characteristics
Size Composition

Annual size frequency distributions of lobsters caught in 8-mm size classes are illustrated in Figure 5. Although the catch of lobsters in all size classes declined during 2002, the pattern of size composition was similar to previous study years and continued to be dominated by pre-recruit (67-74 mm) and recruit (75-82 mm) sized lobsters. A notable feature of the annual size-frequency distributions demonstrating the effects of the intense fishing pressure around MPS is the low number of lobsters in the 83-90 mm and >90 mm size classes. Regardless of the number of lobsters in the recruit size class (75-82 mm CL), few lobsters in the 83-90 mm size class (full recruit) were caught in our traps because they were harvested by commercial
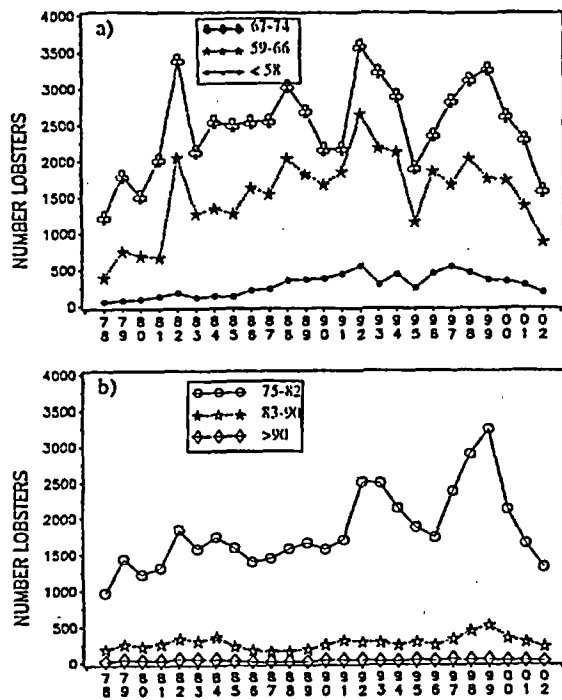
Fig. 5. Number of lobsters caught in each 8-mm carapace length category from (a) <58 mm, 59-66 mm, and 67-74 mm and (b) 75-82 mm, 83-90 mm, and >90 mm (30 wood and 30 wire traps fished from 1978 to 1981; 60 wire traps fished from 1982 to 2002).

lobstermen soon after molting to legal size. Lobsters larger than 90 mm CL were rarely present in the annual catch, which indicates that the LIS fishery is almost entirely (>90%) dependent on new recruits.

Mean carapace length (CL) of all lobsters during 2002 was 71.6 mm, falling within the range of annual average sizes reported previously (69.5-72.2 mm; Table 3). Percentage of legal-size lobsters (≥ 82.6 mm) during 2002 was 5.2%, within the range reported in previous studies when the legal size was ≥ 82.6 mm (1990-99; 3.1-5.7%) and ≥ 81.0 mm (1978-88; 3.2-9.1%). Since 1978, the percentage of legal-size lobsters in our catch has significantly declined (slope=-0.113, p=0.028). The decline can be attributed to changes in fishery regulations (increases in minimum legal size in 1989 and 1990) and to the eightfold increase in the number of traps fished by Connecticut and New York fishermen in LIS since the early 1980s (Crecco and Gottschall 1999; ASMFC 2000).

The mean CL during 2002 was largest at Twotree (72.2 mm), smallest at Intake (71.0 mm), and intermediate at Jordan Cove (71.2 mm; Table 4). Mean sizes at each of the three stations during 2002 were within the range of values reported from 1978 to 2001 (Twotree, 70.0-73.7 mm; Intake 68.9-72.4 mm; Jordan Cove, 68.8-71.8 mm). In contrast to previous

TABLE 3. Summary of lobster carapace length statistics from wire pot catches from May through October, 1978-2002.

	N[a]	Carapace length (mm) Range	Mean ±95% CI	Percentage of legal sizes[b]
1978	1508	53-111	71.4 ± 0.33	7.5
1979	2846	44-100	71.2 ± 0.26	7.6
1980	2531	40-96	70.7 ± 0.27	6.4
1981	1983	43-96	71.0 ± 0.33	8.8
1982	7835	45-103	70.8 ± 0.15	6.7
1983	5432	40-121	71.7 ± 0.19	9.1
1984	6156	45-107	71.8 ± 0.18	8.7
1985	5723	38-101	71.3 ± 0.17	5.9
1986	5961	36-107	70.1 ± 0.17	4.4
1987	5924	36-99	70.2 ± 0.17	3.9
1988	7144	21-97	69.5 ± 0.16	3.2
1989	6713	34-107	69.9 ± 0.17	3.5
1990	6040	36-102	70.2 ± 0.20	4.9
1991	6449	31-101	70.2 ± 0.20	5.0
1992	9594	20-103	70.1 ± 0.15	3.3
1993	8487	30-102	70.8 ± 0.15	3.3
1994	7841	34-100	70.3 ± 0.17	3.1
1995	5472	37-101	71.9 ± 0.20	5.7
1996	6634	16-96	70.0 ± 0.19	3.8
1997	7528	35-100	71.5 ± 0.17	4.8
1998	8958	25-99	71.3 ± 0.16	4.9
1999	9155	32-110	72.2 ± 0.15	5.6
2000	7151	46-110	71.1 ± 0.17	4.7
2001	5888	32-99	71.0 ± 0.19	4.9
2002	4180	43-96	71.6 ± 0.23	5.2

[a] Recaptures not included.
[b] The minimum legal size from 1978 to 1988 was 81.0 mm (3 3/16 in), minimum legal size was increased in 1989 to 81.8 mm (3 7/32 in), and in 1990, to 82.6 mm (3 1/4 in).

TABLE 4. Summary of lobster carapace length statistics from wire pot catches at each station from May through October, during the period from 1978 through 2001 and during 2002 studies.

	Mean carapace length (mm)[a]	Percentage of legals[b]
JORDAN COVE		
1978-2001 range	68.8 - 71.8	2.7 - 8.2
2002 mean	71.2	5.8
INTAKE		
1978-2001 range	68.9 - 72.4	2.8 - 9.2
2002 mean	71.0	5.3
TWOTREE		
1978-2001 range	70.0 - 73.7	3.1 - 14.2
2002 mean	72.2	4.9

[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).

study years when the percentage of legal-size lobsters was highest at Twotree, the percentage during 2002 was highest at Jordan Cove (5.8%) followed by Intake

TABLE 5. Female to male sex ratios[a] of lobsters caught in wire pots from May through October, 1978-2002.

	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
1997	0.35	0.48	0.77	0.52
1998	0.48	0.59	0.92	0.63
1999	0.49	0.60	0.79	0.62
2000	0.37	0.44	0.86	0.55
2001	0.36	0.41	1.01	0.57
2002	0.40	0.38	1.18	0.66

[a] Recaptures not included.

(5.3%) and Twotree (4.9%). Since 1978 the percentage of legal size lobsters has significantly declined at Twotree (slope=-0.210; p=0.009); but not at Jordan Cove or Intake.

Sex Ratios

The sex ratio of lobsters collected during 2002 was 0.66 females per male, compared to a range of 0.48 to 0.97 previously observed (Table 5). Since 1978, annual female to male sex ratios have exhibited a significant declining trend (slope=-0.014; p<0.0001). During 2002, female to male sex ratio was highest at Twotree (1.18), lowest at Intake (0.38) and intermediate at Jordan Cove (0.40). Values at each of the three stations were within the range of values reported over the past 25 years. The occurrence of more females at Twotree than at other stations has been consistent since 1975 (Keser et al. 1983; DNC 2002). Significant declines in female to male ratios were noted from 1978 to 2002 at Jordan Cove (slope=-0.016; p<0.001) and Intake (slope=-0.017; p<0.001), but not at Twotree. The cause for the decline in female to male sex ratios is unclear, although other researchers working in southern New England and New York have observed similar declines in localized nearshore areas (K. Castro URI and K. Graulich NYSDEC, pers.

comm.). Beginning in 1998, we recorded the sex of lobsters caught in our trawl monitoring program at Jordan Cove, Intake and Niantic River (see Fish Ecology section). These data also indicated a predominance of males in nearshore waters around MPS with female to male sex ratios of 0.72 in 1998, 0.74 in 1999, 0.47 in 2000, 0.54 in 2001 and 0.46 in 2002. Other lobster populations typically show a 1:1 sex ratio (Templeman 1936; Ennis 1971, 1974; Stewart 1972; Krouse 1973; Thomas 1973; Briggs and Mushacke 1980; Lawton and Lavalli 1995), although there are some reports of male- and female-dominated populations and seasonal assemblages (Cooper et al. 1975; Briggs and Mushacke 1979; Howell and Watson 1991; Howell et al. 1999). Changes in male and female growth and maturity could cause differential susceptibility of the sexes to the trap fishery and lead to shifts in the sex ratio. In populations that are heavily fished, the sex ratios of individuals just above the legal size are expected to be skewed toward females because berried females are protected from harvest and therefore have lower mortality rates than males (Cobb 1995). Smith (1977) reported female to male sex ratios in the LIS commercial fishery ranging between 1.06 and 1.81 and more recently, sex ratios of lobsters caught in eastern LIS commercial traps were unusually higher, ranging between 2.61 and 6.29 females per male (Blake 1988). During routine trawl surveys conducted from 1986 to 1998 throughout LIS by the CTDEP, female to male sex ratios of lobsters one molt below legal-size averaged 1.44 during the spring and 0.96 during the fall (ASMFC 2000). In contrast, female to male sex ratios of legal lobsters caught in the CTDEP trawl surveys were dominated by males during the fall (0.64) and to a lesser degree during spring (0.96).

Reproduction

Aiken and Waddy (1980) described a number of approaches to determine the size at which females become sexually mature. A simple technique is to examine the size distribution of egg-bearing females in the population. One limitation of this approach is that once females extrude eggs, they are protected from the fishery and the proportion of berried females above the minimum legal size can be overestimated. More accurate methods require sacrificing animals to examine the size and color of ovaries. For the past 22 years, we have used a technique first described by Templeman (1935, 1944) who found that abdominal width measurements of females markedly increase at the first onset of sexual maturation. Female size at sexual maturity was estimated by calculating the ratio
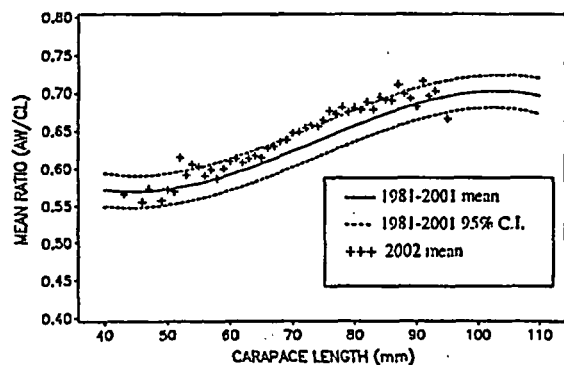
Fig. 6. Morphometric relationship between the mean abdominal width to carapace length ratio (y) and the carapace length (x) of non-berried female lobsters during the period from 1981 to 2001 (——) and during 2002 (+ + +). Regression equation for each period:
1981-2001: $y=0.61-(0.43*10^{-2})x+(0.96*10^{-4})x^2-(0.42*10^{-6})x^3$, $r^2=.95$
2002: $y=1.04-(2.68*10^{-2})x+(4.66*10^{-4})x^2-(2.35*10^{-6})x^3$, $r^2=.95$
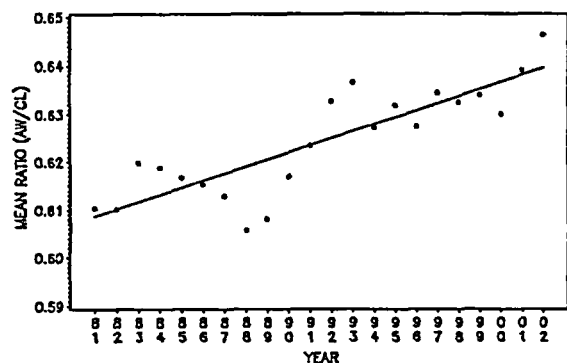
Fig. 7. Annual mean abdominal width to carapace length ratio and linear trend of the ratio for non-berried females from 1981 to 2002.

	Jordan Cove	Intake	Twotree	All Stations
1978	3.5	3.1	7.4	4.7
1979	2.9	3.4	7.9	4.4
1980	2.9	2.4	6.0	3.9
1981	2.1	3.4	8.0	5.1
1982	1.3	0.9	7.2	3.1
1983	3.1	3.2	9.0	5.6
1984	4.3	3.2	12.7	7.5
1985	4.1	4.8	9.6	7.2
1986	4.2	2.6	9.7	6.1
1987	3.8	2.8	10.8	7.0
1988	4.1	2.8	7.3	5.1
1989	4.6	4.6	9.7	6.7
1990	3.7	4.6	14.2	7.8
1991	3.9	1.5	16.5	9.1
1992	4.2	2.4	27.3	15.3
1993	3.9	3.7	27.4	16.0
1994	8.4	5.8	25.1	15.0
1995	7.5	7.2	17.4	12.0
1996	4.3	4.0	17.4	10.4
1997	5.8	3.9	25.8	13.3
1998	8.3	6.0	27.9	13.5
1999	7.9	7.6	18.0	11.7
2000	2.8	6.4	18.5	11.0
2001	5.2	6.4	25.2	14.9
2002	4.9	2.5	20.5	13.4

[a] Recaptures included

of abdominal width to carapace length (AW/CL) and plotting that ratio against carapace length (Skud and Perkins 1969; Krouse 1973). Based on measurements obtained from 67,616 non-berried females, mean AW/CL ratios were calculated for each 1-mm CL and plotted against the carapace length of lobsters collected from 1981 to 2002 (Fig. 6). During 2002, females began to mature between 50 and 55 mm CL, and all females larger than 90 mm CL were mature. When comparing females caught in 2002 with those caught from 1981 to 2001, many of the individual 1-mm mean values for 2002 were above the long-term mean. Closer examination of annual mean AW/CL ratios, revealed a shift in the relationship between female abdominal width and carapace length over the past 22 years. Annual mean AW/CL values fluctuated between 0.610 and 0.620 from 1981 to 1987 and averaged 0.614 (Fig. 7). Beginning in 1988, mean values increased each year from 0.606 to 0.637 in 1993. From 1994 to 2002 annual AW/CL values stabilized at values ranging between 0.627 and 0.646, with an average of 0.633. The discontinuity in the AW/CL

ratio and the changes observed in the proportion and size distribution of berried females (see below) suggests a recent decrease in the size at sexual maturity of female lobsters in LIS (Landers et al. 2001b).

The recent change observed in the time-series of morphometrics of non-berried female lobsters was followed by an increase in the abundance of egg-bearing females. From 1978 through 1990, between 3.1 and 7.8% of females were egg-bearing (Table 6). The percentage of berried females increased sharply to 16.0% in 1993 and remained at above-average levels through 2002, when 13.4% of females were egg-bearing. More pronounced increases in berried female abundance occurred at Twotree, where the percentage of egg-bearing females has been highest since 1978 (Keser et al. 1983; DNC 2002). Between 6.0 and 14.2% of females caught annually at Twotree were egg-bearing from 1978 to 1990, compared to between 16.5 and 27.9% annually from 1991 to 2002 (Table 6). Annual percentages of berried females at the nearshore Jordan Cove and Intake sites were also generally higher over the past decade (2.8-8.4% and 1.5-7.6%, respectively), than during the earlier study years from 1978 to 1990 (1.3-4.6% and 0.9-4.8%, respectively).

Following the recent decrease in female size at onset of sexual maturity, annual average carapace length of egg-bearing females has become smaller and the
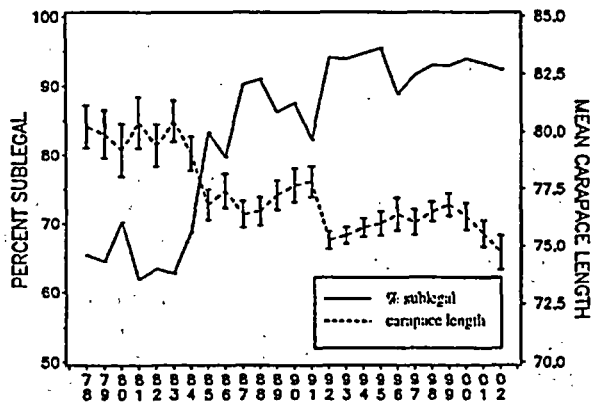
Fig. 8. Percentage sublegal and mean carapace length (± 95% C.I.) of berried females collected from 1978 to 2002.

Fig. 9. a) Cumulative percentage of molting lobsters caught each week and b) molt frequency curves based on the Gompertz function fitted to data in a) of lobsters caught from May 1 (week 1) through October 31 (week 26) during the period from 1980 to 2001 and during 2002.

proportion of sublegal-size egg-bearing lobster has increased. The mean CL of berried females was 74.7 mm during 2002 and 92% were sublegal-size (Fig. 8). Similar average sizes and percentages of sublegal-size were observed in the past 10 years. During the early study years from 1978 to 1987, egg-bearing females averaged 79.2 mm (CL) and 70% were sublegal-size. In recent study years from 1992 to 2002, berried female average size decreased to 75.8 mm (CL) and 93% were below the legal size.

The size at sexual maturity for female lobsters varies widely across the species range (Aiken and Waddy 1980; Waddy et al. 1995). Maturation occurs at smaller sizes in shallow warmwater locations of southern New England and the Gulf of St. Lawrence and at larger sizes in deep coldwater locations offshore and in the Gulf of Maine (Krouse 1973; Briggs and Mushacke 1979; Van Engel 1980; Aiken and Waddy 1986; Fogarty and Idoine 1988; Estrella and McKiernan 1989; Blake 1994). Results from our studies over the past two decades showed that the onset of female lobster sexual maturity occurs between 50 and 60 mm (CL). Based on abdominal growth, our long-term study suggests a recent reduction in the size at which females become sexually mature. Additional evidence supporting this finding includes an increase in the proportion of egg-bearing females and a decrease in their mean carapace length. It is presently unclear whether reductions in female size at maturity resulted from changes in environmental conditions or from selection pressures related to intense fishing pressure (Landers et al. 2001b). Average bottom seawater temperatures from 1979 to 2002 demonstrate a significant increasing trend (see below Molting and Growth) and, as mentioned earlier, the number of traps fished by Connecticut and New York fishers has increased almost eightfold in the past two decades (AFMSC 2000). Water temperatures can directly influence lobster maturation (Aiken and Waddy 1980),
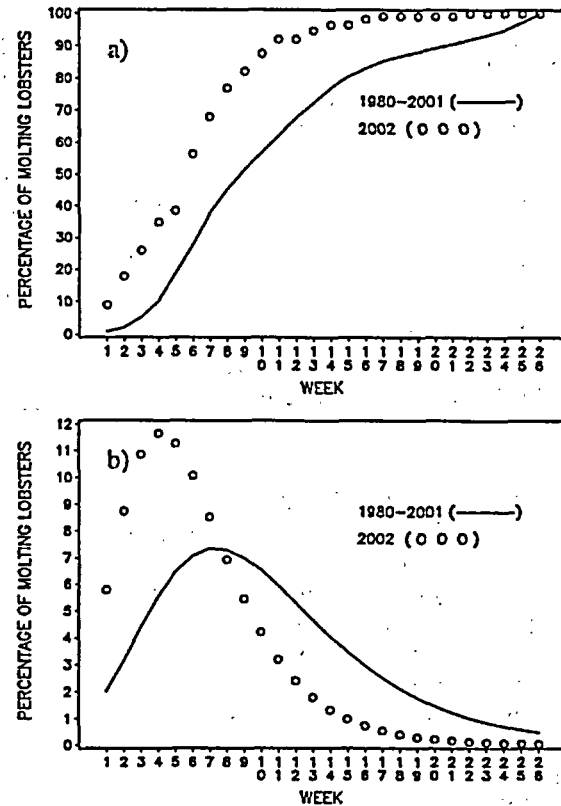
whereas high exploitation rates may be exerting a strong selection for earlier maturing lobsters. While the exact causes for the reduction in female size at maturity are presently unclear, the benefits of earlier maturation may be significant to our local lobster population. Small size at maturity and subsequent egg production from sublegal-size females may explain why LIS lobster populations are so resilient to high levels of fishing pressure.

Molting and Growth

Lobster growth is a function of molt frequency and size (or weight) increase per molt. Water temperature is the most important factor regulating molting events (Aiken 1980). In our study area, the majority of molting lobsters were caught from late spring to early summer, although a secondary molt was observed in autumn of earlier study years (1978-1982) when sampling was conducted through November (Keser et al. 1983). The timing and frequency of lobster molts were examined using weekly cumulative percent-molt data. Examples of these data are presented in Figure 9a
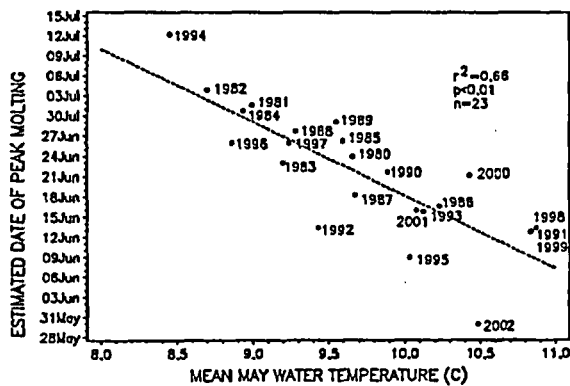
Fig. 10. Relationship between the date of peak molting of lobsters collected near MPS (parameter t from the Gompertz function) and annual mean bottom water temperature during May (1980-2002).

for lobsters caught during 2002 and during the period from 1980 to 2001. The percentage of molting lobsters in the total catch peaked at about 12% during week 4 of the 2002 study compared to a peak of 7% during week 7 of previous studies (Fig 9b). Annual molting peaks were significantly (p<0.05) correlated with mean May bottom water temperatures. 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. During May 2002, bottom water temperature averaged 10.5°C and molting peaked on May 29, which was the earliest molting peak observed in our study (Fig. 10). In general, when bottom water temperatures during May averaged more than 10.0°C, molting peaked during the first or second week of June. Molting peaks were delayed up to one month (e.g., July 12, 1994), when May temperatures averaged below 9.0°C. Aiken and

Waddy (1980) found that when water temperatures rise above 10°C lobsters quickly enter the premolt stage and progress to ecdysis. Early laboratory studies conducted by Templeman (1936) indicated that molting was delayed one week for every 1°C reduction in water temperature. In colder waters of Nova Scotia, differences in the timing of annual molts were linked to average bottom water temperatures in August (Tremblay and Eagles 1997).

Molt increments were determined from lobster tagging studies by comparing CL measurements at the time of tagging with those from recaptured lobsters that had molted only once. Simple linear regressions of pre-molt (tag-size) and post-molt (recapture-size) sizes best describe growth for the size range of lobsters caught during our studies. Linear regression plots of growth for males and females caught in 2002 and those caught from 1979 through 2001 are shown in Figure 11; parameter estimates are provided in Table 7. During 2002, average molt increments of both sexes were smaller than the average increments reported in previous studies. Females grew an average of 7.19 mm per molt during 2002, compared to an average of 8.52 mm from 1979 to 2001. Average male incremental growth of 5.57 mm was smaller than females during 2002 and almost than 3 mm smaller than the average male increment of 8.42 mm from 1979 to 2001. When the three lobster collection stations were compared, molt increments and corresponding percentages of growth per molt during 2002 were largest at Twotree (7.08 mm, 10.9%), and smaller at the nearshore Jordan Cove and Intake stations (6.10 mm, 8.4% and 5.71 mm, 8.5%, respectively; Table 8). Although percent
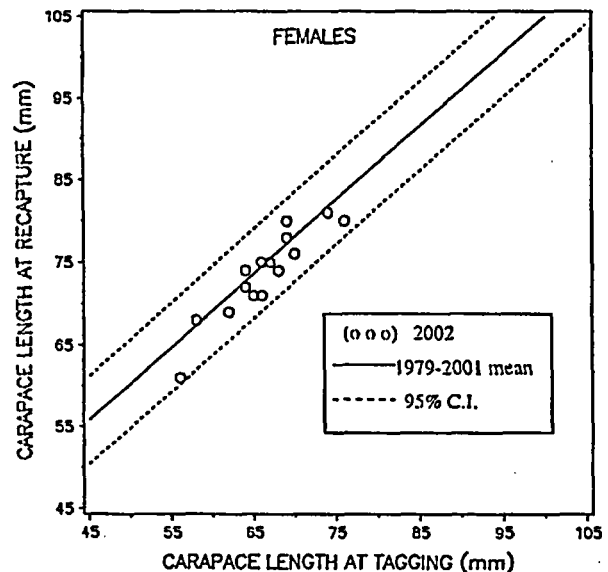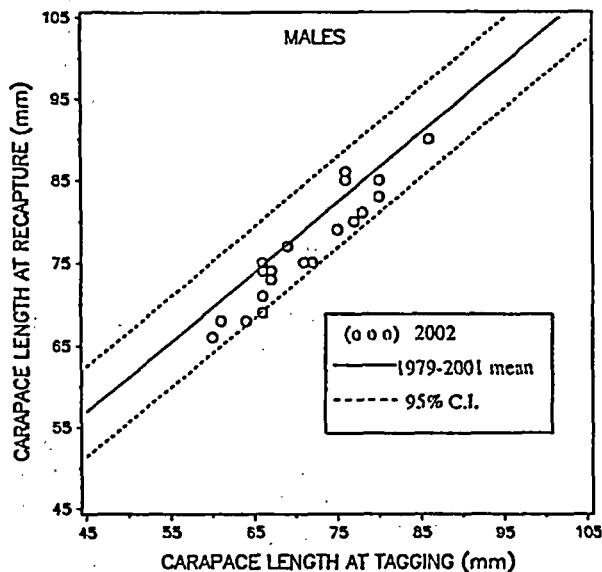
Fig. 11. Relationship between carapace lengths at tagging and carapace lengths at recapture during the period from 1979 to 2001 and during 2002 at sites near MPS of both male and female lobsters.

TABLE 7. Simple linear regression equations describing the relationship between carapace lengths of male and female lobsters at tagging (x) and at recapture (y) in the vicinity of MPS from 1979 to 2001 and during 2002.

		MALES						FEMALES		
	N	Growth model	R^2	Growth increment mean (mm)			N	Growth model	R^2	Growth increment mean (mm)
1979-2001	2076	y=18.878+0.846(x)	0.72	8.42			2090	y=15.650+0.893(x)	0.74	8.52
2002	23	y=13.755+0.885(x)	0.90	5.57			16	y=15.638+0.874(x)	0.84	7.19

TABLE 8. Annual mean growth per molt (incremental and percentage) at each station based on tag and recapture studies conducted from 1979 through 2001 and during 2002.

	1979-2001		2002	
	mean increment (mm)	%	mean increment (mm)	%
Jordan Cove	8.26	12.5	6.10	8.4
Intake	8.38	12.7	5.71	8.5
Twotree	8.70	13.0	7.08	10.9

Fig. 12 a) Annual mean incremental growth per molt of male and female lobsters based on tag and recapture studies conducted from 1979 to 2002and b) means and linear trend (± 95% C.I.) of bottom water temperature measured at 3 stations from May through October 1979 to 2002.

growth per molt was lower during 2002, average values of between 12.5 and 13.0% reported during the period 1979-2001 at the three stations were similar to others reported for LIS lobsters, 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).

Male and female molt increments were highly correlated (r=0.81, p<0.0001) and the time series of annual growth increments for both sexes (Fig. 12a) have exhibited a significant declining trend since 1979 (male slope=-0.100, p=0.002; female slope=-0.091, p<0.0001). Average bottom seawater temperature measured from May through October at the three stations ranged between 15.0 and 17.5°C (Fig. 12b) and has exhibited a significant increasing trend (slope=0.060, p=0.001). A significant inverse correlation was found between bottom seawater temperature and the declining growth rates of male lobsters (r=-0.43, p=0.04), but not for female lobsters in our study area. Molt increments were shown to be smaller in blue crabs raised in warmer water (Leffler 1972) and comparisons of molt increments from lobster tagging studies conducted in cooler offshore waters (Cooper and Uzmann 1971; Fogarty and Idoine 1988) with those in warmer areas (DNC 2002) suggest this is also true for adult lobsters. The declining trend in female growth may be related to earlier maturation,

increased reproductive activity, and changes observed in berried female size structure observed over the past two decades as discussed in the previous section and by Landers et al. (2001b). Females may sacrifice somatic growth for development of ovaries and incubation and development of eggs. Physiological stressors associated with the recent outbreak of shell disease could also be a potential contributing factor to the smaller growth increments observed over the past few years. Finally, high exploitation rates may be exerting a strong selection for slower-growing earlier-maturing lobsters. Similar population responses have been demonstrated for spiny lobster populations under high fishing pressure (Polovina 1989; Chubb 1994) and for

TABLE 9. Percentage of culls (lobsters missing one or both claws) caught in wire pots at sites near MPS 1978-2002.

	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
1997	15.1	12.8	6.2	11.2
1998	13.8	12.5	6.8	11.3
1999	12.9	12.0	7.4	10.7
2000	12.3	10.8	5.9	9.4
2001	15.0	15.0	6.6	11.9
2002	18.1	15.9	7.6	12.7

crayfish in controlled fishing experiments on closed populations (Momot 1998).

Culls

The percentage of culls (i.e., lobsters missing one or both claws) was 12.7% of the total catch during 2002 and within the range of values reported since the study began in 1978 (9.4-15.5%; Table 9). The percentage of culls at the nearshore Jordan Cove and Intake stations during 2002 was 18.1% and 15.9%, respectively, which was more than double the percentage found at Twotree (7.6%). The pattern of higher claw-loss at the male-dominated nearshore sites has been consistent since the study began (Keser et al. 1983) and may be related to the fact that males are more aggressive, more active and more dominant than females (Atema and Voigt 1995). The escape vent regulation instituted in 1984 requires that commercial lobster pots contain an opening to allow escape of sublegal-size lobsters and reduce injury and mortality associated with overcrowded pots (Landers and Blake 1985; Landers and Keser 2000). Since 1984, claw-loss has averaged 11.0%, which was lower than the average value of 12.7% reported before the implementation of escape vents. The 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). Fishery managers have recently implemented a number of additional escape vent requirements including larger vent size, use of biodegradable fasteners when installing vents, and required location of the vent within the trap (parlor section). These measures are expected to further improve lobster survival and recruitment by minimizing trap-related injury and mortality.

Tagging Program

The total number of lobsters tagged during 2002 was 3,892, the lowest number tagged since all wire pots were introduced in 1982 (5,160-9,126; Table 10). The percentage of lobsters tagged in 2002 that were recaptured in Millstone pots during 2002 was 16.7% and within the range of values observed since the study began in 1978 (range=14.4-26.2%). The number of tags returned by commercial lobstermen over the past few years has declined. Commercial lobstermen recaptured 7.4% of the tagged lobsters in 2000, 3.7% in 2001 and 6.8% in 2002, which were the lowest values reported in this study (previous range=7.6-47.6%). Many factors influence the percentage of tags returned annually by commercial lobstermen. Any increase in natural mortality due to the increased prevalence and severity of shell disease would reduce the number of tagged lobsters available for capture by commercial lobstermen. Also, there is lag of about 1 year between the time when lobsters are tagged and the time when commercial lobstermen forward tag return information. The percentage recaptured in commercial traps during 2002 may increase as lobstermen submit their tag return information in 2003. Some lobstermen may have left the industry due to the decline in lobster abundance over the past few years. In 1998 and 1999 when legal catches were high, lobstermen had little time to spend recording tag recovery information. Additionally, the $2.00 reward incentive for each tag return is minimal when compared to the price of their catch (pers. comm., several local lobstermen). Lastly, since the implementation of the escape vent regulation in 1984 and recent amendments to this regulation (i.e., larger vent size), fewer recaptures were made by commercial lobstermen. Most tagged lobsters are sublegal and difficult to retain in commercial traps fitted with escape vents. In contrast, the number of recaptures made by Millstone since 1984 increased, because our traps do not have escape vents and retain more tagged sublegal-size lobsters.

TABLE 10. Lobster tag and recapture statistics from Millstone pots (May-Oct.) and commercial pots (Jan.-Dec.) from 1978 to 2002.

	Number tagged	Millstone				Commercial			
		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	72.9	1183	18.4	20.7	78.2
1990	5741	1066	18.6	5.5	73.3	1007	17.5	26.5	79.3
1991	6136	1109	18.1	7.4	73.4	1228	20.0	33.8	80.8
1992	9126	1842	20.2	3.9	72.4	1559	17.1	23.4	79.5
1993	8177	1708	20.9	3.6	73.4	1768	21.6	27.4	79.4
1994	7533	1974	26.2	3.1	73.4	1020	13.5	21.3	77.3
1995	5307	963	18.1	5.4	75.0	1116	21.0	27.1	80.0
1996	6221	897	14.4	4.2	72.4	537	8.6	28.3	79.4
1997	7153	1251	17.5	5.4	74.1	1134	15.8	30.3	81.5
1998	8586	2033	23.7	5.2	74.0	718	8.4	13.8	82.1
1999	8633	2061	23.9	4.7	74.9	660	7.6	30.7	82.9
2000	6798	1556	22.9	3.3	72.9	504	7.4	34.5	82.5
2001	5465	1380	25.2	6.8	73.6	205	3.7	45.5	83.6
2002	3892	649	16.7	8.2	74.7	265	6.8	32.3	80.9

[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).

The mean CL of lobsters recaptured in Millstone traps was 74.7 mm during 2002, which was within the range of average sizes reported previously (72.0-75.7 mm; Table 10). In contrast, lobstermen consistently recaptured larger lobsters. During 2002, the mean CL of 80.9 mm was among the largest average sizes reported in commercial gear (previous range=75.5-83.6 mm). The average size of tagged lobsters caught in commercial traps has significantly increased over the past 25 years (slope=0.258; p<0.001). Shifts in the size of lobsters recaptured in Millstone and commercial traps were related to escape vent regulations (Landers and Keser 2000). Prior to the regulation, commercial lobstermen recaptured many of the tagged sublegal-size lobsters; with the regulation in force, many of the sublegal-size lobsters escaped from the vented commercial pots, but were retained in unvented Millstone pots. In eastern LIS, Landers and Blake (1985) noted a substantial reduction in the number of sublegal-size lobsters retained in vented pots, without a corresponding decrease in the catch of legal-size lobsters.

Since the tagging study began in 1978, commercial lobstermen have consistently caught a higher percentage of tagged legal-size lobsters than have

Millstone researchers. During 2002, 32.3% of the recaptures in commercial pots were legal-size (≥ 82.6 mm) compared to only 8.2% in Millstone pots (Table 10). The 25-year time series of percentage legal-size lobsters recaptured in Millstone pots has significantly declined (slope=-0.338; p=0.001); no trend was identified in the percentage of legal-size lobsters in commercial traps. The decline in percentage of legal-size lobster recaptures in our traps is not surprising, given the fact that the number of traps fished by Connecticut and New York fishermen in LIS has increased eightfold since the early 1980s (ASMFC 2000). At such high levels of fishing effort, tagged legal-size lobsters are caught by commercial lobstermen soon after they are released and our chances of catching them again are small given the large number of commercial traps present in comparison to the small number of Millstone traps.

Movement

Tag return information from the Millstone monitoring study and commercial lobstermen was used to assess the extent of lobster migrations in the MPS area.

Fig. 13. Location and number of lobsters recaptured in the MPS area by commercial lobstermen from 1978 to 2002.

Fig. 14. Location and number of lobsters caught by commercial lobstermen farther than 5 km from MPS in the vicinity of Long Island and Block Island Sounds from 1978 to 2002.

Fig. 15. Number of tag returns at locations farther than 50 km from MPS from 1978 to 2002.

During 2002, the majority of the lobsters recaptured in Millstone pots were caught at the same station at which they were released (99% at Jordan Cove, 96% at Intake, and 95% at Twotree). This pattern of short-range movement among sublegal-size lobsters was also

observed in previous years and in the recapture information provided by commercial lobstermen (Landers and Keser 1994). Since 1978, 92% of the tagged lobsters recaptured in commercial pots were caught within 5 km of MPS (Fig. 13). Stewart (1972) demonstrated a strong homing behavior for the nearshore eastern LIS lobster population. Individual lobsters leave their burrows at night and return to the same shelters before dawn. This territorial and nocturnal behavior pattern limits their home range. 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; Watson et al. 1999).

Most of the lobsters recaptured more than 5 km from the study area had moved to the east or southeast. Since 1978, only 25 individuals were recaught in western LIS (Fig. 14). Over the past 25 years, more than 1,000 tagged lobsters were caught by commercial lobstermen in The Race, which suggests that 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 in Block Island (43) and Rhode Island Sounds (20). Some lobsters traveled more than 50 km to the east and were caught in Narragansett Bay, RI (34), Buzzards Bay, MA (6), and in waters near Martha's Vineyard (10) and Nantucket Island (3; Fig. 15). Only two individuals were reported being caught in nearshore waters along the south shore of Long Island, compared to 29 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). Similar exchanges between inshore and offshore lobster populations have been observed by other researchers (Saila and Flowers 1968; Uzmann et al. 1977; Cooper and Uzmann 1980; Campbell and Stasko 1985, 1986; Estrella and Morrissey 1997). Based on our tagging studies, the number of lobsters captured offshore seems insignificant. However, fishing effort for lobsters along the continental shelf is low relative to coastal areas; therefore, the probability of capturing our tagged lobsters in this large area is concomitantly low.

Entrainment

Day and night lobster larvae samples were collected at the discharges of Units 2 and 3 during a 10-week period from May 13 to July 19, 2002. The total

Fig. 16. Annual number of lobster larvae and their stage composition (Stage I-IV) collected in samples taken at the MPS discharges from 1984 through 2002.

number of larvae found was 105; as in previous studies, Stage I larvae predominated the collections (n=84), followed by Stage IV (n=11) and Stage II and III (n=5; Fig. 16). The density of lobster larvae in night samples (0.961 per 1000 m³) was nearly four times higher than the density of larvae found in day samples (0.252 per 1000 m³; Table 11); however, the diel variability observed during the 2002 study was not significant. In six of the previous study years (1986, 1989, 1990, 1991, 1994, and 1999), significantly higher larval densities were observed in night samples; lobster larvae were never significantly more abundant in day samples. This observed diel variability was similar to results found elsewhere (review by Ennis 1995). 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 deeper, less turbid 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. More recently, DiBacco and Pringle (1992) found significantly more Stage I larvae during nightime than during daytime in neuston tows made within a protected coastal embayment along the Nova Scotian Shelf. Since the MPS intakes are sited on the shoreline, and draw water from throughout the 8-10 m well-mixed water column, the observed diel variability in our study may be influenced more by wind and tide generated water circulation than by vertical movement by larvae.

The Δ-mean density of lobster larvae collected in entrainment samples during 2002 was 0.603 per 1000 m³, which was within the range of values reported in our entrainment studies when the MPS Units were operational (0.299-2.502; Table 12). An estimated 243,200 lobster larvae were entrained through the MPS

TABLE 11. Annual Δ-mean density of lobster larvae collected in day and night entrainment samples from 1984 through 2002.

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	0.0-0.820
	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	0.0-1.653
	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	0.0-2.132
	Night	1.168	0.0-2.433
1994	Day	0.268	0.085-0.452
	Night	1.505[b]	0.706-2.303
1995	Day	0.594	0.0-1.310
	Night	2.189	0.369-4.009
1996	Day	0.329	0.0-2.057
	Night	0.242	0.0-1.215
1997	Day	0.244	0.0-0.671
	Night	0.122	0.0-0.281
1998	Day	0.500	0.0-1.257
	Night	0.502	0.0-1.150
1999	Day	0.799	0.330-1.268
	Night	4.497[b]	1.281-7.713
2000	Day	0.445	0.101-0.788
	Night	1.034	0.0-2.074
2001	Day	0.290	0.067-0.513
	Night	0.315	0.186-0.444
2002	Day	0.252	0.0-0.594
	Night	0.961	0.0-2.070

[a]Number per 1000 m³.
[b]Significant difference between day and night densities based on 2-sample t-tests (p<0.05).

TABLE 12. Annual Δ-mean density (number per 1000 m³) of lobster larvae in MPS entrainment samples during their season of occurrence and annual entrainment estimates with 95% C.I. from 1984 through 2002.

Year	Time period included	Number of larvae	Δ-mean density	95% C.I.	Cooling Vol. (m³ x 10⁶)	Entrainment estimate x 10³	95% C.I. x 10³
1984	21May-10Jul	102	0.409	0.184-0.635	181.8	74.4	33.5-115.4
1985	15May-16Jul	142	0.504	0.258-0.749	244.9	123.4	63.2-183.4
1986[a]	14May-14Jul	232	0.857	0.418-1.297	639.5	548.1	267.3-829.4
1987	18May-30Jul	184	0.943	0.274-1.613	406.8	383.6	111.5-656.2
1988	16May-01Aug	571	0.717	0.296-1.137	804.1	576.5	238.0-914.3
1989	22May-28Jul	237	0.701	0.358-1.044	540.3	378.8	193.4-564.1
1990	14May-30Jul	280	0.748	0.436-1.060	747.9	559.4	326.1-792.8
1991	07May-22Jul	157	0.525	0.365-0.685	541.6	284.3	197.7-371.0
1992	19May-14Jul	625	1.334	0.652-2.016	449.5	599.6	293.1-906.2
1993	24May-25Jun	218	1.081	0.273-1.889	346.1	374.1	94.5-653.8
1994	25May-04Aug	257	0.908	0.445-1.371	715.4	649.6	318.4-980.8
1995	30May-21Jul	254	1.385	0.470-2.300	476.1	659.4	223.8-1095.0
1996[b]	06Jun-01Jul	19	0.364	0.194-0.535	53.0	19.3	10.3-28.4
1997[b]	30May-16Jul	15	0.186	0-0.395	51.6	9.6	0.0-20.4
1998	20May-23Jul	145	0.511	0.048-0.973	244.5	124.9	11.7-237.9
1999	17May-02Jul	529	2.502	1.107-3.896	237.9	595.2	263.4-926.9
2000	17May-13Jul	155	0.725	0.276-1.173	451.5	327.3	124.6-529.6
2001	21May-02Aug	58	0.299	0.185-0.414	605.1	180.9	111.9-250.5
2002	14May-01Jul	105	0.603	0.092-1.115	403.4	243.2	37.1-449.8

[a] Unit 3 began commercial operation.
[b] All MPS units shutdown.

cooling water system during 2002; this estimate was within the range of previous studies when two or three units operated (74,400-659,400). The lowest entrainment occurred in 1996 and 1997 when MPS units were offline (19,300 and 9,600, respectively). Lobster larvae entrainment is directly related to both the annual larval density and the operational status of the MPS units during the hatching season. During years when lobster larvae abundance is high and units operate at full capacity, cooling-water demands are at a maximum and resulting entrainment estimates are higher. Conversely, entrainment estimates are low when lobster larvae abundance is low and one or more units are shutdown for maintenance or refueling.

Evaluating the effect of entrainment on lobster recruitment is difficult because of the high variability in lobster larvae abundance and stage composition (Bibb et al. 1983; Fogarty 1983; Lux et al. 1983; Blake 1984, 1988), the lack of reliable estimates of larval and post-larval survival rates (Phillips and Sastry 1980; Caddy and Campbell 1986; Cobb 1986; Blake 1991), and the uncertainty regarding post-settlement processes of early benthic phase lobsters controlling recruitment to the fishery (Hudon 1987; Incze and Wahle 1991; Wahle and Steneck 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). Convergence areas were often seen in the MPS area stretching from near Twotree Island into Niantic Bay in 1988 and 1992 and could explain the large number of lobster larvae (all stages) collected in single samples of the cooling water in those years (DNC 2002). Furthermore, based on the short duration of the first larval stage (3-5 days), the source of Stage I larvae collected in the MPS cooling-water was probably from local spawners (e.g., from Twotree, where 20.5% of females were berried during 2002). Stage IV larvae, however, are in the water column between 4 and 6 weeks, and, based on water circulation patterns in LIS, were unlikely to have originated locally. Lund and Stewart (1970) indicated that the large number of berried females found in western LIS compared to eastern LIS may be responsible for recruitment of Stage IV larvae throughout LIS. Stage IV larvae were found to exhibit directional swimming behavior and moved tens of

kilometers from offshore spawning grounds (Cobb et al. 1989; Rooney and Cobb 1991; Katz et al. 1994). The lobster mortality event in western LIS could potentially impact recruitment in the vicinity of MPS, if larvae found in our area originate from the west. A similar 'source/sink' recruitment model was hypothesized by Fogarty (1998); he attributed the stability and resilience of overfished coastal lobster stocks to migration of adults and advection of larvae to coastal waters from less intensively fished areas far from shore on the edge of the continental shelf.

Mortality and Shell Disease

Since 1984, we recorded the presence of dead or dying lobsters caught in our traps during each sampling trip. In addition, we qualitatively assessed external damage to the carapace and abdomen and noted the presence of shell disease (chitinoclasia). Of the 4,829 lobsters collected during the 2002 study, 159 (3.3%) were found dead in the traps or died during the 2 to 3 day holding period in our tanks. More males were found dead (107) than females (49) or egg-bearing females (3; Table 13). Since 1984, the total number of dead or dying male lobsters was 1,464, which was more than double the total number of females (n=671). Since male lobsters predominate in our catches (see Table 5; Sex Ratios), it was not unusual to find more dead males in the traps. The cause of the recent increase in lobster mortality in our area and in western LIS is unclear. Pathologists have examined lobsters from western LIS at the request of NYSDEC and CTDEP but have been unable to identify the cause of the die-offs. Comprehensive examinations of water quality parameters and bottom sediments failed to identify environmental conditions or toxins that could explain the lobster deaths. More recently, histologic examinations of lobsters revealed a systemic inflammatory disease affecting multiple tissues. The nervous system was markedly inflamed and the lesions were associated with a parasitic amoeboid protozoan (French et al. 2000, 2001). At present, the cause of death is hypothesized to be paramoebiasis. Most researchers agree that the die-off in western LIS was caused by multiple disease pathogens exacerbated by environmental stressors, such as increased seawater temperature, anoxia and contaminants. Stewart (1980) indicated that disease and parasitism are dominant sources of natural mortality in wild lobster populations. Mass mortality caused by fatal diseases such as gaffkemia (Snieszko and Taylor 1947) have been reported for lobsters reared in high densities, but little is known about the prevalence and effects of pathogens on natural lobster populations (Martin and Hose 1995).

TABLE 13. Incidence of dead or dying lobsters caught in traps from May through October 1984-2002.

Year	Total collected	Number dead	%	Dead males	Dead females[a]
1984	7587	58	0.8	33	24 (1)
1985	7014	35	0.5	16	19
1986	7211	63	0.9	34	29
1987	7280	52	0.7	33	19
1988	8871	68	0.8	35	33
1989	7950	62	0.8	37	25
1990	7106	72	1.0	48	22 (2)
1991	7597	82	1.1	50	31 (1)
1992	11438	117	1.0	84	33
1993	10195	94	0.9	58	35 (1)
1994	9849	67	0.7	36	31
1995	6435	70	1.1	50	20
1996	7531	256	3.4	200	55 (1)
1997	9026	117	1.3	101	16
1998	10991	104	0.9	79	25
1999	11216	302	2.7	210	89 (3)
2000	8707	157	1.8	111	43 (3)
2001	7268	200	2.8	142	55 (3)
2002	4829	159	3.3	107	49 (3)

[a] Parenthetical values are number berried.

Lobster mortalities in the area around MPS may be related to shell disease, a more common affliction in marine and freshwater crustaceans. Shell disease is characterized as a deterioration of the exoskeleton by chitinoclastic microorganisms (Rosen 1970; Sinderman 1970). Gross signs of the disease are similar in all crustacean species; the exoskeleton is pitted and marred with necrotic lesions and, although the disease is not immediately fatal, death may occur (Fisher et al. 1978). The unsightly appearance of the lobster shell can greatly affect marketability. Chitin-digesting bacteria and fungi have been implicated as causative agents and include isolates of *Vibrio* spp., and several other Gram-negative bacilli (Hess 1937; Getchell 1989). From 1984 to 1997 only a few lobsters (n=7, <0.1%) were found with signs of shell disease. In the fall of 1998, 156 (6.5%) lobsters had signs of minor shell disease. The incidence of shell disease increased markedly in 1999; with 24% and 38% of the total catch infected during September and October, respectively (Fig. 17a). In spring 2000, lobster biologists from southern New England and New York developed standardized procedures for monitoring the extent of shell disease in wild populations. The following index was established based on the percent shell coverage of disease symptoms (e.g., pitting, erosion, lesions) on the total surface area of the lobster: 0=no shell disease symptoms, 1=symptoms on 1-10% of shell surface, 2=symptoms on 11-50% of shell surface, and 3=symptoms on >50% of shell surface. During the past three years, the incidence of shell disease (all severity indices) closely corresponded to the molt cycle observed from May through October. Prior to the molt
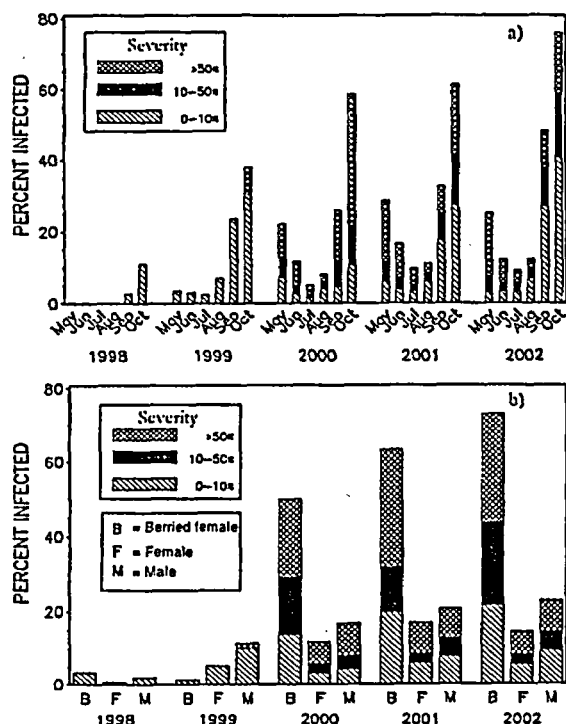
in May 20% to 30% of the catch had shell disease (Fig. 17a). Following the major molt in July, only about 5% of the catch was afflicted. The severity and percentage of diseased lobsters increased in August and September, reaching a peak in October when between 60% and 80% of the catch had symptoms of shell disease. In comparison, prevalence of shell disease in central and western LIS and in offshore canyon areas has never exceeded 5% (Landers et al. 2001a). In eastern LIS and other coastal areas of southern New England, male and female lobsters of all sizes have been observed with shell disease, although larger sized individuals and egg-bearing females had higher incidence and severity of shell disease symptoms. Between 50% and 60% of the egg-bearing females had shell disease during 2000 and 2001; the percentage increased to nearly 80% in 2002, which was considerably higher than the percentages observed in non-berried females and males (2000-2002 range=12-23%; Fig 17b). In nearshore waters of Rhode Island over 50% of the egg-bearing females observed in the trap fishery were infected with shell disease in 1999 (Castro and Angell 2000). This is most likely due to the molt cycle of large lobsters and egg-bearing females. Smaller lobsters molt more frequently (1-2 times/yr or more) and shed their shells before severe shell disease symptoms occur. Large lobsters and egg-

bearing females experience more shell deterioration because they may only molt every 2 years.

The annual percentages of dead and diseased lobsters were highly correlated (r=0.71; p<0.001; Fig. 18). During the 10-year period from 1984 to 1995, <1% of the catch was found dead and no shell disease was observed. Over the past 6 years the percentage of dead lobsters found in traps increased up to 3% and the incidence of shell disease increased each year reaching peak levels of 22% in 2002. Average bottom water temperature measured at the three lobster sampling stations from May through October has steadily increased during the 1984-2002 period. The rise in seawater temperature since 1984 was positively correlated with the increase in shell disease (r=0.49; p=0.034), but not with the increase in dead lobsters (r=0.36; p=0.124).

A number of isolated outbreaks of shell disease have been reported in lobster populations along the New England coast in the past century. However, these reports were limited to impounded lobsters in the Gulf of Maine during the 1930s (Hess 1937; Taylor 1948) and to infrequent occurrences in wild lobsters in the 1980s along Massachusetts coastal waters and the New York Bight (Sinderman et al. 1989; Estrella 1991). The present epizootic is clearly different from any other cases reported for American lobster. The cause of the outbreak is unknown; it may be due to water quality degradation along the coast or to natural environmental factors such as warmer seawater temperature. However, it is surprising that the prevalence of shell disease was low in central and western LIS, two areas known to be more polluted with domestic sewage and industrial contaminants than areas to the east. The etiology of shell disease may be enhanced by increased transmission due to crowding as a result of the sharply higher abundance of lobsters in the late 1990s. Most alarming is the prevalence and severity of shell disease in egg-bearing female lobsters. The effect of the disease on lobster recruitment in southern New England may be significant if egg-bearing females suffer higher natural mortality due to shell disease.

Conclusions

The catastrophic die-off observed in western LIS was not observed in our study area in eastern LIS during 2002. However, the abundance of lobsters (CPUE) in the MPS area declined during 2002 to the lowest levels observed in our 25-year monitoring program. The lobster mortality event in western LIS could be responsible for the declining catch and may affect catches in the future, if lobster recruitment in our area

Fig. 17. Percentage and severity of shell diseased lobsters collected in the vicinity of MNPS from 1998 to 2002 (a) by month and (b) by sex.
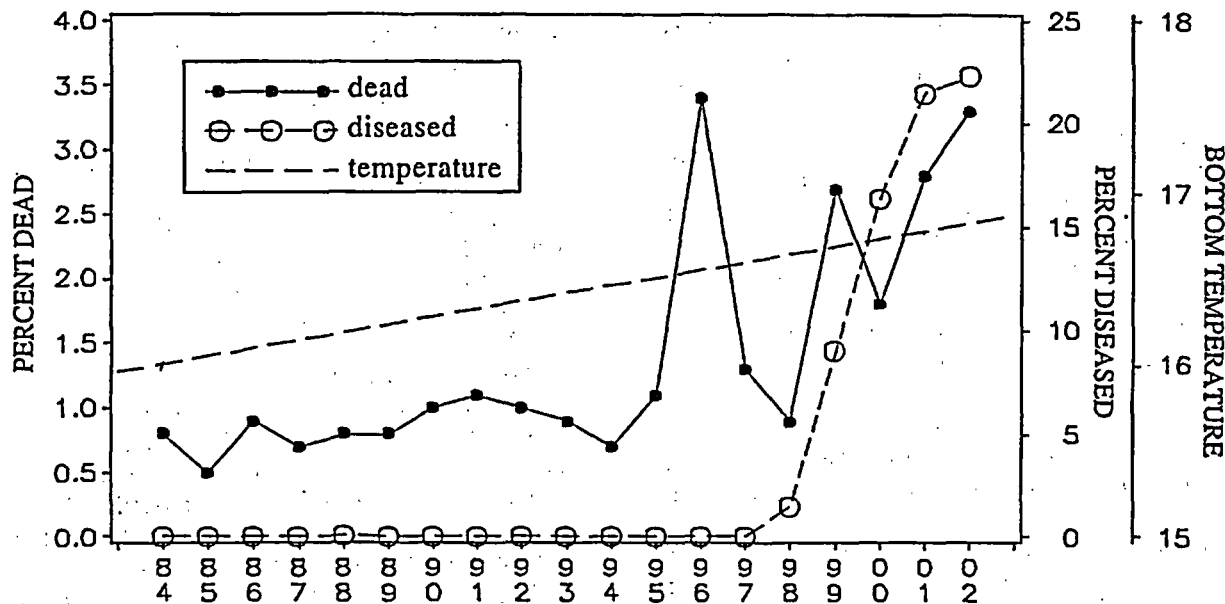
Figure 18. Annual percentages of dead and diseased lobsters and linear trend of bottom water temperatures from May through October at the three MPS lobster stations during the period from 1984 to 2002.

depends on larval production in western LIS. This source-sink relationship between western and eastern LIS and the long-term sustainability of the Connecticut fishery may be further threatened by the widespread outbreak of shell disease in our area over the past four years.

The most notable changes in the population characteristics of local lobsters during 2002 were observed in the proportion of egg-bearing females and their size structure, which may be related to changes in lobster growth and the size at which females become sexually mature. The benefits of earlier maturation may be significant to population growth. Small size at maturity and subsequent egg production from sublegal-sized females may explain why LIS lobsters have been so resilient to high levels of fishing pressure. Currently, the smaller size at sexual maturity allows females to spawn 1 or 2 times before reaching the legal size. However, these benefits may now be offset by the recent outbreak of shell disease. We found that egg-bearing females had the highest prevalence and severity of shell disease, but the effects of the disease on egg-development, hatching success and larval survival are unknown. The events of mass mortality in western LIS and the effects on recruitment in our area due to the recent outbreak of shell disease could be significant, if berried females suffer higher natural mortality.

The die-off in western LIS, the outbreak of shell disease in eastern LIS and changes noted in lobster growth and size at maturity coincided with natural increases in seawater temperature over the past 2-decades. Since LIS is the southern extreme of the

American lobsters range in nearshore waters, a gradual rise in seawater temperature may have profound effects on the biology and physiology of this species. Results from our lobster monitoring program over the past 25 years, suggest that operation of the MPS cooling water system since the early 1970s has not caused a decrease in the local lobster population; however, it is unclear what impacts the power plant may have (if any) on the local lobsters when abundance levels are low. Recent changes in population characteristics and long-term fluctuations in abundance were related more to changes in environmental conditions, most notably seawater temperature, or to other factors mediated by temperature such as increased susceptibility and transmission of diseases. Continued monitoring will help determine if operation of MPS causes changes in the abundance and population characteristics of local lobsters beyond those expected from natural factors such as elevated temperature, disease, and the intense exploitation of this species in coastal waters.

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Rocky Intertidal Studies

Rocky Intertidal Studies

Introduction

Shore habitats in New England, including those near the discharge point of Millstone Power Station (MPS), are often comprised of boulder and exposed bedrock ledge substrata, which support rich and diverse communities of attached algae and animals. These communities are important and productive components of coastal ecosystems. Given their ecological importance and vulnerability to thermal plume impacts, studies of rocky shore communities are often included in ecological monitoring programs designed to assess the impacts of shore-sited power plants (Vadas *et al.* 1976, 1978; Wilce *et al.* 1978; NAESCO 1994; NAI 1999; NUSCO 1999; DNC 2002).

Rocky intertidal studies at MPS are part of a comprehensive environmental monitoring program whose primary objective is to determine whether temporal and spatial differences (*e.g.*, in abundance, distribution or species composition) among communities at several sites in the Millstone Point area can be attributed to construction and operation of MPS. To achieve this objective, studies were designed and implemented to identify attached algal 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 such communities. This research includes qualitative algal sampling, abundance (percentage cover) measurements of intertidal algae and invertebrates, and growth and mortality studies of the brown macroalga, *Ascophyllum nodosum*. The following report discusses results of sampling and analysis in the most recent study year and compares these results to those of previous years, with particular attention to events related to construction and operation of MPS that may have altered environmental conditions on nearby shores.

Materials and Methods

Qualitative Algal 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. A year of qualitative sampling is determined to be from March to the following January,

both months inclusive. The latest year of qualitative algal data (2002) comprises collections from March 2002 to January 2003.

The FE station, approximately 100 m east of the MPS discharges, is directly exposed to the 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 MPS 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 MPS 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 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) and Sears (1998).

Abundance Measurement

Abundance of rocky intertidal organisms was expressed as a percentage of substratum cover. At each qualitative collection station, five permanent strip transects 0.5 m wide were established perpendicular to the water-line, 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 latest year for abundance measurement data is the same as that described for qualitative algal sampling data. The total number of quadrats in each transect depended on the transect slope. The percentage of substratum cover of all organisms and remaining free space in each quadrat was subjectively determined. Understory organisms, *i.e.*, 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
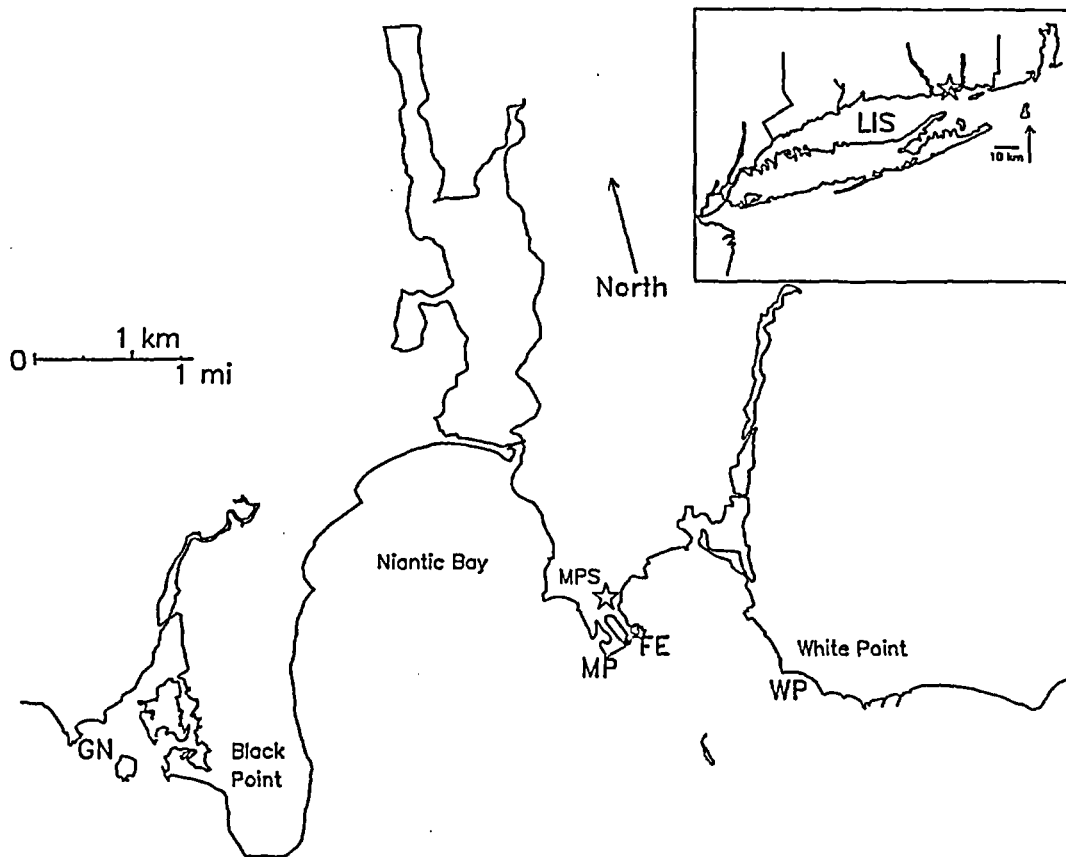
Fig. 1.   Location of the MPS rocky intertidal sampling sites: GN=Giants Neck, MP=Millstone Point, FE=Fox Island-Exposed, WP=White Point.
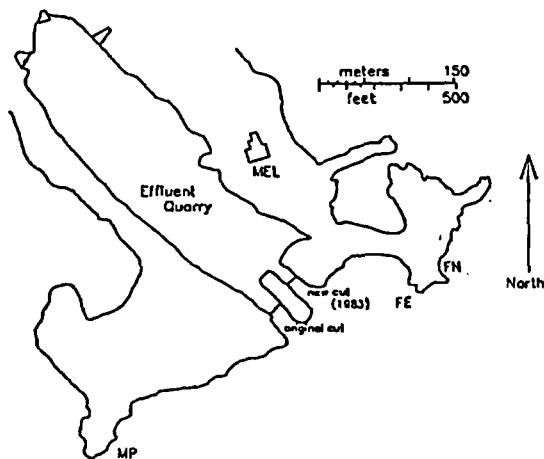
Fig. 2.   Detail map of the MPS vicinity: FN=new experimental *Ascophyllum* site (1985-present); MP and FE as in Fig. 1.

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 and WP mentioned previously; Fig. 1) and a potentially impacted station (FN, about 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 1987).

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 tips were measured on 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 a year). 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 of 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 ($\beta = -0.25$), clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967). Additionally, multi-dimensional scaling ordination (MDS) was used to create a 2-dimensional representation of comparisons of annual collections. This analysis was performed using a group-averaging algorithm in the PRIMER suite of programs developed by the Plymouth Marine Laboratory (Warwick and Clarke 1991; Clarke 1993).

Quantitative analyses included determination of abundance of intertidal organisms as percentage of substratum covered by each taxon. Substratum not occupied by macrobiota was 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. Comparison of station/year collections was done using the same MDS techniques described above for qualitative algal analyses.

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. The first derivative of the Gompertz function represents an instantaneous growth rate, with a maximum value occurring at t_0. Growth data representing the latest growing season (2001-2002) were plotted for all stations together and for each station separately, with summaries of previous years' data. *Ascophyllum* mortality was presented as loss of fronds and tips over time.

Results and Discussion

Qualitative Algal Studies

Water temperature is frequently implicated as a critical environmental factor in determining macroalgal species occurrence and distribution (Hoek 1982, 1984; Breeman 1988; Lüning 1990), and is often an important regulatory cue for algal life cycles (Lüning 1980; Swenarton 1997). Macroalgal communities in the vicinity of MPS are exposed to elevated water temperatures resulting from the thermal effluent discharge, and therefore, alterations of spatial and temporal patterns of 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, by station. The total number of species identified in 2002 was 105. This total was within the range of annual totals for previous study years (81-111). One new taxon, the brown alga *Dictyosiphon chordaria*, was added to our species list in 2002. Since 1979, 154 algal species have been collected in odd-numbered months at the current four sampling sites; this compares with 161 species reported in the last year that nine stations were sampled (until 1995; see NUSCO 1996). Of the 'lost' species, only *Laminaria digitata* had occurred as more than a trace component of our flora. *L. digitata* had been relatively common, but only at the Twotree Island (TT) sampling site, where sampling has been suspended since 1995.

TABLE 1. Qualitative algal collections (Mar. 1979 - Jan. 2003) by month, and by station. Values represent number of times found, as a percentage of possible times found. A dash before a species name indicates that it was collected 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 details. The FE, by group columns refer to the dendrogram groupings in Fig. 3; the unimpacted station average is the mean of GN, MP and WP ('T'=present, but <1%).

| | by month | | | | | | by station | | | | FE, by group | | | | | | | unimpact. |
Rhodophyta	J	M	M	J	S	N	FE	GN	MP	WP	fe1	fe2	fe3	fe4	fe5	fe6	fe7	sta. avg.
Stylonema alsidii	4	2	0	6	19	5	8	8	0	9	0	33	8	0	0	11	0	6
-Erythrotrichopeltis ciliaris	25	11	11	12	28	31	26	25	5	21	13	75	25	58	0	26	6	17
-Erythrotrichia carnea	10	4	3	4	13	8	8	13	2	4	0	0	0	0	17	15	11	6
Erythrocladia subintegra	1	0	0	0	1	3	2	0	2	0	0	8	0	0	0	4	0	1
Erythropeltis discigera	2	1	0	1	4	6	6	1	1	1	0	8	33	8	0	4	6	1
-Bangia atropurpurea	74	82	33	10	26	48	44	43	49	47	38	33	25	50	50	50	44	46
-Porphyra leucosticta	72	78	61	17	10	31	44	46	52	40	29	33	33	67	42	46	56	46
-Porphyra carolinensis	8	5	3	1	4	6	12	0	6	1	0	0	0	0	0	4	83	2
-Porphyra umbilicalis	56	76	86	42	25	39	61	43	62	51	71	33	33	100	67	70	28	52
-Porphyra linearis	15	8	3	0	0	0	2	1	11	4	0	0	0	0	17	0	6	5
Audouinella purpurea	3	1	2	1	1	2	5	1	1	1	25	0	0	0	0	0	6	1
-Audouinella secundata	31	20	20	16	18	15	22	20	22	17	21	33	8	8	17	28	22	20
Audouinella daviesii	6	2	2	4	4	5	7	4	2	3	4	0	0	8	0	4	33	3
-Audouinella saviana	10	13	14	4	11	14	14	13	6	10	4	25	25	17	0	15	17	10
Audouinella sp.	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	T
Audouinella dasyae	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	T
-Gelidium pusillum	40	33	29	34	35	43	45	65	1	28	0	0	0	0	92	67	100	31
Nemalion helminthoides	0	0	0	5	0	0	0	0	4	0	0	0	0	0	0	0	0	1
-Bonnemaisonia hamifera	1	4	9	12	0	1	0	1	1	16	0	0	0	0	0	0	0	6
'Traillliella intricata'	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	T
-Agardhiella subulata	25	18	12	22	27	28	65	3	3	14	17	25	75	67	50	87	89	7
Polyides rotundus	3	2	6	12	9	6	6	2	5	13	8	0	0	0	17	7	6	6
-Cystoclonium purpureum	73	66	74	44	19	53	24	67	61	68	79	42	0	0	33	6	22	65
-Gracilaria tikvahiae	15	8	2	2	14	12	33	0	0	1	0	8	0	25	33	50	72	T
-Ahnfeltia plicata	37	38	34	41	30	33	30	12	57	46	92	67	0	0	50	9	11	38
-Phyllophora pseudoceranoides	19	11	4	9	5	12	5	9	6	20	17	0	0	0	8	4	0	12
-Coccotylus truncatus	8	14	11	5	6	10	2	5	6	23	8	0	0	8	0	0	0	11
-Chondrus crispus	97	97	97	98	97	97	88	100	100	100	100	75	0	83	100	100	100	100
-Mastocarpus stellatus	57	54	52	53	52	62	6	31	98	91	21	25	0	0	0	0	0	73
-Hypnea musciformis	2	1	1	1	2	2	6	0	0	0	0	0	0	0	0	0	50	0
Rhodophysema georgii	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	T
-Corallina officinalis	76	72	71	71	73	74	97	5	99	94	96	100	92	100	100	96	100	66
-Dumontia contorta	22	67	69	6	1	1	8	42	26	33	33	25	0	0	8	0	0	34
Gloiosiphonia capillaris	1	2	6	0	0	0	5	0	1	1	4	0	0	0	17	7	0	T
-Choreocolax polysiphoniae	9	13	6	8	3	3	1	19	6	1	8	0	0	0	0	0	0	9
Hildenbrandia rubra	0	0	0	1	1	0	0	1	0	1	0	0	0	0	0	0	0	T
-Palmaria palmata	18	29	26	24	6	12	6	21	22	28	13	8	0	8	0	4	6	24
-Champia parvula	28	12	6	59	77	62	41	32	33	56	21	58	33	50	25	43	61	41
-Lomentaria baileyana	2	0	0	9	46	6	15	13	1	13	8	33	25	25	0	11	17	9
-Lomentaria clavellosa	6	11	9	2	2	3	1	6	3	12	0	8	0	0	0	0	6	7
Lomentaria orcadensis	1	1	1	0	5	0	1	1	0	3	0	0	0	0	0	2	6	1
-Antithamnion cruciatum	24	2	9	46	39	38	18	25	20	41	33	25	33	17	0	17	0	29
-Antithamnion pectinatum	62	34	27	46	65	63	50	38	67	46	0	0	0	58	42	85	78	50
Callithamnion corymbosum	0	0	0	0	2	2	1	0	0	2	0	0	0	0	0	2	0	1
-Aglaothamnion roseum	4	2	1	8	26	17	22	6	6	6	33	0	8	33	0	33	0	6
-Callithamnion tetragonum	35	19	10	11	17	30	15	18	29	22	63	42	0	8	0	0	0	23
-Aglaothamnion byssoides	0	0	0	5	0	0	1	2	0	0	0	0	0	0	0	2	6	1
-'Callithamnion baileyi'	23	9	4	24	24	38	3	27	32	19	0	0	0	0	8	7	0	26
-Ceramium deslongchampii	2	0	1	1	8	3	0	7	0	3	0	0	0	0	0	0	0	3
-Ceramium diaphanum	1	0	2	31	40	11	10	12	10	24	4	0	8	0	8	13	28	15
-Ceramium rubrum	83	84	86	89	83	84	70	91	83	95	100	92	42	83	50	74	28	90
Ceramium fastigiatum	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	T
-Spermothamnion repens	55	33	27	45	49	61	13	56	45	67	21	17	0	0	8	15	11	56
-Spyridia filamentosa	1	0	0	6	24	8	0	19	1	5	0	0	0	0	0	0	0	8

TABLE 1. (cont.)

Rhodophyta	J	M	M	J	S	N	FE	GN	MP	WP	fe1	fe2	fe3	fe4	fe5	fe6	fe7	unimpact. sta. avg.
			by month					by station					FE, by group					unimpact.
Scagelia pylaisaei	1	0	2	1	2	2	1	1	1	3	0	0	0	8	0	0	0	2
Griffithsia globulifera	0	0	0	1	8	0	1	0	0	5	0	0	0	0	0	0	6	2
-Grinnellia americana	3	1	0	8	14	20	16	2	0	12	4	0	8	8	0	30	22	5
Phycodrys rubens	0	3	6	2	3	2	1	1	0	10	4	0	0	0	0	0	0	3
-Dasya baillouviana	15	0	0	22	52	40	33	22	10	19	4	25	25	42	25	46	39	17
Chondria sedifolia	0	0	0	3	8	0	1	4	1	1	0	0	0	0	0	2	6	2
-Chondria baileyana	1	1	1	3	18	1	3	8	6	1	0	0	0	0	0	2	17	5
Chondria capillaris	0	0	0	1	4	0	0	3	0	0	0	0	0	0	0	0	0	1
-Polysiphonia denudata	5	0	2	3	3	4	9	0	1	2	4	0	0	0	8	13	22	1
-Polysiphonia harveyi	97	58	55	98	98	100	94	74	87	82	71	92	100	100	100	100	100	81
-Polysiphonia lanosa	77	73	74	71	71	71	13	90	100	92	63	33	0	0	0	0	0	94
-Polysiphonia nigra	6	12	24	0	1	3	9	6	1	15	0	0	0	0	25	15	11	7
-Polysiphonia fucoides	28	19	22	31	33	24	14	26	5	57	25	0	0	17	17	13	17	29
-Polysiphonia stricta	34	49	58	13	6	6	14	29	28	41	46	0	0	0	33	2	22	33
-Polysiphonia elongata	1	2	2	0	1	3	0	0	1	6	0	0	0	0	0	0	0	2
Polysiphonia fibrillosa	2	0	0	1	1	0	1	1	1	0	4	0	0	0	0	0	0	1
-Polysiphonia flexicaulis	5	0	0	2	0	3	2	1	1	3	0	0	0	8	0	0	11	2
-Rhodomela confervoides	9	15	8	0	0	0	3	3	10	6	13	0	0	0	8	0	0	6

Phaeophyta	J	M	M	J	S	N	FE	GN	MP	WP	fe1	fe2	fe3	fe4	fe5	fe6	fe7	unimpact. sta. avg.
			by month					by station					FE, by group					unimpact.
-Ectocarpus fasciculatus	3	13	32	10	19	19	11	14	21	19	21	33	17	8	25	2	0	18
-Ectocarpus siliculosis	25	49	51	43	33	30	26	49	32	47	50	42	0	8	8	28	17	43
Ectocarpus sp.	2	3	0	2	2	0	1	1	3	1	4	8	0	0	0	0	0	2
-Hincksia granulosa	3	9	10	2	3	4	8	4	4	5	0	0	17	0	0	15	6	4
-Hincksia mitchelliae	17	10	22	18	42	32	42	18	10	22	29	50	50	50	25	41	61	17
-Pilayella littoralis	15	18	37	11	10	9	3	43	2	16	13	0	0	8	0	0	0	20
-Spongonema tomentosum	10	32	20	0	0	1	12	11	10	9	17	17	0	8	8	13	11	10
-Acinetospora sp.	0	0	1	0	2	0	0	1	0	1	0	0	0	0	0	0	0	1
-Ralfsia verrucosa	53	43	41	59	62	57	24	60	52	74	63	58	50	17	0	7	0	62
-Elachista fucicola	67	75	83	83	72	61	67	77	74	76	79	50	0	58	100	67	89	76
Halothrix lumbricalis	1	3	3	2	2	0	0	3	2	3	0	0	0	0	0	0	0	3
-Leathesia difformis	0	0	51	53	0	0	9	20	22	18	25	0	17	8	8	4	6	20
-Chordaria flagelliformis	0	1	31	31	18	2	5	8	29	16	8	0	0	8	17	4	0	18
Sphaerotrichia divaricata	0	0	2	0	0	0	0	0	1	1	0	0	0	0	0	0	0	T
Eudesme virescens	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	T
-Pogotrichum filiforme	0	3	2	0	1	1	0	1	1	3	0	0	0	0	0	0	0	2
Punctaria tenuissima	3	9	3	2	0	0	2	3	0	6	0	0	0	0	8	4	0	3
Phaeosaccion collinsii	0	2	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	T
-Punctaria latifolia	8	12	6	1	0	3	8	3	2	6	4	0	0	8	17	11	11	4
-Punctaria plantaginea	2	4	3	6	2	2	1	7	2	3	0	0	0	0	0	0	11	4
-Petalonia fascia	81	82	89	41	3	42	51	56	52	65	58	50	50	42	83	43	50	58
-Scytosiphon lomentaria	34	94	94	60	0	12	43	52	48	53	46	42	50	50	58	39	33	51
Dictyosiphon chordaria	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	T
Desmarestia aculeata	2	5	9	8	2	10	1	6	2	14	4	0	0	0	8	0	0	7
-Desmarestia viridis	0	30	45	2	0	0	3	12	17	20	8	0	0	0	8	4	0	16
Chorda filum	0	0	8	12	0	0	1	1	1	10	0	0	0	0	0	2	0	4
Halosiphon tomentosus	0	5	10	3	0	0	1	0	3	8	8	0	0	0	0	0	0	4
-Laminaria longicruris	15	6	13	9	10	11	1	8	13	21	0	0	0	0	0	2	0	14
-Laminaria saccharina	48	60	81	87	51	52	47	69	69	69	79	33	17	58	67	37	39	69
-Sphacelaria cirrosa	62	38	45	42	48	59	76	63	25	30	79	25	33	92	75	89	83	39
Sphacelaria rigidula	0	0	0	0	1	2	0	1	0	1	0	0	0	0	0	0	0	1
-Ascophyllum nodosum	90	94	94	94	90	90	69	100	100	100	100	75	0	0	83	94	28	100
-Fucus distichus s edentatus	12	16	10	3	3	1	5	2	23	2	21	0	0	8	0	0	6	9
Fucus distichus s evanescens	2	12	9	2	1	2	3	5	8	3	17	0	0	0	0	0	0	5
-Fucus spiralis	10	5	8	13	13	6	0	4	34	1	0	0	0	0	0	0	0	13
-Fucus vesiculosus	98	99	99	100	98	98	94	100	100	100	100	75	58	100	100	100	100	100
-Sargassum filipendula	14	13	12	13	17	15	52	1	2	0	0	0	0	25	17	98	94	1

TABLE 1. (cont.)

Chlorophyta	by month						by station				FE, by group							unimpact.
	J	M	M	J	S	N	FE	GN	MP	WP	fe1	fe2	fe3	fe4	fe5	fe6	fe7	sta. avg.
-Ulothrix flacca	48	57	28	4	5	20	22	31	28	28	54	17	8	33	17	11	22	29
-Urospora penicilliformis	69	66	18	2	4	33	30	29	33	37	38	17	33	67	33	22	22	33
Urospora wormskjoldii	17	16	10	8	2	5	17	6	8	8	13	0	8	58	25	19	0	7
'Urospora collabens'	3	2	2	2	0	0	3	1	1	1	0	8	17	8	0	0	0	1
Acrochaete viridis	1	0	0	1	0	0	0	0	1	1	0	0	0	0	0	0	0	T
-Monostroma grevillei	8	49	37	0	1	2	8	20	18	18	21	25	17	0	8	2	0	19
-Protomonostroma undulatum	6	75	71	1	0	1	14	31	28	31	33	25	0	17	25	7	0	30
Monostroma oxysperma	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	T
-Spongomorpha arcta	8	42	65	6	0	1	11	20	30	21	33	0	0	25	25	4	0	24
-Spongomorpha aeruginosa	2	4	51	2	1	0	3	10	17	10	13	0	8	0	8	0	0	13
'Codiolum gregarium'	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	T
Capsosiphon fulvescens	0	0	1	1	1	0	0	1	0	1	0	0	0	0	0	0	0	1
Capsosiphon groenlandicum	0	3	0	1	0	1	1	1	2	1	4	0	0	0	0	0	0	1
-Blidingia minima	75	66	78	71	72	60	79	56	79	68	75	67	67	75	83	91	67	68
Blidingia marginata	1	0	0	4	0	1	1	2	0	1	8	0	0	0	0	0	0	1
-Enteromorpha clathrata	3	1	8	27	38	9	14	16	4	22	25	33	33	17	0	6	6	14
-Enteromorpha flexuosa	58	54	56	54	69	69	76	51	48	62	63	50	75	92	58	85	89	54
-Enteromorpha intestinalis	15	26	32	46	29	13	20	29	23	35	38	25	67	8	0	13	6	29
-Enteromorpha linza	75	71	86	84	86	78	85	68	88	81	75	50	42	92	100	96	100	79
-Enteromorpha prolifera	18	16	23	18	28	23	14	27	10	32	25	33	25	17	8	6	6	23
Enteromorpha torta	1	0	3	4	0	0	0	3	0	3	0	0	0	0	0	0	0	2
Enteromorpha ralfsii	0	0	0	8	3	0	1	3	0	2	4	0	0	0	0	2	0	2
Percursaria percursa	0	0	1	1	2	0	0	1	0	2	0	0	0	0	0	0	0	1
-Ulva lactuca	97	83	94	97	97	98	90	95	96	95	100	100	67	100	100	85	89	95
-Prasiola stipitata	15	15	15	20	16	17	1	63	1	0	0	0	0	0	0	2	0	21
-Chaetomorpha linum	38	18	33	81	74	65	30	56	63	58	88	67	8	17	17	13	11	59
Chaetomorpha melagonium	0	0	3	0	0	0	1	1	1	0	0	0	0	0	0	0	6	T
-Chaetomorpha aerea	70	46	51	55	57	61	77	35	52	61	63	75	58	83	83	78	100	50
-Cladophora albida	0	1	8	13	11	0	3	6	4	8	13	0	0	0	0	2	6	6
-'Cladophora flexuosa'	11	2	19	46	29	12	18	14	25	24	0	25	42	33	25	13	22	21
'Cladophora glaucescens'	0	0	1	1	0	0	0	0	1	1	0	0	0	0	0	0	0	T
Cladophora laetevirens	0	0	0	1	1	0	0	0	1	1	0	0	0	0	0	0	0	T
-'Cladophora refracta'	8	3	8	10	12	5	8	3	7	12	25	8	0	17	0	6	0	7
-Cladophora sericea	11	17	52	46	38	26	35	28	27	35	46	33	58	25	33	26	44	30
'Cladophora crystallina'	0	0	0	0	2	0	0	1	0	1	0	0	0	0	0	0	0	T
-Cladophora hutchinsiae	6	3	5	5	5	11	6	7	7	5	8	25	0	0	0	4	6	6
-Cladophora rupestris	1	2	6	13	13	5	10	3	10	4	4	0	0	0	0	15	28	6
-Cladophora ruchingeri	1	1	4	22	15	5	15	3	4	10	0	0	8	8	8	24	28	6
-Rhizoclonium riparium	8	13	18	34	19	5	10	33	4	17	25	17	8	0	0	7	6	18
'Rhizoclonium kerneri'	0	2	1	1	0	0	0	2	0	1	0	0	0	0	0	0	0	1
'Rhizoclonium tortuosum'	0	0	0	0	0	1	1	0	0	0	4	0	0	0	0	0	0	0
-Bryopsis plumosa	12	0	2	5	13	14	17	4	2	6	21	17	33	25	0	17	11	4
-'Bryopsis hypnoides'	5	0	8	22	9	10	13	9	4	8	0	17	0	8	0	24	17	7
Derbesia marina	4	2	1	2	1	4	8	1	0	1	33	17	0	8	0	0	0	1
-Codium fragile	92	84	84	91	95	91	99	78	89	92	100	100	100	100	100	98	100	86

Because elevated temperatures may cause temporal shifts in occurrence of seasonal species, identification of components of the flora which exhibit natural seasonality provide a baseline from which power plant-induced changes can be assessed. A characteristic suite of species typical of cold-water period (January-May) collections in the Millstone area includes *Dumontia contorta, Polysiphonia stricta, Spongonema tomentosum, Desmarestia viridis, Halosiphon tomentosus, 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, Aglaothamnion roseum, Ceramium diaphanum, Grinnellia americana, Dasya baillouviana, Polysiphonia harveyi, Hincksia mitchelliae, Enteromorpha clathrata, Chaetomorpha linum, Bryopsis plumosa* and *B. hypnoides*.

Shifts in natural occurrence patterns related to thermal plume exposure (*i.e.*, 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 species frequencies at stations potentially exposed to the thermal plume to the frequencies at other stations beyond its influence. 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 local winter/spring flora and were found in collections at unimpacted stations (GN, MP and WP) an average of 34% and 33% of the time, respectively (Table 1). However at FE, these two species were only found in 8% and 14% of all collections, and most of these specimens were found prior to thermal plume effects caused by the opening of the second quarry cut in 1983, *i.e.*, in group fe1 described below. Other cold-water species (*Desmarestia viridis, Monostroma grevillei, Protomonostroma undulatum* and *Spongomorpha arcta*) occurred occasionally at FE, but much less frequently than at the other three stations. It is important to note that all these cold-water species were found at FE during the 1996-98 station shutdown period, presumably because of the ambient temperature conditions in the absence of cooling water discharge. By contrast, a number of warm-water seasonal species were more common at FE than at other sites. For example, *Aglaothamnion roseum* and *Hincksia mitchelliae* occurred in 22% and 42%, respectively, of the collections at FE, but in only 6% and 17% of collections at unimpacted stations, respectively. Other warm-water seasonals found with higher frequency at FE included *Grinnellia americana, Dasya baillouviana,* and *Bryopsis hypnoides*. These species were less common at FE

during the shutdown period, which we attribute to the lack of thermal addition to FE discussed above.

Several perennial species exhibited shifts in occurrence patterns at FE. We documented the establishment of populations of species with geographical distributions that extend into warm temperate and tropical regions, and are therefore tolerant of elevated temperature regimes at FE (*e.g., Gracilaria tikvahiae, Agardhiella subulata, Sargassum filipendula,* and most recently, *Hypnea musciformis*; Taylor 1957; Lüning 1990). Similarly, some species near the southern limit of their normal geographical ranges, such as *Mastocarpus stellatus* and *Polysiphonia lanosa*, experienced elimination at FE either directly, from exposure the MPS thermal plume, or indirectly, from reduction of suitable substrata.

Cluster 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 (Fig. 3a-c) separate into early and later sampling years, with the point of separation around 1986-88. This separation was influenced by the increasing contribution in recent years of species like *Antithamnion pectinatum*, an introduced species (Verlaque and Riouall 1989; Villalard-Bohnsack 1995). This species was first observed in the MPS area in 1986 and now common at all study sites (Foertch *et al.* 1995). Additionally, *Gelidium pusillum* has been considerably more abundant during 3-unit operation at GN and WP. Even with such floristic separation, all annual collections at GN, MP and WP clustered at greater than 50-60% similarity, indicating a high degree of consistency in the year-to-year algal flora at these sites.

In contrast, the overall similarity of annual collections at FE was only about 35%, with seven groupings apparent at about the 70% similarity level (Fig. 3d; Table 1). The first group represents collections made during 2-unit 1-cut operational years (1979-82), when the flora at FE was similar to that observed at other unimpacted stations (Table 1). Temperature conditions were severely altered when the second quarry cut was opened in 1983, and account for the relatively low algal diversity at FE, and increased occurrence of some of warm-water species discussed above (*e.g., Lomentaria baileyana, Champia parvula,* and *Hincksia mitchelliae*) in 1983 and 1984 (Group 2). Algal diversity remained low in Group 3 years (1985 and 1986), when dominant perennials (*Chondrus crispus, Ascophyllum nodosum* and *Ahnfeltia plicata*) and their associated epiphytes (*e.g., Ceramium rubrum, Protomonostroma undulatum, Elachista fucicola, Cystoclonium purpureum* and *Polysiphonia lanosa*) were eliminated. Opportunistic species, including *Codium fragile, Polysiphonia harveyi* and *Enteromorpha intestinalis* became more common. Another floristic shift occurred during the early years of 3-unit operation (1987 and 1988; Group 4). Increased
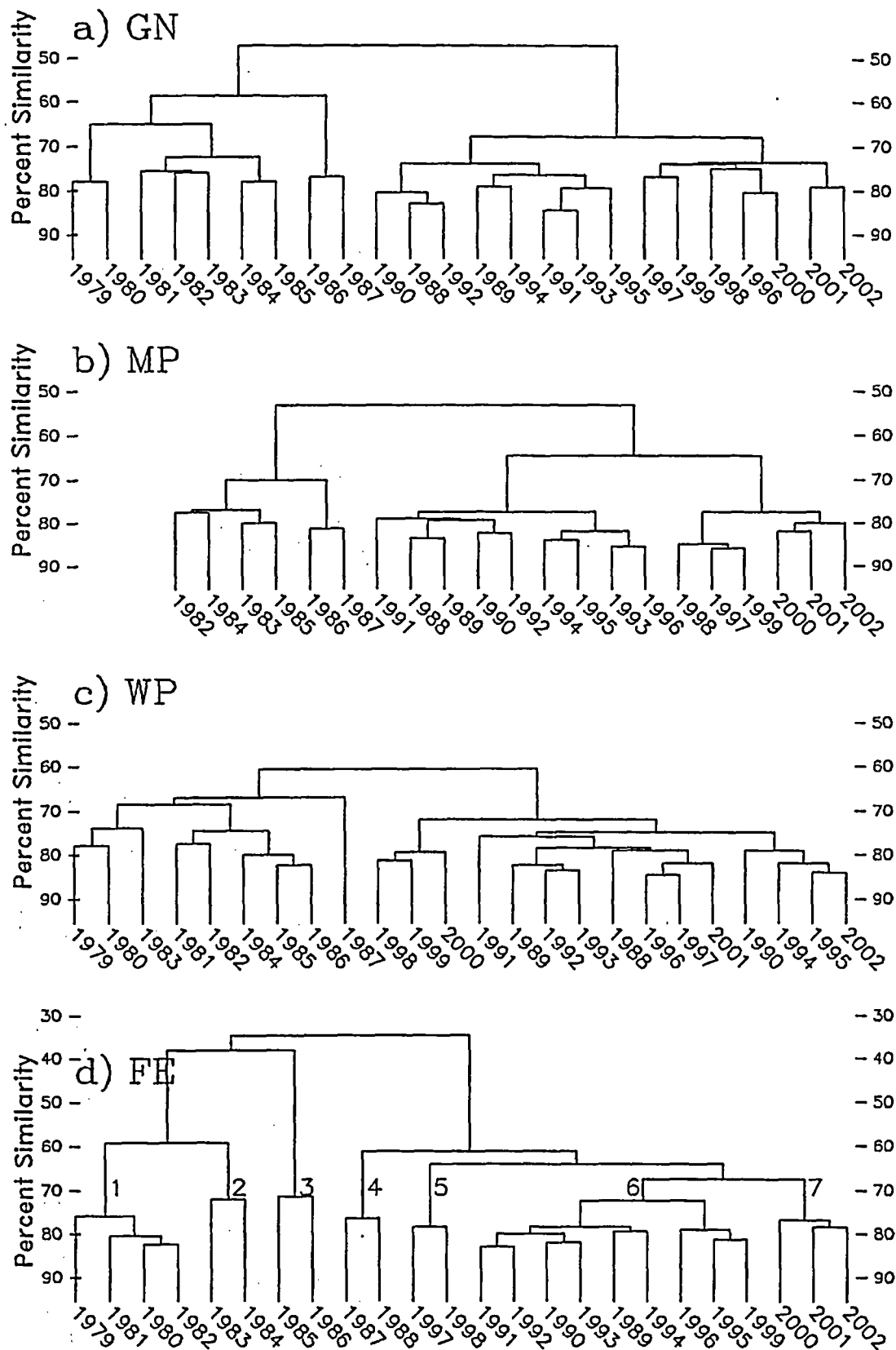
Fig. 3. Clustering dendrogram of percent similarity of qualitative algal collections, by year, at rocky intertidal sampling stations. Numbered groups at FE are further characterized in Table 1.

water flow out of the quarry cuts from the Unit 3 discharge reduced temperature extremes at FE, allowing for reestablishment of some species (*e.g.*, *Chondrus crispus, Polysiphonia fucoides, Elachista fucicola*, and *Spongomorpha arcta*), and initial colonization by new populations (*Sargassum filipendula* and *Gracilaria tikvahiae*). This group was further distinguished by the initial appearance *Antithamnion pectinatum*, which occurred throughout the study area during this period.

More consistent water temperature regimes occurred over the next eight years of the 3-unit operating period (1989-1996) which are included in Group 6. This relative consistency allowed for some further successional development of what has become a unique flora at FE, characterized by perennial populations of *Sargassum filipendula* and *Gracilaria tikvahiae* and shifts in seasonal species occurrence patterns described above. Other researchers reported similar shifts of attached algae near thermal effluents (Vadas *et al.* 1976; Wilce *et al.* 1978; Schneider 1981). However, this algal community exhibited a degree of resiliency to more than two years of plant shutdown (1996-1998), as Group 6 also includes 1999, the first year when both Units 2 and 3 operated after the shutdown. Detectable changes to the FE algal flora occurred during the shutdown period, as indicated by the separate grouping of 1997 and 1998 (Group 5). Some examples include reduced occurrence of established perennial populations of *Sargassum* and *Gracilaria*, brief reoccurrence of cold-water annuals discussed above, and reduced occurrence of warm-water annuals relative to previous years during 3-unit operation. Collections in 2000-2002 (Group 7) were characterized by increased occurrence of species such as *Porphyra carolinensis* and *Hypnea musciformis*, but the close linkage to Group 6 indicates the persistence of the thermally-influenced Fox Island algal community that returned following the restart of Units 2 and 3.

Relationships among all station/year collections were examined using multi-dimensional scaling (MDS) techniques. The resulting MDS plot (Fig. 4) revealed two distinct clusters. The largest cluster is comprised of all annual collections made at unimpacted sites (GN, MP and WP) along with FE collections made prior to observable thermal impacts (1979-1983). Initial years of thermal impacts at FE (1984-1986) appear as spatially distinct outliers in the MDS plot, and represent early successional years during the most severe thermal loading after the opening of the second quarry cut. The unique algal community that developed at FE after Unit 3 start-up in 1986, including the years of shutdown and reactor restarts, are represented by annual collections included in a smaller cluster. Years comprising this cluster are the same as those included in Groups 5-7 of the dendrogram discussed above (1987-2002). The temporal trajectory of algal community development under various thermal regimes following Unit 3 start-up can be tracked in this

cluster. This pattern demonstrates the establishment of a new environmental domain at FE during 1987-2002 (*sensu* Bradbury *et al.* 1984) that exhibits some degree of resilience to fluctuation in thermal regimes, such as ambient conditions during station shutdown from 1996 to 1998.

Abundance Measurement

Thermal impacts also change abundance and distribution of existing species; such changes could be undetected based solely on qualitative sampling described in the previous section. Therefore, more quantitative assessments of distribution and abundance patterns of dominant intertidal organisms, based on percentage substratum cover of some seaweeds discussed above, and also of several invertebrates, were conducted using permanently marked transects. This study was 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 could then be related to site-specific physical and biological controlling mechanisms including, at stations near the MPS discharge, exposure to elevated temperature regimes. Abundance patterns of ecologically important intertidal organisms follow, along with analyses of overall community structure.

Barnacles

Barnacles (primarily *Semibalanus balanoides*) occupy a large proportion of intertidal rock surfaces on local shores, relative to other sessile invertebrates. They also exhibit spatial and temporal patterns of abundance that can be related to specific environmental factors. Barnacle abundance is generally highest in the mid-intertidal zone, where habitat conditions are optimal relative to upper and lower zone conditions. Barnacle abundance in the upper intertidal zone is limited by shorter immersion time, reducing time for larval settlement and feeding, and creating more physical stress from desiccation and temperature extremes (Connell 1961; Grant 1977; Gaines and Roughgarden 1985; Connolly and Roughgarden 1999). Longer immersion time of low intertidal surfaces dramatically increases larval supply and, coupled with extensive algal canopy, improves survival conditions (Leonard 1999). Higher predation rates and interspecific competition for space (lack of available substratum) substantially limits abundance (Connell 1961; Underwood and Denley 1984; Minchinton and Scheibling 1991, 1993; Bertness *et al.* 1999). Seasonally, 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
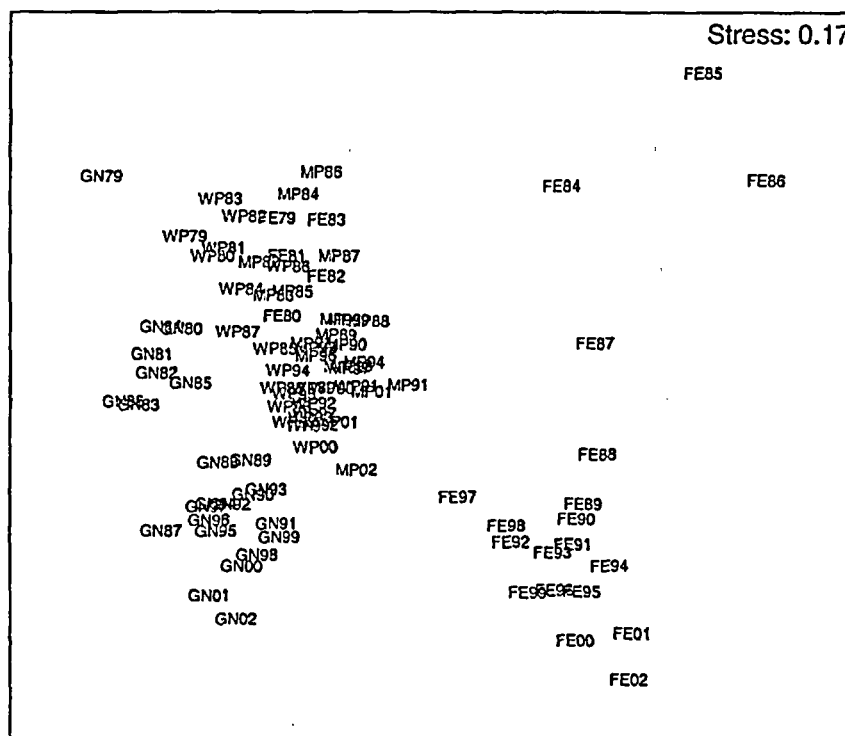
Fig. 4. Two-dimensional MDS plot for annual qualitative algal collections from rocky intertidal transects at FE, GN, and WP (1979-2002), and MP (1982-2002).

winter due to a combination of overcrowding, predation and physical disturbance (Connell 1961; Menge 1976; Hughes and Griffiths 1988; Bertness 1989; NUSCO 1993; Bertness *et al.* 1998). In addition, long-term monitoring studies in Plymouth, U.K. summarized by Southward (1991) have revealed trends in barnacle abundance and distribution over periods of 10-25 years that may be related to sea water temperature, solar (sunspot) cycles, or changing weather patterns and other effects of global climate shift.

Barnacle abundance patterns described above, both seasonal and spatial were observed at all Millstone study sites over the 24-year study period (Fig. 5). Maximum barnacle cover in the high intertidal (Zone 1) during 2002 ranged from 6% (GN) to 22% (MP). Minimum coverage in Zone 1 ranged from ≤2% at GN and WP to 7% at MP. In the mid intertidal (Zone 2), maximum barnacle cover in 2002 was lowest at FE (12%) and highest at GN (60%); minimum cover in Zone 2 was lowest at FE (<2%) and highest at GN (20%). Low intertidal (Zone 3) maxima during 2002 ranged from 0% (FE) to 17% (GN). The annual minimum in Zone 3 in 2002 was 0% at FE and MP, <1% at WP, and 6% at GN. Peak abundance of predatory snails (primarily the oyster drill, *Urosalpinx cinera*) in 2002 at all study sites occurred in July or September, and ranged from 0% (GN) to 2% (MP).

While predictable annual barnacle abundance cycles have been observed at each site over the study period,

recruitment and peak abundance levels have been more variable from year to year. Peak abundance estimates in 2002 were low at most sites (excluding GN) and have been low regionally for four of the past five sampling years, with 2000 being the exception. These, and other studies (*e.g.*, Underwood and Denley 1984; Minchinton and Scheibling 1991; Miron *et al.* 1999), show a strong relationship between larval settlement and peak abundance levels. In 2000 and most study years, seasonal cycles were characterized by heavy set of cyprid larvae in spring, followed by growth to a summer peak in percent cover, as described above. The specific factors influencing barnacle recruitment from year to year are unclear, but the regional nature of recruitment success suggests natural, possibly climatic, environmental factors unrelated to power plant operation.

Natural regional environmental factors were also important in determining barnacle population structure at FE, as evidenced by low peak abundance in recent years similar to unimpacted sites. Barnacle abundance at FE was higher during 3-unit operation (1986-1995), and particularly high during the 1996-98 shutdown period. Since 1998, peak barnacle abundances at FE were generally low, with near complete elimination of barnacles in Zones 2 and 3 each summer. This sharp a decline had not been observed in previous 3-unit operational years, and may be due to a combination of thermal effects from current 2-unit operating conditions,

Fig. 5. Abundance of barnacles in each zone, and of predatory snails in Zone 3, of undisturbed transects, from March 1979 through January 2003.

Fig. 5. (cont.)

and regionally poor barnacle settlement and recruitment discussed above. Clearly, regional conditions for barnacle recruitment were better in 2000, and seasonal patterns of abundance at FE were more consistent with those observed during most 3-unit operational years (*i.e.*, 1988-1995). The thermal regime at FE under current 2-unit operation (Units 2 and 3; temperatures 6-7°C above ambient) is not as stressful as that observed after the opening of the second quarry cut in 1983, and prior to initial Unit 3 start-up in 1986. The 2-unit (Units 1 and 2) 2-cut discharge during this period elevated temperatures by 9-11°C at FE and caused complete elimination of barnacles in Zones 2 and 3 by October (Fig. 5; NUSCO 1992). This further supports the hypothesis that low barnacle abundance in recent years (*i.e.*, 1998-2002) at FE is partially related to regional factors, but is also related to site-specific community changes at FE during this period, discussed below.

Fucus

Fucoids are major contributors to seaweed biomass in many parts of the world, including arctic, temperate, tropical and antarctic waters (Lüning 1990). Ecological studies of fucoids began in earnest at the turn of the century; however, most progress in understanding the demography and community ecology of these plants has been made since the early 1970s (Chapman 1995). Recent research has shown that *Fucus* canopy can be a major influence on abundance of other algae, sessile invertebrates, snails, and on the recruitment, regeneration and growth/self thinning of *Fucus* itself (McCook and Chapman 1997; Jenkins *et al.*1999). On local shores near MPS, *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.

Fucus populations in the MPS area exhibit distribution patterns and seasonal abundance cycles similar to those reported elsewhere in the North Atlantic Ocean (Lubchenco 1980, 1983; Topinka *et al.* 1981; Creed *et al.* 1996; Johnson *et al.* 1998; Karez and Chapman 1998). At most MPS 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. 6). Peak abundance in Zone 1 during 2002 was greatest at FE (71%) and lowest at GN (<1%). *Fucus* abundance was highest in Zone 2, relative to upper and lower zones, at all stations except FE (owing to the high *Fucus* cover in Zone 1, resulting from the increased degree of exposure to waves and spray). *Fucus*

abundance reached its highest at MP, peaking at 83%; Zone 2 abundance peak in 2002 was lowest at WP (50%). In Zone 3, maximum *Fucus* cover during 2002 was greatest at GN (60%); the lowest abundance peak occurred at FE (2%). Adult *Fucus* is relatively resistant to herbivory, but newly settled germlings are vulnerable. Grazing snails (primarily the common periwinkle *Littorina littorea*) were most abundant at GN with maximum cover estimates in Zone 3 reaching 5%. Grazing snails were absent from low intertidal collections at MP in 2002, and have not been observed in Zone 3 quadrats at FE since 1998.

Fucus abundance patterns varied among study populations, reflecting environmental conditions unique to each site. In general, *Fucus* is most abundant on moderately exposed shores, such as those at FE and MP. *Fucus* abundance is limited at highly exposed sites by physical stress from wave shock, while at more sheltered sites, like WP and GN, these species are often outcompeted for space by another fucoid, *Ascophyllum nodosum* (Schonbeck and Norton 1978, 1980; Keser and Larson 1984). Vertical distribution patterns of intertidal *Fucus* are generally determined by desiccation rate gradients controlled by the degree of wave exposure, as well as slope of available substratum (Johnson *et al.* 1998). More detailed description of the role these natural, site-specific characteristics play in determining *Fucus* zonation patterns at each study site is provided in previous reports (*e.g.*, NUSCO 1992, 1993).

Since 1983, MPS operation has had a measurable impact on the *Fucus* abundance at FE. Prior to that, the two operating units discharging through a single quarry cut resulted in only a small thermal addition (<2°C) to the shoreline at FE, which had no detectable effects to the intertidal community there (NUSCO 1992). Dramatic community changes were observed after the opening of the second quarry cut, particularly during the summer of 1984 with two units operating. The perennial *Fucus* population occupying all three intertidal zones was eliminated by temperature increases of up to 13°C. As discussed previously in the barnacle section of this report, Unit 3 start-up reduced thermal stress to the adjacent shoreline, as increased discharge velocity carried the zone of extreme temperature conditions beyond the FE study site during most of the tidal cycle. As a result, perennial *Fucus* populations returned to mid and upper intertidal zones at FE, because they were exposed to air at low tide, the time of maximum thermal incursion. However, *Fucus* in Zone 3 was exposed to temperature increases of 8-9°C for several hours during each tidal cycle; this regime would totally eliminate *Fucus* by September of most 3-unit operational years. A similar regime was apparent at FE from 1999 to 2002 as Zone 3 *Fucus* was eliminated in each summer with both Units 2 and 3 operating.

Fucus exhibited long-term abundance cycles at study sites more distant from the discharge than FE; these are
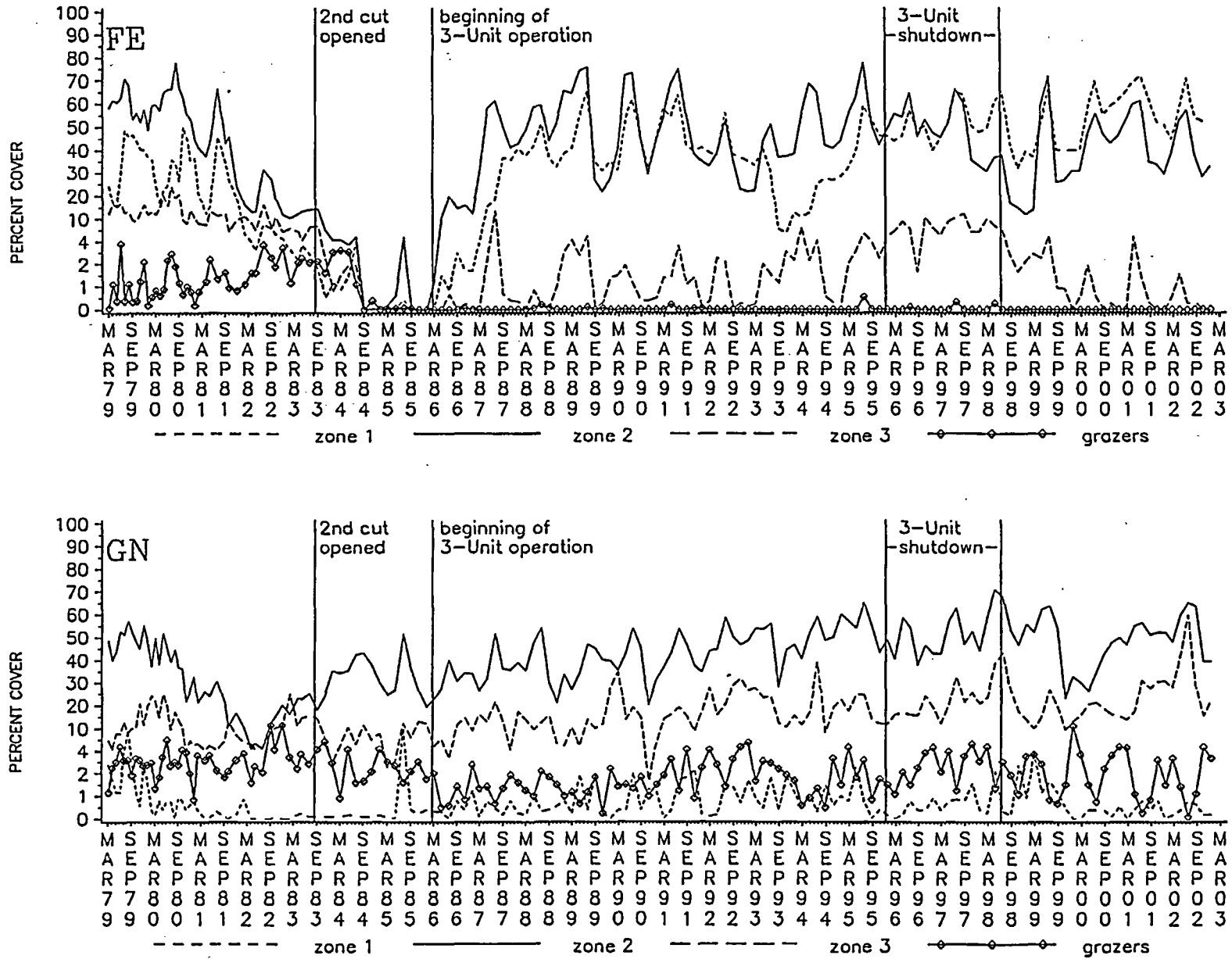
Fig. 6. Abundance of *Fucus* in each zone, and of grazing snails in Zone 3, of undisturbed transects, from March 1979 through January 2003.
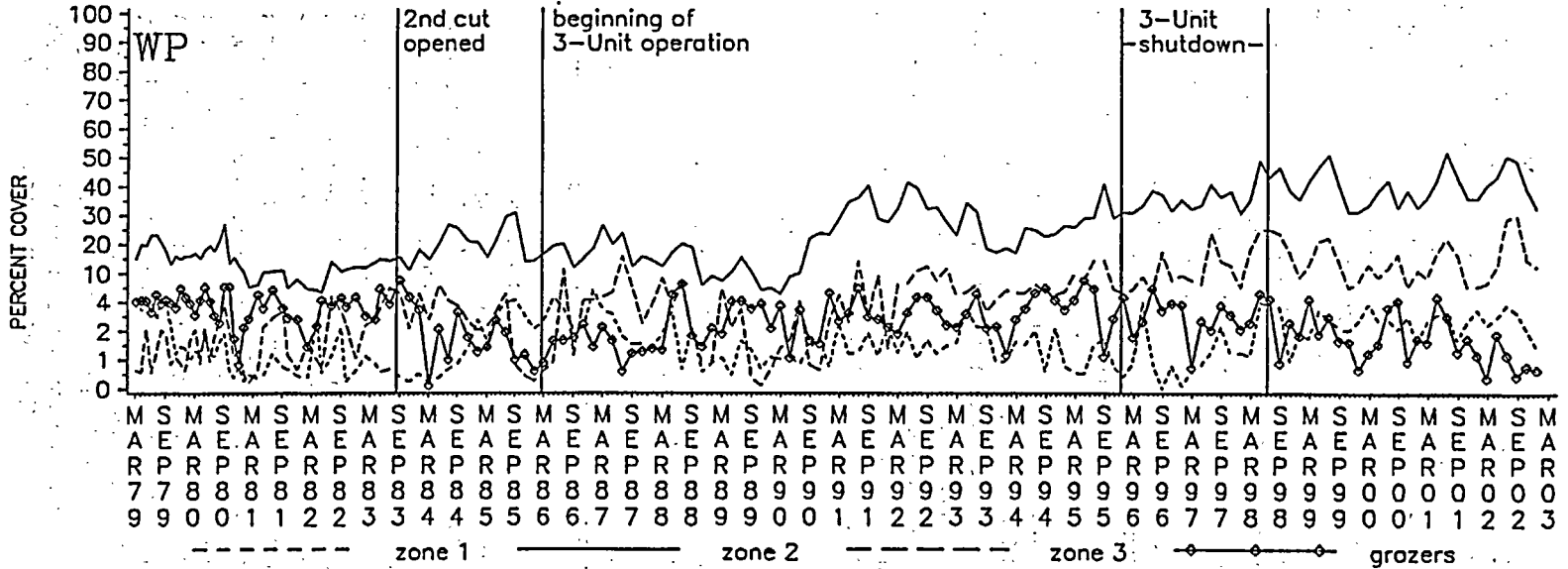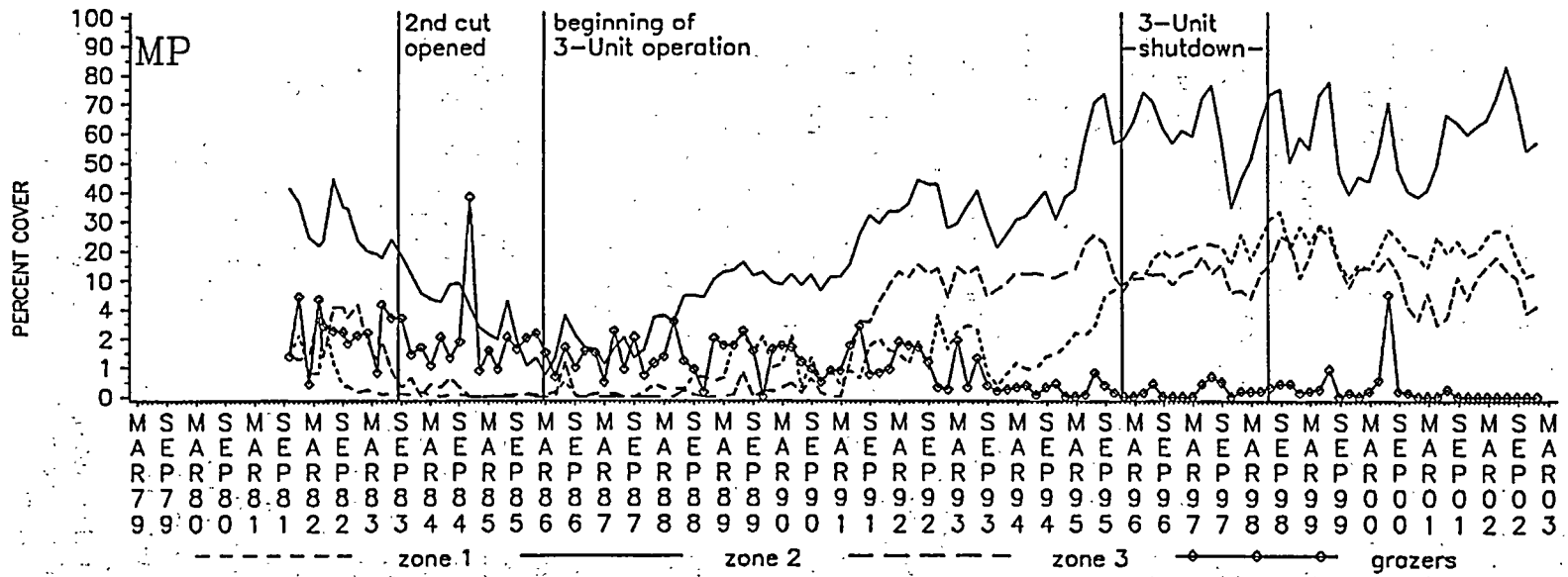
Fig. 6. (cont.)

likely unrelated to MPS operation. Most notable of these is the protracted decline/recovery cycle at MP discussed in previous reports (*e.g.*, NUSCO 1996, 1998). *Fucus* abundance in Zone 2 at MP reached an historical high in 2002, and has plateaued at high levels established in recent years in Zones 1 and 3. We have no explanation for this long-term abundance cycle at MP. Proximity of MP to the MPS discharge (*ca.* 300 m to the east) and the moderate temperature increases measured directly there (2-3°C above ambient during slack tides; NUSCO 1994) suggest the possibility of a power plant impact. A direct thermal impact is unlikely, however, as the present *Fucus* population at FE recovered relatively rapidly after Unit 3 start-up, even under much greater temperature extremes than those at MP. Increase in *Fucus* abundance at MP in recent years could be related to lower abundance of grazing snails since 1993. It is unclear whether this decline in snail abundance at MP is related to the thermal plume or other factors. However, it is interesting to note that grazing snail abundance at MP in July 2000 was almost 7% cover, and concurrent peak summer *Fucus* cover was the lowest in six years. Data from WP and GN suggest that the increasing trend at MP may be part of an area-wide trend, as a long-term increase in *Fucus* abundance has become apparent over the last 15-20 years in Zones 2 and 3 at both stations.

Chondrus and common epiphytes

The common red alga *Chondrus crispus* is among the most abundant low intertidal organisms on New England rocky shores (Menge 1976; Lubchenco 1980; Lubchenco and Menge 1983) including those near MPS. As discussed in previous sections, low intertidal habitat near the MPS discharge is more susceptible to thermal impacts than are higher zones. Therefore, documentation of abundance patterns of *Chondrus* and its associated epiphytes is critical to our ecological monitoring program. Perennial stands of *Chondrus* exclude many other species from Zone 3, including *Fucus vesiculosus* discussed above (Lubchenco 1980). Two seasonally abundant algal taxa coexist as epiphytes on *Chondrus* (*i.e.*, *Monostroma* spp. (including *Protomonostroma*) and *Polysiphonia* spp.) instead of competing directly for primary space. Spatial and temporal distribution patterns of these algae in the Millstone area are discussed below.

Extensive, well-established *Chondrus* populations are documented at three of the four study sites (all but FE) during the current study period. Given its perennial habit, *Chondrus* abundance fluctuates little seasonally or from year to year at these three sites. During 2002, percent cover estimates were consistent with historical levels, peaking at 56% at GN, 83% at MP, and 78% at WP (Fig. 7).

The *Chondrus* population at FE had abundance levels similar to sites mentioned above prior to 1984 (40-75%),

but has since been reduced to scattered individual plants, with abundance estimates rarely exceeding 3%. Elevated water temperatures from the 2-cut 2-unit discharge eliminated the extensive *Chondrus* population at FE in 1984 (NUSCO 1987). Since that time, only a few scattered *Chondrus* plants have been observed in upper Zone 3 study quadrats. These plants are present during cooler months, but their upright portions are typically eliminated each summer. This summer decline was attributed to elevated water temperatures from the 2-cut 3-unit discharge. However, a similar decline was observed in both 1996 and 1997 during plant shutdown, suggesting other mechanisms, possibly seasonal overgrowth by *Codium fragile* and blue mussels (*Mytilus edulis*). These two species, and their influence on low intertidal community structure, will be discussed further in the next section. Regardless of the causal mechanisms, the pattern of summer disappearance of *Chondrus*, established under 3-unit operating conditions, was observed in each of the summers after the MPS shutdown period ended, including 2002.

Common epiphytes that exhibit warm-water and cold-water seasonality in the local low intertidal zone have shown temporal shifts in abundance in response to altered temperature regimes, typical of those at FE during MPS operation (NUSCO 1997). *Polysiphonia* spp. (mostly *P. harveyi*) are common warm-water epiphytes on *Chondrus*, *Ascophyllum* and *Codium* and also grow on rocky substrata. A late summer peak characterizes the annual abundance cycle of *Polysiphonia* spp., with cover declining to near 0% by winter at most study sites (Fig. 7). Peak abundance during 2002 was lowest at GN (3%), intermediate at MP and WP (53% and 22%, respectively), and highest at FE (70%). The annual cycle in *Polysiphonia* spp. abundance has been consistent among 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, as well as by decreasing the abundance of grazing snails (Fig. 6). 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. The return of ambient temperature conditions at FE following the 3-unit shut-down has resulted in a winter/spring decline in *Polysiphonia* abundance similar to that observed at the other rocky shore stations, and at FE prior to the opening of the second quarry cut. Following MPS restart (July 1998), high fluctuations in *Polysiphonia* abundance occurred, with peak abundance often exceeding 80%. Along with these record high peak abundances, late winter declines have been less predictable at FE since 1998, with abundance dropping below 1% in 2000 and 2001, but remaining above 50%

Fig. 7. Abundance of *Chondrus* and major epiphytes in Zone 3 of undisturbed transects, from March 1979 through January 2003.
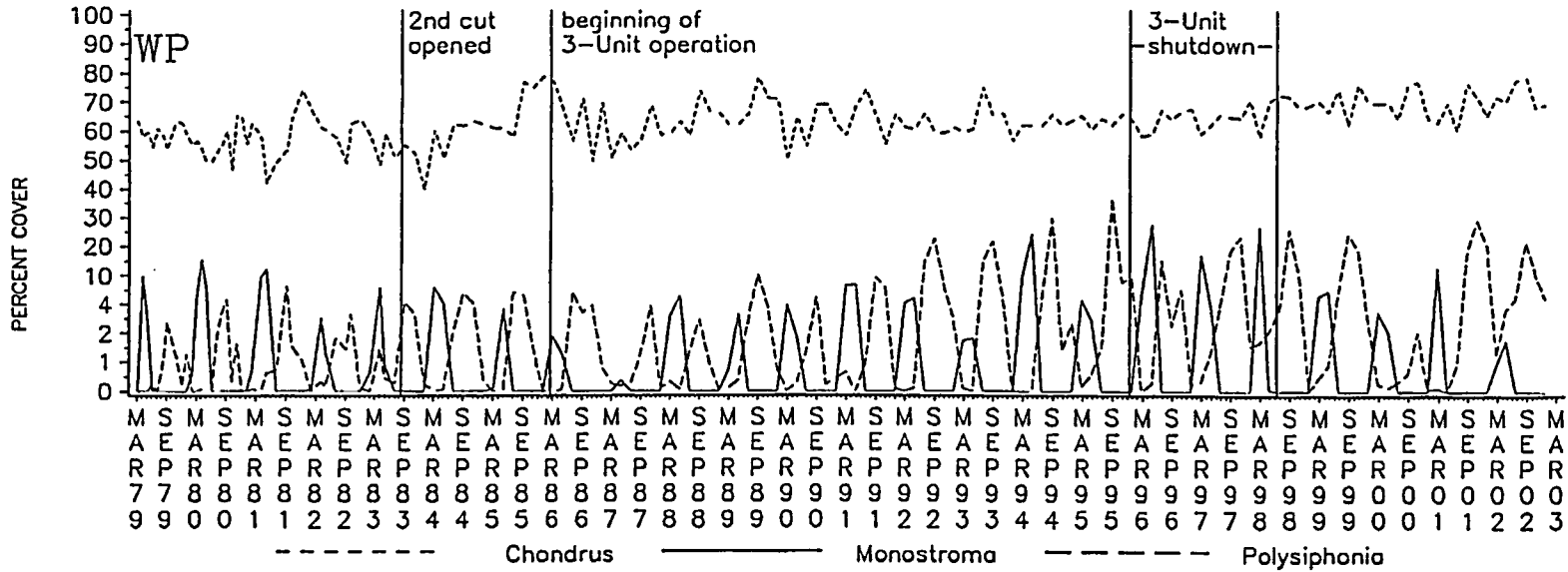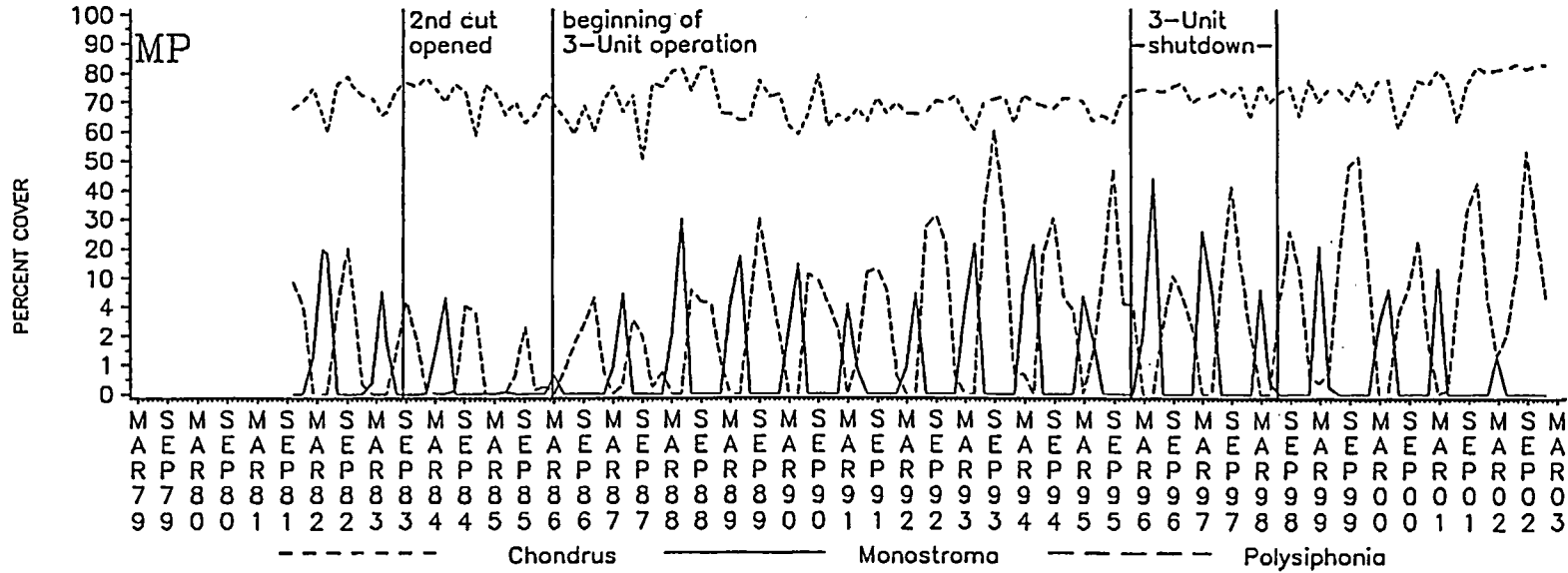
Fig. 7. (cont.)

during 2002. This inconsistency may be related to recent shifts in species dominance and community composition in Zone 3 FE, including a decline in the previously dominant Codium fragile population, and the establishment and expansion of Gelidium pusillum and Hypnea musciformis populations, discussed below.

The annual abundance cycle of Monostroma spp. (M. grevillei and Protomonostroma undulatum) is out of phase with that described for Polysiphonia spp., i.e., peak abundance is observed during cold-water months (late winter/early spring) and these species disappear during warm-water months (July-December; Table 1, Fig. 7). This annual abundance cycle occurred consistently over the study period at all study sites except FE, although peak Monostroma abundances in 2002 were unusually low. Since 1984 and excluding the recent extended shutdown period, 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%. Monostroma occurred in low abundance (<1% cover) in FE study quadrats during the three cold-water periods of the 1996-98 shutdown, but has not been observed since the plants were restarted. While Monostroma is clearly temperature-limited during thermal discharge, other environmental factors may also inhibit its abundance at FE, such as limited suitable substratum, e.g., a host species (Chondrus crispus) and bare rock.

Additional Taxa at Fox Island-Exposed

A number of other low intertidal taxa at FE have become important to the monitoring program because they have exhibited localized shifts in abundance. Population shifts of these taxa reflect acute effects of power plant operational changes and long-term successional development of the thermally altered community in the nearfield discharge area. Therefore, time-series of abundance of these taxa are presented in Figure 8, and discussed below, to provide a more comprehensive description of ecologically significant changes to the intertidal community at FE.

Codium fragile - This siphonaceous green alga is now a common component of the algal flora of eastern Long Island Sound, and is frequently found at all MPS monitoring stations (Table 1). Codium is an introduced species, first collected on the northeast coast of North America in 1957 at nearby East Marion, Long Island, NY (Bouck and Morgan 1957). This opportunistic or 'weed' species quickly colonized other coastal areas of New England, including Connecticut. Optimal light requirements of Codium are comparable to those of other low intertidal algae such as Chondrus (Mathieson and Burns 1971). However, Codium is competitively inferior to Chondrus (through competitive exclusion), as

demonstrated by low Codium abundance (<5% cover) at sites where extensive Chondrus canopy is found (DNC 2001). Codium is considered a community dominant only at FE, with maximum cover often approaching 90 % since 1984. Elimination of the FE Chondrus population in 1984 by elevated temperatures allowed Codium to develop a perennial population during the 3-unit operational period, which persisted through the 1996-98 shutdown period. However, since 2000, Codium cover at FE has declined precipitously, and remained <2% in Zone 3 throughout 2002. This decline has coincided with expansion of populations of Gelidium pusillum, Corallina officinalis, and most recently, Hypnea musciformis discussed below.

Mytilus edulis - Blue mussels were observed periodically at most study sites (DNC 2001), and have often been among the dominant low intertidal taxa, particularly at FE (Fig. 8). Temporal mussel abundance patterns were altered at FE after Unit 3 startup in 1986. Prior to 1986, mussel abundance at FE never exceeded 5%, but during 3-unit operation, annual abundance peaks often reached 20-30% and in one instance (1994) exceeded 90%. Mussel cover in Zone 3 at FE in 2001 peaked at 32%, but mussels were absent in 2002. The higher settlement of mussels at FE since 1986 was previously attributed to hydrodynamic characteristics of the 3-unit thermal plume, but may also be related to indirect effects such as the ability of mussels to settle in the extensive low intertidal Codium canopy. Following settlement, initial higher growth and subsequent high mortality were attributed to elevated water temperatures at FE. Thermal plume incursion at FE produced optimum temperature conditions for growth (10 to 20°C; Seed and Suchanek 1992) coinciding with spring and early summer phytoplankton blooms. However, thermal incursion proved detrimental by late summer, as temperatures approached and often exceeded 27°C, the maximum temperature for adult survival (Gonzales and Yevich 1976). High mussel cover was observed in 2000 and 2001 even with the sharp decline in Codium abundance, indicating that replacement species in Zone 3 (e.g., Gelidium pusillum discussed below) provide suitable substratum for settlement. Low mussel abundance, such as that observed in 2002, has been observed in previous years at FE (e.g., 1988, 1993, and 1996), and was attributed to regionally poor settlement and/or recruitment.

Antithamnion pectinatum – Another introduction to the Millstone area low intertidal zone, A. pectinatum (Foertch et al. 1995), occurs commonly as an epiphyte on Chondrus, Corallina and Ulva. This species is common in warm temperate waters of the western Pacific, but has also been identified as a recent introduction to the Mediterranean Sea (Verlaque and Riouall 1989) and the Azores (Athanasiadis and Tittley 1994). Since its introduction to the Millstone area in 1987, A. pectinatum

Fig. 8. Abundance of additional taxa of interest in Zone 3 of undisturbed transects at FE, from March 1979 through January 2003.

has been found at all sampling sites, including FE near the MPS discharge (DNC 2001), and in Rhode Island (Villalard-Bohnsack 1995). Because this species is present at all of our study sites, including FE, its occurrence is considered an area-wide phenomenon unrelated to MPS. In the past several years, *A. pectinatum* has become increasingly common at FE, with peak cover estimates at times exceeding 30%. In 2002, maximum cover of this species FE was 25%. The generally higher abundance of *A. pectinatum* in recent years is likely due to expansion of the low intertidal host species *Corallina officinalis* coupled with the return of elevated temperature regime after the 1996-98 MPS shut-down period.

Other species - Abundance time-series of four perennial macroalgae (*Gelidium pusillum*, *Corallina officinalis*, *Sargassum filipendula*, and *Hypnea musciformis*) now considered low intertidal community dominants at FE are presented in Figure 8. All four species have biogeographical ranges that included Long Island Sound, but also extend into tropical waters (Taylor 1957). Qualitative sampling indicates that *Gelidium* occurrence has increased regionally since 1986 (Table 1), and has now established an extensive turf in the low intertidal at FE at times exceeding 65% cover. Reduced competition for space with *Codium* in recent years has allowed for population expansion. The *Corallina* population at FE also appears to have benefitted from reduced *Codium* abundance at FE, as Zone 3 cover has exceeded 30% in recent years. *Sargassum* was first observed in 1986 at FE, and has occurred periodically primarily during warm water months, but has rarely been collected at other sampling sites (Table 1). Declines noted in winter at FE are attributed to loss of upright thalli, presumably a response to colder water temperatures. The *Sargassum* population at FE appears dependent on thermal input from MPS, as few plants were observed during the 1996-98 shutdown period. After this period, extended operation of Units 2 and 3 provided beneficial conditions for the FE population as *Sargassum* abundance exceeded 30% cover at times, the highest level observed since the study began. Since its initial discovery at FE in 2001, *Hypnea musciformis* has become the dominant low intertidal species there. Summer abundance exceeded 70% cover in 2002.

Community Analysis

Over one hundred macroalgal and invertebrate taxa occur in local rocky shore communities. Abundance and distribution of these taxa are influenced by complex interactions between physical (*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 taxa not normally considered intertidal organisms, such as

fish and shorebirds), growth and reproductive cycles). Characterization of these communities may be descriptive; abundance of populations that are stable or predictably variable may be represented as time-series of percentage of substratum coverage, as described 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 taxa found in the transects, even those that are rare or unpredictable in their occurrence.

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. 9). Annual samples at FE (Fig. 9a) were much more dissimilar than those at other stations; at the 50% similarity level, years grouped into four distinct clusters. Group I (1979-83) is comprised of early study years prior to observable effects from the opening of the second quarry cut. This community was characterized by high *Chondrus*, *Fucus* and barnacle coverage, with an appreciable amount of available free space (rock); it was similar to communities found at nearby unimpacted sites.

Group II comprises 1984-86, when the opening of the second cut substantially altered the community at FE. *Fucus* and *Chondrus* abundance declined considerably, and these species were replaced with *Codium* and ephemeral green algae (*Enteromorpha* spp.). The next 12 study years make up Group III (1987-1998), which is further divided into two subgroupings at the 60% similarity level. Group IIIa is comprised of the 3-unit operating period from 1987 to 1996 (excluding 1994) and is characterized by the return of *Fucus* to Zone 2, expansion of the *Codium* population in Zone 3, and less bare rock. Group IIIb (1994 and 1997-1998) is distinguished from Group IIIa by lower *Codium* cover, high cover of *Mytilus* and more diatoms. The last four years (1999-2002) are included in Group IV, and are characterized by low abundance of *Codium* and barnacles and high abundance of *Polysiphonia harveyi*, *Corallina officinalis*, *Gelidium pusillum*. Increased dominance of *Sargassum filipendula* and *Hypnea musciformis* also distinguish Group IV from other groups.

The high among-year similarity illustrated by the dendrograms for the other three rocky intertidal monitoring stations reflects the more stable environmental conditions at these sites. Annual collections at GN formed three distinct groupings at the 70% similarity level (Fig. 9b). Group I (1982 and 1983) was distinguished from Groups II (1979-1981, 1984-1991, and 2000) and III (1992-1999, 2001, and 2002) by lower abundances of barnacles and *Fucus*, and more bare rock. Group III collections also had higher abundance of *Gelidium pusillum*. At MP, three groupings were noted at the 70% similarity level (Fig. 9c): Group I (1982-1987), Group II (1988-1994) and Group III (1995-2002).
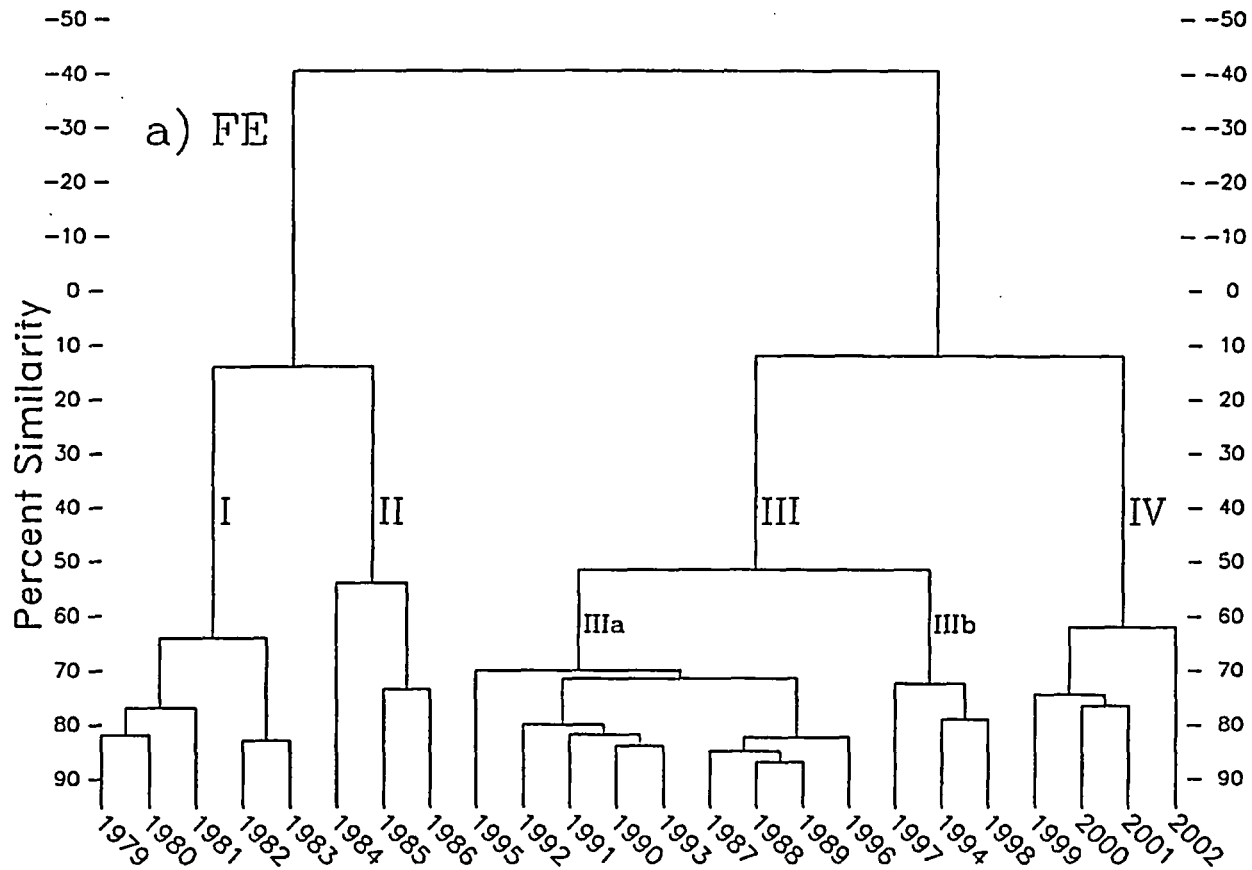
taxon	Group I	Group II	Group III	Group IIIa	Group IIIb	Group IV
Fucus vesiculosus	24.33	2.76	25.00	25.45	23.65	21.68
Codium fragile	0.95	24.98	31.53	36.05	17.97	4.01
Polysiphonia harveyi	5.89	6.03	11.98	12.61	9.73	26.92
Semibalanus balanoides	18.94	15.29	10.21	10.78	8.52	3.16
Mytilus edulis	0.34	2.55	14.15	7.61	33.75	8.51
rock	16.91	16.04	3.43	3.48	3.26	2.52
Chondrus crispus	30.28	4.88	0.59	0.48	0.93	1.66
Enteromorpha linza	1.23	6.31	8.49	9.06	6.75	5.89
Enteromorpha flexuosa	0.92	14.54	4.64	4.92	3.78	4.30
Ulva lactuca	3.09	3.00	5.73	5.95	5.06	3.44
diatoms	0.11	1.41	4.52	2.85	9.52	5.75
Gelidium pusillum	0.00	0.00	0.87	0.77	1.19	17.13
Corallina officinalis	0.56	0.36	1.12	0.89	1.82	9.44
Hypnea musciformis	0.00	0.00	0.00	0.00	0.00	4.82
Sargassum filipendula	0.00	0.00	0.46	0.41	0.62	3.07
Enteromorpha clathrata	0.00	3.74	0.16	0.02	0.57	0.00

Fig. 9.   Clustering dendrogram of percent similarity of undisturbed communities, by year, at rocky intertidal sampling stations: a) Fox Island - Exposed, b) Giants Neck, c) Millstone Point, and d) White Point. Each dendrogram is followed by a listing of taxa whose mean percent substratum coverage (by station) was at least 3% in at least one of the groupings determined by community analysis; group numbers correspond to those in the dendrograms.

b) GN
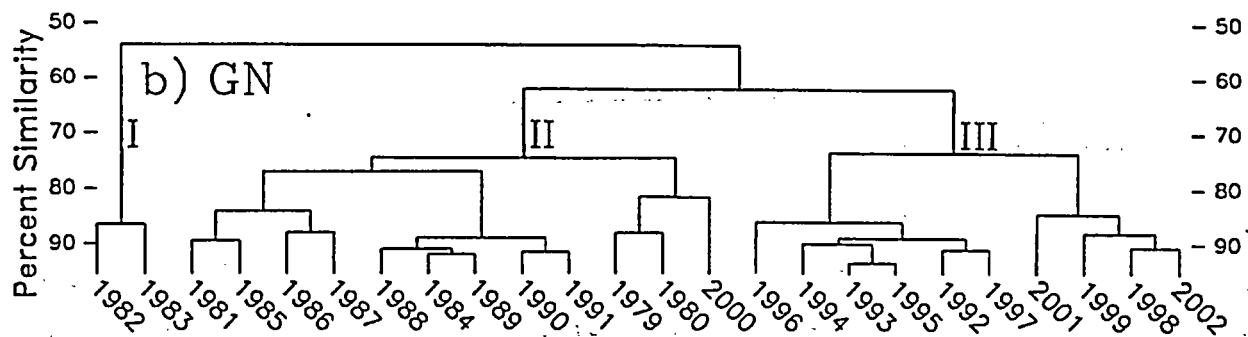
taxon	Group I	Group II	Group III
Semibalanus balanoides	23.66	35.21	31.87
Fucus vesiculosus	13.90	24.18	36.53
Chondrus crispus	24.35	24.91	26.51
rock	28.75	19.16	13.71
Ascophyllum nodosum	5.33	5.78	5.74
Littorina littorea	4.60	2.28	2.12
Gelidium pusillum	0.00	0.65	4.60
debris	0.00	3.22	0.81

c) MP
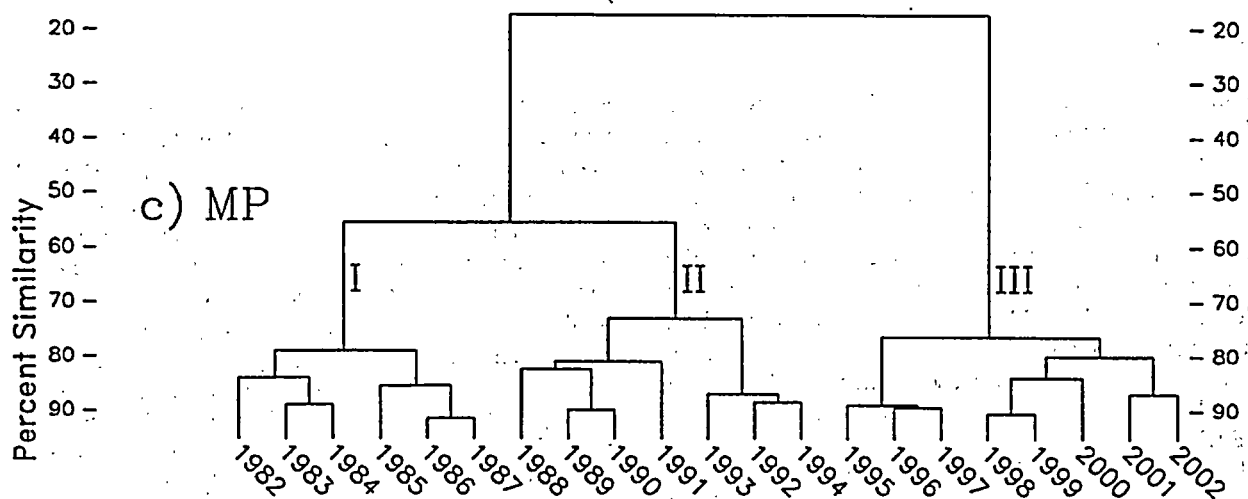
taxon	Group I	Group II	Group III
Chondrus crispus	36.49	35.97	44.72
Semibalanus balanoides	24.40	26.38	24.42
Fucus vesiculosus	5.35	13.96	41.19
rock	22.38	12.55	5.82
Enteromorpha linza	1.15	5.69	6.91
Polysiphonia harveyi	1.09	4.97	7.07
Ulva lactuca	1.10	3.96	7.26
Mytilus edulis	0.77	3.69	7.10
Ralfsia verrucosa	3.63	2.28	0.10
Littorina littorea	3.13	1.41	0.14

Fig. 9. (cont.)

d) WP
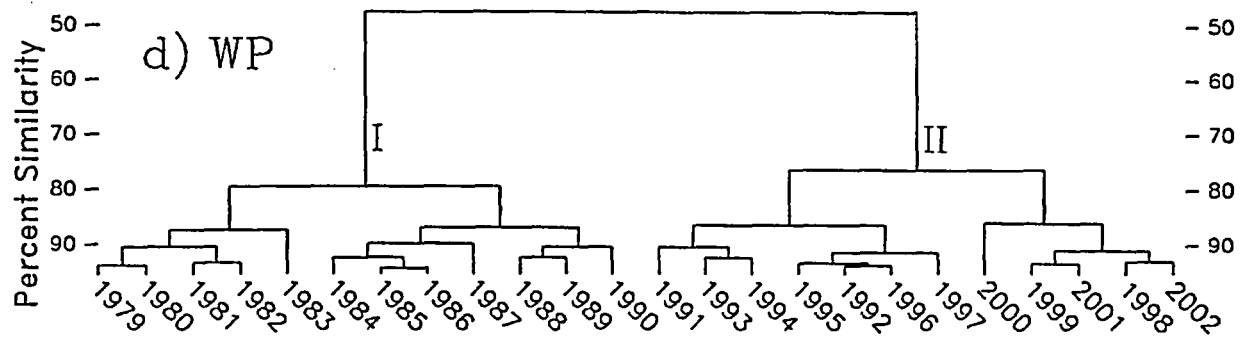
taxon	Group I	Group II
Chondrus crispus	33.83	37.01
rock	24.01	16.32
Semibalanus balanoides	17.79	15.25
Fucus vesiculosus	9.20	22.93
Ascophyllum nodosum	14.01	12.54
Ralfsia verrucosa	2.52	3.10
Polysiphonia harveyi	0.88	3.52

Fig. 9. (cont.)

A pronounced increasing trend in *Fucus* abundance at MP discussed earlier, along with more subtle increases in ephemeral green algae (*Ulva lactuca* and *Enteromorpha linza*), *Polysiphonia harveyi*, and *Mytilus* and a decline in *Littorina littorea* abundance and · available rock substratum over the years, were the major community changes that explain among-group dissimilarities. In addition, Group III at MP was further distinguished from other groups by higher *Chondrus* cover. Similar temporal . trends in *Fucus* abundance and available rock factored strongly in groupings determined by cluster analysis at WP (Fig. 9d).

To provide a comparison of community states at FE to those observed at unimpacted stations, multi-dimensional scaling techniques were applied to a Bray-Curtis similarity matrix comprised of station/year collections from all stations. These techniques have proven useful elsewhere for understanding spatial and temporal patterns of complex assemblages on rocky shores (Dye 1998; Underwood and Chapman 1998). The resulting MDS plot (Fig. 10) has a low stress level (0.09), indicating a good representation of the ordination in two dimensions (Clarke 1993), and reveals two distinct groupings with several outliers. Most collections fall into a relatively tight grouping, which comprises all annual collections at the unimpacted stations, along with annual collections from FE prior to the opening of the second quarry cut (1979-1983). The compactness of this cluster, with considerable overlap among collections, suggests that stations unaffected by MPS form a distinct community

state relative to impacted areas. The changing community states at FE are illustrated by the relationships among the remaining collections in this MDS plot and these relationships are similar to those in the clustering dendrogram discussed above (Fig. 9). Annual collections at FE from 1984 to 1986 are spatially distinct, due to dramatic community changes at FE following the opening of the second quarry cut (see subgroup II in the dendrogram). The other grouping in Figure 10 includes FE annual collections from 1988 to 2002, years of development and maintenance of the thermal community established under 3-unit operation. Recent years (*i.e.*, 1999-2002) are becoming increasingly dissimilar from other FE collection years, due to the large decline in *Codium*, and increases of *Mytilus, Corallina, Gelidium,* and *Hypnea* discussed for Group IV in Figure 9, changes particularly obvious in 2002.

Ascophyllum nodosum Studies

Populations of the rockweed *Ascophyllum nodosum* form dense, nearly monospecific, stands on sheltered rocky shores locally, including some areas near the MPS outfall. Monthly monitoring of population growth and mortality has been conducted at three locations since 1979 to assess possible effects of MPS operation. *Ascophyllum* exhibits easily quantifiable · growth · responses to even slight changes in temperature, which makes this species a critical biomonitoring tool for studies of the ecological effects of thermal effluents. This
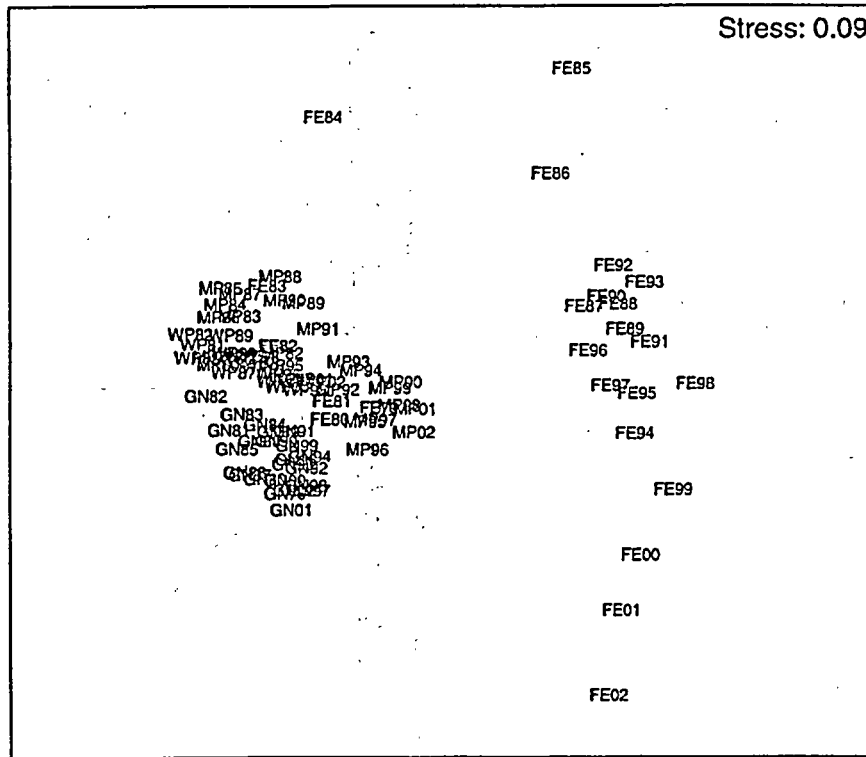
Fig. 10. Two-dimensional MDS plot for annual percent cover data from rocky intertidal transects at FE, GN, and WP (1979-2002), and MP (1982-2002).

attribute makes *Ascophyllum* a key species within the MPS ecological monitoring program and these studies, as elsewhere (*e.g.*, Vadas *et al.* 1976, Wilce *et al.* 1976), document the value of this species as a sensitive indicator of local environmental conditions. Review of phenological, ecological and applied monitoring studies of *Ascophyllum* was presented in NUSCO (1993). Results of 2001-2002 *Ascophyllum* growth and mortality studies, compared with results from previous years, are presented below.

Growth

Annual *Ascophyllum* growth is described using a Gompertz growth model (Gendron 1989) fitted to monthly *Ascophyllum* tip length data (Fig. 11). Parameters of this model provide useful indicators of *Ascophyllum* population growth characteristics. The α-parameter of the model, used as an estimate of total annual growth, was higher at FN and WP in 2000-2001 (101.4 mm at both sites) than at GN (92.0 mm; Fig. 11a). *Ascophyllum* annual growth estimates at FN and WP in 2000-2001 were significantly higher (P<0.05) than the estimate at GN, but there was no difference between annual growth at FN and WP. Growth rates were

determined by extracting the first derivative of the Gompertz growth model and plotting results on the same set of axes (Fig. 11b). The inflection point, a parameter of the Gompertz model, which identifies the time of maximum growth rate, occurred earliest in 2000-2001 at FN (19 July; 17.0 mm/mo), followed by GN (23 July; 15.1 mm/mo) and WP (27 July; 16.6 mm/mo). The annual growth estimate at GN during 2001-2002 was not significantly different from the mean of all previous years (96.5 mm; Fig. 12). Growth at FN during 2001-2002 was lower than the previous years' mean of 115.5 mm, and this year's growth at WP was higher than past years' mean (89.5 mm). The inflection points for 2001-2002 growth data at all stations were 10-14 days earlier than the means of previous years.

Relationships of *Ascophyllum* growth characteristics among the three populations monitored in 2001-2002 differed from those observed in recent years. Growth estimates at all sites were within the ranges of previous years', but the 2001-2002 value at FN was the fifth lowest since this station was established in 1985 (range 79-148 mm). In contrast, 2001-2002 growth at WP was the fourth highest in 23 years (range 62-109 mm); this year's growth at GN was more towards the middle of previous years' 69-130 mm range. We have previously identified
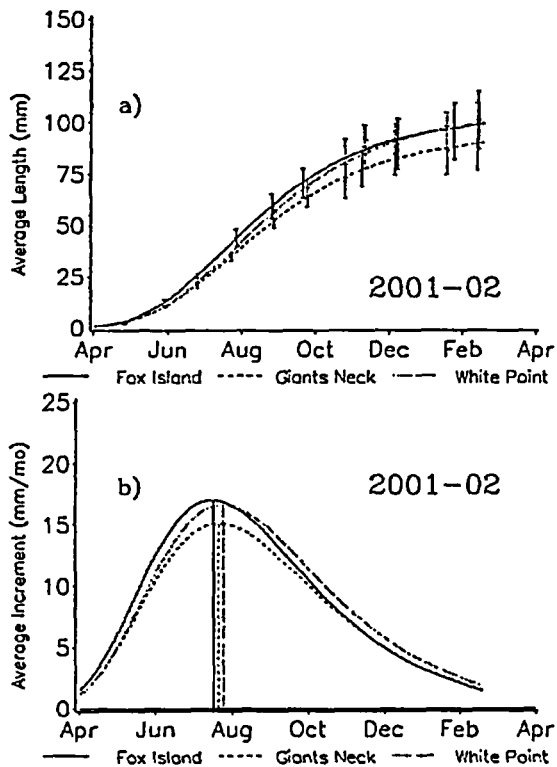
Fig. 11. *Ascophyllum* growth during 2001-2002: a) the Gompertz growth model fitted to monthly tip length data (error bars represent monthly mean lengths ± 2 SE), and b) instantaneous growth rate based on the first derivative of the Gompertz model, including inflection points.

Fig. 12. *Ascophyllum* growth and growth rates during 2001-2002 (——) compared to all previous years (- - - -), including inflection points. Error bars represent monthly mean lengths ± 2 SE.

a pattern of highest growth frequently occurring at the site nearest the discharge (FN), with lower growth at GN and WP. In addition, earlier and higher peak growth rate at FN compared to WP and GN was also frequently noted in previous years. Until recent years, the MPS thermal plume was thought to be a dominant influence on *Ascophyllum* growth at FN. During the first year of monitoring at FN (1985-86), with only one unit operating and temperature increases of only about 1°C for 1-2 hours each tidal cycle, growth was not significantly different from GN or WP. Higher temperature increases (up to 3-4°C for 3-4 hours each tidal cycle) resulted from Unit 3 startup during the 1986-87 growing season. This level of thermal input was believed to create 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). During

subsequent 3-unit years, there appeared to be a relationship between the thermal load produced by the power plant (affected by the number and duration of unit outages) and the degree of growth enhancement at FN (NUSCO 1992). A similar relationship was observed at our original experimental population (FO) closer to the discharge from 1979 to 1983, prior to thermal effluent-related elimination after the opening of the second quarry cut (NUSCO 1992).

However, higher growth continued at FN through 1996-98 MPS shutdown period when no thermal effluent was present, suggesting that natural factors at FN were more

influential than previously thought. Possible natural factors included solar warming of water from nearby sandflats and water circulation patterns in Jordan Cove (NUSCO 1999, 2000). Further confounding our hypothesis of power plant-related growth enhancement at FN were atypical relationships among monitored populations during 1998-99 while a thermal effluent was present (*i.e.*, growth at FN was significantly lower than growth at GN, and not significantly different from growth at WP). High growth at GN and WP in 1998-99 cannot be explained by water temperature, as ambient temperatures were well within historical ranges (NUSCO 1999). Thermal effects have never been observed at WP, even during years of 3-unit operation, and we have no explanation for the relatively high growth at WP (compared to most previous years, and to the other stations) in 2001-2002. The lack of a clear relationship between seawater temperature and *Ascophyllum* growth at all three sample sites provides evidence that other regional or site-specific environmental factors, possibly nutrients or light, account for much of the spatial and temporal variability in growth observed since 1985.

Mortality

Ascophyllum populations endure considerable stress from a variety of biotic and abiotic factors in the intertidal zone (NUSCO 1992), often resulting in breakage and loss (mortality) of the upright shoots or fronds. Thalli compensate for this loss of biomass through growth of suppressed fronds, described as meristem banks (Cousens 1986; Vadas *et al.* 1990) that were previously covered and shaded by a well-developed main shoot canopy. Population mortality is monitored in this study by examining patterns of frond base tag loss (referred to as plant loss; Fig. 13) and apical tag loss (tip loss; Fig. 14). Plant loss at FN during 2001-2002 (26 remaining plants, 48% lost) was considerably less than the historic mean of 60% lost. Plant losses at GN and WP in 2001-2002 (16 and 18 remaining plants, 68 and 64% lost, respectively) were higher than previous years' means of 57 and 56% lost, respectively. During the 2001-2002 study year, the most rapid loss of plants occurred between October and November sampling trips at all sites, particularly at GN and WP. These losses were attributed to strong autumn storms, with associated winds and waves from the southwest, a direction from which the FN *Ascophyllum* population is relatively more protected.

Patterns of tip loss at each station were similar to those described above for plant loss, although actual losses were higher. The measure of tip loss at FN in 2001-2002 (95 remaining tips, 62% lost) was lower than the historic mean of 80% tip loss, and lower than 2001-2002 tip losses at GN and WP (41 and 59 remaining tips, 84 and 76% lost, respectively). Historic tip losses at GN and
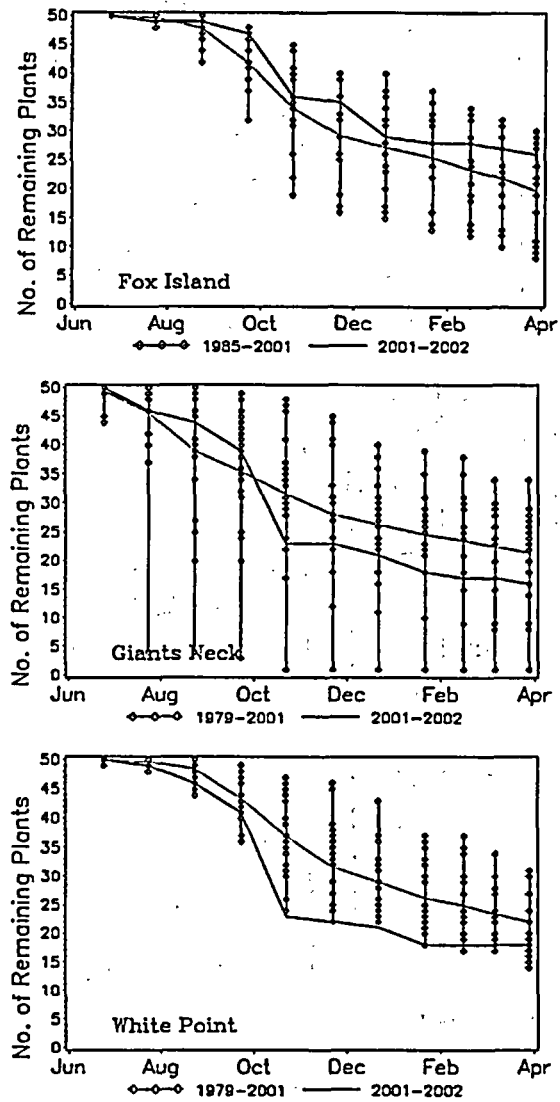
Fig. 13. *Ascophyllum* mortality, as number of remaining tagged plants, at each station.

WP had averaged 76 and 73%, respectively. Again, most rapid losses were associated with autumn storms.

Ascophyllum mortality trends observed throughout this study reveal no evidence of power plant impact, with the exception of elimination at the original experimental study site (FO) in 1984 (NUSCO 1992). 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. Throughout the study, mortality rates were highest during the months of August through November, when strong storms and high energy waves were frequent; plant and tip loss could often be attributed to an individual storm. 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
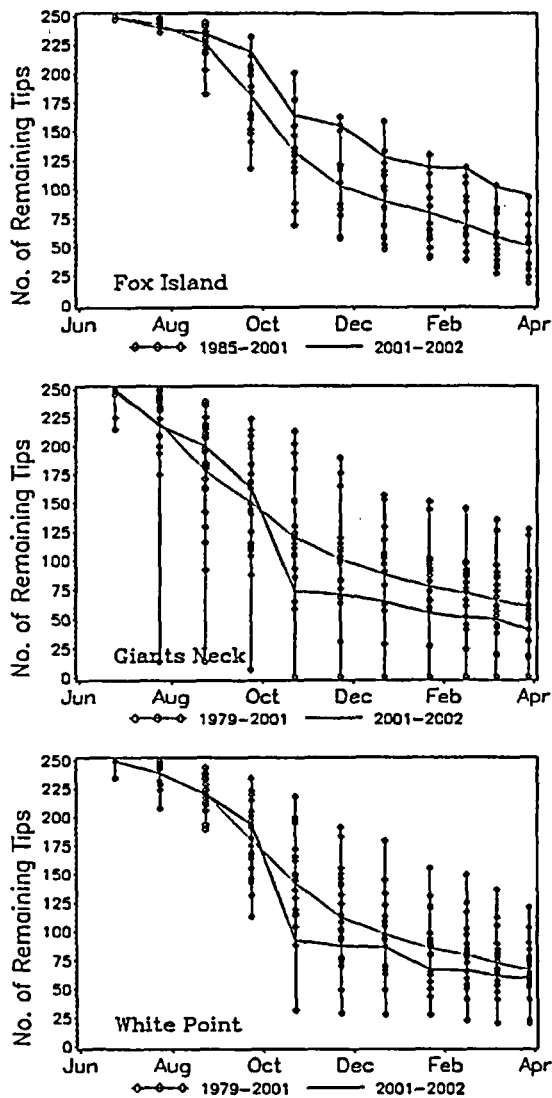
Fig. 14. *Ascophyllum* mortality, as number of remaining tagged tips, at each station.

Demetropoulos 1968; Vadas *et al.* 1976, 1978; Wilce *et al.* 1978; Cousens 1982, 1986; Vadas and Wright 1986).

The *Ascophyllum* population at our original impacted site (FO) has shown few signs of recovery following power plant-induced elimination 1984. As discussed in previous reports (NUSCO 1996, 1997), some individual thalli have settled, grown and persisted at FO during 3-unit operation and more recent years; however, no significant recovery has occurred to date. In addition, no observable recruitment of *Ascophyllum* occurred during more than two years of the recent station shutdown. Given its documented poor recruitment under natural conditions (Vadas *et al.* 1990), particularly at more exposed sites like FO (Bertness *et al.* 1999), it appears likely that post-operational recovery of this population will be slow.

Conclusions

More than 20 years of ecological monitoring studies have documented that impacts associated with operation of MPS to rocky intertidal habitats remain restricted to approximately 150 m of shoreline on the east side of the discharge to Long Island Sound. This area, which includes the Fox Island study sites, has been exposed to the thermal effluent for most years since the opening of the second quarry cut in 1983, and has supported a unique community of seaweeds and invertebrates that has undergone a degree of successional development over the last 18 years. This community has exhibited resilience to recent operational events (*e.g.*, more than two years of plant shutdown, Unit 3 restart in 1998, Unit 2 restart in 1999), exhibiting only minor and transient community changes.

The intertidal community at FE that developed under modified thermal regimes in the discharge area was characterized by shifts in occurrence and abundance of many intertidal species at FE, *e.g.*, absence or abbreviated season for species with cold-water affinity (*Chondrus, Monostroma, Dumontia*), and presence or extended season of occurrence for species with warm-water affinity (*Codium, Sargassum, Gracilaria*). In 2001, a new species to include in the latter category, *Hypnea musciformis*, established a population at FE that expanded considerably in 2002. Most of these community characteristics remained evident during the shutdown period from 1996 to 1998. Subtle community changes attributed to the return of ambient temperature conditions to FE included: 1) higher frequency of some cold water annuals (*e.g.*, *Monostroma* spp., *Spongomorpha* spp. and *Polysiphonia stricta*) and lower frequency of warm-water annuals (*e.g.*, *Bryopsis* spp., *Polysiphonia harveyi* and *Grinnellia americana*); 2) higher mid and low intertidal abundance of barnacles and *Fucus* throughout the year; and 3) small increases in abundance of *Chondrus* in the low intertidal. However, these responses to ambient temperature conditions were reversed following MPS restart.

High *Ascophyllum* growth at FN compared to other sites has been noted during many previous study years, and has been attributed to elevated temperatures from the MPS discharge. However, a similar relationship was observed during two recent growing seasons (1996-97 and 1997-98) while all three units were shutdown, and was not evident during the first year following Unit 3 restart (1998-99) or during 2001-02 when 2 units were operating. Natural influences of other factors such as ambient temperature conditions, nutrients and light may play a more important role in determining *Ascophyllum* growing conditions at FN than does thermal plume incursion from the MPS discharge.

These studies were also sufficiently sensitive to detect important regional changes to rocky shore communities unrelated to MPS operation. These included the introduction and spread of an exotic red alga *Antithamnion pectinatum*, a species native to the Pacific Ocean and not previously reported in the North Atlantic, and an increase in abundance of *Fucus* over the last 15 years.

In summary, the current rocky intertidal monitoring program provides sufficient characterization of local rocky shore communities to detect and document ecologically significant changes, both related and unrelated to MPS operation. Equally important, this long-term monitoring database provides a useful foundation from which further changes related to MPS operation or other factors can be assessed.

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Benthic Infauna

Benthic Infauna

Introduction

Sedimentary benthic habitats in the vicinity of Millstone Power Station (MPS) and elsewhere support rich and diverse infaunal invertebrate communities. These communities are important as a source of food for numerous animal 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 natural activities of infauna, such as feeding, burrowing, and tube-building, also promote nutrient recycling from sediments to the water column (Goldhaber *et al.* 1977; Aller 1978; Gaston and Nasci 1988), and aid transfer of oxygenated water deeper into sediments. It is clear that the presence of a diverse benthic infaunal community is essential to the health of coastal marine ecosystems.

The close association of benthic communities with sediments, where most pollutants ultimately accumulate, also makes infaunal communities effective indicators of acute and chronic environmental stresses (Warwick 1986; Diaz and Schaffner 1990; Warwick *et al.* 1990; Somerfield *et al.* 1995; Zajac *et al.* 1998). Changes in benthic community structure and abundance that follow disturbance (Boesch and Rosenburg 1982; Young and Young 1982; Warwick *et al.* 1987, 1990; Gaston and Nasci 1988; Regnault *et al.* 1988; Rees and Eleftheriou 1989; NAESCO 1994; Prena 1995; Somerfield *et al.* 1995) provide a useful baseline in the evaluation of impacts and recovery sequences that take place in benthic marine systems.

Coastal benthic communities are subject to a high degree of environmental variability (Holland 1985; Nichols 1985; Holland *et al.* 1987; Warwick 1988; Rees and Eleftheriou 1989; Boero 1994). The interaction of many physical and biological factors creating this natural variability, and their effects on the structural and functional ecology of benthic communities (Diaz and Schaffner 1990; Hutchings 1998; Zajac *et al.* 1998, 2000), hinders prediction of how these communities respond to disturbance. Long-term monitoring studies are essential to assess changes in marine benthic communities (Thrush *et al.* 1994; Prena 1995). Such studies are the principal means of documenting and characterizing changes in community structure and fluctuations in species abundance, which occur in response to climatic conditions (Boesch *et al.* 1976; Flint 1985; Jordan and Sutton 1985), to variations in biological factors (Levinton and Stewart 1982; Woodin 1982; Kneib 1988), and to human activities. Studies of subtidal benthic infauna have identified impacts to communities that were attributed

to Unit 3 intake construction (NUSCO 1987, 1988a) and to 3-unit operations (NUSCO 1997), as well as to regional shifts in species composition and abundance that are assumed the result of natural events. The focus of this monitoring program is to measure infaunal species composition and abundance, to identify spatial and temporal patterns in community structure and abundance, and to assess whether observed changes could possibly have been the result of construction and operation of MPS. This report presents results from the 2002 sampling year, and compares them to results summarized from previous years (1980-2001).

Materials and Methods

Subtidal infaunal communities in the vicinity of MPS were sampled twice per year (June and September) from 1980 through 2002 at four stations (Fig. 1). The Giants Neck station (GN), located 6 km west of MPS, is outside the area potentially affected by power plant operations. Data from GN are 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 MPS 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 MPS. The area encompassing this station experiences increases in surface water temperatures of 0.8 to 2.2°C above ambient during some tidal stages (primarily ebb tide) due to the 3-unit thermal discharge of MPS (NUSCO 1988b).

At each station, ten replicate samples (0.0079 m² 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 returned to the laboratory. Samples 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 with 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
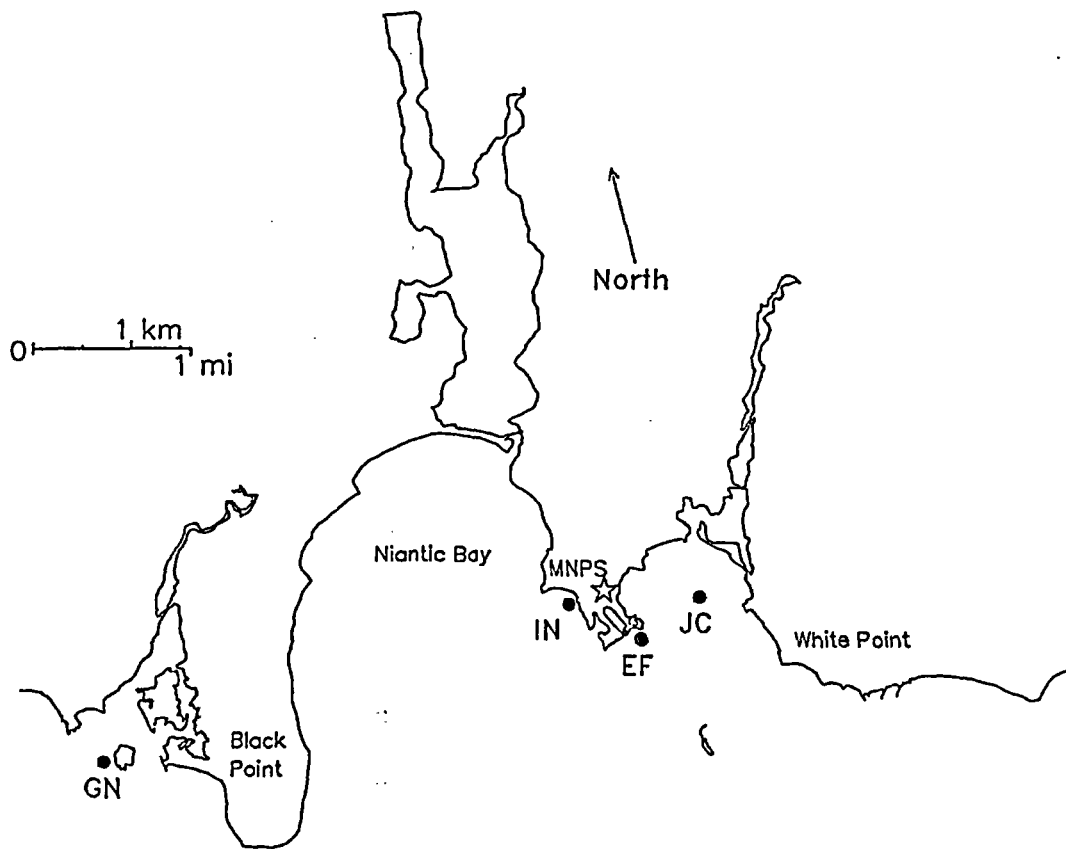
Fig. 1. Map of the Millstone Point area showing the location of infaunal sampling stations (EF=Effluent, GN=Giants Neck, IN=Intake, JC=Jordan Cove) established as part of the long-term monitoring program for Millstone Power Station.

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 and the smaller 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

Trend Analysis

The nonparametric (*i.e.*, distribution-free) Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether time-series data on community abundance, numbers of species, and selected taxa over the entire study period exhibited significant trends. Monthly (June and September) data were plotted

against time; however, annual means of community and specific population abundance were used as variables in the trend analysis.

Community Analyses

Comparisons of annual collections were made at each station 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 group-average-sorting, clustering algorithm was applied to the resulting similarity matrix (Lance and Williams 1967). Linear regression methods were used to examine long-term trends in community density at each station.

Non-Metric Multi-Dimensional Scaling (MDS)

Non-Metric Multi-Dimensional Scaling (MDS) was applied to our data using methods introduced by Shepard (1962) and Kruskal (1964). MDS was applied to the same similarity matrix used in the classification and cluster analyses. This analysis was performed with the PRIMER suite of programs developed by the Plymouth Marine Laboratory (Warwick and Clarke 1991; Clarke 1993). The purpose of MDS is to construct a map or configuration of samples in a two-dimensional plot. There are no absolute similarity values associated with MDS. Interpretation of sample placement on a configuration/map is relative in nature, i.e., more similar samples are closer together. The iterative process of MDS seeks to position the elements in the configuration/map with the least amount of distortion (stress) derived from the original data matrix. Stress values less than 0.2 indicate a potentially useful 2-dimensional picture. Results of the cluster analyses are compared with the ordination plot as a check of the adequacy and mutual consistency of both representations.

Results

Sedimentary Environment

Sedimentary environments at infaunal sampling stations in the vicinity of MPS were analyzed for mean grain size and silt/clay content (Fig. 2). Sediment grain size means in 2002 (June and September, respectively) were 0.51 mm and 0.49 mm at EF, 0.31 mm and 0.43 mm at GN, 0.24 mm and 0.31 mm at IN, and 0.16 mm and 0.27 mm at JC (Fig. 2). Mean grain sizes during 2002 were within the ranges of previous years at all stations. Sediment silt/clay contents in 2002 were highest at JC (17.9% and 11.4% in June and September, respectively). At all other stations, sediment silt/clay contents were less than 10% during 2002: EF (1.4% in June and September); GN (7.5% and 3.1%); and IN (6.7% and 4.6%) in June and September, respectively. Silt/clay estimates at all stations in 2002 were within historical ranges, although

silt/clay content at GN in September 2002 was the second lowest observed during the study, next to the historical low value recorded in the previous September.

General Community Composition

Descriptions of benthic community composition include total numbers of individuals and species, along with numbers of individuals and species within major taxonomic groups. Station totals for number of individual organisms collected in 2002 (Fig. 3) were highest at JC (6,556), and lowest at EF (2,553) and intermediate IN (3,883) and GN (3,073). Totals at JC and IN were within historical ranges for each station. However, totals at EF and GN were the lowest observed at these sites during the study.

Polychaetes were the most abundant in terms of individuals collected at all four stations in 2002: 4,752 polychaetes were collected at JC, 2,689 at IN, 1,807 at GN and 1,080 at EF (Fig. 4). Polychaete abundance in 2002 at GN was the lowest observed historically at this site. Oligochaetes were the second most abundant group at all sites in 2002: annual counts at JC (1,118), EF (988), GN (967), and IN (866) were all within historical ranges.

Arthropods were the third-ranked taxonomic group in 2002 at EF, GN and IN (331, 255 and 211 individuals, respectively), and ranked fourth in total abundance at JC (323). Arthropod abundances in 2002 increased from the historical lows recorded in 2001at EF and GN. Molluscs ranked fourth in abundance at EF (96), GN (35) and IN (104) while ranking third at JC (343). At all stations, mollusc abundances in 2002 were within historical ranges.

Annual total numbers of species in 2002 (Fig. 5) were highest at IN (130), followed by JC (106), EF (103), and GN (99). Similar to total abundance discussed above, most of the species identified were polychaetes: 72 at IN, 63 at JC, 53 at GN and 52 at EF. Total and Polychaete species numbers for all stations were within their historical ranges. Numbers of arthropod species in 2002 were highest at IN (31) followed by GN (28), EF (26), and JC (20); all station values were within historical ranges. The relationship of numbers of arthropod species among stations has remained relatively stable over years, with IN generally having highest arthropod species numbers and JC most often the lowest. Numbers of mollusc species in 2002 was highest at IN (24), with lower, more similar numbers at EF (18), JC (18), and GN (14). During the 2002 study, IN was distinct in having highest numbers of species in all the above categories.
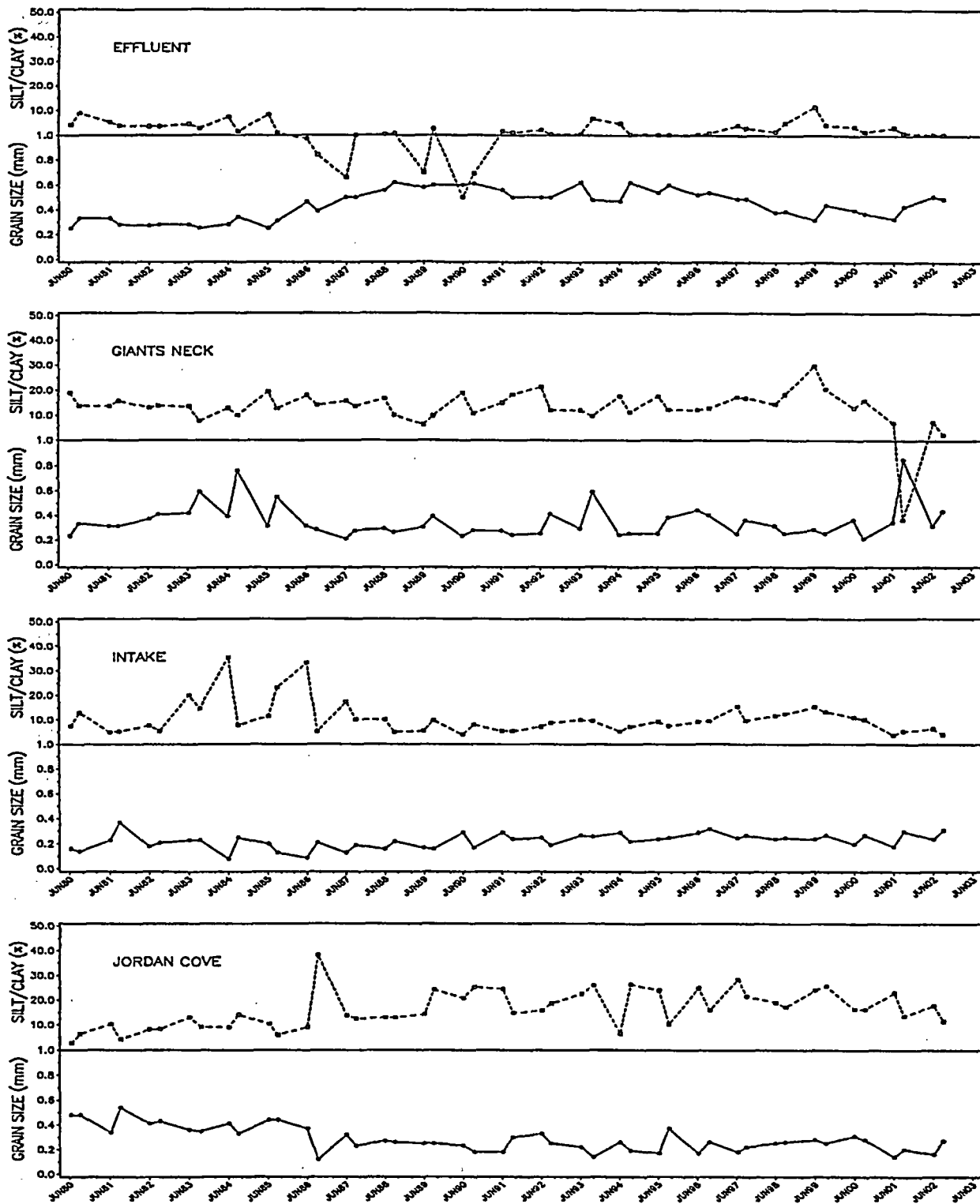
Fig. 2. Mean grain size (mm) and silt/clay (%) of sediments at MPS infaunal stations for June and September from 1980 to 2002.
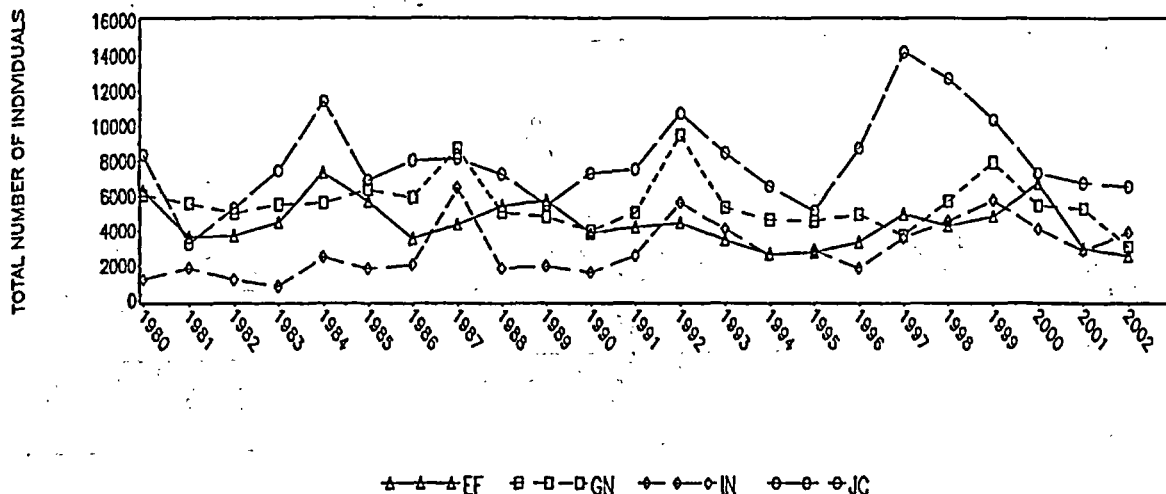
Fig. 3. Total annual abundance of individual organisms collected during 1980-2002 at MPS infaunal stations.

Faunal Abundance

Mean faunal densities (individuals per core) in June and September 2002 collections, respectively, were 105 and 150 at EF, 161 and 146 at GN, 191 and 198 at IN, and 284 and 372 at JC (Fig. 6). Densities were within historic ranges established since 1980 at IN and JC. However, 2002 faunal densities at EF in June and at GN in September were the lowest observed since 1980. Analyses of long-term mean community abundance since 1980 indicated a significant increase in number of individuals collected at IN. There were no significant trends in mean density evident at any other station during the 1980-2002 period.

Numbers of Species

Mean numbers of species per core in 2002 June and September (respectively) were 21 and 26 at EF, 24 and 21 at GN, 31 and 25 at IN, and 28 and 29 at JC (Fig. 7). All 2002 means were within the range of respective means observed at each station established for previous study years. Trend analysis revealed a significant long-term increase in annual mean species numbers per core over the entire study period (1980-2002) at IN. There was a significant decreasing long-term trend at EF. There was no long-term trend in annual mean number of species per core at GN or JC.

Community Dominance

Comparison of relative abundances of common taxa provides insight into annual variation and long-term infaunal community structure. The relative abundances of infaunal taxa were calculated for each

station during 2002 and compared to means over the entire 23-year study period (1980-2002; Table 1). Of the many taxa collected in 2002, only four (representatives of the class Oligochaeta, and the polychaetes *Aricidea catherinae*, *Tharyx* spp. and *Protodorvillea gaspeensis*) had relative abundances of >10% at one or more stations. The five top-ranked taxa at each station in 2002 accounted for 64% or more of the total individuals, and were: Oligochaeta, and the polychaetes *Protodorvillea gaspeensis*, *Polycirrus eximius*, *Parapionosyllis longicirrata*, and the amphipod *Microdeutopus gryllotalpa* at EF; Oligochaeta, and the polychaetes *Tharyx* spp., *Aricidea* catherinae, *P. eximius*, and *P. gaspeensis* at GN; *A. catherinae*, Oligochaeta, and the polychaetes *Pygospio elegans*, *Tharax* spp. and *Exogone hebes* at IN; and *A. catherinae*, Oligochaeta, the polychaetes *Tharyx* spp. and *Scoletoma tenuis*, and the mollusk *Nucula annulata* at JC. In 2002, four taxa were among the top ten dominants at all sites: Oligochaeta, *Polycirrus eximius*, *Protodorvillea gaspeensis*, and *Tharyx* spp.

Variability in relative abundance levels and shifts in rankings among dominant taxa from year-to-year have been typical of benthic communities at each monitoring site, and are often related to short-term pulses in abundance of particular taxa. Some of these pulses occur at more than one site, suggesting regional factors. For example, both *Mediomastus ambiseta* and *Prionospio steenstrupi* have exhibited abundance cycles historically that included strong regional pulses. Both species were in the low abundance phase of their cycle at all stations in 2002. At times, pulses of a particular species only occur at a specific station. In 2002, we observed an increase in relative abundance of the amphipod
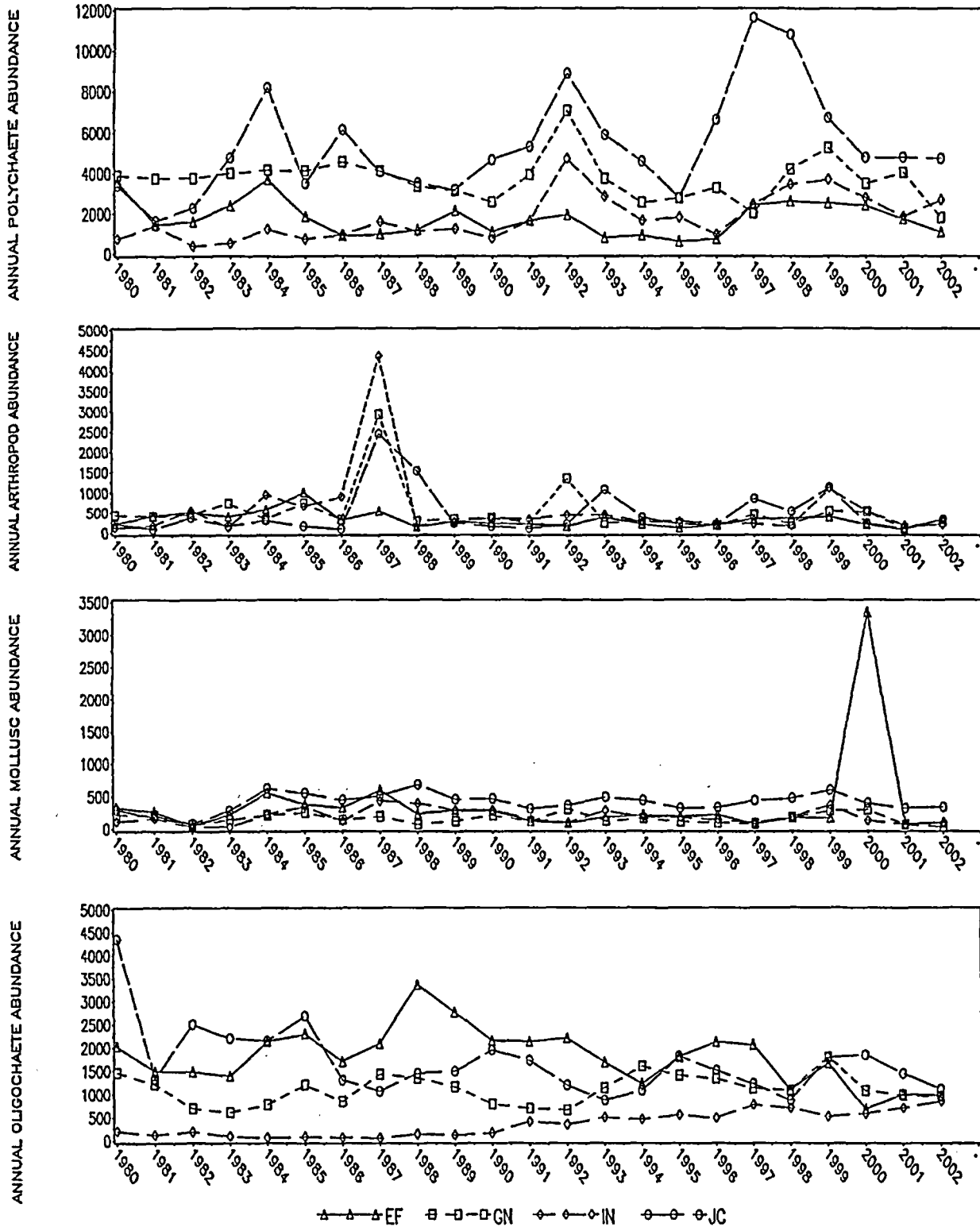
Fig. 4. Total abundance of individuals by major taxonomic groups at the MPS infaunal stations 1980-2002.
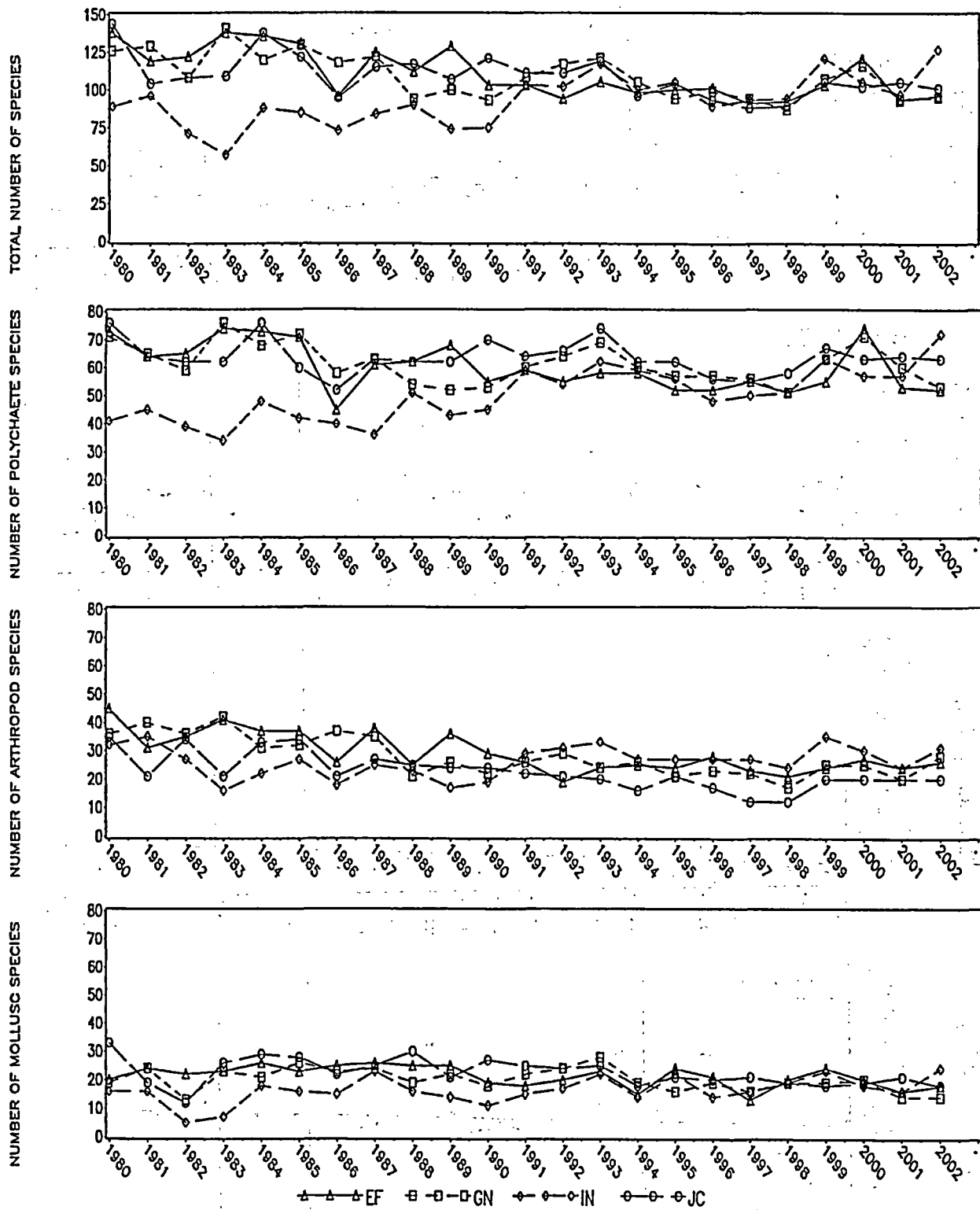
Fig. 5. Total number of species within major taxonomic groups, and overall total collected, at the MPS infaunal stations during 1980-2002.
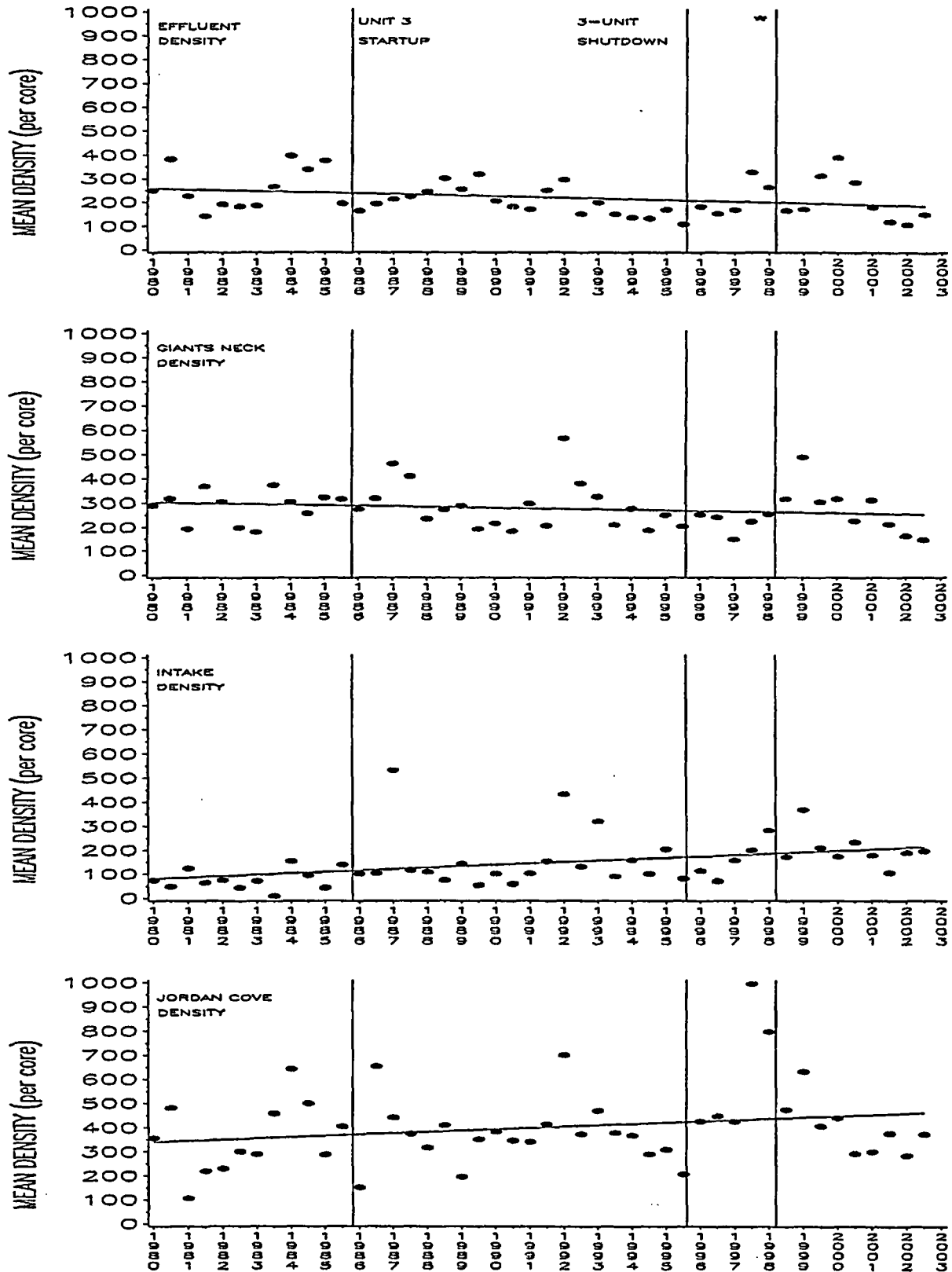
Fig. 6. Mean faunal density (per core) at Millstone subtidal stations in June and September for every year from 1980 through 2002. Vertical line marked with (*) indicates Unit 3 restart in July 1998.
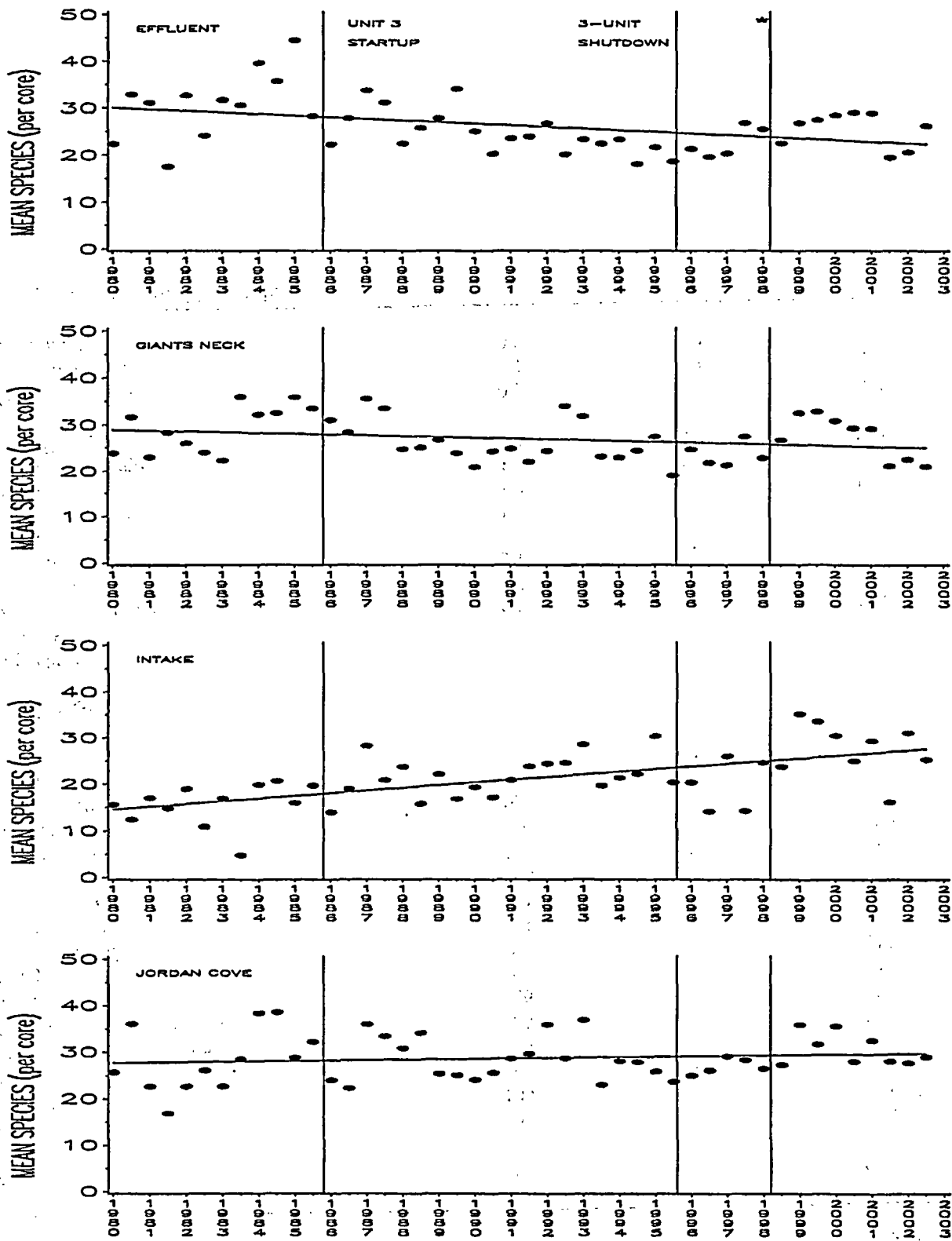
Fig. 7. Mean number of species (per core) at subtidal stations in June and September for every year from 1980 through 2002. Vertical line marked with (*) indicates Unit 3 restart in July 1998

TABLE 1. Relative abundance (%) and coefficient of variability (C.V.) of each of the ten most abundant taxa collected at Millstone infaunal monitoring stations during 2002 and for the overall study period (1980-2002).

	2002	1980-2002	1980-2002
EFFLUENT	%	%	C.V.
Oligochaeta	39.4	43.0	7.0
Protodorvillea gaspeensis	10.8	5.3	9.1
Polycirrus eximius	5.4	6.3	26.0
Microdeutopus gryllotalpa	4.5	0.3	41.9
Parapionosyllis longicirrata	4.5	2.9	29.4
Aricidea catherinae	3.6	1.9	16.7
Tharyx spp.	2.9	4.6	40.0
Ampelisca vadorum	2.5	1.5	22.7
Scoletoma tenuis	1.8	1.2	15.0
Pagurus acadianus	1.8	4.7	40.4
Mediomastus ambiseta	0.3	2.6	36.8
Mytilus edulis	0.1	2.6	84.2
Tellina agilis	0.7	2.1	16.6
Rhyncocoela	1.4	1.9	21.7
GIANTS NECK			
Oligochaeta	31.5	21.1	6.1
Tharyx spp.	30.8	23.2	9.2
Aricidea catherinae	6.6	11.8	14.8
Polycirrus eximius	4.7	3.0	15.6
Protodorvillea gaspeensis	3.1	3.0	7.8
Ampelisca vadorum	2.5	1.8	40.5
Ampharete americana	1.5	0.6	18.0
Exogone dispar	1.4	2.2	10.9
Polydora caulleryi	1.1	1.1	22.4
Capitella spp.	0.9	0.8	14.8
Mediomastus ambiseta	0.8	6.8	24.7
Prionospio steenstrupi	0.1	4.3	55.3
Scoletoma tenuis	0.2	1.7	15.1

C.V. = (Standard Error/Mean) x 100

TABLE 1. (cont.)

	2002	1980-2002	1980-2002
INTAKE	%	%	**C.V.**
Aricidea catherinae	32.5	13.2	22.4
Oligochaeta	22.3	13.1	15.0
Pygospio elegans	6.0	3.6	40.4
Tharyx spp.	5.6	3.9	15.8
Exogone hebes	5.1	5.1	15.3
Protodorvillea gaspeensis	3.1	2.5	16.3
Euclymene spp.	2.9	0.5	29.9
Mediomastus ambiseta	2.2	8.4	31.7
Leptocheirus pinguis	1.7	4.2	69.9
Capitella spp.	1.1	3.1	22.2
Prionospio steenstrupi	0.1	4.8	69.7
Nucula annulata	0.5	2.7	25.8
Tellina agilis	0.5	2.7	16.6
JORDAN COVE			
Aricidea catherinae	37.3	19.9	13.6
Oligochaeta	17.1	23.6	9.3
Tharyx spp.	5.5	4.3	12.0
Scoletoma tenuis	4.8	5.0	9.9
Nucula annulata	3.6	1.7	14.1
Exogone hebes	3.4	1.3	24.0
Protodorvillea gaspeensis	3.2	0.7	20.2
Leptocheirus pinguis	3.0	4.2	31.7
Mediomastus ambiseta	2.7	13.9	26.9
Maldanidae	2.2	0.8	50.8
Prionospio steenstrupi	0.0	4.0	50.3
Polycirrus eximius	1.2	3.7	18.5
Capitella spp.	0.7	1.4	15.9

C.V. = (Standard Error/Mean) x 100

Microdeutopus gryllotalpa at EF. This species was not among the overall study period dominants, and has never been among the annual dominant taxa in previous study years. Some other site-specific shifts in relative abundance noted during 2002 include increases of *Pygospio elegans* and *Euclymene* spp. at IN, and of *Exogone hebes* and *Protodorvillea gaspeensis* at JC.

Other shifts in relative abundance in 2002 were not consistent among sites. For example, when compared to overall 1980-2002 study period, *Aricidea*

catherinae abundance in 2002 was high at IN and JC, but low at GN and EF. Oligochaete relative abundance in 2002 was high at GN (31.5%) and IN (22.3%) when compared to the overall study period (21.1% and 13.1%, respectively), but low at JC (17.1%) compared to 1980-2002 (23.6%). Relative abundance of *Tharyx* spp. was also high in 2002 at GN (30.8% versus 23.2% for 1980-2002) and IN (5.6% versus 3.9%), but lower at EF (2.9% versus 4.6%).

Dominant Taxa

Seven infaunal taxa were selected as representative of sites affected by construction and operation of MPS. These taxa are useful biomonitoring tools because they are also indicators of natural environmental stress (*e.g.*, storms). Trends in their abundance were examined using the same techniques as those applied to time-series of overall community abundance and numbers of species.

Oligochaetes - Representatives of the class Oligochaeta are among the dominant taxa throughout the study period at all stations, accounting for 13-43% of the organisms collected since 1980. Oligochaete abundance during June and September 2002 averaged 43 and 56/core at EF, 57 and 40/core at GN, 47 and 40/core at IN, and 64 and 48/core at JC. Oligochaete densities in 2002 were within the ranges of densities for previous study years at all stations (Fig. 8a-d). Long-term (1980-2002) trend analysis of oligochaete abundance revealed a significant increase at IN (Fig. 8c) and a significant decrease at JC (Fig. 8d). No significant long-term trends were detected at either EF or GN.

Aricidea catherinae - The polychaete *A. catherinae* was also among the top ten dominant taxa at all stations in 2002 (Table 1). Average densities in June and September 2002 (Fig. 8e-h) were highest at JC (59 and 185/core, respectively) and IN (25 and 101/core, respectively). Lower average densities occurred at EF (2 and 7/core) and GN (6 and 14/core). Average densities at all stations in 2002 were within historical ranges.

Long-term (1980-2002) trend analysis indicated a significant increase in abundance of *Aricidea catherinae* at IN and JC, and a significant decreasing trend at GN. *Aricidea* abundances at EF have been low throughout the study period, with no apparent trend.

Mediomastus ambiseta - Although among the ten numerically abundant taxa at IN and JC in 2002 (Table 1), *Mediomastus* has exhibited periodic pulses in abundance that has placed it among the historical dominants at all stations. Average June and September densities in 2002 were among the lowest values recorded at all stations. Highest *M. ambiseta* densities in 2002 were at JC (11 and 6/core in June and September, respectively (Fig. 8i-l). At IN, *M. ambiseta* densities were lower, 5 and 3/core in June and September, respectively. *Mediomastus ambiseta* averaged between 1-5/core at the other stations. Although low, mean densities during 2002 were within the range of values observed at all stations

since 1980. There were no significant long-term trends in abundance at any station over the study period (1980-2002).

Tharyx spp. - *Tharyx* spp. ranked second in relative abundance in 2002 at GN (30.8%), third at JC (5.5%), fourth at IN (5.6%), and seventh at EF (2.9%; Table 1). Densities during June and September 2002 (Fig. 8m-p) were highest at GN (52 and 43/core), with lower densities observed at JC (17 and 20/core), IN (14 and 8/core), and EF (2 and 5/core). Analysis of long-term data (1980-2002) indicated a significant increasing trend at GN, IN and JC. There was no trend in *Tharyx* spp. density at EF.

Polycirrus eximius - The polychaete *P. eximius* was among the dominant taxa during 2002 at EF and GN, ranking third and fourth, respectively (Table 1). *Polycirrus eximius* has historically been among the dominants at JC also, although not in 2002. June and September mean densities in 2002 were 5 and 8/core at EF, 5 and 9/core at GN and 2 and 6/core at JC (Fig. 8q-s). High annual variability in density has been characteristic of *P. eximius* over the study period at EF and JC, with pulses of abundance observed both before and after Unit 3 startup in 1986 and following the MPS extended shutdown from 1996-98. Densities have been lower and appear to exhibit a gradual increasing tendency. Smaller pulses coinciding with those observed at EF and JC also occurred at GN. The densities observed at all stations in 2002 were among the lowest recorded during the entire study. The erratic nature of population density data over the study period (1980-2002) at each station resulted in no significant long-term trends.

Protodorvillea gaspeensis - *P. gaspeensis* was among the top ten dominant taxa in 2002 at all stations (Table 1). *P. gaspeensis* rankings during 2002 were: second at EF, fifth at GN, sixth at IN, and seventh at JC. In June and September densities were 15 and 12/core at EF, 5/core in each sample period at GN, 4 and 8/core at IN, and 18 and 13/core at JC. Densities at EF, GN and IN in 2002 were within historical ranges; JC densities were the highest values observed since 1980 (Fig. 8t-w). Trend analysis on the entire time-series (1980-2002) indicated a significant increasing trend in *P. gaspeensis* density at IN and JC. No significant long-term trends were detected at either EF or GN.

Nucula annulata - The bivalve mollusc *N. annulata* was among the infaunal community dominants only at JC in 2002, ranking fifth in relative abundance (Table 1). This species has also been among the overall study period dominants at JC,
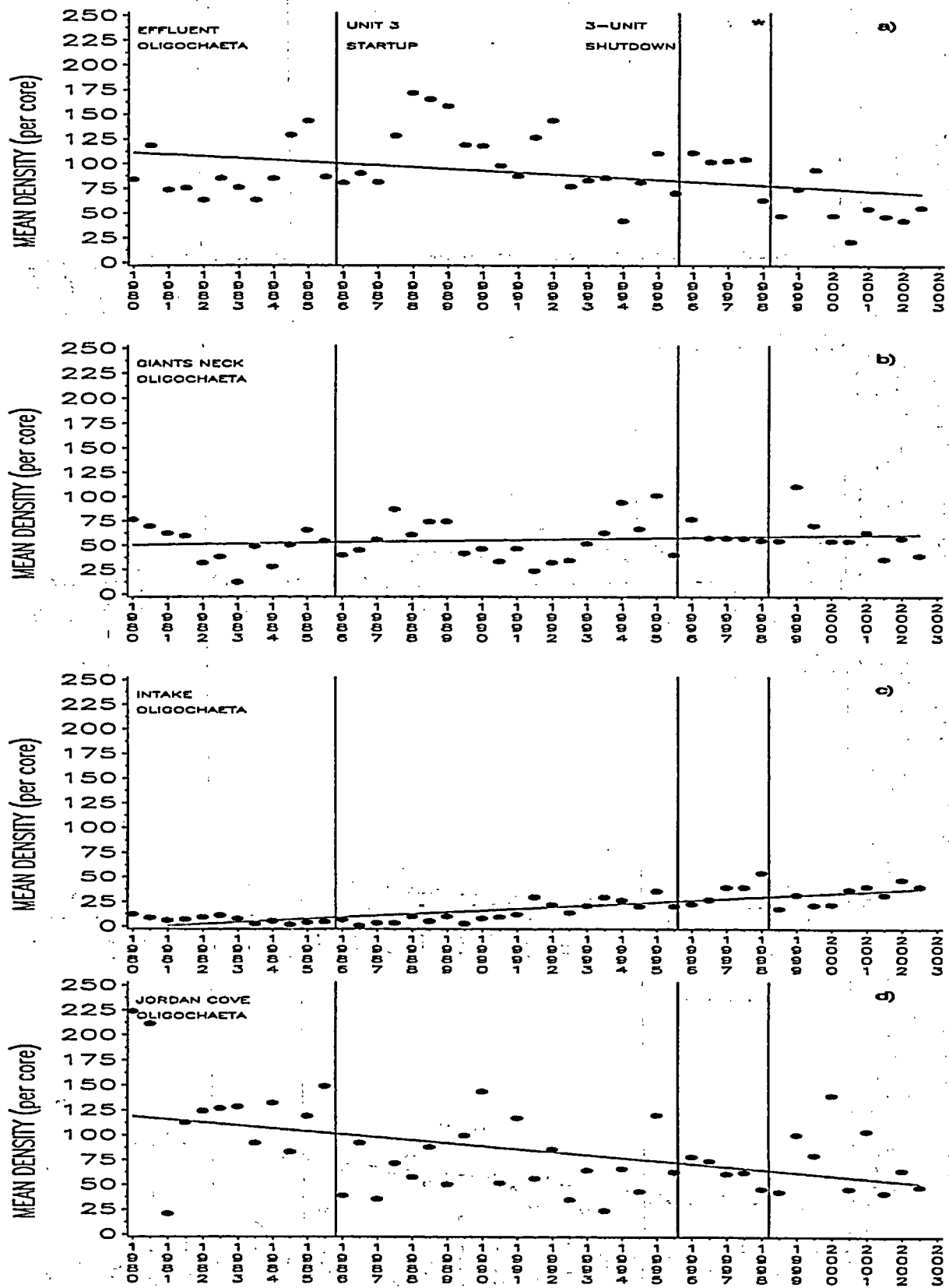
Fig. 8. Mean density of selected taxa in June and September and linear regression of the 1980-2002 period. Vertical line marked with (*) indicates Unit 3 restart in July 1998.
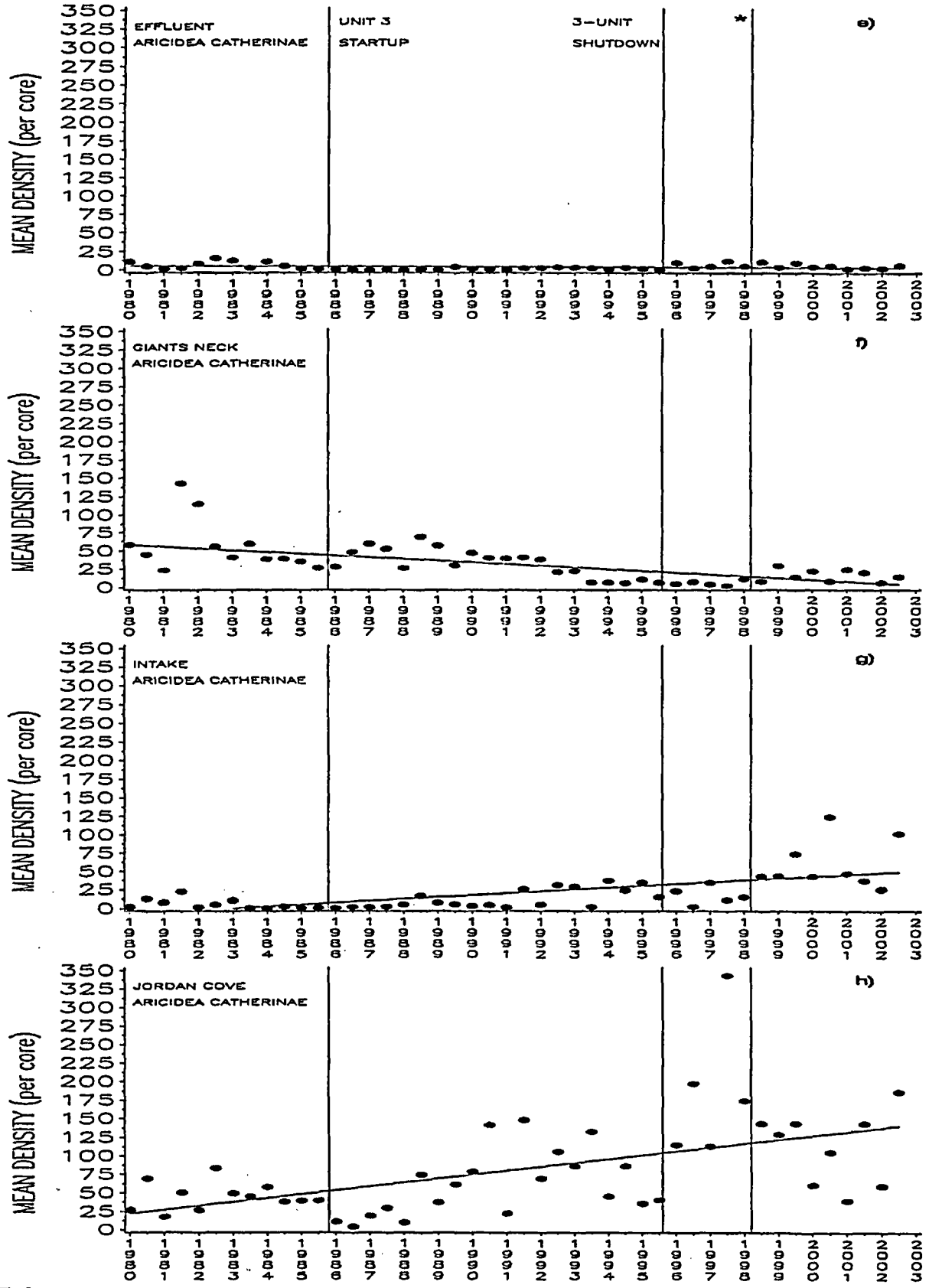
Fig 8. (cont.)
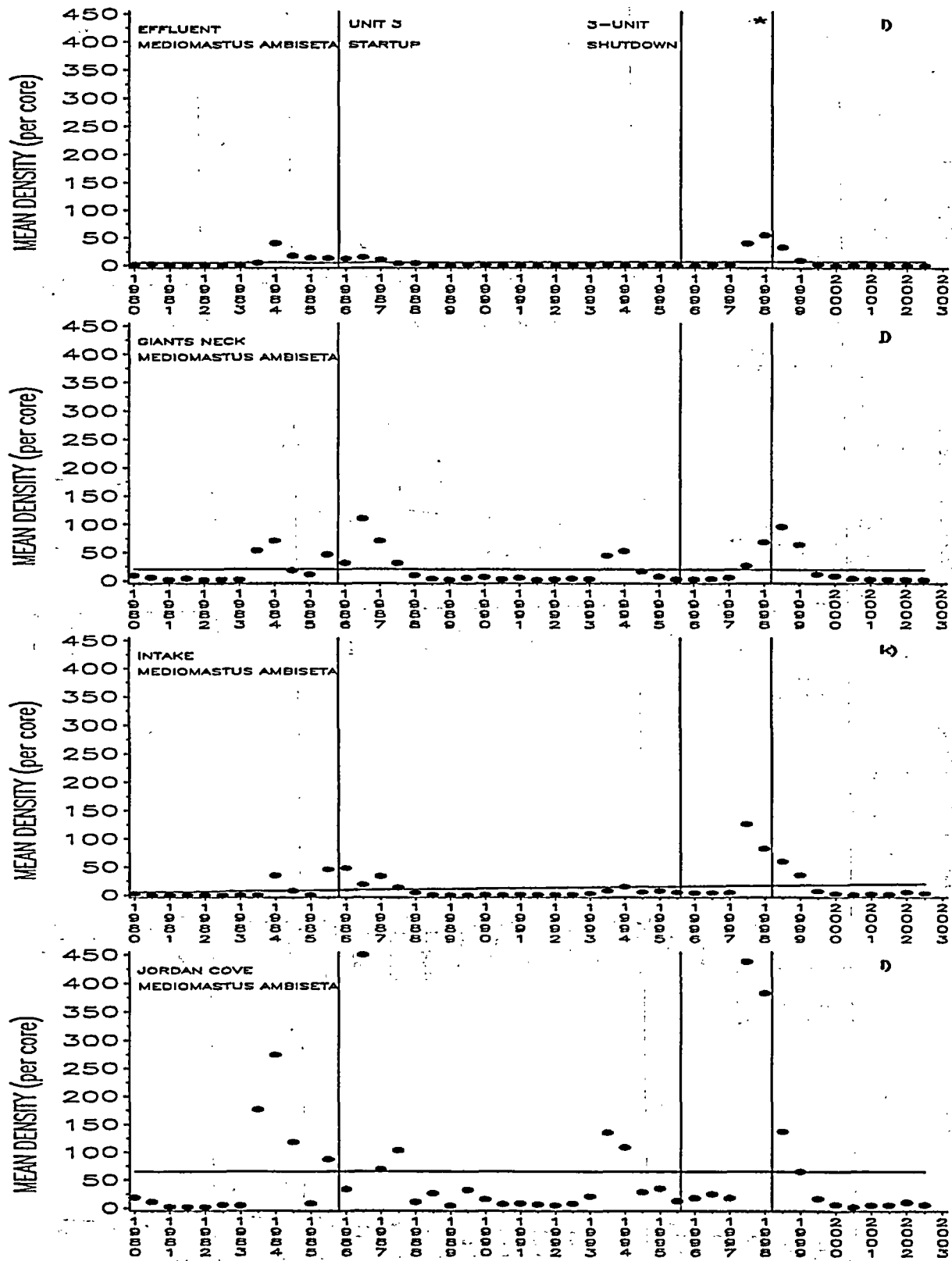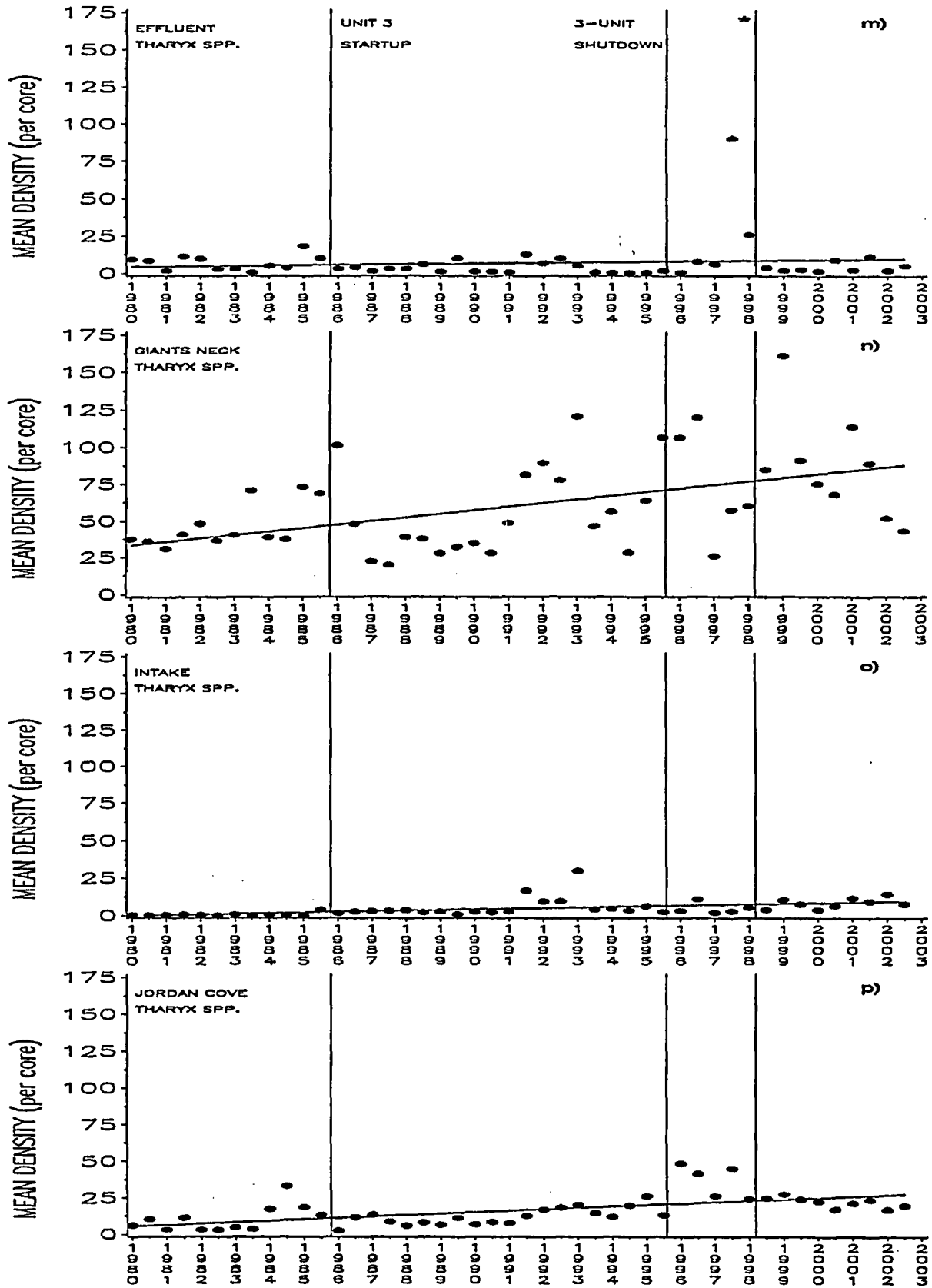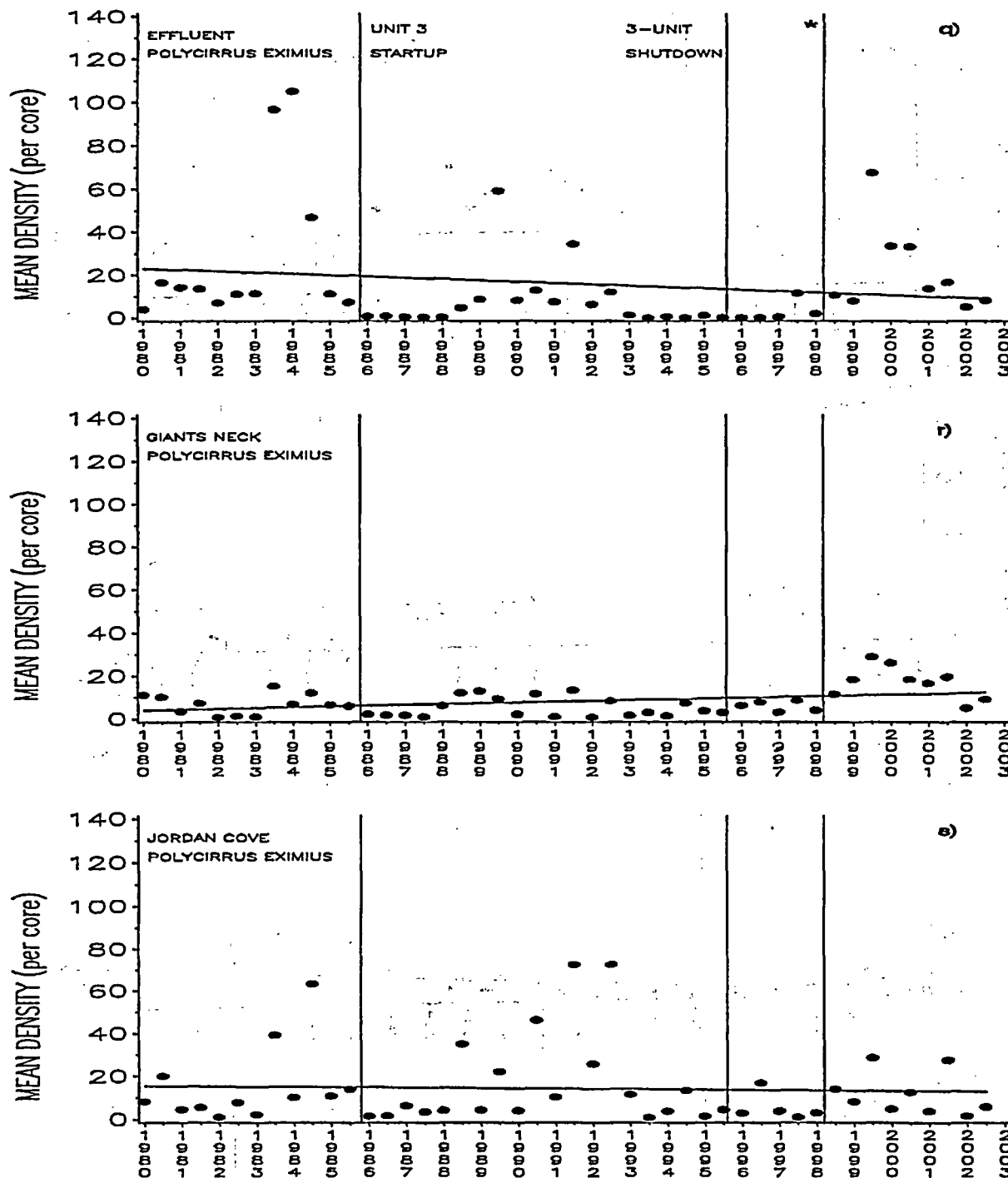
Fig. 8 (cont.)

Fig. 8. (cont.)

Fig. 8. (cont.)

ranking ninth. Data from IN are also included because this species has exhibited increases in abundance concurrent with construction and operation of Unit 3 during the 1984-1990 period. June and September mean *N. annulata* densities were 17/core and 7/core, respectively at JC, and 1/core in

both periods at IN (Fig. 8x-y). These density estimates were within the range of estimates from previous years. Trend analysis of the 23-year time-series revealed a significant increasing trend in *N. annulata* density at JC, but no significant trend at IN.
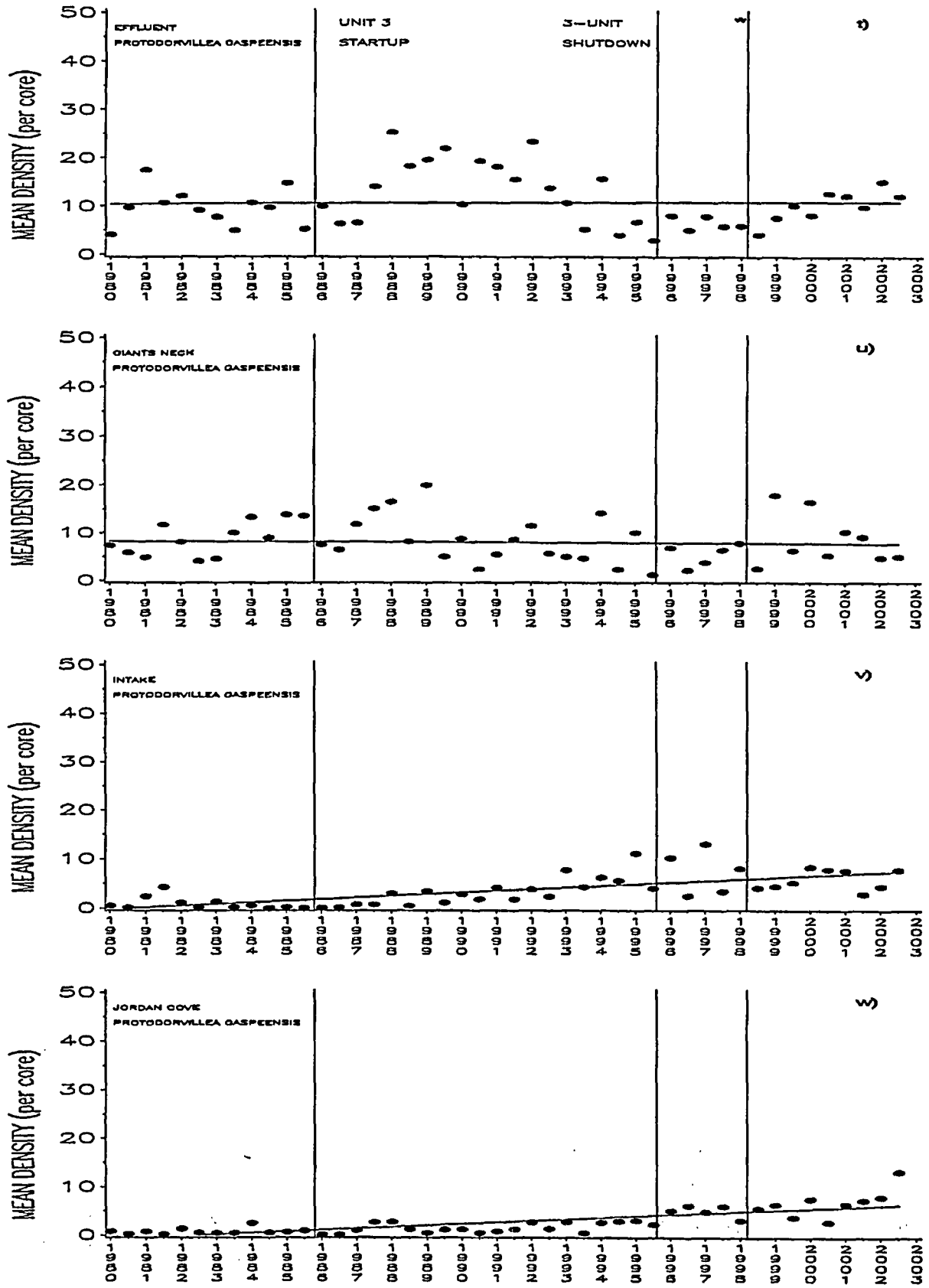
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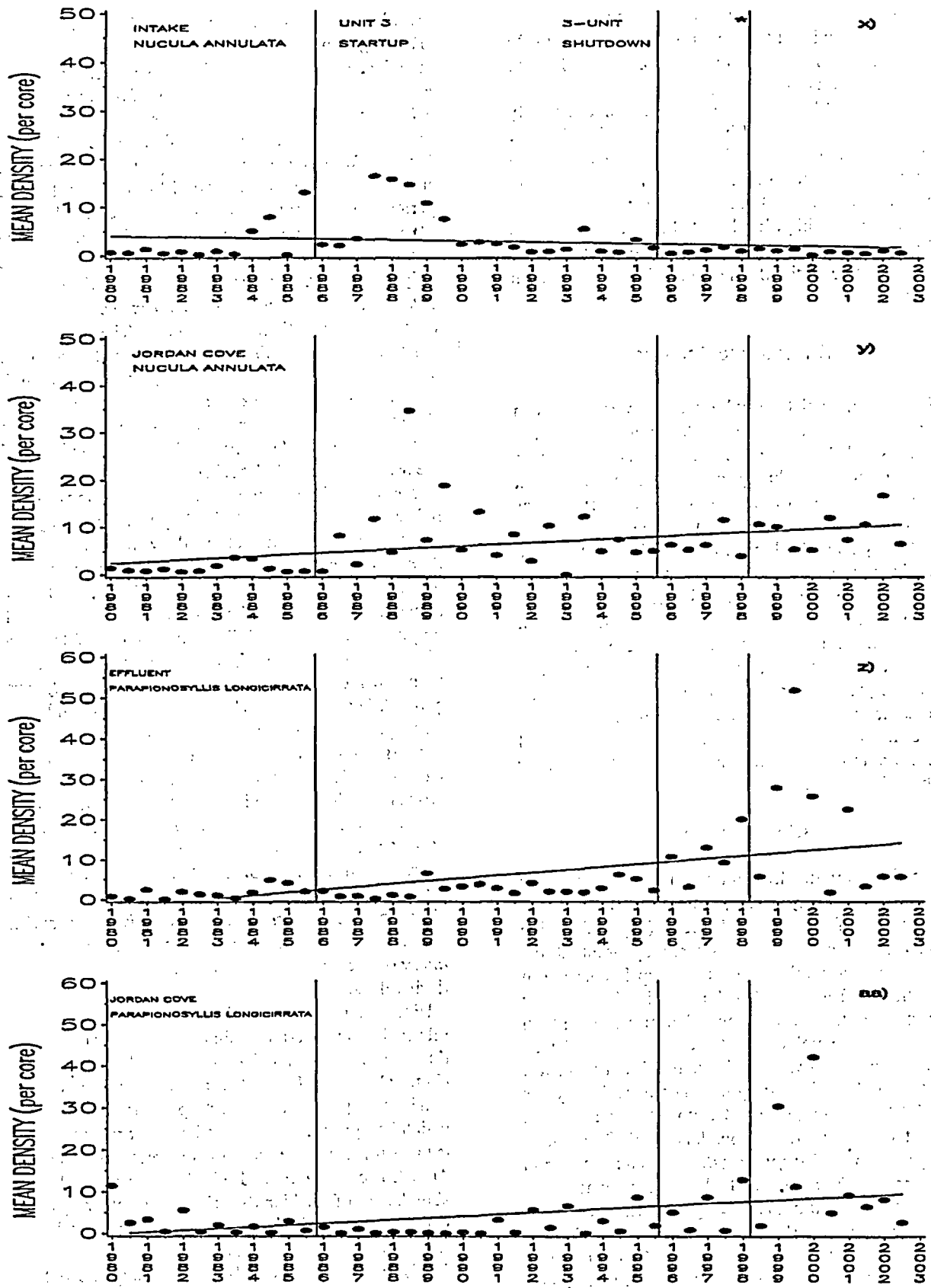
Fig. 8. (cont.)

Parapionosyllis longicirrata - *P. longicirrata* was among the dominant taxa at EF in 2002, ranking fifth in relative abundance (Table 1). At JC, *P. longicirrata* was not among the dominants in 2002, but is included here because it has shown a relatively steady increase since 1991. Mean *P. longicirrata* densities during June and September 2002 were 8 and 3/core at JC, and 6/core at EF in both sampling periods. Densities at both stations have decreased in 2002 relative to high levels noted in recent past years (Fig. 8z-aa); however, analyses of the 23-year time-series indicate a significant increasing trend at both EF and JC.

Multivariate Analyses

Several infaunal taxa exhibited temporal trends in abundance associated with construction and operation activities at MPS. Trends in the abundances of other taxa were common throughout the greater MPS area, but were unrelated to the power plant. Each species represents one component of a complex community, affected by many biological and physical factors. To provide a more complete characterization of local infaunal communities, 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. Relationships of annual collections based on comparisons of these indices are illustrated as a clustering dendrogram and as a two-dimensional representation of a multi-dimensional scaling plot for each station (Fig. 9).

Cluster Analysis

The clustering dendrogram based on EF data shows an overall similarity among collections from the 23 study years of approximately 63% (Fig. 9a). Five groups of annual collections were apparent at the 70% or greater similarity level. Group I consisted of collections from 1980-1985, and these years distinguished themselves from other groups with high densities of the polychaete taxa *Polycirrus eximius*, *Tharyx* spp., *Eumida sanguinea*, and *Exogone hebes*, and the mollusc *Tellina agilis*. The next three years (1986-1988) comprised Group II, and were characterized by lowest abundances of *P. eximius*, *Tharyx* spp., *Parapionosyllis longicirrata* and *Aricidea catherinae*, and high abundances of oligochaetes, *Protodorvillea gaspeensis*, and the anemone *Haliplanella luciae*. Group II collection years coincided with the period of scour-related sediment changes (see Fig 2.), associated with Unit 3 start-up and early operational years. Group III

included annual collections from 1989 through 1996. This group was characterized by moderate abundances of a number of polychaete species (e.g., *A. catherinae*, *Parapionosyllis longicirrata*, *P. eximius* and *Protodorvillea gaspeensis*), low densities of *Mediomastus ambiseta* and high densities of *Prionospio steenstrupi*. Group IV consisted of annual collections made from 1997 through 1999, a period which encompasses the extended shutdown of MPS. The infaunal community at EF during these years was characterized by high abundances of the polychaetes *P. longicirrata*, *M. ambiseta*, *A. catherinae*, *P.eximius* and *Tharyx* spp. and the amphipod *Ampelisca vadorum*. The last three collection years (2000-2002) comprised Group V. This group was characterized primarily by high abundances of *P. eximius*, *P. gaspeensis*, and *Polydora caulleryi*, in addition to the highest abundance of *Mytilus edulis* and lowest abundance of the *M. ambiseta* and oligochaetes.

Overall similarity among annual collections at the reference site (GN) was 65% (Fig. 9b). Group I years (1980-1986) were characterized by high densities of *Aricidea catherinae* and *Mediomastus ambiseta* and the amphipod *Gammarus lawrencianus*, and low abundances of oligochaetes and the amphipod *Ampelisca vadorum*. Collections from 1987 through 1991 comprise Group II, and were characterized by high densities of *Protodorvillea gaspeensis*, *Polydora quadrilobata*, *Microphthalmus aberrans* and the amphipods *Ampelisca vadorum* and *Leptocheirus pinguis* and lower abundances of oligochaetes. Group III (1992-1999) was distinguished from other groups by high abundances of the *Tharyx* spp., oligochaetes, *Exogone dispar* and *Prionospio steenstrupi* concurrent with low abundances of *Aricidea catherinae*. Group IV at GN contained the most recent collection years (2000-2002). High abundances of *Polycirrus eximius*, *Polydora socialis*, and *P. caulleryi*, and the lowest abundance of *M. ambiseta* characterized the infaunal community during these years.

High infaunal community variability resulted in relatively low overall similarity (<55%) among the three groups of annual collections identified at IN (Fig. 9c). Group I included initial study years (1980-1983) and post Unit 3 start-up years (1988-1990) which had generally low abundances of most infaunal taxa, with the exception of moderate numbers of *Aricidea catherinae* and *Capitella* spp., and high abundance of maldanids relative to other groups. Unit 3 intake dredging, cofferdam removal and startup all occurred during the four study years (1984-1987) included in Group III. This group showed the lowest similarity to other collection years with high abundance of amphipods (*Leptocheirus pinguis*,

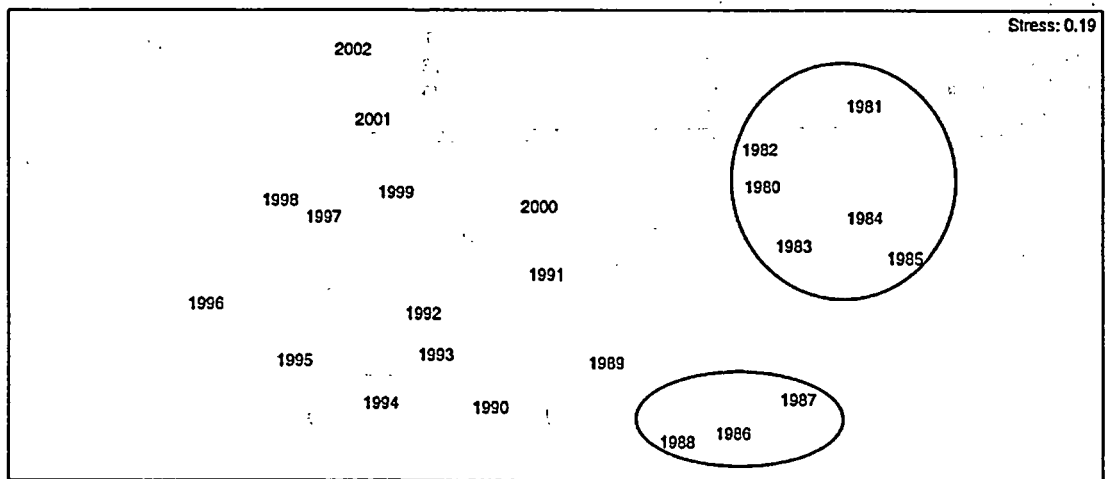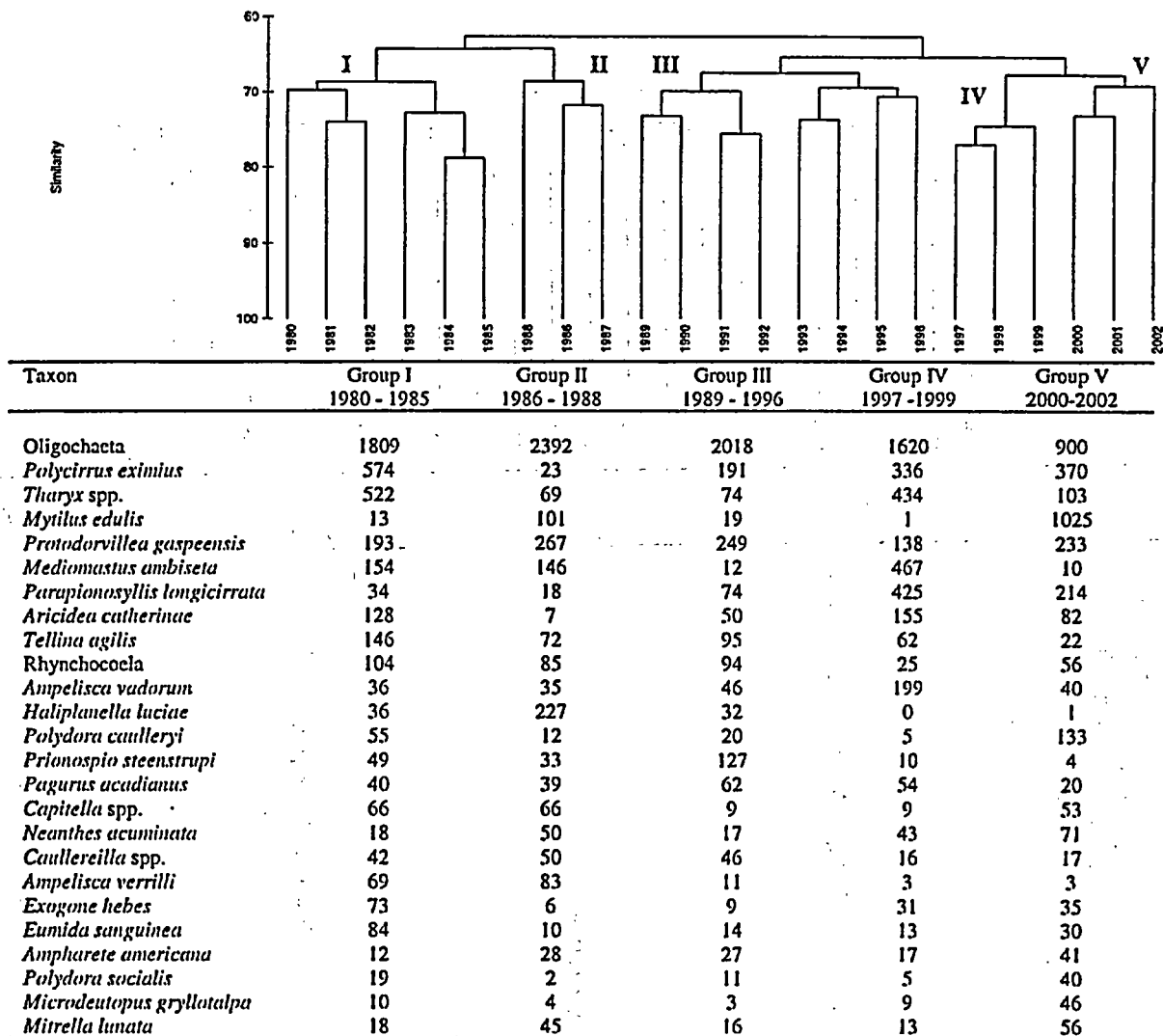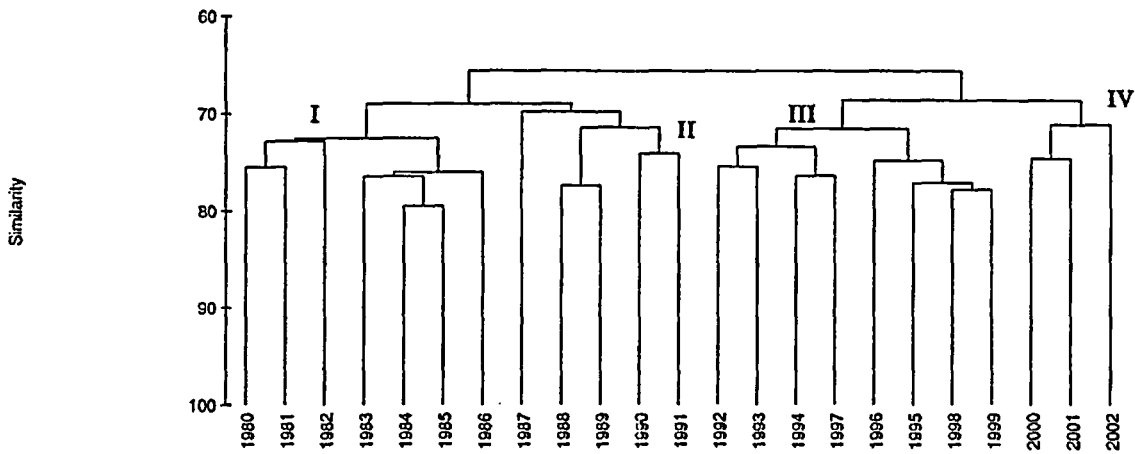| Taxon	Group I 1980 - 1985	Group II 1986 - 1988	Group III 1989 - 1996	Group IV 1997 -1999	Group V 2000-2002
Oligochaeta	1809	2392	2018	1620	900
Polycirrus eximius	574	23	191	336	370
Tharyx spp.	522	69	74	434	103
Mytilus edulis	13	101	19	1	1025
Protodorvillea gaspeensis	193	267	249	138	233
Mediomastus ambiseta	154	146	12	467	10
Parapionosyllis longicirrata	34	18	74	425	214
Aricidea catherinae	128	7	50	155	82
Tellina agilis	146	72	95	62	22
Rhynchocoela	104	85	94	25	56
Ampelisca vadorum	36	35	46	199	40
Haliplanella luciae	36	227	32	0	1
Polydora caulleryi	55	12	20	5	133
Prionospio steenstrupi	49	33	127	10	4
Pagurus acadianus	40	39	62	54	20
Capitella spp.	66	66	9	9	53
Neanthes acuminata	18	50	17	43	71
Caullereilla spp.	42	50	46	16	17
Ampelisca verrilli	69	83	11	3	3
Exogone hebes	73	6	9	31	35
Eumida sanguinea	84	10	14	13	30
Ampharete americana	12	28	27	17	41
Polydora socialis	19	2	11	5	40
Microdeutopus gryllotalpa	10	4	3	9	46
Mitrella lunata	18	45	16	13	56

Fig. 9. Clustering dendrogram of natural log-transformed density data of annual infaunal communities, and list of taxa (mean density/year) associated with major group divisions in classification and cluster analyses, and Multidimensional Scaling two dimensional plots for the years 1980-2002.

GIANTS NECK

Taxon	Group I 1980-1986	Group II 1987-1991	Group III 1992-1999	Group IV 2000-2002
Tharyx spp.	1233	740	1620	1468
Oligochaeta	979	1090	1278	1016
Aricidea catherinae	1075	917	243	315
Mediomastus ambiseta	521	263	504	56
Prionospio steenstrupi	101	316	526	17
Polycirrus eximius	123	142	148	317
Protodorvillea gaspeensis	170	201	133	168
Exogone dispar	110	114	146	112
Ampelisca vadorum	23	218	133	108
Leptocheirus pinguis	32	297	14	33
Scoletoma tenuis	140	152	43	6
Phoxocephalus holbolli	127	86	70	6
Polydora caulleryi	95	51	22	113
Polydora socialis	16	16	25	120
Microphthalmus aberrans	22	91	19	5
Polydora quadrilobata	32	68	32	5
Ampharete americana	13	27	38	59
Pagurus acadianus	27	55	36	18
Rhynchocoela	37	36	33	25
Mitrella lunata	28	15	44	30
Gammarus lawrencianus	110	54	14	17
Exogone hebes	9	9	11	48
Pholoe minuta	26	13	5	1
Phyllodoce mucosa	17	43	16	3
Mytilus edulis	2	9	14	45

Fig. 9. (cont)

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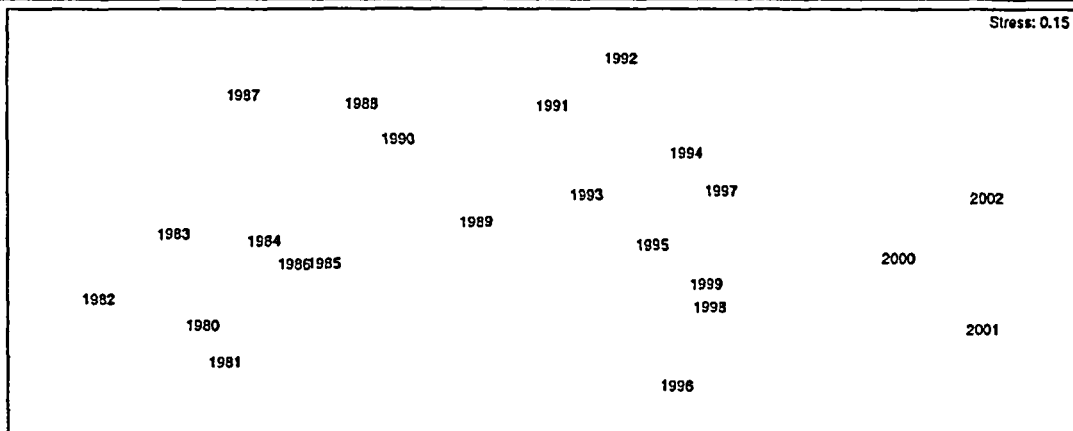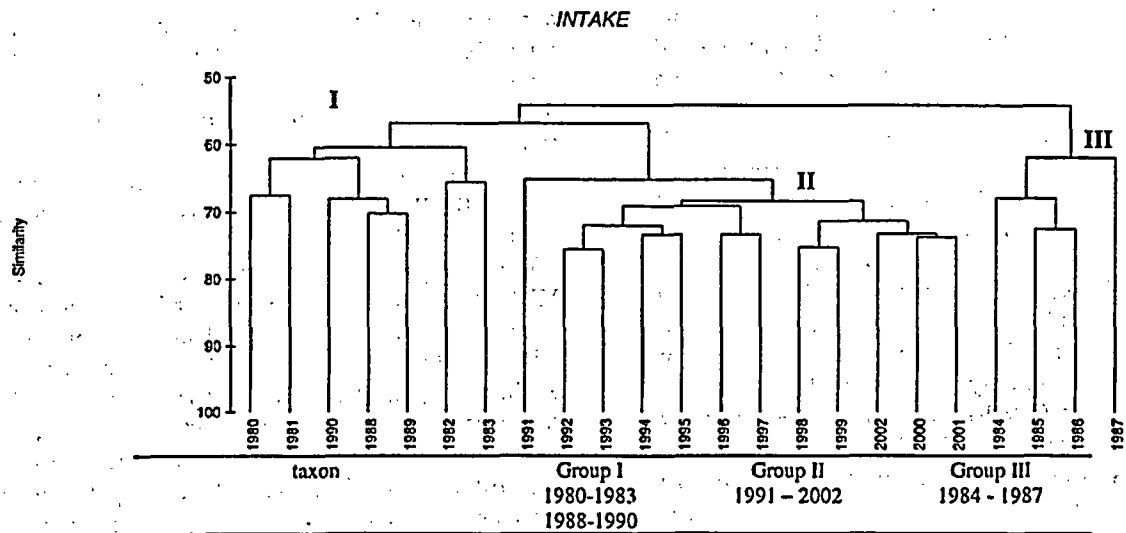
taxon	Group I 1980-1983 1988-1990	Group II 1991 – 2002	Group III 1984 - 1987
Leptochierus pinguis	13	62	894
Mediomastus ambiseta	14	326	509
Aricidea catherinae	110	689	30
Oligochaeta	87	585	72
Prionospio steenstrupi	15	340	42
Pygospio elegans	6	249	16
Ampelisca vadorum	6	99	145
Tharyx spp.	39	157	53
Ampelisca verrilli	28	18	170
Exogone hebes	17	162	19
Nucula annulata	44	27	126
Ampelisca abdita	3	0	185
Capitella spp.	70	41	61
Protodorvillea gaspeensis	12	117	6
Owenia fusiformis	4	18	112
Unciola irrorata	1	39	90
Polydora cornuta	3	14	112
Tellina agilest	13	62	48
Maldanidae	84	33	1
Pagurus longicarpus	1	0	97
Polydora quadrilobata	25	13	15
Microphthalmus aberrans	9	22	21
Clymenella torquata	16	23	8
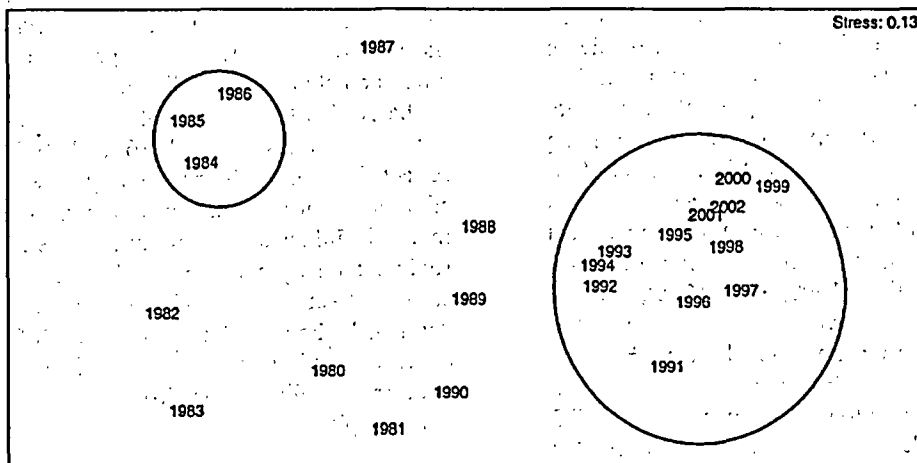
Fig. 9. (cont.)

Ampelisca verrilli, Ampelisca vadorum and *Ampelisca abdita*), the mollusc *Nucula annulata* and the polychaetes *Mediomastus ambiseta, Owenia fusiformis* and *Polydora cornuta,* and low abundance of *A. catherinae* and oligochaetes. Abundances of all amphipods were lower during Group II study years (1991-2002), as populations of *A. catherinae* and oligochaetes rebounded and highest abundances were observed for the polychaetes *Exogone hebes Tharyx* spp., *Prionospio steenstrupi, Protodorvillea gaspeensis* and the mollusc *Tellina agilis.* Collections from JC clustered into three groups at about a 63% similarity level (Fig. 9d). Group I collection years (1980-1986) were characterized by the highest density of oligochaetes and the polychaetes *Mediomastus ambiseta* and *Polydora caulleryi,* and low abundance of *Aricidea catherinae, Prionospio steenstrupi, Leptocheirus pinguis, Nucula annulata* and juveniles of the polychaete family Maldanidae. Group II (1987-1994) collections were distinguished by high densities of *P. steenstrupi, L. pinguis, Polycirrus eximius, Capitella* spp. and *N. annulata,* and by low oligochaete and *Mediomastus ambiseta* densities. Group III was made up of collections from 1995 through 2002, characterized by highest densities of the polychaetes *A. catherinae,* and *Tharyx* spp. Several other taxa (*e.g.,* Maldanidae, *Protodorvillea gaspeensis, Exogone hebes, Mediomastus ambiseta* and *Parapionsyllis longicirrata*) were also abundant during this period at JC.

Multi-Dimensional Scaling (MDS)

The MDS plots included in Figure 9 illustrate temporal community shifts and groupings of annual collections that generally support the associations in their respective clustering dendrograms. However, these two-dimensional plots are more amenable to multiple year comparisons, are more sensitive to less abundant taxa, and reveal the degree of community shifts based on distance between points representing annual collections. For example, at EF, there are four island groupings that correspond to pre-Unit 3 years (1980-1985), the initial three years following Unit 3 start-up (1986-1988), a loose grouping of years during Unit 3 operation (1990, 1992-1995), and the period of extended shutdown (1997-1999), with other collections representing transitional years between community states. Examination of spatial relationships among individual points in the MDS plot for EF reveals large year to year shifts that occurred between 1985 and 1986, when increased water flow from Unit 3 start-up scoured the discharge area and resulted in sudden sediment changes in the discharge area (see Fig. 2.) and infaunal shifts

described for Group II of the dendrogram. The remaining collections in the MDS plot (1989-2002) represents continued successional community development under 3-unit operation, an extended shutdown period (1997-1999), and 2-unit operation after the restart of Units 2 and 3, corresponding to Groups III, IV, and V, respectively, of the clustering dendrogram (Fig 9a).

The arrangement of annual collections in the GN MDS show little evidence of island groupings, but rather gradual community changes over time, primarily attributed to continued long-term increases of *Tharyx* spp., *Polydora caulleryi, Protodorvillea gaspeensis, Polycirrus eximius* and oligochaetes, and a concomitant decrease of *Aricidea catherinae* (Fig. 9b). Additionally, the MDS plots reflect pulses in the abundance of some taxa in recent years, such as that of *Polydora socialis* in 2001.

The MDS plot of annual IN collections (Fig 9c) illustrates high variability in community structure during the period from 1980-1990. The greatest community shift, based on the MDS plot, occurred between 1983 and 1984, when Unit 3 intake cofferdam removal and dredging activities were conducted, reflected in changes in sediment composition shown in Fig. 2. The years immediately following these activities (1984-1987) form an isolated grouping, consistent with Group III of the cluster dendrogram. Collections from 1988 to 1990 represent transition toward the community state reflected in the grouping composed of 1991-2002 collections, also identified as Group II in the clustering dendrogram. The distinct, relatively cohesive island grouping formed by more recent years demonstrates the successional development of the post-disturbance community at IN, with a trend toward more consistent community structure. During these recent years, taxa such as *Aricidea catherinae* have shown increases over previous years, while amphipod abundance declined.

The Jordan Cove MDS plot (Fig. 9d) reveals two island groupings (1980-1986 and 1987-2002) that reflect shifts in the JC community state caused by the siltation event immediately following Unit 3 start-up in 1986. Infaunal community changes resulting from sediment changes related to Unit 3 start-up had a strong effect on the JC infaunal community, as described above. Some of these shifts were first observed during the second half of 1986, which appears spatially distinct from previous years within that grouping.

Discussion

Monitoring studies of benthic infaunal communities inhabiting sediments in the MPS vicinity document

JORDAN COVE
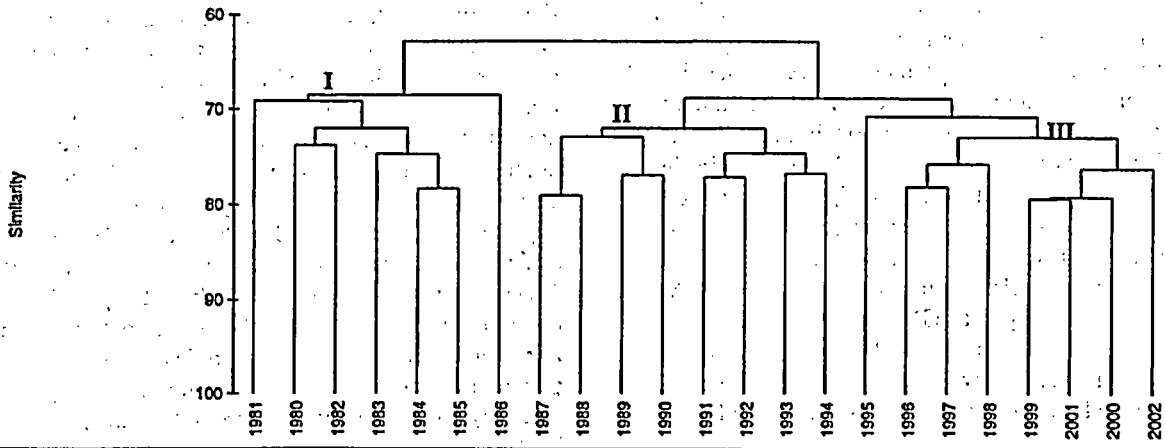
Taxon	Group I 1980 - 1986	Group II 1987 - 1994	Group III 1995 - 2002
Oligochaeta	2365	1361	1460
Aricidea catherinae	795	1427	2526
Mediomastus ambiseta	1707	739	1487
Prionospio steenstrupi	65	902	82
Tharyx spp.	282	238	523
Leptocheirus pinguis	53	616	361
Scoletoma tenuis	349	407	390
Polycirrus exinius	273	422	177
Nucula annulata	38	185	159
Tellina agilis	117	86	102
Capitella spp.	102	120	77
Exogone hebes	36	45	203
Parapionosyllis longicirrata	48	29	195
Maldanidae	2	23	189
Microphthalmus aberrans	44	84	84
Polydora caulleryi	140	12	13
Protodorvillea gaspeensis	13	31	111
Phyllodoce mucosa	33	32	56
Mitrella lunata	52	42	26
Rhynchocoela	46	49	24
Pygospio elegans	1	0	34
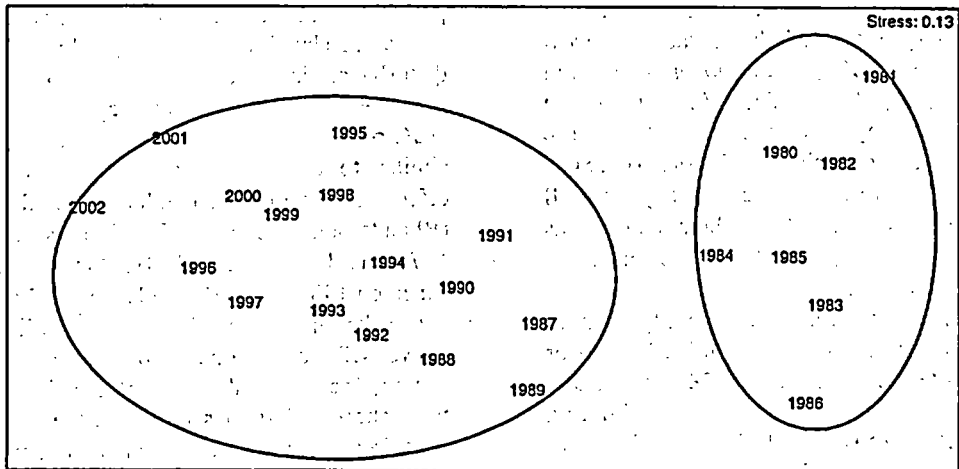
Fig. 9. (cont.)

both long-term trends and short-term changes in community and population-level parameters and sedimentary characteristics. This study has provided evidence of impacts from disturbance events related to MPS construction and operation, along with assessment of the degree to which factors unrelated to MPS operation contribute to observed community changes. Infaunal communities exhibiting evidence of MPS impacts were observed at the stations in the immediate vicinity of MPS (IN, EF and JC). Community changes at these sites were closely associated with changes in sediment composition related to power plant-induced disturbance. The infaunal community at GN reference site was unaffected by plant operation and provided a baseline from which variability associated with natural environmental cycles and disturbances could be assessed.

The GN station exhibited changes in sediments and community structure over the entire study period. Sediment mean grain size was most variable at this site, but silt/clay content was least variable and no long-term trends were apparent for either parameter. The GN infaunal community has generally been dominated by three taxa over the study period (*Tharyx* spp., oligochaetes and *Aricidea catherinae*), and some long-term trends in abundance of these taxa continued through the 2002. *Tharyx* spp. abundance continued to increase, while *A. catherinae*, abundant in early monitoring years, declined. Some species at GN exhibited relatively high year-to-year fluctuations. In many cases, these fluctuations proved useful in validating area-wide shifts in species abundance and community structure in response to natural causes and unrelated to MPS operation. For example, there was a pulse in abundance of the polychaete *Polydora socialis* at all stations in 2001, indicating a regional phenomenon. Other similar population shifts include area-wide increases of the opportunistic polychaete *Mediomastus ambiseta* (NUSCO 1989, 1996, 1999) and *Pygospio elegans* in 1998 (NUSCO 1993, 1999). These increases could not be explained by changes in site-specific sedimentary or regional abiotic factors (NUSCO 1989, 1993, 1999). However, because they were all observed at reference site GN, their cause was determined to be independent of power plant construction or operation.

Physical disturbances initiated community changes at IN and JC in the period between 1980-1987. Prior impacts were noted from 1983 to 1985 at IN, and resulted from dredging and cofferdam removal during Unit 3 construction (NUSCO 1987). Sedimentary characteristics at IN in 2002 (primarily silt/clay content levels) indicate further stabilization and more consistent habitat. Evidence of some infaunal

community stabilization and recovery was also apparent. In particular, numbers of individuals and species richness at IN have steadily increased over the study period, as have abundances of organisms typically more common in early study years or at other sites, such as *Aricidea catherinae*. These trends, along with concomitant decreases in abundance of *Nucula annulata*, which were not among the dominants in 2002, and of other opportunistic species (*e.g.*, the amphipod *Leptocheirus pinguis* and polychaete *Mediomastus ambiseta*), indicate some degree of recovery at IN. Other organisms that established post-impact community dominance, such as oligochaetes and *Protodorvillea gaspeensis*, indicate a changed community may persist indefinitely. Long-term recovery following disturbance is typical of marine benthic communities (Kaplan *et al.* 1974; Swartz *et al.* 1980; Nichols 1985; Berge 1990; Zajac *et al.* 1998).

Changes in community composition that result from increased silt deposition at JC following Unit 3 start-up in 1986 were continued through the 2002 sampling year. Silt deposition and concurrent large infaunal community changes occurred over a short period at the beginning of the 3-unit operational period. The infaunal community exhibited changes in recent years moving toward pre-impact community structure. For example, *A. catherinae* and *Tharyx* spp. abundances both advanced to levels observed prior to 1986 within a few years of Unit 3 start-up. However, continued trends toward recovery noted in previous years (e.g., reduced silt/clay content during 1994 and 1995, and rebounding abundances of oligochaetes and *P. eximius* through 1993; NUSCO 1994, 1995) were short-term habitat or community changes not reflected in more recent data. Additionally, the opportunistic mollusc *Nucula annulata* maintained a population at JC through 2002 that remains well above pre-Unit 3 levels. This biological evidence of disturbance is consistent with trends in sediment characteristics; elevated silt/clay levels in sediments at JC persisted through 2002. These observations are similar to 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; Currie and Parry 1998).

The benthic infaunal site that still experiences active MPS operational impact processes is the EF station, located in the path of the discharge 100 m from shore. Current scour from the MPS discharge modified sediments and the infaunal community at EF immediately after Unit 3 startup in 1986. Relatively coarse sediment with low silt/clay levels was characteristic of the 3-unit benthic habitat at EF.

While sediment characteristics remain different from those observed prior to 1986, the altered sedimentary environment at EF, and the infaunal community it supports, stabilized under the new environmental conditions created by the 3-unit discharge. The relative stabilization of the sediments at EF allowed for rebounds of *Tharyx* spp. and *Aricidea catherinae*, taxa common during 2-unit operation. In addition, oligochaete and *Protodorvillea gaspeensis* abundances have generally decreased from high abundances during the early 3-unit period. Other indications of sediment stabilization at EF are population increases consistent with regional pulses of populations discussed above (*e.g., Mediomastus ambiseta, Polycirrus eximius,* and *Polydora socialis*). Plant discharge effects overrode natural regional factors during earlier years of 3-unit operation, and region-wide pulses in the above taxa had not been observed at EF.

Changes observed in infaunal community structure at impacted stations indicate permanent shifts in the dynamics of benthic communities in the vicinity of MPS. Shifts in benthic community structure at EF were similar to those caused by commerical scallop dredging in Port Phillip, Australia, noted over a 20-year period (Currie and Parry 1998). However, establishing which components of benthic community structure are due to natural cycles or man-induced disturbance is difficult because of the unpredictable length of natural cycles (Gray and Christie 1983). Additional monitoring will determine if these trends are indicative of current operating conditions (Units 2 and 3 only), or are simply short-term fluctuations related to other factors.

Conclusions

Benthic habitats and their associated infaunal communities at IN, JC and EF during 2002 continue to exhibit patterns in community structure typical of disturbance related to construction and operation of Unit 3. While sediment grain size exhibited the highest variability at the unimpacted site GN, silt/clay content and infaunal community structure at this site exhibited highest stability over the study period relative to the other monitoring sites. Some indications of increased stability and limited 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 but progression is slow. Community parameters of species composition and population abundance show similarities to the pre-disturbance period, but also clearly show differences related to past disturbance at both stations. Data from the

recent two-year shutdown period (1996-1998) and from the years following restart provided additional evidence that the MPS discharge has been a dominant factor in structuring both the sedimentary environment and infaunal community in the immediate vicinity of the discharge. In the absence of discharge effects, the EF community exhibited detectable shifts in some populations that had not been seen during 3-unit operation. Aside from these population changes, overall recovery of the community and sedimentary environment during this period was limited, indicating that post-operational recovery will be slow. Reversal of most trends observed during the shutdown period were noted soon after MPS restart, owing to full operation of Units 2 and 3 for much of the last four years.

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Eelgrass

Eelgrass

Introduction

Eelgrass (*Zostera marina* L.) is the dominant marine angiosperm in temperate coastal regions of the Northern Hemisphere (Setchell 1935; Thayer et al. 1984). Extensive eelgrass meadows found in shallow estuaries and lagoons provide multi-level ecological benefits that make them key components to many coastal marine systems. High eelgrass primary productivity is utilized directly by many consumers including numerous waterfowl, fish and invertebrate grazers (see review by Valentine and Heck 1999), but much of the production enters the foodweb through the detrital pathway (Thayer et al. 1984). Increased habitat complexity within eelgrass meadows enhances primary and secondary production and species diversity in coastal ecosystems. Eelgrass meadows support productive epiphytic communities (Thayer et al. 1984; Nelson and Waaland 1997), and provide feeding and nursery grounds for many species, including commercially important finfish and invertebrates (Orth 1973; Heck et al. 1989, 1995; Gotceitas et al. 1997; Mattila et al. 1999; Short et al. 2001).

Historical reports from eastern Long Island Sound (LIS) of eelgrass populations (e.g., Marshall 1947, 1960, 1994; Koch and Beer 1996) describe fluctuations in abundance and distribution over the last century similar to those observed elsewhere in the western North Atlantic Ocean (Tutin 1942; Rasmussen 1977; Orth and Moore 1983; Short et al. 1996). Populations along the coastline of Connecticut were almost completely eliminated by 1933. Some recovery occurred by the late 1950s, and by the early 1970s eelgrass beds in southeastern Connecticut were again extensive (Knight and Lawton 1974; McGill 1974; Marshall 1994). Since the early 1980s, however, eelgrass populations in Connecticut and elsewhere have declined (e.g., Orth and Moore 1983; Short 1988; Koch and Beer 1996; Short et al. 1996). Because of the ecological importance and high historical variability of eelgrass, and the prediction that the 3-unit thermal plume from Millstone Power Station (MPS) could reach to the nearby populations in Jordan Cove (Knight and Lawton 1974; NUSCO 1988), the present study to monitor local eelgrass populations was initiated in 1985. Objectives of the present study are to identify temporal patterns of eelgrass population characteristics in the vicinity of MPS and to determine the extent to which changes in these patterns are the result of natural variability or MPS operation.

Materials and Methods

Three eelgrass study sites in the vicinity of MPS were sampled during 2002 (White Point-WP, Jordan Cove-JC, and Niantic River-NR; Fig. 1). The WP and JC stations, are located 1.6 km and 0.5 km east of the power plant discharge, respectively, were within the area potentially influenced by the 3-unit thermal plume (NUSCO 1988). The NR sites, located about 3 km from Millstone Point, are 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. A new sampling site was established in the Niantic River in 2000 (NR5) due to bed die-off at the 1999 site (NR4). Location changes and periods of sampling for all Niantic River sites are described in Figure 1.

Samples were collected monthly at each site from June through September, the period of maximum standing stock and plant density. At each station, SCUBA divers collected 16 samples 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, returned to the laboratory and analyzed. A 3.5 cm diameter x 5 cm deep core was taken concurrently with eelgrass samples for analysis of sedimentary characteristics at each station. Water temperature at the JC site was measured by an encased thermistor-recorder, suspended just above the sediment surface. Temperature measurements have been recorded in Jordan Cove since 1991.

All shoots collected were counted in the laboratory and the longest blade of each shoot (up to 20 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. 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 2002 were estimated from the wet-weight/dry-weight relationship obtained above.
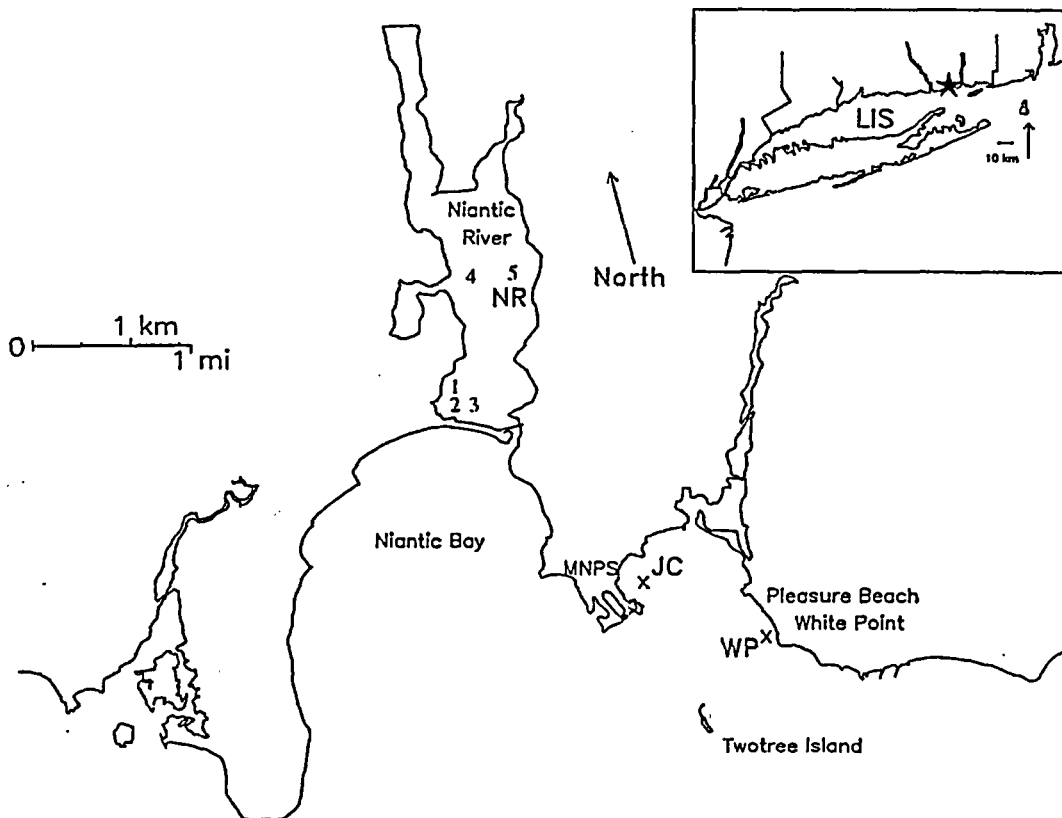
Fig. 1. Map of the Millstone Point area, showing the location of eelgrass sampling stations: JC=Jordan Cove, NR= Niantic River (1=sampled 1985-June 1986 and 1993-1994, 2=sampled July-September 1986, 3=sampled 1987-1992, 4=sampled 1995-1999, 5=sampled 2000-2002), WP=White Point.

Linear regression methods were used to examine long-term trends in eelgrass shoot density, shoot length and standing stock. These methods were applied to all the data, except for outliers identified by the SAS (1999) univariate procedure, and removed prior to analysis. This analysis was not conducted on Niantic River data due to the lack of a continuous time-series at any given sampling location resulting from periodic sampling bed die-off.

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 seawater temperatures at the MPS intakes and discharge, and at the JC sampling station, during June through September 2002 are presented in Figure 2. Temperatures at the power plant intakes and at the eelgrass station showed typical seasonal patterns, with peak daily temperature occurring on August 2 at JC (24.1°C) and on August 5 at the intakes (22.9°C). Average seawater temperatures at the JC eelgrass station were generally warmer than ambient temperatures by 1-2°C, especially in July and August. Effluent temperatures averaged 10-11°C above intake and JC temperatures, but reflected a short shutdown of Unit 2 in early August, and Unit 3 refueling in September; peak daily average temperature at the discharge occurred on September 23 (34.8°C).

Sediments

Monthly measures (June-September) of mean grain size, silt/clay and organic content have been used to describe eelgrass sites monitored since 1985 (Fig. 3). Sediments at stations nearest MPS (JC and WP) have shown a high degree of stability relative to the variable nature of those at NR. Much of the variability at NR has been attributed to repeated relocation of the sampling site, the patchy distribution of eelgrass within the Niantic River, and subsequent exposure to water currents and settlement of sediment fine fractions.
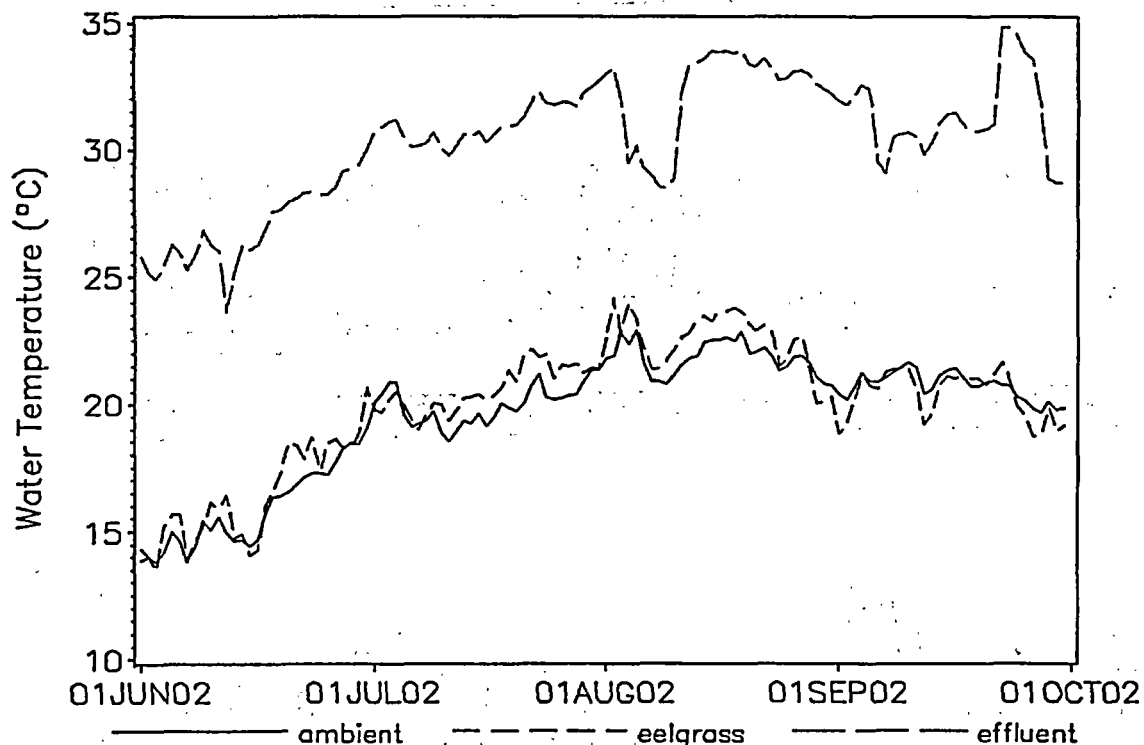
Fig. 2. Mean daily water temperature at the JC eelgrass station (eelgrass), at MPS intakes (ambient), and at MPS quarry cuts (effluent), from June through September 2002.

Sediments collected during 2002 were coarser at JC (mean grain size 0.24-0.30 mm) than those at WP (0.12-0.16 mm) or NR5 (0.08-0.09 mm). Sediment organic content in 2002 was higher at NR5 (7.2-8.5%) than at WP (2.1-3.8%) or JC (1.3-1.9%). Silt/clay content in 2002 was highest at NR5 (39.2-42.8%), intermediate at WP (6.2-11.8%), and lowest at JC (2.2-4.7%). Sediment parameter estimates relationships among stations in 2002 were within historical ranges and similar to those established since 1985.

Shoot Density

Annual mean shoot density (shoots m^{-2}) among eelgrass stations in 2002 was similar to previous years: highest at JC (506), and lower at WP (289) and at NR5 (268; Fig. 4). Shoot densities at all stations in 2002 were within historical ranges, although mean annual density at NR5 has exhibited an increase over the last three years. Monthly shoot densities in 2002 ranged from 348 (June) to 616 (August) at JC, 203 (June) to 416 (July) at WP, and 189 (September) to 380 (July) at NR5. Monthly mean densities at all stations were within the ranges of previous years. Linear regression analysis of shoot density over the entire time-series indicated no significant trend at JC or WP.

Shoot Length

Annual mean shoot lengths for 2002 were longest at WP (96 cm), and shorter at NR5 (47 cm), and at JC (37 cm; Fig. 5). Annual means at all stations were within historical ranges observed since 1985. Maximum monthly shoot lengths in 2002 were highest in September at JC (46 cm), in June at NR5 (52 cm), and in July at WP (106 cm). Linear regression of shoot length data over the entire time-series indicated significantly decreasing slopes at JC (n=1446; slope= -0.09 cm month^{-1}; p<0.0001) and WP (n=1127; slope= -0.07 cm month^{-1}; p<0.0001).

Standing Stock

Eelgrass standing stock (g dry wt m^{-2}) annual mean for 2002 was higher at WP (206) than at JC (106) or NR5 (70; Fig. 6). These annual standing stock estimates were within the historic ranges. Monthly standing stock estimates in 2002 ranged from 73 (June) to 148 (August) at JC, from 123 (September) to 283 (July) at WP, and from 31 (September) to 109 (July) at NR5. Linear regression analysis of standing stock data over the entire time-series indicated a significantly decreasing trend at JC (n =1142; slope= -0.46 g m^{-2} month^{-1}; p<0.0001) and WP (n=1125; slope= -0.11 g m^{-2} month^{-1}; p=0.041).
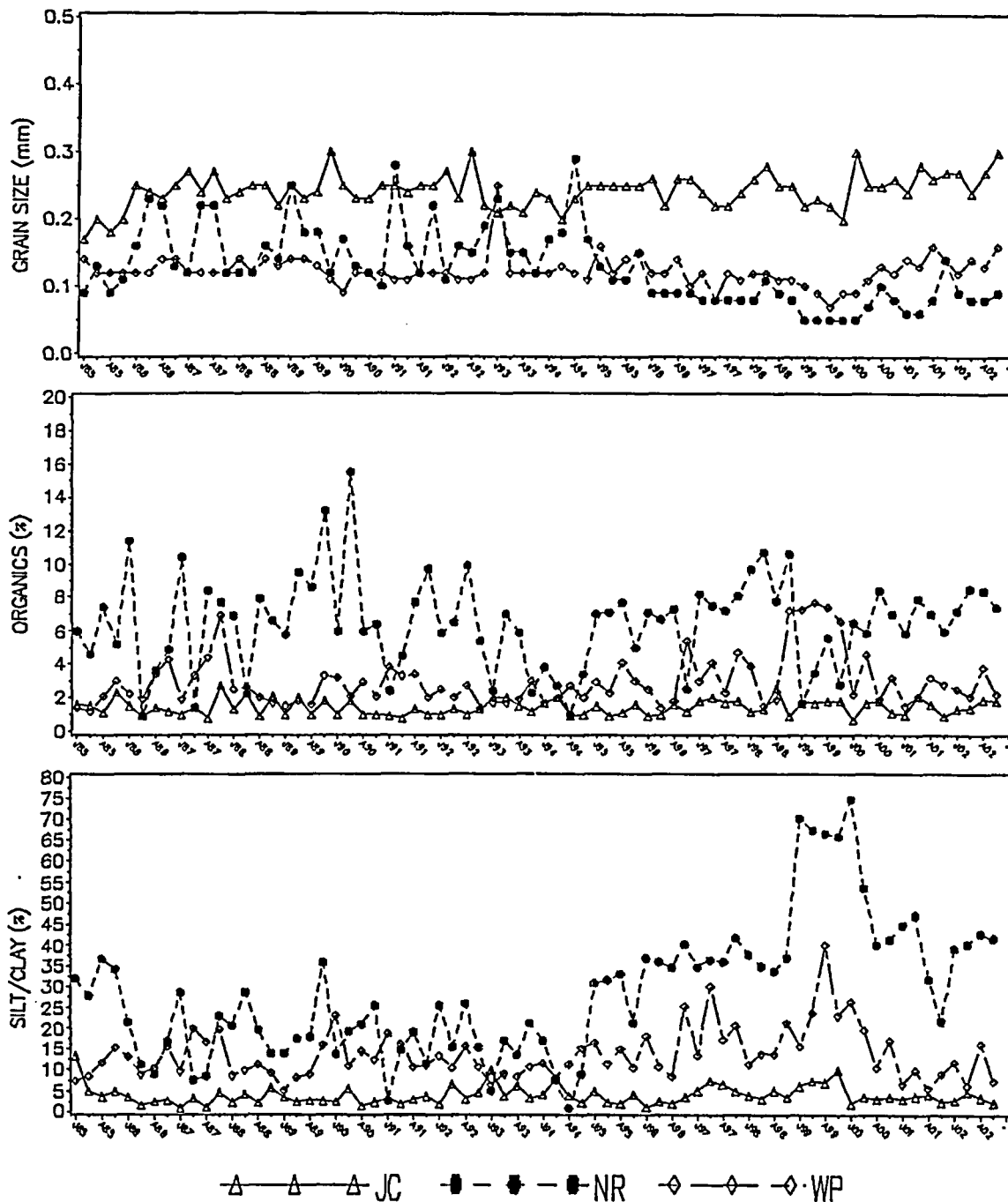
Eelgrass 273

Fig. 3. Mean grain size, organic content, and silt/clay content of sediments at MPS eelgrass stations Jordan Cove (JC), Niantic River (NR), and White Point (WP), sampled June-September from 1985 through 2002.

Seed-Bearing Shoots

Annual and monthly numbers of seed-bearing shoots collected at each station are presented in Figure 7. Total seed-bearing shoot abundance in 2002 was highest at NR5 (103) followed by WP (32) and JC (25). Seed bearing shoots in 2002 were found in June and July at all stations, with the highest number of shoots occurring in June (55 at NR5, 16 at

WP, and 13 at JC). Only one seed-bearing shoot was collected in August 2002 (JC); there has been a general regional trend over time that seed-bearing shoots have occurred less frequently during August and September. Seed-bearing shoots were consistently observed in samples collected from all sites in August and September prior to 1991, but were less common in those months in subsequent years.
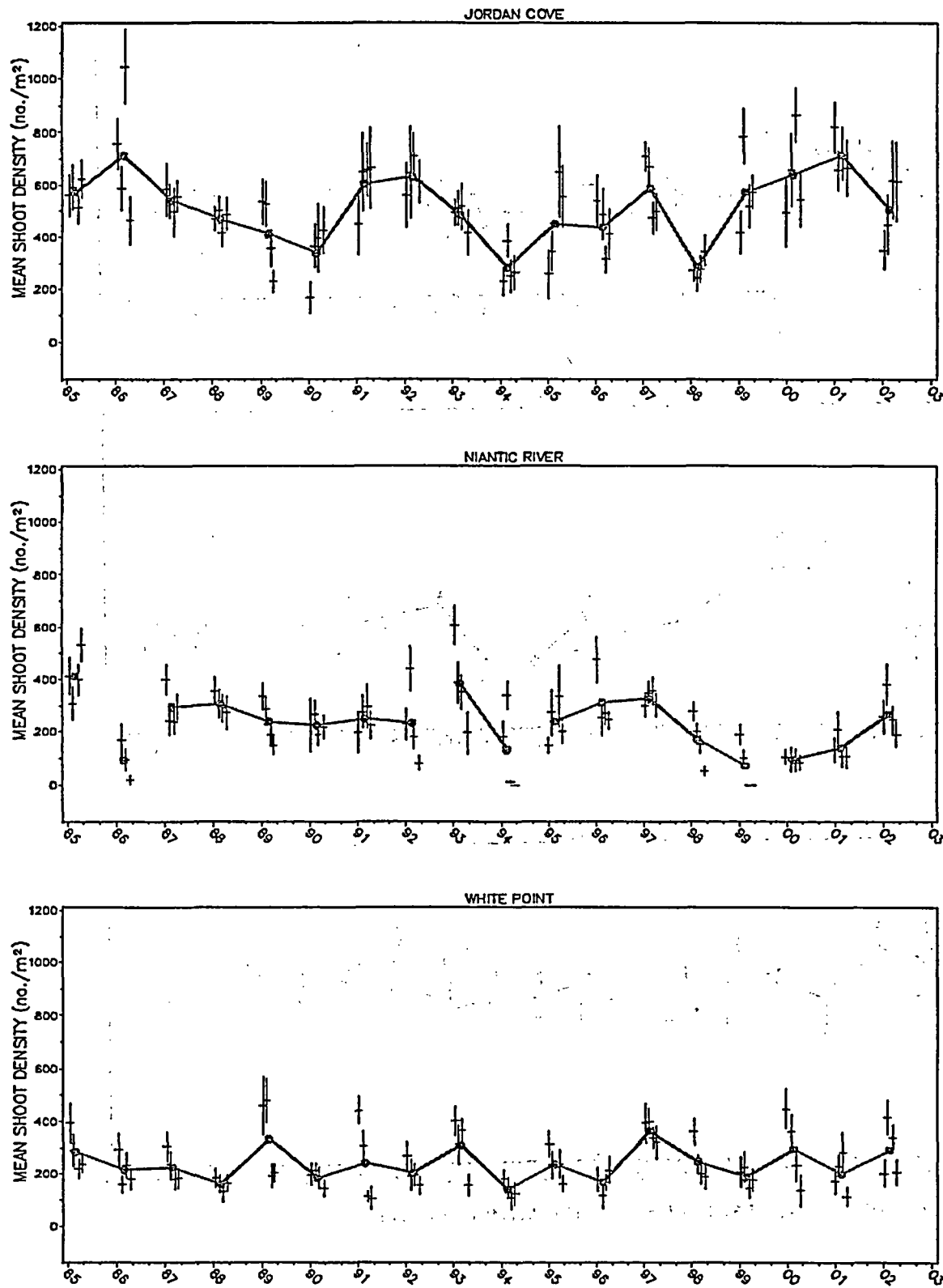
Fig. 4. Monthly mean shoot density (± 95% C.I.) and annual mean density at MPS eelgrass sites.
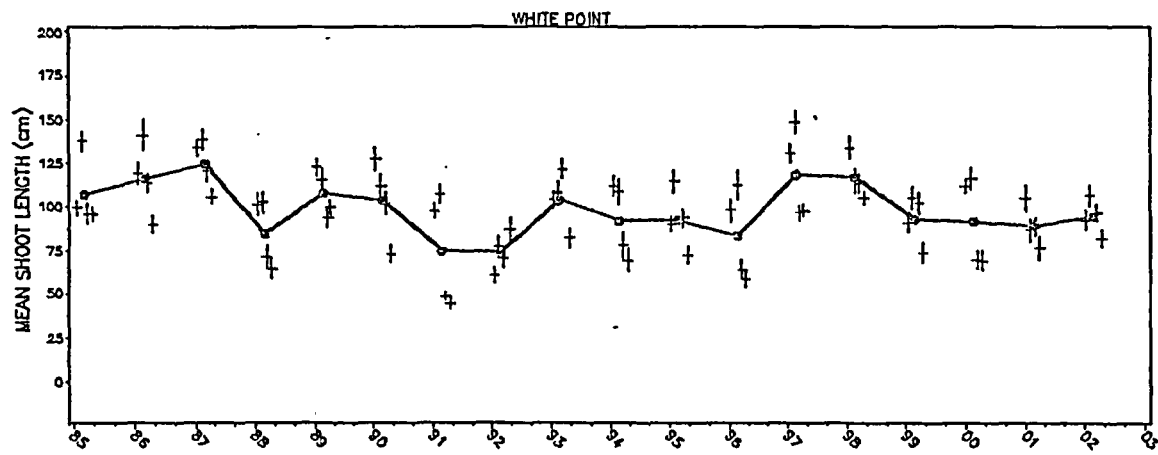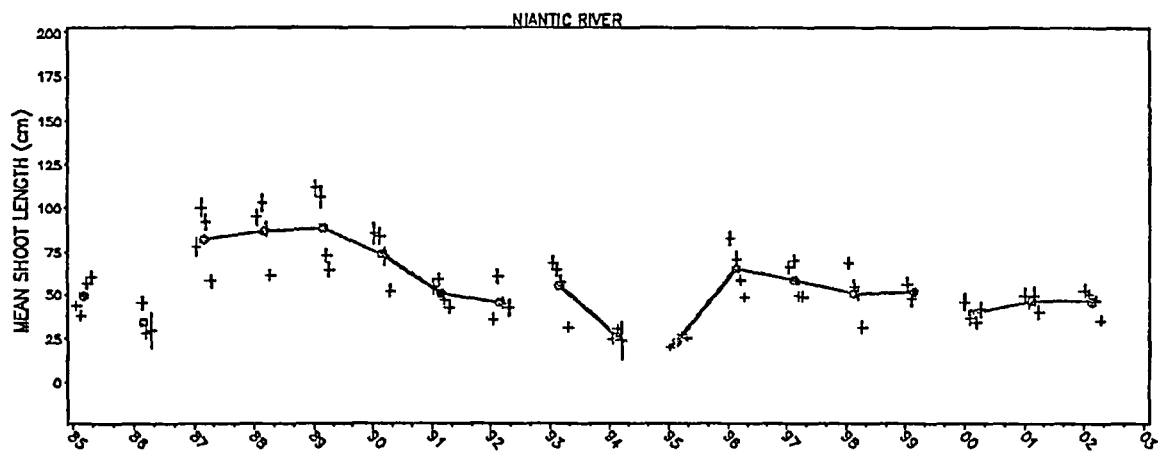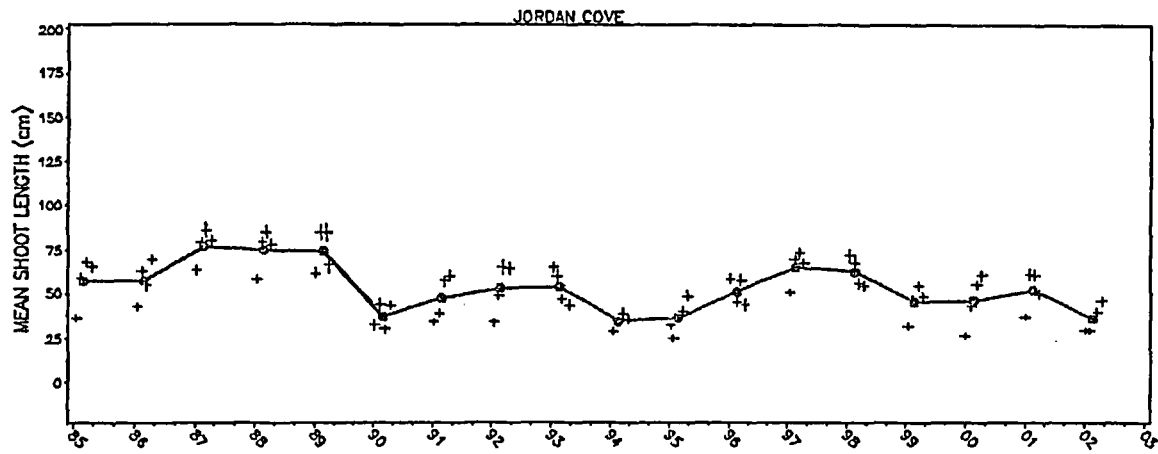
Fig. 5. Monthly mean shoot length (± 95% C.I.) and annual mean length at MPS eelgrass sites.
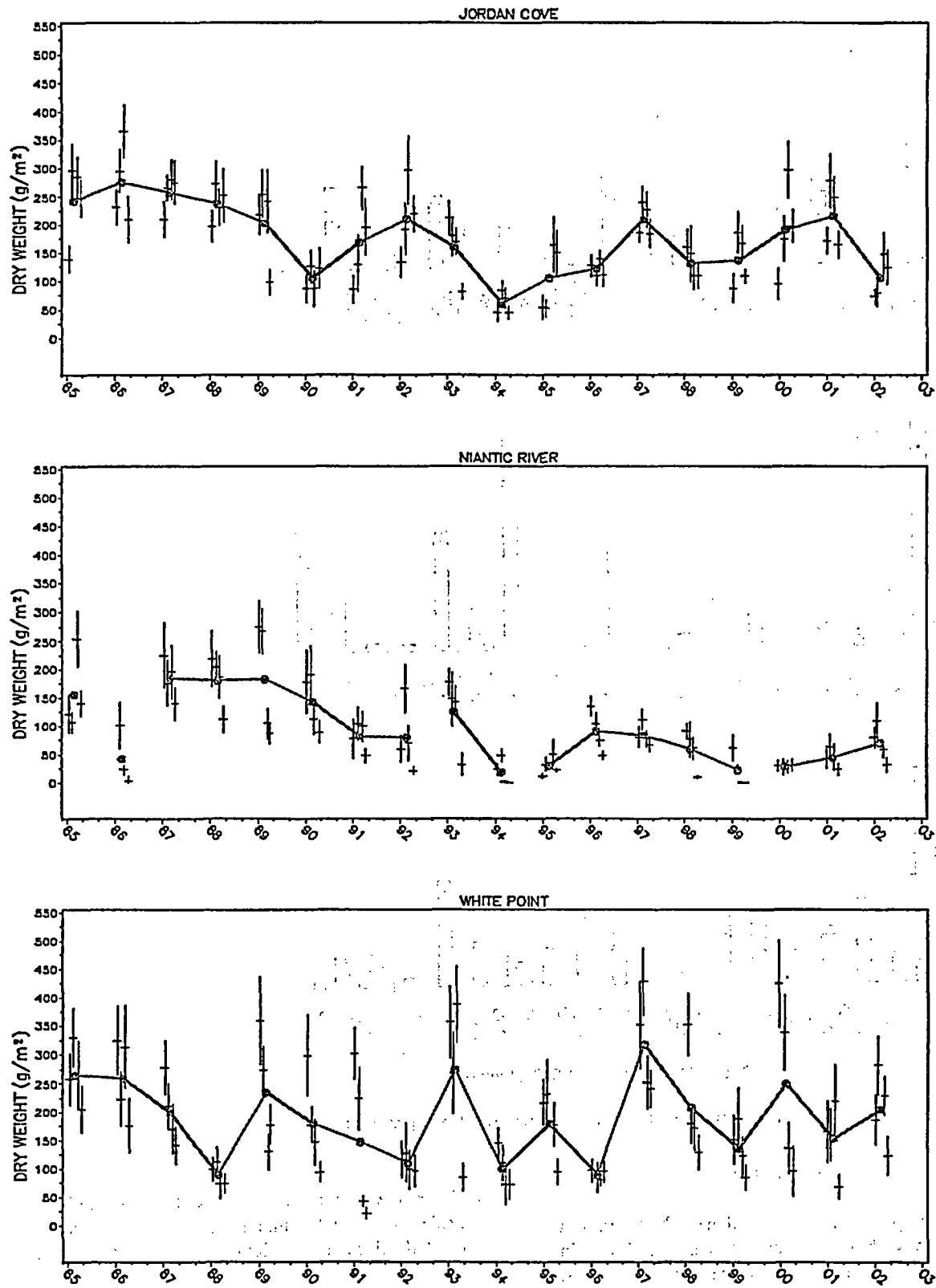
Fig. 6. Monthly mean standing stock biomass (± 95% C.I.) and annual mean biomass at MPS eelgrass sites.
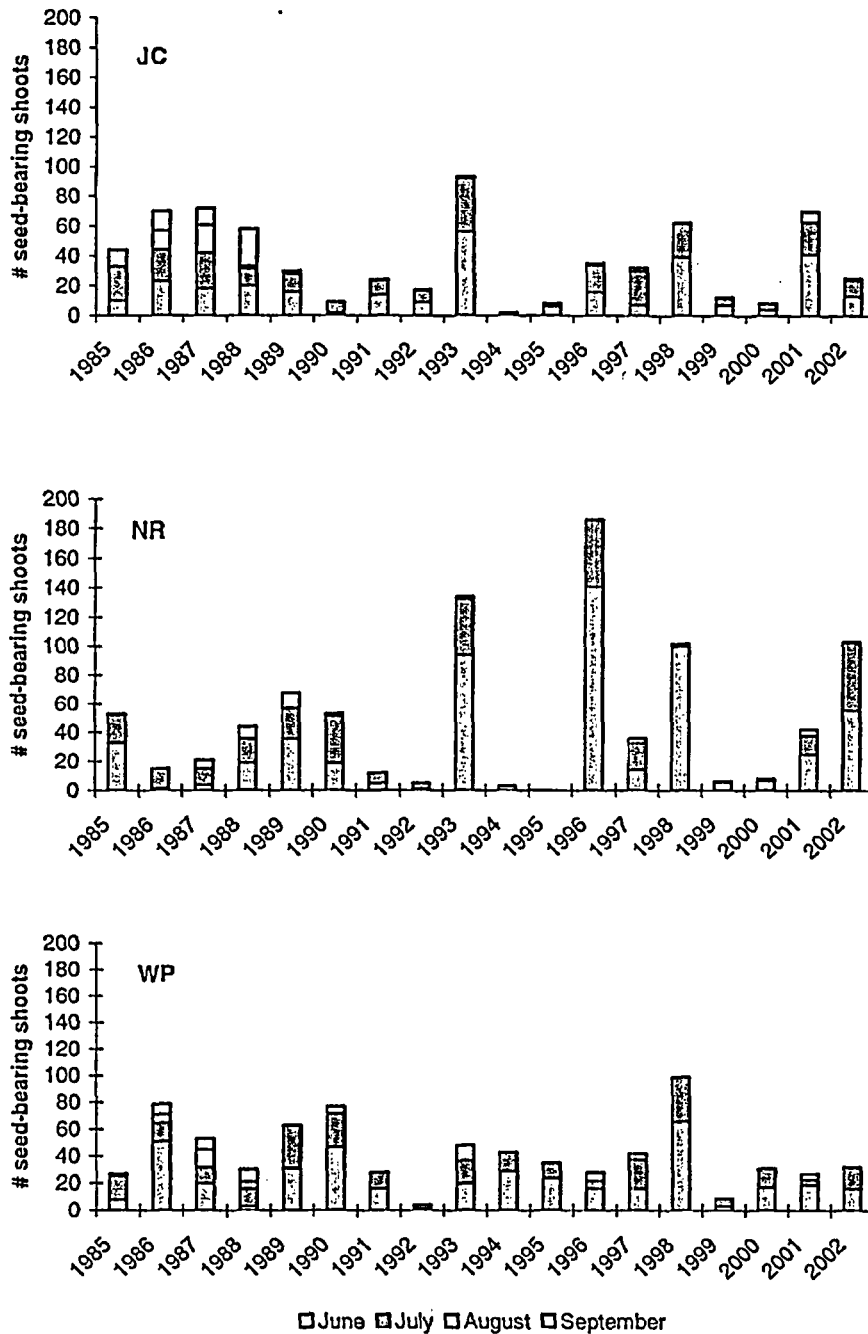
Fig. 7. Total number of seed-bearing shoots collected monthly and annually at MPS eelgrass sites.

Discussion

Eelgrass populations monitored near MPS exhibit considerable within- and among-year variability in population parameters at all three study locations. Statistically significant declines in eelgrass shoot length and biomass were observed at sites nearest the MPS thermal plume (JC and WP), although these eelgrass populations have persisted as healthy beds over the course of this study. In contrast, eelgrass

beds in the Niantic River (NR 1-5), located well outside the thermal plume area, have experienced the largest declines in population characteristics of any site.

Others have shown that increased water temperatures (e.g., from heated power plant effluents) eliminated eelgrass from nearby areas (Phillips 1974; Thayer et al. 1984). Similarly, decline of the seagrasses *Thalassia testudinum* and *Halodule*

wrightii in Florida (Roessler and Zieman 1969; Wood et al. 1969; Zieman 1970; Roessler 1971; Blake et al. 1976; Thorhaug et al. 1979) and the marsh grass *Spartina alterniflora* in Maine (Keser et al. 1978) were directly attributed to temperature increases from power plant effluents. However, effluent-related water temperature increases experienced by populations in the above-cited studies were considerably higher than those observed in this study.

Daily temperature fluctuations of up to 5°C were measured at our JC study site during mid-day in summer, and were most pronounced on sunny days at low tide. This diurnal pattern during summer suggests that increases in seawater temperature at this site are primarily attributed to solar warming. Any thermal input to JC from the MPS cooling water discharge (<1°C; NUSCO 1988) is difficult to distinguish from natural fluctuations, suggesting very limited thermal effects (Keser et al. 2003).

This study provides more evidence of long-term trends in eelgrass population characteristics from a region that has experienced widespread declines over the last two decades. Similar trends of eelgrass populations have been documented elsewhere in eastern North America, including the Chesapeake Bay (e.g., Orth 1976; Orth and Moore 1983, 1986; Moore et al. 1996, 1997; Moore and Wetzel 2000) and coastal embayments of Rhode Island (Harlin and Thorne-Miller 1981; Thorne-Miller and Harlin 1984; Short et al. 1996), Massachusetts (Dexter 1947, 1985; Roman and Able 1988; Short and Burdick 1996) and New Hampshire (Riggs and Fralick 1975; Short et al. 1986). This study complements the few reports available on temporal trends of eelgrass in LIS (Dexter 1946; Marshall 1947, 1960, 1994; Knight and Lawton 1974; Koch and Beer 1996). These reports indicate that eelgrass populations in LIS have fluctuated since the widespread die-off in the 1930s, similar to long-term trends reported elsewhere in eastern North Atlantic. Several reports note that eelgrass populations in the vicinity of MPS had experienced considerable recovery by the 1970s (Knight and Lawton 1974; Marshall 1994), with a gradual decline beginning in the 1980s (Vozarik et al. 2000; DNC 2001). Eelgrass distribution along the north shore of LIS once reached as far west as Westchester County, NY, but now is limited to the easternmost third of the Connecticut coastline (Koch and Beer 1996; Randall et al. 1999). Our results show that this west-to-east trend in eelgrass decline may be progressing. Given that much of the shoreline and watershed areas of LIS also experience the type of expanding industrial and residential development thought to be detrimental to eelgrass elsewhere (e.g., Orth and Moore 1983; Thayer et al. 1984; Short et al. 1996), there is need to better document changes in eelgrass populations, to more definitively identify and possibly reduce sources of suggested impact responsible for observed declines.

It is often not clear how trends and fluctuations in eelgrass population characteristics observed in this study are related to physical factors such as temperature or sedimentary characteristics. We observed abbreviated or shifted periods of occurrence of seed-bearing plants, which were more commonly found at all stations throughout the June-September during the period 1985-89 than during later years, when they occurred primarily in June and July. Analysis of long-term temperature data from the MPS intakes revealed a significant warming trend in seawater temperatures over the period 1976-2000, and this trend was particularly pronounced during the winter/spring (January-June; Foertch 2000). The winter/spring temperature rise may have resulted in earlier onset of both the optimum temperature range for eelgrass reproduction (9-15°C; Setchell 1929; Thayer et al. 1984) and of more stressful temperatures above 15-20°C when eelgrass ceases to produce seeds (Burkholder and Doheny 1968; Orth and Moore 1983). Seawater temperature rise may have contributed to declines observed in other *Zostera* population parameters as well.

Two short-term population declines in shoot density and standing stock biomass were directly associated with localized population overgrowth. In July 1991, a bloom of the filamentous green alga *Cladophora* spp. covered the entire WP eelgrass bed. As a result of this bloom, shoot density declined and the sparse remaining plants appeared pale yellow in August and September. Studies in Oregon, USA (Kentula and McIntire 1986) and Hampshire, UK (den Hartog 1994) found similar demise of eelgrass following green algal blooms of *Enteromorpha prolifera* and *E. radiata*, respectively. Additionally, loss of plants from NR3 in 1992 was attributed to fouling by blue mussels (*Mytilus edulis*), which covered blades so heavily that they sank to the bottom where they were completely overgrown. Reusch et al. (1994) suggested that blue mussels can be beneficial to eelgrass through biodeposition, water nutrient enhancement, and turbidity reduction. However, mussels in their study were rarely observed attached to blades or rhizomes, but rather to each other forming an understory mat below the eelgrass canopy. The NR3 bed had not recovered from this event through 2002.

Extensive die-off of most of the remaining eelgrass in the Niantic River, including sampling bed for site NR4, occurred between July and August 1999. This die-off coincided with a large increase in sediment silt/clay content at NR4. The cause of this sedimentary change was not determined. Turbidity and high siltation rates over short periods have been shown to be detrimental to seagrasses (Moore et al.

1996, 1997; Vermaat et al. 1996). In addition, we observed thick mats (up to 25 cm) of the red macroalga *Agardhiella subulata* covering the sediment surface and lower portions of eelgrass blades at NR4 preceding the die-off. This habitat change in the Niantic River was also reported by Goldberg et al. (2000), which noted evidence of hypoxia and migration of scallops out of the NR4 eelgrass bed. Hauxwell *et al.* (2001, 2003) and others (see review of McGlatherty 2001) demonstrated that this condition was detrimental to eelgrass by limiting light and creating anoxia and high ammonia levels within the mat through algal decomposition and remineralization. At the most recently established Niantic River sampling station (NR5), there have been lower levels of silt/clay, and increases in shoot density and biomass over the past three years.

The spatial pattern of decline among study populations in the vicinity of MPS suggests a relationship to nutrient enrichment. The location exhibiting highest variability in eelgrass population parameters monitored in this study is the Niantic River, where a patchy population of transient beds has been observed since 1985. This population was extensive during the 1970s (McGill 1974; Marshall 1994), but nearly nonexistent by 1999 (Vozarik et al. 2000; Keser et al. 2003). Marshall (1994) and Short (1988) suggested that the decline of eelgrass in the Niantic River was due to a combination of poor water quality and the presence of *Labyrinthula*, a marine slime mold implicated as the causative factor in eelgrass wasting disease. The Niantic River study populations experience the least tidal flushing and are nearest to freshwater input sources and dense year-round housing, compared to the other two study populations near MPS. Thus, Niantic River eelgrass beds may be more susceptible to nutrient loading from fertilizers and domestic septic systems than beds in Jordan Cove. As with many areas in the northeast U.S., considerable housing development has occurred along the Niantic River shoreline and within its watershed over the past 20 years. The JC site receives more tidal flushing and is farther removed from coastal nutrient input sources, and eelgrass population declines there were quite minor by comparison. The study population most removed from these sources and located along an open coastline with unrestricted tidal flow, WP, showed the least change of the three study areas over the 18-yr study period.

Studies conducted in Ninigret Pond, RI and Waquoit Bay, MA linked both long- and short-term declines in eelgrass populations to increased housing development in the watershed (Short and Burdick 1996; Short et al. 1996; Bowen and Valiela 2001). Specifically, increased surface run-off and high groundwater discharge of nutrients from on-site septic

systems were implicated as causal factors in declines of eelgrass population parameters in these studies. In another study of *Zostera* in Ninigret Pond, Thorne-Miller and Harlin (1984) found a negative correlation between eelgrass production-to-biomass ratio (P/B) and distance from the breachway to the Atlantic Ocean, suggesting a positive effect of ocean water on eelgrass production. These coastal systems are similar to the Niantic River in that they are enclosed embayments where environmental conditions are strongly influenced by local landuse practices. While municipal sewerage has been installed in some parts of the Niantic River watershed (e.g., the eastern shore of the River), most of the developed portions still rely on on-site septic systems. In addition, old septic systems no longer in use may still be sources of nutrient input to the River, especially after periods of heavy precipitation. Moore et al. (1997) attributed losses and lack of recovery of eelgrass in an upriver section of the York River in Chesapeake Bay to seasonal pulses of higher turbidity relative to downriver sites where turbidity was generally lower and eelgrass persisted.

Biological disturbances may also have contributed to eelgrass losses in the Niantic River. Increased numbers of waterfowl known to reduce eelgrass coverage (Brent geese *Branta bernicla*, and swans *Cygnus* spp.; Valentine and Heck 1999) have been noted grazing on eelgrass in the Niantic River (pers. obs.). In addition, large increases in green crabs (*Carcinus maenas*) occurred in the River over the last 20 yrs (DNC 2001); their burrowing activities have been shown to damage eelgrass (Davis et al. 1998).

Conclusions

Eelgrass beds at the two monitoring sites nearest MPS (JC and WP) exhibited periodic fluctuations in shoot density and standing stock biomass, but generally support healthy populations, based on parameters monitored, throughout the 18-year study period. These two populations are considered potentially impacted by the MPS thermal plume, but temperature monitoring has not provided evidence of a power plant influence. Rather, observed variability in water temperatures is indicative of natural solar warming and hydrodynamic conditions in Jordan Cove. These natural factors are particularly influential at JC, which is the most shallow of the study sites, and is immediately adjacent to extensive shallow sand flats vulnerable to solar warming. With Unit 1 permanently retired, the likelihood of thermal plume incursion at JC and WP in the future is further reduced.

Population variability has been highest during the study period in the Niantic River; since 1985, complete die-off of five separate eelgrass beds within

the river has been observed. During 2002, the population at NR5 showed some signs of improvement. The exact cause of the long-term decline of eelgrass in the Niantic River is unknown. Because the Niantic River is located well away from any influence of the MPS thermal plume, declines there have been, and continue to be, related to other environmental factors such as nutrient input from domestic septic systems, disease, or increased turbidity.

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