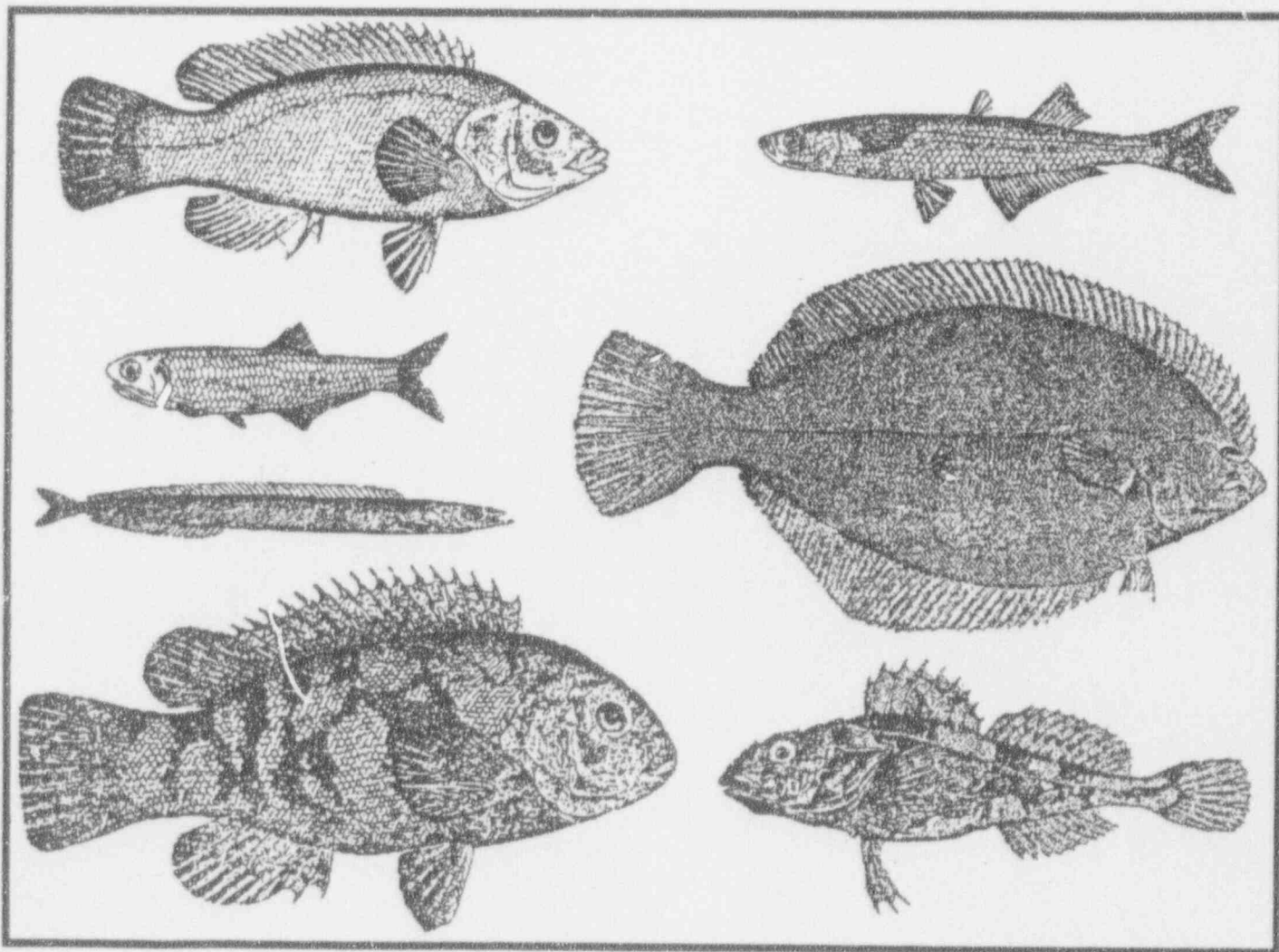


# Monitoring the Marine Environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut

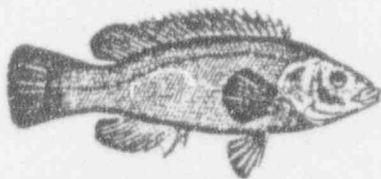
## Annual Report 1993



Northeast  
Utilities System

Northeast Utilities Service Company  
Environmental Services Division  
Aquatic Services Branch  
N U Environmental Laboratory  
April 1994

9405030186 931231  
PDR ADDCK 05000423  
R PDR



Cunner (*Tautoglabrus adspersus*)



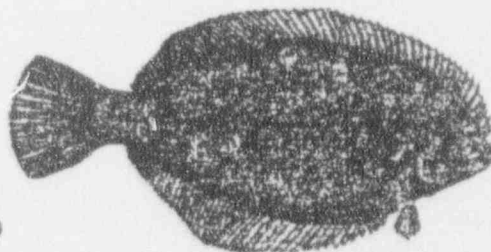
Silverside (*Menidia* spp.)



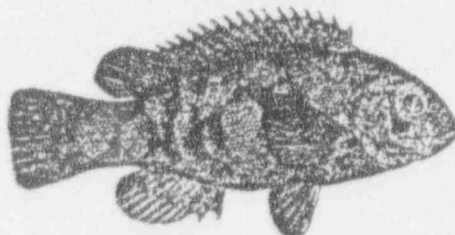
Anchovy (*Anchoa* spp.)



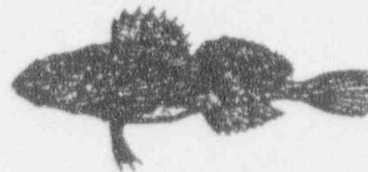
American sand lance (*Ammodytes americanus*)



Winter Flounder (*Pleuronectes americanus*)



Tautog (*Tautoga onitis*)



Grubby (*Myoxocephalus aeneus*)

The picture on the cover depicts six fish genera common in Long Island Sound and in samples collected as part of the Millstone Nuclear Power Station fish ecology monitoring program.



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

**1993 Annual Report**

Prepared by:                      Staff of  
Northeast Utilities Service Company  
Environmental Services Division  
Aquatic Services Branch  
NU Environmental Laboratory

Approved by: *Milan Keser*  
Dr. Milan Keser

*Ronald C. Nichols*  
Ronald C. Nichols

April 1994



We dedicate this report to Dr. Linda E. Bireley, who recently took the position of Manager-Generation and Environmental Licensing, and who, for the past twenty years, had been involved with virtually every step in the development of the MNPS Environmental Monitoring Program.

## Acknowledgements

This report was prepared by the staff of Northeast Utilities Service Company (NUSCO), Environmental Services Division, Aquatic Services Branch, located at N U Environmental Laboratory (NUEL), Millstone Nuclear Power Station, PO Box 128, Waterford, CT 06385. Staff members include:

Dr. Milan Keser, Manager

Ronald C. Nichols, Supervisor

John A. Castleman

Donald J. Danila

David G. Dodge

Christine P. Gauthier

Donald F. Landers

J. Dale Miller

John T. Swenarton

David P. Colby

Gregory C. Decker

James F. Foertch

Raymond O. Heller

Dr. Ernest Lorda

Douglas E. Morgan

Joseph M. Vozarik

Special appreciation is extended to the Environmental Programs Branch staff and our summer and intern staff (Kirsten Berg, Calli Conway, Paul Jacobson, Jr., Clay Livingston, Karen Main, Jessica Spelke, Kim Tuttle, Cynthia Walker, and Keith Wheeler) for their untiring efforts in field and laboratory support. Additional thanks to JoAnne K. DeRico, Richard A. Larsen and Henry R. Paul who no longer work for Aquatic Services Branch, but whose contributions to the monitoring program are gratefully acknowledged. Critical reviews of this report were provided by the following members of the Millstone Ecological Advisory Committee: Dr. John Tietjen (City University of New York), Dr. Nelson Marshall (emeritus, University of Rhode Island), Dr. Saul Salla (Professor emeritus, University of Rhode Island), Dr. William Percy (Oregon State University), Dr. Robert Wilce (University of Massachusetts), and Dr. Robert Whitlatch (University of Connecticut).



## Executive Summary

### Lobster Studies

The American lobster, *Homarus americanus*, is one of the most valuable species in the Connecticut fishing industry. Between 0.8 and 2.7 million pounds have been landed annually since 1978 yielding between 2.4 and 8.4 million dollars to lobstermen employed in the fishery. Lobsters are highly exploited throughout their range; in the Connecticut lobster fishery, more than 90% of legal-sized lobsters are harvested each year. New fishery regulations have been implemented throughout the lobster range to reduce the high fishing mortality rates and to increase larval production and subsequent recruitment. Since 1984, Connecticut lobstermen have been required to install escape vents in traps; the escape vents allow sublegal-sized lobsters to escape from traps and thereby reduce injury and mortality to this portion of the population. The minimum legal size (carapace length) of lobster was gradually increased in Connecticut from 81.0 mm ( $3 \frac{3}{16}$  in) in 1988 to 82.6 mm ( $3 \frac{1}{4}$  in) in 1990. Because of the regional economic importance of lobsters, the local population of lobsters in the Millstone Point area have been sampled annually from May through October since 1978 to determine if operation of Millstone Nuclear Power Station (MNPS) has caused changes in local lobster abundance beyond those expected from natural variability and high fishing mortality rates. Some changes were observed in abundance and population characteristics of lobsters during 1993, but they were most likely related to high fishing levels and changes in fishery regulations (escape vents, minimum legal size increase) rather than to power plant impacts.

The total number of lobsters caught and catch per unit effort (CPUE) of all sizes of lobster reached record levels in 1992 and remained high during 1993. Total number caught (10,195) and CPUE (2.301) in 1993 were the second highest reported (previous ranges were 6,376-11,438 and 0.904-2.457). While a significant increasing trend was observed in total CPUE from 1978 to the present, legal lobster catches (those individuals  $\geq 82.6$  mm carapace length) have significantly declined since the study began in 1978. The CPUE of legal-sized lobsters was 0.080 during 1993 and within the range reported in previous 3-unit studies, but lower than the range reported during 2-unit studies (0.098-0.173).

The mean carapace length of lobsters caught during 1993 (70.8 mm) was larger than previous 3-unit values (69.5-70.2 mm), but within the range of values reported in 2-unit study years. Only 3.3% of the catch was of legal-size during 1993, within the range of 3-unit years but below the range of 2-unit years (5.9-9.1%). Percentage of females carrying external eggs (berried) during 1993 was 12.2% and higher than in any previous study year (3.1-12.1%). Berried females were, however, smaller during 3-unit years (76.5 mm) than during 2-unit years (79.4 mm), reflecting the high proportion (90%) of berried females below the minimum legal size of 82.6 mm.

The most important factor regulating molting and growth of lobsters is water temperature. Water temperatures during the 3-unit study years were, on average, slightly warmer than during 2-unit study years. As a result, the peak in catches of molting lobsters in 3-unit years occurred earlier (by 9 days) than during 2-unit years. Lobster growth per molt, as determined by tag and recapture studies, averaged 13.7% for both males and females in 3-unit studies, and was slightly higher than growth per molt observed during 2-unit studies (males 13.3% and females 13.0%).

Results from tag and recapture studies indicate that the overall percentage of recaptures in our traps was similar during 2- and 3-unit years (19% vs. 20%), whereas the percentage of recaptures by commercial lobstermen declined from 33% during 2-unit years to 18% during 3-unit years. This decline of recaptures in commercial traps was related to the 1984 escape vent regulation and not to MNPS operation. Installation of escape vents, coupled with the fact that most of our tagged lobsters are sublegal, resulted in fewer tagged lobsters retained in commercial traps. Conversely, our traps did not have escape vents and retained similar numbers of tagged sublegal lobsters. Lobster tagging also indicated that local individuals are predominantly nonmigratory. Over 90% of the tagged lobsters recaptured in commercial traps were caught within 5 km of Millstone Point. The average distance traveled by lobsters before they were caught in commercial pots was similar during 2- and 3-unit years (2.4 vs. 2.9 km). Although a predominance of localized movement was observed in our study, a number of lobsters (113) were reported caught outside LIS along the Rhode Island and Massachusetts



coasts, and in offshore deep water canyons on the edge of the continental shelf.

Since 1984, lobster larvae have been sampled from May to August at the discharges of Units 1, 2 or 3. The average density of larvae collected during 1993 was 1.081 per 1000 m<sup>3</sup> of cooling water, which was the second highest value reported (previous ranges were 3-unit 0.525-1.334; 2-unit 0.409 and 0.504). Estimates of total lobster larvae entrained through the plants cooling water system were based on sample density of larvae and total MNPS cooling water demand during the May to August hatching period. During 1993 an estimated 389,767 larvae were entrained, within the range of previous 3-unit years (296,173-615,285), but higher than the 2-unit estimates of 77,458 and 128,550. Entrainment numbers have been substantially higher since Unit 3 began operating because the cooling water demand of Unit 3 alone is about the same as that of Units 1 and 2 combined. The potential impact of lobster larvae entrainment is difficult to assess because of the uncertainty that exists concerning larval origin and larval survival and recruitment rates to legal size. Since lobsters require 4-5 years of growth before they are vulnerable to capture, and an additional 2 years of growth to reach legal size, a decline in local lobster abundance caused by larval entrainment would not be apparent for several years.

At present, fishery regulations implemented in 1984 (escape vents) and 1988 (increased minimum size) to preserve the LIS lobster resource appear to be effective. The percentage of berried females has increased each year since the minimum legal size was first increased and lower incidence of claw loss and reduced retention of sublegal sized lobsters in commercial traps were attributed to the use of escape vents. However, fishing effort has more than doubled since 1978 and further increases may offset some of the benefits of the new regulations.

### Eelgrass

Eelgrass is the predominant marine flowering plant in estuaries and lagoons of temperate and warm boreal coasts in the Atlantic and Pacific Oceans. In the vicinity of Millstone Point, eelgrass populations can exhibit wide temporal and spatial variability in shoot length, plant density, standing stock biomass and other population parameters. During 1993, these parameters were measured at three sites in the MNPS area: Jordan Cove (JC), White Point (WP) and Niantic River (NR).

Eelgrass population parameters (plant density, shoot length, standing stock biomass and percentage of reproductive shoots) and sedimentary characteristics (mean grain size, organic content and silt/clay percentage) measured in 1993 were generally within the ranges of previous years; the only exceptions were annual biomass at WP, which was the highest yet reported, and percentage of reproductive shoots at JC and NR, which were also maxima.

The eelgrass population at WP remains unaffected by MNPS operation, as population parameters have been generally stable since 1985. At NR, wide fluctuations of abundance have occurred, including localized elimination of plants from NR #1 (1986-87), from NR #2 (1987), and from NR #3 (1987-93). These abundance fluctuations, however, are related to factors other than MNPS operation because the Niantic River is not impacted by the thermal effluent. Furthermore, in recent years NR #1 has been recolonized and the eelgrass population has apparently recovered completely. At JC, changes in eelgrass abundance may be related to changes in water temperature, but, at least to date, these changes appear to be the result of natural variability rather than an impact of 3-unit operation.

### Rocky Intertidal Studies

Attached rocky shore communities, as described by the NUSCO monitoring program, continue to serve as effective integrators of local environmental conditions in the vicinity of MNPS. Conditions resulting in much of the variability among communities at sampling sites outside the influence of MNPS were related to natural factors including site orientation to prevailing wind-generated waves, the ability of available substratum (slope) to dissipate the horizontal force of those waves, and the character of that substratum (e.g., boulders, bedrock ledge, etc.). Community differences beyond those attributed to natural factors occurred within the thermal plume area at sites located on Fox Island (FE and FN), and were directly attributed to MNPS operation. Various aspects of the impact-related community changes at Fox Island were identified through separate studies which included qualitative algal sampling, estimation of intertidal organism abundance, and studies of local *Ascophyllum nodosum* populations.

Elevated temperature conditions caused by the 3-unit thermal plume allowed development of a unique flora at FE. The most notable shifts in species occurrence, revealed by qualitative algal sampling, were the

presence of warm water-tolerant species not typical of other sites (*Agardhiella subulata*, *Gracilaria tikvahiae* and *Sargassum filipendula*), absence of common cold water species (*Mastocarpus stellatus*, *Dumontia contorta* and *Polysiphonia lanosa*) and extended or reduced periods of occurrence of seasonal species with warm or cold water affinities, respectively.

During 1993, power plant impacts on dominant species abundance patterns, caused by 2-cut water circulation patterns and by 3-unit operations, were observed only at FE, and were most pronounced in the low intertidal, where temperature conditions were most severe. The low intertidal community at FE, which prior to 1983 had been unimpacted and characterized by perennial populations of *Fucus*, *Chondrus* and *Ascophyllum* and predictable seasonal peaks in barnacle abundance, has been replaced by a persistent community dominated by *Codium*, *Ulva*, *Enteromorpha* and *Polysiphonia*. Also, populations of species observed in undisturbed transects only at FE (*Sargassum*, *Gracilaria*) persisted during 1993.

*Ascophyllum* populations at three stations in the vicinity of MNPS continued to be monitored in 1993. Elevated temperatures (2-3°C above ambient) at our station nearest the discharge (FN) caused *Ascophyllum* to grow longer and more rapidly at this site, relative to stations farther away. A moderate level of growth enhancement was observed at FN during 1992-93, when compared to previous years, attributed to lessened thermal plume incursion resulting from an extended outage of Unit 2 for much of the peak growing season. As in previous years, *Ascophyllum* mortality, or loss of tagged plants and tips, at our present sampling sites was not related to proximity to the power plant but rather to degree of exposure to prevailing winds and waves.

### Benthic Infauna

During 1993, infaunal communities inhabiting soft-bottom subtidal habitats in the vicinity of MNPS were sampled quarterly as part of a long-term monitoring program. These communities were characterized in terms of species composition, abundance, and sedimentary parameters in order to identify spatial and temporal patterns in community structure and to assess whether observed changes can be attributed to construction or operation of the power plant.

Changes in sediments resulting from Unit 3 construction and initial operation events have resulted in alterations to associated infaunal communities in recent years. During 1993, the effluent site (EF) con-

tinued to show evidence of sediment scour, and Jordan Cove (JC) sediment deposition. The intake site (IN) exhibited continued recovery from dredging activities near the intakes in the early 1980s, and Giants Neck (GN), as the control site, continued to show little temporal variation and no effect of MNPS operation.

The dominant taxa collected during 1993 at subtidal stations included the polychaete species *Aricidea catherinae*, *Tharyx* spp., *Prionospio steenstrupi*, *Polycirrus eximius*, *Scoletema tenuis*, *Protodorvillea gaspeensis*, *Mediomastus ambiseta*, *Pygospio elegans*, the arthropods *Ampelisca vadorum*, *A. verrilli* and *Leptocheirus pinguis* and representatives of the class Oligochaeta. The top four ranked taxa at each station in 1993 accounted for 50% or more of all individuals. In most cases, these organisms have been the dominant subtidal taxa in both 2-unit and 3-unit operational periods. Most stations were characterized by one or more clearly dominant taxon (oligochaetes at EF, GN and JC, *Aricidea catherinae* at GN and JC and *Tharyx* spp. at GN) during both operational periods. There has been no single dominant taxon at IN during either operational period, where mean relative abundance of any single taxon rarely exceeded 10%. Analyses of local benthic communities have identified changes and long-term trends in community parameters, and have permitted distinction between changes related to natural variability, and those caused by power plant operation.

### Marine Woodborer Study

The Marine Woodborer Study report describes the local distribution of *Teredo bartschi*, a semitropical shipworm common from Texas to South Carolina, but capable of establishing isolated populations near thermal discharges in more northern climates. *T. bartschi* remains in MNPS discharge waters and, in 1993, it was collected for the first time in panels 500 m from the quarry cuts. Reduced currents around the rock outcroppings at this new site may trap discharge water and increase the probability of collecting this immigrant species. The absence of *T. bartschi* at EF in 1993 is probably related to unusual conditions resulting from Unit 3 being off-line from August to November, during the peak recruitment period for this species. The distribution of *T. bartschi* remains closely associated with the discharge waters of MNPS, suggesting that the discharge population has not adapted to ambient temperature conditions at White Point, 1700 m from the quarry cuts. Even though the current program represents a large

reduction in sampling effort relative to previous study years, it continues to effectively monitor the abundance of *T. bartschi* in the Millstone area.

### Fish Ecology Studies

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

Trawl, seine and ichthyoplankton monitoring programs were established in 1976 to determine the impact of MNPS on local fish assemblages. Of the 120 different fish taxa that have been collected since then in these programs, seven taxa (American sand lance, anchovy, grubby, silversides, tautog, cunner and winter flounder) have been identified as having the potential to be impacted by MNPS either by entrainment or because of their susceptibility to thermal impacts.

Abundance data were analyzed separately for two-unit (1976-1985) and three-unit (1986-1992) operational periods and for the entire 17-year data series (both periods combined) to determine if changes in fish abundance have occurred. Larvae of sand lance and anchovy have declined as have adult tautog and cunner. Because so many factors affect the abundance of these taxa the reasons for these declines are difficult to ascertain. American sand lance larvae has ranked third among fish larvae entrained and it has significantly decreased in abundance in the entrainment and offshore samples. The bay anchovy is typically the most abundant ichthyoplankton species collected in estuaries within its range and it was the dominant larval taxa entrained at MNPS. Similar to the sand lance, this fish also exhibits large natural abundance fluctuations. Along the Connecticut coast, the Atlantic silverside and the inland silverside are among the most common shore-zone species. Typical of short-lived species, the catches of Atlantic silverside by trawl and seine were highly variable and annual catch indices ranged over two orders of magnitude. The trawl CPUE of Atlantic silversides was at a 17-year high at the two Niantic Bay Stations (NB and IN), and all 1992-93 trawl catches were

above the two-unit annual averages. The catches of Atlantic and inland silversides in seines were all within historic ranges and above the two-unit period average, except for Atlantic silversides at JC. The grubby is unique because unlike the other potentially impacted fishes it experiences no fishing pressure. Both larval and adult grubby abundance indices have been stable throughout the 17 years of monitoring. Tautog has been the second-most abundant egg taxon entrained and has accounted for more than 30% of the total eggs collected since 1979. The tautog egg entrainment estimate for this report year was the lowest since three-unit operation began and the average density of eggs at the entrainment site (EN) was the second lowest since sampling began. Larval tautog average densities at EN were within their historic range. The cunner egg entrainment estimate also was the lowest since Unit 3 began operating. Cunner larval densities were within their historic range. Prior to 1992-93, the trawl catch of cunner had been decreasing at all six stations. This year's trawl catches were below the two-unit operational catches at two inshore stations, but were at a historic high in Niantic Bay. Both tautog and cunner young-of-the-year have accounted for a high proportion of the fish caught in the trawl since three-unit operation began.

Because over 85% of the eggs entrained at MNPS were tautog and cunner eggs, special studies were conducted in 1993 to determine the entrainment mortality of these eggs. The average hatching rate was 4%. To examine daily fluctuations of egg abundance, samples were collected every two hours during three 24-hour periods. Examination of the geometric mean for each 2-hour sampling period showed that on the average, daily peak spawning for cunner and tautog occurred at about 1800 hours and then declined rapidly. Estimated mortality rate during this rapid decline was 44% per hour for cunner and 47% per hour for tautog. These very high egg mortality rates may account for the low numbers of cunner and tautog larvae collected compared to the large number of eggs of these two fishes.

### Winter Flounder Studies

The local Niantic River population of the winter flounder (*Pleuronectes americanus*) is potentially affected by the operation of MNPS, particularly by entrainment of larvae through the cooling-water systems of the units. As a result, intensive studies of the life history and population dynamics of this



valuable sport and commercial species have been undertaken since 1976.

The median trawl catch-per-unit-effort (CPUE) of fish larger than 15 cm collected in the Niantic River during the February-April 1993 spawning season was 1.9. This value was only about 30% of the CPUE of 6.2 for last year, and it was the smallest CPUE in the 18-year time-series. The Jolly stochastic model was used with mark and recapture data to estimate the absolute abundance of the adult spawning population (all winter flounder larger than 20 cm, which includes some immature fish). The most recent abundance estimate of 12,178 fish for 1992 was only 19% of the 1991 estimate and further illustrated the severe decline of winter flounder abundance in recent years.

Each year, about one-third to one-half of the winter flounder found in the Niantic River during the spawning period have been mature females. Using available information on sex, age, and size composition, the annual female winter flounder parental stock sizes have been estimated for the past 17 years. These estimates have ranged from 7,821 (1993) to 78,629 (1982) fish, with corresponding total egg deposition ranging from about 6.4 (1993) to 45.6 billion (1982).

Estimates of larval winter flounder abundance at the MNPS discharge (entrainment sampling) have been obtained since 1976, at a station in mid-Niantic Bay since 1979, at three stations in the Niantic River since 1983, and at the mouth of the Niantic River during 1991-93. The low abundance of newly-hatched larvae in Niantic Bay compared to the Niantic River suggested that most local spawning occurred in the river. Larval abundance in 1993 was the lowest in both the Niantic Bay and River since sampling began in 1976 and 1983, respectively. Annual larval abundances in the bay for 1976-93 appeared to reflect region-wide trends, because they were highly correlated to abundance indices in Mount Hope Bay, MA and RI.

Larval developmental stage and length were closely related. Smaller larval size-classes predominated in the river and larger size-classes were more prevalent in the bay. In Niantic Bay, growth and development were correlated with water temperature, and in the river growth appeared to be related to both water temperature (positively) and larval density (negatively). Estimated mortality of larvae in the Niantic River for 1984-93 ranged from about 82 to 98%. Mortality was consistently highest during Stage 2 of development (3- to 4-mm size-classes), which is when feeding first occurs; larval abundance

declined about 90% during this stage in 1993. This stage may include a "critical period" for winter flounder as survival rates generally improve progressively for larger size-classes.

Young-of-the-year winter flounder have been collected during late spring and summer in the Niantic River since 1983 and in Niantic Bay since 1988. In 1993, abundance of newly metamorphosed young was relatively low, particularly in the Niantic River. Mortality was once again apparently quite high in the bay, with no fish caught there by mid-summer. Late-season median densities at two stations in the river were among the lowest found since 1983.

An index of abundance was calculated for young-of-the-year fish taken during the late fall and early winter at the trawl monitoring program stations. The 1992-93 abundance index (1992 year-class) was 31.1, the highest in the 17-year series. This was consistent with the high abundance of these fish found during the summer of 1992 and was indicative of the strength of the 1992 year-class. However, relatively few of these fish were taken within the Niantic River during the adult spawning population surveys in early 1993. Young-of-the-year abundance indices were not significantly correlated with those for age-4 and 5 female adult spawners. Therefore, none of the early life stages could be used as a reliable index of year-class strength for Niantic River winter flounder stock.

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

The number of larvae entrained through the condenser cooling-water system at MNPS is the most direct measure of potential impact on winter flounder. Annual estimates of entrainment were related to larval densities in Niantic Bay, as well as to plant operation. The entrainment estimate for 1993 of 41.1 million was the lowest since three-unit operation began in 1986 and was one of the lowest in the 18-year series. Low entrainment was attributed to low larval abundance, as all MNPS units operated throughout much of the larval winter flounder season during 1993.

The impact of larval entrainment on the Niantic River stock depends upon the fraction of its production that is entrained each year. Empirical mass-balance calculations for 1984-93 indicated that a large number of entrained larvae came from areas of Long Island Sound other than the Niantic River. An estimated 11 to 35% of the larvae entrained at MNPS appeared to have originated from the Niantic River during these years. Percentages of the river production that were entrained annually ranged from about 4 to 21% and the estimated fraction of Niantic River winter flounder production that would be entrained under full (100% capacity) three-unit operation ranged from about 5 to 24%.

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

calculations. All SPDM runs were stochastic and consisted of 100 Monte Carlo replicates for each yearly stock projection over a 100-year period (1960-2060).

An initial stock size of 97,075 lbs was used to represent the theoretical (no fishing assumed) maximum spawning potential (MSP) of the Niantic River female spawning stock. When fishing effects were simulated, the annual projections of the initially unfished stock become the baseline time series of annual spawning biomass for Niantic River winter flounder subjected to fishing, but in the absence of any plant impact. Under the exploitation rates simulated, the stochastic mean stock size of the baseline declined to about 48,300 lbs in 1971 and to 12,300 lbs in 1993. The latter value was about one-half of a generally accepted critical stock size, defined as 25% of MSP. Following simulated reductions in fishing, however, the stock rapidly recovered. A new series of stock size projections were then simulated by adding the effect of larval entrainment at MNPS. The lowest projected stock biomass under simultaneous fishing and effects of MNPS occurred in 1993 (10,600 lbs), whereas the greatest absolute decline relative to the baseline occurred in 2001 (a difference of 7,800 lbs). Generally, however, greater reductions in stock biomass resulted from fishing than from larval entrainment, because fishing tends to remove larger fish and reduce average weight of the remaining spawners. The simulated spawning stock returned to near-baseline levels about 6 years after the scheduled termination of Unit 3 operation in 2025.

The probabilities that the Niantic River female spawning stock biomass would fall below selected reference sizes (25, 30, and 40% of MSP) were determined to help assess the long-term effects of MNPS operation. A stock less than 25% of MSP is considered overfished, whereas one that is at 40% of MSP can maximize yield to the fisheries while remaining stable. For both baseline and MNPS-impact simulations, it was likely ( $p \geq 0.87$ ) that the stocks were greater than 40% of MSP in 1970. At the lowest point of both stock projections in 1993, all replicates were less than 25% of MSP. Simulated reductions in fishing allowed for a rapid increase in spawner biomass and it was likely greater than 30% of MSP by 2010 ( $p \geq 0.97$ ) and had a better than even ( $p = 0.56$ ) chance of being greater than 40% of MSP by 2020. These increases, however, assumed that changes in fishing regulations would be implemented as scheduled and that they would achieve the expected reductions in fishing mortality.



## Table Of Contents

Acknowledgements . . . . .	iii
Executive Summary . . . . .	v
Introduction . . . . .	1
Lobster Studies . . . . .	9
Eelgrass . . . . .	35
Rocky Intertidal Studies . . . . .	51
Benthic Infauna . . . . .	81
Marine Woodborer Studies . . . . .	103
Fish Ecology Studies . . . . .	111
Winter Flounder Studies . . . . .	141



## Introduction

### Reporting Requirements

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

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

In addition, paragraph 7 of the permit requires that:

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

Furthermore, a decision and order of the Connecticut Siting Council (CSC) requires that NNECO inform the Council of results of MNPS environmental impact monitoring studies and any modifications made to these studies (paragraph 6 of the proceeding

entitled "Docket No. 4, Certificate of Environmental Compatibility and Public Need for an Electric Generating Facility Identified as 'Millstone Nuclear Power Station, Unit 3,' located in the Town of Waterford, Connecticut" and dated March 22, 1974). This report satisfies the requirements of the NPDES permit and of the CSC by updating and summarizing various studies conducted at MNPS that were presented most recently in NUSCO (1993).

### Study Area

MNPS is situated on Millstone Point, about 8 km west-southwest of New London on the Connecticut shore of LIS (Fig. 1). The property, covering an area of about 200 ha, is bounded to the west by Niantic Bay, to the east by Jordan Cove, and to the south by Twotree Island Channel. The MNPS monitoring programs sample a study area of approximately 50 km<sup>2</sup> 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<sup>3</sup>·sec<sup>-1</sup> and at maximum is about 8,500 m<sup>3</sup>·sec<sup>-1</sup> (NUSCO 1983). Current velocities are about 1 to 1.8 knots in the channel, slightly less (1 to 1.5 knots) near the plant and in Niantic Bay, and relatively weak in Jordan Cove and in the upper Niantic River. The currents are driven by semi-diurnal tides that have a mean and maximum range of 0.8 and 1.0 m, respectively. Thermal and salinity induced stratification may occur in regions unaffected by strong tidal currents. The greatest temperature variation has been observed in nearshore areas where water temperature can vary from -3 to 25°C; salinity varies much less and ranges from 26 to 30‰. The bottom is generally composed of fine to medium sand throughout the area, but also includes some rock outcrops and muddy sand, especially near shore. Strong winds, particularly from the southwest, can at times result in locally heavy seas (up to 1.5 m or greater) near Millstone Point. Additional information on local hydrography and meteorology can be found in NUSCO (1983).

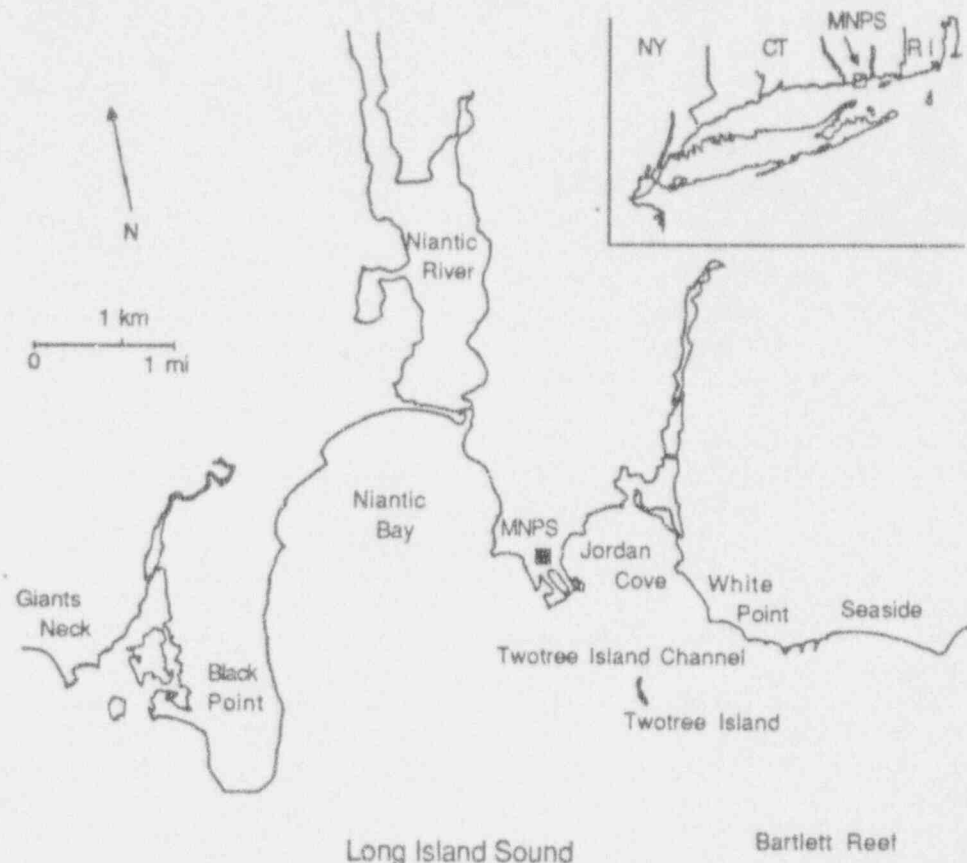


Fig. 1. The area in which biological monitoring studies are being conducted to assess the effects of the operation of MNPS.

### Millstone Nuclear Power Station

The MNPS complex consists of three operating nuclear power units; a detailed description of the station was given in NUSCO (1983). Unit 1, a 660-MWe boiling water reactor, began commercial operation on November 29, 1970; Unit 2 is an 870-MWe pressurized water reactor that began commercial operation in December 1975; and Unit 3 (1150-MWe pressurized water reactor) commenced commercial operation on April 23, 1986. All three units use once-through condenser cooling water systems with rated circulating water flows of 26.5, 34.6, and 56.6  $\text{m}^3\text{-sec}^{-1}$  for Units 1 through 3, respectively. Cooling water is drawn from depths of about 1 m below mean sea level by separate shoreline intakes located on Niantic Bay (Fig. 2). The intake structures, typical of many coastal power plants, have coarse bar racks (6.4 cm on center, 5.1-cm gap) preceding vertical traveling screens to protect the

plants from debris. Units 1 and 2 have always had 9.5-mm mesh traveling screens. Unit 3 originally had 4.8-mm mesh screens, a combination of 9.5- and 4.8-mm mesh screens from early 1990 through summer 1992, and only 9.5-mm mesh screens as of August 15, 1992. Fish return systems (sluiceways) were installed at Unit 1 in December 1983 and at Unit 3 during its construction to return aquatic organisms washed off the traveling screens back to LIS. The installation and operation of these sluiceways have minimized the impact of impingement at MNPS (NUSCO 1986, 1988a). A chronology of significant events associated with MNPS construction and operation, including installation of devices designed to mitigate environmental effects and unit operational shutdowns exceeding 2 weeks, are found in Table 1. Capacity factors (electricity produced as a percentage of maximum possible production) during 1993 were 92.8% for Unit 1, 82.5% for Unit 2, and 65.0% for Unit 3.

TABLE 1. Chronology of major construction and operation events at MNPS through 1993.

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



TABLE 1. (cont.).

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

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

MNPS cooling water is nominally heated in Units 1, 2, and 3 from ambient temperature to a maximum of 13.9, 12.7, and 9.5°C, respectively. Each unit has separate discharge structures that release the effluent into an abandoned granite quarry (ca. 3.5 ha surface area, maximum depth of approximately 30 m). The thermal discharge (about 11°C warmer than ambient under typical three-unit operation) exits the quarry through two channels (cuts), whereupon the thermal effluent mixes with LIS water (Fig. 2). The cuts are equipped with fish barriers consisting of 19-mm metal grates, which serve to keep larger fish out of the quarry. The thermal plume is warmest in the immediate vicinity of the cuts and within about 1,100

m of the quarry the surface-oriented plume cools to within 2.2°C above ambient. Beyond this distance the plume is highly dynamic and varies with tidal currents (Fig. 3). Hydrothermal surveys conducted at MNPS were described in NUSCO (1988b).

### Monitoring Programs

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

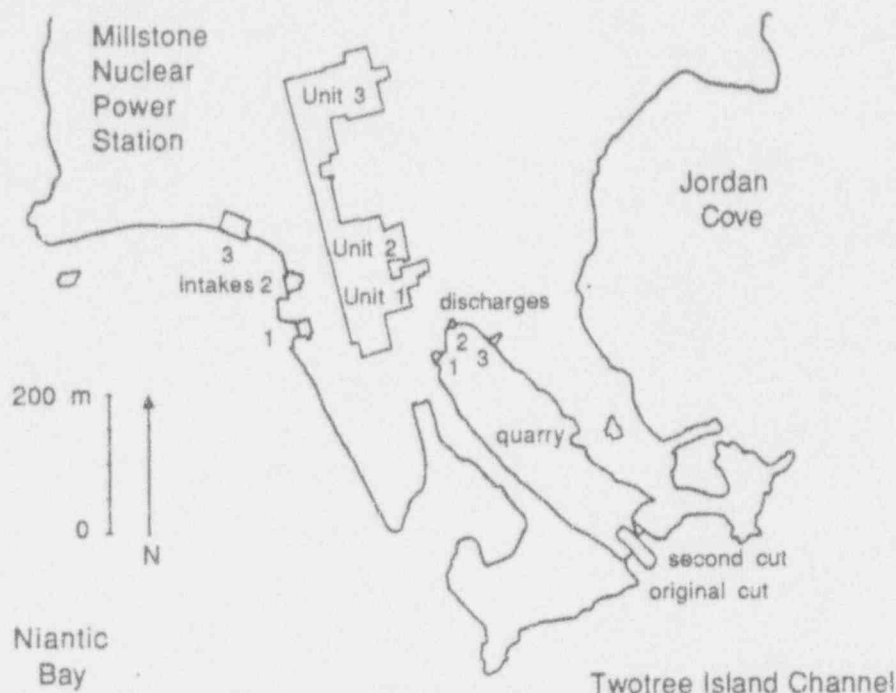


Fig. 2 The MNPS site, showing the intake and discharge of each unit, the quarry, and the two quarry discharge cuts.

nificance of changes found for the 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, Lobster Studies, Fish Ecology Studies, Benthic Infauna, Rocky Intertidal Studies, Marine Woodborer Studies, and Eelgrass. Reporting periods for each section vary and were predicated on biological considerations and processing time necessary for samples, as well as on regulatory requirements. In cases where the seasonal abundance of organisms differed from arbitrary annual reporting periods, the periods chosen were adjusted to best define the season of interest for a particular species or community.

### References Cited

- NUSCO. (Northeast Utilities Service Company). 1973. Environmental effects of site preparation and construction. Pages 4.4-1 to 4.5-1 in *Millstone Nuclear Power Station, Unit 3, Environmental report. Construction permit stage.*
- NUSCO. 1978. Impingement studies. Millstone Units 1 and 2, 1977. Pages 1-1 to 4-2 in *Annual report, ecological and hydrographic studies, 1977. Millstone Nuclear Power Station.*
- NUSCO. 1983. Millstone Nuclear Power Station Unit 3 environmental report. Operating license stage. Vol. 1-4.
- NUSCO. 1986. The effectiveness of the Millstone Unit 1 sluiceway in returning impinged organisms to Long Island Sound. Enclosure to Letter D01185 dated May 27, 1986 from R.A. Reckert, NUSCO, to S.J. Pac, Commissioner, CT DEP. 18 pp.
- NUSCO. 1988a. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to Letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP. 21 pp.
- NUSCO. 1988b. Hydrothermal studies. Pages 323-355 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-87.*
- NUSCO. 1993. Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1992. 269 pp.

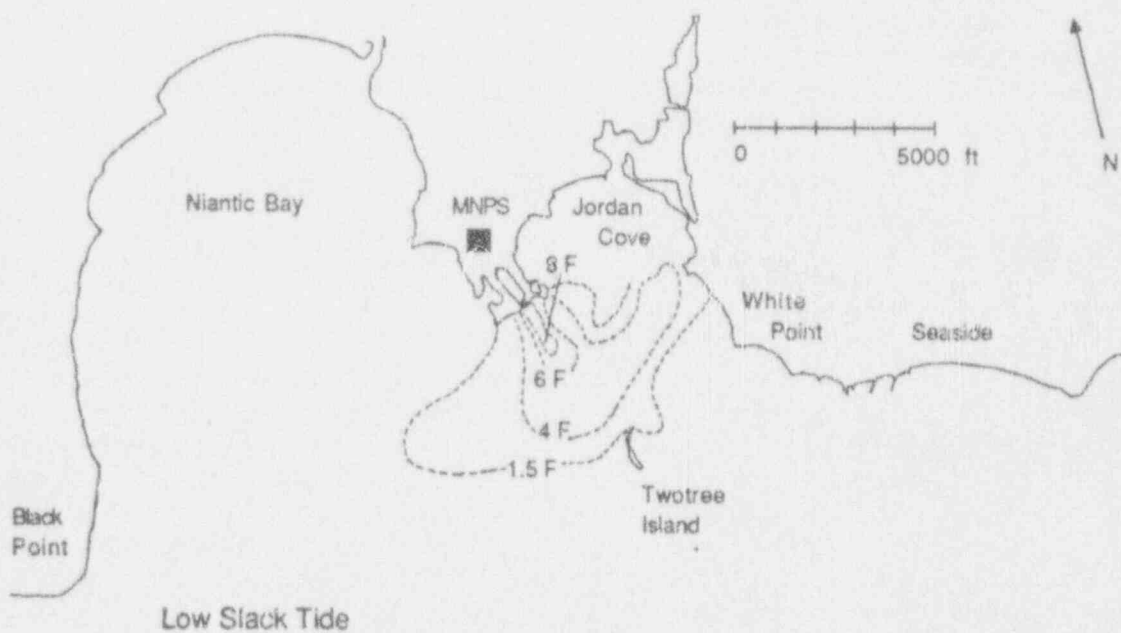
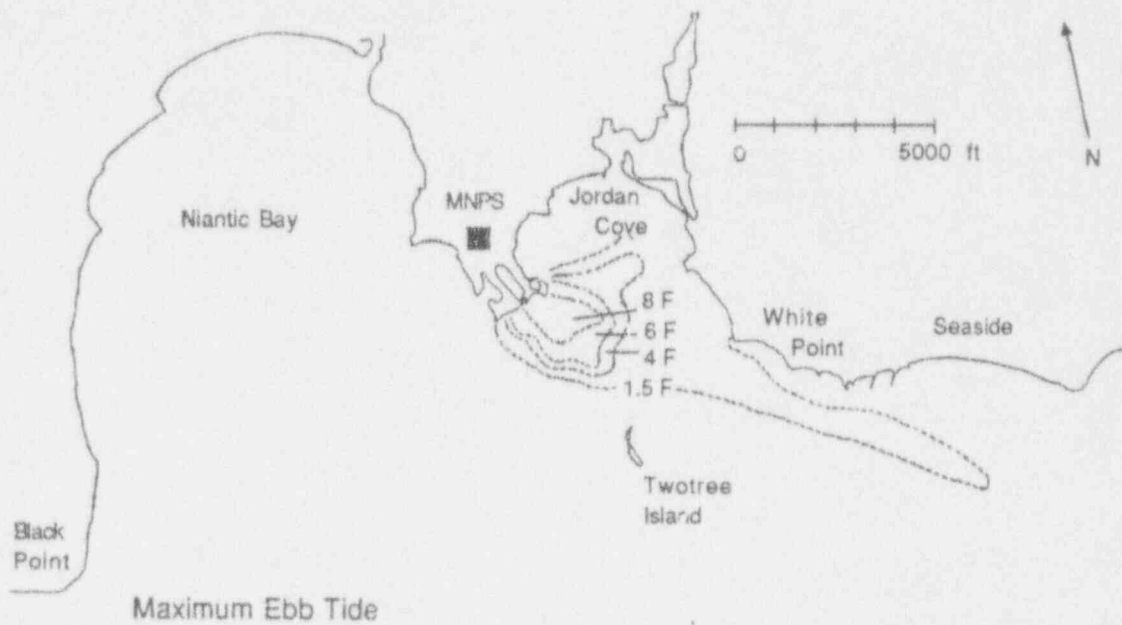


Fig. 3. Locations of selected three-unit thermal plume isotherms (1.5°F, 4°F, 6°F, and 8°F) under various tidal conditions.

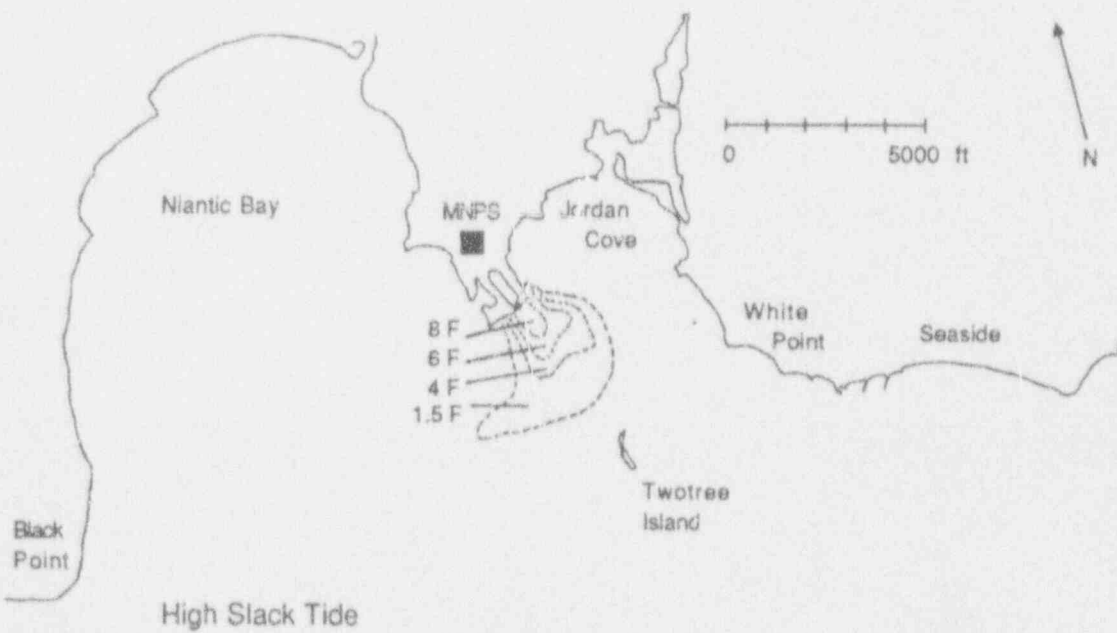
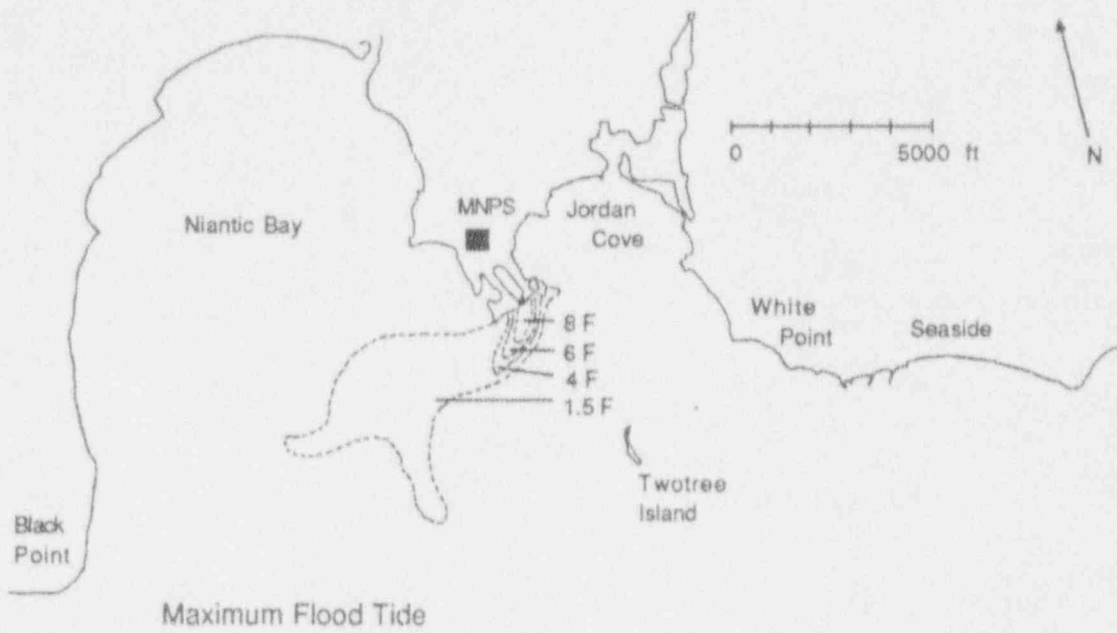


Fig. 3. (continued).





## Lobster Studies

Introduction .....	11
Materials and Methods .....	11
Results and Discussion .....	13
Temperature and Salinity .....	13
Abundance and Catch-per-Unit-Effort .....	14
Population Characteristics .....	16
Size Frequency .....	16
Sex Ratios .....	19
Reproduction .....	20
Molting and Growth .....	21
Culls .....	24
Tagging Program .....	24
Movement .....	25
Entrainment .....	26
Conclusions .....	28
References Cited .....	29



# Lobster Studies

## Introduction

The American lobster, *Homarus americanus*, is one of the most valuable species in the Connecticut fishing industry (Blake and Smith 1984). Annual landings in Long Island Sound (LIS) of 0.8 to 2.7 million pounds from 1978 to 1993 yielded between 2.4 and 8.4 million dollars to lobstermen employed in the fishery (Smith et al. 1989; Connecticut Department of Environmental Protection CT DEP, Marine Fishery Statistics). Nearly 30% of the total Connecticut landings during 1993 were made in New London county, which includes the Millstone Point area.

Lobsters are highly exploited throughout their range and overfishing could impact recruitment and lead to a decline in abundance of coastal populations (Anthony and Caddy 1980). To address the high fishing mortality rates, and to improve larval production and subsequent recruitment, the New England Fishery Management Council (NEFMC) recommended an increase of minimum legal size of lobsters in lobster producing states. In Connecticut, new lobster fishery regulations were implemented beginning in 1988, which increased the minimum legal size (carapace length) from 81.0 mm ( $3 \frac{3}{16}$  in) to 81.8 mm in 1989 ( $3 \frac{7}{32}$  in) and to 82.6 mm ( $3 \frac{1}{4}$  in) in 1990. Increases in minimum legal size should eventually improve yield per recruit, unless an escalation of fishing effort offsets the benefits of raising the minimum size. In 1992, while fishing effort continued to rise, landings in the United States and Canada substantially declined (down 13% and 20%, respectively) prompting further management action by the NEFMC. The Council proposed an amendment to the American Lobster Fishery Management Plan which incorporates measures to reduce fishing mortality (e.g., maximum size limit, trap limits, seasonal closures and closed areas) if the management objective of resource preservation is not met after three years.

The lobster fishery in Connecticut is almost completely dependent on new animals that molt into legal size each year. Lobsters in the Millstone Point area have been studied extensively since 1978 to determine if operation of the Millstone Nuclear Power Station (MNPS) caused changes beyond

those expected from natural variability and the high rate of fishing. The potential impacts of power plant operations on the local population of lobsters include entrainment of larval lobsters through the cooling water systems, impingement of juveniles and adults on the intake traveling screens, and effects of the heated discharge. Entrainment and impingement contribute additional mortality to the local lobster population and thereby may alter recruitment patterns.

The objectives of the lobster program are to evaluate year-to-year, seasonal, and among station changes in catch-per-unit-effort, as well as population characteristics such as size frequency, growth rate, sex ratios, female size at sexual maturity, characteristics of egg-bearing females, and lobster movements. Additionally, since 1984, studies have been conducted during the hatching season to estimate the number of lobster larvae entrained through cooling water systems. Impacts associated with plant operations on the local lobster population were assessed by comparing results of the 1993 study to other 3-unit operational study years (1986-1992) and to data collected during 2-unit operations (1978-1985). Results from the 2-unit period were also compared to combined 3-unit operational data (1986-1993) to assess impacts associated with the addition of Millstone Unit 3. When appropriate, results of our lobster study were compared to other studies conducted in LIS and throughout the range of the American lobster.

## Materials and Methods

Full description of methods used to conduct lobster population studies is in NUSCO (1982, 1987a). Four pot-trawls, each consisting of five double-entry wire pots (76 x 51 x 30 cm; 2.5 cm<sup>2</sup> mesh) equally spaced along a 50-75 m line buoyed at both ends, were used to collect lobsters from May through October. Pot-trawls were set near rocky outcrops at three stations (Fig. 1). Pots set in Jordan Cove (average depth 6 m) were 500 m east of the Millstone discharge. The Intake station (average depth 5 m) was 600 m west of the discharge near the power plant intake structures, and the Twotree station (average depth 12 m) was



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

located south of Millstone Point, about 1600 m offshore near Twotree Island. Beginning in 1984, pots were individually numbered to determine the variability in catch among pots, and to provide more accurate values for catch-per-pot than an average catch-per-pot based on a total of 20 pots per sampling location. Pots were hauled on Monday, Wednesday, and Friday of each week, weather permitting; on holiday weeks pots were checked on the first and last work days of each week. On each sampling trip, surface and bottom water temperatures and salinities were recorded at each station. Lobsters larger than 55 mm carapace length were banded to restrain chelipeds, brought to the laboratory, and kept in three separate tanks supplied with a continuous flow of seawater. After lobsters were removed, pots were rebaited with flounder carcasses and reset in the same area. On Fridays, lobsters caught that week were examined and the following data recorded: sex, presence of eggs (berried), carapace length (CL), crusher claw position, missing claws, and molt stage (Aiken 1973). Lobsters were tagged with a serially

numbered international orange sphyron tag (Scarratt and Elson 1965; Scarratt 1970), and released at the site of capture. Recaptured tagged lobsters, severely injured or newly molted 'soft' lobsters, and those smaller than 55 mm CL were released untagged after recording the above data.

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

Lobster larvae have been sampled from 1984 to 1993 during the period of their occurrence (May through July) at the discharges of Units 1, 2 and 3. Samples were collected with a 1.0 x 6.0 m conical plankton net of 1.0 mm mesh. The volume of cooling water sampled was estimated from the average readings of four General Oceanic

flowmeters located in the mouth of the net; about 4000 m<sup>3</sup> of cooling water were filtered in each sample by deploying the net for 45-60 minutes. Day and night samples were collected four days a week in all study years. Each sample was placed in a 1.0 mm mesh sieve and kept in tanks supplied with a continuous flow of seawater. Shortly after collection, samples were sorted in a white enamel pan; larvae were examined for movement and classified as live or dead. Lobster larvae were also classified by stage according to the criteria established by Herrick (1911). The abundance of larvae in entrainment samples was standardized as the number of larvae per unit-volume. The seasonal (May through July) mean density was calculated as the mean of the assumed "delta" distribution, referred to as  $\Delta$ -mean (Pennington 1983; NUSCO 1988a). To estimate the total number of larvae entrained, the  $\Delta$ -mean density was scaled by the total volume of water pumped through the plants during the sampling period.

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

Catch-per-unit-effort (CPUE, i.e., the number of lobsters caught per pot-haul) was used to describe the annual abundance of lobsters in the MNPS area. Because these CPUE data are ratios, which are not additive and have an asymmetric distribution about the arithmetic mean, the geometric mean was the chosen statistic to analyze trends in CPUE. The geometric mean is better suited for constructing asymmetric confidence intervals for skewed data (Snedecor and Cochran 1967; McConnaughey and Conquest 1993). Annual geometric mean CPUEs were calculated for all lobster sizes. The annual abundance (CPUE) of legal-size lobsters in the MNPS area was estimated by using the  $\Delta$ -mean. The  $\Delta$ -mean was a more appropriate statistic for describing the CPUE of legal-size lobster, since a large number of zero observations were present in the data (i.e.,

many pots contained no legal-size lobsters). Both geometric means of all lobsters and  $\Delta$ -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  $\Delta$ -mean abundance of legal-size lobsters is referred to as "mean legal CPUE". The distribution-free, Mann-Kendall test (Hollander and Wolfe 1973) was used to determine presence of significant trends in the time series of annual CPUE data, and of several other selected population characteristics. Trend slopes, when significant, were calculated using Sen's estimator of the slope (Sen 1968; Gilbert 1987).

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

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

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

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

## Results and Discussion

### Temperature and Salinity

Water temperature and salinity were measured on each sampling trip from May through October (1979-1993). Monthly mean surface water temperature at all stations ranged from 9.4 to



TABLE 1. Mean monthly surface and bottom water temperatures measured at each station during 2-unit (1979-85) and 3-unit studies (1986-93).

	Temperature			
	Surface	Bottom	Surface	Bottom
	2-Unit	3-Unit	2-Unit	3-Unit
<u>Jordan Cove</u>				
MAY	10.2	12.4	9.2	9.9
JUN	15.1	17.1	13.9	14.4
JUL	19.5	20.8	18.0	18.1
AUG	21.2	22.2	19.9	20.0
SEP	20.3	21.4	19.2	19.2
OCT	16.8	17.2	16.0	16.1
<u>Intake</u>				
MAY	10.1	11.1	9.3	10.1
JUN	14.9	15.7	14.1	14.7
JUL	19.2	19.4	18.3	18.5
AUG	20.7	21.0	20.1	20.2
SEP	19.8	20.1	19.4	19.5
OCT	16.1	16.5	15.9	16.1
<u>Twotree</u>				
MAY	9.4	9.9	8.9	9.4
JUN	14.2	14.7	13.7	14.1
JUL	18.3	18.4	17.7	17.8
AUG	20.0	20.2	19.6	19.7
SEP	19.3	19.3	19.1	19.0
OCT	15.9	15.9	15.9	15.8

21.2°C during 2-unit studies and was lower than the range of monthly means during 3-unit studies (9.9-22.2°C; Table 1). Bottom water temperatures ranged from 8.9 to 20.1°C during 2-unit studies and from 9.4 to 20.2°C during 3-unit studies. When water temperatures from the three stations were compared, the highest temperatures were recorded at Jordan Cove during both 2-unit (10.2-21.2°C) and 3-unit studies (12.4-22.2°C). In general, temperatures (surface and bottom) at Jordan Cove and Intake (4-6 m depth) were similar and consistently warmer than at Twotree (12 m depth). These temperature data agreed with results of hydrothermal studies, which indicated that a 2.2°C isotherm resulting from 3-unit operation could extend into the Jordan Cove area. In addition, a 0.8°C isotherm extends 600 m from the discharge to a depth of 3 to 5 m (NUSCO 1988c), and may reach the bottom 500 m from the discharge at the Jordan Cove and Intake sites where some of the pot-trawls are deployed.

Salinities were similar during 2-unit and 3-unit operation; mean surface and bottom water salinities ranged between 29.4 and 31.6‰. Due to the spring freshwater runoff, salinities were generally lower at each station in May and June.

### Abundance and Catch-per-Unit-Effort

Number of lobsters caught during 1993 (10,195) was the second highest annual total catch observed during either 2-unit (6,376-9,109) or previous 3-unit (7,106-11,438) operational study periods when 20 traps were used at each station (Table 2). The lower catches from 1978 to 1981 of 2-unit studies (1,824-3,259) occurred when only 10 wire pots were used at each station. The geometric mean total CPUE for 1993 of 2.301 lobsters/pot was also the second highest value reported since the study began in 1978 (2-unit range 0.904-2.006; previous 3-unit range 1.531-2.457). In general, total CPUE during 2-unit studies (1978-85) was lower (1.364) than that observed during 3-unit studies (1986-93; 1.810), and a significant increasing trend (slope=0.048,  $p=0.03$ ) was observed in the time series (1978-1993) of total CPUE. The  $\Delta$ -mean CPUE of legal-size lobsters ( $\geq 82.6$  mm) was 0.080 in 1993, which was within the range of values reported in other 3-unit studies when the legal size was 81.0 mm (1986-88  $\Delta$ -mean range=0.079-0.086), 81.8 mm (1989  $\Delta$ -mean=0.065) and 82.6 (1990-92  $\Delta$ -mean range=0.076-0.091). However, legal CPUE during 1993 was lower than any legal CPUE reported in 2-unit studies (1978-85  $\Delta$ -mean range=0.098-0.173). Legal catches in our traps have significantly declined since 1978 (slope = -0.005,  $p=0.002$ ), most likely due to a twofold escalation in fishing effort since 1979 (NMFS 1993) and the increase in minimum legal size in 1988.

In our study area, more than 90% of the legal-size lobsters had recently molted from the sublegal size class. Since the legal size was increased from 81.0 mm in 1988, through 81.8 mm in 1989, to 82.6 mm at present, the  $\Delta$ -mean CPUE for lobsters  $\geq 81.0$  mm (the old legal size) increased each year from 1989 (0.112) to 1992 (0.208). The 1993 CPUE of lobsters  $\geq 81.0$  mm (0.197) was the second highest reported in this study. The greater CPUE of lobsters between 81.0 and 82.6 mm indicates that the regulation increasing the legal size has been effective. The increases in total

TABLE 2. Catch statistics for lobsters caught in wire<sup>a</sup> pots from 1978 to 1993.

	Total number lobster caught	Number pots hailed	Geometric mean total CPUE	95% C.I.	Delta mean legal CPUE <sup>b</sup>			95% C.I. ≥ 81.0 mm
					≥ 81.0	≥ 81.8	≥ 82.6	
1978	1824	1026	1.600	1.454 - 1.761	<u>0.173</u>	0.118	0.096	0.144 - 0.202
1979	3259	2051	1.404	1.302 - 1.513	<u>0.128</u>	0.101	0.079	0.107 - 0.148
1980	2856	2116	1.103	0.997 - 1.221	<u>0.109</u>	0.076	0.063	0.092 - 0.126
1981	2236	2187	0.904	0.839 - 0.974	<u>0.098</u>	0.079	0.069	0.083 - 0.113
1982	9109	4340	2.006	1.925 - 2.089	<u>0.165</u>	0.126	0.106	0.144 - 0.186
1983	6376	4285	1.331	1.250 - 1.418	<u>0.148</u>	0.109	0.093	0.128 - 0.168
1984	7587	4550	1.607	1.540 - 1.677	<u>0.159</u>	0.120	0.104	0.140 - 0.179
1985	7014	4467	1.352	1.252 - 1.460	<u>0.105</u>	0.080	0.068	0.090 - 0.120
1986	7211	4243	1.585	1.501 - 1.673	<u>0.086</u>	0.060	0.049	0.074 - 0.097
1987	7280	4233	1.633	1.562 - 1.707	<u>0.079</u>	0.054	0.046	0.070 - 0.089
1988	8871	4367	1.929	1.846 - 2.015	<u>0.079</u>	0.052	0.047	0.068 - 0.091
1989	7950	4314	1.729	1.645 - 1.817	0.112	<u>0.065</u>	0.053	0.097 - 0.126
1990	7106	4350	1.531	1.455 - 1.610	0.161	0.102	<u>0.076</u>	0.143 - 0.179
1991	7597	4404	1.542	1.437 - 1.654	0.183	0.117	<u>0.091</u>	0.159 - 0.206
1992	11438	4427	2.457	2.352 - 2.565	0.208	0.114	<u>0.085</u>	0.186 - 0.229
1993	10195	4194	2.301	2.198 - 2.408	0.197	0.111	<u>0.080</u>	0.175 - 0.219
<hr/>								
2-Unit 78-85	40261	2502	1.364	1.337 - 1.403	0.134	0.100	0.085	0.127 - 0.141
3-Unit 86-93	67648	345	1.810	1.775 - 1.846	0.138	0.084	0.066	0.131 - 0.144

<sup>a</sup> 10 wire pots used at each station from August through October 1978, and from May through October 1979-81; 20 wire pots used at each station from May through October 1982-93.

<sup>b</sup> The minimum legal-size from 1978 to 1988 was 81.0 mm ( $3\frac{1}{16}$  in); minimum legal-size was increased in 1989 to 81.8 mm ( $3\frac{3}{32}$  in), and in 1990 to 82.6 mm ( $3\frac{1}{4}$  in).

CPUE during the past two years, coupled with the greater number of lobsters between 81.0 and 82.6 mm observed in recent years, may lead to higher legal catches in the near future as pre-recruit lobsters (one or two molts from legal size) molt into the legal size class. However, continued increases in fishing effort may preclude any increases in legal CPUE. A strong recruit class was observed in 1982 and the percentage of legal caught in 1983 increased by almost 3%. Record landings were reported for LIS in 1983 and 1984. Another strong recruit class was observed during 1988 and although the minimum legal size was increased from 1988 to 1989, higher landings were reported in LIS in 1989 (2.7 million pounds) compared to 1988 (2.4 million pounds).

Total and legal CPUE at each station are presented in Figure 2. Total CPUE of 2.957 lobsters/pot at Twotree during 1993 was the highest reported in this study (previous range 0.988-2.941; Fig. 2). At the nearshore Jordan Cove and Intake stations, the total CPUEs during 1993 of 2.306 and 1.786, respectively, were within the

range of CPUEs reported since 1978 (Jordan Cove 0.753-2.642 and Intake 0.920-1.908). A significant increasing trend (from 1978 to 1993) in total CPUE was identified at Twotree (slope=0.101,  $p=0.003$ ). No trends in total CPUE were evident at Jordan Cove or Intake. The pattern of legal CPUE among stations during 1993 was similar to that of total CPUE. Legal CPUE (lobsters ≥ 82.6 mm) was highest at Twotree (0.107) and lower at the nearshore Jordan Cove (0.065) and Intake (0.069) stations.

Seasonal patterns of lobster abundance have been observed since 1978; in general, lobster catches (total and legal) were highest in June or July, and lowest in October. During 2-unit studies (1978-85), overall total CPUE was highest in July, whereas peak catches occurred earlier (June) during 3-unit studies (1986-93; Fig. 3). Similarly, the abundance of legal-size lobsters peaked in June during 3-unit studies, which was earlier than during 2-unit studies (July). Catchability of lobsters is directly influenced by water temperature; when water temperature rises above 10°C, lobster activity

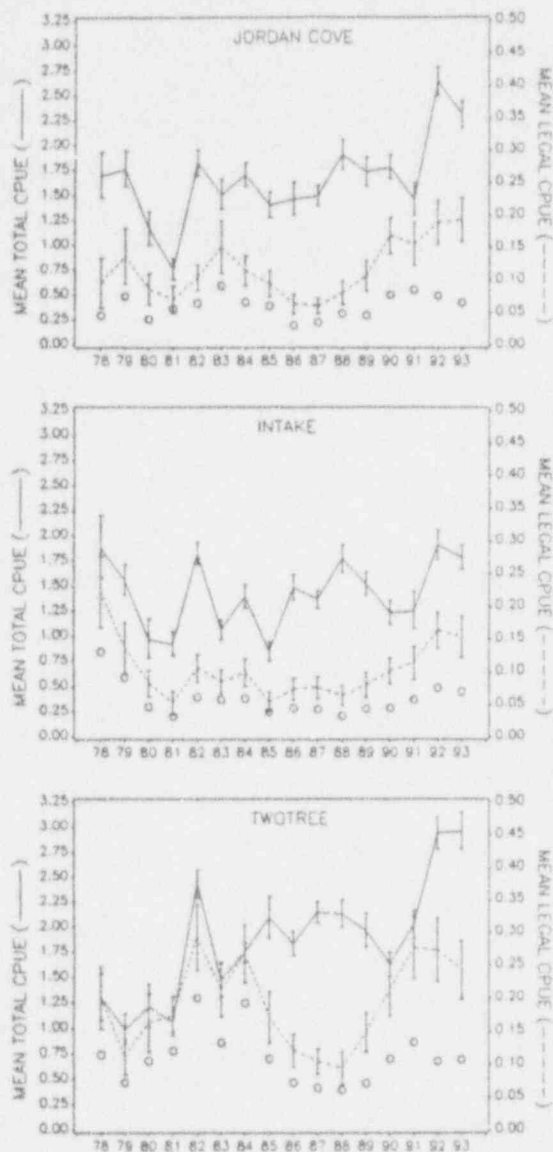


Fig. 2. Mean total CPUE (geometric mean  $\pm$  95% C.I.) and mean legal CPUE ( $\Delta$ -mean  $\pm$  95% C.I.) for lobsters  $\geq$  81.0 mm caught at each station from 1973 to 1993 ( $O = \Delta$ -mean legal CPUE for lobsters  $\geq$  82.6 mm).

(e.g., feeding, movement, and molting) increases (McLeese and Wilder 1958; Dow 1966, 1969, 1976; Flowers and Salla 1972; NUSCO 1993). Accordingly, warmer sea temperatures during the 3-unit study period (1986-93), relative to the 2-unit studies (1978-85; Table 1), may be responsible for the earlier peaks in lobster abundance.

The inherent variability in lobster catch associated with the use of passive gear (lobster traps) was examined in our study. In addition to

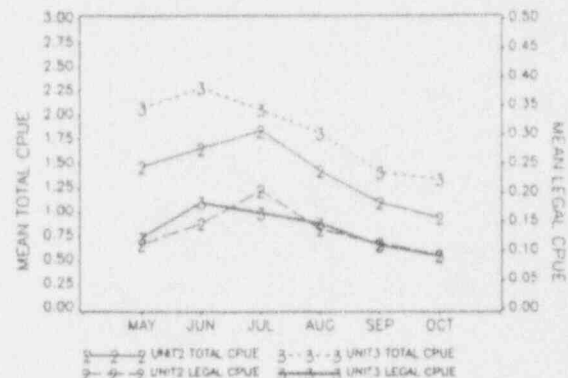


Fig. 3. Monthly mean total CPUE and mean legal CPUE for lobsters  $\geq$  81.0 mm caught during 2-unit studies (1978-1985) and 3-unit studies (1986-1993).

the significant ( $p < 0.05$ ) variation in lobster catch among the 20 traps used at each station, the incidental catch of other species in the traps was shown to influence lobster catches in previous studies (NUSCO 1987a, 1993). During 1993, lobster CPUE was influenced by catches of spider and hermit crabs at Intake, and of whelks and spider crabs at Twotree (Table 3). No species affected lobster catch at Jordan Cove during 1993. Spider crab catches at Intake continued to be high during 1993 ( $n=9,253$ ) and have influenced lobster catches in all but one year since 1984. Additionally, catches of whelks have had a significant influence on lobster CPUE in seven previous study years (Table 3). Incidental catches of other species influence lobster catches by occupying space in the trap, consuming the bait and blocking the entrance to the trap. Throughout the North Atlantic Ocean, researchers have demonstrated competition, niche segregation and interactions between lobsters and crabs in field and laboratory studies (Richards et al. 1983; Cobb et al. 1986; Richards and Cobb 1987; Hudon and Lamarche 1989; Miller 1989).

## Population Characteristics

### Size Frequency

The mean carapace length (CL) of 70.8 mm during 1993 was larger than the range of previous 3-unit CLs (69.5-70.2 mm) but within the range of CL means reported in 2-unit studies (range of 70.7-71.8 mm; Table 4). Legal-size lobsters ( $\geq$  82.6

TABLE 3. Total number of lobsters and incidental catch of other species caught in traps from 1984-1993.

	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993
Lobster	7587	7014	7211	7280	8871	7930	7106	7597	11438	10195
Rock, Jonah crab	465	177	158	108	79	583*	843*	1063*	2033*	1130
Spider crab	3237*	1950	1344*	1754*	7238*	6938*	11228*	9716*	13086*	10722*
Hermit crab	428	496*	435	721*	711	590	470	324	286	403*
Blue crab	40	21	26	44	71	43	63	148	110	70
Winter flounder	45	40*	19*	30	28	8	13	22	10	11
Summer flounder	60	24*	38*	35	28	4	16	16	14	11
Skates	15	17	33	14	16	54	40	47	53	47
Oyster toadfish	76	67	58	14	33	10	16	12	10	8
Scup	27	90	288	169	97	84	237	176	185	95
Cunner	141	207	206	167	181	67	71	76	152	75
Tautog	39	250	196	208	44	83	50	82	69	119
Sea raven	20	19	6	2	0	2	4	1	0	0
Whelks	66	78*	164*	132*	27	84*	44*	56*	178*	67*

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

TABLE 4. Summary of lobster carapace length statistics for wire pot catches from May through October, 1978-1993.

	N <sup>a</sup>	Carapace length (mm)		Percentage of legal sizes <sup>b</sup>		
		Range	Mean $\pm$ 95% CI	$\geq 81.0$	$\geq 81.8$	$\geq 82.6$
1978	1508	53-111	71.4 $\pm$ 0.33	<u>7.5</u>	5.9	4.8
1979	2846	44-100	71.2 $\pm$ 0.26	<u>7.6</u>	6.6	5.1
1980	2531	40-96	70.7 $\pm$ 0.27	<u>6.4</u>	5.0	4.1
1981	1983	43-96	71.0 $\pm$ 0.33	<u>8.8</u>	7.6	6.6
1982	7835	45-103	70.8 $\pm$ 0.15	<u>6.7</u>	5.7	4.7
1983	5432	40-121	71.7 $\pm$ 0.19	<u>9.1</u>	7.4	6.3
1984	6156	45-107	71.8 $\pm$ 0.18	<u>8.7</u>	7.3	6.4
1985	5723	38-101	71.3 $\pm$ 0.17	<u>5.9</u>	5.1	4.3
1986	5961	36-107	70.1 $\pm$ 0.17	<u>4.4</u>	3.6	3.0
1987	5924	36-99	70.2 $\pm$ 0.17	<u>3.9</u>	3.2	2.7
1988	7145	21-97	69.5 $\pm$ 0.16	<u>3.2</u>	2.6	2.3
1989	6715	34-107	69.9 $\pm$ 0.17	4.5	<u>3.5</u>	2.9
1990	6040	36-102	70.2 $\pm$ 0.20	7.9	5.9	<u>4.5</u>
1991	6449	31-101	70.2 $\pm$ 0.20	8.5	6.5	<u>5.0</u>
1992	9594	20-103	70.1 $\pm$ 0.15	6.4	4.3	<u>3.3</u>
1993	8487	30-102	70.8 $\pm$ 0.15	6.7	4.6	<u>3.3</u>
2-Unit 78-85	34014	38-121	71.3 $\pm$ 0.07	7.5	6.3	5.3
3-Unit 86-92	56315	20-107	70.1 $\pm$ 0.06	5.7	4.3	3.3

<sup>a</sup> Recaptures not included.

<sup>b</sup> The minimum legal size from 1978 to 1988 was 81.0 mm ( $3 \frac{1}{16}$  in), minimum legal size was increased in 1989 to 81.8 mm ( $3 \frac{7}{32}$  in), and in 1990 to 82.6 mm ( $3 \frac{1}{4}$  in).

mm) comprised 3.3% of the total catch during 1993, which was within the range of catch percentages reported from 1986 to 1988 (3.2-4.4%) when the legal size was 81.0 mm and from 1990 to 1992 (3.3-5.0%) when the legal size was 82.6 mm.

However, the 1993 percentage was lower than the 3.5% reported in 1989 when the legal size was 81.8 mm. Both the percentage of lobsters in each legal size category (i.e., 81.0, 81.8, 82.6 mm) and mean carapace length were lower during 3-unit studies

TABLE 5. Summary of lobster carapace length statistics for wire pot catches from May through October, 1978-1993.

JORDAN COVE	N <sup>a</sup>	Carapace length (mm) Range	Mean $\pm$ 95% CI	Percentage of legal sizes <sup>b</sup>		
				$\geq 81.0$	$\geq 81.8$	$\geq 82.6$
1978	489	54-111	70.3 $\pm$ 0.54	<u>3.9</u>	3.5	2.7
1979	1138	46-96	70.7 $\pm$ 0.39	<u>6.7</u>	5.7	4.2
1980	831	40-93	70.2 $\pm$ 0.45	<u>4.8</u>	3.5	2.5
1981	556	45-93	70.6 $\pm$ 0.64	<u>7.2</u>	6.7	5.9
1982	2323	49-96	69.8 $\pm$ 0.26	<u>4.7</u>	4.0	3.2
1983	1965	40-100	71.0 $\pm$ 0.32	<u>8.2</u>	6.5	5.5
1984	1999	52-107	70.7 $\pm$ 0.29	<u>5.7</u>	4.4	4.0
1985	1722	48-96	71.1 $\pm$ 0.32	<u>5.9</u>	5.0	4.1
1986	1748	38-99	69.8 $\pm$ 0.31	<u>3.5</u>	2.6	2.1
1987	1690	44-95	70.2 $\pm$ 0.32	<u>3.7</u>	3.0	2.5
1988	2239	21-97	69.2 $\pm$ 0.29	<u>3.3</u>	2.7	2.3
1989	2077	36-98	69.8 $\pm$ 0.30	4.0	<u>3.0</u>	2.6
1990	2108	36-94	69.4 $\pm$ 0.33	7.0	4.9	<u>4.0</u>
1991	1834	38-101	69.3 $\pm$ 0.39	7.6	6.1	<u>4.8</u>
1992	3125	20-103	69.0 $\pm$ 0.27	5.5	3.7	<u>2.8</u>
1993	2627	44-102	70.0 $\pm$ 0.28	6.5	4.3	2.7
2-Unit 78-85	11023	40-111	70.6 $\pm$ 0.13	6.0	4.9	4.1
3-Unit 86-93	17448	20-103	69.5 $\pm$ 0.11	5.2	3.8	2.9
INTAKE						
1978	645	55-110	71.8 $\pm$ 0.50	<u>9.2</u>	6.7	5.7
1979	1087	50-100	71.4 $\pm$ 0.41	<u>7.6</u>	6.6	5.4
1980	855	46-95	70.6 $\pm$ 0.45	<u>5.6</u>	4.3	3.6
1981	686	43-95	69.2 $\pm$ 0.53	<u>4.4</u>	3.4	2.9
1982	2402	51-103	70.2 $\pm$ 0.27	<u>5.0</u>	4.1	3.2
1983	1436	52-110	71.2 $\pm$ 0.37	<u>6.5</u>	5.5	4.9
1984	1830	45-105	70.5 $\pm$ 0.32	<u>6.1</u>	4.9	4.2
1985	1215	44-99	71.2 $\pm$ 0.37	<u>5.6</u>	4.9	4.3
1986	1888	50-107	69.3 $\pm$ 0.31	<u>4.3</u>	3.6	3.0
1987	1687	47-94	70.2 $\pm$ 0.32	<u>4.7</u>	3.5	3.0
1988	2253	39-95	69.2 $\pm$ 0.27	<u>2.9</u>	2.0	1.8
1989	2005	39-98	69.0 $\pm$ 0.32	4.0	<u>3.0</u>	2.4
1990	1721	36-102	69.4 $\pm$ 0.36	6.1	4.4	<u>3.1</u>
1991	1877	31-100	68.9 $\pm$ 0.35	6.5	5.1	<u>3.6</u>
1992	2575	22-97	69.5 $\pm$ 0.29	6.4	4.6	<u>3.5</u>
1993	2275	47-95	70.1 $\pm$ 0.30	6.7	4.9	3.6
2-Unit 78-85	10156	43-110	70.7 $\pm$ 0.13	6.0	4.9	4.2
3-Unit 86-93	16281	22-107	69.4 $\pm$ 0.11	5.2	3.9	3.0
TWO TREE						
1978	374	53-94	72.2 $\pm$ 0.67	<u>9.4</u>	7.8	5.9
1979	621	44-94	71.8 $\pm$ 0.58	<u>9.0</u>	8.1	6.0
1980	845	40-96	71.3 $\pm$ 0.49	<u>8.8</u>	7.1	6.0
1981	741	48-96	73.0 $\pm$ 0.54	<u>14.2</u>	12.2	10.4
1982	3110	45-102	72.0 $\pm$ 0.25	<u>9.4</u>	8.1	7.0
1983	2031	43-121	72.8 $\pm$ 0.32	<u>11.8</u>	9.7	8.1
1984	2327	50-105	73.7 $\pm$ 0.29	<u>13.4</u>	11.7	10.3
1985	2786	38-101	71.5 $\pm$ 0.25	<u>6.1</u>	5.2	4.4
1986	2325	36-97	71.0 $\pm$ 0.27	<u>5.3</u>	4.4	3.6
1987	2547	36-99	70.2 $\pm$ 0.27	<u>3.6</u>	3.1	2.6
1988	2653	36-95	70.0 $\pm$ 0.27	<u>3.5</u>	3.0	2.8
1989	2633	34-107	70.6 $\pm$ 0.28	5.4	<u>4.2</u>	3.4
1990	2211	39-102	71.7 $\pm$ 0.33	10.0	7.9	<u>6.1</u>
1991	2738	38-98	71.6 $\pm$ 0.30	10.5	7.7	<u>6.2</u>
1992	3894	32-96	71.3 $\pm$ 0.23	7.1	4.5	<u>3.5</u>
1993	3585	30-92	71.9 $\pm$ 0.23	6.7	4.6	3.6
2-Unit 78-85	12835	38-121	72.3 $\pm$ 0.12	10.0	8.5	7.3
3-Unit 86-93	22586	30-107	71.1 $\pm$ 0.10	6.5	4.9	3.9

<sup>a</sup> Recaptures not included. <sup>b</sup> The minimum legal size from 1978 to 1988 was 81.0 mm (3 1/16 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).



(5.6%, 4.2%, 3.4% and 70.0 mm, respectively), than during 2-unit studies (7.5%, 6.3%, 5.3%, and 71.3 mm). The percentages of legal lobsters in our catch have significantly declined since 1978 (slope = -0.328,  $p < 0.01$ ), probably due to increased fishing effort, which has more than doubled since 1978 (Blake 1991; NMFS 1993).

Carapace length statistics of lobsters caught at each station (Table 5) followed trends similar to those observed for the total catch. Largest lobsters were caught at Twotree (mean CL range 70.0-73.7 mm) and the mean size ranges of lobsters caught at the nearshore Jordan Cove and Intake stations were 69.2-71.1 mm and 68.9-71.8 mm, respectively. The mean CL at Twotree during 1993 (71.9 mm) was the largest observed in 3-unit studies (70.0-71.7 mm), while those at Jordan Cove and Intake were within the range of previous 3-unit studies (Table 5). Mean lobster size at all stations was larger during 2-unit studies than during 3-unit studies (Jordan Cove 70.6 vs. 69.5, Intake 70.7 vs. 69.4, Twotree 72.3 vs. 71.1 mm). The percentage of legal lobsters was also greater during 2-unit studies; at both Jordan Cove and Intake, 6.0% of the catch was legal size ( $\geq 81.0$  mm) during 2-unit studies and 5.2% during 3-unit studies (Table 5). The Twotree station consistently yielded more legal-size lobsters than did nearshore sites; 10.0% of the catch was legal size during 2-unit studies and 6.5% during 3-unit studies. Since 1978, legal catches at each station have significantly declined and lobstermen in our area have reported a 15-20% decline in their legal catches over the past few years. Larger declines (50%) were reported from lobstermen west of the Connecticut River. In 1992, substantial declines in landings were observed throughout the range of lobsters (U.S. -13% and Canada -20%; NMFS 1993).

### Sex Ratios

The sex ratio of lobsters collected during 1993 was 0.84 females per male, compared to a range of 0.71-0.88 in prior years of 3-unit operation and 0.79-0.97 during 2-unit operation (Table 6). Sex ratios during 1993 at the two nearshore sites, Jordan Cove (0.47) and Intake (0.59), were within the range of previous 3-unit studies (0.43-0.71 and 0.47-0.73, respectively), but below the range of 2-unit studies (0.60-0.79 and 0.66-0.97, respectively). Female to male sex ratios at Twotree have

TABLE 6. Female to male sex ratios<sup>a</sup> of lobsters caught in wire pots from May through October, 1978-1993.

	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
2-Unit 78-85	0.67	0.72	1.21	0.86
3-Unit 86-93	0.56	0.62	1.24	0.80

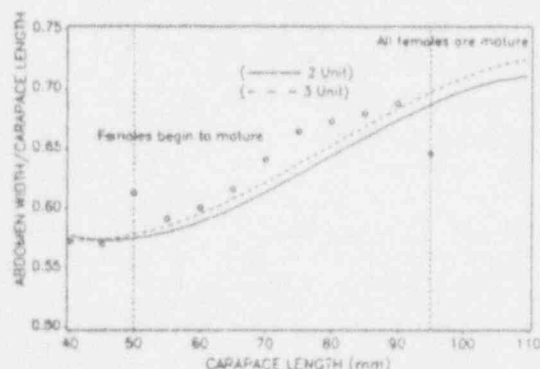
<sup>a</sup> Recaptures not included.

increased each year over the past four years, with the 1993 sex ratio of 1.59 the highest reported in this study (previous range 0.90-1.45). The overall ratio of females to males was higher during 2-unit studies (0.86 female per male) than during 3-unit studies (0.80 females per male). At Jordan Cove and Intake, sex ratios during 2-unit studies were 0.67 and 0.72 females per male, higher than the ratios during 3-unit studies (0.56 and 0.62, respectively), but at Twotree, the female-to-male ratio was lower during 2-unit (1.21) than during 3-unit studies (1.24). The occurrence of more females than males at Twotree has remained consistent since 1975 and different from sex ratios at other stations (Keser et al. 1983). Female-to-male ratios of lobsters caught by commercial lobstermen in LIS ranged between 1.06 and 1.81 (Smith 1977) and more recently, Blake (1988) reported higher sex ratios of lobsters caught in the eastern LIS commercial fishery (range 2.61-6.29). Several factors have been suggested to cause a predominance of females in the commercial fishery: differences in female lobster behavior related to molting and reproduction, fishery regulations designed to protect egg bearing females, and the fact that mature females molt less

frequently than males (Ennis 1980). The overall sex ratio of lobsters in the MNPS area is close to the 1:1 sex ratio reported by other researchers for predominantly sublegal (< 81.0 mm CL) populations of lobsters (Herrick 1911; Templeman 1936; Ennis 1971, 1974; Stewart 1972; Krouse 1973; Thomas 1973; Cooper et al. 1975; Briggs and Mushacke 1980).

### Reproduction

Several methods have been used to determine the size at which females become sexually mature, with the presence of external eggs the obvious indication of female lobster maturity. Additionally, the method first described by Templeman (1935) is that female abdominal width markedly increases during maturation. Calculation of the abdominal width to carapace length ratio and comparison to CL provides an index of female size at sexual maturity (Skud and Perkins 1969; Krouse 1973). Mean ratios of abdominal width to carapace length were calculated for each 5 mm CL and plotted against the carapace length of lobsters collected for 2-unit (1981-85) and 3-unit (1986-93) operations and for 1993 alone (Fig. 4). During 1993, females began to mature at about 55 mm CL, and all females were mature above 90 mm CL. The close correspondence between the 2- and 3-unit curves in Figure 4 indicates that female size at sexual maturity was similar during both operational periods. The size of the smallest berried females



$$\begin{aligned} \text{2-unit: } y &= 1.28 - (3.13 \times 10^{-3})x + (4.40 \times 10^{-6})x^2 - (1.85 \times 10^{-6})x^3, r^2 = .30 \\ \text{3-unit: } y &= 1.03 - (2.32 \times 10^{-2})x + (3.61 \times 10^{-5})x^2 - (1.61 \times 10^{-6})x^3, r^2 = .32 \\ \text{1993: } y &= 1.34 - (4.09 \times 10^{-3})x + (6.85 \times 10^{-6})x^2 - (3.45 \times 10^{-6})x^3, r^2 = .31 \end{aligned}$$

Fig. 4. Morphometric relationship between the abdominal width to carapace length ratio ( $y$ ) and the carapace length ( $x$ ) for female lobsters during 2-unit (—) and 3-unit studies (---) and during 1993 ( $\circ \circ \circ$ ).

collected in 2-unit (62 mm) and 3-unit studies (60 mm) supported the results of the morphometric relationship between the abdominal width and carapace length. These individuals were between 50-55 mm CL when oviposition first occurred (assuming 14% growth per molt). Briggs and Mushacke (1979), using the same morphometric technique, found that females in western LIS begin to mature at 60 mm CL and most are mature at about 80 mm CL. In contrast, Gulf of Maine females seldom become sexually mature at less than 81 mm CL, and only a small percentage are mature between 81 and 90 mm CL (Krouse 1973; Krouse et al. 1993). Earlier maturation of females in LIS than in the Gulf of Maine can be attributed to the warmer LIS water temperatures (Smith 1977; Aiken and Waddy 1980). The sexual maturity of males was not investigated in our study because other researchers documented that the size at which males become mature varies only slightly throughout the range of lobsters. In western LIS, males are mature (i.e., produce mature spermatozoa) at 40 to 44 mm CL, and over half are mature at 50 to 54 mm CL (Briggs and Mushacke 1979); in northern waters (Maine), males also begin to mature at relatively small sizes (50% mature at 44 mm CL; Krouse 1973).

The percentage of berried females collected during 1993 was 12.2%, which was the highest value reported since the study began in 1978 (previous range 3.1-12.1%; Table 7). More berried females were caught at Twotree (19.4%) than at the nearshore Jordan Cove (3.1%) or Intake (2.7%) stations; this distribution pattern has been consistent since 1975 (Keser et al. 1983). The percentage at Twotree was the highest since the study began (previous range 5.3-19.3%), while the percentages at Jordan Cove and Intake were within the range of previous 3-unit (2.4-3.4% and 1.5-4.0%, respectively) and 2-unit percentages (0.8-3.6% and 0.9-4.5%, respectively). The percentage of berried females was higher during 3-unit (7.7%) than during 2-unit operations (4.3%). The mean carapace length of 75.6 mm for berried females collected during 1993 was within the range of average sizes reported in previous 3-unit studies (75.3-78.1 mm), but below the range reported in 2-unit studies (77.0-81.2 mm; Table 7). On average, berried females collected during 3-unit studies were smaller (76.5 mm) than during 2-unit studies (79.4 mm), due to the larger proportion of sublegal-sized berried females collected since 1986.

TABLE 7. Percentage of berried females caught at each station and annual carapace length statistics from 1978-93.

	Percentage of berried females				N <sup>a</sup>	Range	Carapace Length (mm) Mean $\pm$ 95% C.I.	Percentage of sublegal sizes <sup>b</sup>		
	All stations	Jordan Cove	Intake	Twotree				<81.0	<81.8	<82.6
1978	3.4	1.4	2.6	5.3	58	74 - 88	80.1 $\pm$ 1.04	<u>73</u>	78	78
1979	3.1	1.9	2.7	7.2	70	64 - 93	80.5 $\pm$ 1.28	<u>59</u>	64	70
1980	3.3	3.5	1.8	5.6	71	66 - 93	79.1 $\pm$ 1.27	<u>70</u>	73	79
1981	4.2	1.6	2.7	7.1	82	69 - 97	81.2 $\pm$ 1.35	<u>55</u>	59	62
1982	3.1	0.8	0.9	6.1	108	64 - 99	80.0 $\pm$ 1.08	<u>60</u>	66	70
1983	4.7	2.1	3.2	8.5	123	66 - 103	80.5 $\pm$ 1.04	<u>63</u>	65	67
1984	6.2	3.6	3.5	10.6	173	62 - 95	79.1 $\pm$ 0.87	<u>69</u>	75	76
1985	6.2	3.5	4.5	8.5	171	63 - 94	77.0 $\pm$ 0.81	<u>82</u>	85	86
1986	4.8	3.0	2.3	8.0	135	65 - 94	78.0 $\pm$ 0.95	<u>77</u>	80	83
1987	5.7	3.2	1.9	9.6	158	62 - 90	76.5 $\pm$ 0.67	<u>92</u>	92	93
1988	3.8	2.4	1.9	6.4	124	63 - 90	76.9 $\pm$ 0.82	<u>89</u>	90	90
1989	5.4	2.8	3.3	8.2	161	65 - 98	77.3 $\pm$ 0.78	<u>82</u>	<u>85</u>	88
1990	6.6	2.7	4.0	11.2	165	65 - 102	78.1 $\pm$ 0.82	75	81	<u>87</u>
1991	8.2	3.2	1.5	13.5	226	62 - 96	78.0 $\pm$ 0.75	71	78	<u>82</u>
1992	12.1	3.4	1.7	19.3	491	60 - 93	75.3 $\pm$ 0.44	89	92	<u>94</u>
1993	12.2	3.1	2.7	19.4	476	62 - 90	75.6 $\pm$ 0.43	88	91	<u>93</u>
2-Unit 78-85	4.3	2.0	2.2	7.1	856	62 - 103	79.4 $\pm$ 0.39	68	72	74
3-Unit 86-93	7.7	2.9	2.4	12.8	1936	60 - 102	76.5 $\pm$ 0.23	84	88	90

<sup>a</sup> Recaptures not included.<sup>b</sup> The minimum legal size from 1978 to 1988 was 81.0 mm ( $3\frac{3}{16}$  in), minimum legal size was increased in 1989 to 81.8 mm ( $3\frac{7}{32}$  in), and in 1990 to 82.6 mm ( $3\frac{1}{4}$  in).

During 1993, 93% of the berried females were below the minimum legal size of 82.6 mm. High rates of fishing remove most females shortly after they reach legal size or after berried females release eggs. The apparent stability of the LIS lobster population, despite current high exploitation rates, may be due to the fact that females become mature and bear eggs at sizes well below the legal size. While undocumented, stock mean fecundity may be lower, as a result of relying on smaller berried females to sustain recruitment, which could affect the long-term health of the LIS fishery. The regulation to increase the minimum legal size appears to be effective as the percentage of berried females in our catch has increased each year. The increase in minimum legal size should improve egg production and subsequent recruitment by allowing more females to spawn before reaching legal size. However, if fishing effort continues to escalate, higher yields (CPUE) due to increased egg production may not materialize.

### Molting and Growth

The most important factor regulating molting and growth of lobsters is temperature (Aiken 1980), resulting in variable seasonal and annual catches of soft or newly molted lobsters. During 1993, the majority of molting lobsters were caught from late spring (end of May) to early summer (middle of June). However, in several of the previous study years, a second peak in the catch of molting lobsters was observed in autumn (Keser et al. 1983). The frequency and timing of lobster molts were examined using the Gompertz growth function fitted to cumulative percent-molt data for 2- and 3-unit studies (Fig. 5). The inflection points of the growth curves were used to estimate annual dates of peak molting. Annual molting peaks were significantly ( $p < 0.05$ ) correlated with mean May bottom water temperatures and indicated that molting occurred earlier when May water temperatures were warmer than average. Conversely, peaks occurred later when water temperatures were colder than average. Peak molt during 2-unit studies occurred on 27 June, which

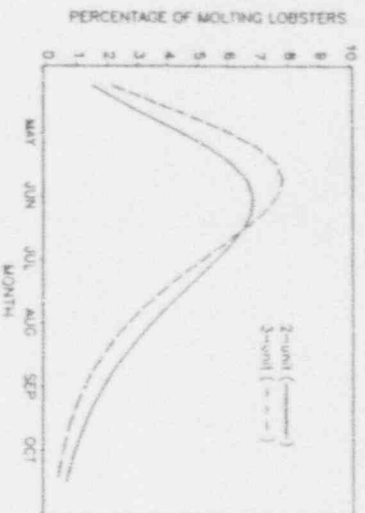


Fig. 5. Molt frequency curves estimated from the Gompertz function for lobsters caught during 2-unit studies (1978-85; —) and 3-unit studies (1986-93; - - -).

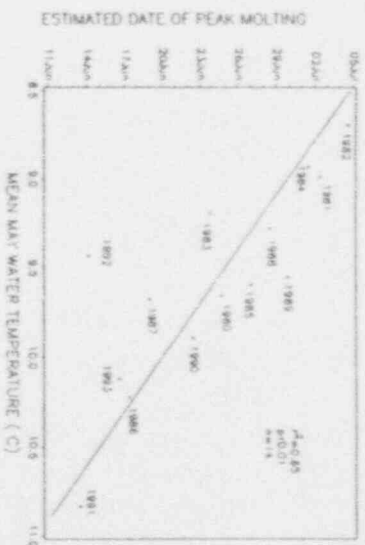
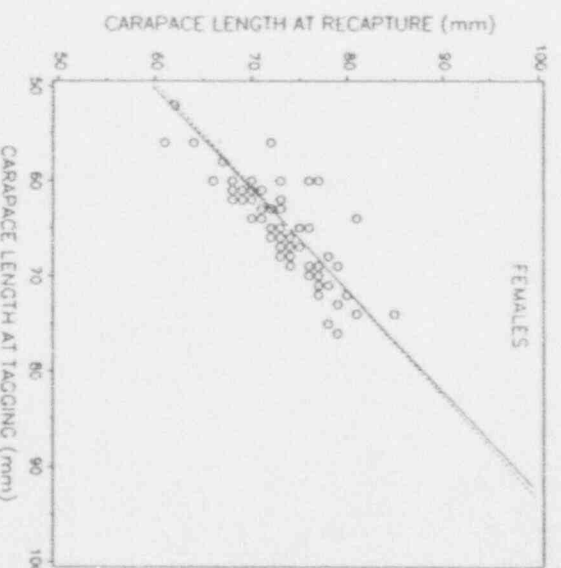
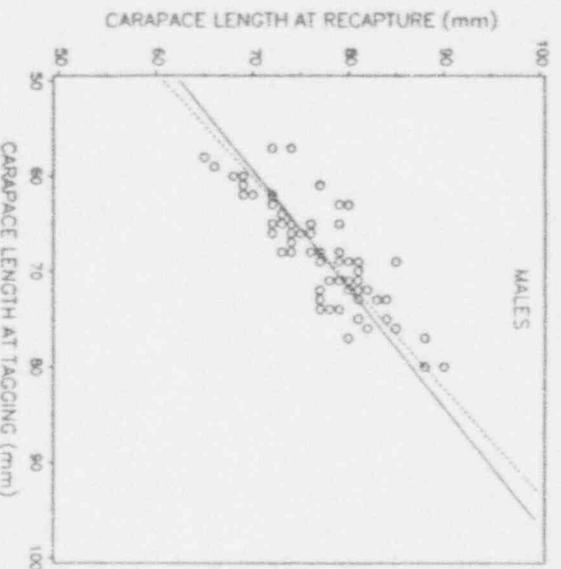


Fig. 6. Relationship between the date of peak molting (1 from the Gompertz function) and annual mean bottom water temperature during May.



MALES				FEMALES			
Growth			R <sup>2</sup>	Growth			R <sup>2</sup>
N				N			
2-Unit	379	y = 22.168 + 0.805(x)	0.70	2-Unit	586	y = 12.678 + 0.942(x)	0.79
3-Unit	627	y = 15.422 + 0.907(x)	0.77	3-Unit	833	y = 14.577 + 0.914(x)	0.73
1993	99	y = 20.457 + 0.829(x)	0.73	1993	78	y = 24.003 + 0.756(x)	0.70

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

Fig. 7. Linear regressions and parameter estimates for carapace lengths at tagging and recapture times for male and female lobsters caught during 2-unit studies (1978-85; —), 3-unit studies (1986-92; - - -), and during 1993 (O).

was nine days later than the peak molt observed during 3-unit studies (18 June; Fig. 5); the corresponding average bottom water temperatures during May of the two study periods were 9.2°C and 9.9°C, respectively. During 1993, the peak molt occurred on 15 June when May water

temperatures averaged 10.1°C. The earliest molting peak occurred on 12 June 1991 and the latest on 3 July 1982; the average bottom water temperatures during May in those years were 10.8°C and 8.7°C, respectively (Fig. 6). Templeman (1936) correlated the timing of molts

TABLE 8. Summary of lobster growth (in mm and as a percentage) at each station for wire pot catches from May through October 1979 to 1993.

	<u>Jordan Cove</u>				<u>Intake</u>				<u>Twotree</u>		
	N	Growth (mm)	Percentage	N	Growth (mm)	Percentage		N	Growth (mm)	Percentage	
1979	33	7.3	10.6	22	8.8	12.8		21	11.1	16.3	
1980	38	8.6	12.7	21	8.7	12.5		33	10.1	14.8	
1981	29	7.9	11.8	24	8.9	13.1		40	10.3	15.4	
1982	48	9.0	13.3	55	7.8	12.0		96	9.1	13.2	
1983	40	8.8	13.4	23	9.5	14.2		71	9.6	14.5	
1984	85	9.0	13.8	44	7.8	12.3		79	8.8	12.9	
1985	63	8.4	12.8	25	8.8	13.7		77	8.0	11.7	
1986	61	9.1	13.5	39	7.5	11.6		69	8.6	12.9	
1987	71	7.9	12.0	41	8.6	12.8		67	8.9	13.2	
1988	93	8.5	12.8	58	9.5	15.2		104	9.6	14.7	
1989	82	9.3	14.3	72	9.5	14.4		80	9.2	14.1	
1990	93	9.1	13.9	51	9.2	14.2		58	10.2	15.5	
1991	57	8.4	12.6	65	8.9	13.4		65	9.8	14.7	
1992	107	8.9	13.8	48	8.8	13.2		81	9.4	14.6	
1993	68	8.5	13.0	35	8.4	12.7		76	8.6	13.2	
<hr/>											
2-Unit 79-85	336	8.6	12.9	214	8.5	12.8		417	9.2	13.6	
3-Unit 86-93	632	8.7	13.6	409	8.9	13.6		600	9.3	14.1	

with summer water temperatures in the Canadian Maritimes and suggested that a 1°C drop in water temperature delays the first molting period by a week or more. The influence of varying water temperature on the molt cycle was examined by Aiken and Waddy (1980). Moreover, at 10°C lobsters quickly entered the premolt stage and progressed to ecdysis.

Lobster growth was determined from carapace length measurements of those lobsters that had molted between tagging and recapture. Incremental growth per molt for the size range of lobsters caught in our studies is best described using simple linear regressions (Wilder 1953; Kurata 1962; Mauchline 1976). Regression plots and parameter estimates of growth for males and females caught during 1993 and in 2-unit and 3-unit studies are presented in Figure 7. Growth increments averaged 8.9 mm (13.3%) and 8.7 mm (13.0%) for males and females, respectively during 2-unit studies, which was smaller than the average incremental growth during 3-unit studies (males 9.1 mm, 13.7%; females 8.9 mm, 13.7%). The fact that females exhibited slightly lower growth was expected because of their reproductive cycle; energy that could be used for carapace growth is

diverted to widening of the abdomen and development of ovaries. Growth increments at each station were slightly larger during 3-unit studies than during 2-unit studies (Table 8). Growth per molt was higher at Twotree during 2-unit (9.2 mm, 13.6%) and 3-unit studies (9.3 mm, 14.1%), than at the nearshore stations, Jordan Cove (2-unit 8.6 mm, 12.9%; 3-unit 8.7 mm, 13.6%) and Intake (2-unit 8.5 mm, 12.8%; 3-unit 8.9 mm, 13.6%; Table 8). Lobster growth determined from our tag and recapture studies is similar to that from other studies conducted in eastern LIS, where growth has averaged between 12.7% and 15.8% for males and between 12.8% and 15.4% for females (Stewart 1972; Blake 1991). In western LIS, growth per molt ranged between 13.0 and 14.5% for males and between 12.5 and 13.5% for females (Briggs and Mushacke 1984; Blake 1991). Offshore lobsters were reported to exhibit larger growth (18.7% for males and 16.7% for females) than inshore lobsters, which was attributed to seasonal migration of offshore lobsters to maintain a temperature regime of between 8° and 14°C (Cooper and Uzmann 1971; Uzmann et al. 1977).



TABLE 9. Percentage of culls (lobsters missing one or both claws) caught in wire pots from 1978-1993.

	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
<hr/>				
2-Unit 78-85	14.4	15.2	7.6	12.1
3-Unit 86-93	12.5	14.0	7.4	10.9

### Culls

The percentage of lobsters missing one or both claws (culls) was 10.1% of the total catch during 1993, which was within the range of previous 3-unit studies (10.0-12.2%), but lower than the values reported in 2-unit studies (10.8-15.5%; Table 9). During 1993, Twotree had the lowest percentage of culls (7.7%), followed by Jordan Cove (11.2%) and Intake (12.6%); this ranking of the three stations has remained unchanged since the study began (NUSCO 1987a, 1993). Claw-loss was lower during combined 3-unit studies (10.9%) than during the 2-unit study period (12.1%), likely due to the implementation of the escape vent regulation in 1984. This regulation requires that pots contain a 1 $\frac{3}{4}$  by 6 inch opening to allow escape of sublegal-sized lobsters, and thereby reduces injury and mortality associated with overcrowded pots (Landers and Blake 1985). Trap related injuries were associated with water temperature, fishing pressure (i.e., handling by lobstermen), trap soaktime, and shell hardness (Pecci et al. 1978). Of these factors, a positive correlation between fishing pressure and the incidence of culls along the coast of Maine was

demonstrated by Krouse (1976). Throughout New England, researchers have reported the benefits of incorporating escape vents in lobster traps (Krouse and Thomas 1975; Fair and Estrella 1976; Krouse 1978; Pecci et al. 1978; Fogarty and Borden 1980; Krouse et al. 1993).

### Tagging Program

The number of lobsters tagged during 1993 (8,177) was within the range of previous 3-unit years (5,680-9,126), but higher than the 2-unit study range of 2,768 to 7,575 (Table 10). The percentage recaptured in our traps was 20.9% and within the recapture rates observed in both 3-unit (18.1-25.2%) and 2-unit studies (14.4-23.9%; Table 10). Relative to recaptures in NUSCO traps, the percentage recaptured in commercial traps was smaller (14.8%) during 1993 and this value was lower than percentages reported since 1978 (previous range 17.0-47.6%). The overall percentage recaptured in NUSCO traps was 2% higher during 3-unit than during 2-unit studies, whereas the percentage of recaptures by commercial lobstermen declined from 33.0% during 2-unit studies to 18.4% during 3-unit studies. The shift in percentages recaptured in NUSCO and commercial traps during 2- and 3-unit operations appears due to the escape vent regulation implemented in 1984 and not to plant operation. Installation of escape vents, coupled with the fact that most of our tagged lobsters are sublegal, resulted in fewer tagged lobsters retained in commercial traps. Conversely, NUSCO traps do not have escape vents and retain greater numbers of tagged sublegal lobsters. The mean CL of lobsters recaptured in NUSCO and in commercial traps during 1993 were 73.4 and 79.1 mm, respectively; both values were within the range of those in previous 2-unit and 3-unit studies (Table 10). Differences between the size of lobsters recaptured in NUSCO and in commercial traps during the two operational study periods were also due to the implementation of the escape vent regulation. The overall mean CL in NUSCO traps was smaller during 3-unit (72.8 mm) than during 2-unit studies (73.9 mm) and larger in commercial traps during the two operational study periods (79.0 vs. 77.1 mm). Before escape vents were required (1978-83), commercial lobstermen recaptured many of the sublegal-sized tagged

TABLE 10. Lobster tag and recapture statistics for NUSCO pots (May-Oct.) and commercial pots (Jan.-Dec.) from 1978 to 1993.

	NUSCO					Commercial			
	Number tagged	Number recaptured	Percentage recaptured	Percentage legal <sup>a</sup>	Mean CL (mm)	Number recaptured	Percentage recaptured	Percentage legal <sup>a</sup>	Mean CL (mm)
1978	2768	498	18.0	16.7	75.5	884	31.9	43.6	81.1
1979	3732	722	19.4	11.5	75.1	1778	47.6	27.2	77.6
1980	3634	522	14.4	18.8	75.7	1363	37.5	27.5	76.4
1981	4246	707	16.7	12.0	74.8	1484	35.0	25.9	76.3
1982	7575	1282	16.9	10.4	73.2	2519	33.2	23.0	75.5
1983	5160	932	18.1	11.3	73.6	2266	43.9	27.6	76.9
1984	5992	1431	23.9	8.4	73.0	1290	21.5	34.3	78.8
1985	5609	1216	21.7	7.7	73.2	1185	21.1	29.3	78.3
1986	5740	1194	20.8	4.7	72.3	1177	20.4	27.5	78.2
1987	5680	1356	23.9	5.5	72.8	1160	20.4	25.3	78.9
1988	6837	1725	25.2	4.3	72.0	1383	20.2	26.7	78.0
1989	6438	1233	19.2	4.4 (9.3)	72.9	1183	18.4	20.7 (24.8)	78.2
1990	5741	1066	18.6	5.5 (12.7)	73.3	1007	17.5	26.5 (32.8)	79.3
1991	6136	1109	18.1	7.4 (13.9)	73.4	1228	20.0	33.9 (41.5)	80.8
1992	9126	1842	20.2	3.9 (9.3)	72.4	1552	17.0	23.3 (28.5)	79.5
1993	8177	1708	20.9	3.6 (9.8)	73.4	1213	14.8	25.4 (47.5)	79.1
2-Unit 78-85	38716	7310	18.9	11.0	73.9	12769	33.0	27.5	77.1
3-Unit 86-93	53875	11233	20.9	4.7 (8.3)	72.8	9903	18.4	26.1 (31.7)	79.0

<sup>a</sup> The minimum legal size from 1978 to 1988 was 81.0 mm ( $3\frac{3}{16}$  in), minimum legal size was increased in 1989 to 81.8 mm ( $3\frac{7}{32}$  in), and 1990 82.6 mm ( $3\frac{1}{4}$  in). Parenthetical values for percentage legal from 1989 to 1993 and 3-unit studies (1986-93) represent lobsters  $\geq$  81.0 mm carapace length.

lobsters. Since the regulation was enforced, many sublegals escaped from the vented commercial pots, but were still retained in unvented NUSCO pots. In eastern LIS, Landers and Blake (1985) noted a substantial reduction in the number of sublegal-sized lobsters retained in vented pots, without a corresponding decrease in the catch of legal-sized lobsters. In Maine waters, Krouse et al. (1993) examined lobster catches in traps equipped with a variety of escape vent sizes ( $1\frac{3}{4}$ ,  $1\frac{13}{16}$ ,  $1\frac{7}{8}$  by  $5\frac{3}{4}$  in). They found that  $1\frac{7}{8} \times 5\frac{3}{4}$  in vents retained fewer sublegals than did traps with smaller escape vents, and that the overall catch of legal was comparable for the  $1\frac{3}{4}$  and  $1\frac{7}{8}$  in vented traps.

The declining trend observed in the percentage of legal-sized lobsters in the overall lobster catches was reflected in the proportion of recaptured lobsters that were of legal size. During 1993, only 3.6% of the recaptures in NUSCO pots were of legal size ( $\geq 82.6$  mm), which was the lowest value reported (previous range 3.9-18.8%; Table 10). The overall percentage of legal-sized lobsters

recaptured in NUSCO traps during 3-unit studies (4.7%) was substantially lower than the percentage during 2-unit studies (11.0%). Declines were also noted in the percentage of legal-sized lobsters recaptured by commercial lobstermen during the two operational periods (26.1% in 3-unit vs. 27.5% in 2-unit studies). The declines in percentage of legal-sized recaptures are attributed to an increase in fishing effort, which has more than doubled since 1978 and, in part, to the increase in minimum legal size beginning in 1989.

### Movement

Lobster movement was examined using tag return information obtained from commercial lobstermen. The average distance traveled by lobsters before they were caught in commercial traps was 5.06 km during 1993, which was the longest distance reported in our tagging study (previous range 1.70-3.16 km; Table 11). During 1993, 91% of the recaptured lobsters were caught by lobstermen fishing within 5 km of MNPS.

TABLE 11. The average distance (km) travelled by lobsters from Millstone Point for all commercial pots.

	<u>All recaptures</u>		<u>Within 5 km of MNPS</u>		<u>More than 5 km from MNPS</u>	
	Number of tags returned	Average distance(km)	Number (%) of tags returned	Average distance(km)	Number (%) of tags returned	Average distance(km)
1978	798	3.01	725 (91)	1.71	73 (9)	15.92
1979	1733	1.70	1665 (96)	1.31	68 (4)	11.28
1980	1303	2.09	1257 (96)	1.25	46 (4)	25.17
1981	1478	1.89	1451 (98)	1.49	27 (2)	23.49
1982	2509	2.34	2343 (93)	1.58	166 (7)	13.04
1983	2258	2.88	2111 (93)	1.70	147 (7)	19.78
1984	1288	2.33	1230 (95)	1.78	58 (5)	13.93
1985	1183	2.84	1077 (91)	1.81	106 (9)	13.40
1986	1172	2.64	1112 (95)	1.76	60 (5)	18.82
1987	1157	2.87	1124 (97)	1.77	33 (3)	40.09
1988	1371	3.16	1286 (94)	1.81	85 (6)	23.72
1989	1165	1.97	1147 (98)	1.79	18 (2)	12.86
1990	1004	2.18	979 (98)	1.80	25 (2)	17.06
1991	1222	2.47	1181 (97)	1.80	41 (3)	21.80
1992	1536	2.68	1485 (97)	1.80	51 (+)	28.09
1993	1209	5.06	1096 (91)	1.96	113 (9)	35.15
<hr/>						
2-Unit 78-85	12550	2.36	11859 (94)	1.57	691 (6)	15.95
3-Unit 86-93	9836	2.89	9410 (96)	1.81	426 (4)	26.82

within the range of percentages reported during 2-unit studies (91-98%), but lower than during previous 3-unit studies (94-98%; Table 11). Lobsters recaptured within 5 km of Millstone Pt. moved an average of 1.96 km during 1993, longer than the distances reported previously (1.25-1.81 km; Table 11). An earlier tagging study conducted in eastern LIS by Stewart (1972) demonstrated a strong homing behavior of the nearshore lobster population. Because lobsters are territorial and nocturnal, individuals have a limited home range; they leave their burrows at night and return to the same shelters before dawn. Our tagging studies indicate a predominance of localized movement which is typical of nearshore lobsters in coastal waters of eastern North America (Templeman 1940; Wilder and Murray 1958; Willet 1963; Cooper 1970; Stewart 1972; Cooper et al. 1975; Fogarty et al. 1980; Krouse 1980, 1981; Campbell 1982; Ennis 1984).

Although our tag and recapture studies indicate that most lobsters are nonmigratory and remain in the local area, since 1978 1,117 have been recaptured more than 5 km away from Millstone Pt. (Table 11). Of these, several hundred lobsters were recaptured by lobstermen in The Race, a deep water channel 10.5 km from MNPS between

LIS and Block Island Sound, which suggests a migration route for lobsters that exit LIS. A number of lobsters (113) were reported caught outside LIS along the Rhode Island coast (28), off of Block Island (19) and in waters along the south shore of Cape Cod (13). Tag returns were also reported by lobstermen fishing the offshore waters, in canyons on the edge of the continental shelf (Block n=7, Hudson n=10, Atlantis n=5, Veatch n=2). Similar offshore migrations were demonstrated by other researchers working in waters from Canada to southern New England (Saila and Flowers 1968; Uzmann et al. 1977; Cooper and Uzmann 1980; Campbell and Stasko 1985, 1986).

### Entrainment

The total number of lobster larvae collected in samples of the MNPS cooling water was 219 during 1993, which was within the range of values for 3-unit studies (157-625), but higher than the number collected in 2-unit studies (102 and 143). Occurrence of larvae in entrainment samples coincided with the catch of berried females and their egg development. Observations of the shape, color and development of embryos indicated that

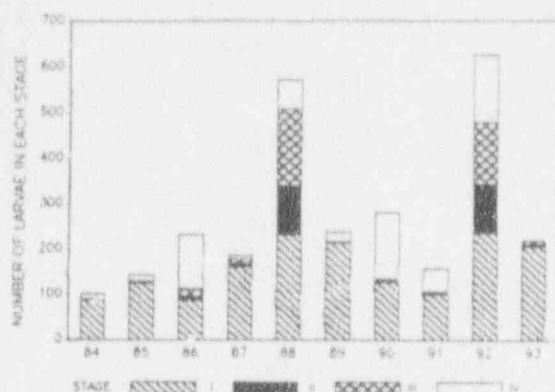


Fig. 8. Annual number of lobster larvae (Stage I-IV) collected in samples of the MNPS discharges from 1984 to 1993.

TABLE 12. Delta mean density (number per 1000 m<sup>3</sup>  $\pm$  5% C.I.) of lobster larvae collected in day and night entrainment samples from 1984 to 1993.

Year	Time of day	Delta mean density <sup>a</sup>	95% C.I.
1984	Day	0.158	0.061-0.256
	Night	0.737	0.138-1.336
1985	Day	0.390	-0.041-0.820
	Night	0.620	0.290-0.951
1986	Day	0.324	0.063-0.585
	Night	1.399 <sup>b</sup>	0.556-2.242
1987	Day	0.791	0.040-1.542
	Night	0.667	0.205-1.129
1988	Day	0.727	-0.199-1.653
	Night	0.688	0.271-1.106
1989	Day	0.158	0.087-0.229
	Night	1.403 <sup>b</sup>	0.537-2.269
1990	Day	0.341	0.101-0.581
	Night	1.167 <sup>b</sup>	0.569-1.765
1991	Day	0.287	0.131-0.442
	Night	0.756 <sup>b</sup>	0.502-1.010
1992	Day	1.299	0.043-2.555
	Night	1.369	0.530-2.209
1993	Day	0.963	-0.207-2.132
	Night	1.168	-0.097-2.433

<sup>a</sup> Number per 1000 m<sup>3</sup>.

<sup>b</sup> Significant difference between day and night densities 2-sample t-tests ( $p < 0.05$ ).

most hatching occurred during late May and June. As stated earlier, only a few berried females were caught in July, indicating the completion of hatching. Stage I larvae accounted for 94% of the four larval stages collected during 1993 (Fig. 8). Similarly, Stage I larvae predominated in the collections during previous 3-unit (38-90%) and 2-unit studies (86 and 87%; Fig. 8). Stage II and III larvae were rarely collected in our entrainment samples, and with the exception of the 1988 and 1992 collections, these larval stages have accounted for only 5% of the total larvae collected since the studies began in 1984. Only two Stage IV larvae were collected during 1993, accounting for < 1% of the total larvae collected. Stage IV larvae comprised between 4 and 52% of the four larval stages collected in previous studies (Fig. 8). Other researchers in southern New England found similar high variability in numbers and stage composition of lobster larvae (Bibb et al. 1983; Fogarty 1983; Lux et al. 1983; Blake 1984, 1988).

The  $\Delta$ -mean density of lobster larvae collected in 1993 night samples (1.168 per 1000 m<sup>3</sup>) was similar to the density in day samples (0.963/1000 m<sup>3</sup>; Table 12). However, density of lobster larvae night samples had been significantly higher than day samples in four previous 3-unit study years (1986, 1989, 1990, 1991; Table 12). The diel variability observed in our studies was similar to what other researchers found throughout the range of lobsters. Positive phototaxis of Stage I larvae was demonstrated in early laboratory studies by Templeman (1937, 1939), which contrasts with results of more recent field surveys conducted in Canadian waters by Harding et al. (1987). Their work on Browns Bank, southwest of Nova Scotia, indicated that most Stage I larvae were collected between 15 and 30 m during the day and rarely found below 10 m at night. Thermal gradients were recently shown to influence the vertical migration of lobster larvae, with all four stages seeking the warm water above the thermocline regardless of time of day (Boudreau et al. 1991).

The 1993 annual  $\Delta$ -mean density of lobster larvae collected in entrainment samples (1.081 per 1000 m<sup>3</sup>) was the second highest value reported since the entrainment studies began in 1984 (3-unit 0.525-1.334; 2-unit 0.409 and 0.504; Table 13). The estimated number of lobster larvae entrained through the MNPS cooling water system was 389,767, which was within the range of entrained larvae reported in other 3-unit studies (296,173-



TABLE 13. Annual mean density (number per 1000 m<sup>3</sup>) of lobster larvae in entrainment samples during their season of occurrence and annual entrainment estimates with 95% C.I. for MNPS from 1984 to 1993.

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

<sup>a</sup> Mean densities are based on the delta-mean (NUSCO 1988b and Pennington 1983).

<sup>b</sup> Unit 3 began commercial operation.

615,285), but higher than the 2-unit estimates of 77,458 and 128,550. Because total entrainment estimates were based on cooling water volumes and larval density, the highest entrainment estimates were observed in 1986, 1988, and 1990 when all units operated at or near full power from May through July, and in 1992 when larval densities were highest. Since Unit 3 began operating, entrainment estimates have been substantially higher, because the cooling water demand of Unit 3 alone is approximately the volume required by Units 1 and 2 combined.

Impacts of entrainment of lobster larvae on lobster recruitment are difficult to assess due to the unreliable estimates of larval and post-larval survival rates (Phillips and Sastry 1980; Caddy and Campbell 1986; Cobb 1986; Blake 1991). In addition, researchers disagree on the source and dispersion mechanism of lobster larvae. As a result, larval survival estimates are wide ranging. In Canadian waters perhaps less than 1% of newly hatched larvae survive to the fourth stage (Scarratt 1964, 1973; Harding et al. 1982), but in LIS more than 50% were estimated to survive through the entire larval phase (Lund and Stewart 1970; Blake 1991). Most Stage I larvae entrained through the plants cooling water system probably originate from the berried females in the MNPS area, because Stage I larvae are only in the water column for 3 to 5 days. It is unlikely that Stage IV larvae originate from the local population because they are in the water column for between 4 and 6 weeks and water currents would carry them to other areas

of LIS. Furthermore, Stage IV larvae exhibit directional swimming behavior and can move tens of kilometers from the origin of hatching (Cobb et al. 1989; Rooney and Cobb 1991). Lund and Stewart (1970) indicated that the large number of berried females found in western LIS (27%; Smith 1977) may be responsible for recruitment of Stage IV larvae in middle and eastern LIS. Stage IV larvae in our area may also originate from outside LIS, as Cobb et al. (1989) observed Stage IV larvae in Block Island Sound swimming in a northerly and easterly direction. Recently, commercial lobstermen in western LIS reported a 50% decline in legal lobster abundance and the overall U.S. landings were 13% lower, which may lead to a decline in our area if local recruitment depends on larval production in other areas. The decline in legal lobster abundance is probably related to overfishing, and in western LIS the decline may be exacerbated by lobster kills due to disease (*Gaffkemia*), eutrophication, and hypoxia of bottom waters during the summer.

## Conclusions

Since our studies began in 1978, fishing effort in Connecticut waters has more than doubled. The fishery is almost completely dependent on new animals molting into legal size; each year nearly all the lobsters above the minimum legal size are removed by fishing. The total number of lobsters caught and total CPUE in our study area reached record levels in 1992 and remained high during



1993. However, legal lobster catches have significantly declined since the NUSCO study began in 1978. Legal catches were expected to improve in 1993, after large numbers of lobsters, observed to be just below legal size in the 1992 catches, molted to legal size. Instead, legal CPUE declined during 1993 in contrast with recruitment patterns of previous study years when strong recruit classes were followed by increased legal catches one year later. The observation that legal catches did not increase in 1993 may be strong indication that the lobster resource is currently overfished.

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

The density of lobster larvae was higher during 3-unit studies due to the higher percentage of berried females. Estimated numbers of larvae entrained through the MNPS cooling water systems were also higher during 3-unit operation, in part due to the higher cooling water demand of Unit 3. Higher larval entrainment may affect subsequent legal lobster abundance, but quantification of this impact is difficult given the uncertainty of larval origin and larval survival and recruitment rates to legal size. Since lobsters require 4-5 years of growth before they are vulnerable to capture, and an additional 2 years of growth to reach legal size, a decline in local lobster abundance caused by larval entrainment would not be apparent for several years. Continued monitoring of lobsters and lobster larvae will demonstrate the effects, if any, of 3-unit operation on the local lobster population.

## References Cited

- Aiken, D.E. 1973. Proecdysis, setal development, and molt prediction in the American lobster, (*Homarus americanus*). J. Fish. Res. Board Can. 30:1337-1344.
- Aiken, D.E. 1980. Molting and growth. Pages 91-163 in J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol. I. Academic Press, Inc., New York.
- Aiken, D.E., and S.L. Waddy. 1980. Reproductive biology. Pages 215-276 in J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol. I, Academic Press, Inc., New York.
- Anthony, V.C., and J.F. Caddy. 1980. Proceedings of the Canada-U.S. workshop on status of assessment science for N.W. Atlantic lobster (*Homarus americanus*) stocks (St. Andrews, N.B., Oct 24-26, 1978). Can. Tech. Rep. Fish. Aquat. Sci. 932. 186 pp.
- Bibb, B.G., R.L. Hersey, and R.A. Marcello, Jr. 1983. Distribution and abundance of lobster larvae (*Homarus americanus*) in Block Island Sound. NOAA Tech. Rep. NMFS SSRF-775:15-22.
- Blake, M.M. 1984. Annual progress report Connecticut lobster investigations, January-December 1983. NOAA-NMFS Project No. 3-374-R. 47 pp.
- Blake, M.M. 1988. Final Report Connecticut lobster investigations January 1, 1983-December 31, 1987. NOAA-NMFS Project No. 3-374-R. 103 pp.
- Blake, M.M. 1991. Connecticut lobster (*Homarus americanus*) population recruitment studies January 1, 1988-December 31, 1990. NOAA-NMFS Project No. 31J4. 87 pp.
- Blake, M.M., and E.M. Smith. 1984. A marine resources plan for the state of Connecticut. Connecticut Dept. of Environ. Protection, Mar. Fish. 244 pp.
- Boudreau, B., Y. Simard, and E. Bourget. 1991. Behavioural responses of the planktonic stages of American lobster *Homarus americanus* to thermal gradients, and ecological implications. Mar. Ecol. Prog. Ser. 76:13-23.
- Briggs, P.T., and F.M. Mushacke. 1979. The American lobster in western Long Island Sound. NY Fish Game J. 26:59-86.
- Briggs, P.T., and F.M. Mushacke. 1980. The American lobster and the pot fishery in the

- inshore waters off the south shore of Long Island, New York. NY Fish Game J. 27:156-178.
- Briggs, P.T., and F.M. Mushacke. 1984. The American lobster in western Long Island Sound: Movement, growth and mortality. NY Fish Game J. 31:21-37.
- Caddy, J.F., and A. Campbell. 1986. Summary of session 9: summary of research recommendations. Can. J. Fish. Aquat. Sci. 43:2394-2396.
- Campbell, A. 1982. Movements of tagged lobsters released off Port Maitland, Nova Scotia, 1944-80. Can. Tech. Rep. Fish. Aquat. Sci. No. 1136. 41 pp.
- Campbell, A., and A.B. Stasko. 1985. Movements of tagged American lobsters, *Homarus americanus*, off southwestern Nova Scotia. Can. J. Fish. Aquat. Sci. 42:229-238.
- Campbell, A., and A.B. Stasko. 1986. Movements of lobsters (*Homarus americanus*) tagged in the Bay of Fundy, Canada. Mar. Biol. 92:393-404.
- Cobb, J.S. 1986. Summary of session 6: ecology of population structures. Can. J. Fish. Aquat. Sci. 43:2389-2390.
- Cobb, J.S., D. Wang, R.A. Richards, and M.J. Fogarty. 1986. Competition among lobsters and crabs and its possible effects in Narragansett Bay, Rhode Island. Pages 282-290 in G.S. Jamieson and N. Bourne, eds. North Pacific Workshop on stock assessment and management of invertebrates. Can. Spec. Publ. Fish. Aquat. Sci. 92.
- Cobb, J.S., D. Wang, D.B. Campbell, and P. Rooney. 1989. Speed and direction of swimming by postlarvae of the American lobster. Trans. Am. Fish. Soc. 118:82-86.
- Cooper, R.A. 1970. Retention of marks and their effects on growth, behavior and migrations of the American lobster, *Homarus americanus*. Trans. Amer. Fish. Soc. 99:409-417.
- Cooper, R.A., R.A. Clifford, and C.D. Newell. 1975. Seasonal abundance of the American lobster, *Homarus americanus*, in the Boothbay Region of Maine. Trans. Amer. Fish. Soc. 104:669-674.
- Cooper, R.A., and J.R. Uzmann. 1971. Migrations and growth of deep-sea lobsters, *Homarus americanus*. Science 171:288-290.
- Cooper, R.A., and J.R. Uzmann. 1980. Ecology of juvenile and adult *Homarus americanus*. Pages 97-142 in J.S. Cobb, and B.F. Phillips, eds. The biology and management of lobsters, Vol II, Academic Press, Inc., New York.
- Dow, R.L. 1966. The use of biological, environmental and economic data to predict supply and to manage a selected marine resource. The Amer. Biol. Teacher 28:26-30.
- Dow, R.L. 1969. Cyclic and geographic trends in seawater temperature and abundance of American lobster. Science 164:1060-1063.
- Dow, R.L. 1976. Yield trends of the American lobster resource with increased fishing effort. Mar. Technol. Soc. 10:17-25.
- Ennis, G.P. 1971. Lobster (*Homarus americanus*) fishery and biology in Bonavista Bay, Newfoundland, 1966-70. Fish. Mar. Serv. Tech. Rep. 289. 46 pp.
- Ennis, G.P. 1974. Observations on the lobster fishery in Newfoundland. Fish. Mar. Serv. Tech. Rep. 479. 21 pp.
- Ennis, G.P. 1980. Size-maturity relationships and related observations in Newfoundland populations of the lobster (*Homarus americanus*). Can. J. Fish. Aquat. Sci. 37:945-956.
- Ennis, G.P. 1984. Small-scale seasonal movements of the American lobster, *Homarus americanus*. Trans. Am. Fish. Soc. 113:336-338.
- Fair, J.J., and B. Estrella. 1976. A study on the effects of sublegal escape vents on the catch of lobster traps in five coastal areas of Massachusetts. Unpublished manuscript, Mass. Div. Mar. Fish. 9 pp.
- Flowers, J.M., and S.B. Saila. 1972. An analysis of temperature effects on the inshore lobster fishery. J. Fish. Res. Board Can. 29:1221-1225.
- Fogarty, M.J. 1983. Distribution and relative abundance of American lobster, *Homarus americanus* larvae: New England investigations during 1974-79. NOAA Tech. Rep. NMFS SSRF-775. 64 pp.
- Fogarty, M.J., and D.V.D. Borden. 1980. Effects of trap venting on gear selectivity in the inshore Rhode Island American lobster, *Homarus americanus*, fishery. Fish. Bull., U.S. 77:925-933.
- Fogarty, M.J., D.V.D. Borden, and H.J. Russell. 1980. Movements of tagged American lobster, *Homarus americanus*, off Rhode Island. Fish. Bull., U.S. 78:771-780.
- Gilbert, R.O. 1987. Statistical methods for environmental pollution monitoring. Van

- Nostrand Reinhold Co., New York. 320 pp.
- Harding, G.C., W.P. Vass, and K.F. Drinkwater. 1982. Aspects of larval American lobster (*Homarus americanus*) ecology in St. Georges Bay, Nova Scotia. *Can. J. Fish. Aquat. Sci.* 39:1117-1129.
- Harding, G.C., J.D. Pringle, W.P. Vass, S. Pearre Jr., and S.J. Smith. 1987. Vertical distribution and daily movement of larval lobsters *Homarus americanus* over Browns Bank, Nova Scotia. *Mar. Ecol. Prog. Ser.* 49:29-41.
- Herrick, F.H. 1911. Natural history of the American lobster. *Bull. U.S. Bureau Fish.* 29:149-408.
- Hollander, M., and D.A. Wolfe. 1973. Nonparametric statistical methods. John Wiley and Sons. New York. 503 pp.
- Hudon, C., and G. Lamarche. 1989. Niche segregation between American lobster *Homarus americanus* and rock crab *Cancer irroratus*. *Mar. Ecol. Prog. Ser.* 52:155-168.
- Keser, M., D.F. Landers, Jr., and J.D. Morris. 1983. Population characteristics of the American lobster, *Homarus americanus*, in eastern Long Island Sound, Connecticut. NOAA Tech. Rep. NMFS SSRF-770. 7 pp.
- Krouse, J.S. 1973. Maturity, sex ratio, and size composition of the natural population of American lobster, *Homarus americanus*, along the Maine coast. *Fish. Bull., U.S.* 71:165-173.
- Krouse, J.S. 1976. Incidence of cull lobsters, *Homarus americanus*, in commercial and research catches off the Maine coast. *Fish. Bull., U.S.* 74:719-724.
- Krouse, J.S. 1978. Effectiveness of escape vent shape in traps for catching legal-sized lobster, *Homarus americanus*, and harvestable-sized crabs, *Cancer borealis* and *Cancer irroratus*. *Fish. Bull., U.S.* 76:425-432.
- Krouse, J.S. 1980. Summary of lobster, *Homarus americanus*, tagging studies in American waters (1898-1978). *Can. Tech. Rep. Fish. Aquat. Sci.* 932:135-140.
- Krouse, J.S. 1981. Movement, growth, and mortality of American lobsters, *Homarus americanus*, tagged along the coast of Maine. NOAA Tech. Rep. NMFS SSRF-747. 12 pp.
- Krouse, J.S., and J.C. Thomas. 1975. Effects of trap selectivity and some population parameters on the size composition of the American lobster, *Homarus americanus*, catch along the Maine coast. *Fish. Bull., U.S.* 73:862-871.
- Krouse, J.S., K.H. Kelly, D.B. Parkhurst Jr., G.A. Robinson, B.C. Scully, and P.E. Thayer. 1993. Maine Department of Marine Resources Lobster Stock Assessment Project 3-1J-61-1. Annual report April 1, 1992 through January 31, 1993. 61 pp.
- Kurata, H. 1962. Studies on the age and growth of Crustacea. *Bull. Hokkaido Reg. Fish. Res. Lab.* 24:1-115.
- Landers, D.F., Jr., and M.M. Blake. 1985. The effect of escape vent regulation on the American lobster, *Homarus americanus*, catch in eastern Long Island Sound, Connecticut. *Trans. 41st Annual Northeast Fish Wild. Conf.* 9 pp.
- Lund, W.A., Jr., and L.L. Stewart. 1970. Abundance and distribution of larval lobsters, *Homarus americanus*, off southern New England. *Proc. Natl. Shellfish. Assoc.* 60:40-49.
- Lux, F.E., G.F. Kelly, and C.L. Wheeler. 1983. Distribution and abundance of larval lobsters (*Homarus americanus*) in Buzzards Bay, Massachusetts, in 1976-79. NOAA Tech. Rep. NMFS SSRF-775:29-33.
- Mauchline, J. 1976. The Hiatt growth diagram for Crustacea. *Mar. Biol.* 35:79-84.
- McConnaughey, R.A., and L.L. Conquest. 1993. Trawl survey estimation using a comparative approach based on lognormal theory. *Fish. Bull., U.S.* 91:107-118.
- McLeese, D.W., and D.G. Wilder. 1958. The activity and catchability of the lobster (*Homarus americanus*) in relation to temperature. *J. Fish. Res. Board Can.* 15:1345-1354.
- Miller R.J. 1989. Catchability of American Lobsters (*Homarus americanus*) and Rock Crabs (*Cancer irroratus*) by traps. *Can. J. Fish. Aquat. Sci.* 46:1652-1657.
- NMFS (National Marine Fisheries Service). 1993. Report of the 16th Northeast Regional Stock Assessment Workshop. *Northeast Fish. Sci. Cen. Ref. Doc.* 93-18. NOAA/NMFS Northeast Fisheries Science Center, Woods Hole, MA. 118 pp.
- NUSCO (Northeast Utilities Service Company). 1982. Lobster Population Dynamics-A Review and Evaluation. Pages 1-32 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Resume 1968-1981.*
- NUSCO. 1986. The effectiveness of the Millstone

- Unit 1 sluiceway in returning impinged organisms to Long Island Sound. Enclosure to letter D01185 dated May 27, 1986 from R.A. Reckert, NUSCO, to S.J. Pac, Commissioner, CTDEP. 18 pp.
- NUSCO. 1987a. Lobster population dynamics. Pages 1-42 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of studies prior to Unit 3 operation 1987.
- NUSCO. 1987b. The effectiveness of the Unit 3 fish return system 1987. 20 pp.
- NUSCO. 1988a. The usage and estimation of DELTA means. Pages 311-320 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.
- NUSCO. 1988b. The effectiveness of the Millstone Unit 3 fish return system. Appendix 1 to Enclosure 3 to letter D01830 dated January 29, 1988 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CTDEP. 21 pp.
- NUSCO. 1988c. Hydrothermal Studies. Pages 323-355 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.
- NUSCO. 1993. Lobster studies. Pages 7-32 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1992.
- Pecci, K.J., R.A. Cooper, C.D. Newell, R.A. Clifford, and R.J. Smolowitz. 1978. Ghost fishing of vented and unvented lobster, *Homarus americanus*, traps. Mar. Fish. Rev. 40:9-43.
- Pennington, M. 1983. Efficient estimators of abundance, for fish plankton surveys. Biometrics 39:281-286.
- Phillips, B.F., and A.N. Sastry. 1980. Larval ecology. Pages 11-57 in J.S. Cobb, and B.F. Phillips, eds. The biology and management of Lobsters, Vol II, Academic Press, Inc., New York.
- Richards, R.A., J.S. Cobb, and M.J. Fogarty. 1983. Effects of behavioral interactions on the catchability of American lobster, *Homarus americanus*, and two species of *Cancer* crab. Fish. Bull., U.S. 81:51-60.
- Richards, R.A., and J.S. Cobb. 1987. Use of avoidance responses to keep spider crabs out of traps for American lobsters. Trans. Amer. Fish. Soc. 116:282-285.
- Rooney, P., and J.S. Cobb. 1991. Effects of time of day, water temperature, and water velocity on swimming by postlarvae of the American Lobster, *Homarus americanus*. Can. J. Fish. Aquat. Sci. 48:1944-1950.
- Saila, S.B., and J.M. Flowers. 1968. Movements and behavior of berried female lobsters displaced from offshore areas to Narragansett Bay, Rhode Island. J. Cons. Int. Explor. Mer. 31:342-351.
- Scarratt, D.J. 1964. Abundance and distribution of lobster larvae (*Homarus americanus*) in Northumberland Strait. J. Fish. Res. Board Can. 21:661-680.
- Scarratt, D.J. 1970. Laboratory and field tests of modified sphyron tags on lobsters (*Homarus americanus*). J. Fish. Res. Board Can. 27:257-264.
- Scarratt, D.J. 1973. Abundance, survival, and vertical and diurnal distribution of lobster larvae in Northumberland Strait 1962-63, and their relationships with commercial stocks. J. Fish. Res. Board Can. 30:1819-1824.
- Scarratt, D.J., and P.F. Elson. 1965. Preliminary trials of a tag for salmon and lobsters. J. Fish. Res. Board Can. 22:421-423.
- Sen, P.K. 1968. Estimates of the regression coefficient based on Kendall's tau. J. Am. Stat. Assoc. 63:1379-1389.
- Skud, B.E., and H.C. Perkins. 1969. Size composition, sex ratio and size at maturity of offshore northern lobsters. U.S. Fish Wildl. Spec. Sci. Rep. Fish. 598. 10 pp.
- Smith, E.M. 1977. Some aspects of catch/effort, biology, and the economics of the Long Island lobster fishery during 1976. NOAA-NMFS, Commer. Fish. Res. Dev. Act. Project No. 3-253-R-1. 97 pp.
- Smith, E.M., E.C. Mariani, A.P. Petrillo, L.A. Gunn, and M.S. Alexander. 1989. Principal fisheries of Long Island Sound, 1961-1985. Connecticut Dept. Environ. Prot., Div. of Conservation and Preservation, Bureau of Fisheries, Mar. Fish. Prog.
- Snedecor, G.W., and W.G. Cochran. 1967. Statistical methods. The Iowa State University



- Press, Ames, IA. 593 pp.
- Stewart, L.L. 1972. The seasonal movements, population dynamics and ecology of the lobster, *Homarus americanus* (Milne-Edwards), off Ram Island, Connecticut. Ph.D. Thesis, University of Connecticut, Storrs, CT. 112 pp.
- Templeman, W. 1935. Local differences in the body proportions of the lobster, *Homarus americanus*. J. Biol. Board Can. 1:213-226.
- Templeman, W. 1936. Local differences in the life history of the lobster (*Homarus americanus*) on the coast of the maritime provinces of Canada. J. Biol. Board Can. 2:41-88.
- Templeman, W. 1937. Habits and distribution of larval lobsters (*Homarus americanus*). J. Biol. Board Can. 3:343-347.
- Templeman, W. 1939. Investigations into the life history of the lobster (*Homarus americanus*) on the west coast of Newfoundland, 1938. Newfoundland Dep. Nat. Resour. Res. Bull. (Fish) 7. 52 pp.
- Templeman, W. 1940. Lobster tagging on the west coast of Newfoundland 1938. Newfoundland Dep. Nat. Resour. Res. Bull. (Fish) 8. 16 pp.
- Thomas, J.C. 1973. An analysis of the commercial lobster (*Homarus americanus*) fishery along the coast of Maine August 1966 through December 1970. NOAA-NMFS Tech. Rept. SSRF-667. 57 pp.
- Uzmann, J.R., R.A. Cooper, and K.J. Pecci. 1977. Migrations and dispersion of tagged American lobsters, *Homarus americanus*, on the southern New England Continental Shelf. NOAA Tech. Rep. NMFS SSRF-705. 92 pp.
- Wilder, D.G. 1953. The growth rate of the American lobster (*Homarus americanus*). J. Fish. Res. Board Can. 10:371-412.
- Wilder, D.G. 1963. Movements, growth and survival of marked and tagged lobsters liberated in Egmont Bay, Prince Edward Island. J. Fish. Res. Board Can. 20:305-318.
- Wilder, D.G., and R.C. Murray. 1958. Do lobsters move offshore and onshore in the fall and spring? Fish. Res. Board Can. Atl. Prog. Rep. 69:12-15.





## Eelgrass

Introduction . . . . .	37
Materials and Methods . . . . .	37
Results . . . . .	39
Temperature . . . . .	39
Sediments . . . . .	41
Shoot Density . . . . .	41
Shoot Length . . . . .	42
Standing Stock . . . . .	43
Reproductive Shoots . . . . .	45
Discussion . . . . .	46
Conclusions . . . . .	47
References Cited . . . . .	47



# Eelgrass

## Introduction

Eelgrass (*Zostera marina* L.) is the predominant marine flowering plant in the northern hemisphere. It has colonized estuaries and lagoons of temperate and warm boreal coasts in the Atlantic and Pacific Oceans (Setchell 1935). Populations of this species have successfully adapted to wide ranges of temperature, salinity and water depth (Osterhout 1917; Setchell 1929; Uphof 1941; Burkholder and Doheny 1968; Dillon 1971; Thayer et al. 1984). Although few animal species consume the plant directly, eelgrass provides large quantities of energy and nutrients to consumers through the food chain. Besides providing habitat for higher trophic level fish and crustacean consumers, eelgrass beds have physical, chemical and biological effects on coastal ecosystems. These include: wave action and water movement reduction (Fonseca et al. 1982) and hence prevention of erosion; trapping and binding of sediments (Scoffin 1970) and organic detritus (Walker and McComb 1985); provision of surfaces for colonization by epiphytes (Harlin 1975); high rates of production (Hillman et al. 1989); contribution of calcium carbonate to sediments via epiphyte decomposition (Walker and Woelkerling 1988); and nutrient trapping and recycling (Hemminga et al. 1991). These functions were interrupted when the "Wasting Disease" of the early 1930s destroyed most of eelgrass populations in eastern North America and western Europe (Tutin 1942; Rasmussen 1973, 1977). Following the destruction of these populations, increased wave scour and changes in current patterns caused shoreline erosion, and declines in abundance of many animal species, including commercially important fishes and lobster were observed (Stauffer 1937; Dexter 1947; Milne and Milne 1951; Orth 1973, 1977; Rasmussen 1973, 1977; Thayer et al. 1975; Zieman 1982).

Recovery of eastern North Atlantic eelgrass populations began in the late 1950s, and by the early 1970s populations in the Niantic River were so dense that dynamite was used to clear areas through eelgrass beds to maintain water circulation (Klotz and Knight 1973). Eelgrass populations were also dense in the vicinity of Millstone Nuclear Power Station (MNPS), and large amounts of eelgrass washed ashore at local beaches due to

the natural senescence of plants in late summer and early autumn. Local residents attributed the die off of eelgrass to the operation of Millstone Unit 1. Northeast Utilities initiated studies in 1973 to address these concerns; the studies included mapping of eelgrass beds in Jordan Cove and measurement of environmental (e.g., water temperature and sediment characteristics) and eelgrass population parameters (e.g., shoot lengths and reproductive status) (Klotz and Knight 1973; Knight and Lawton 1974). These studies found no relationship between MNPS operation and the phenology of eelgrass populations. In addition, hydrographic and hydrothermal studies indicated that the thermal plume of Millstone Unit 1 did not reach the eelgrass populations in Jordan Cove or the Niantic River (VAST 1972; NUSCO 1979).

The present study was initiated in 1985 following the prediction that the 3-unit thermal plume would extend into Jordan Cove (ENDECO 1977). Throughout the range of eelgrass, researchers have demonstrated that high water temperature affects populations by reducing growth rate, lowering resistance to disease, and reducing the production and germination of seeds (Burkholder and Doheny 1968; Phillips 1974, 1980; Orth and Moore 1983). The objectives of the present study are to identify temporal patterns of eelgrass distribution, abundance and reproduction and to determine the extent to which these patterns may be affected by natural variability or by MNPS operation.

## Materials and Methods

Three eelgrass study sites in the vicinity of MNPS were sampled during 1993 (White Point-WP, Jordan Cove-JC, Niantic River-NR) (Fig. 1). The WP and JC stations are located 1.6 km and 0.5 km east of the power plants discharge, respectively, and are within the area potentially influenced by the 3-unit thermal plume (ENDECO 1977; NUSCO 1988). The NR site, located about 3 km from Millstone Point, is a control station in an area unaffected by power plant operation (Fig. 1). Water depths (measured below mean low water) were 2.5 m at WP, 1.5 m at NR and 1.1 m at JC.

The WP and JC sites have been sampled since 1985. The NR site has been relocated several

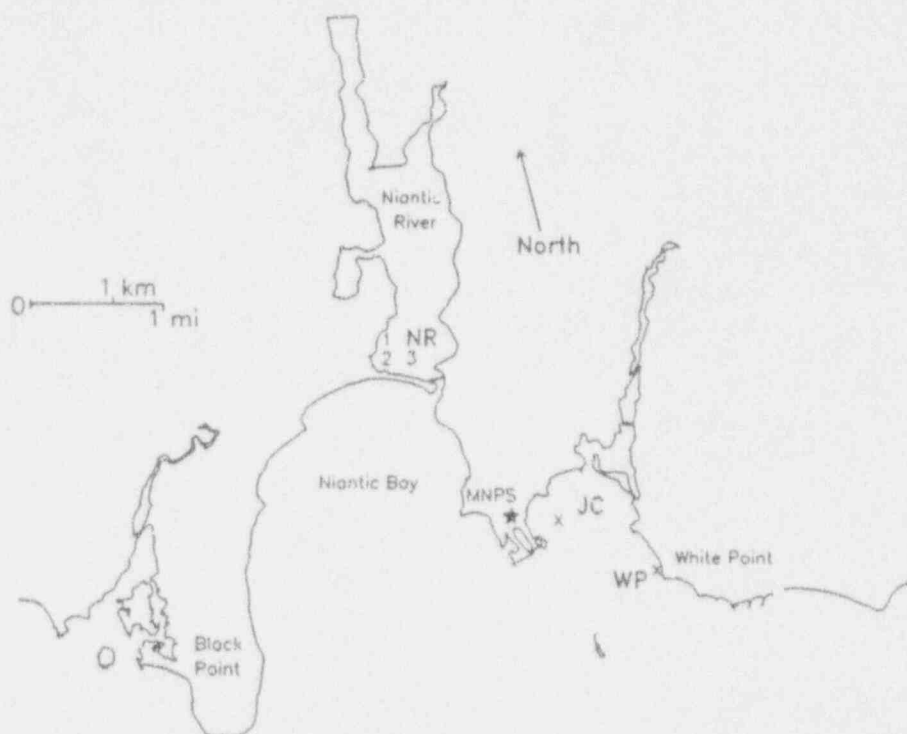


Fig. 1. Map of the Millstone Point area showing the location of eelgrass sampling stations. JC=Jordan Cove (x), NR=Niantic River (1=sampled 1985-June 1986 and 1993, 2=sampled July 1986-September 1986, 3=sampled 1987-1992, was sampled in 1993, but no eelgrass was present), WP=White Point (x).

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

Samples were collected monthly from June

through September 1993, the period of maximum standing stock and plant density. At each station, 16 samples were collected by SCUBA divers randomly tossing a quadrat (25x25 cm, 0.0625 m<sup>2</sup>) within a 10 m radius of the station marker. The upright shoots from plants within each quadrat were harvested, placed in a 0.333 mm mesh bag, and taken to the laboratory for processing. A 3.5 cm diameter x 5 cm deep core was taken together with eelgrass samples for analysis of sedimentary characteristics at each station. Temperature in Jordan Cove was measured by submerging a self-contained thermistor-recorder. Continuous temperature measurements have been recorded there since 1991. Temperatures reported herein cover the period from June 1 through September 30, when additional temperature increases above ambient from the 3-unit operation of MNPS could be most detrimental to eelgrass in Jordan Cove.

All shoots collected were counted in the laboratory and the longest blade of each shoot (up to 20 plants per sample) was measured to the nearest centimeter. The number of reproductive shoots in each sample was recorded and used to



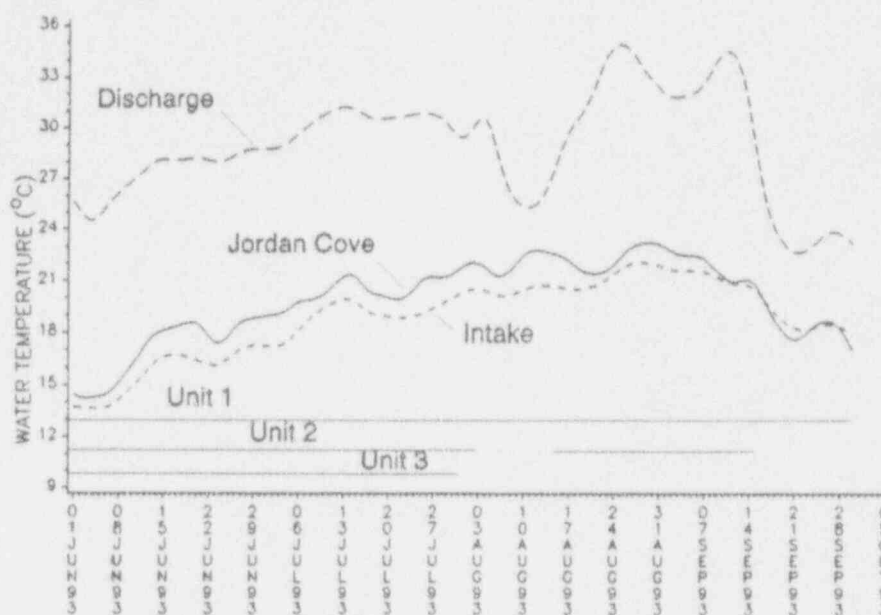


Fig. 2. Daily average water temperature at the Jordan Cove eelgrass station and at the MNPS intake and discharge June 1993 through September 1993. Units 1, 2, and 3 operation are also indicated for the same period.

estimate the percentage of reproductive shoots in the population. Shoots were rinsed in freshwater to remove invertebrates and at the same time, all epiphytes were removed. Because the presence of epiphytes on eelgrass shoots was minimal throughout the study, epiphyte weights were not recorded. Eelgrass standing stock was estimated as the weight of the shoots taken from each quadrat. From 1985 to 1987, shoots were weighed, then dried in an oven at 80 °C to a constant weight. Because dry-weights closely correlated with wet-weights ( $r^2=0.94-0.99$ ), dry weights from 1988 to 1993 were estimated from the wet-weight/dry-weight relationship and presented in this report as grams dry weight/m<sup>2</sup> (g/m<sup>2</sup>). Nonparametric methods were used to examine trends in the time series of eelgrass shoot density and standing stock. The distribution-free, Mann-Kendall test (Hollander and Wolfe 1973) was used to determine whether the time series of mean monthly standing stock biomass or shoot density exhibited significant trends. The slope of the trend, when significant, was estimated by Sen's estimator of the slope (Sen 1968). Eelgrass shoot length was not statistically analyzed because growth occurs at the base of the shoot (from a basal meristem) and tips continuously erode, and because leaf turnover rate

is highest during the summer (Roman and Able 1988). Mean sediment grain size and silt/clay content were determined using the dry sieving method (Folk 1974). Sediment samples were heated to 500 °C for 24 h to determine organic content, estimated as the difference between dry-weight and ash-weight. Both silt/clay and organic content were recorded as a percentage of the total sediment sample weight. Additional sediment samples collected at the NR site #3 were analyzed to evaluate whether any sediment changes noted could be connected to the decline of eelgrass at this site.

## Results

### Temperature

Water temperatures at the JC eelgrass site have been measured with continuous temperature recorders from May 1991 through December 1993. Daily average water temperatures during 1993 (June through September) at JC and the MNPS intake and discharge are shown in Figure 2. Temperatures at JC and the intakes ranged between 13.4°C and 23.5°C and between 13.5°C and

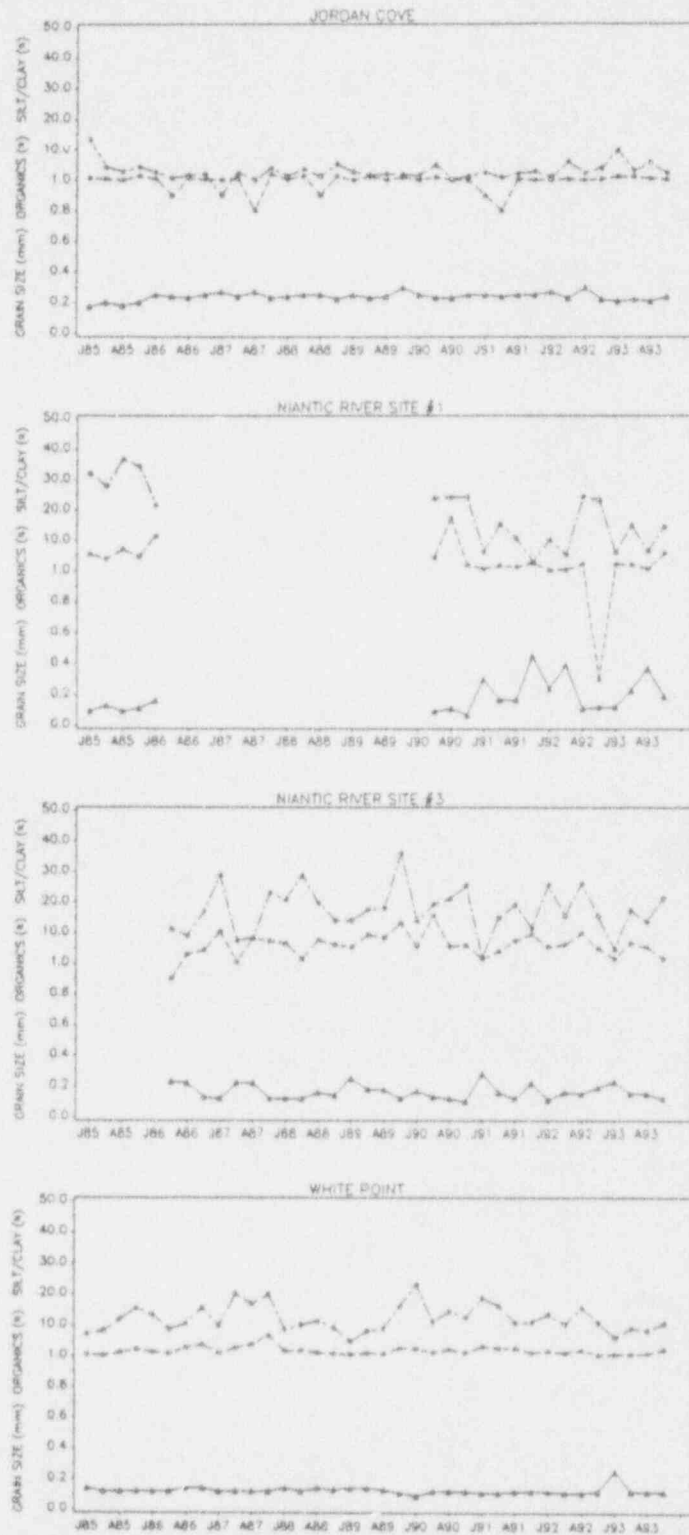


Fig. 3. Mean grain size (▲), organic content (\*) and silt/clay content (●) of sediments at Millstone eelgrass stations at Jordan Cove, Niantic River (Site #1 and Site #3) and White Point sampled during the period June-September from 1985 through 1993.

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

	ANNUAL MEANS									1993			
	1985	1986	1987	1988	1989	1990	1991	1992	1993	Jun	Jul	Aug	Sept
<u>Shoot Density</u>													
Jordan Cove	572	713	542	468	411	338	603	630	484	493	510	516	417
Niantic River	413	72	294	307	240	225	249	233	385	607	387	350	196
White Point	286	218	227	161	335	185	242	204	310	403	313	368	156
<u>Shoot Length</u>													
Jordan Cove	57	57	77	75	74	38	48	53	54	65	60	47	43
Niantic River	50	39	81	88	94	73	51	48	58	68	64	57	31
White Point	107	116	126	86	110	106	87	72	107	104	107	121	82
<u>Standing Stock</u>													
Jordan Cove	243	276	258	238	202	105	169	210	160	211	179	169	81
Niantic River	156	32	184	181	183	143	81	79	125	177	147	143	31
White Point	265	260	201	90	236	180	148	110	275	358	269	388	85

22.2°C, respectively, and were slightly warmer than those measured during the 1992 study (JC range 13.1-22.5°C and intakes 11.9-20.4°C). Daily average water temperature at JC was up to 2.5 °C warmer than at the intakes.

### Sediments

Sediment mean grain size, silt/clay and organic content for the months June through September from 1985 to 1993 are presented in Figure 3. Mean grain size at JC and WP during 1993 ranged from 0.21 to 0.24 mm and from 0.12 to 0.25 mm, respectively. Silt/clay content at JC (3.2-9.9%) was lower than at WP (6.4-10.7%). Sediment organic content at JC was also slightly lower (1.2-2.0%) than at WP (1.7-3.0). With the exception of the high mean grain size value at WP in June, all of the above 1993 values at JC and WP were within the range of previous years (1985-1992).

Mean grain size at NR site #1 in 1993 ranged from 0.12 to 0.37 mm. Organic and silt/clay content were 1.7-6.4% and 6.5-15%, respectively. Values for all sediment parameters observed in 1993 at NR site #1 were within the range of previous years (1985, 1990-92).

Values for all sediment parameters measured at NR site #3 in 1993 were also within the range of previous years (1987-1992). Mean grain size in 1993 ranged between 0.12 and 0.23 mm compared to 0.10 - 0.28 mm during the previous six years. The ranges in silt/clay and organic content in 1993 were 5.1-21.3% and 2.3-7.0%, respectively, compared to 2.6-36.0% and 1.4-15.5%, respectively during 1987-1992.

### Shoot Density

Spatial patterns of eelgrass shoot density among sampling stations during 1993 (June-Sept.) were similar to results from 1985 to 1992. Average shoot density during 1993 was highest at JC (484 shoots/m<sup>2</sup>) and lowest at WP (310 shoots/m<sup>2</sup>; Table 1). The nine year time-series (1985-1993) of shoot density at JC and WP showed no significant trends (Fig. 4). The average density at NR #1 (385 shoots/m<sup>2</sup>) was similar to that in 1985 (413 shoots/m<sup>2</sup>), but annual trend analysis was not performed on NR #1 shoot densities, due to insufficient data for that site.

Monthly shoot densities were highest in June at NR #1 (607 shoots/m<sup>2</sup>) and WP (403 shoots/m<sup>2</sup>),

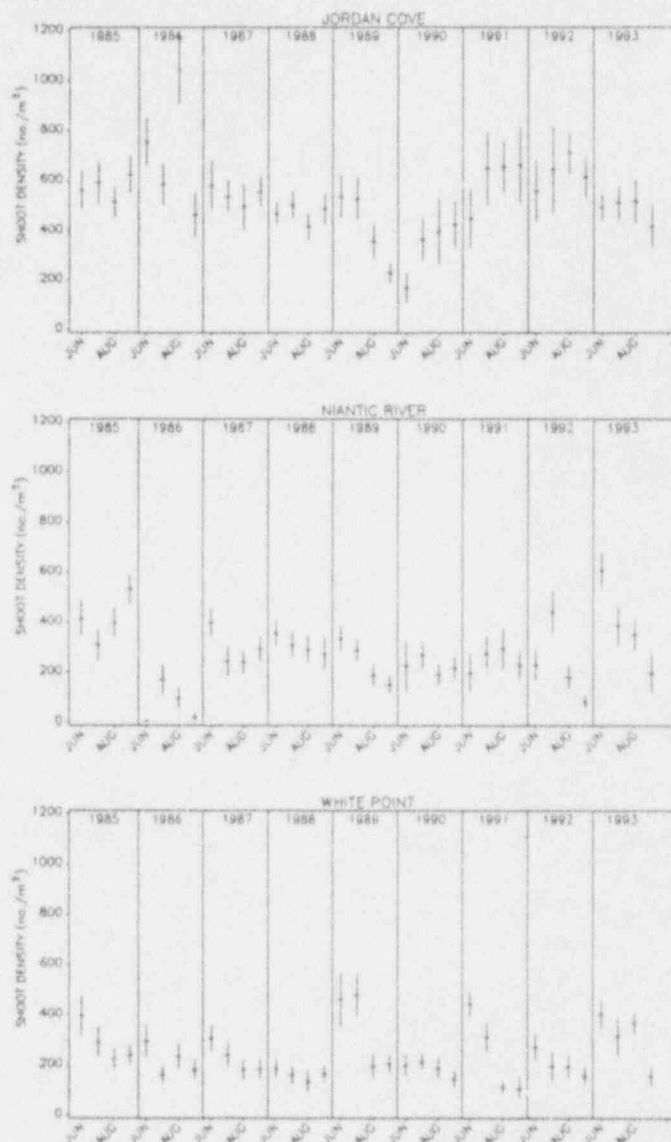


Fig. 4. Mean number of eelgrass shoots per  $m^2 \pm 95\%$  C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1993.

and highest in August at JC (516 shoots/ $m^2$ ; Fig. 4). Shoot densities in June, July and August 1993 at NR (607, 387 and 350 shoots/ $m^2$ , respectively), and at WP (403, 313 and 368 shoots/ $m^2$ , respectively) were, for each station, among the highest observed for those same months over the entire study period (Fig. 4). The 1993 monthly densities at JC (417-516 shoots/ $m^2$ ) were comparable to those from other years.

### Shoot Length

Yearly (1985-1993) and 1993 monthly (June-September) average shoot lengths are presented in Table 1. During 1993, shoots were longest at WP (107 cm), shortest at JC (54 cm) and intermediate at NR #1 (58 cm). The 1993 average shoot length values at JC and WP stations were within the ranges of values from previous years, (43-65 cm and 82-121 cm, respectively). The 1993 shoot length average at NR #1 (58 cm) was within the overall range of all annual means in the Niantic

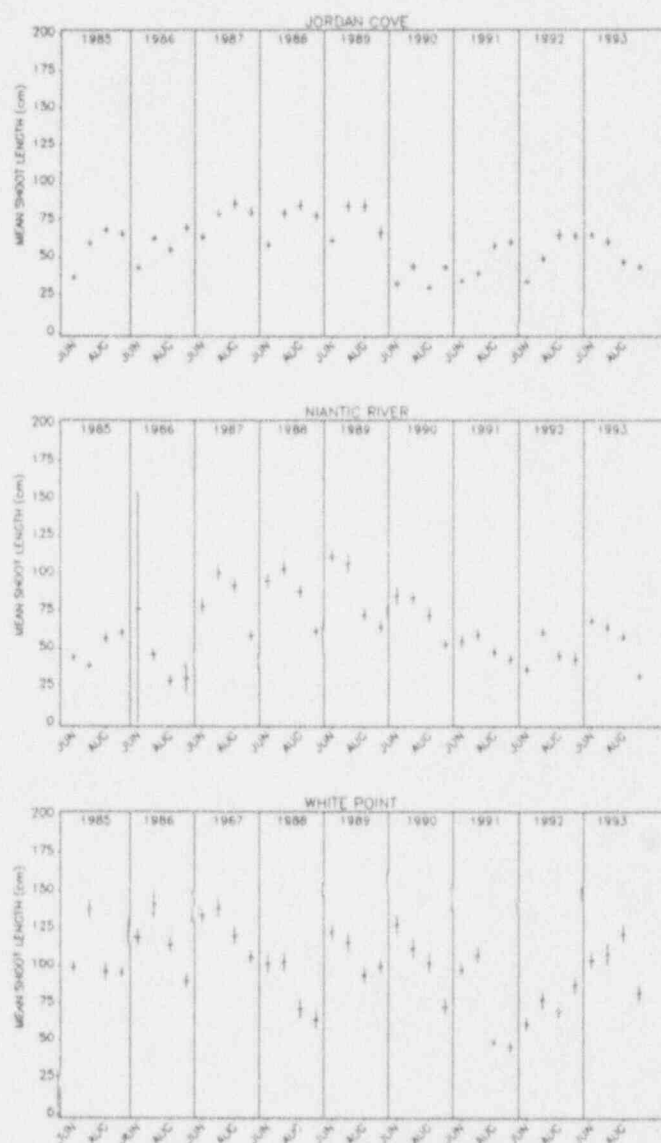


Fig. 5. Mean shoot length (cm)  $\pm$  95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1993.

River since 1985 (39-94 cm), and was very similar to the 1985 average (50 cm), when this site was last sampled.

Monthly shoot lengths in 1993 at JC and NR #1 declined from June (65 and 68 cm, respectively) through September (43 and 31 cm, respectively; Fig. 5); shoot length at WP increased from 104 cm in June to 121 cm in August, then declined to 83 cm in September.

## Standing Stock

Eelgrass standing stock during 1993 was highest at WP (275 g/m<sup>2</sup>) followed by JC (160 g/m<sup>2</sup>) and NR (125 g/m<sup>2</sup>; Table 1). Annual mean standing stock at WP during 1993 was the highest recorded to date (previous range, 90-265 g/m<sup>2</sup>), but the 1993 mean standing stock at JC was the second lowest reported since 1985 (previous range, 105-276 g/m<sup>2</sup>). The 1993 standing stock at NR #1 (125 g/m<sup>2</sup>) was slightly lower than in 1985 (156 g/m<sup>2</sup>), but within



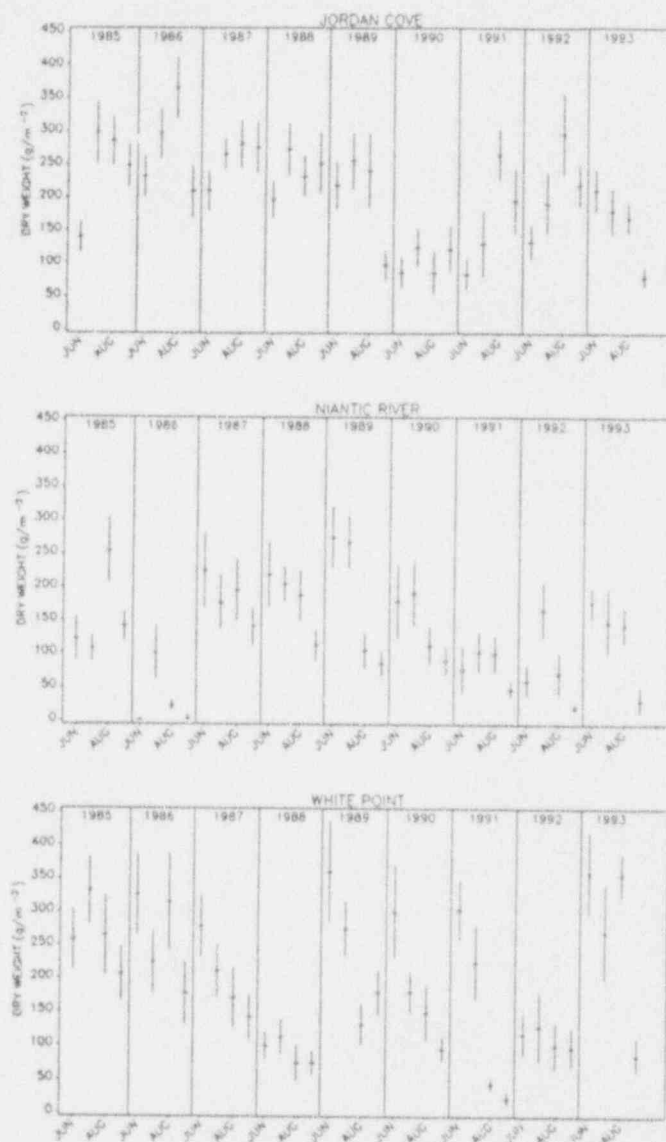


Fig. 6. Mean dry weight (grams per  $m^2$ )  $\pm$  95% C.I. at Millstone eelgrass stations sampled during the period June-September from 1985 through 1993.

the range of annual means at NR #3 from 1987-1992. Trend analysis indicated that standing stocks had significantly declined since 1986 at JC (slope = -3.180,  $p < 0.002$ ). There was no significant trend in standing stock at WP during the nine year (1985-1993) period. Trend analysis was not performed on the NR data, due to frequent station relocation.

Standing stocks in 1993 were highest in June at JC (211  $g/m^2$ ) and NR (177  $g/m^2$ ) and highest in August at WP (388  $g/m^2$ ) (Table 1). Monthly mean standing stocks for June, July and August

(1993) at NR and JC were within the range of monthly means reported since 1985 (Fig. 6). The September 1993 mean at NR (although within the range of previous years) was among the lowest reported during the study; the September 1993 mean at JC was also among the lowest standing stock measurements made at this site in the last nine years. In contrast, 1993 June, July and August monthly means at WP were among the highest reported at that site to date (Fig. 6). However, September standing stock at WP was among the lowest reported at this site.

TABLE 2. Number of reproductive shoots, total number of shoots and percentage of reproductive shoots at eelgrass sampling stations from June 1985 through September 1993.

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

<sup>a</sup> Total number of reproductive shoots.

<sup>b</sup> Total number of shoots (vegetative + reproductive).

### Reproductive Shoots

Percentage of reproductive eelgrass shoots in 1993 was highest in June at JC (11.4%) and NR (15.4%) and highest in July at WP (5.4%) (Table 2). Percentage of reproductive shoots declined by August to <0.1% at JC and NR, and to 3.0% at

WP. No reproductive shoots were collected at any station in September. Annual reproductive shoot percentages in 1993 were the highest values observed since 1985 at JC (4.8%) and all NR sites (8.7%). The 1993 value at WP (3.9%) was within the range of previous years (0.4-10.4%).

## Discussion

Density, length and standing stock of eelgrass shoots were examined during 1993 as part of a long-term sampling program to assess whether any changes in eelgrass populations have resulted from 3-unit operation at Millstone Nuclear Power Station. At WP, no significant trends were evident in the nine year time-series of plant density or standing stock since 1985. In 1993, the WP study parameters (i.e. shoot length, percentage of reproductive shoots and sediment characteristics) were within the ranges of previous years. No long-term changes, either power plant-induced or natural, have been identified with this population.

Niantic River eelgrass populations are not exposed to the MNPS effluent. Therefore, factors other than the effluent discharge were responsible for the eliminations of populations at NR #1 and NR #2 in 1986 and early 1987, and the gradual loss of plants at NR #3 from 1987 to 1993. Eelgrass populations throughout the North Atlantic are characterized by high variability and fluctuations in abundance. Therefore, the eelgrass population losses in the Niantic River are not considered to be merely a local phenomenon, but rather, are examples of such large-scale variability in population parameters. Loss of eelgrass has been attributed to a variety of causes, ranging from natural, e.g., 'wasting disease' (den Hartog 1987), severe storms (Patriquin 1975) to human activities, e.g. eutrophication (Bulthuis 1983; Orth and Moore 1983; Cambridge and McComb 1984; Neverauskas 1985; Burkholder 1993), land reclamation, or changes in near-shore land use (Kemp et al. 1983). The disappearance of eelgrass from the Niantic River in the late 1980s was attributed to a decline in water quality and the presence of *Labyrinthula* (Short 1988), but none of these factors were implicated in the recent loss of plants from the lower river (NR #3). Instead, their loss was associated with a heavy *Mytilus* set in 1991, which may have aided in uprooting, smothering or reducing light availability to plants. Recovery of the eelgrass population at NR #1, first observed in 1989 and continuing through 1993, was a relatively quick process. Others have suggested that recolonization on a global scale can take from 80-200 years (Clark and Kirkman 1989). The apparent full recovery of the NR #1 population is attributed to recruitment of new

plants through germination of seeds transported from healthy beds at the mouth of the river or in Niantic Bay, and the subsequent vegetative rhizome propagation. Although eelgrass meadows persist mainly by vegetative growth (Tomlinson 1974), seed production and dispersal are important mechanisms for maintenance of a healthy population, and are the only recolonization mechanisms. Such patterns of recolonization were documented in the Chesapeake Bay (Orth 1976; Orth and Moore 1981, 1983, 1986); as long as nearby populations remain healthy, and a source of seeds (e.g., at the re-established NR #1), it is reasonable to expect similar recolonization in the lower Niantic River. Continued monitoring will provide information concerning eelgrass populations at these sites.

In Jordan Cove, density, growth and standing stock of eelgrass shoots were higher during the first year of 3-unit operation (1986), followed by a general decline from 1987 to 1990; in recent years, these population parameters have exhibited considerable recovery. Jordan Cove is shallow, and has large sand flats that are exposed to heating in summer, and freezing in winter. As a result, natural fluctuations in water temperature create a more stressful environment in Jordan Cove than at White Point.

The importance of temperature in regulating eelgrass growth and development was first stressed by Setchell (1929). It was later shown that eelgrass is sensitive to small temperature variations (Thayer et al. 1984). Eelgrass plants do not produce seeds at temperatures above 15-20 °C (Burkholder and Doheny 1968; Orth and Moore 1983). Higher water temperatures, e.g., from heated effluents of power plants, could eliminate eelgrass from nearby areas (Phillips 1974; Thayer et al. 1984). The high annual percentage of reproductive plants at JC during 1993 suggests that temperature increases from 3-unit operation did not affect the reproductive cycle at this site. Studies of another seagrass, *Thalassia*, in Florida (Roessler and Zieman 1969; Wood et al. 1969; Zieman 1970; Roessler 1971) and of *Spartina alterniflora* in Maine (Keser et al. 1978), showed a significant decline in abundance of these plants in the vicinity of power plant effluents. Elevated water temperatures increased respiration beyond levels that could be supported by plant photosynthesis. Recognizing that eelgrass meadows are among the

most productive of marine systems (Mann 1973; McRoy and McMillan 1977; Zieman and Wetzel 1980) and act to stabilize sediments (Wood et al. 1969; Zieman 1972; Orth 1977), changes in eelgrass abundance at Jordan Cove could also affect the movement of sediments and species abundance of associated infaunal communities.

Further recognizing the importance of water temperature in regulating eelgrass populations, and that the thermal plume resulting from 3-unit operation was predicted to extend into Jordan Cove (NUSCO 1988), the JC eelgrass station has been closely monitored for the past nine years. To date, the temperature extremes at this site appear to be more related to insolation of the shallow-water sand flats than to incursion of the thermal plume. However, when eelgrass plants are near their physiological limits, even a slight increase above ambient temperature could be detrimental. Further monitoring at various levels of power plant operation, will allow determination of the effects of temperature on the JC eelgrass population, and the extent to which MNPS affects those temperatures.

### Conclusions

Eelgrass population parameters and sediment characteristics, sampled during 1993, were generally consistent with those sampled since 1985. At WP, most parameters in 1993 were within the range of previous years, and none appear to be influenced by MNPS operation. At NR, wide fluctuations of abundance have occurred, including localized elimination of plants from NR #1 (1986-1987), from NR #2 (1987), and from NR #3 (1987-1993). However, in recent years NR #1 has been recolonized, and the eelgrass population apparently recovered completely. These fluctuations were related to factors other than MNPS operation. At JC, changes in eelgrass abundance may be related to changes in water temperature, but, at least to date, these changes appear to be the result of natural variability rather than an impact of 3-unit operation.

### References Cited

- Bulthuis, D.A. 1983. Effects of *in situ* light reduction on density and growth of the seagrass *Heterozostera tasmanica* (Martens ex Aschers.) den Hartog in Western Port, Victoria, Australia. *J. Exp. Mar. Biol. Ecol.* 67:91-103.
- Burkholder, P.R., and T.E. Doheny. 1968. The biology of eelgrass. Contribution No. 3. Dept. Conservation and Waterways, Town of Hempstead, Long Island. Contribution No. 1227 Lamont Geological Observatory, Palisades, New York. 120 pp.
- Burkholder, J. 1993. Botanist Investigates Impact of Nitrate on Seagrasses. *Coastlines* 3:6.
- Cambridge, M.L., and A.J. McComb. 1984. The loss of seagrass from Cockburn Sound, Western Australia. I. The time course and magnitude of seagrass decline in relation to industrial development. *Aquat. Bot.* 20:229-243.
- Clark, S., and H. Kirkman. 1989. Seagrass stability and dynamics. A.W.D. Larkum, A.J. McComb and S.A. Shepherd (eds). Pages 304-345 in *Seagrasses: A Treatise on the Biology of Seagrasses with Special Reference to the Australian Region*. Elsevier, North Holland.
- den Hartog, C. 1987. "Wasting Disease" and other dynamic phenomena in *Zostera* beds. *Aquat. Bot.* 27:3-14.
- Dexter, R.W. 1947. The marine communities of a tidal inlet at Cape Ann, Massachusetts: A study in bio-ecology. *Ecol. Monogr.* 17:261-294.
- Dillon, C.R. 1971. A comparative study of the primary productivity of estuarine phytoplankton and macrobenthic plants. Ph.D. Dissertation. Univ. North Carolina, Chapel Hill. 112 pp.
- ENDECO (Environmental Devices Corporation). 1977. Postoperational Units 1 and 2, preoperational Unit 3 hydrothermal survey of the Millstone Power Station. Rpt. to Northeast Utilities Service Co.
- Folk, D. 1974. *Petrology of Sedimentary Rocks*. Hemphill Pub. Co., Austin, Texas. 192 pp.
- Fonseca, M.S., J.S. Fisher, J.C. Zieman and G.W. Thayer. 1982. Influence of seagrass *Zostera marina* L., on current flow. *Estuar. Coast. and Shelf Sci.* 15:387-364.
- Harlin, M.M. 1975. Epiphyte-host relationships in seagrass communities. *Aquat. Bot.* 1:125-131.
- Hemminga, M.A., P.G. Harrison and F. Vanlent. 1991. The balance of nutrient losses in seagrass meadows. *Mar. Ecol. Prog. Ser.* 71:85-96.
- Hillman, K., D.I. Walker, A.J. McComb and A.W.D. Larkum. 1989. Productivity and nutrient availability. Pages 635-685 in A.W.D.



- Larkum, A.J. McComb and S.A. Shepherd (eds). Seagrasses: A Treatise on the Biology of Seagrasses with Special Reference to the Australian Region. Elsevier, North Holland.
- Hollander, M., and D.A. Wolfe. 1973. Non-parametric statistical methods. John Wiley and Sons, New York. 503 pp.
- Kemp, W.M., W.R. Boynton, R.R. Twilley, J.C. Stevenson and J.C. Means. 1983. The decline of submerged vascular plants in Upper Chesapeake Bay: summary of results concerning possible causes. *Mar. Tech. Soc. J.* 17:78-89.
- Keser, M., B.R. Larson, R.L. Vadas, and W. McCarthy. 1978. Growth and ecology of *Spartina alterniflora* in Maine after a reduction in thermal stress. Pages 420-433 in J.H. Thorpe and J.W. Gibbons (eds). *Energy and Environmental Stress in Aquatic Systems*. DOE Symposium Series (CONF-771114). Nat. Tech. Infor. Ser., Springfield, VA.
- Klotz, R.L., and J.L. Knight. 1973. The ecology of eelgrass (*Zostera marina*). Rpt. to Northeast Utilities Service Co. 14 pp.
- Knight, J.L., and R.B. Lawton. 1974. Report on the possible influence of thermal addition on the growth of eelgrass (*Zostera marina*) in Jordan Cove, Waterford, Connecticut. Rpt. to Northeast Utilities Service Co.
- Mann, K.H. 1973. Seaweeds: Their productivity and strategy for growth. *Science* 182:975-981.
- McRoy, C.P., and C. McMillan. 1977. Production, ecology and physiology of seagrasses. Pages 53-81 in C.P. McRoy and C. Helfferich (eds). *Seagrass Ecosystems: A Scientific Perspective*. Marcel Dekker Inc., New York. 314 pp.
- Milne, L.J., and M.J. Milne. 1951. The eelgrass catastrophe. *Sci. Am.* 184:52-55.
- Neverauskas, V.P. 1985. Port Adelaide sewage treatment works sludge outfall. Effect of discharge on the adjacent marine environment. Progress report, July 1982-May 1984. EWS Rpt. 85/6.
- NUSCO (Northeast Utilities Service Company). 1979. Millstone Point Units 1 and 2 hydrothermal survey report, July 25-August 2, 1977. Submitted to Nuclear Regulatory Commission, January 1979.
- NUSCO. 1988. Hydrothermal Studies. Pages 323-355 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Three-Unit Operational Studies 1986-1987.
- NUSCO. 1993. Eelgrass. Pages 33-48 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. 1992 Annual Report.
- Orth, R.J. 1973. Benthic infauna of eelgrass, *Zostera marina*, beds. *Chesapeake Sci.* 14:258-269.
- Orth, R.J. 1976. The demise and recovery of eelgrass, *Zostera marina*, in the Chesapeake Bay, Virginia. *Aquat. Bot.* 2:141-159.
- Orth, R.J. 1977. The importance of sediment stability in seagrass communities. Pages 281-300 in B.C. Coull (ed). *Ecology of Marine Benthos*. Univ. South Carolina Press, Columbia, SC.
- Orth, R.J., and K.A. Moore. 1981. Submerged aquatic vegetation of the Chesapeake Bay: past, present and future. *Trans. N. Am. Wildl. Nat. Resour. Conf.* 46:271-283.
- Orth, R.J., and K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. *Science* 222:51-52.
- Orth, R.J., and K.A. Moore. 1986. Seasonal and year-to-year variations in the growth of *Zostera marina* L. (eelgrass) in the lower Chesapeake Bay. *Aquat. Bot.* 24:335-341.
- Osterhout, W.J.V. 1917. Tolerance of fresh water by marine plants and its relation to adaptations. *Bot. Gaz.* 63:146-149.
- Patriquin, D.G. 1975. 'Migration' of blowouts in seagrass beds at Barbados and Carriacou, West Indies, and its ecological and geographical implications. *Aquatic Bot.* 1:163-189.
- Phillips, R.C. 1974. Transplantation of seagrasses with special emphasis on eelgrass, *Zostera marina* L. *Aquaculture* 4:161-176.
- Phillips, R.C. 1980. Responses of transplanted and indigenous *Thalassia testudinum* Banks ex König and *Halodule wrightii* Aschers. to sediment loading and cold stress. *Contrib. Mar. Sci.* 23:79-87.
- Rasmussen, E. 1973. Systematics and ecology of the Isefjord marine fauna (Denmark). *Ophelia* 11:1-495.
- Rasmussen, E. 1977. The wasting disease of eelgrass (*Zostera marina*) and its effect on environmental factors and fauna. Pages 1-51 in C.P. McRoy and C. Helfferich (eds). *Seagrass Ecosystems: A Scientific Perspective*. Marcel Dekker Inc., New York. 314 pp.



- Roessler, M.A. 1971. Environmental change associated with a Florida power plant. *Mar. Poll. Bull.* 2:87-90.
- Roessler, M.A., and J.C. Zieman, Jr. 1969. The effects of thermal additions on the biota of southern Biscayne Bay, Florida. Pages 136-145 in *Proceed. Gulf and Caribbean Fish. Inst. Contrib. No. 1165, 22nd Ann. Sess.*
- Roman, C.T., and K.W. Able. 1988. Production ecology of eelgrass (*Zostera marina* L.) in a Cape Cod salt marsh-estuarine system, Massachusetts. *Aquat. Bot.* 32:353-363.
- Scoffin, T.P. 1970. The trapping and binding of subtidal carbonate sediments by marine vegetation in Mimini Lagoon, Bahamas. *J. Sedim. Petrol.* 40:249-273.
- Sen, P.K. 1968. Estimates of regression coefficients based on the Kendall's tau. *J. Am. Stat. Assoc.* 63:1379-1389.
- Setchell, W.A. 1929. Morphological and phenological notes on *Zostera marina* L. *Univ. Calif. Publ. Bot.* 14:389-452.
- Setchell, W.A. 1935. Geographic elements of the marine flora of the North Pacific Ocean. *Am. Nat.* 69:560-577.
- Short, F.T. 1988. Eelgrass-scallop research in the Niantic River: Final report to the Waterford-East Lyme Shellfish Commission. November 15, 1988. 12 pp.
- Stauffer, R.C. 1937. Changes in the invertebrate community of a lagoon after disappearance of the eelgrass. *Ecology* 18:427-431.
- Thayer, G.W., S.M. Adams, and M.W. LaCroix. 1975. Structural and functional aspects of a recently established *Zostera marina* community. Pages 518-540 in L.E. Cronin (ed). *Recent Advances in Estuarine Research*. Academic Press, New York.
- Thayer, G.W., W.J. Kenworthy, and M.S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic coast: A community profile. FWS/OBS-84-02. 147 pp.
- Tomlinson, P.B. 1974. Vegetative morphology and meristem dependence-the foundation of productivity in seagrasses. *Aquaculture* 4:107-130.
- Tutin, T.G. 1942. *Zostera*. *J. Ecol.* 30:217-226.
- Uphof, J.C.T. 1941. Halophytes. *Bot. Rev.* 7:1-58.
- VAST, Inc. 1972. Thermal survey and dye study Millstone Point, Connecticut, September-November 1971. Rpt. to Millstone Point Company.
- Walker, D.I., and A.J. McComb. 1985. Decomposition of leaves from *Amphibolis antarctica* and *Posidonia australis*, the major seagrass species in Shark Bay, Western Australia. *Bot. Mar.* 18:407-413.
- Walker, D.I. and Wm.J. Woelkerling. 1988. A quantitative study of sediment contribution by epiphytic coralline red algae in seagrass meadows in Shark Bay, Western Australia. *Mar. Ecol. Prog. Ser.* 43:71-77.
- Wood, E.J.F., W.E. Odum, and J.C. Zieman. 1969. Influence of seagrasses on the productivity of coastal lagoons. Pages 495-502 in A. Ayala Castanares and F.B. Phleger (eds). *Coastal Lagoons*. Universidad Nacional Autonoma de Mexico, Ciudad Universitaria, Mexico, D.F.
- Zieman, J.C. Jr. 1970. The effects of a thermal effluent stress on the seagrasses and macro-algae in the vicinity of Turkey Point, Biscayne Bay, Florida. Ph.D. Dissertation, Univ. Miami. Coral Gables, Fla. 129 pp.
- Zieman, J.C. Jr. 1972. Origin of circular beds of *Thalassia* (Spermatophyta: Hydrocharitaceae) in Southern Biscayne Bay, Florida, and their relationship to mangrove hammocks. *Bull. Mar. Sci.* 22:559-574.
- Zieman, J.C. Jr. 1982. The ecology of the seagrasses of South Florida: A community profile. U.S. Fish and Wildlife Service, FWS/OBS-82/25.124. 26 pp.
- Zieman, J.C. Jr., and R.G. Wetzel. 1980. Productivity in seagrasses: Methods and rates. Pages 87-116 in R.C. Phillips and C.P. McRoy (eds). *Handbook of Seagrass Biology: An ecosystem perspective*. Garland STPM Press, New York, NY.



## Rocky Intertidal Studies

Introduction .....	53
Materials and Methods .....	53
Qualitative Sampling .....	53
Abundance Measurement .....	53
<i>Ascophyllum nodosum</i> Studies .....	54
Data Analysis .....	55
Results and Discussion .....	55
Qualitative Studies .....	55
Abundance Measurement .....	64
Barnacles .....	64
<i>Fucus</i> .....	67
<i>Chondrus</i> and common epiphytes .....	67
Similarity Dendrograms .....	72
<i>Ascophyllum nodosum</i> Studies .....	74
Growth .....	74
Mortality .....	75
Conclusions .....	77
References Cited .....	77



## Rocky Intertidal Studies

### Introduction

Shore communities are particularly vulnerable to anthropogenic impacts, particularly thermal effects of heated effluents from coastal power plants. In the vicinity of Millstone Nuclear Power Station (MNPS), as with much of coastal New England, a substantial proportion of the shoreline is composed of rock ledge and boulders supporting a rich and diverse community of attached biota. Rocky shore community studies have been, and continue to be, an important aspect of biological monitoring programs associated with nuclear power plants along the New England coastline (Vadas et al. 1976, 1978; Wilce et al. 1978; NAI 1993; NUSCO 1993).

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

### Materials and Methods

#### Qualitative Sampling

Qualitative algal collections were made monthly at nine rocky intertidal stations (Fig. 1). These stations are, in order of most to least exposed to

prevailing winds and storm forces: Bay Point (BP), Fox Island-Exposed (FE), Millstone Point (MP), Twotree Island (TT), White Point (WP), Seaside-Exposed (SE), Seaside-Sheltered (SS), Giants Neck (GN), and Fox Island-Sheltered (FS). The MP and TT stations were added in September 1981; all other stations have been sampled since March 1979. Each 'year' of qualitative sampling extends from March of that year to the following February, i.e., the latest year of qualitative algal data (1992) comprises collections from March 1992 to February 1993. The 1985 sample year (3/85 - 2/86) terminated the 2-unit operational period; the 1986 sample year (3/86 - 2/87) was the first of the 3-unit operational period.

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

Qualitative collections were used to characterize the attached flora at each site during each sampling period. 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.

#### Abundance Measurement

The abundance of rocky intertidal organisms was expressed as percentage of substratum cover. At each qualitative collection station except TT (because of insufficient exposed bedrock), five permanent strip transects were established perpendicular to the water-line, 0.5 m wide and extending from Mean High Water to Mean Low Water levels. Each transect was subdivided into 0.5 m x 0.5 m quadrats and was non-destructively sampled six times per year, in odd numbered months (or a total of 46 times in the Unit 3 operational period to date). These transects are considered 'undisturbed', as they experienced no experimental manipulations. The total number of quadrats in each transect depended on the slope of the transect. The percentage of substratum cover of all organisms and remaining free space in each





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

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 its tidal height: Zone 1 (high intertidal), Zone 2 (mid intertidal), or Zone 3 (low intertidal).

### *Ascophyllum nodosum* Studies

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

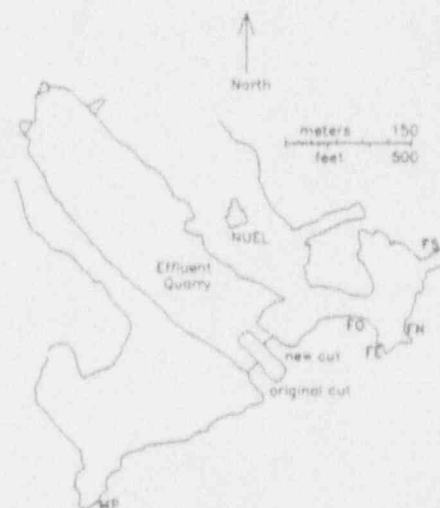


Fig. 2. Detail map of the MNPS vicinity. FO=original experimental *Ascophyllum* site (1979-84). FN=new experimental *Ascophyllum* site (1985-present). MP, FE, FS as in Fig. 1.

elevated temperatures from the thermal plume discharged through two quarry cuts (NUSCO 1985).

*Ascophyllum* plants were measured monthly,

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

### Data Analysis

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

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

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

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

period groupings.

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

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

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

where  $L_t$  is the predicted length at time  $t$ ,  $\alpha$  is the asymptotic length (limit of total increase for the growing season),  $k$  is the rate of decrease of specific growth (shape parameter), and  $t_0$  denotes the time at which the inflection point occurs (time when length is increasing most rapidly). Growth curve parameters were compared among stations and between periods using 2-sample t-tests ( $P=0.05$ ), based on the asymptotic standard errors of the parameter estimates. Growth data representing the latest growing season (1992-1993) were plotted for all stations together and for each station separately, with summaries of 2-unit (1979-1986) and 3-unit (1986-1993) operational data. Because the FN station was established in 1985, 2-unit operational data from this site included only the 1985-86 growing season.

## Results and Discussion

### Qualitative Studies

Spatial and temporal occurrence patterns of the local benthic marine flora were documented through various floristic analyses, based on monthly qualitative algal collections. This floristic tabulation includes all attached, macroscopic algal species. Excluded from these studies are diverse diatom taxa, blue-green algae and some crustose, endophytic or endozooic species. These elements of the microbiota are present but too difficult to consistently collect, and, for many species, to identify. Also included were taxa that are, or may be, conspecific or subspecific forms, or alternate

life history stages of erect macroalgae. For simplicity, we refer to each of these entities as a species throughout this report. Except where noted, nomenclature follows that of South and Tittley (1986).

A summary of results from qualitative sampling studies conducted since 1979 is presented as percent frequency of species occurrence during 3-unit and 2-unit operational studies by month (Table 1) and by station (Table 2). A total of 126 taxa were collected and identified during 1992, which was within the range of annual totals for 2-unit (101-131) and 3-unit (118-126) years (NUSCO 1993). Three species, new to our studies, were collected during 1992; *Phyllophora vrainii* was collected once at FS, *Eudesme virescens* once at WP, and *Griffithsia globulifera* was collected four times, twice at both WP and SS. These additional three species bring the overall total of species collected since 1979 to 161.

These floristic analyses have been successful in documenting occurrence and distributional trends of attached algal species throughout the Millstone area. Many of these floristic differences, identified over time (among seasons or years, or between operational periods) and among stations, provide information on natural factors which produce variability in the local flora and establish a baseline from which power plant impacts can be assessed. For example, seasonal floristic groupings, or suites of species characteristic of warm-water collections (e.g., *Champia parvula*, *Lomentaria baileyana*, *Ceramium diaphanum*, *Dasya baillouviana*, *Polysiphonia novae-angliae*, *Giffordia mitchelliae*, and *Enteromorpha clathrata*) and cold-water collections (e.g., *Dumontia contorta*, *Polysiphonia urceolata*, *Spongonema tomentosum*, *Desmarestia viridis*, *Spongomorpha arcta*, *Monostroma pulchrum* and *M. grevillei*) noted in 1992, were consistent with those identified over the entire study period and discussed in detail in previous reports (NUSCO 1992, 1993). Similarly, consistent occurrence of species which have historically exhibited site specificity (e.g., *Prasiola stipitata* and *Gelidium pusillum*) was also observed during 1992.

Relationships among composite collections made during 2-unit and 3-unit periods at each station were examined using cluster analysis techniques and illustrated by the resulting clustering dendrogram (Figure 3a). In short, the addition of 1992 data has resulted in no substantive change in

major groupings, relative to recent previous years (e.g., NUSCO 1992, 1993). Excluding the FE 3-unit collection, primary groupings of all other collections are distinguished by floras which develop on differing substrata. Collection areas at sites comprising Group I are composed of bedrock ledge, with subgroups separated into exposed sites (WP, MP, BP and FE 2-unit; Subgroup 1a) and sheltered sites (GN and FS; Subgroup 1b). Collection substrata at Group II sites (TT, SS and SE) are primarily large boulders and relatively unstable cobble with few horizontal surfaces. The strong dissimilarity of the FE 3-unit collection (Group III) to all other collections is due to the unique floristic assemblage that has developed at this site in response to exposure to the 3-unit thermal plume.

A similar analysis was applied to annual collections at FE (Fig. 3b) to illustrate yearly changes to the algal community brought about by important power plant operational events. Group I represents collections made during 2-unit 1-cut operation, when the unimpacted flora at FE was similar to that observed at other exposed sites (see Fig. 3a and NUSCO 1987). Temperature conditions were altered substantially due to operational changes occurring in 1984 (2-unit, 2-cut operation) and 1986 (Unit 3 start-up), and this situation was reflected in the characteristic disturbed or early successional stage flora collected at FE from 1984 to 1987 (Group II). Elevated temperature conditions persisted, but were more consistent in subsequent years comprising collections in Group III. These conditions allowed for more long-term development of the unique flora now observed at FE, characterized by the presence of warm water-tolerant species not typical of other sites (*Agardhiella subulata*, *Gracilaria tikvahiae* and *Sargassum filipendula*), absence of common cold water species (*Mastocarpus stellatus*, *Dumontia contorta* and *Polysiphonia lanosa*) and extended or reduced periods of occurrence of the seasonal species mentioned previously with warm water and cold water affinities, respectively. Similar floristic shifts have been observed by other researchers studying attached algae near thermal effluents (Vadas et al. 1976; Wilce et al. 1978; Schneider 1981).

TABLE 1. Qualitative algal collections (Mar. 1979-Feb. 1993) by month, during 2-unit (3/79-2/86) and 3-unit (3/86-2/93) operating periods. Values represent number of times found, as a percentage of possible times found. Taxa enclosed in quotes are, or may be, conspecific or subspecific forms, or alternate life-history stages (see text for additional explanation); a dash before a species indicates that it was included in collections made in the latest report year.

Rhodophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
<i>Stylonema alsidii</i>	5	2	2	2	0	4	9	12	21	33	7	5	3	5	2	2	0	2	2	8	10	10	3	5
<i>Erythrotrichopeltis ciliaris</i>	32	18	18	12	16	11	11	18	32	49	32	25	24	21	13	17	13	6	8	19	29	46	25	29
<i>Erythrotrichia carnea</i>	4	4	0	0	2	2	0	0	9	4	4	2	2	5	3	11	2	0	5	3	14	14	6	8
<i>Erythrocladia subintegra</i>	0	0	2	0	0	2	0	7	2	5	5	5	2	0	0	2	0	0	0	3	2	3	3	0
<i>Erythrotrichia discigera</i>	7	4	2	0	4	0	2	2	7	5	5	5	2	3	5	2	2	0	2	0	5	10	8	5
<i>Bangia atropurpurea</i>	65	79	77	86	26	11	4	7	18	21	35	54	60	73	95	75	38	17	6	6	30	27	35	44
<i>Porphyra leucosticta</i>	46	68	61	65	44	26	12	9	9	19	18	25	68	78	86	84	75	32	11	6	6	30	32	43
<i>Porphyra umbilicalis</i>	46	53	77	77	93	72	58	40	25	14	23	28	59	57	76	90	83	59	43	37	29	38	43	59
<i>Porphyra linearis</i>	0	0	0	0	0	0	0	0	0	0	0	2	2	2	3	0	0	0	0	0	0	0	0	3
<i>Porphyropsis coccinea</i>	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
<i>Audouinella purpurea</i>	4	4	0	4	2	2	2	4	2	0	5	2	0	0	0	0	2	0	0	0	0	0	0	0
<i>Audouinella secundata</i>	35	53	37	35	35	40	25	23	21	37	18	19	30	30	25	35	25	17	16	11	10	13	8	3
<i>Audouinella daviesii</i>	9	0	2	7	4	4	5	2	2	2	7	0	0	0	3	2	3	0	5	3	0	5	2	0
<i>Audouinella saviana</i>	16	5	14	12	9	11	5	5	19	18	7	11	6	5	8	24	16	8	3	5	17	16	16	8
<i>Audouinella</i> sp.	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
<i>Audouinella dasyae</i>	0	0	0	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	2
<i>Gelidium pusillum</i>	7	9	4	9	7	9	11	9	9	14	12	12	25	22	21	19	19	16	21	22	22	25	33	32
<i>Nemalion helminthoides</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	11	3	0	0	0	0
<i>Bonnemaia hamilera</i>	9	9	18	18	26	53	33	9	0	0	5	5	2	6	6	13	25	43	37	10	3	2	0	5
<i>Trailiella intricata</i>	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
<i>Agardhiella subulata</i>	5	5	5	4	11	14	12	12	18	11	12	12	13	13	10	5	3	10	11	11	14	13	16	16
<i>Polydora rotundus</i>	5	11	0	7	11	4	11	12	14	14	19	12	5	3	3	2	3	6	6	8	3	0	3	5
<i>Cystoclonium purpureum</i>	81	77	68	74	79	79	46	23	12	47	56	68	68	68	57	56	65	65	29	11	14	19	43	48
<i>Gracilaria tikvahiae</i>	0	0	0	0	0	0	0	2	2	0	0	0	10	6	3	3	2	0	0	3	5	3	6	8
<i>Ahnfeltia plicata</i>	47	51	49	49	42	44	37	37	37	46	47	56	22	22	17	14	19	22	21	13	14	13	14	22
<i>Phyllophora pseudoceranoides</i>	23	14	16	16	11	11	12	12	4	19	16	16	8	11	2	3	6	8	3	5	8	6	10	
<i>Phyllophora trillii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
<i>Phyllophora truncata</i>	12	18	7	5	9	12	12	9	11	11	16	19	5	6	14	6	6	10	3	3	2	2	5	3
<i>Chondrus crispus</i>	96	96	98	98	98	98	98	98	96	96	96	96	98	98	97	97	97	97	98	98	98	97	98	98
<i>Mastocarpus stellatus</i>	74	56	53	47	56	58	60	56	58	65	77	65	67	70	65	68	65	70	63	67	68	60	68	68
<i>Petrocelus muldendorffii</i>	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0	0	0	0	0
<i>Rhodophyxa georgii</i>	0	2	7	2	9	2	2	4	4	0	0	0	0	0	3	0	6	3	0	0	0	0	0	0
<i>Corallina officinalis</i>	60	61	58	56	51	51	61	63	60	68	58	67	71	76	57	62	63	68	59	67	60	62	62	76
<i>Dumontia contorta</i>	46	65	81	82	81	47	5	2	2	0	2	7	33	49	65	76	75	32	10	2	0	0	0	6
<i>Glossiphonia capillaris</i>	2	0	2	0	0	0	0	0	6	0	0	0	0	0	0	0	3	2	0	0	0	0	0	0
<i>Choreocolax polysiphoniae</i>	9	12	12	9	9	2	4	4	2	5	2	9	14	11	8	10	6	5	8	3	6	10	5	6
<i>Hildenbrandia rubra</i>	4	2	2	2	0	0	2	0	2	5	0	2	2	0	0	0	0	5	2	0	3	2	0	2
<i>Palmaria palmata</i>	32	33	49	44	39	46	33	28	21	12	28	30	19	27	22	19	22	19	29	21	14	14	10	19
<i>Champia parvula</i>	35	21	11	7	4	4	35	65	74	79	65	46	14	6	8	5	3	8	49	76	79	67	49	38
<i>Lomentaria baileyana</i>	4	0	0	0	0	0	5	30	49	28	7	2	0	0	0	0	0	2	6	19	24	14	3	0
<i>Lomentaria clavellata</i>	11	5	9	16	7	2	2	4	7	4	7	4	5	5	8	8	6	0	2	2	0	0	2	2
<i>Lomentaria orcadensis</i>	2	2	4	0	0	0	0	0	2	5	0	0	2	2	0	0	0	0	0	0	2	0	0	2
<i>Anathamnion cruciatum</i>	47	25	5	18	7	16	46	63	70	74	74	67	24	16	10	11	17	22	43	52	29	33	30	16
<i>Antithamnion nipponicum</i>	0	0	0	0	0	0	0	0	0	0	0	0	73	71	43	25	35	40	51	63	75	70	79	81
<i>Callithamnion corymbosum</i>	0	0	0	0	0	2	0	7	9	5	5	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Callithamnion roseum</i>	7	2	2	0	0	0	9	18	35	18	14	5	0	0	3	2	3	5	6	17	22	8	8	5
<i>Callithamnion tetragonum</i>	65	46	23	33	21	11	25	26	44	46	72	53	10	17	16	17	10	3	5	8	10	13	10	13
<i>Callithamnion byssoides</i>	0	0	0	0	0	0	4	4	2	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0
<i>Callithamnion baileyi</i>	0	0	0	0	0	0	0	0	0	5	7	2	43	29	21	13	8	11	27	25	32	38	52	38
<i>Ceramium deslongchampsii</i>	4	4	0	2	0	2	0	4	4	4	4	9	3	2	0	2	3	0	0	0	2	3	3	2
<i>Ceramium diaphanum</i>	7	0	0	2	0	0	25	68	49	51	11	12	0	0	0	0	2	5	30	57	52	33	14	2
<i>Ceramium rubrum</i>	88	88	74	81	89	91	95	84	95	93	91	88	89	92	75	79	81	89	90	90	89	86	87	87
<i>Ceramium fastigiatum</i>	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Spermothamnion repens</i>	54	33	28	26	18	28	28	35	40	72	74	70	43	29	27	17	24	41	30	48	32	41	40	49
<i>Spyridia filamentosa</i>	0	0	0	0	0	2	2	4	12	4	2	2	0	0	2	0	0	0	0	3	6	10	6	3
<i>Scagelia pylaisaei</i>	5	2	2	4	5	0	0	2	2	4	4	4	0	0	5	3	3	2	2	3	0	0	0	2
<i>Griffithsia globulifera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	2	2	0	0

TABLE 1. (cont.)

Rhodophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Grinnellia americanum	4	0	0	2	2	0	4	9	2	11	12	7	5	0	2	0	0	3	8	6	13	11	16	6
-Phycodrys rubens	2	4	5	12	7	7	4	5	5	7	7	0	0	0	2	6	6	8	3	2	2	3	2	5
-Dasya baillouviana	7	2	0	0	0	0	7	39	30	25	23	11	6	2	0	2	0	2	13	35	27	38	30	27
-Chondria sedifolia	0	0	0	0	0	0	0	4	5	5	0	0	0	0	0	0	0	0	2	5	2	0	0	0
-Chondria baileyana	2	0	2	0	2	0	0	4	16	11	0	5	2	0	0	0	0	0	3	5	11	6	3	2
-Chondria tenuissima	0	0	0	2	0	0	4	2	4	0	0	0	0	0	0	0	0	0	3	5	2	2	0	0
-Chondria dasyphylla	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Polysiphonia denudata	0	0	0	0	0	0	5	0	4	9	5	7	2	2	2	0	5	3	2	3	0	2	0	3
-Polysiphonia harveyi	70	39	25	23	19	53	82	95	91	65	68	63	17	8	6	8	11	27	35	40	38	21	29	17
-Polysiphonia lanosa	82	74	68	60	70	63	61	60	65	72	65	74	73	70	70	65	71	76	70	65	68	67	75	76
-Polysiphonia nigra	5	9	7	11	18	16	2	4	2	4	4	7	2	5	10	8	17	13	0	0	0	3	2	3
-Polysiphonia nigrescens	19	14	18	32	19	28	23	16	18	25	23	21	13	22	5	13	16	19	16	16	16	11	11	17
-Polysiphonia urceolata	19	16	30	54	60	32	18	4	4	2	2	4	14	21	19	24	32	22	5	5	3	2	2	11
-Polysiphonia elongata	0	0	0	0	0	2	0	2	0	0	0	4	2	0	0	0	0	2	0	0	0	0	0	0
-Polysiphonia fibrillosa	5	2	0	0	0	6	2	0	0	2	0	12	3	0	0	0	0	0	0	0	2	2	0	3
-Polysiphonia flexicaulis	0	0	0	0	0	0	0	0	0	0	0	2	13	2	0	0	0	0	0	3	0	8	3	5
-Polysiphonia novae-angliae	70	61	42	30	39	41	68	74	74	88	86	84	94	87	56	40	40	76	90	97	98	97	98	98
-Rhodomela confervoides	7	9	30	19	9	4	0	2	2	0	2	2	2	3	3	6	3	0	0	0	0	0	0	0
Phaeophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Ectocarpus fasciculatus	7	18	12	26	33	35	25	16	28	32	19	9	0	3	16	21	30	32	17	14	22	19	25	11
-Ectocarpus siliculosus	19	32	47	53	60	70	60	39	28	26	23	14	16	32	40	51	44	46	32	30	22	17	29	19
-Ectocarpus sp.	5	12	9	7	0	5	7	4	7	2	5	4	2	0	3	2	0	2	2	2	0	0	0	2
-Giffordia granulosa	5	2	4	4	4	7	2	0	5	2	5	4	2	3	2	5	8	13	3	0	2	0	8	3
-Giffordia mitchelliae	4	7	2	5	14	19	19	32	32	28	14	7	10	8	5	3	14	13	13	32	43	43	22	11
-Pilayella littoralis	21	18	25	35	51	32	12	16	14	21	12	18	11	16	22	27	38	38	6	11	3	19	8	16
-Spongonema tomentosum	18	28	42	30	19	0	2	4	0	7	2	2	3	22	43	25	14	3	2	3	0	6	0	5
-Entonema aecidioides	0	0	0	4	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Acinetospora sp.	0	0	0	4	2	0	0	2	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
-Feldmannia sp.	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0
-Ralfsia verrucosa	42	56	37	39	46	47	58	65	68	67	54	46	65	65	52	59	43	44	67	78	76	63	70	65
-Elachista fucicola	42	51	61	72	86	82	70	81	77	49	35	26	54	59	67	79	75	87	87	81	73	60	59	52
-Halothrix lumbiculis	0	2	0	0	5	4	2	0	2	2	0	0	0	0	3	3	6	2	2	0	0	0	0	0
-Leathesia difformis	0	2	0	9	19	26	26	11	0	0	0	0	0	0	0	16	35	46	37	5	0	0	0	0
-Chordaria flagelliformis	0	0	0	5	19	49	37	30	19	5	2	4	0	0	2	2	11	19	22	13	6	3	0	0
-Sphaerotrichia divaricata	0	0	0	2	5	11	2	0	0	0	0	0	0	0	2	0	3	0	0	0	0	0	0	0
-Cladosiphon zosterae	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Eudesme virescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
-Pogotrichum filiforme	0	14	5	2	5	0	0	0	2	0	0	0	0	3	3	3	0	0	0	0	0	0	0	0
-Desmotrichum undulatum	0	2	14	7	4	2	0	0	0	4	0	4	0	8	6	6	6	5	2	0	0	0	0	0
-Phaeosaccion collinsii	0	4	2	0	0	0	0	0	0	0	0	0	0	2	5	0	0	0	0	0	0	0	0	0
-Punctaria latifolia	2	9	16	12	4	9	0	0	0	5	4	4	3	5	8	3	8	6	2	0	0	0	3	6
-Punctaria plantaginea	2	4	4	2	5	7	7	5	4	0	2	0	0	3	0	2	6	3	3	2	2	2	3	3
-Petalonion fascia	70	84	68	86	84	84	72	9	5	12	42	63	70	90	84	97	92	81	51	8	2	6	38	54
-Scytosiphon lomentaria	46	79	93	95	93	95	86	18	5	9	21	32	27	70	90	98	94	90	65	8	0	2	6	17
-Delamarea attenuata	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Desmarestia aculeata	7	2	9	16	9	12	2	7	4	7	11	4	8	8	5	8	13	8	19	3	5	8	6	5
-Desmarestia viridis	2	4	30	44	49	39	2	0	0	0	2	0	0	5	24	48	54	35	0	2	0	0	0	0
-Chorda filum	0	0	0	2	9	19	16	0	0	0	0	0	0	0	0	2	8	25	17	5	0	0	0	0
-Chorda tomentosa	0	0	5	18	28	2	9	0	0	0	0	0	0	5	16	29	21	6	0	0	0	0	0	0
-Laminaria digitata	2	0	0	0	4	0	0	2	0	0	0	4	0	2	2	5	0	0	0	0	2	2	0	0
-Laminaria longicruris	9	12	9	14	14	11	12	11	12	18	12	7	11	13	11	14	17	16	17	25	17	22	16	13
-Laminaria saccharina	53	37	53	63	82	77	82	75	60	58	49	58	51	38	57	68	87	97	78	75	62	63	59	54
-Sphacelaria cirrosa	30	16	9	12	16	16	19	21	21	32	37	39	33	27	22	17	27	22	24	19	30	37	37	41
-Sphacelaria rigidula	0	0	0	0	0	0	0	0	2	0	0	0	0	2	0	0	2	0	0	2	0	2	3	0
-Ascophyllum nodosum	96	96	98	98	98	98	98	98	96	96	96	96	95	95	95	95	95	95	95	95	95	95	95	95
-Fucus distichus s. edentatus	7	11	18	21	16	4	2	2	0	2	5	4	6	8	13	13	3	3	0	5	6	2	0	3
-Fucus distichus s. evanescens	14	12	21	19	23	7	2	5	0	4	9	0	2	8	13	13	3	2	5	5	2	3	2	2
-Fucus spiralis	2	2	2	9	5	11	7	7	14	7	7	4	10	5	6	11	10	14	11	10	16	13	6	10
-Fucus vesiculosus	96	96	98	98	100	100	100	96	96	96	96	96	100	100	100	100	100	100	100	100	100	100	100	100
-Sargassum filipendula	0	0	0	0	0	0	0	2	0	0	0	0	8	8	6	6	6	6	6	8	8	8	8	8



TABLE 1. (cont.)

Chlorophyta	2-Unit Operation												3-Unit Operation											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
-Ulothrix flacca	53	61	74	70	47	7	5	0	2	9	23	32	51	70	70	70	29	8	6	2	5	8	16	30
-Urospora penicilliformis	61	72	74	70	30	7	2	4	2	7	28	44	60	57	62	46	25	8	10	8	8	16	30	48
-Urospora wormskjoldii	9	7	9	16	2	7	4	0	0	0	0	2	8	6	17	6	11	5	8	6	5	10	5	16
-Urospora collabens	7	19	9	2	5	2	2	0	0	0	0	4	2	2	2	0	3	0	2	0	0	0	0	0
-Acrochaete viridis	2	2	0	2	0	0	0	0	0	2	2	2	0	0	0	0	2	2	2	0	0	0	0	0
-Monostroma grevillei	25	58	54	60	51	12	2	2	2	0	7	5	3	51	54	73	46	2	0	0	0	0	0	3
-Monostroma pulchrum	19	44	88	91	86	16	4	0	0	4	2	2	0	24	76	79	71	5	0	0	0	0	0	0
-Monostroma oxysperma	2	0	0	2	2	0	0	0	0	0	0	0	0	2	3	6	6	2	0	0	0	0	0	0
-Spongomorpha arcata	7	18	39	51	54	32	5	0	0	0	4	4	5	17	24	30	46	22	5	0	0	0	0	2
-Spongomorpha aeruginosa	4	2	7	14	18	16	5	4	4	0	0	0	2	0	2	21	33	25	3	0	0	2	0	0
-Codiolum gregarium	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Capsosiphon fulvescens	0	2	0	5	4	4	2	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
-Capsosiphon groenlandicum	2	12	12	14	11	7	2	0	0	0	2	11	3	2	2	2	2	2	0	0	2	0	0	2
-Blidingia minima	58	44	44	53	70	65	47	67	58	49	47	53	65	78	65	75	79	81	73	81	78	62	71	67
-Blidingia marginata	9	4	0	0	2	0	4	0	0	4	2	2	0	0	0	0	2	2	3	0	0	0	0	0
-Enteromorpha clathrata	4	5	2	12	18	19	37	46	37	28	5	4	3	0	3	5	8	10	25	37	41	16	3	2
-Enteromorpha flexuosa	33	33	25	28	37	40	40	32	46	60	53	35	52	51	59	40	51	38	44	51	70	56	63	60
-Enteromorpha intestinalis	25	26	35	44	49	47	46	51	35	25	21	16	17	17	25	40	32	37	46	35	30	29	11	14
-Enteromorpha liza	51	30	32	56	67	63	65	60	60	70	63	49	48	49	54	76	76	70	68	76	73	65	56	
-Enteromorpha prolifera	42	39	33	28	35	37	25	33	32	47	44	47	14	14	6	16	16	21	10	14	10	22	17	24
-Enteromorpha torta	2	0	0	0	4	7	5	7	2	5	2	0	0	0	0	0	2	5	3	0	0	0	0	0
-Enteromorpha ralfsii	0	0	0	0	0	11	7	4	5	2	2	0	0	2	2	0	2	5	6	6	3	3	0	0
-Percursaria percursa	2	0	0	5	4	9	4	2	2	2	0	0	0	0	2	0	0	0	0	0	2	2	0	0
-Ulva lactuca	96	83	77	84	89	98	93	91	96	98	96	96	94	94	76	83	89	97	97	98	100	95	95	94
-Prasiola stipitata	19	21	23	23	23	28	25	25	25	19	30		37	35	24	27	30	32	35	29	32	27	29	32
-Chaetomorpha linum	79	58	40	37	56	89	91	95	96	95	84	74	56	33	19	21	48	70	90	90	86	81	70	54
-Chaetomorpha melagonium	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
-Chaetomorpha aerea	30	25	26	21	19	26	37	28	42	30	37	30	44	46	27	41	46	40	51	51	38	48	46	44
-Cladophora albida	0	0	2	2	11	9	16	11	12	4	4	2	0	0	0	3	0	3	0	5	5	2	0	0
-Cladophora flexuosa	14	2	4	7	14	25	37	26	18	28	12	11	5	2	2	5	24	59	78	40	44	33	14	8
-Cladophora glaucescens	0	0	0	0	2	2	2	2	0	0	0	0	0	0	0	2	0	0	2	2	0	0	0	0
-Cladophora laevigata	0	0	0	2	2	4	4	0	4	2	0	0	0	0	0	0	0	0	2	0	0	0	0	0
-Cladophora refracta	9	7	2	2	5	32	33	26	28	37	18	16	3	5	5	2	5	11	3	3	2	3	2	6
-Cladophora sericea	12	5	5	25	53	42	35	37	23	21	18	14	11	2	14	24	35	27	35	35	24	22	14	22
-Cladophora crystallina	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	5	3	0	0	0	0
-Cladophora hutchinsiae	2	4	4	2	9	9	7	12	12	14	11	2	3	3	2	3	5	11	10	6	6	5	5	2
-Cladophora rupestris	0	2	0	2	5	2	9	7	0	2	2	0	2	2	2	0	10	10	6	5	3	2	3	0
-Cladophora ruchingeri	0	0	0	0	0	0	0	2	0	0	0	0	0	0	2	0	5	6	14	22	11	13	10	0
-Rhizoclonium riparium	9	30	25	16	18	30	30	30	28	23	12	16	8	0	10	5	8	25	24	27	13	14	3	8
-Rhizoclonium kernerii	2	0	4	0	0	4	0	0	0	0	0	2	0	0	2	0	2	2	3	2	0	0	0	0
-Rhizoclonium tortuosum	0	0	0	0	0	5	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
-Bryopsis plumosa	7	0	0	0	4	2	12	16	11	12	7	7	6	3	0	0	0	2	13	6	11	3	6	8
-Bryopsis hypnoides	0	0	0	0	2	5	9	2	5	7	2	4	2	0	2	0	6	13	22	17	3	8	10	5
-Derbesia marina	7	5	4	9	0	0	2	4	4	9	12	11	0	0	0	0	2	0	2	2	2	5	0	2
-Codium fragile	89	79	68	75	75	82	81	84	88	95	89	86	94	92	73	75	81	89	95	97	95	90	89	90

TABLE 2. Qualitative algal collections (Mar. 1979-Feb. 1993) by station, during 2-unit (3/79-2/86) and 3-unit (3/86-2/93) operating periods. Taxa enclosed in quotes are, or may be, conspecific or subspecific forms, or alternate life-history stages, see text for additional explanation. Values represent number of times found, as a percentage of possible times found. The last three columns represent 2-unit, 3-unit, and overall study summaries ("T" = present, <1%).

Rhodophyta	2-Unit Operation										3-Unit Operation										Summaries		
	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot		
<i>Stylonema alsidii</i>	14	11	6	2	12	11	10	1	6	13	4	0	1	10	2	7	0	0	8	4	6		
<i>Erythrotrichopeltis ciliaris</i>	32	20	19	23	26	20	27	11	23	40	14	8	12	46	19	29	4	14	23	21	22		
<i>Erythrotrichia carnea</i>	4	2	0	2	1	4	4	4	0	12	1	6	1	17	10	5	1	2	2	6	4		
<i>Erythrocladia subintegra</i>	1	4	4	2	4	0	0	1	2	0	0	2	0	8	0	0	0	0	2	1	2		
<i>Erythropeltis discigera</i>	4	6	6	0	5	2	1	2	5	0	7	2	2	10	0	2	4	4	4	3	3		
<i>Bangia atropurpurea</i>	48	45	54	35	42	30	49	38	25	50	50	46	40	42	32	48	43	30	40	42	41		
<i>Porphyra leucosticta</i>	42	27	56	44	32	23	37	31	24	50	43	62	46	51	49	40	42	30	33	46	40		
<i>Porphyra umbilicalis</i>	64	42	50	54	54	42	52	64	33	42	50	73	50	76	49	54	74	37	50	56	53		
<i>Porphyra linearis</i>	0	0	0	0	0	0	1	0	0	0	0	2	1	0	0	1	2	0	T	1			
<i>Porphyropsis coccinea</i>	0	0	0	0	0	2	0	0	0	0	0	1	0	0	1	0	0	0	T	T	T		
<i>Audouinella purpurea</i>	2	0	0	0	14	0	1	0	1	0	0	1	0	0	0	0	0	0	2	T	1		
<i>Audouinella secundata</i>	39	49	44	23	24	24	30	27	25	24	32	27	8	15	10	10	24	18	31	19	25		
<i>Audouinella daviesii</i>	5	4	2	6	2	2	4	4	4	2	2	1	0	2	1	2	2	2	4	2	3		
<i>Audouinella saviana</i>	17	12	19	15	12	13	8	2	6	17	8	5	7	18	13	19	2	10	11	11	11		
<i>Audouinella sp.</i>	1	1	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	T	T	T		
<i>Audouinella dasyae</i>	0	2	0	0	0	0	0	0	0	0	0	0	0	0	2	1	0	0	T	T	T		
<i>Gelidium pusillum</i>	7	0	0	4	1	63	1	0	0	71	0	1	0	24	100	10	1	1	9	23	17		
<i>Nemalion helminthoides</i>	0	0	0	0	0	0	0	0	0	0	7	4	2	0	0	0	0	0	0	1	1		
<i>Bonnemaisonia hamifera</i>	1	20	2	33	0	1	20	29	33	1	2	2	33	0	1	14	21	37	15	13	14		
<i>'Trailliella intricata'</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	T	T		
<i>Agardhiella subulata</i>	5	7	8	2	24	8	17	4	12	0	2	1	0	80	6	6	1	4	10	11	11		
<i>Polyides rotundus</i>	4	8	8	8	5	8	18	2	26	1	0	5	8	6	0	5	1	10	10	4	7		
<i>Cystoclonium purpureum</i>	58	50	67	65	58	58	71	49	62	61	24	52	68	7	32	54	52	57	59	45	52		
<i>Gracilaria tikvahiae</i>	0	1	0	0	1	0	0	0	0	0	0	1	0	33	1	1	0	0	1	4	2		
<i>Ahnfeltia plicata</i>	20	39	58	92	73	19	52	24	55	8	7	31	42	1	6	33	14	18	45	18	31		
<i>Phyllophora pseudoceranoides</i>	3	7	10	29	10	8	40	7	11	5	1	5	11	2	1	17	4	10	14	6	10		
<i>Phyllophora trillii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1			
<i>Phyllophora truncata</i>	11	11	13	10	11	11	21	5	13	4	2	2	18	1	1	12	4	5	12	5	8		
<i>Chondrus crispus</i>	100	100	100	100	79	100	100	100	100	100	100	100	100	80	100	100	100	100	97	98	98		
<i>Mastocarpus stellatus</i>	25	61	90	100	17	21	65	96	98	29	43	95	100	0	43	100	95	95	60	67	64		
<i>Petrocelis middendorffii</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	T	T	T		
<i>Rhodophysema georgii</i>	0	1	0	0	0	2	4	1	12	0	1	0	1	0	1	0	1	5	2	1	2		
<i>Corallina officinalis</i>	2	100	100	31	95	70	82	32	27	1	98	99	31	94	100	94	54	18	60	65	63		
<i>Dumontia contorta</i>	39	19	40	48	26	45	33	32	39	39	10	31	50	0	42	25	19	45	35	29	32		
<i>Glossiphonia capillaris</i>	0	0	2	0	1	0	0	0	0	0	0	0	0	1	0	2	0	0	T	T	T		
<i>Chortocolax polysiphoniae</i>	14	21	4	2	2	0	5	0	6	11	26	14	4	0	0	2	0	12	6	8	7		
<i>Hildenbrandia rubra</i>	1	0	0	15	0	1	1	0	1	0	0	0	8	0	0	1	0	1	2	1	1		
<i>Palmaria palmata</i>	36	39	17	69	8	8	44	25	58	15	12	6	58	2	2	23	15	42	33	20	26		
<i>Champia parvula</i>	37	35	15	33	31	38	61	26	46	30	25	27	36	42	35	42	26	40	37	34	35		
<i>Lomentaria baileyana</i>	21	7	2	6	17	13	17	2	2	11	0	0	0	12	14	10	1	4	10	6	8		
<i>Lomentaria clavellosa</i>	5	5	0	10	2	5	10	7	12	1	0	5	4	0	0	10	1	8	6	3	5		
<i>Lomentaria orcadensis</i>	1	0	0	8	0	0	2	0	1	0	0	0	1	0	0	2	0	1	1	1	1		
<i>Antithamnion cruciatum</i>	44	52	33	40	33	40	62	30	43	19	27	29	27	25	23	33	18	26	43	25	33		
<i>Antithamnion nipponicum</i>	0	0	0	0	0	0	0	0	0	38	69	85	65	68	52	61	46	45	0	59	31		
<i>Callithamnion corymbosum</i>	0	7	0	2	2	1	5	1	1	0	0	0	0	0	0	0	0	0	2	0	1		
<i>Callithamnion roseum</i>	10	6	10	4	18	11	10	4	8	4	2	4	7	29	5	6	1	2	9	7	8		
<i>Callithamnion tetragonum</i>	38	37	52	63	50	25	45	23	31	2	21	20	18	1	8	17	0	10	39	11	24		
<i>Callithamnion byssoides</i>	1	5	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	T	T		
<i>'Callithamnion baileyi'</i>	2	2	2	2	0	0	1	0	1	25	49	45	39	1	21	37	18	17	1	28	15		
<i>Ceramium deslongchampsii</i>	13	1	0	4	0	4	1	1	0	2	4	1	1	0	0	2	2	1	3	2	2		
<i>Ceramium diaphanum</i>	18	29	2	33	8	13	26	19	19	8	20	5	26	4	5	29	24	26	19	16	17		
<i>Ceramium rubrum</i>	88	95	88	92	87	77	94	87	86	85	99	76	89	76	87	96	83	85	88	86	87		
<i>Ceramium fastigiatum</i>	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	T		
<i>Spermothamnion repens</i>	46	58	40	46	21	27	68	26	48	44	46	19	27	11	25	60	38	45	42	35	38		
<i>Spyridia filamentosa</i>	12	1	0	0	0	4	1	0	0	15	2	1	1	0	1	1	0	0	2	3	2		
<i>Scaglia pylaisaei</i>	1	4	6	2	0	1	6	1	4	0	2	0	1	1	1	5	2	3	3	2	2		
<i>Griffithsia globulifera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	2	0	1	T		

TABLE 2. (cont.)

	2-Unit Operation										3-Unit Operation										Summaries									
	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	GN	BP	MP	TT	FE	FS	WP	SE	SS	2U	3U	tot
<b>Rhodophyta</b>																														
<i>Grinnellia americana</i>	1	1	2	2	1	1	13	2	12	0	0	0	4	23	2	10	2	12	0	0	0	4	23	2	10	2	12	4	6	5
<i>Phycodrys rubens</i>	0	0	2	8	1	2	18	6	11	0	1	0	10	0	0	10	1	7	0	1	0	10	0	0	10	1	7	5	3	4
<i>Dasya baillouviana</i>	8	14	2	10	11	21	18	4	13	19	13	10	6	45	13	21	2	6	19	13	10	6	45	13	21	2	6	12	15	14
<i>Chondria sedifolia</i>	5	1	0	2	0	1	0	0	1	4	0	0	0	0	2	0	0	0	4	0	0	0	0	0	2	0	0	1	1	1
<i>Chondria baileyana</i>	5	5	13	4	0	4	4	0	1	10	1	4	1	1	6	0	0	1	10	1	4	1	1	6	0	0	1	3	3	3
<i>Chondria tenuissima</i>	5	0	0	0	1	1	0	0	0	1	0	0	0	1	6	0	0	0	1	0	0	0	1	6	0	0	0	1	1	1
<i>Chondria dasyphylla</i>	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	0	0	0	0
<i>Polysiphonia denudata</i>	0	4	0	2	5	2	6	1	1	0	5	1	1	5	0	2	1	1	0	5	1	1	5	0	2	1	1	2	2	2
<i>Polysiphonia harveyi</i>	58	63	50	50	70	54	60	51	57	11	20	29	19	39	21	20	14	19	11	20	29	19	39	21	20	14	19	58	21	39
<i>Polysiphonia lanosa</i>	86	86	100	44	46	27	81	50	94	80	99	100	39	0	40	98	81	98	80	99	100	39	0	40	98	81	98	68	71	69
<i>Polysiphonia nigra</i>	1	17	0	2	4	7	12	6	11	11	5	1	1	4	1	15	6	2	11	5	1	1	4	1	15	6	2	7	5	6
<i>Polysiphonia nigrescens</i>	19	23	4	15	15	20	55	10	20	25	17	1	4	11	4	49	7	14	25	17	1	4	11	4	49	7	14	21	15	18
<i>Polysiphonia urceolata</i>	42	20	23	27	24	5	29	4	13	26	18	24	7	0	0	36	6	2	26	18	24	7	0	0	36	6	2	20	13	17
<i>Polysiphonia elongata</i>	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	1	1	1
<i>Polysiphonia fibrillosa</i>	5	1	0	2	4	0	2	1	1	0	1	1	1	1	0	0	2	0	0	1	1	1	1	1	0	0	2	2	1	1
<i>Polysiphonia flexicaulis</i>	0	0	2	0	0	0	0	0	0	2	7	1	2	1	0	5	6	0	2	7	1	2	1	0	5	6	0	1	3	2
<i>Polysiphonia novae-angliae</i>	60	62	73	58	70	60	68	57	64	80	85	81	77	99	75	87	70	75	80	85	81	77	99	75	87	70	75	63	81	73
<i>Rhodomela confervoides</i>	14	8	2	10	5	2	6	5	10	1	2	0	2	0	0	1	2	4	1	2	0	2	0	0	1	2	4	7	1	4
<b>Phaeophyta</b>																														
<i>Ectocarpus fasciculatus</i>	17	20	48	33	27	10	20	20	15	17	27	10	39	5	7	21	17	15	17	27	10	39	5	7	21	17	15	22	18	20
<i>Ectocarpus siliculosus</i>	49	38	25	50	44	37	45	35	29	48	31	26	36	24	23	45	32	19	48	31	26	36	24	23	45	32	19	39	31	35
<i>Ectocarpus</i> sp.	2	6	17	13	2	4	2	4	8	2	1	2	1	0	0	0	1	1	2	1	2	1	0	0	0	1	1	6	1	3
<i>Giffordia granulosa</i>	2	4	8	6	5	2	2	1	4	5	1	2	8	10	1	6	2	0	5	1	2	8	10	1	6	2	0	4	4	4
<i>Giffordia mitchelliae</i>	18	13	6	8	32	15	25	6	6	23	21	12	17	56	11	20	1	1	23	21	12	17	56	11	20	1	1	15	18	17
<i>Pilayella littoralis</i>	65	7	6	21	6	58	20	7	6	40	5	1	26	5	51	20	5	8	40	5	1	26	5	51	20	5	8	23	18	20
<i>Spongonema tomentosum</i>	17	20	10	17	12	5	11	13	11	13	8	11	42	13	7	12	12	7	13	8	11	42	13	7	12	12	7	13	11	12
<i>Entonema acidoides</i>	0	2	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	0	1
<i>Acinetospora</i> sp.	0	1	0	0	0	2	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	1	1	1
<i>Feldmannia</i> sp.	0	1	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	1	0	1
<i>Ralfsia verrucosa</i>	76	58	42	15	61	60	68	27	42	73	81	67	69	11	70	79	61	51	73	81	67	69	11	70	79	61	51	52	62	57
<i>Elachista fucicola</i>	70	62	69	58	61	45	61	73	54	76	76	71	70	58	57	73	80	63	76	76	71	70	58	57	73	80	63	61	69	65
<i>Halothrix lumbricalis</i>	1	4	0	0	0	0	4	0	2	2	4	2	4	0	0	0	0	0	2	4	2	4	0	0	0	0	0	1	1	1
<i>Leathesia difformis</i>	12	1	27	4	14	1	12	1	4	25	8	25	12	5	11	15	1	1	25	8	25	12	5	11	15	1	1	8	12	10
<i>Chordaria flagelliformis</i>	15	23	27	15	5	2	27	12	7	2	25	12	10	1	0	6	1	1	2	25	12	10	1	0	6	1	1	14	6	10
<i>Sphaerotrichia divaricata</i>	0	1	2	4	0	0	4	4	1	0	1	1	1	0	0	0	0	0	0	1	1	1	1	0	0	0	0	2	1	1
<i>Cladosiphon zosterae</i>	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	0	1	0	1
<i>Eudesme virescens</i>	0	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	0	0	1	1
<i>Pogotrichum filiforme</i>	4	1	4	4	0	2	5	1	1	0	1	0	2	1	0	1	0	1	0	1	0	2	1	0	1	0	1	2	1	2
<i>Desmotrichum undulatum</i>	7	6	0	0	0	2	5	0	4	5	4	0	8	5	1	1	0	1	5	4	0	8	5	1	1	0	1	3	1	3
<i>Phaeosaccion collinsii</i>	1	0	0	0	0	1	1	0	0	0	0	1	1	0	0	1	0	1	0	0	1	1	0	0	1	0	1	1	1	1
<i>Punctaria latifolia</i>	5	5	2	10	2	11	5	5	4	2	5	1	6	5	4	4	4	4	2	5	1	6	5	4	4	4	4	5	4	4
<i>Punctaria plantaginea</i>	5	4	4	0	0	8	7	0	1	11	2	0	1	0	6	1	0	0	11	2	0	1	0	6	1	0	0	3	2	3
<i>Petalonia fascia</i>	54	74	63	46	54	56	70	55	38	61	68	60	64	48	57	61	50	37	61	68	60	64	48	57	61	50	37	57	56	56
<i>Scytosiphon lomentaria</i>	64	74	48	48	49	64	60	49	40	50	58	55	45	39	55	52	42	30	50	58	55	45	39	55	52	42	30	56	47	51
<i>Delamarea attenuata</i>	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	0	1	0	1
<i>Desmarestia aculeata</i>	2	1	4	13	5	10	17	5	11	12	1	1	21	0	0	15	11	10	12	1	1	21	0	0	15	11	10	7	8	8
<i>Desmarestia viridis</i>	12	17	17	25	8	11	17	8	19	13	7	13	26	5	11	19	13	18	13	7	13	26	5	11	19	13	18	14	14	14
<i>Chorda filum</i>	2	1	0	13	1	1	11	4	4	1	2	1	14	0	0	12	5	7	1	2	1	14	0	0	12	5	7	4	5	4
<i>Chorda tomentosa</i>	0	4	2	23	2	4	6	4	8	2	5	2	27	0	1	5	7	7	2	5	2	27	0	1	5	7	7	5	6	6
<i>Laminaria digitata</i>	0	0	0	13	0	0	0	0	0	0	0	0	7	0	0	0	0	1	0	0	0	7	0	0	0	0	0	1	1	1
<i>Laminaria longicurvis</i>	11	4	13	42	1	1	19	14	14	7	0	7	62	0	5	25	18	21	7	0	7	62	0	5	25	18	21	12	16	14
<i>Laminaria saccharina</i>	62	69	63	96	60	50	61	62	54	71	76	70	88	39	48	75	73	51	71	76	70	88	39	48	75	73	51	62	66	64
<i>Sphacelaria cirrosa</i>	49	25	10	6	49	27	18	0	4	65	17	7	0	75	52	25	8	2	65	17	7	0	75	52	25	8	2	22	28	25
<i>Sphacelaria rigidula</i>	0	0	0	0	0	0	1	0	0	2	0	1	0	0	2	0	1	0	2	0	1	0	0	2	0	1	0	1	1	1
<i>Ascophyllum nodosum</i>	100	100	100	100	79	100	100	100	100	100	100	100	100	57	100	100	100	100	100	100	100	100	57	100	100	100	100	97	95	96
<i>Fucus distichus s. edentatus</i>	6	7	13	23	17	6	2	7	1	2	5	10	17	1	0	4	6													

TABLE 2. (cont.)

Chlorophyta	2-Unit Operation										3-Unit Operation										Summaries		
	GN	BP	MP	TT	FE	FS	WP	SE	SS		GN	BP	MP	TT	FE	FS	WP	SE	SS		2U	3U	tot
<i>Ulothrix flacca</i>	37	37	27	17	32	29	45	31	24		40	32	31	36	18	31	32	26	26		32	30	31
<i>Urospora penicilliformis</i>	38	42	40	33	31	21	43	31	24		35	45	36	27	29	33	31	31	17		33	31	32
<i>Urospora wormskjoldii</i>	5	8	6	0	7	4	7	2	0		12	5	6	2	24	8	13	4	4		5	9	7
' <i>Urospora collabens</i> '	6	6	6	4	4	1	2	2	6		0	4	0	0	2	0	1	0	0		4	1	2
<i>Acrochaete viridis</i>	0	2	6	0	0	1	0	0	0		1	1	0	0	0	0	1	0	0		1	T	1
<i>Monostroma grevillei</i>	26	23	19	19	23	32	23	15	25		20	21	18	27	1	19	19	26	21		23	19	21
<i>Monostroma pulchrum</i>	32	33	25	29	24	23	33	33	31		20	19	23	30	5	24	26	26	25		30	21	25
<i>Monostroma oxysperma</i>	0	1	0	0	0	0	0	0	2		0	1	1	1	0	0	1	2	7		T	2	1
<i>Spongomorpha arcta</i>	29	27	31	25	18	4	18	10	7		19	12	32	18	5	6	12	7	2		18	13	15
<i>Spongomorpha aeruginosa</i>	10	5	15	0	6	6	8	5	1		8	14	17	5	1	2	7	6	5		6	7	7
' <i>Codiolum gregarium</i> '	1	0	0	0	0	0	0	0	0		0	0	0	0	0	0	0	0	0		T	0	T
<i>Capsosiphon fulvescens</i>	1	1	0	0	0	2	6	0	2		0	0	0	0	0	0	0	0	0		2	0	1
<i>Capsosiphon groenlandicum</i>	6	20	8	2	2	2	5	2	5		0	1	0	2	1	2	1	1	1		6	1	3
<i>Blidingia minima</i>	61	55	71	71	70	8	64	43	62		67	68	89	83	81	49	76	60	83		55	73	64
<i>Blidingia marginata</i>	1	2	0	2	4	1	4	1	2		4	1	0	0	0	0	0	0	0		2	1	1
<i>Enteromorpha clathrata</i>	27	4	10	4	25	33	33	1	14		18	14	6	0	14	21	23	4	14		18	13	15
<i>Enteromorpha flexuosa</i>	38	40	35	27	63	37	54	12	33		63	50	58	42	85	40	60	30	49		38	53	46
<i>Enteromorpha intestinalis</i>	52	39	40	19	32	27	55	15	30		38	36	30	15	19	20	39	18	35		35	28	31
<i>Enteromorpha linza</i>	50	74	75	40	71	35	67	48	42		68	85	86	55	83	60	73	50	32		55	66	61
<i>Enteromorpha prolifera</i>	42	37	35	40	35	35	54	15	40		21	12	10	10	15	15	23	12	20		37	15	26
<i>Enteromorpha torta</i>	5	1	2	0	1	6	6	0	2		1	0	0	1	1	1	2	0	0		3	1	2
<i>Enteromorpha ralfsii</i>	4	1	0	0	4	8	4	0	0		5	0	1	2	1	7	2	2	0		2	2	2
<i>Percursaria percura</i>	4	1	0	0	1	8	4	0	1		0	1	0	0	0	0	2	0	0		2	T	1
<i>Ulva lactuca</i>	96	90	96	94	96	89	98	86	86		93	96	95	90	90	92	95	93	88		92	93	92
<i>Prasiola stipitata</i>	51	1	4	90	1	1	0	75	8		61	0	0	92	1	0	1	76	44		24	31	27
<i>Chaetomorpha linum</i>	56	81	88	79	69	68	75	79	85		62	60	57	71	13	52	67	80	76		75	60	67
<i>Chaetomorpha melagonium</i>	1	0	0	0	0	0	0	0	0		1	0	0	0	0	0	0	0	0		T	T	T
<i>Chaetomorpha aerea</i>	23	26	29	2	74	40	45	2	10		55	38	65	8	71	52	60	7	35		29	44	37
<i>Cladophora albida</i>	5	4	4	0	6	8	14	2	6		1	1	1	0	2	5	2	0	0		6	1	4
' <i>Cladophora flexuosa</i> '	8	26	31	17	13	11	19	17	12		19	32	29	18	19	29	39	23	27		16	26	21
' <i>Cladophora glaucescens</i> '	0	0	2	0	0	0	4	0	0		0	0	0	0	0	2	1	0	0		1	T	T
<i>Cladophora laetevirens</i>	0	0	2	0	0	1	2	1	5		0	0	0	0	0	0	0	0	1		1	T	1
' <i>Cladophora refracta</i> '	8	37	23	8	18	12	24	17	12		1	6	6	2	6	5	4	6	1		18	4	11
<i>Cladophora sericea</i>	24	20	13	2	40	32	40	11	20		35	17	14	5	25	43	33	5	23		24	22	23
' <i>Cladophora crystallina</i> '	0	0	0	0	0	0	0	2	0		1	0	0	0	1	1	2	0	0		T	1	T
<i>Cladophora hutchinsiae</i>	5	1	10	2	13	7	12	2	11		10	4	10	1	5	6	4	2	5		7	5	6
<i>Cladophora rupestris</i>	2	2	2	4	2	4	4	0	2		1	2	4	2	8	5	5	1	4		2	4	3
<i>Cladophora ruchingeri</i>	0	0	0	0	1	0	0	0	0		4	5	4	0	19	18	11	1	1		7	7	4
<i>Rhizoclonium riparium</i>	29	19	8	6	19	49	29	10	10		31	10	8	4	5	20	20	8	2		21	12	16
' <i>Rhizoclonium kernerii</i> '	2	1	0	0	0	1	0	1	1		4	0	0	0	0	1	1	1	0		1	1	1
' <i>Rhizoclonium tortuosum</i> '	1	0	0	0	2	0	1	0	0		0	0	0	0	0	0	0	0	0		1	0	T
<i>Bryopsis plumosa</i>	7	5	0	6	19	4	5	6	4		2	2	2	6	21	2	4	0	4		6	5	6
<i>Bryopsis hypnoides</i>	2	4	4	0	4	6	1	1	4		10	2	2	8	17	7	10	5	5		3	7	5
<i>Dertbesia marina</i>	6	2	0	2	25	7	0	0	2		1	0	0	0	5	0	1	0	2		5	1	3
<i>Codium fragile</i>	81	85	83	100	100	87	81	64	69		92	81	86	88	100	98	96	75	80		83	88	86

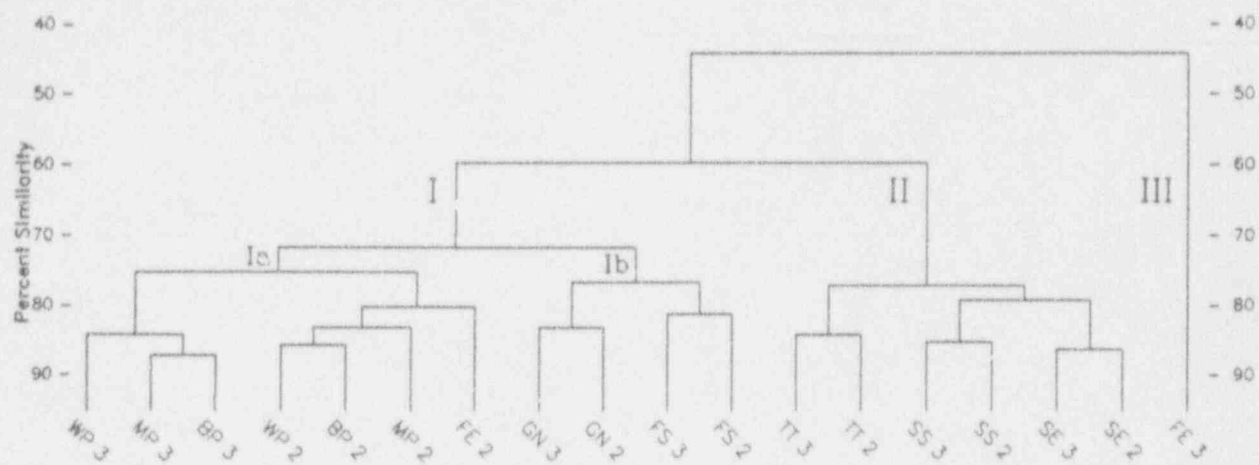


Fig. 3a. Clustering dendrogram of percent similarity of qualitative algal collections, by station and operational period (2-unit + 3-unit).

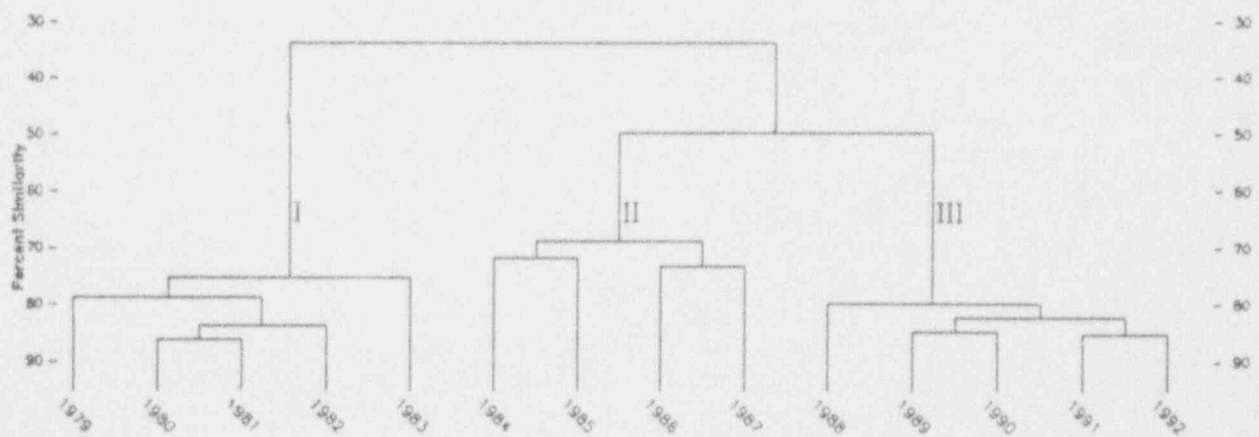


Fig. 3b. Clustering dendrogram of percent similarity of qualitative collections, by year, at Fox Island-Exposed.



## Abundance Measurement

A variety of interacting physical and biological processes influence abundance and distribution of rocky shore organisms. In the local intertidal zone, these include tidal fluctuation, degree of exposure to wave energy, annual cycles in many physical parameters (e.g., light, nutrients, air and water temperatures), predation, competition, reproduction, propagule transport and behavior, and recruitment. Much research has been conducted describing the effect of these processes on community organization, both singly and, to some degree, in combination, as reviewed in NUSCO (1993). The purpose of these quantitative studies was to sample organism abundance over an area sufficiently large to accurately describe large scale patterns of abundance at each sampling site, to relate these patterns to site-specific physical and biological controlling mechanisms and, of primary concern, to determine if any of these mechanisms result from, or are influenced by, the operation of MNPS. Subsections describing abundance patterns of important intertidal organisms, i.e., barnacles, *Fucus*, and *Chondrus*, are included below.

### Barnacles

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

Barnacle and predatory snail abundance patterns at eight sites in the Millstone area are presented in Figure 4 as time-series covering the fifteen year study period (1979-93). At all sites, and in all three zones, the annual abundance cycle was characterized by minimum coverage in late winter/early spring and maximum in summer. Maximum barnacle cover in the high intertidal (Zone 1) during 1992-93 ranged from 4% (GN) to 45% (FE). Minimum coverage in Zone 1 ranged from 1% at FS and GN to 22% at FE. In the mid intertidal (Zone 2), maximum barnacle cover was

lowest at FE (24%) and highest at GN (77%); minimum cover was lowest at FE (12%) and highest at GN and FS (43%). Low intertidal (Zone 3) maxima during 1992-93 ranged from 1% (FE) to 38% (GN); minima were lowest at FE and MP (0%), and highest at GN and SE (6%).

Relationships among barnacle abundance patterns at NUSCO study sites (excluding FE) have been remarkably consistent over the study period, including the present sampling year. Variability among study populations has been considerable and primarily attributed to natural factors. Degree of site exposure to wind and waves and slope of available substratum appear to be the most important controlling mechanisms and their effects on local barnacle population dynamics are described in detail in NUSCO (1993).

These natural controlling mechanisms play an important role in community organization at our study site nearest the discharge (FE) as well; however, the effect of periodic thermal plume incursion is superimposed on these natural mechanisms and has resulted in significant impacts to the shore biota there, including barnacles. Due to the influence of tides, these impacts are most notable in the low intertidal (Zone 3). Zone 3 barnacles are exposed to elevated discharge temperatures for 9-10 hours each tidal cycle during 3-unit operation, whereas barnacles in Zones 1 and 2 are exposed to air during times of maximum 3-unit thermal plume incursion. These conditions have directly and indirectly modified the pattern of barnacle abundance in Zone 3 at FE. Elevated temperatures directly impacted low intertidal barnacles by causing complete population mortality in late summer every year since Unit 3 start-up, including the present study year. Reduced early summer maxima is an indirect effect of 3-unit conditions and resulted from the development of an extensive low intertidal *Codium fragile* population at FE, which persists to the present. As with *Chondrus* at other sites, *Codium* and associated silt accumulated on adjacent areas exclude barnacles and other species from Zone 3 through preemption of habitat space (Underwood and Denley 1984; NUSCO 1993). The return of predatory snails *Urosalpinx cinerea* and *Anachis lafresnayi* in recent years to pre-1984 levels (Fig. 4), coupled with other impacts mentioned above, may have also contributed to reduced barnacle abundances observed in Zone 3 at FE.

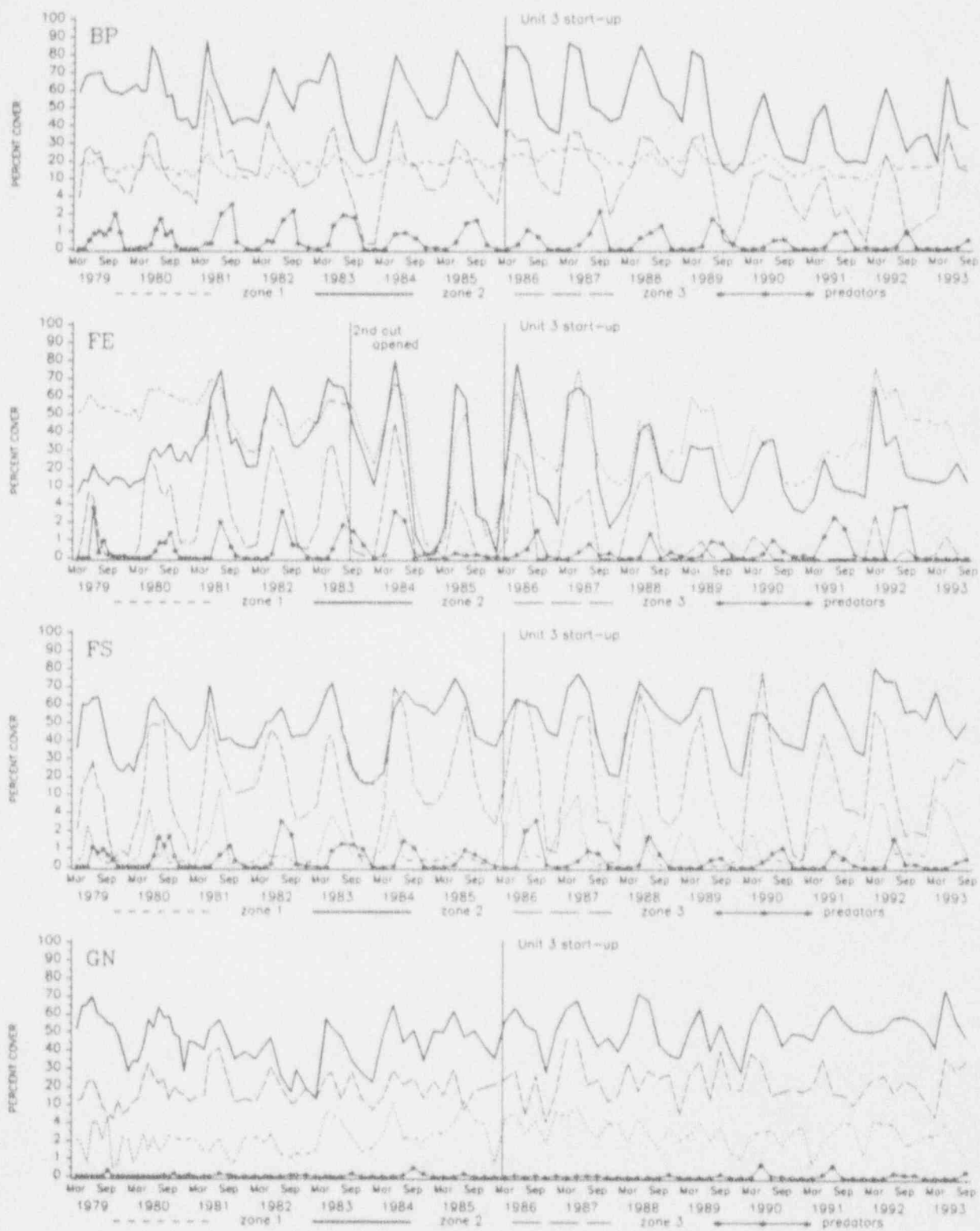


Fig. 4. Abundance of barnacles in each zone, and of predatory snails in Zone 3, of undisturbed transects, from 3/79 - 9/93.

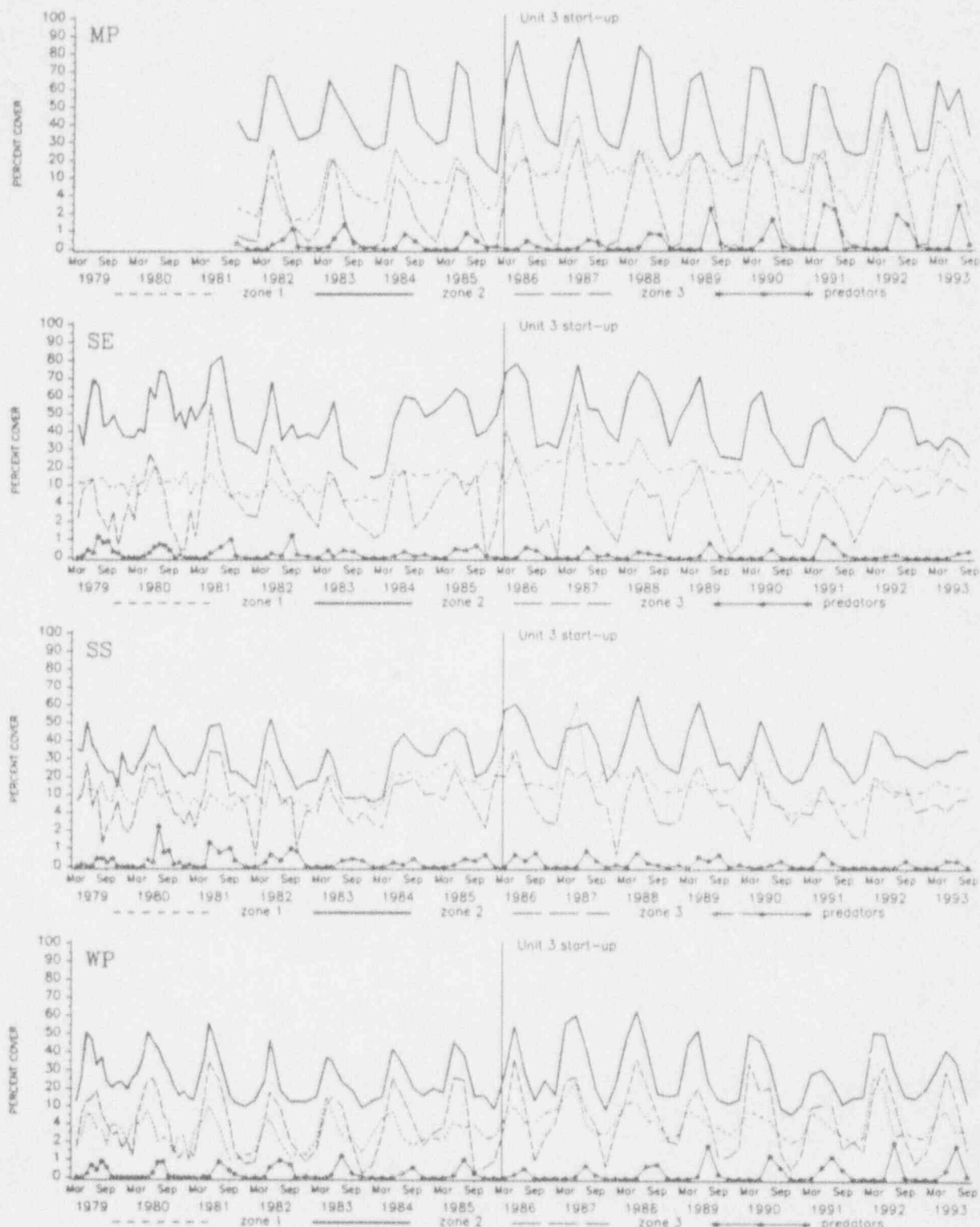


Fig. 4. (cont.)

## *Fucus*

*Fucus vesiculosus* is the most common species of macroalgae in local mid and upper intertidal zones but also occurs in lower abundance in the low intertidal. Other species of *Fucus* are found occasionally at our study sites, including *F. distichus* subsp. *edentatus*, *F. distichus* subsp. *evanescens* (both occur mostly subtidally) and occasionally *F. spiralis*, which occurs in the high intertidal. Our estimates of *Fucus* abundance combine abundances of all the above mentioned species.

Local distribution patterns of *Fucus*, presented in Figure 5, are similar to those reported elsewhere in New England (Lubchenco 1980, 1983; Topinka et al. 1981). *Fucus* abundance maxima during 1992-93 in Zone 1 were highest at FE (42%), with the low maximum of 2% at FS and GN (Fig. 5). Highest Zone 2 maxima during 1992-93 occurred at FS (84%); maximum cover was lowest at BP and WP (35%). In Zone 3, maximum *Fucus* cover during 1992-93 reached a high of 43% at FS; the low peak abundance value occurred at FE (2%).

In general, *Fucus* is most abundant on moderately exposed shores, common conditions at most of our study sites. *Fucus* abundance is limited at highly exposed sites by physical stress from wave shock, while at sheltered sites these species are often outcompeted for space by another fucoid, *Ascophyllum nodosum* (Schonbeck and Norton 1978, 1980; Keser and Larson 1984). At both sheltered and exposed sites, maximum *Fucus* abundance typically occurs in late summer/early autumn. Local vertical distribution patterns of intertidal *Fucus* are determined, in large part, by mechanisms similar to those discussed for barnacles (i.e., related to exposure to waves and slope of available substratum) and are discussed in previous reports (NUSCO 1992, 1993).

Physical stress from the MNPS discharge, in the form of heat, is an important mechanism controlling *Fucus* abundance in Zone 3 at FE. Elevated temperatures during periods of thermal plume incursion have resulted in annual elimination of *Fucus* in Zone 3 since the opening of the second quarry cut in 1983 and throughout 3-unit operation, including 1992-93 (Fig. 5). Thermal stress was most severe at FE in Zone 3, because organisms there were submerged and exposed to elevated temperatures for much of the

tidal cycle. After Unit 3 went on line, thermal stress at mid and upper intertidal levels was substantially reduced, and *Fucus* populations in Zones 1 and 2 returned to abundance levels similar to those observed from 1979-1983. Reduced cover minima and maxima in Zones 1 and 2 were recorded in the past two years (1991-92 and 1992-93), compared to other 3-unit years; however, these values were still within the range of values recorded prior to 1984, when impacts were first detected at FE.

In previous reports (e.g., NUSCO 1992, 1993), we have discussed the possibility of power plant impact at MP causing the decline of *Fucus* from 1981-1986, followed by a prolonged period of recovery (6 years). Although continuous temperature recorders deployed at MP over the last several years show water temperatures 2-3°C above ambient during slack tides with one to three units operating, we have no direct evidence linking the pattern of *Fucus* abundance at MP to power plant impact. Elevated temperatures may have indirectly impacted *Fucus* recruitment at MP, by enhancement of earlier seasonal migration and higher feeding rates of grazers (mostly *Littorina littorea*; Newell et al. 1971, NUSCO 1993). *Fucus* now appears to be well established at MP, as continued high cover in Zones 2 and 3 was observed during 1992-93 (>40%). Similar, although less marked, increases in *Fucus* abundance in recent years have also been observed at BP, SE, SS and WP, which may indicate an area-wide trend not related to power plant operation.

## *Chondrus* and common epiphytes

As discussed in previous sections, power plant impacts to the local rocky intertidal have occurred primarily in Zone 3 at our sampling site nearest the discharge (FE). Documentation of abundance patterns of the dominant low intertidal alga, *Chondrus crispus*, and its associated seasonally abundant epiphytes (e.g., *Monostroma* spp. and *Polysiphonia* spp.) is therefore critical to our assessment of these impacts.

*Chondrus* abundance time-series are presented in Figure 6. Perennial populations of *Chondrus* have occurred in the low intertidal at all study sites, excluding FE. Abundance maxima at these sites during 1992-93 ranged from 10% at FS to

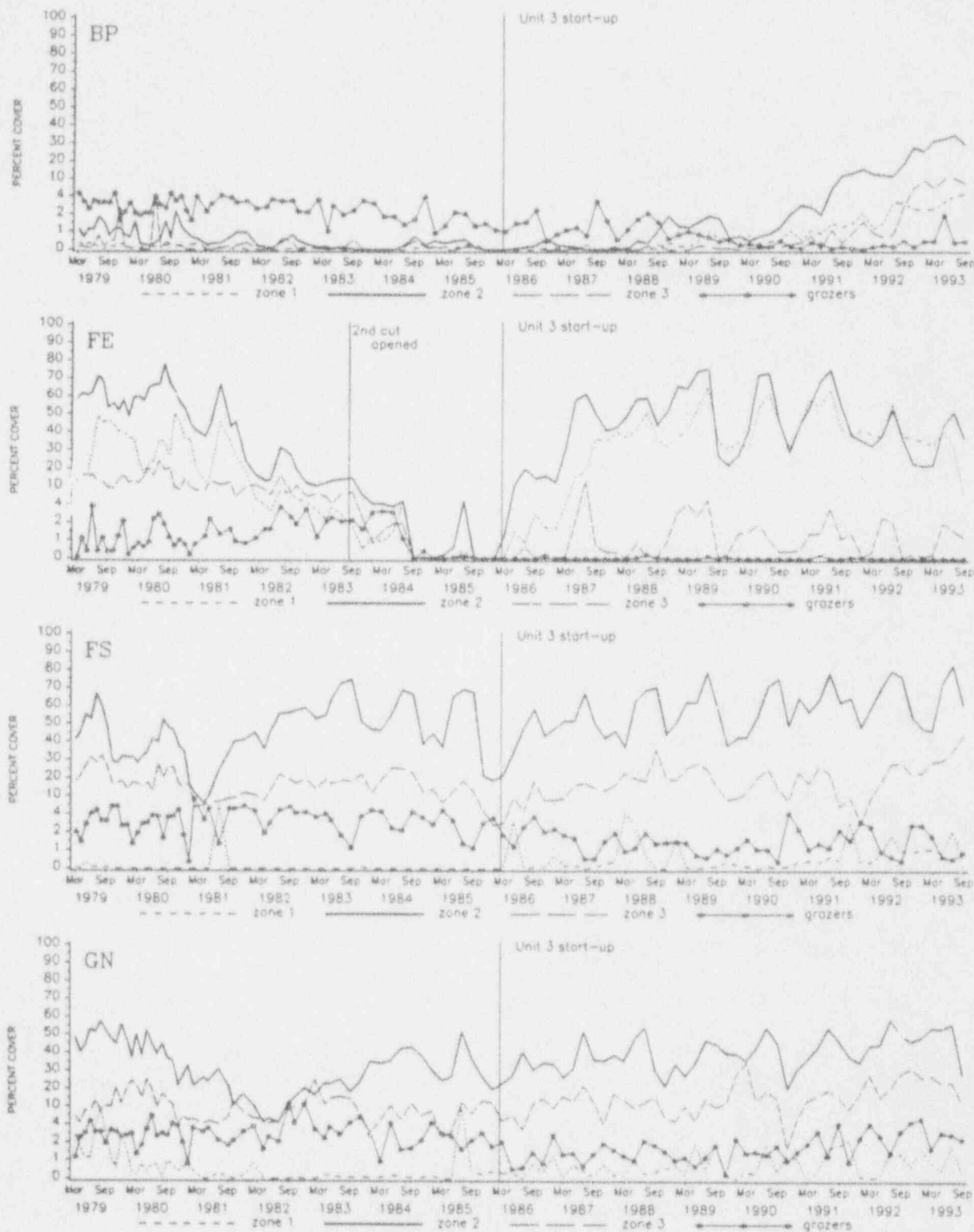


Fig. 5. Abundance of *Fucus* in each zone, and of grazing snails in Zone 3, of undisturbed transects, from 3/79 - 9/93.



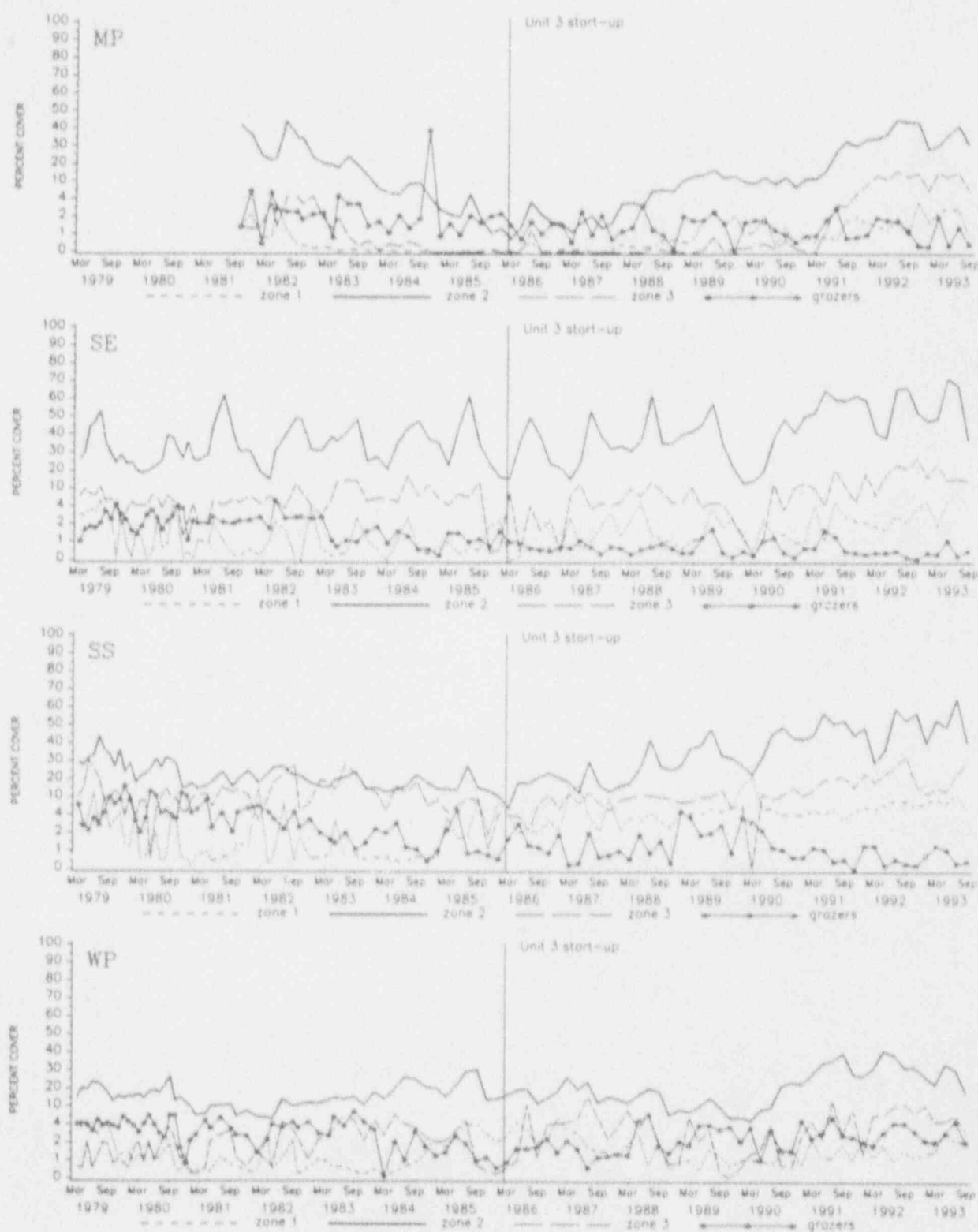


Fig. 5. (cont.)

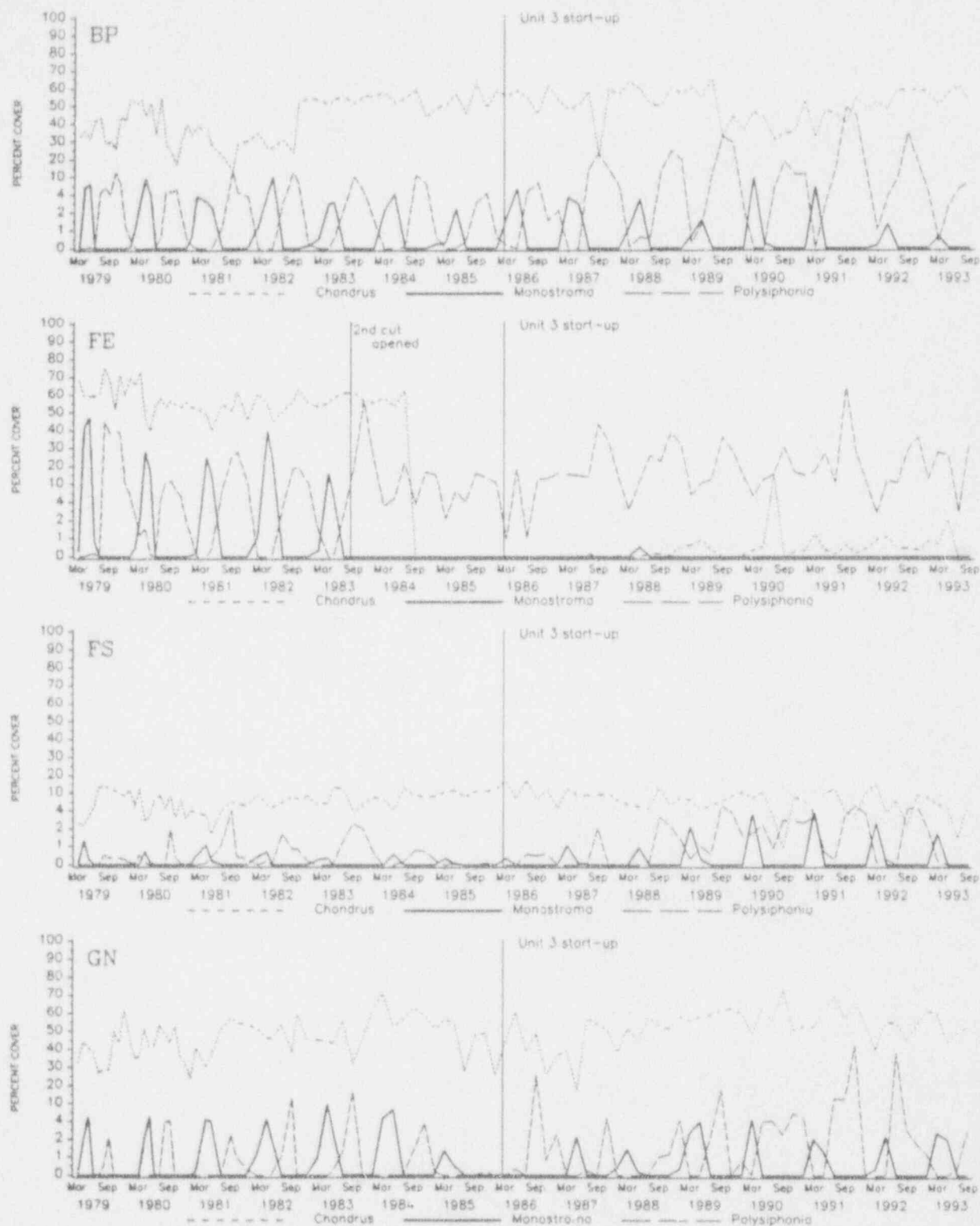


Fig. 6. Abundance of *Chondrus* and major epiphytes in Zone 3 of undisturbed transects, from 3/79 - 9/93.

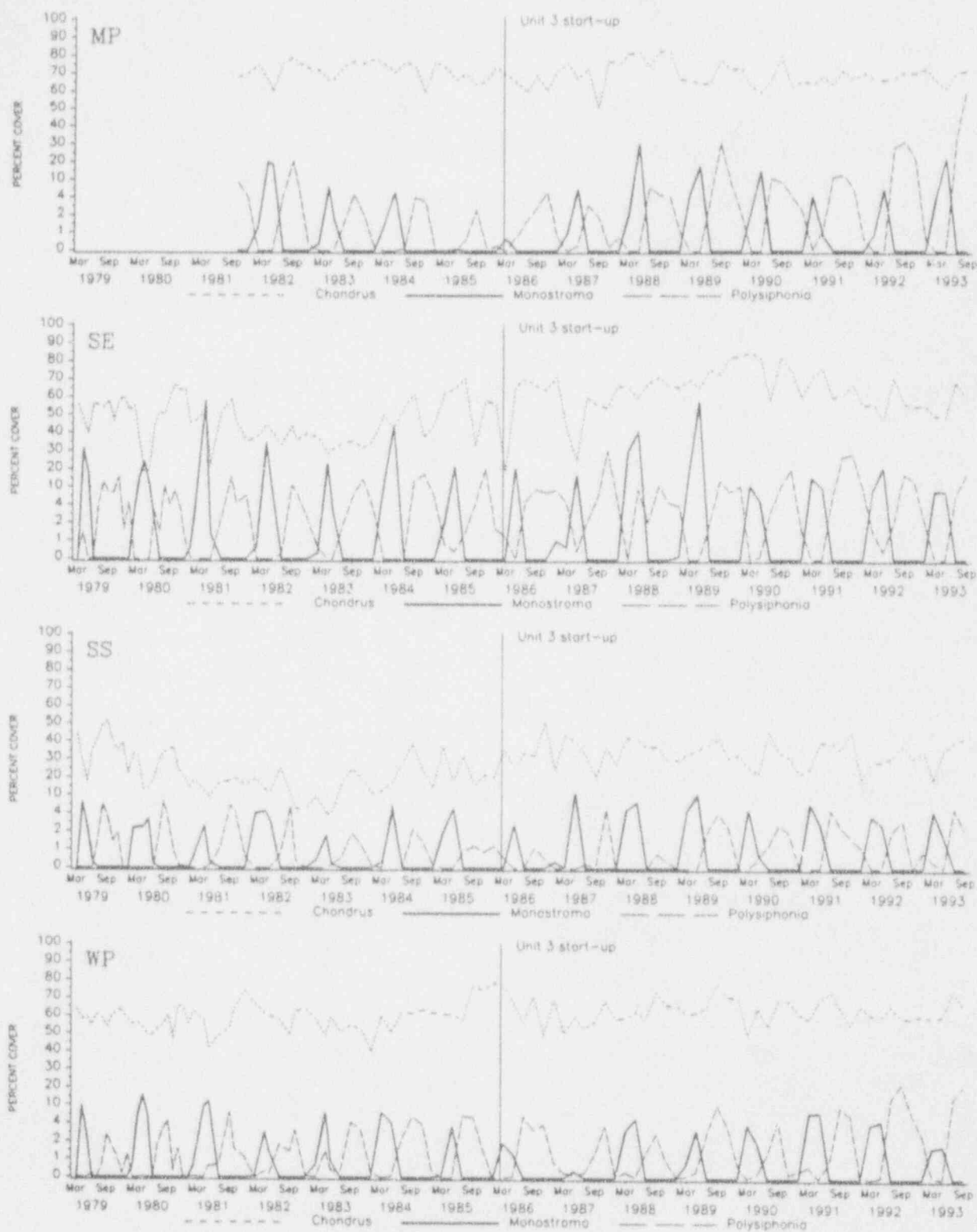


Fig. 6. (cont.)

approximately 70% at MP, SE and WP. In general, *Chondrus* abundances observed during 1992-93 at all sites except FE were similar to those observed throughout our studies.

Maximum *Chondrus* cover at FE during 1992-93 was only 2%, which was within the range of covers recorded at that site after 1984 (0-14%), but well below those recorded prior to 1984 (40-75%). This extensive population was eliminated in 1984 by elevated water temperatures from the 2-cut 2-unit discharge (NUSCO 1987). Since that time, only a few scattered plants have been observed in upper Zone 3 study quadrats; abundance estimates have generally been less than 1%, and declined to near 0% during each summer. These declines are no doubt caused by elevated summer water temperatures from the 3-unit 2-cut discharge, which appear to preclude any successful reestablishment of the *Chondrus* population at FE. The low intertidal community at FE is now composed primarily of an extensive *Codium* population and persistent populations of ephemeral algae including *Ulva lactuca*, *Enteromorpha* spp. and *Polysiphonia* spp. This warm-water tolerant community may also competitively exclude *Chondrus* from Zone 3 through preemption of suitable substratum.

Power plant impacts are also reflected in temporal abundance shifts of seasonal epiphytes. *Polysiphonia* spp. (mostly *P. novae-angliae* and *P. harveyi*) are common warm-water epiphytes on *Chondrus*, *Ascophyllum* and *Codium*, and also grow attached to rock. The annual abundance cycle of *Polysiphonia* spp. is characterized by a late summer peak, with cover declining to near 0% by winter (Fig. 6.). Peak abundances during 1992-93 ranged from 3% at FS to 60% at MP. In general, these estimates are consistent with those observed throughout the NUSCO monitoring program, again excepting FE. Elevated temperature regimes at FE since the opening of the second quarry cut (1983) have produced favorable conditions for these species by extending the season of occurrence and increasing the levels of peak abundance. These temperature regimes at FE have also allowed *Polysiphonia* spp. to persist through cold water months, when these species are typically absent from other sites, including FE prior to 1983.

The annual abundance cycle of *Monostruma* spp. (*M. grevillei* and *M. pulchrum*) is nearly opposite to

that described for *Polysiphonia* spp. i.e., peak abundance is observed during cold water months (late winter/early spring) and virtual absence is noted during warm-water months (July-December; Table 1). *Monostruma* spp. occurred at every site except FE during 1992-93 (Fig. 6); peak abundance ranged from 1% (BP) to 21% (MP). Abundance patterns of *Monostruma* at all sites except FE in 1992-93 were similar to those observed during previous 2-unit and 3-unit years. *Monostruma* has only occurred once in undisturbed transects at FE since 1983 (<1% cover in May 1988), where spring cold water temperatures required for *Monostruma* rarely occur.

### Similarity Dendrograms

The Bray-Curtis Similarity Index is an analytical technique applied previously to qualitative data and used here to evaluate communities in the Millstone area based on the abundances of all species observed in Zones 2 and 3 of undisturbed transects. The resulting clustering dendrogram illustrates multiple pair-wise comparisons. This technique is used to compare communities at all sites over 3-unit and 2-unit periods, and to examine annual communal changes at our experimental site, FE.

Similarities of species composition among stations in mid and low intertidal zones during each operational period are illustrated in Figure 7a. Similarities were highest (>75%) between operational collections at the same station for all stations except FE, indicating a relatively consistent species composition at these sites throughout 3-unit and 2-unit operation. Similarities were less for among-station comparisons, but were primarily related to natural factors affecting abundances of *Fucus* and *Chondrus* (e.g., Groups I, II and III; NUSCO 1993). Group IV consisted of the 3-unit operational collection at FE, which is greatly dissimilar to all other collections. The community at FE during 3-unit operation reflects power plant impacts, and is characterized by the absence of *Chondrus* and higher abundances of *Codium* and ephemeral algae.

To better illustrate annual community responses to separate operational events, including those prior to Unit 3 start-up, a clustering dendrogram was generated based on annual collections at FE

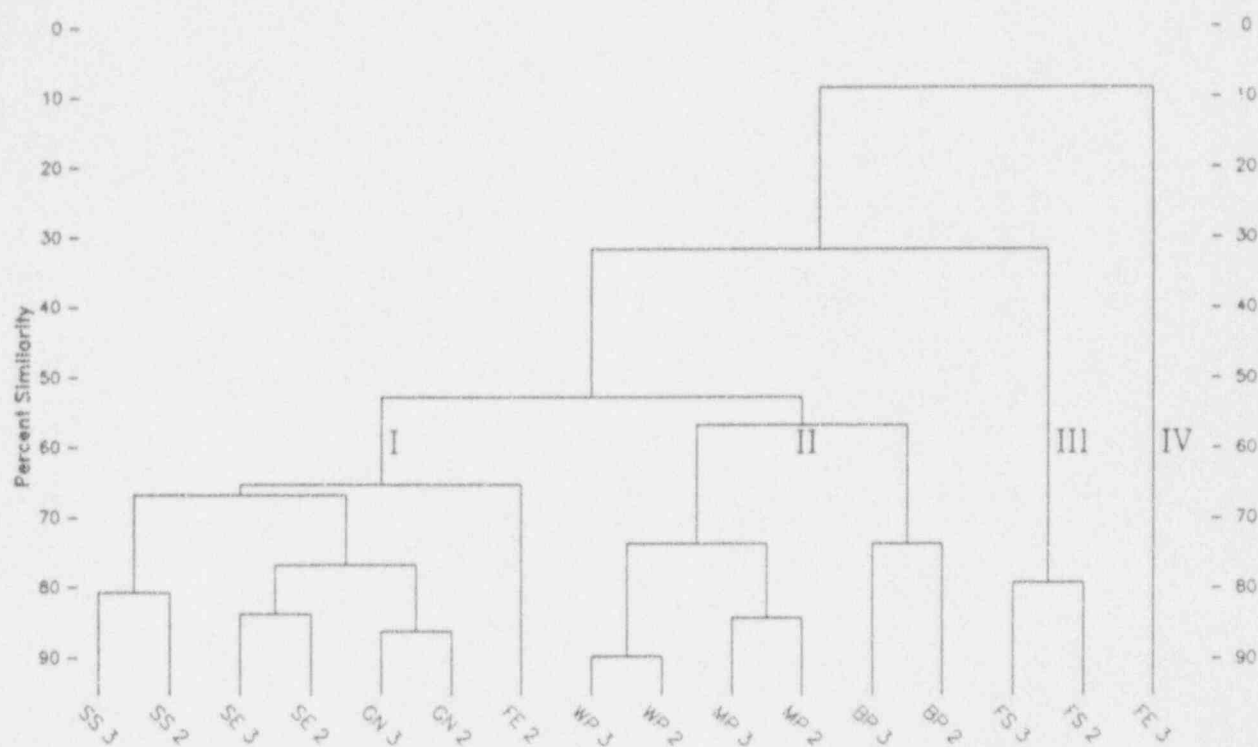


Fig. 7a. Clustering dendrogram of percent similarity of undisturbed communities, by station and operational period (2-unit - 3-unit).

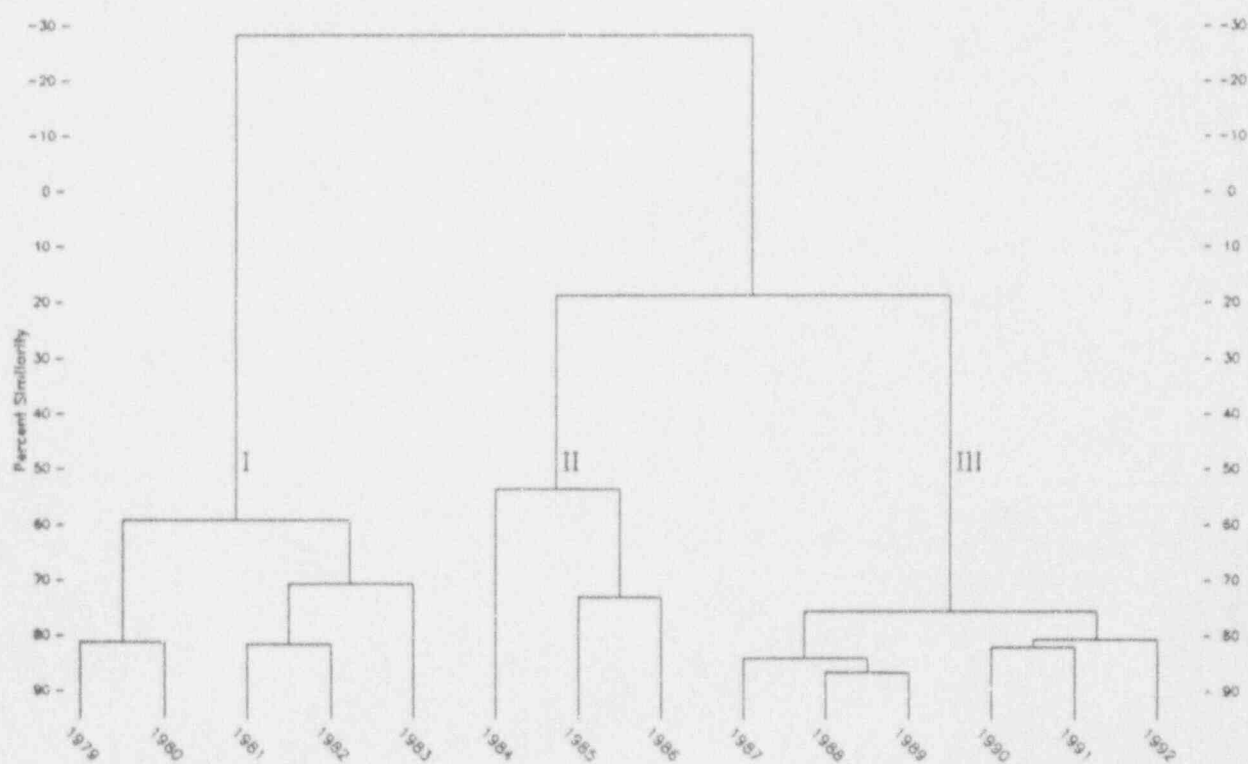


Fig. 7b. Clustering dendrogram of percent similarity of undisturbed communities, by year, at Fox Island-Exposed.



(Figure 7b). This method showed three groupings at the 50% similarity level. Group I represented an unimpacted community at FE and consisted of annual collections made before changes associated with the opening of the second quarry cut were observed. During these years (1979-83), the FE community was similar to those observed at other moderately exposed sites, with extensive *Fucus* and *Chondrus* populations and consistent occurrences of species such as *Monostroma*, *Polysiphonia*, and other local annuals. Group II consisted of annual collections made during years of maximum thermal incursion created by 2-unit 2-cut operating conditions. *Chondrus* was eliminated after 1984, and barnacle and *Fucus* populations were eliminated annually in summer after that time until 1987. This period was also marked by high abundances of *Polysiphonia* and ephemeral green algae.

Group III represented years of community development at FE under the less stressful thermal conditions of 3-unit operation. Group III was distinguished from Group II by the return of a substantial *Fucus* population and small amounts of *Chondrus* and the persistence of an extensive *Codium* population. Other significant events during this period were the appearance, persistence and expansion of populations of the perennials *Sargassum filipendula* and *Gracilaria tikvahiae*, species previously not observed in undisturbed transects. The high level of similarity among annual collections in recent years at FE suggests a relatively consistent species composition, which has developed in response to present operating conditions.

### *Ascophyllum nodosum* Studies

Growth and mortality of three local populations of the perennial brown alga, *Ascophyllum nodosum*, continued to be monitored during the 1992-93 sampling year. Many factors contribute to the overall value of this species as a useful and sensitive biomonitoring tool. An extensive review of phenological, ecological and applied monitoring studies of *Ascophyllum* is presented in NUSCO (1993). Growth and mortality results from the most recent sampling year (1992-93) are presented below and compared with results from overall 2-unit and 3-unit operational periods.

### Growth

*Ascophyllum* growth, or annual tip elongation described by the  $\alpha$  parameter in the Gompertz growth model fitted to the data, is presented in Figure 8. *Ascophyllum* growth during 1992-93 (Fig. 8a) was significantly higher ( $P < 0.05$ ) at FN (116.3 mm) than growth at both GN (105.9 mm) and WP (97.8 mm). Growth differences between GN and WP were also significant. The inflection point of the growth curve, which identifies the time of maximum growth rate, occurred earliest at FN in 1992 (July 16). Growth rate peaked later at GN and WP in 1992 (July 24 and July 29, respectively). Growth at GN during 1992-93 was significantly higher than growth over both the 3-unit (97.8 mm) and 2-unit (90.1 mm) operational periods (Fig. 8b). The difference between growth estimates during 3-unit and 2-unit operation at GN was also significant. Although growth at WP over the 1992-93 season was also higher than growth during 3-unit operation (89.0 mm) and during 2-unit operation (90.2 mm; Fig. 8c), the growth difference between operational periods was not significant. At FN, growth during 1992-93 was significantly higher than the 1985-86 2-unit year (90.5 mm), but was not different from growth over the 3-unit period (Fig. 8d). Growth during 3-unit years at FN was also significantly higher than growth during 1985-86.

Relatively consistent spatial relationships among *Ascophyllum* populations studied around MNPS have been observed throughout 3-unit operation. The most obvious and important aspect of these relationships is the phenomenon of enhanced growth at the site nearest the discharge cuts, FN, when compared to sites farther away. Growth characteristics at GN and WP were more similar during the entire study period, including 2-unit operational years; when differences were observed, such as during the most recent sampling year, most often higher growth occurred at GN, the control site well beyond any power plant influence (NUSCO 1992, 1993). Therefore, we attribute differences between reference sites to spatial variability of natural environmental factors and not to power plant operation.

By contrast, increased growth and accelerated growth rates of FN *Ascophyllum* plants are considered to be directly related to power plant operation, in particular, to periodic exposure to

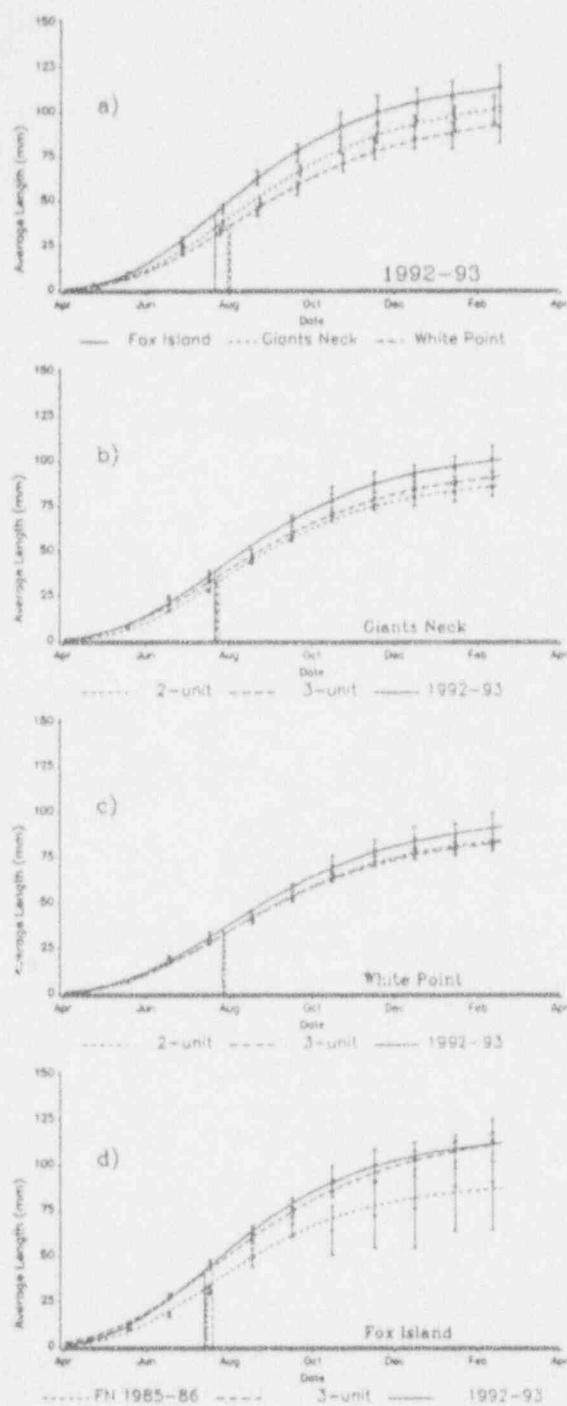


Fig. 8. *Ascophyllum* growth: a) during 1992-1993, b-d) present year, 3-unit and 2-unit operational periods at each station. Curves are the Gompertz growth model fitted to tip length data, including inflection points. Error bars represent monthly mean lengths  $\pm 2$  SE.

the discharge thermal plume. Water temperatures at FN are elevated 3-4°C for 3-4 hours each tidal cycle when all three units are operating at full power. This level of temperature increase creates near ideal conditions for *Ascophyllum* growth at FN by: 1) extending the period of "normal" or "ambient" peak growing conditions for local *Ascophyllum* populations (18-21°C; Kanwisher 1966; Chock and Mathieson 1979), 2) more closely synchronizing these periods of optimal growing temperatures with the period of maximum daily solar irradiance (June), and 3) elevating temperatures in late summer above normal maxima but below stress levels (22-25°C), increasing plant respiration and growth rates without exceeding photosynthate production (Brinkhuis et al. 1976; Stromgren 1977, 1981; Vadas et al. 1978). Similar conditions have been implicated when *Ascophyllum* growth enhancement has been reported near other coastal power plants (Vadas et al. 1976, 1978; Wilce et al. 1978).

Year-to-year variability in *Ascophyllum* growth has been noted at FN during 3-unit operation and appears related to the degree of thermal load produced by the power plant. In other words, growth at FN is highest when periods of 3-unit operation are longer or more frequent during the peak growing season (May-November), compared to years when one or more unit outages occurs during that time (NUSCO 1992, 1993). The degree of growth enhancement at FN during 1992-93 was intermediate compared to previous years and likely due to the extended outage of Unit 2 during the second half of 1992.

### Mortality

*Ascophyllum* plant breakage, referred to here as mortality, was assessed by examining patterns of frond base tag loss (plant loss; Fig. 9) and apical tip tag loss (tip loss; Fig. 10). Plant loss at GN during 1992-93 (32%) was lower than mean plant loss during the 3-unit period (58%) and the 2-unit period (52%). Similarly, plant loss at FN during 1992-93 was 58%, which was lower than both the 3-unit and 1985-86 (2-unit) plant losses of 66% and 80%, respectively. Conversely, plant loss at WP during 1992-93 (62%) was higher than both operational means (58% and 55% during 3-unit and 2-unit periods, respectively). Tip loss at GN was 51% during 1992-93, lower than 3-unit and 2-unit tip

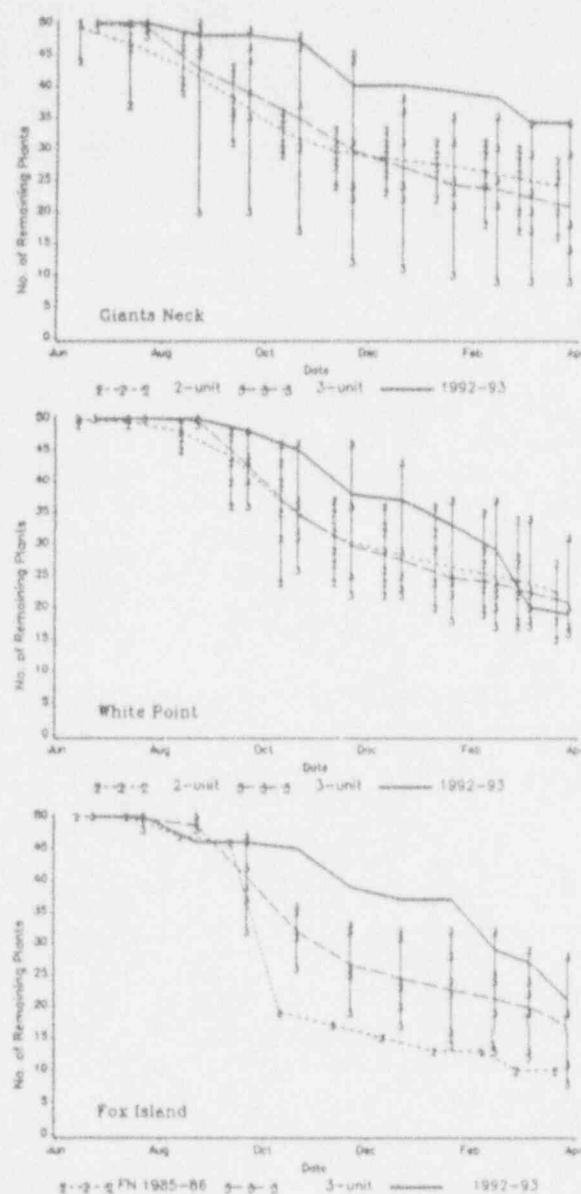


Fig. 9. *Ascophyllum* mortality, as number of remaining tagged plants, at each station.

losses (76% and 75%, respectively). Tip loss was also lower at FN during 1992-93 (77%) than during the overall 3-unit period (83%) and the 1985-86 2-unit year (90%). As with GN and FN, the 1992-93 tip loss at WP (71%) was also lower than both operational means; 72% during 3-unit operation, 75% during 2-unit operation.

*Ascophyllum* mortality studies during 3-unit operation, including the most recent sampling year (1992-93), have revealed no power plant-related mortality to populations around MNPS. Overall,

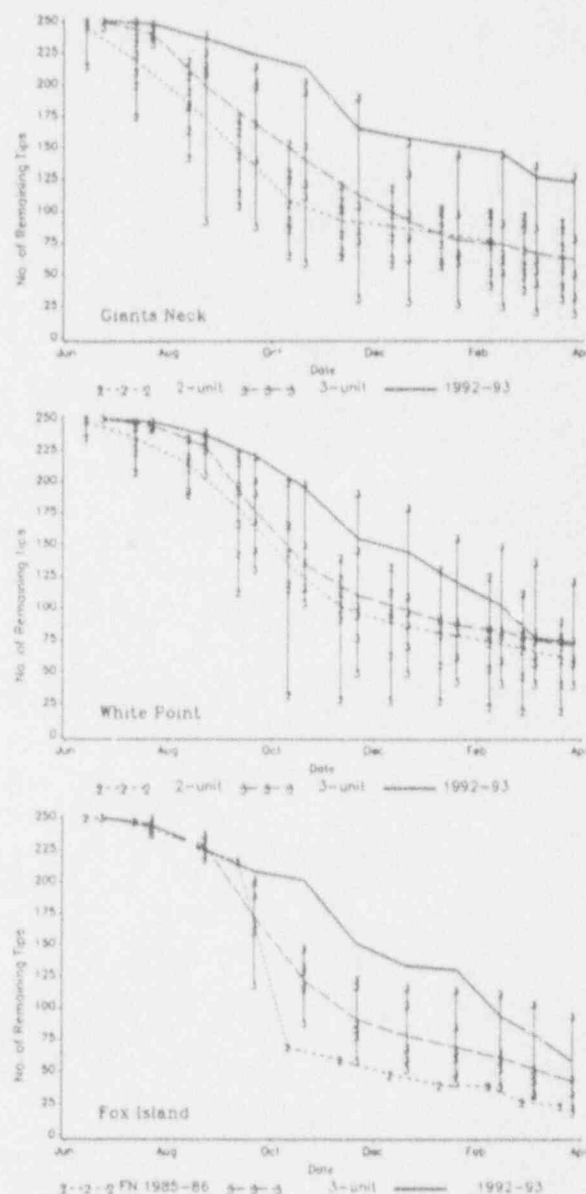


Fig. 10. *Ascophyllum* mortality, as number of remaining tagged tips, at each station.

somewhat higher mortality rates were observed at FN, our sampling site nearest the discharge, than at reference sites. However, these higher mortality rates do not appear to be related to proximity to the discharge, but rather to the exposed orientation of this site, in contrast to the more sheltered reference sites. An area-wide seasonal pattern of mortality has been observed throughout our studies which further implicates wave-induced stress as a major cause of mortality. During both 2-unit and 3-unit operational periods, mortality

rates were highest during the months of August through November, when strong storms and high energy waves were frequent. Many studies elsewhere point to stress from wave action as the most important factor and report a strong relationship between mortality and degree of site exposure to prevailing winds and storms (Baardseth 1955, 1970; Jones and Demetropoulos 1968; Vadas et al. 1976, 1978; Wilce et al. 1978; Cousens 1982, 1986; Vadas and Wright 1986).

We continue to monitor FO, our original experimental *Ascophyllum* site, for recovery following power plant-induced population elimination in 1984 (NUSCO 1987, 1993). Some individual plants have settled, grown and persisted at FO during 3-unit operation, however, no significant population recovery has occurred to date. Environmental conditions at FO created by 3-unit operation, although less stressful than those during 2-unit, 2-cut operation, may be outside the extremely limited range of conditions required for successful recruitment of this characteristically slow recolonizer.

### Conclusions

NUSCO Rocky Intertidal Studies have successfully characterized attached shore biota by identifying variability in population and community parameters and relating this variability to the influence of site-specific controlling mechanisms. At most sites, degree of exposure to wave energy (through site orientation to prevailing wind-generated waves and ability of available substratum (slope) to dissipate the horizontal force of those waves), and the character of that substratum, are the direct or underlying causes of the most notable among-site differences in the occurrence and distribution of local species outside the influence of MNPS.

Community differences which could not be explained by these natural mechanisms and occurred within the thermal plume area, such as those observed in the Fox Island area (FE and FN), are directly attributed to the operation of MNPS. Detection of these differences was accomplished by comparison of present population and community parameters at FE to those at unimpacted sites farther away from the discharge and to those at FE prior to community breakdown in 1984. For example, qualitative studies

continued to document shifts in occurrence of the algal flora at FE during 1992-93, which included presence or extended season of occurrence for species with warm-water affinity and absence or abbreviated season for species with cold-water affinity.

Shifts in abundance observed only at FE during 1993 were most pronounced in the low intertidal, where elevated temperature conditions were most pronounced. The low intertidal community at FE, which prior to 1983 was unimpacted and characterized by populations of *Chondrus* and *Ascophyllum*, has been replaced by a persistent community dominated by *Codium*, *Ulva*, *Enteromorpha* and *Polysiphonia*. Also, populations of species observed in undisturbed transects only at FE (*Sargassum*, *Gracilaria*) continued to persist and expand during 1993.

Elevated temperatures (2-3°C above ambient) at the *Ascophyllum* station nearest the discharge (FN) caused plants to grow longer and more rapidly at that site during 1992-93, relative to growth of *Ascophyllum* at more distant stations. The degree of growth enhancement at FN during 1992-93 was intermediate, compared to previous years at that site, likely due to lessened thermal plume incursion resulting from an extended outage of Unit 2 for much of the peak growing season. *Ascophyllum* plant and tip mortality were associated primarily with exposure to storm forces, rather than with proximity to the discharge.

### References Cited

- Baardseth, E. 1955. Regrowth of *Ascophyllum nodosum* After Harvesting. Inst. Ind. Res. Stand., Dublin. 63 pp.
- Baardseth, E. 1970. Seasonal variation in *Ascophyllum nodosum* (L.) Le Jol. in the Trondheimsfjord with respect to the absolute live and dry weight and the relative contents of dry matter, ash and fruit bodies. Bot. Mar. 13:13-22.
- Bertness, M.D. 1989. Intraspecific competition and facilitation in a northern acorn barnacle population. Ecology 70:257-268.
- Brinkhuis, B.H., N.R. Campbell, and R.F. Jones. 1976. Photosynthesis and respiration of exposed salt marsh fucoids. Mar. Biol. 34:349-359.
- Chock, J.S., and A.C. Mathieson. 1979. Physiological ecology of *Ascophyllum nodosum*



- (L.) Le Jolis and its detached ecad *scorpioides* (Hornemann) Hauck (Fucales, Phaeophyta). Bot. Mar. 22:21-26.
- Clifford, H.T., and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press, New York. 229 pp.
- Connell, J.H. 1961. Effects of competition, predation, by *Thais lapillus* and other factors on natural populations of the barnacle, *Balanus balanoides*. Ecol. Monogr. 31:61-104.
- Cousens, R. 1982. The effect of exposure to wave action on the morphology and pigmentation of *Ascophyllum nodosum* (L.) Le Jolis in south-eastern Canada. Bot. Mar. 25:191-195.
- Cousens, R. 1986. Quantitative reproduction and reproductive effort by stands of the brown alga *Ascophyllum nodosum* (L.) Le Jolis in south-eastern Canada. Estuar. Coast. Shelf Sci. 22:495-507.
- Draper, N., and H. Smith. 1981. Applied Regression Analysis. John Wiley and Sons, New York. 709 pp.
- Gendron, L. 1989. Seasonal growth of the kelp *Laminaria longicruris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. Bot. Mar. 32:345-354.
- Jones, J.E., and A. Demetropoulos. 1968. Exposure to wave action: Measurements of an important ecological parameter on rocky shores on Anglesey. J. Exp. Mar. Biol. Ecol. 2:46-63.
- Kanwisher, G.W. 1966. Photosynthesis and respiration in some seaweeds. Pages 407-420 in H. Barnes (ed.). Some Contemporary Studies in Marine Science. George Allen Unwin Ltd., London.
- Keser M., and B.R. Larson. 1984. Colonization and growth dynamics of three species of *Fucus*. Mar. Ecol. Prog. Ser. 15:125-134.
- Lance, G.N., and W.R. Williams. 1967. A general theory of classificatory sorting strategies. I. Hierarchical systems. Comput. J. 9:373-380.
- Lubchenco, J. 1980. Algal zonation in the New England rocky intertidal community: an experimental analysis. Ecology 61:333-244.
- Lubchenco, J. 1983. *Littorina* and *Fucus*: effects of herbivores, substratum heterogeneity, and plant escapes during succession. Ecology 64:1116-1123.
- Menge, B.A. 1976. Organization of the New England rocky intertidal community: role of predation, competition and environmental heterogeneity. Ecol. Monogr. 46:355-393.
- NAI (Normandeau Associates, Inc.). 1993. Seabrook environmental studies, 1992. A characterization of environmental conditions in the Hampton-Seabrook area during the operation of Seabrook Station. Tech. Rpt. XXIV-1.
- Newell, R.C., V.I. Pye, and M. Ahsanullah. 1971. Factors affecting the feeding rate of the wrinkle *Littorina littorea*. Mar. Biol. 9:138-144.
- NUSCO (Northeast Utilities Service Company). 1987. Rocky Intertidal Studies. Pages 1-66 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Summary of studies prior to Unit 3 operation.
- NUSCO. 1992. Rocky Intertidal Studies. Pages 237-292 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1991.
- NUSCO. 1993. Rocky Intertidal Studies. Pages 49-92 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford Connecticut. Annual Report, 1992.
- Schonbeck, M.W., and T.A. Norton. 1978. Factors controlling the upper limits of fucoid algae on the shore. J. Exp. Mar. Biol. Ecol. 31:303-313.
- Schonbeck, M.W., and T.A. Norton. 1980. Factors controlling the lower limits of fucoid algae on the shore. J. Exp. Mar. Biol. Ecol. 43:131-150.
- Schneider, C.W. 1981. The effect of elevated temperature and reactor shutdown on the benthic marine flora of the Millstone thermal quarry, Connecticut. J. Therm. Biol. 6:1-6.
- South, G.R., and I. Tittley. 1986. A checklist and distributional index of the benthic marine algae of the North Atlantic Ocean. Huntsman Marine Laboratory and British Museum (Nat. Hist.), St. Andrews and London. 76 pp.
- Stromgren, T. 1977. Short-term effects of temperature upon the growth of intertidal fucals. J. Exp. Mar. Biol. Ecol. 29:181-195.
- Stromgren, T. 1981. Individual variation in apical growth rate in *Ascophyllum nodosum* (L.) Le Jolis. Aquat. Bot. 10:377-382.
- Topinka, J., L. Tucker, and W. Korjeff. 1981. The distribution of fucoid macroalgal biomass along



- central coastal Maine. *Bot. Mar.* 24:311-319.
- Underwood, A.J., and E.J. Denley. 1984. Paradigms, explanations and generalizations in models for the structure of intertidal communities of rocky shores. Pages 151-180 in D.R. Strong, Jr., D. Simberloff, L.G. Abele and A.B. Thistle (eds.). *Ecological Communities: Conceptual Issues and the Evidence*. Princeton University Press, Princeton N.J. 611 pp.
- Vadas, R.L., M. Keser, and P.C. Rusanowski. 1976. Influence of thermal loading on the ecology of intertidal algae. Pages 202-251 in G.W. Esch and R.W. MacFarlane (eds.). *Thermal Ecology II*. ERDA Symposium Series, Augusta, GA.
- Vadas, R.L., M. Keser, and P.C. Rusanowski. 1978. Effect of reduced temperature on previously stressed populations of an intertidal alga. Pages 434-451 in J.H. Thorp and G.W. Gibbons (eds.). *DOE Symposium Series*. Springfield, VA. (CONF-771114, NTIS).
- Vadas, R.L., and W.A. Wright. 1986. Recruitment, growth and management of *Ascophyllum nodosum*. *Actas II Congr. Algas Mar. Chilenas*:101-113.
- Wilce, R.T., J. Foertch, W. Grocki, J. Kilar, H. Levine, and J. Wilce. 1978. Flora: Marine Algal Studies. Pages 307-656 in *Benthic Studies in the Vicinity of Pilgrim Nuclear Power Station, 1969-1977*. Summary Rpt., Boston Edison Co.



## Benthic Infauna

Introduction .....	83
Materials and Methods .....	83
Data Analyses .....	84
Sediments .....	84
Multiple Regression Analyses .....	85
Response Modeling and Trend Analysis .....	86
Results .....	86
Sedimentary Environment .....	86
General Community Composition .....	89
Community Abundance .....	89
Number of Species .....	90
Community Dominance .....	90
Dominant Taxa .....	91
Cumulative Abundance Curves .....	97
Discussion .....	98
Conclusions .....	99
References Cited .....	100



## Benthic Infauna

### Introduction

Benthic infauna, in soft-bottom subtidal habitats in the vicinity of Millstone Nuclear Power Station (MNPS), have been monitored since 1973. Environmental variability characteristic of estuarine systems (Holland 1985; Nichols 1985; Holland et al. 1987; Warwick 1988; Rees and Eleftheriou 1989), together with our inability to completely understand how physical and biological factors combine to impose structure on, and control the functions of, benthic communities (Diaz and Schaffner 1990), make long-term monitoring studies necessary to assess the impacts of human activities on marine environments (Thrush et al. 1994). Such studies are the principal means of characterizing species composition and fluctuations in abundance which might occur in response to acute or chronic climatic conditions (Boesch et al. 1976; Flint 1985; Jordan and Sutton 1985), to variations in biological factors such as competition and predation (e.g., Levinton and Stewart 1982; Woodin 1982; Kneib 1988), or to human activities.

Infaunal studies are an important component of environmental impact studies for several reasons. Infauna are a source of food for numerous invertebrate and vertebrate species, including demersal fishes (Richards 1963; Moeller et al. 1985; Watzin 1986; Horn and Gibson 1988; Commito and Boncavage 1989; Franz and Tanacredi 1992). Sediment reworking resulting from the burrowing and tube-building activities of infauna can promote nutrient recycling from the sediments to the water column (Goldhaber et al. 1977; Aller 1978; Gaston and Nasci 1988). The close association of benthic communities with the sediments, where most pollutants ultimately accumulate, also make them an effective integrator of short and long term environmental conditions (Diaz and Schaffner 1990; Warwick et al. 1990). Lastly, because many studies have documented changes in benthic communities following disturbance (Boesch and Rosenberg 1982; Young and Young 1982; Gaston and Nasci 1988; Regauault et al. 1988; Rees and Eleftheriou 1989; Warwick et al. 1990), a framework of baseline studies exists to aid in evaluating impacts of human activities on marine benthic systems.

The infaunal monitoring program at Millstone was designed to measure infaunal species composition and abundance, to identify spatial and temporal patterns in community structure and abundance, and to assess whether observed changes might have been the result of construction and operation of MNPS. To date, Millstone studies have identified impacts to infaunal communities that were attributed to Unit 3 intake construction (NUSCO 1987) and to 3-unit operations (NUSCO 1988a), as well as regional shifts in species composition and abundance that apparently were the result of natural events. This report presents data collected during the 1993 sampling year, and compares them to results of monitoring local infaunal communities during 2-unit (1979-85) and 3-unit (1986-93) operational periods at MNPS.

### Materials and Methods

Subtidal infaunal communities in the vicinity of MNPS were sampled quarterly (September, December, March and June) from 1979 through 1993 at four stations (Fig. 1). The Giants Neck station (GN), located 6 km west of MNPS, is outside the area potentially affected by power plant operations. This station was used to identify possible region-wide shifts in infaunal community structure and composition which occur independently of power plant operations. The intake station (IN) is located 100 m seaward of MNPS Unit 2 and Unit 3 intake structures, and is exposed to scour produced by inflow of cooling water and the effects of periodic dredging. The effluent station (EF), located approximately 100 m offshore from the station discharge into Long Island Sound, is exposed to increased water temperatures, scour, and to chemical or heavy metal additions to the cooling water discharge. The Jordan Cove station (JC) is located 500 m east of MNPS and is considered potentially impacted by 3-unit operations. The area encompassing this station can experience increases in water temperatures of 0.8 to 2.2°C above ambient during certain tidal stages (principally ebb tide) due to the 3-unit thermal discharge of MNPS (NUSCO 1988b). At each station, ten replicate samples (0.0079 m<sup>2</sup> each) were collected by SCUBA divers using a hand-held coring device 10 cm in diameter



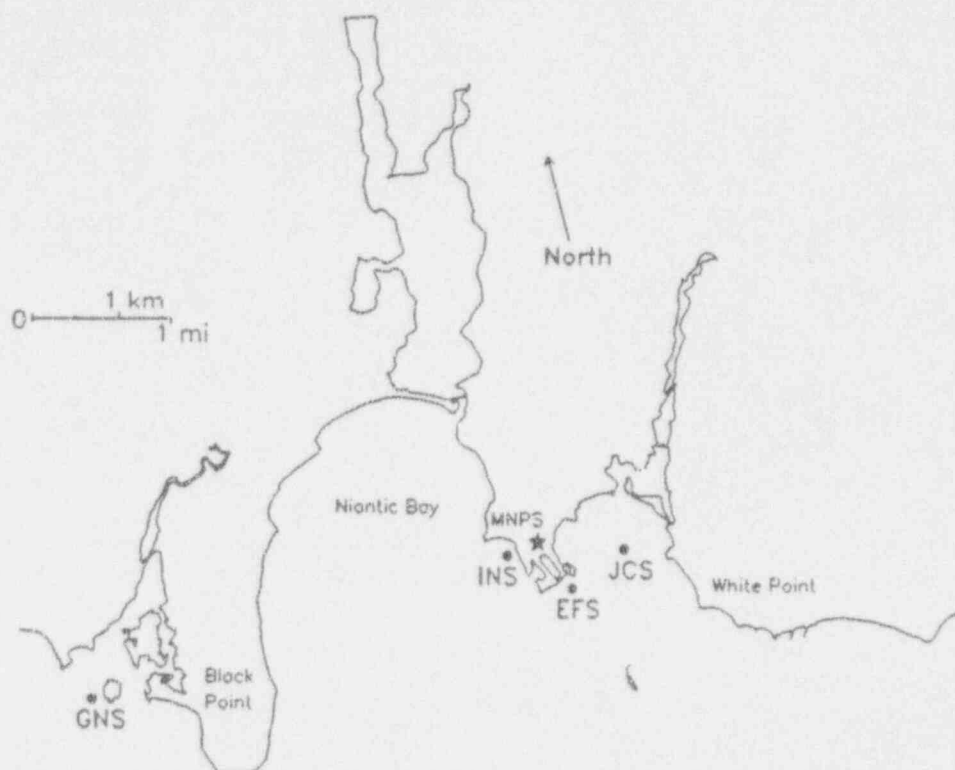


Fig. 1. Map of the Millstone Point area showing the location of subtidal infaunal stations (EF=Effluent, GN=Giants Neck, IN=Intake, JC=Jordan Cove).

x 5 cm deep. Each sample was placed in a 0.333 mm mesh Nitex bag and was brought to the surface. When taken to the laboratory, samples were fixed with a 10% buffered formalin/Rose Bengal solution and after a minimum of 48 h, organisms were floated from the sediments onto a 0.5 mm mesh sieve and preserved in 70% ethyl alcohol. Samples were examined using dissecting microscopes (10x) and organisms were sorted into major groups (annelids, arthropods, molluscs, and others) for later identification to the lowest practical taxon and counted. Oligochaetes and rhynchocoels were each treated in aggregate because of the difficulties associated with identifying these organisms. Organisms that were too small to be quantitatively sampled by our methods (e.g., nematodes, ostracods, copepods, and foraminifera) were not sorted. Grain size and the silt/clay fraction were determined from a 3.5 cm diameter x 5 cm core, taken at the time of infaunal sampling. Sediment samples were analyzed using the dry sieving method described by Folk (1974).

## Data Analyses

This report summarizes results of the macrofaunal sampling program conducted from September 1979 to June 1993. The period extending from September 1979 through June 1985 is considered the 2-unit operational period while September 1985 through June 1993 is referred to as the 3-unit period. A sampling year encompasses quarterly collections made from September through June of the following calendar year.

## Sediments

Sediment sieve fraction weights were used to construct cumulative curves for 2-unit (1980-85) and 3-unit (1986-93) operational periods by pooling quarterly weights from each sieve used for grain size analysis, in each 2-unit operational year and each 3-unit operational year, with years serving as replicates. Shifts in sedimentary environments over the 2-unit and 3-unit operational periods were then quantitatively assessed using the Gompertz function. This function has a sigmoid shape and

can describe cumulative data (e.g., growth data) that are not necessarily symmetrical about the midpoint of their range (Draper and Smith 1981). This feature provides the flexibility to fit cumulative data with or without an inflection point (s-shaped versus parabolic) within the observational range. The form of the Gompertz function used was:

$$C_t = 100 \exp[-e^{-k(t-p)}]$$

where  $C_t$  = cumulative sediment weight at point  $t$ ,  $p$  = location of inflection point in units of grain size and  $k$  = the shape parameter (Gendron 1989).

This function was fitted to data separately for 2-unit and 3-unit operational periods using non-linear regression methods (SAS 1985). Two-sample t-tests were used to test for differences between the  $k$  parameters of curves based on data collected during the 2-unit and the 3-unit operational periods.

### Multiple Regression Analyses

Multiple regression techniques were used to minimize the unexplained temporal variation in community abundance, in number of species and in the abundance of numerically dominant taxa. Several explanatory variables (described below) were used to remove variation that was attributable to fluctuations in sediments, reproductive or recruitment cycles, or climatic conditions. This technique was used to improve the sensitivity of analyses later performed to identify and compare long-term trends in data from the 2-unit and 3-unit sampling periods. Analyses were based on average quarterly abundance data after  $\ln(x+1)$  transformation and on species number collected from September 1979 through June 1993. Explanatory variables used in the regression analyses were as follows:

**Precipitation** - Daily precipitation records compiled by the U.S. Weather Bureau at the Groton Filtration Plant, Groton, CT were obtained from June 1976 through June 1993. Values to the nearest 0.01 inch were used as "rain" data.

**Water and Air Temperatures** - Ambient water temperatures (at the intake structures) and air temperatures (recorded at the 33 foot level of the Millstone meteorological tower) were obtained

from the Northeast Utilities Environmental Data Acquisition Network (EDAN). Daily averages of 15-minute values were calculated for the period June 1976 to June 1993.

**Wind Speed and Direction** - Wind speed and direction (at the 33 foot level of the Millstone meteorological tower) were extracted from the EDAN database for each 15-min interval from June 1976 to June 1993. These values were used to calculate a wind index, which was the wind speed weighted by wind direction. A navigational chart of the sampling area was used to calculate site-specific wind directional weighting coefficients. The directional weight ranged from 0, when wind could not influence the station, to a maximum of 1, when wind-induced waves could directly affect the area. The wind index was then computed by multiplying the directional weight by the wind speed. Because the effect of wind was assumed to be cumulative, daily averages were derived using only wind index values greater than 0 (that is, when the wind was from a direction which could produce wind effects).

**Climatic Extremes (Deviations)** - Additional explanatory variables were created to represent unusual climatic conditions which occurred during the sampling period. High or low deviations (i.e., extremes) were derived for wind, rain, water and air temperature data and calculated as the difference between the quarterly mean or daily value and the 16-year mean (1977-93) for that quarter. Deviations based on quarterly means reflect the effects of longer-term extremes (e.g., an unusually cold winter), while those based on daily values tend to remove the effects of shorter-term episodic events (e.g., storms). Daily deviations were averaged and also summed in each sampling quarter to assess cumulative effects.

**Sedimentary Parameters** - Sediment mean grain size and silt/clay content were obtained as part of the monitoring studies and these quarterly values were used directly as explanatory variables in the multiple regression models.

**Reproductive - Recruitment Component** - Infaunal organisms in the Millstone area exhibit annual peaks in abundance, often reflecting the seasonal nature of reproduction and recruitment cycles or periods of favorable climatic conditions. Spectral analyses of quarterly data showed annual cycles in community abundance and number of species. To account for this periodicity, harmonic

terms (Lorda and Salla 1986) having a period of 1 year were also included as explanatory variables in the regression models.

In all, 32 variables were initially used during model selection steps. These variables included two sedimentary parameters, two seasonal/reproductive components and seven climatic variables, each of which had four values representing daily and quarterly high and low extremes.

## Response Modeling and Trend Analysis

Quarterly abundance and species number data from September 1979 to June 1985 (2-unit period) and September 1985 to June 1992 (3-unit period) were separately detrended using a linear regression model. A step-wise multiple regression was then applied to the residuals of quarterly data (i.e., the variability or "noise" about the linear trend) over the entire sampling period to identify explanatory factors and combinations of factors whose regression coefficients were significantly different from zero ( $p \leq 0.05$ ). This probability level was chosen to guard against fitting more parameters than could be reliably estimated, given the sample size. The model that minimized the mean-square-error and maximized the  $R^2$  was selected to "clean" the original data, and produce a time-series free from variation attributable to concomitant physical factors or known biotic processes such as reproductive/recruitment cycles. Linear models were then refitted to the variance-reduced time-series corresponding to the 2-unit and 3-unit periods. The nonparametric (i.e., distribution-free) Mann-Kendall test (Hollander and Wolfe 1973) was next used to determine whether these 2-unit and 3-unit series exhibited significant trends and Sen's nonparametric estimator of the slope (Sen 1968) was used to test for trend differences. These two tests were suggested by Gilbert (1989) as particularly well suited for analyzing environmental monitoring data because no distributional assumptions are required, and because relatively short time series ( $n < 10$ ) are acceptable. In this report, plots of the original quarterly data, adjusted data and a graphical representation of the linear trend are provided for community abundance, numbers of species and for selected taxa.

Abundances of the top ten numerically dominant

taxa collected at each station were used to construct cumulative abundance curves (k-dominance curves) for 2-unit and 3-unit operational periods. Comparison of k-dominance curves has been suggested as a means of assessing shifts in the structure of macrofaunal communities (Warwick 1986; Warwick et al. 1987). Curves were constructed by plotting percentages of cumulative abundance (ordinate) versus the natural logarithm of a taxon's rank (abscissa). To assess possible shifts in infaunal community structure between 2-unit and 3-unit operational periods, the same Gompertz function used for sediment data analysis was fitted to cumulative abundance data by substituting species abundance for sediment weight, and species rank for particle size in the equation. Two-sample t-tests were used to compare parameters of curves representing data collected during the 2-unit and 3-unit operational periods.

## Results

### Sedimentary Environment

Sediments at subtidal sampling stations in the vicinity of Millstone during 1993 were comprised of fine to coarse sands. The finest sediments were found at IN, where quarterly grain size means were 0.18 to 0.30 mm (Fig. 2). Sediments were most coarse at EF; the quarterly grain size range was 0.50-0.62 mm. Intermediate sizes were observed at JC and GN (ranges of 0.21-0.25 mm and 0.29-0.50 mm, respectively). Quarterly silt-clay contents were highest at JC (18.7-22.5%), lowest at EF (1.3-3.8%) and intermediate at GN and IN (12.1-13.6% and 5.7-10.1%, respectively). Sediment grain size and silt-clay contents observed during 1993 were within the ranges of these two sediment parameters during both 2-unit and 3-unit operational periods.

Cumulative curves based on sediment sieve fraction weights (Fig. 3) were used to characterize subtidal environments, and allowed us to statistically compare sediments collected at each station during 1993 to those collected over 2-unit (1979-85) and 3-unit (1986-93) operational periods. Based on t-tests of Gompertz parameters derived from these curves, significant differences between 2-unit and 3-unit periods were noted only at EF and JC. Differences between 2-unit and 1993 sediments were also significant at both stations.

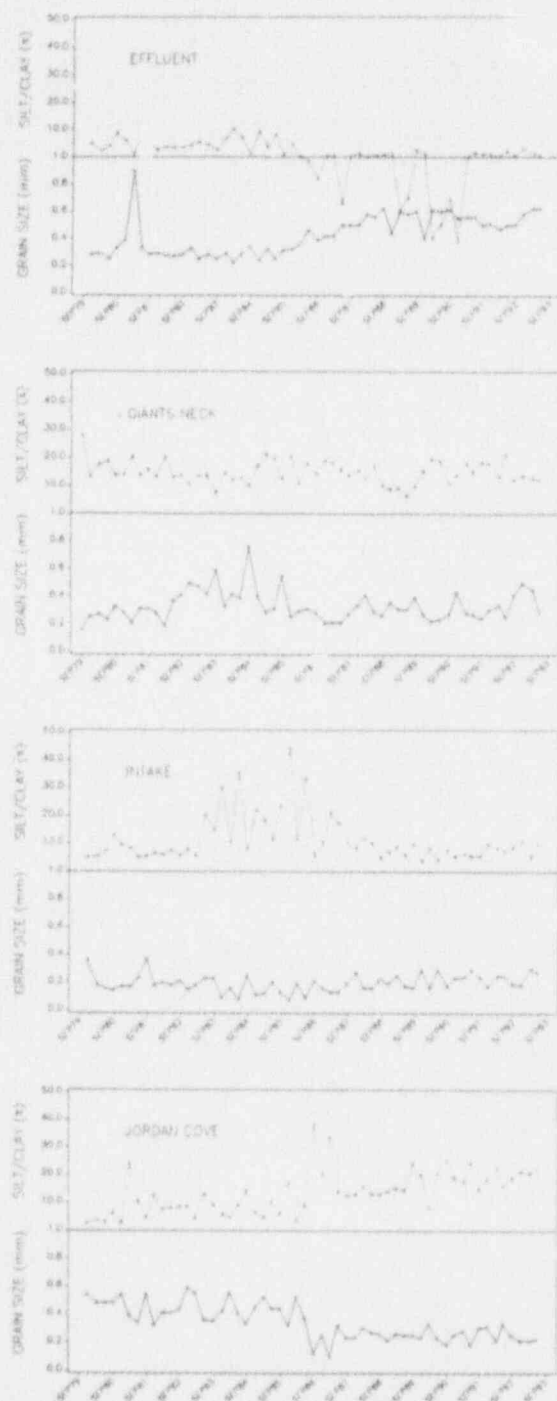


Fig. 2. Quarterly mean grain size (mm) and silt/clay content (%) of sediments at Millstone subtidal stations from September 1979 to June 1993.

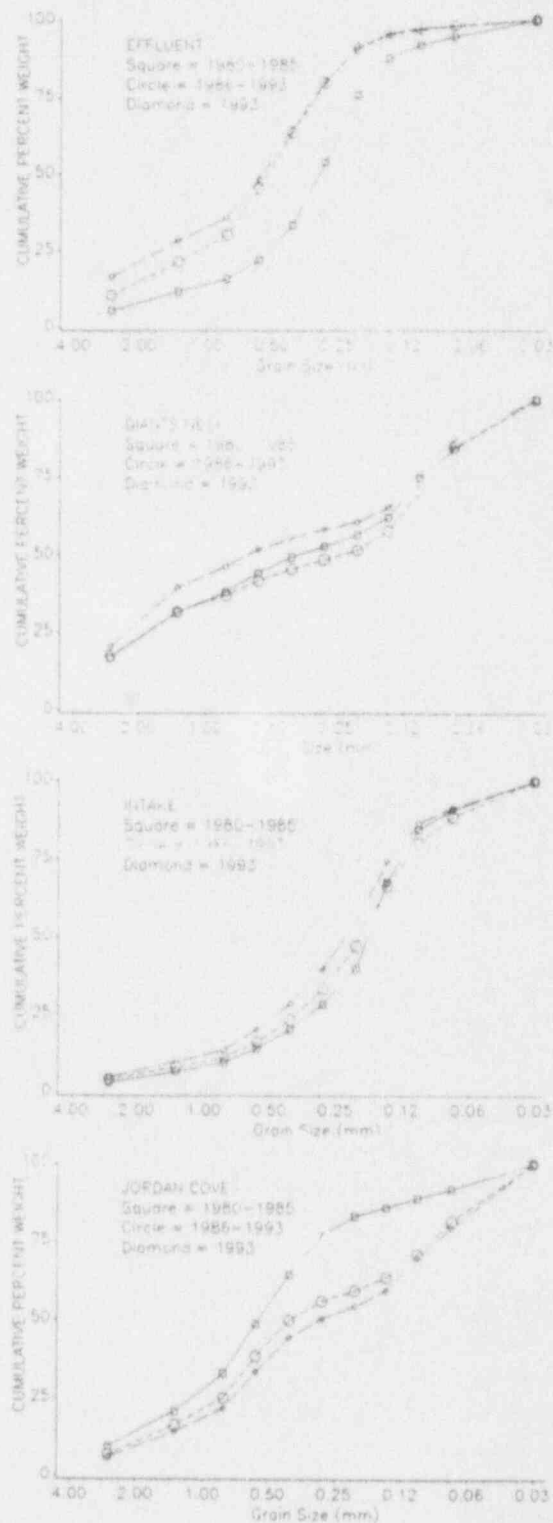


Fig. 3. Cumulative curves based on fractional weights of sediments collected during the 2-unit (1980-1985) and 3-unit (1986-1993) operational periods, and during 1993, at Millstone subtidal stations.

TABLE 1. Annual mean number of species (S), number of individuals (N) of each major taxon collected in 1993, during 2-unit (1980-1985) and 3-unit (1986-1993) operational years at Millstone subtidal stations.

	1993		2-Unit Period (1980-1985)				3-Unit Period (1986-1993)			
	(S)	(N)	MEAN (S)	CV*	MEAN (N)	CV	MEAN (S)	CV	MEAN (N)	CV
<u>Effluent</u>										
Polychaeta	60	2433	67	2.7	4675	17.7	62	2.8	2579	12.2
Oligochaeta	-	2703	-	-	2885	13.9	-	-	4289	11.0
Mollusca	23	214	29	4.8	497	29.3	27	4.1	472	19.1
Arthropoda	25	418	39	4.5	723	21.9	30	5.2	416	9.5
Rhynchocoela	-	532	-	-	138	23.2	-	-	182	28.6
'Others'	5	51	4	20.4	11	48.7	4	8.5	146	30.7
Total	113	6351	139		8930		123		8084	
<u>Giants Neck</u>										
Polychaeta	63	9256	67	4.4	6683	12.9	60	5.1	7128	8.5
Oligochaeta	-	2717	-	-	1932	12.6	-	-	2253	5.4
Mollusca	29	416	20	9.9	260	20.7	26	5.8	282	9.9
Arthropoda	29	1670	35	4.6	624	5.8	32	8.6	1052	21.6
Rhynchocoela	-	172	-	-	62	20.4	-	-	74	20.1
'Others'	3	46	3	26.4	8	43.2	3	12.4	17	28.9
Total	124	14277	125		9569		121		10806	
<u>Intake</u>										
Polychaeta	63	5070	43	3.3	1110	9.3	52	3.9	3000	21.8
Oligochaeta	-	706	-	-	253	16.2	-	-	397	23.5
Mollusca	22	303	18	10.8	199	27.0	20	4.6	503	13.8
Arthropoda	32	858	25	8.2	829	47.4	27	5.0	1710	53.1
Rhynchocoela	-	39	-	-	15	26.4	-	-	25	19.8
'Others'	1	1	1	68.3	1	74.2	1	33.3	2	35.4
Total	118	6977	87		2405		100		5637	
<u>Jordan Cove</u>										
Polychaeta	72	10153	64	4.7	6513	23.2	63	2.7	7887	13.2
Oligochaeta	-	2070	-	-	4124	24.2	-	-	2706	7.3
Mollusca	25	708	24	12.8	446	24.6	28	3.5	717	8.0
Arthropoda	26	1608	27	6.2	641	55.4	27	6.0	964	35.1
Rhynchocoela	-	172	-	-	79	12.3	-	-	91	13.8
'Others'	3	7	3	33.1	4	28.1	4	7.3	12	33.3
Total	126	14718	118		12110		122		12377	

\* C.V. of the mean estimate = (Standard Error/Mean) x 100



Sediments at IN in 1993 were significantly different (coarser) from those collected during both 3-unit and 2-unit periods. The shift at EF reflected the declining silt/clay fraction and the increasing grain size since Unit 3 began operation (Fig. 2). Conversely, silt/clay content increased and average grain size decreased over the same time period at JC.

### General Community Composition

Mean numbers of species and of individuals in major invertebrate groups collected during 1993, and during 2-unit and 3-unit operating periods are presented in Table 1. The annual mean numbers of species at subtidal stations in 1993 ranged from 113 (EF) to 126 (JC). The 1993 mean at EF was lower than the means for 2-unit and 3-unit periods (139 and 123, respectively); during both operational periods, mean number of species had been highest at EF in relation to operational means at other stations. In contrast, the 1993 means at IN (118) and JC (126) were higher than 2-unit (87 and 118, respectively) and 3-unit (100 and 121, respectively) operational means. Number of species at GN during 1993 (124) was lower than the 2-unit mean (125), but higher than the 3-unit mean (121).

Similar to spatial trends in number of species, the largest number of individual organisms was collected at JC (14,718), and the smallest at EF (6,351) in 1993. Means were also highest at JC over both 2-unit (12,110) and 3-unit periods (12,376); low 2-unit and 3-unit means were both recorded at IN (2,405 and 5,635, respectively). The 1993 mean number of organisms at IN was higher (6,977) than the two operational period means. The number of individuals collected at GN in 1993 (14,277) was also higher than the means of 9,569 and 10,806 for 2-unit and 3-unit periods, respectively. Fewer individuals were collected at EF in 1993 than over the two operational periods (2-unit mean: 8,930, 3-unit mean: 8,084).

Polychaetes were the dominant group in terms of numbers of species at all stations during 1993, ranging from 60-72 species. Polychaetes were also the most abundant taxon in the 2-unit and 3-unit operating periods (ranges 43-67 and 52-63, respectively). Except at EF during the 3-unit period and during 1993, polychaetes also dominated in terms of numbers of individuals

(nearly 50% of total individuals). Mollusc and arthropod species were not as abundant as polychaetes; ranges of numbers of species were 22-29 and 25-32, respectively. Numbers of molluscs and arthropods were generally similar between the operating periods. At EF during 3-unit operation, including 1993, oligochaetes were dominant. At GN and JC, oligochaetes were second in abundance (after polychaetes) followed by arthropods and molluscs. Arthropods ranked second in abundance at IN, followed by oligochaetes and molluscs. Rhynchocoels and "Others" contributed little to total abundance during either operational period.

Some notable differences in general community composition between operational periods are evident, based on species number. During the 3-unit period, the polychaete-dominated community at EF present during the 2-unit operation was replaced by one dominated by oligochaetes. Abundances of polychaetes and oligochaetes were similar during 1993. Also, the number of arthropod species at EF was lower during 3-unit operation than during the 2-unit period. The opposite trend was observed at JC; i.e., more polychaetes and arthropods, and fewer oligochaetes were collected, on average, in the 3-unit period than in the 2-unit period. At IN during 3-unit operation numbers in all taxonomic groups increased. At GN, there was little change in numbers between operational periods.

### Community Abundance

Ranges of average quarterly abundance (per core) at subtidal stations during 1993 were 134-194 at EF, 279-408 at GN, 119-314 at IN, and 292-369 at JC (Fig. 4). At each station, 1993 densities were within the range for their respective 14-year time series. In general, infaunal abundance at all stations during the 3-unit period (1986-1993) was similar to that observed during the 2-unit operational years. Overall, seasonal and annual fluctuations were lowest at EF and highest at IN; however, no consistent seasonal periodicity was evident in community abundance at any station during the 14-year period. Analyses of long-term trends in community abundance indicated a significant ( $p > 0.01$ ) increasing trend at EF during 2-unit operation, resulting primarily from peak abundances recorded near the end of that period

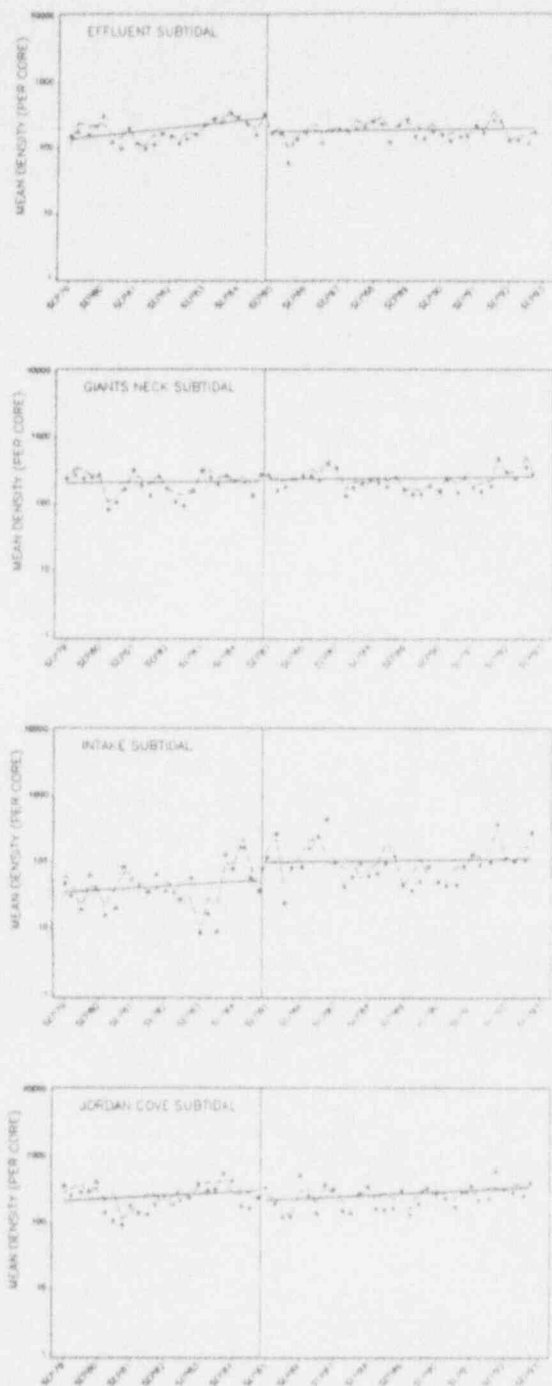


Fig. 4. Quarterly abundance data (dots), and variance-reduced data (dashed line), and linear-trends for subtidal communities before and after Unit 3 operation at MNPS.

(1984). During the 3-unit operating period, a significant increasing trend was detected only at JC.

## Numbers of Species

The mean number of species (per core) collected during 1993 ranged from 21-28 at EF, 26-35 at GN, 25-30 at IN, and 30-38 at JC (Fig. 5). These means were within the range of annual means observed at each station over the previous 14-year period. A significant increasing trend in quarterly species number was evident at EF and GN during 2-unit operation, and coincided with high species richness evident at most stations during the period 1984-87. During the 3-unit operating period, there were no significant trends in species richness evident at EF, GN, or JC. However, at IN, there was a significant increasing trend in quarterly numbers of species evident over the 3-unit operating period (slope=0.194).

## Community Dominance

The dominant taxa collected during 1993 at subtidal stations included the polychaete species *Aricidea catherinae*, *Tharyx* spp., *Prionospio steenstrupi*, *Polycirrus eximius*, *Scoletema tenuis*, *Protodorvillea gaspeensis*, *Mediomastus ambiseta*, *Pygospio elegans*, and the arthropods *Ampelisca vadorum*, *A. verrilli* and *Leptocheirus pinguis* and representatives of the class Oligochaeta (Table 2). The top four ranked taxa at each station in 1993 accounted for 50% or more of all individuals, and were: Oligochaeta, *Protodorvillea gaspeensis*, Rhynchocoela, *Tharyx* spp. at EF; *Tharyx* spp., Oligochaeta, *Aricidea catherinae*, *Ampelisca vadorum* at GN; *Aricidea catherinae*, *Pygospio elegans*, *Prionospio steenstrupi*, Oligochaeta at IN; *Aricidea catherinae*, Oligochaeta, *Leptocheirus pinguis*, *Polycirrus eximius* at JC (Table 2). In most cases, these organisms have been the dominant subtidal taxa in both 2-unit and 3-unit operational periods. There was a large increase in relative abundance of the polychaete *Aricidea catherinae* at IN and JC in 1993, which accounted for 16.3% and 26.0% of the individuals collected at these stations, respectively.

Increase in abundances of *Pygospio elegans* and *Prionospio steenstrupi* were also noted at IN in 1993, and a decrease in abundance of oligochaetes was observed at JC. *Tharyx* spp. relative abundance was higher at all sites in 1993, and particularly at GN, where this taxon accounted for 34.1% of individuals, compared to 13.8% and

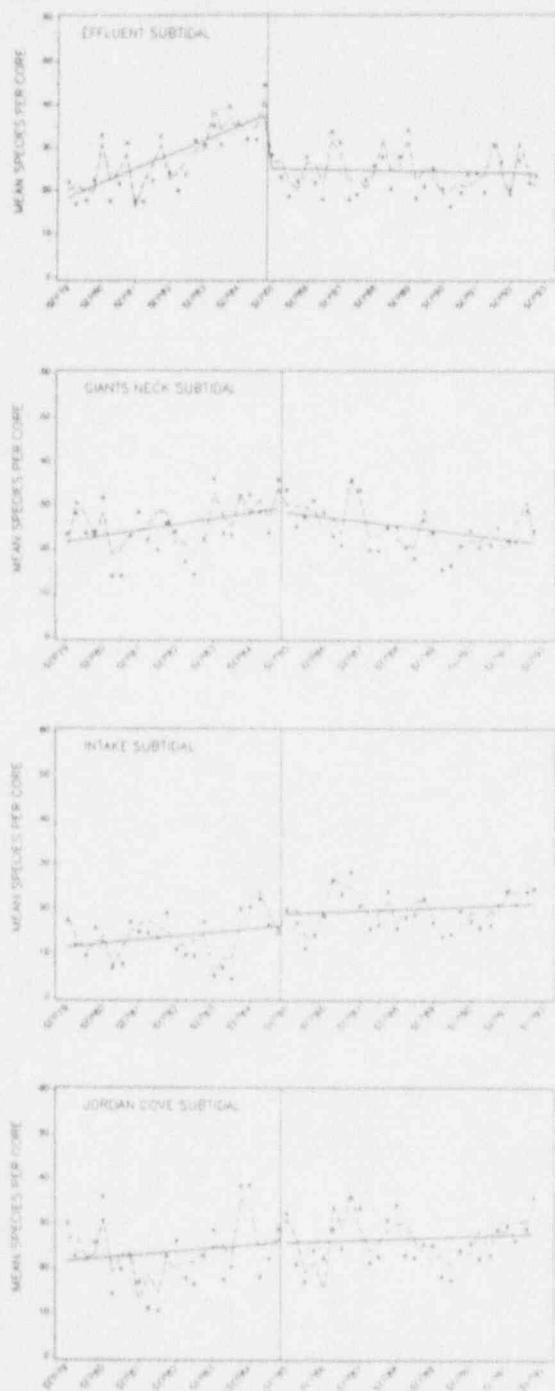


Fig. 5. Quarterly number of species data (dots), and variance-reduced data (dashed line), and linear-trends for subtidal communities before and after Unit 3 operation at MNPS.

18.5% over 2-unit and 3-unit years, respectively. The increase of *Tharyx* spp. at GN was mirrored by a marked decrease in relative abundance of *Aricidea catherinae*.

Relative abundances of dominant taxa between operational periods have exhibited considerable consistency, with a few exceptions. For example, oligochaetes continued to be the most common taxon overall during 3-unit operation (accounting for 8.0 to 52.8% of individuals), as they were during 2-unit operation (10.3 to 40.8%); however, as discussed in the previous section, oligochaetes were more abundant at EF and less abundant at JC during 3-unit operation. Most stations were characterized by one or more clearly dominant taxon (oligochaetes at EF, GN and JC, *Aricidea catherinae* at GN and JC and *Tharyx* spp. at GN) during both operational periods. There has been no single dominant taxon at IN during either operational period, where mean relative abundances of any single taxon collected there rarely exceeded 10%.

### Dominant Taxa

Eight subtidal taxa have been identified as being affected or potentially affected by construction and operation of MNPS. Trends in the abundance of these taxa were examined using the same techniques as those applied to overall community abundance and numbers of species. For a review of the general ecology of these dominant taxa, refer to NUSCO (1992).

**Oligochaetes** - Oligochaetes were ranked first in overall abundance during both 2-unit and 3-unit operating periods, accounting for 10-41% and 8-53% of total individuals, respectively (Table 2). During 1993, oligochaete abundances were generally highest at GN (35-92/core) and EF (45-75/core), lowest at IN (14-21/core) and intermediate at JC (33-65/core). These densities were within the ranges of densities in previous study years.

Analyses of trends in oligochaete abundances revealed an increase at EF, and a decline at GN during 2-unit operation (Fig. 6a and b). Trend analysis for the 3-unit period indicated a significant increase in oligochaete abundance at IN (Fig. 6c). At EF, GN and JC, oligochaete abundance has remained at similar levels during the 3-unit operating period (Fig. 6a, b and d).

***Aricidea catherinae*** - *A. catherinae* was among the dominant taxa at all subtidal stations during 2-unit

TABLE 2. Mean relative abundance<sup>a</sup> (%) and coefficient of variability (CV<sup>b</sup>) of each of the ten most abundant taxa collected at the Millstone subtidal stations during 1993, 2-Unit operational years (1980-1985) and 3-Unit operational years (1986-1993).

	1993	2-Unit Period (1980-1985)		3-Unit Period (1986-1993)	
		MEAN	CV	MEAN	CV
	%	%		%	
<u>Effluent</u>					
<i>Oligochaeta</i>	42.6	32.9	4.4	52.8	1.4
<i>Protodrilus guspeensis</i>	8.9	4.6	13.3	8.0	4.0
<i>Rhynchocoela</i>	8.4	2.4	12.2	3.0	15.9
<i>Tharyx</i> spp.	4.8	2.6	28.1	3.0	18.2
<i>Polycirrus eximius</i>	3.2	10.8	10.2	3.3	23.7
<i>Aricidea catherinae</i>	3.0	4.1	20.6	1.6	31.4
<i>Ampharete americana</i>	1.9	-	-	1.6	24.1
<i>Scoletema tenuis</i>	1.9	-	-	1.9	14.5
<i>Ampelisca vadorum</i>	1.7	-	-	1.5	30.8
<i>Chyrenella mucosa</i>	1.6	-	-	1.5	32.5
<i>Parapionosyllis longicirrata</i>	-	1.6	18.8	2.1	14.5
<i>Haliplanella luciae</i>	-	1.3	70.7	2.2	26.8
<i>Pagurus acadianus</i>	-	-	-	2.0	13.4
<i>Prionospio steenstrupi</i>	-	-	-	1.9	32.6
<i>Spiophanes bombyx</i>	-	-	-	1.6	28.4
<i>Harmothoe imbricata</i>	-	-	-	1.4	86.8
<i>Eumida sanguinea</i>	-	-	-	1.3	24.5
<u>Giants Neck</u>					
<i>Tharyx</i> spp.	34.1	13.8	3.9	18.5	4.8
<i>Oligochaeta</i>	19.0	20.8	3.9	22.1	2.2
<i>Aricidea catherinae</i>	7.4	19.6	5.0	15.3	5.7
<i>Ampelisca vadorum</i>	4.8	1.2	18.5	2.7	24.0
<i>Exogone dispar</i>	3.5	2.7	19.6	3.1	7.1
<i>Prionospio steenstrupi</i>	3.0	2.0	16.1	4.0	27.9
<i>Ampharete americana</i>	1.9	-	-	1.5	26.1
<i>Scoletema tenuis</i>	1.9	3.1	14.1	3.6	6.9
<i>Polydora quadrilobata</i>	1.5	-	-	1.8	26.8
<i>Mitrella lunata</i>	1.4	-	-	1.4	30.8
<i>Capsella</i> spp.	-	1.8	19.4	1.8	10.7
<i>Tellina agilis</i>	-	1.5	25.8	1.5	38.4
<i>Pagurus acadianus</i>	-	1.3	34.3	1.6	21.1
<i>Mediomastus ambiseta</i>	-	4.8	27.5	4.3	25.9
<i>Phoxocephalus holboellii</i>	-	2.7	13.0	2.0	13.5

<sup>a</sup> Based on log-transformed data

<sup>b</sup> C.V. of the mean estimate = (Standard Error/Mean) X (100)

- = Not among the dominant taxa

TABLE 2. continued.

	1993	2-Unit Period (1980-1985)		3-Unit Period (1986-1993)	
	%	MEAN %	CV	MEAN %	CV
<u>Intake</u>					
<i>Aricidea catherinae</i>	16.3	6.7	22.4	6.0	17.3
<i>Pygospio elegans</i>	12.4	-	-	3.4	28.6
<i>Prionospio stenstrupi</i>	10.5	2.2	35.8	4.3	29.7
Oligochaeta	10.1	10.3	11.1	8.0	10.9
<i>Tharyx</i> spp.	9.7	-	-	4.3	13.4
<i>Exogone hebes</i>	6.8	3.6	29.1	5.2	20.9
<i>Ampelisca vadorum</i>	5.0	-	-	3.1	22.4
<i>Ampelisca verrilli</i>	2.5	5.1	31.1	3.2	20.5
<i>Protodorvillea gaspensis</i>	2.4	-	-	2.6	22.1
<i>Spiophanes bombyx</i>	2.0	2.5	30.6	2.2	16.6
<i>Capitella</i> spp.	-	3.0	25.3	4.2	21.4
<i>Sabellaria vulgaris</i>	-	-	-	1.3	60.2
<i>Nucula proxima</i>	-	2.9	26.2	4.3	20.8
<i>Mediomastus ambiseta</i>	-	-	-	4.0	27.2
<i>Leptochieris pinguis</i>	-	-	-	3.0	39.2
<i>Ampelisca abdita</i>	-	-	-	3.0	53.0
<i>Tellina agilis</i>	-	3.9	19.3	3.9	17.9
Maldanidae	-	-	-	1.5	19.3
<i>Owenia fusiformis</i>	-	-	-	2.5	26.3
<i>Gammarus lawrencianus</i>	-	2.2	40.0	1.5	54.9
<u>Jordan Cove</u>					
<i>Aricidea catherinae</i>	26.0	14.8	6.1	15.9	8.3
Oligochaeta	14.1	40.8	3.9	23.0	4.0
<i>Leptochieris pinguis</i>	9.3	1.5	34.5	3.9	30.6
<i>Polycirrus eximius</i>	7.8	4.3	14.2	5.0	12.4
<i>Tharyx</i> spp.	5.2	3.1	13.3	4.1	5.0
<i>Prionospio stenstrupi</i>	4.7	1.4	24.8	5.0	18.6
<i>Scoletema tenuis</i>	4.6	4.7	13.1	5.9	6.3
<i>Mediomastus ambiseta</i>	3.9	7.2	26.3	8.3	16.8
<i>Polydora quadrilobata</i>	1.5	-	-	1.5	38.2
<i>Spiophanes bombyx</i>	1.5	-	-	1.4	37.4
<i>Nucula proxima</i>	-	1.3	13.1	2.8	17.3
<i>Capitella</i> spp.	-	2.2	17.9	2.3	17.2
<i>Tellina agilis</i>	-	2.2	23.6	1.9	14.9
<i>Microphthalmus aberrans</i>	-	1.5	22.6	2.9	18.6
<i>Cossura longocirrata</i>	-	-	-	1.5	26.1



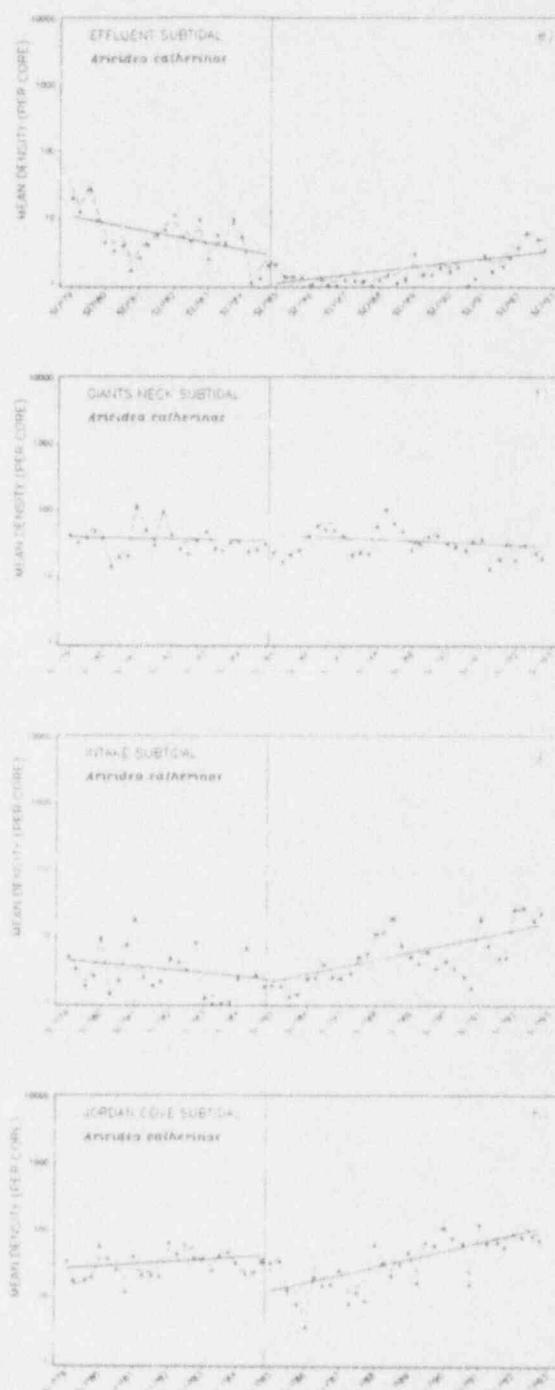
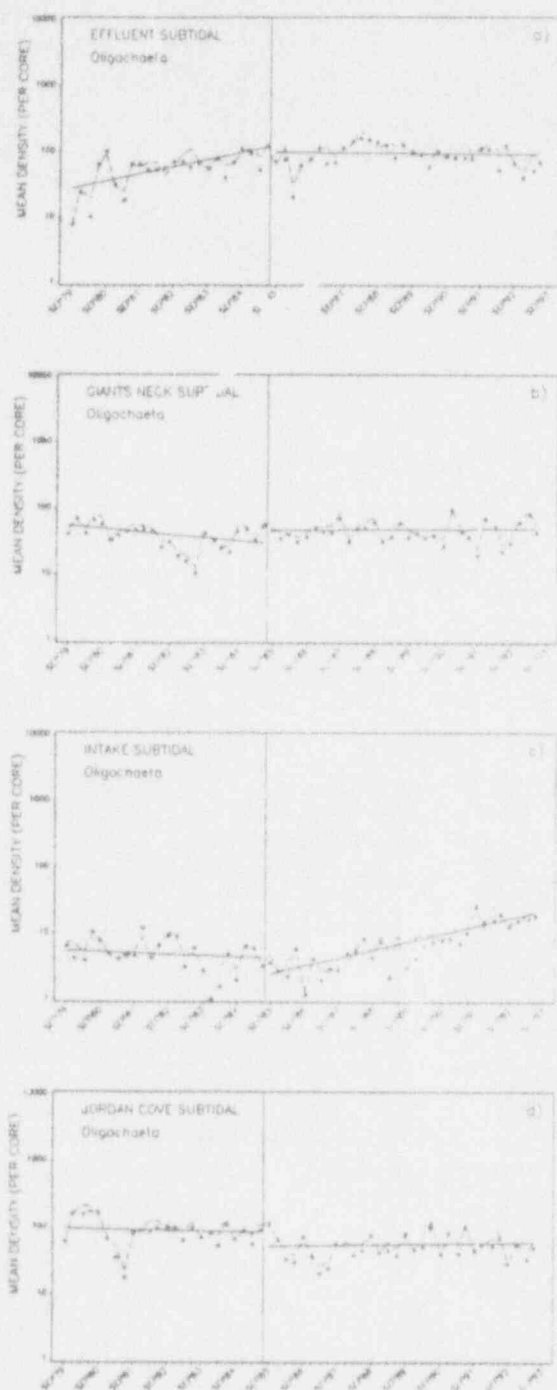


Fig. 6. Quarterly abundance data (dots), and variance-reduced data (dashed line), and linear-trends for selected dominant organisms comprising Millstone subtidal communities before and after Unit 3 operation at MNPS.

operation, and at most subtidal stations, except EF, during the 3-unit operating period (Table 2). The range in average density during 1993 was 4-7/core

Fig. 6. continued.

at EF, 19-31/core at IN, 20-34/core at GN, and 84-103 at JC.

The average quarterly abundance of *Aricidea catherinae* exhibited several trends over both 2-unit and 3-unit operating periods (Fig. 6e-h). Relationships among operational period trends at

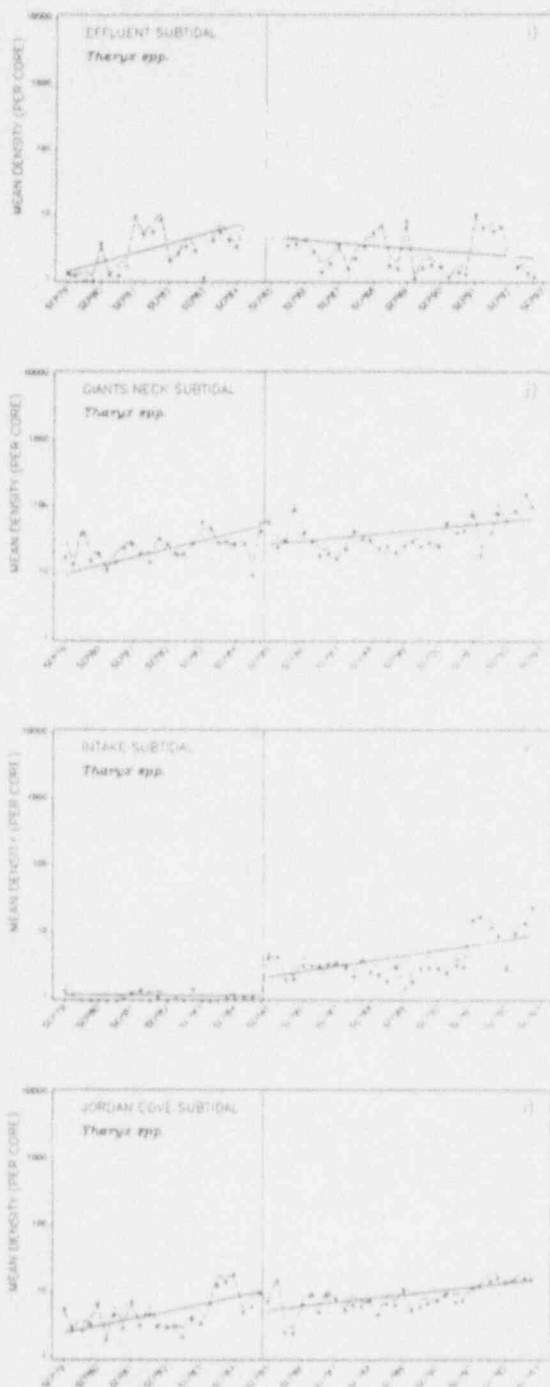


Fig. 6, continued.

EF and IN were similar; at both sites, abundance of *A. catherinae* significantly declined during 2-unit operation, and significantly increased after Unit 3 start-up. Abundance of *A. catherinae* also increased at JC during 3-unit operation. Abundances have been consistently high at GN

throughout our studies with no apparent trend in either operational period.

*Tharyx* spp. - *Tharyx* spp. were among the dominant taxa at most subtidal stations during 2-unit and 3-unit operating periods. During 1993, densities of *Tharyx* exhibited high variability among subtidal stations, ranging from 1-3/core at EF, 69-171/core at GN, 3-26/core at IN and 15-18/core at JC (Fig. 6i-iv). At the GN reference station, *Tharyx* spp. densities were consistently high, ranking third, second and first in the 2-unit, 3-unit operating periods and 1993, respectively (Table 2). At EF and JC, *Tharyx* spp. ranking was also consistent, but at a lower level over the entire study period, ranking fifth during both operating periods and fourth in 1993 at EF, and sixth during both periods and fifth in 1993 at JC. *Tharyx* spp. were among the numerical dominants at IN only during the 3-unit operating period, ranking fifth in abundance in 1993, and fourth for the overall 3-unit period. This taxon was virtually absent from that site during the 2-unit period.

Results of trend analysis on *Tharyx* spp. abundance during 2-unit operation indicate a significant increasing trend at EF, GN and JC. Trend analysis of 3-unit operating data indicated a continued significant increase in the abundance of *Tharyx* spp. at JC and the reference station GN. During the same period, there were no trends evident at IN, despite increases in recent years, or at EF.

*Polycirrus eximius* - During 1993, *P. eximius* was among the dominant taxa at JC and EF, ranking fourth and fifth (Table 2), and averaging 12-72/core and 2-12/core, respectively (Fig. 6m-n). Trend analysis indicated significant increasing densities at both stations during both operational periods. However, the 1993 March and June abundances at EF were among the lowest values reported for the 3-unit operating period. In contrast, JC densities were within the range of values reported for the 3-unit period, with the September value among the highest reported during both 2-unit and 3-unit operating periods. Historically, *P. eximius* has exhibited both seasonal periodicity and regional long-term cycles at all stations except IN (NUSCO 1993).

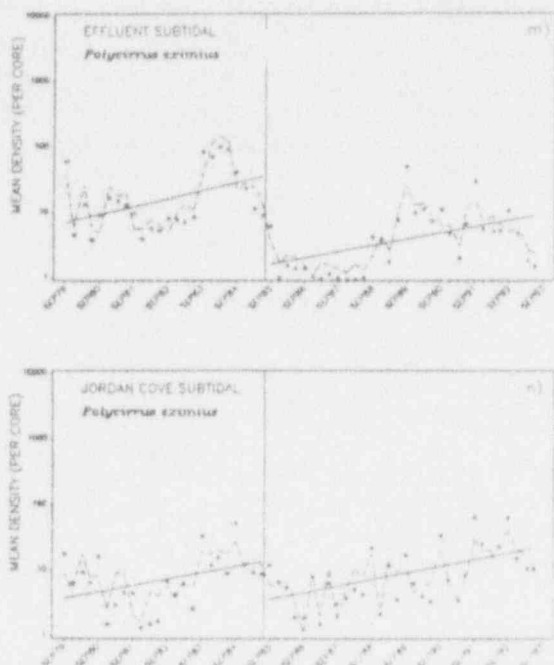


Fig. 6, continued.

*Scoliotema tenuis* - *S. tenuis* was a dominant component of infaunal communities during the 2-unit and 3-unit operating period at JC (ranking fourth and third, respectively) and GN (ranking fifth for both periods; Table 2). Average densities during 1993 were 3-12/core and 8-33/core at GN and JC, respectively (Fig 6o-p). These densities were within the range of density values from previous years. At both GN and JC, an increasing trend was observed during 2-unit operation, likely due to low abundances of *S. tenuis* at the beginning of the time series at both sites. Low abundances in 1993 resulted in a significant decreasing trend for the 3-unit period at GN that had not been apparent in previous years (e.g., NUSCO 1992, 1993). Abundances at JC have been more consistent during 3-unit operation, with no significant trend detected over that period.

*Mediomastus ambiseta* - *M. ambiseta* abundances continued to decline in 1993 relative to the period 1984-87, when an area-wide pulse in abundance was reported (NUSCO 1984). The density range at the GN reference station during 1993 was 2-4/core, and at JC, *M. ambiseta* density ranged from 8-22/core (Fig. 6q-r). Trend analysis of 3-unit operating data indicated significant decreasing densities at both GN and JC. A similar, although

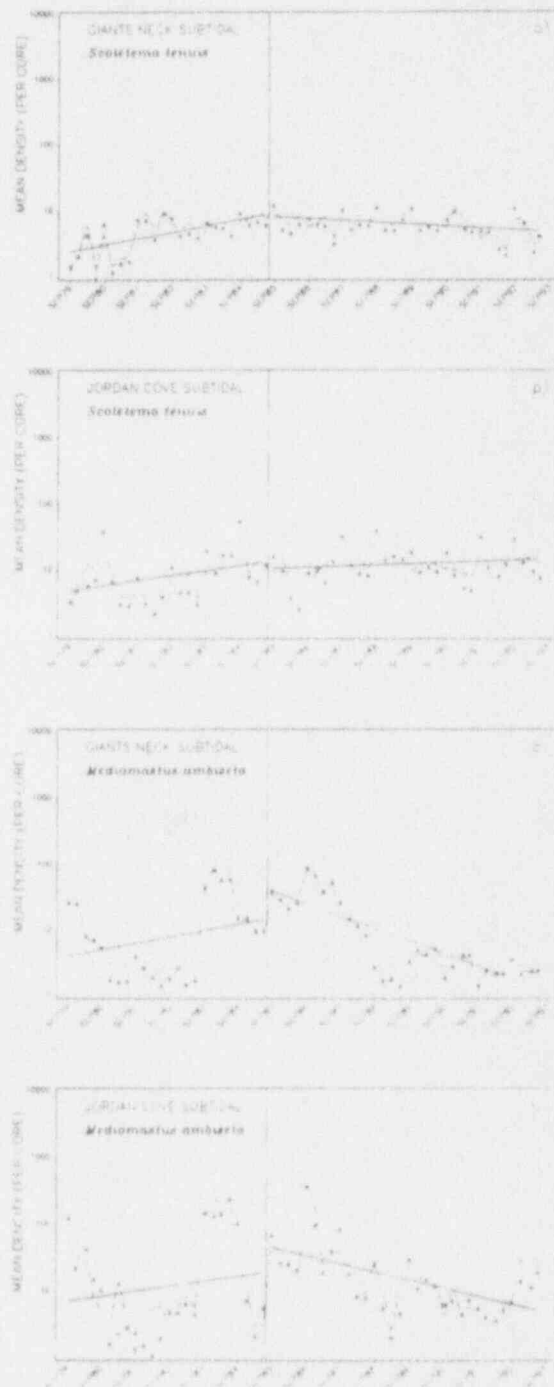


Fig. 6, continued.

less extreme, pulse/decline cycle was also noted at EF and IN, where, in 1993, *M. ambiseta* was not among the dominant taxa.

*Protodorvillea gaspeensis* - *P. gaspeensis* was among the dominant subtidal organisms at EF ranking third and second in the 2-unit and 3-unit operating periods, respectively (Table 2). During 1993, *P. gaspeensis* ranked second in abundance; quarterly values of *P. gaspeensis* were within the range of previously reported values, averaging 8-16/core (Fig. 6s). Significant increasing trends occurred at EF during both 2-unit and 3-unit operating periods. The increasing trend during the 2-unit period was attributed to low values in 1980. The 3-unit trend resulted from continued steady increases in the abundance of this species following Unit 3 start-up.

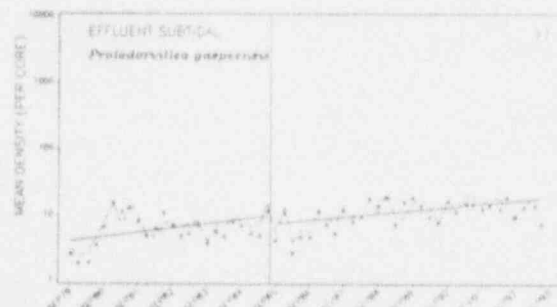


Fig. 6, continued.

*Nucula proxima* - This small bivalve ranked only seventh at IN during 2-unit operation, but was the fourth (tie) most dominant taxon during 3-unit operation (Table 2). *N. proxima* was not among the top ten dominants at IN during 1993; quarterly density values ranged from 1-2/core (Fig. 6t). This species was the tenth most dominant taxon during 3-unit operation at JC, but was not among dominant taxa during 2-unit operation or during 1993. Quarterly densities for 1993 at JC were 1-11/core (Fig. 6u). No significant trends were observed at either IN or JC during 2-unit operation. During 3-unit operation, *N. proxima* exhibited a decreasing trend at IN, and conversely, an increasing trend at JC.

### Cumulative Abundance Curves

Cumulative abundance curves representing subtidal communities at each station sampled over the 2-unit and 3-unit operational periods are

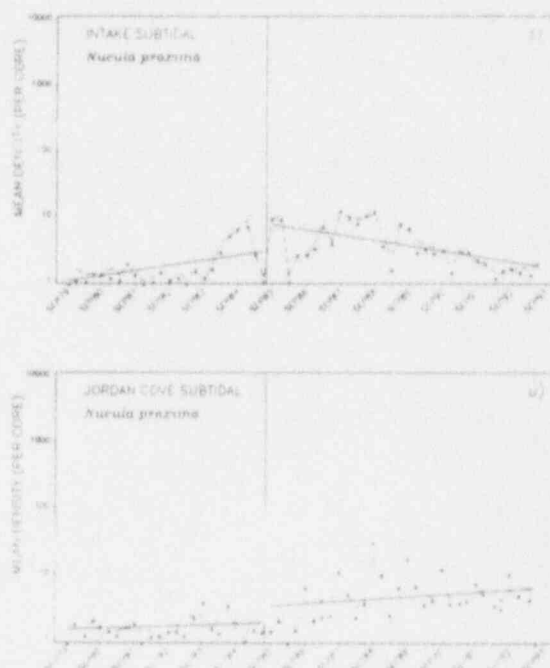


Fig. 6, continued.

presented in Fig. 7. At EF and JC, the location parameter (i.e., percent contribution of the top ranked organism) was significantly different between the two operating periods, reflecting changes in the overall contribution of the dominant taxon (i.e., oligochaetes). Additionally, changes in the abundances of other taxa from the 2-unit to 3-unit operational period caused the overall shapes of these curves to significantly differ between the two periods. There was no significant difference in the shape parameter between the 2-unit and 3-unit periods at GN or IN. This similarity reflects the consistent contribution of oligochaetes, *Aricidea catherinae* and *Tharyx* spp. at GN, and of oligochaetes, *A. catherinae* and *Ampelisca verrilli* at IN. The lower position of the 3-unit curve at JC reflects a shift toward increased equitability in the taxonomic distribution. In contrast, the high starting point of the 3-unit EF curve indicates the numerical dominance of the top ranked taxon, oligochaetes, during that period. The low starting point of the IN curves in both operating periods indicate that no single taxon was overwhelmingly dominant in either period (see Table 2).

## Discussion

Effective long-term monitoring of benthic infaunal communities is strengthened by identification of structuring mechanisms which produce characteristic fluctuations in species composition, abundance and dominance. However, even the first step in this process, i.e., separation of naturally induced physical and biological mechanisms, including naturally varying levels of mortality, recruitment, competition and vagaries in local physico-chemical conditions is often difficult (Watling 1975; Flint and Younk 1983; Nichols 1985; Watzin 1986; Rees and Eleftheriou 1989). An attempt has been made here (through modelling and regression analysis) to account for many of the important natural factors reported to effect differences observed in local benthic communities, and thereby isolate other factors possibly related to construction and operation of MNPS. Also, the establishment of two control strategies, i.e., temporal (sampling a potentially impacted site prior to impacts) and spatial (a site well beyond any power plant influence), has further improved our ability to separate natural factors from those related to MNPS. These methods of impact assessment, integrated into a single study design, have allowed us to document community changes that were independent of power plant operations, as well as those directly attributed to the construction and operation of Millstone Unit 3.

Several significant area-wide shifts in species abundance and community structure have been observed over the course of these studies, and have been described in detail in previous reports as unrelated to MNPS operation (e.g., NUSCO 1987, 1993). These included large increases in abundance of the opportunistic polychaete, *Mediomastus ambiseta*, and the amphipods, *Leptochemus pinguis* and *Annelisca* spp. which occurred over several years (1983-88; NUSCO 1989), as well as annual pulses in species abundance such as the substantial increase in abundance of the spionid polychaete, *Prionospio steenstrupi*, observed in 1992 (NUSCO 1993). These increases could not be explained by changes in site-specific sedimentary or regional climatic factors (NUSCO 1989, 1993); however, because these changes occurred at all stations, including our GN reference station located beyond any

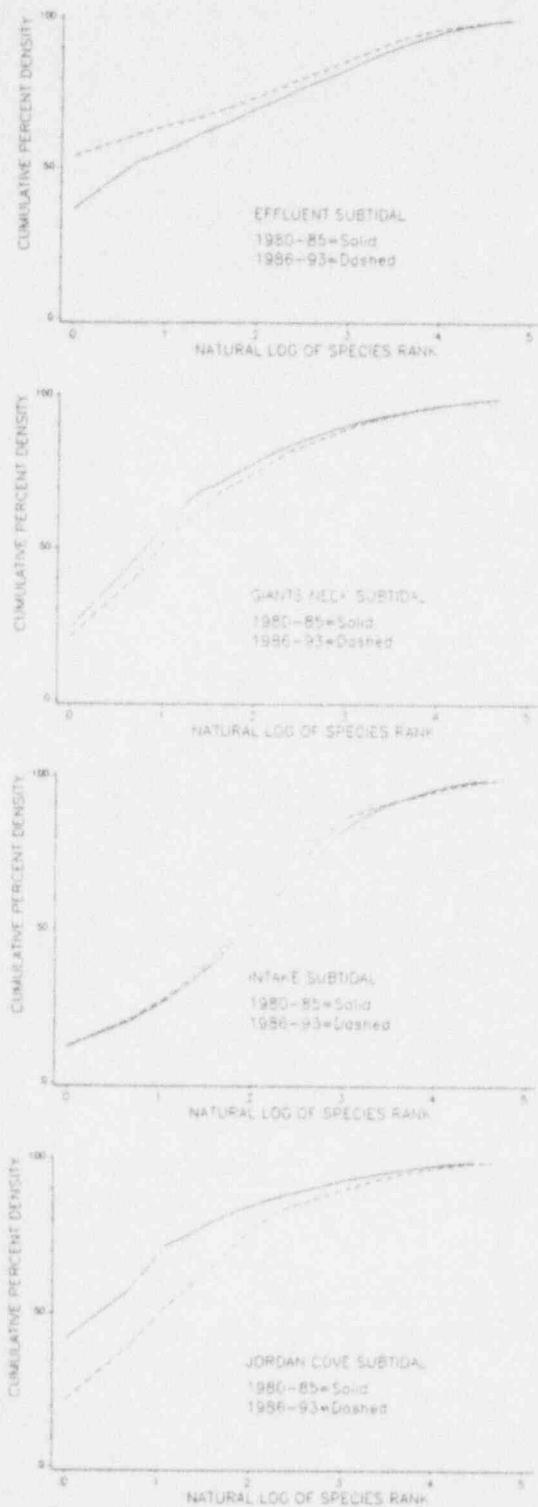


Fig. 7. Cumulative species abundance curves based on the ten most abundant organisms collected during the 2-unit (1980-1985) and 3-unit (1986-1993) operational periods at MNPS.



effects of MNPS, their cause was assumed to be of natural origin and independent of power plant construction or operation.

Along with substantiating the area-wide trends described above, GN showed long-term stability in sedimentary characteristics and infaunal communities throughout the 14-yr Millstone study. Sediments showed no significant changes over the entire study, either in terms of grain size or silt/clay content, and patterns of total infaunal abundance and community composition were, in general, consistent throughout the study. This community was numerically dominated by the same three taxa (oligochaetes, *Tharyx* spp. and *Aricidea catherinae*) in generally the same rank order and relative abundance, during both 2-unit and 3-unit periods.

In contrast to the temporal consistency observed at GN, all stations located near the power plant exhibited changes in both sedimentary character and infaunal communities beyond those believed to reflect natural processes. Rather, these site-specific changes are attributable to events associated with either construction (at IN) or operations (at EF and JC) at Millstone (NUSCO 1985, 1986, 1987). The impact of each of these events was most severe following initial detection, and lessened in subsequent years, when a degree of recovery was observed at each station.

Impacts were noted earliest at IN, and were associated with disturbances resulting from dredging and coffer dam removal during Unit 3 construction from 1983-85 (NUSCO 1987). Since that time, sediments (primarily silt/clay content levels) have stabilized and have become more similar to pre-impact years, and ongoing community recovery is evident. In particular, increases in the abundance of common species prior to 1983 (e.g., oligochaetes, *Aricidea catherinae*; Fig. 6c and g), with concomitant decreases in abundance of *Nucula proxima* (Fig. 6t) and other opportunistic species (NUSCO 1993) indicate that a recovery process has continued through 1993. However, other species which have established post-impact populations, such as *Tharyx* spp. (Fig. 6k), maintained a degree of dominance in 1993, which suggests that recovery at this site is not complete. Changes such as these are typical of those in marine benthic communities following disturbance (Kaplan et al. 1974; Swartz et al. 1980; Nichols 1985; Berge 1990).

A similar, relatively short-term, disturbance event resulted in sedimentary and infaunal changes at JC. In 1986, silt was scoured from the area of the Unit 3 discharge and settled at JC, increasing silt/clay content of sediments in this area. These substratum changes resulted in decreased abundances of the previously dominant taxa oligochaetes and the polychaetes *Polycirrus eximius* and *Aricidea catherinae* (NUSCO 1988a). This depositional event likely occurred over a relatively short period of time (months), and its impact has evidently lessened since 1986. Despite the long-term persistence of some of the deposited silt/clay at JC, reduced levels of silt/clay relative to those observed in 1987 have allowed some community recovery. For example, the abundance in 1993 of oligochaetes, and the polychaetes *A. catherinae* and *P. eximius* abundances in 1993 were at levels comparable to those observed during 2-unit years (Fig. 6d, h and m). These observations are consistent with those of other researchers studying the effects of siltation on benthic infaunal communities (Rhoads and Young 1970; Jumars and Fauchald 1977; Turk and Risk 1981; Mauer et al. 1986; Emerson 1989; Brey 1991).

In contrast to the short-term disturbance events described above at IN and JC, scour produced by the 3-unit discharge continues to affect both the sediments and the infaunal community at EF. Increased grain size and decreased silt/clay levels continued to be observed at EF during 1993. However, it appears that in recent years, sediment, community and species parameters have stabilized under the new environmental conditions created by the 3-unit discharge. Oligochaete abundance has been remarkably consistent over the 3-unit period, although at a level considerably higher than during the 2-unit period (Fig. 6a). Despite distinctly different sedimentary characteristics, this stabilization has allowed for the return of species common during 2-unit operation, such as *Aricidea catherinae* and *Polycirrus eximius* (Fig. 6e and n). The community at EF should continue to develop under 3-unit operating conditions, but true recovery of this community is not expected until power plant-induced scour ceases.

## Conclusions

Benthic infaunal studies continued to monitor subtidal soft bottom habitats in the vicinity of

MNPS during 1993 for changes in sedimentary parameters and faunal characteristics (total abundance, species number and species composition). We have documented the effects of both short-term episodic (e.g., dredging and construction activities at IN, and siltation at JC) and long-term continuous (effluent scour at EF) disturbance events, all attributed to MNPS operation, along with subsequent community recovery and/or development at all impacted sites throughout the period of 3-unit operation. Detection of these impacts was enhanced by successful identification of area-wide community changes (primarily fluctuations in species abundance) attributed to natural factors, unrelated to power plant operation. Distinction between natural and power plant effects on the infaunal communities is on-going and the primary objective of this program.

### References Cited

- Aller, R.C. 1978. Experimental studies of changes produced by deposit-feeders on pore water, sediment, and overlying water chemistry. *Am. J. Sci.* 278:1185-1234.
- Berge, J.A. 1990. Macrofaunal recolonization of subtidal sediments. Experimental studies on defaunated sediment contaminated with crude oil in two Norwegian fjords with unequal eutrophication status. I. community responses. *Mar. Ecol. Prog. Ser.* 66:103-115.
- Boesch, D.F., R.J. Diaz, and R.W. Virnstein. 1976. Effects of tropical storm Agnes on soft-bottom macrobenthic communities of the James and York estuaries and the lower Chesapeake Bay. *Chesapeake Sci.* 17:246-259.
- Boesch, D.F., and R. Rosenberg. 1982. Response to stress in marine benthic communities. Pages 179-200 in G.W. Barrett and R. Rosenberg, eds. *Stress Effects on Natural Ecosystems*. John Wiley, New York.
- Brey, T. 1991. The relative significance of biological and physical disturbance: an example from intertidal and subtidal sandy bottom communities. *Estuar. Coast. Shelf Sci.* 33:339-360.
- Commuto, J.A., and M. Boncavage. 1989. Suspension-feeders and coexisting infauna: an enhancement counterexample. *J. Exp. Mar. Biol. Ecol.* 125:33-42.
- Diaz, R.J., and L.S. Schaffner. 1990. The functional role of estuarine benthos. Pages 25-56 in *Contrib. 1595*. College of William and Mary, Virginia Inst. of Mar. Sci.
- Draper, N., and H. Smith. 1981. *Applied regression analysis*. John Wiley and Sons, New York. 709 pp.
- Emerson, C.W. 1989. Wind stress limitation of benthic secondary production in shallow, soft-sediment communities. *Mar. Ecol. Prog. Ser.* 55:65-77.
- Flint, R.W. 1985. Long-term estuarine variability and associated biological response. *Estuaries* 8:159-169.
- Flint, R.W., and J.A. Young. 1983. Estuarine benthos: long-term community structure variations, Corpus Christi Bay, Texas. *Estuaries* 6:126-141.
- Folk, D. 1974. *Petrology of Sedimentary Rocks*. Hemphill Publishing Company, Austin, Texas. 182 pp.
- Franz, D.R. and J.T. Tanacredi. 1992. Secondary production of the amphipod *Ampelisca abdita* Mills and its importance in the diet of juvenile winter flounder (*Pleuronectes americanus*) in Jamaica Bay, New York. *Estuaries* 15:193-203.
- Gaston, G.R., and J.C. Nasci. 1988. Trophic structure of macrobenthic communities in the Calcasieu Estuary, Louisiana. *Estuaries* 11:201-211.
- Gendron, L. 1989. Seasonal growth of the kelp, *Laminaria longicruris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. *Bot. Mar.* 32:345-354.
- Gilbert, R.O. 1989. *Statistical methods for environmental pollution monitoring*. Van Nostrand Reinhold Company, New York. 320 pp.
- Goldhaber, M.B., R.C. Aller, J.K. Cochran, J.K. Rosenfield, C.S. Martens, and R.A. Berner. 1977. Sulfate reduction, diffusion bioturbation Long Island Sound sediments: Report of the FOAM Group. *Am. J. Sci.* 277:193-237.
- Holland, A.F. 1985. Long-term variation of macrobenthos in a mesohaline region of Chesapeake Bay. *Estuaries* 8:93-113.
- Holland, A.F., A.T. Shaughnessy, and M.H. Hiegel. 1987. Long-term variation in mesohaline Chesapeake Bay macrobenthos: Spatial and temporal patterns. *Estuaries* 10:227-245.
- Hollander, M., and D.A. Wolf. 1973. Non-

- parametric statistical methods. John Wiley and Sons. New York. 503 pp.
- Horn, M.H., and R.N. Gibson. 1988. Intertidal fishes. *Sci. Am.* 256:64-70.
- Jordan, R.A., and C.E. Sutton. 1985. Oligohaline benthic invertebrate communities at two Chesapeake Bay power plants. *Estuaries* 7:192-212.
- Jumars, P.A., and K. Fauchald. 1977. Between-community contrasts in successful polychaete feeding strategies. Pages 1-20 in B.C. Coull, ed. *Ecology of Marine Benthos*. Univ. of South Carolina Press, Columbia, S.C. 467 pp.
- Kaplan, E.H., J.R. Welker, and M.G. Kraus. 1974. Some effects of dredging on populations of macrobenthic organisms. *Fish. Bull.* 72:445-480.
- Kneib, R.T. 1988. Testing for indirect effects of predation in an intertidal soft-bottom community. *Ecology* 69:1795-1805.
- Levinton, J.S., and S. Stewart. 1982. Marine succession: The effect of two deposit-feeding gastropod species on the population growth of *Paranais litoralis* Muller 1784 (Oligochaeta). *J. Exp. Mar. Biol. Ecol.* 59:231-241.
- Lorda, E., and S.B. Salla. 1986. A statistical technique for analysis of environmental data containing periodic variance components. *Ecol. Model.* 32:59-69.
- Maurer, D., R.T. Keck, J.C. Tinsman, W.A. Leathem, C. Wethe, C. Lord and T.M. Church. 1986. Vertical migration and mortality in marine benthos in dredged material: a synthesis. *Int. Rev. ges. Hydrobiol.* 71:49-63.
- Moeller, P., L. Pihl, and R. Rosenberg. 1985. Benthic faunal energy flow and biological interaction in some shallow marine soft bottom habitats. *Mar. Ecol. Prog. Ser.* 27:109-121.
- Nichols, F.H. 1985. Abundance fluctuations among benthic invertebrates in two Pacific estuaries. *Estuaries* 8:136-144.
- NUSCO. (Northeast Utilities Service Company). 1985. Benthic Infauna. Pages 1-39 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1984.*
- NUSCO. 1986. Benthic Infauna. Pages 1-52 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1985.*
- NUSCO. 1987. Benthic Infauna. Pages 1-51 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of studies prior to Unit 3 operation. Annual Report 1986.*
- NUSCO. 1988a. Benthic Infauna. Pages 59-117 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.*
- NUSCO. 1988b. Hydrothermal Studies. Pages 323-354 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies 1986-1987.*
- NUSCO. 1989. Benthic Infauna. Pages 38-98 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1988.*
- NUSCO. 1992. Benthic Infauna. Pages 187-222 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1991.*
- NUSCO. 1993. Benthic Infauna. Pages 115-150 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual Report 1992.*
- Rees, H.L. and A. Eleftheriou. 1989. North Sea benthos: A review of field investigations into the biological effects of man's activities. *J. Cons. int. Explor. Mer.* 45:284-305.
- Regnault, M., R. E. Juchet-Rodoni, G. Boucher, and P. Lasserre. 1988. Effects of macrofauna excretion and turbulence on inorganic nitrogenous exchanges at the water-sediment interface. *Cah. Biol. Mar.* 29:427-444.
- Rhoads, D.C., and D.K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *J. Mar. Res.* 28:150-178.
- Richards, S.W. 1963. The demersal fish population of Long Island Sound. *Bull. Bingham Oceanogr. Coll.* 8:1-101.
- Sen, P.K. 1968. Estimates of regression coefficients based on the Kendall's tau. *J. Am. Stat. Assoc.* 63:1379-1389.

- Swartz, R.C., W.A. DeBen, F.A. Cole, and L.C. Bentsen. 1980. Recovery of the macrobenthos at a dredge site in Yaquina Bay, Oregon. Pages 391-408 in R.A. Baker, ed. *Contaminants and Sediments*, Vol. 2. Ann Arbor Science Publisher, Inc., Ann Arbor, Mich.
- Thrush, S.F., R.D. Pridmore, and J.E. Hewitt. 1994. Impacts on soft-sediment macrofauna: the effects of spatial variation on temporal trends. *Ecol. Appl.* 4:31-41.
- Turk, T.R., and M.J. Risk. 1981. Effects of sedimentation on infaunal invertebrate populations of Cobequid Bay, Bay of Fundy. *Can. J. Fish. Aquat. Sci.* 38:642-648.
- Warwick, R.M. 1986. A new method for detecting pollution effects on marine macrobenthic communities. *Mar. Biol.* 92:557-562.
- Warwick, R.M. 1988. Effects on community structure of a pollutant gradient-introduction. *Mar. Ecol. Prog. Ser.* 46:149.
- Warwick, R.M., T.H. Pearson and Ruswahyuni. 1987. Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance biomass method. *Mar. Biol.* 95:193-200.
- Warwick, R.M., H.M. Platt, K.R. Clark, J. Agard and J. Gobin. 1990. Analysis of macrobenthic and meiobenthic community structure in relation to pollution and disturbance in Hamilton Harbour, Bermuda. *J. Exp. Mar. Biol. Ecol.* 138:119-142.
- Watling, L. 1975. Analysis of structural variations in a shallow estuarine deposit-feeding community. *J. Exp. Mar. Biol. Ecol.* 19:275-313.
- Watzin, M. C. 1986. Larval settlement into marine soft-sediment systems: Interactions with the meiofauna. *J. Exp. Mar. Biol. Ecol.* 98:65-113.
- Woodin, S.A. 1982. Browsing: important in marine sedimentary environments? Spionid polychaete examples. *J. Exp. Mar. Biol. Ecol.* 60:35-45.
- Young, M.W., and D.K. Young. 1982. Marine macrobenthos as indicators of environmental stress. Pages 527-539 in G.F. Mayer, ed. *Ecological Stress and the New York Bight: Science and Management. Proceedings of the symposium; 1979 June 10-15; New York, New York. Estuarine Research Federation, Columbia, S.C.* 715 pp.

## Marine Woodborer Study

Introduction . . . . .	105
Materials and Methods . . . . .	105
Results and Discussion . . . . .	106
Conclusions . . . . .	108
References Cited . . . . .	108





## Marine Woodborer Study

### Introduction

Woodborers fill an important niche in marine ecosystems by degrading wood that is carried downriver into coastal waters. Unfortunately, they also destroy wooden structures (e.g., boats, piling, lobster pots) that have been intentionally placed in coastal areas. This study considers the wood-boring effects in Long Island Sound (LIS) of a single genus of marine woodborer, *Teredo*, which is represented by two species (*T. navalis* and *T. bartschi*) in the vicinity of Millstone Nuclear Power Station (MNPS). *Teredo navalis* is the only shipworm native to LIS; *T. bartschi* is a semitropical shipworm common from Texas to South Carolina (Turner 1966), but capable of establishing isolated populations near thermal discharges in more northern climates. In the 1970s, *T. bartschi* caused considerable economic damage to marinas within the influence of the warm discharge waters of Oyster Creek Nuclear Generating Station, NJ (Turner 1973). In July of 1975, *T. bartschi* was first documented in the discharge waters of MNPS (Battelle 1976) and has since persisted in the quarry (NUSCO 1993).

*Teredo bartschi* was targeted for a principal monitoring effort because it was not common to Long Island Sound waters and it appeared to be able to maintain a resident population in the warm discharge water of MNPS. Marine Woodborer Studies were redesigned in 1993 to more intensively monitor the abundance and distribution of *T. bartschi* in the immediate vicinity of MNPS. Increased number of sites and a modified sampling design were selected as a result of data that suggested *T. bartschi* set most successfully on surface panels, in contrast to the native shipworm, *T. navalis*, which appears to prefer panels near the bottom (NUSCO 1993).

The population of *Teredo bartschi* in the MNPS discharge waters was postulated to have the potential of adapting to the cold water temperatures common to the winter months of LIS, and thus, widening its distribution in the northeast (Hoagland 1981, 1983). The present study could detect such an event by monitoring trends in the abundance of this shipworm at all

our sites. This report covers the first year of collections and data analysis from a redesigned *T. bartschi* study.

### Materials and Methods

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

At EB, HR, JC and WP, two sets of three panels were attached 1 m below floats (double lobster pot buoys) at each site and anchored by a weighted (80 lbs.) wire lobster pot with no entry funnels (Fig. 2). At EF, the buoyed panels were attached to a pulley-and-line system to facilitate retrieval, and at QC, panels were attached to a buoy and stiff-arm system to ensure that the panels remained submerged at the proper depth in the eddy currents. Placement of panels in the water column (near the surface versus near the bottom) was shown to affect shipworm recruitment beyond the quarry cuts (NUSCO 1991, 1992, 1993). Surface locations had ended to reduce *Teredo navalis* settlement and enhance *T. bartschi* settlement during the 1990-1992 Discharge Study.



Fig. 1. Location of woodborer sampling sites in the vicinity of Millstone Nuclear Power Station (EF = Effluent - undiluted discharge waters, QC = Quarry Cuts - 1 m beyond fish barriers, EB = 100 m beyond QC, HR = High Rock - between rock outcroppings about 500 m beyond QC, JC = 500 m beyond QC, WP = White Point - 1700 m beyond QC).

Each panel (clear pine 25.4 x 8.9 x 1.9 cm) was secured in a separate section of PVC pipe (35 cm in length x 10.2 mm diameter). The panel arrays were deployed in May 1993 and collected in November 1993 (redundant sampling was used to minimize data loss). Three panels (replicates 1 and 2 from one array and replicate 4 from the second array, if present) at each site were processed by scraping fouling organisms and debris from all surfaces, X-raying (250 kV, 5 mA, for 45 s) each panel, and removing all or at least 70 shipworms per panel. Radiographs were used to locate shipworms within the panels and to estimate total shipworm abundance; shipworms were identified after removal from each panel to determine the percentage composition of *T. bartschi* and *T. navalis*. Shipworms smaller than 5 mm in tube length were classified only as juvenile teredinids and, although included as a component of shipworm abundance, were not included in the 70 shipworm subsample used for identifications.

## Results and Discussion

In 1993, *Teredo bartschi* was collected at QC, EB, and HR (Table 1). The abundance of *T. bartschi* at these three sites was low compared to that reported at EF during previous exposure panel studies (NUSCO 1993). However, 1993 represents the first occurrence of this species at a 500 m distance from the quarry cuts (HR). Except for three individuals collected in surface panels at 300 m in 1990 (Discharge Study; NUSCO 1991),

14 individuals collected on bottom panels at 100 m in 1985 (NUSCO 1986) and a few hundred individuals on surface panels at 100 m in 1990 (NUSCO 1991), from 1985 to 1992, *T. bartschi* had not been consistently collected in panels beyond the quarry cuts, either at bottom or surface locations (NUSCO 1993). The HR site was selected specifically for its reduced exposure to waves and currents, with the expectation that discharge water might be retained in this area for longer periods of time and favor the setting of *T. bartschi*. Doochin and Walton-Smith (1951) and NUSCO (1993) reported that strong currents inhibited shipworm settlement.

*Teredo bartschi* was absent from EF in 1993. A complete failure in recruitment of *T. bartschi* during a May to November exposure period had only occurred once before in 1981 (a 2-unit operational year). The latest absence of *T. bartschi* at EF, and the low abundance of this shipworm at QC, may be related to an atypical set of physical conditions of the MNPS discharge waters during 1993. Unit 3 was off-line from August to November in 1993, representing the longest period of 2-unit operation since Unit 3 start-up in 1986. This prolonged summer shut-down affected both water flow (without Unit 3 cooling water flow, the discharge volume was approximately halved) and water temperature (design  $\Delta T$  for Units 1 and 2 are 13.9 and 12.7 °C, respectively, compared to 9.5 °C for Unit 3) in the quarry. In 1993, the average daily discharge temperature in August and September exceeded 33 °C for more than 14 days. Adult *T. bartschi* tolerate temperatures of 6-35 °C, but their pediveliger larvae only remain alive between 5 and 32 °C (Hoagland 1983).

Water temperature is not the only factor affecting *Teredo* larval behavior and recruitment; settlement has been reported to be mediated by phototaxis (Isham et al. 1951), geotaxis (Isham et al. 1951; Nair 1962) and chemotaxis (Hillman et al. 1987). As mentioned previously, *T. bartschi* tends to set on panels near the surface (cf. Edmonson 1942; NUSCO 1992), and *T. navalis* on the bottom (cf. Schellema and Truitt 1956). *Teredo bartschi* has a short planktonic life and larvae can settle immediately after release (Isham and Tierney 1953), while *T. navalis* larvae may delay settlement for weeks (Grave 1928; Turner and Johnson 1971). Both species are affected by strong water currents (Doochin and Walton-Smith 1951; NUSCO 1993).

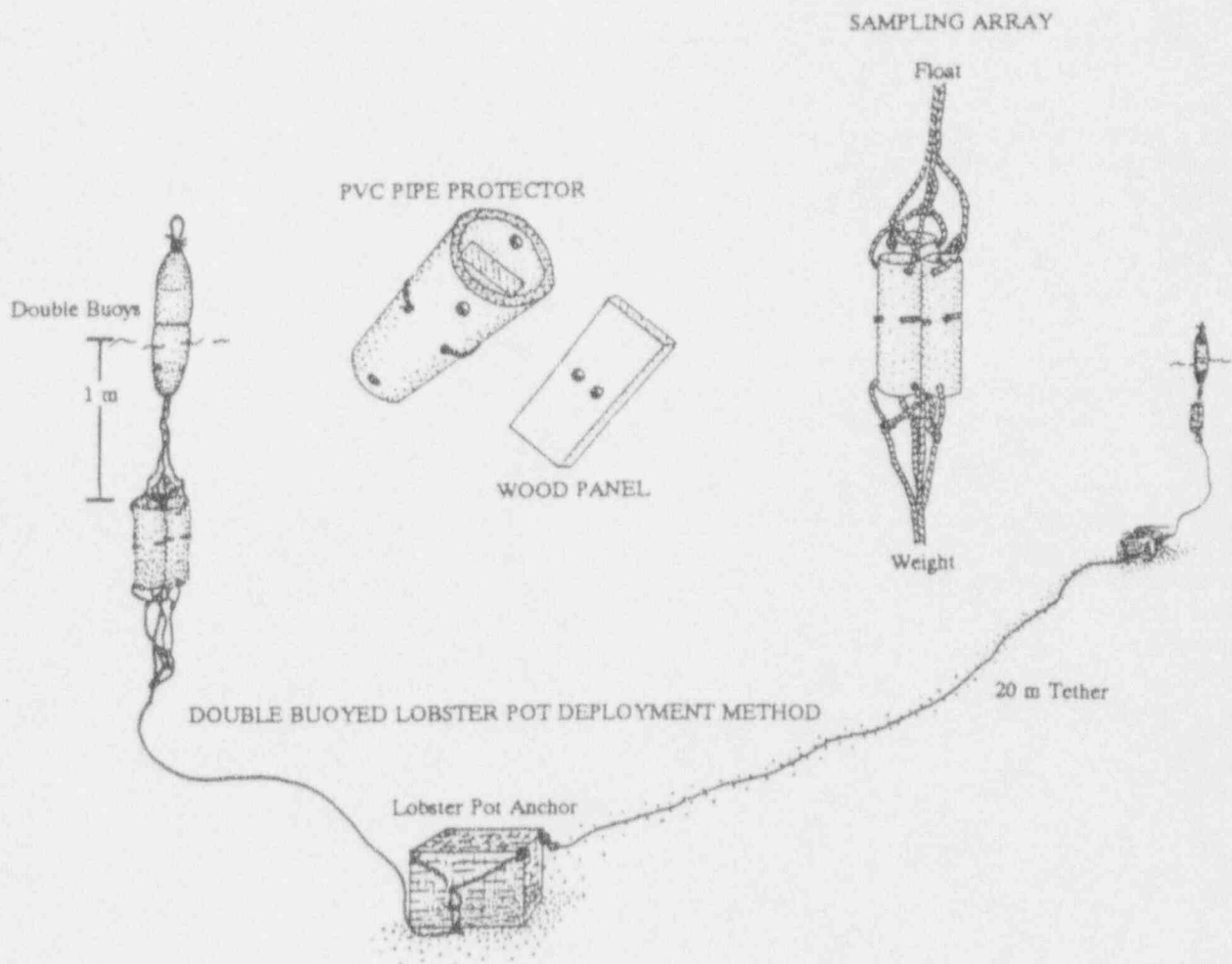


Fig. 2. The double buoyed lobster pot deployment method used to place wood panels 1 m below the surface at EF, HR, JC, and WP in the Woodborer Study at MNPS. Panels at QC are placed in a sampling arrays and held in place by stiff-arm assemblies and panels at EB are placed in sampling arrays and attached 1 m below the Effluent Benthic Buoy



Table 1. Average density (Ave.) and standard error of mean for shipworms, and associated wood-loss, using surface panels in the vicinity of MNPS from May to November 1993.

SPECIES	EF		QC		EB		HR		JC		WP	
	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE	Ave.	SE
<i>Teredo bartschi</i>	0.0	0.0	15.7	8.1	0.7	0.7	1.0	0.6	0.0	0.0	0.0	0.0
<i>Teredo navalis</i>	34.7	2.9	26.7	1.9	68.3	6.7	38.3	3.4	14.7	2.3	64.0	13.7
Teredinid juveniles	0.0	0.0	0.3	0.3	3.0	3.0	2.0	0.0	2.0	0.58	5.0	1.5
Percentage Wood-loss	28.3	1.6	27.3	8.2	55.0	5.0	45.0	8.7	6.3	0.9	45.0	2.9

It is unclear to what extent these environmental cues were altered in the quarry as a result of the atypical operating conditions in 1993. It is also unclear what effect the retubing of Unit 1 condensers (and concomitant decrease in the use of wood-chips) will have on the population of *T. bartschi* in the quarry.

In contrast, the continued absence of *Teredo bartschi* panels at WP indicates that this immigrant species has not adapted to ambient water conditions. Although the abundance of untreated oak piling at White's Dock provides an attractive food resource for this shipworm, the cold winter conditions appear too harsh for this species to establish a resident population beyond the influences of the warm discharge waters. The lack of *T. bartschi* in our panels at WP suggests they have not colonized this area, verifying that the EF population has not yet adapted to survive at ambient conditions in the Millstone area.

### Conclusions

*Teredo bartschi* remains in MNPS discharge waters. This shipworm was collected at HR, which is the first time *T. bartschi* has occurred in panels 500 m from the quarry cuts. Reduced currents around the rock outcroppings at HR may trap discharge water and increase the probability of collecting this warm water immigrant. The absence of *T. bartschi* at EF in 1993 is probably related to unusual conditions resulting from Unit 3 being off-line from August to November, during the peak recruitment period for this species. The distribution of this warm water immigrant remains closely associated with the discharge waters of MNPS and *T. bartschi* appears not to have adapted to ambient conditions at WP, 1700 m from the quarry cuts.

### References Cited

- Battelle (Columbus Lab., W.F. Clapp Lab.). 1976. Exposure Panels. Pages A1-A20 in A monitoring program on the ecology of the marine environment of the Millstone Point, Connecticut area. As submitted to Northeast Utilities Service Company. Ann. Rpt., No. 14673, 1975.
- Doochin, H., and F.G. Walton-Smith. 1951. Marine boring and fouling in relation to velocity of water currents. Bull. Mar. Sci. Gulf Carib. 1:196-208.
- Grave, B.H. 1928. Natural history of shipworm, *Teredo navalis*, at Woods Hole Massachusetts. Biol. Bull. Woods Hole. 55:260-282.
- Hillman, R.E., C. Werme, and M.J. Kennish. 1987. Setting of larval shipworms *Teredo bartschi* Clapp stimulated by malic acid and woodborer metabolites. Technical report to Environmental Controls Department, GPU Nuclear, Forked River, N.J. 25 pp.
- Hoagland, K.E. 1981. Life history characteristics and physiological tolerances of *Teredo bartschi*, a shipworm introduced into two temperate zone nuclear power plant effluents. In 3rd Int. Waste Heat Meetings Proc. 14 pp.
- Hoagland, K.E. 1983. Ecological studies of wood-boring bivalves and fouling organisms in the vicinity of the Oyster Creek Nuclear Generating Station. Final Report, September 1976 - December 1982. U.S. Nuclear Regulatory Comm. Wash. D.C. NRC FIN B8138. 173 pp.
- Isham, L.B., F.G. Walton-Smith and V. Springer. 1951. Marine borer attack in relation to conditions of illumination. Bull. Mar. Sci. Gulf Carib. 1:46-63.



- Nair, N.B. 1962. Ecology of marine fouling and wood-boring organisms of Western Norway. *Sarsia* 8:1-88.
- NUSCO (Northeast Utilities Service Company). 1982. Exposure Panels. Pages 1-32 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1981.
- NUSCO. 1987. Exposure Panel Program. Pages 1-45 in *Monitoring the marine environment of Long Island Sound at the Millstone Nuclear Power Station, Waterford Connecticut, Summary of studies prior to Unit 3 operation*. Ann. Rpt. 1986.
- NUSCO. 1988. Hydrothermal Studies. Pages 323-354 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1987.
- NUSCO. 1990. Marine Woodborer Studies. Pages 221-242 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1989.
- NUSCO. 1991. Marine Woodborer Studies. Pages 201-220 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1990.
- NUSCO. 1992. Marine Woodborer Studies. Pages 295-315 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1991.
- NUSCO. 1993. Marine Woodborer Studies. Pages 95-112 in *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Ann. Rpt. 1992.
- Scheltema, R.S., and R.V. Truitt. 1956. The shipworm *Teredo navalis* in Maryland coastal waters. *Ecology* 37:841-843.
- Turner, R.D. 1966. A survey and illustrated catalogue of the Teredinidae (Mollusca: Bivalvia). The Museum of Comparative Zoology, Harvard U., Cambridge, MA. 265 pp.
- Turner, R.D. and A.C. Johnson. 1971. Biology of marine wood-boring molluscs. Pages 259-301 in E.B.G. Jones and S.K. Eltringham (eds). *Marine borers, fungi, and fouling organisms of wood*. Proc. OECD Workshop, March 1968. Organization for Economic Cooperation and Development, Paris, France.
- Turner, R.D. 1973. In the path of a warm saline effluent. *Am. Malacol. Union Bull.* 39:36-41.



## Fish Ecology Studies

Introduction .....	113
Materials and Methods .....	113
Ichthyoplankton program .....	113
Entrainment mortality studies .....	115
Trawl program .....	115
Seine program .....	115
Data analyses .....	115
Abundance estimates .....	115
Entrainment estimates .....	117
Results and Discussion .....	117
American sand lance .....	120
Anchovies .....	121
Silversides .....	122
Grubby .....	125
Tautog .....	126
Cunner .....	127
Entrainment mortality studies .....	128
Conclusions .....	130
References Cited .....	131
Appendix .....	132



# Fish Ecology Studies

## Introduction

Fish populations that inhabited the area around Millstone Nuclear Power Station (MNPS) were monitored since 1976 to determine the effects of station operations. Abundance and distribution of marine fish populations can be affected by water temperature, salinity, density-dependent growth and mortality, fecundity of individual species, age structure of the population and life history strategies. Moreover, fish species in the MNPS area could be adversely affected by losses due to impingement of juvenile and adult fish on the intake screens, entrainment of fish eggs and larvae or by changes in the thermal regime or physical habitat. Long-term monitoring is needed to assess the impacts of power generation on fish assemblages because of the inherent variability of fish populations.

Changes in the abundance and distribution of fish species which would affect community structure have been defined as power plant related effects. Ichthyoplankton, trawl and seine monitoring programs are used to ascertain these effects on local fish populations. Additionally, impingement on the intake screens removes juvenile and adult fish from populations, although this impact has been mitigated with the addition of fish returns at Units 1 and 3. Fish eggs and larvae suffer mortality when they are entrained through the condenser cooling water system. Effects of increased mortality rates on the size of fish populations depend on size or age at which mortality occurs, age structure, and the effectiveness of compensatory mechanisms. Spatial distribution of local fish populations may vary with thermal addition or habitat alterations. Data from the trawl and seine monitoring programs, on entrained larvae from June 1976 through May 1993 and on entrained eggs from June 1979 through May 1993, are summarized in this report along with a brief synopsis of the corresponding sampling programs.

## Materials and Methods

Species-specific analyses are based on the actual periods of occurrence instead of being constrained to June 1 as the common starting point. A reporting year comprises the 12-month period 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 the June 1 endpoint. When the season of occurrence of a species crossed a calendar year, the year was reported as "1992-93", but when the species occurred only within a calendar year, the year was reported as "1993". Materials and methods of the 1992-93 reporting period are essentially the same as those used in earlier years.

## Ichthyoplankton program

Entrained ichthyoplankton (fish eggs and larvae) samples were collected both day and night three times each week from June through September 1992 and once per week from October through December 1992. Sampling was reduced during 1993 after an evaluation of the ichthyoplankton sampling program demonstrated that sample reductions did not substantially change  $\Delta$ -mean indices and therefore, did not reduce the sensitivity of the program to detect changes related to MNPS operations. During January 1993, only one day sample per week was collected (a reduction of one sample per week). In February 1993, samples were collected both day and night one time per week (no reduction) and in March through June 1993, three times per week (a reduction of two samples per week). Generally, samples were collected each week at only one of the three plant discharges (station EN, Fig. 1), with the location usually alternating weekly between Unit 1 or 2. To collect samples from the discharge water a 1.0 x 3.6-m conical plankton net with 333- $\mu$ m mesh was deployed with the aid of a gantry system. Four General Oceanic flowmeters (Model 2030) were mounted in the mouth of this net and positioned to account for horizontal and vertical flow variations. Sample volume (about 400 m<sup>3</sup> in 1992 and 200 m<sup>3</sup> in 1993, except during periods of high plankton or detritus concentrations) was determined by an average of the volume estimates from the four flowmeters.

Fish larvae were also collected in mid-Niantic Bay at station NB during 1992 (Fig. 1). Two day and two night samples were taken weekly from April through August, and one day and one night sample were taken biweekly from September through December.



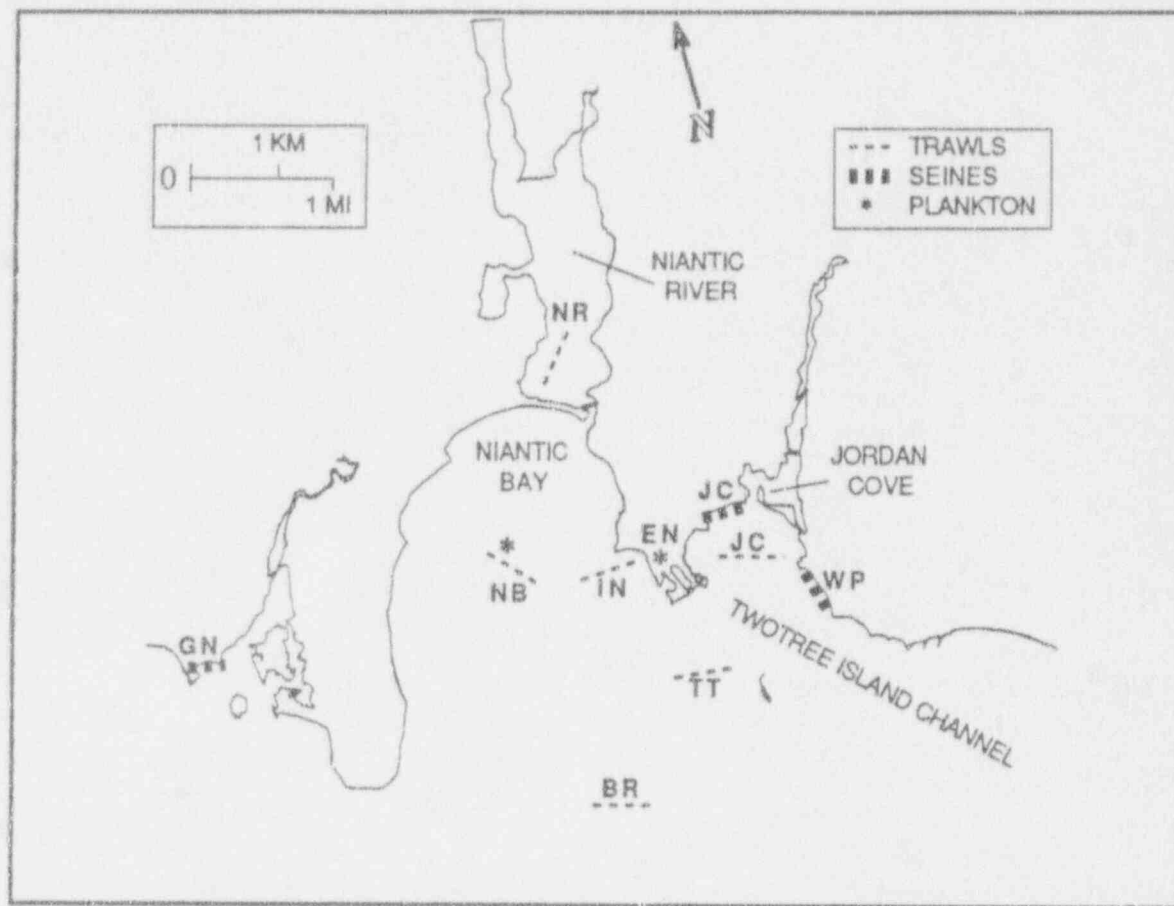


Fig. 1. Location of trawl, seine and ichthyoplankton sampling stations.

Samples were collected with a bongo sampler (two 0.61-m x 3.3-m conical plankton nets). The bongo sampler was towed at approximately 2 knots for 6 to 15 minutes using a stepwise oblique pattern with equal sampling time at surface, mid-depth, and near-bottom. Sample volumes were measured using one General Oceanics flowmeter in each net and approximately 300 m<sup>3</sup> of seawater were filtered for each sample (except during periods of high plankton or detritus concentrations). Net mesh size was 333  $\mu$ m, except during a period from mid-February through March, when 202- $\mu$ m mesh nets were used to minimize the extrusion of yolk-sac winter flounder larvae. Sampling at NB was discontinued in 1993.

Plankton samples were split using a NOAA-Bourne splitter (Botelho and Donnelly 1978); ichthyoplankton were removed from the samples with the aid

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

## Entrainment mortality studies

Special studies to estimate wrasse egg entrainment mortality were conducted in 1993. To decrease the volume of water filtered through the plankton net and reduce the pressure on the eggs in the net, a 20.5 cm Pitot tube sampler was deployed using the same gantry system used in regular entrainment collections at the discharge (Fig. 2). One General Oceanic (Model 2030) flowmeter was positioned inside the Pitot tube to determine the sample volume. A tight mesh nylon sleeve was secured around the net to reduce the shear on the net which could damage the eggs already caught in the net. These studies were conducted June 21-22, June 22 and June 28-29. Samples were also collected in front of the MNPS intakes in Niantic Bay (IN) and were used as a control to determine natural and sampling mortality. At IN, the sampler was placed in a styrofoam float and towed at a similar velocity as encountered at station EN.

Samples were brought to the laboratory, placed in running seawater and viewed immediately with a dissecting microscope. Approximately 100 wrasse eggs were removed from samples collected at each station. Ten eggs were put into each of ten 30-mL Nalgene® beakers with three 1.5 cm diameter openings covered with 110- $\mu$ m mesh netting and placed in a flow-through sea water bath. Samples were viewed at least once per day to determine how many eggs had hatched; eggs that had not hatched at the end of a 72-hour period were considered to be dead. After hatching, larvae were identified to species.

To determine daily fluctuations in abundance, three 24-hour studies of wrasse egg densities were conducted June 8-9, June 15-16 and July 19-20. A total of 39 samples were collected at EN (using sampling methods described above for the 1-m 333- $\mu$ m mesh net) every 2 hours during a 24-hour period. Approximately 100 eggs were removed from each sample and identified as cunner, tautog or other species. Densities were reported as number of eggs per 500m<sup>3</sup>.

## Trawl program

Triplicate bottom tows were made using a 9.1-m otter trawl with a 0.6-cm codend liner. Demersal fishes were collected biweekly throughout the year at six stations: Niantic River (NR), Jordan Cove (JC), Twotree Island Channel (TT), Bartlett Reef (BR), Intake (IN) and Niantic Bay (NB) (Fig. 1). Rarely,

only two replicate tows were taken at a station because of damage to gear or severe weather. A standard tow was 0.69 km and this distance was measured using onboard radar. When the trawl net became loaded with macroalgae and detritus, tow distances were shortened and standardized to 0.69 km. Catch was expressed as the number of fish per standardized tow (CPUE). Up to 50 randomly chosen individuals of certain selected species per station were measured (total length) to the nearest millimeter.

## Seine program

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

## Data analyses

### *Abundance estimates*

The occurrence, distribution, and abundance of selected potentially impacted fish, as well as their observed spatial and temporal fluctuations, were analyzed to assess possible plant-related impacts. Indices of fish abundance were selected on the basis of underlying distributional assumptions; failure of the data to conform to these assumptions may reduce the precision of the estimates or, worse, provide biased results. Thus, the  $\Delta$ -mean was used as an index of abundance and the  $\alpha$ -parameter from the Gompertz function was used to estimate entrainment of fish eggs and larvae (NUSCO 1990).

The  $\Delta$ -mean was selected to describe annual abundance trends because it is the best estimator of the mean of a population that approximately follows the lognormal distribution and contains many zeros (Hennemuth et al. 1980; Pennington 1983, 1986). The calculation of this index and its variance estimate

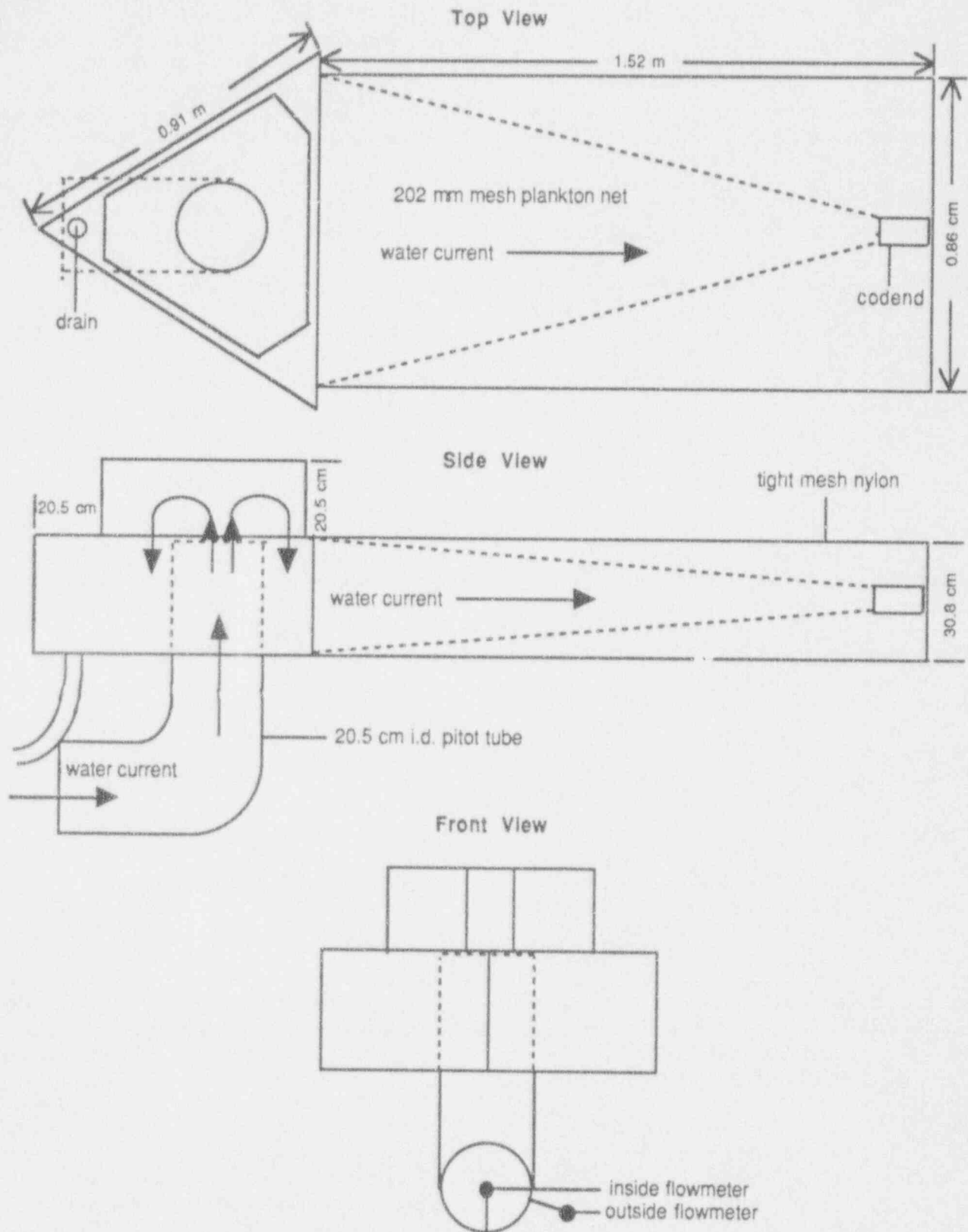


Fig. 2. Schematic of Pitot tube sampler use for the wrasse egg mortality study sampling at MNPS discharges.

was described in detail in NUSCO (1988). The  $\Delta$ -mean was used as an index of abundance for juvenile and adult fish collected in trawl and seine programs, for larvae and fish eggs collected at EN, and for larvae collected at NB. The  $\Delta$ -mean indices of ichthyoplankton species were weighted by the largest number of samples collected in a week to standardize data across weeks and years. For any species that occurred seasonally, the data used for calculating the  $\Delta$ -mean were restricted to its period of occurrence to reduce the number of zero values in the distribution tails. Two-unit operational period  $\Delta$ -means were calculated from the beginning of two-unit operation (1976) to the beginning of three-unit operation (1986).

### Entrainment estimates

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

$$C_t = \alpha \exp(-\exp[-k(t-p)]) \quad (1)$$

where

$C_t$  = cumulative density at time  $t$

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

$\alpha$  = total or asymptotic cumulative density

$p$  = inflection point in days since first occurrence date

$k$  = shape parameter

The origin of the time scale was set to the date when the eggs or larvae generally first appeared in the waters off MNPS. Least-squares estimates, standard errors, and asymptotic 95% confidence intervals of these parameters were obtained by fitting the above equation to the cumulative abundance data using

nonlinear regression methods (SAS Institute Inc. 1985). The cumulative data were obtained as the running sums of the weekly geometric means of the abundance data per unit volume.

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

$$d_t = \alpha' k \exp(-\exp[-k(t-p)] - k(t-p)) \quad (2)$$

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

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

### Results and Discussion

One hundred and twenty species of fish eggs, larvae, juvenile and adult fish were collected in the MNPS monitoring programs from June 1976 through May 1993 (Appendix 1). Winter flounder (*Pleuronectes americanus*), anchovies (*Anchoa mitchilli* and *A. hepsetus*), silversides (*Menidia menidia* and *M. beryllina*), grubby (*Myoxocephalus aeneus*), American sand lance (*Ammodytes americanus*), skates (*Raja erinacea*, *R. ocellata* and *R. eglanteria*), scup (*Stenotomus chrysops*), windowpane flounder (*Scophthalmus aquosus*), tautog and cunner were the most common fish collected.

Sixty-one taxa were represented in ichthyoplankton samples and of these 7 egg and 20 larval taxa were found in sufficient numbers to calculate  $\Delta$ -mean densities (Tables 1, 2, and 3). All 1992-93 egg densities at EN were within historic ranges. However, densities of winter flounder larvae at EN were the lowest ever recorded since sampling began in 1976-77, and densities of anchovy larvae at EN and NB were the second lowest recorded with only densities in 1987-88 lower. Except for winter flounder larvae, other egg and larval entrainment estimates were within historic ranges.

Over the past 17 years, 105 fish taxa were caught in trawls and 50 were captured in seines (Appendix I). Six taxa in trawl samples and one taxon in seine samples accounted for over 80% of the catch in each



TABLE 1. The  $\Delta$ -mean<sup>a</sup> density (no./500 m<sup>3</sup>) of the most abundant fish eggs collected at EN for each report year from June 1979 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

Taxon	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
<i>T. adspersus</i>	5,870	8,223	5,171	5,501	7,068	5,719	7,484	2,969	5,002	5,395	6,904	4,998	6,954	4,416
<i>T. onitis</i>	1,364	2,842	2,647	2,244	2,114	2,157	3,237	2,756	3,011	2,269	2,887	2,060	1,878	1,449
<i>Anchoa</i> spp.	1,447	1,245	1,080	765	2,257	4,880	145	910	89	38	54	127	476	107
<i>S. aequosus</i>	50	63	65	34	19	71	365	181	520	178	94	76	64	72
<i>Prionotus</i> spp.	61	206	398	385	425	156	367	82	63	89	64	15	14	52
<i>S. chrysops</i>	21	1	133	113	98	194	25	69	31	4	36	19	23	34
<i>E. cimbrius</i>	22	11	31	34	8	10	14	55	58	65	65	24	117	86

<sup>a</sup> Data seasonally restricted to May 22-July 23 for *T. adspersus*, May 23-August 25 for *T. onitis*, June 15-August 5 for *Anchoa* spp., May-August for *S. aequosus*, July-August for *Prionotus* spp., May-July for *S. chrysops*, and April-August for *E. cimbrius*.

TABLE 2. The  $\Delta$ -mean<sup>a</sup> density (no./500 m<sup>3</sup>) of the most abundant fish larvae collected at EN for each report year from June 1976 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

Taxon	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93
<i>Anchoa</i> spp.	1,152	931	483	2,168	2,430	5,768	816	1,421	302	1,102	1,244	126	359	619	1,122	799	178
<i>P. americanus</i>	106	143	114	285	129	233	297	210	180	87	109	116	203	106	99	388	21
<i>A. americanus</i>	94	318	119	111	136	21	27	18	9	3	13	41	31	24	7	18	28
<i>M. aeneus</i>	41	38	36	38	107	72	68	50	68	34	29	95	63	30	24	58	34
<i>P. gunnellus</i>	13	13	16	13	58	27	13	14	14	22	4	26	9	6	3	15	8
<i>B. tyrannus</i>	5	4	4	0	3	1	11	23	2	41	3	2	6	72	18	97	41
<i>T. adspersus</i>	29	58	1	13	58	78	31	49	4	12	4	5	9	14	68	209	8
<i>T. onitis</i>	37	36	1	11	46	83	44	33	3	15	3	7	17	15	33	99	13
<i>Liparis</i> spp.	27	30	10	16	22	5	13	8	36	1	4	42	18	12	3	23	14
<i>U. subbifurcata</i>	5	9	14	14	16	17	6	4	60	7	9	23	41	51	34	28	2
<i>S. fuscus</i>	4	7	4	9	8	13	7	9	9	5	4	6	7	5	3	5	3
<i>E. cimbrius</i>	2	8	6	8	6	1	6	13	5	8	8	12	45	31	37	98	5
<i>S. aequosus</i>	10	11	1	5	5	5	2	13	3	1	4	3	5	3	4	12	2
<i>P. triacanthus</i>	14	3	1	2	11	17	9	9	1	2	3	0	9	5	29	10	2
Gobiidae	6	3	1	0	1	0	0	1	4	3	3	2	4	8	7	12	2
<i>Prionotus</i> spp.	2	2	0	1	3	18	0	4	1	6	0	0	1	1	7	3	0
<i>M. octodecemspinosus</i>	1	1	1	1	1	3	4	4	1	0	0	0	1	1	0	2	0
<i>S. chrysops</i>	5	8	0	4	6	8	1	0	0	0	0	0	0	2	0	0	0
<i>C. harengus</i>	1	1	1	0	6	1	0	1	0	2	1	14	1	1	2	9	7
<i>C. regalis</i>	1	3	0	6	0	7	6	3	0	2	1	0	1	1	0	0	0

<sup>a</sup> Data seasonally restricted to July-September for *Anchoa* spp., March-June for *P. americanus*, December-May for *A. americanus*, February-May for *M. aeneus*, January-May for *P. gunnellus*, July-December for *B. tyrannus*, to June-August for *T. adspersus*, June-August for *T. onitis*, March-May for *Liparis* spp., April-September for *S. fuscus*, April-June for *U. subbifurcata*, April-July for *E. cimbrius*, May-October for *S. aequosus*, June-September for *P. triacanthus*, June-November for Gobiidae, June-September for *Prionotus* spp., January-May for *M. octodecemspinosus*, June-August for *S. chrysops*, February-May for *C. harengus* and June-August for *C. regalis*.

TABLE 3. The  $\Delta$ -mean<sup>a</sup> density (no./500 m<sup>3</sup>) of the most abundant fish larvae collected at NB for each report year from June 1979 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

Taxon	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
<i>Anchoa</i> spp.	3,801	5,716	7,873	2,103	2,860	327	1,117	1,224	79	303	462	1,214	711	181
<i>P. americanus</i>	222	129	158	317	187	85	80	146	147	111	109	128	364	.. <sup>b</sup>
<i>A. americanus</i>	113	238	20	54	58	8	3	13	83	35	16	5	9	.. <sup>b</sup>
<i>T. adspersus</i>	92	143	96	151	198	37	30	8	10	28	39	141	282	19
<i>M. aeneus</i>	31	54	28	39	24	18	26	28	119	44	25	23	72	.. <sup>b</sup>
<i>T. onitis</i>	49	89	91	110	119	28	44	12	13	29	28	60	123	25
<i>E. cimbrius</i>	19	10	3	24	19	23	9	11	19	54	42	90	131	17
<i>B. tyrannus</i>	0	2	2	36	9	24	12	0	1	1	4	1	4	5
<i>S. aequosus</i>	24	12	7	17	30	14	5	8	7	10	7	12	13	5
<i>P. gunnellus</i>	5	24	14	7	8	2	8	1	7	5	5	1	9	.. <sup>b</sup>
<i>P. triacanthus</i>	9	18	37	31	35	2	18	6	1	11	8	54	15	2
<i>Liparis</i> spp.	12	23	4	15	4	14	1	5	59	12	18	3	29	.. <sup>b</sup>
<i>S. fuscus</i>	5	7	9	5	12	7	2	2	4	4	2	1	5	2
<i>U. subbifurcata</i>	10	9	20	8	5	7	5	6	11	12	32	9	7	.. <sup>b</sup>
<i>Prionotus</i> spp.	5	4	34	13	8	3	7	5	0	2	2	16	6	2
<i>M. octodecemspinosus</i>	2	3	4	6	4	1	0	1	1	1	1	0	3	.. <sup>b</sup>
<i>P. oblongus</i>	1	3	8	6	9	1	5	1	1	3	3	4	4	2
<i>C. regalis</i>	6	0	11	13	0	0	1	1	0	0	1	0	1	0
<i>S. chrysops</i>	9	16	6	1	0	0	0	0	0	0	1	2	7	1

<sup>a</sup> Data seasonally restricted to July-September for *Anchoa* spp., to March-June for *P. americanus*, December-May for *A. americanus*, June-August for *T. adspersus*, June-August for *T. onitis*, February-May for *M. aeneus*, July-December for *B. tyrannus*, April-July for *E. cimbrius*, May-October for *S. aequosus*, January-May for *P. gunnellus*, June-September for *P. triacanthus*, March-May for *Liparis* spp., April-September for *S. fuscus*, June-September for *Prionotus* spp., April-June for *U. subbifurcata*, January-May for *M. octodecemspinosus*, June-August for *P. oblongus*, June-August for *C. regalis*, June-August for *S. chrysops*.

<sup>b</sup> Sampling at NB was discontinued after December 1992 thus no data was obtained for these species during the report year 1992-1993.



TABLE 4. The  $\Delta$ -mean<sup>a</sup> catch (no./0.69 km) of the most abundant fish collected by trawl for each report year from June 1976 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

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

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

monitoring program (Appendices II, III, IV and V). Winter flounder continued to dominate the trawl catches, accounting for 37% of the catch from 1976-77 through 1992-93, and 29% of the catch during 1992-93 (Appendix II). During two-unit operations winter flounder accounted for 41% of the trawl catch but only 31% of the catch during the three-unit operational period. Scup also comprised 29% of the catch during 1992-93 and 21% from 1976-77 through 1992-93. During the two-unit operational period scup only accounted for 14% of the trawl catch but increased to 28% of the catch during the three-unit operational period. In 1992-93, winter flounder and skates accounted for 5% and 6%, respectively which was just below the 17 year (1976-77 through 1992-93) average of 7% each. During two-unit operations windowpane flounder accounted for 8% of the trawl catch while skates accounted for 5.8%. These percentage were reversed during three-unit operations with windowpane flounder accounted for 5.8% of the catch and skates accounting for 8%. More silversides were collected by trawl in 1992-93 than in the previous 17 years (1976-77 through 1992-93). Historically (1976-77 through 1992-93) silversides have accounted for 4% of the trawl catch but this year (1992-93) they accounted for 11% of the catch. The percent contribution of silversides to the trawl catch increased from 4% during two-unit operations to 5% during three-unit operations. Only 0.2% of the catch in 1992-93 were anchovies, although historically (1976-77 through 1992-93) they have accounted for 6% of the catch. The  $\Delta$ -mean for each of these taxa were all within the range of previous trawl catches (Table 4). Silversides dominated seine catches accounting for 81% of the catch from 1976-77 through 1992-93 and 93% of the catch in 1992-93 (Appendix IV). During the two-unit operational period silversides accounted for 85% of the catch and 78% of the catch during the three-unit operational period.

Cunner, tautog and anchovies accounted for over 90% of all eggs entrained from June 1976 through

May 1993 (Table 5). Anchovies, winter flounder, American sand lance and grubby accounted for 80% of the entrained larvae during the same period (Table 5). Except for winter flounder, entrainment estimates of eggs and larvae were within historic ranges (Tables 6 and 7). Although the volume of cooling water pumped through MNPS during the 1993 season of occurrence for winter flounder was the highest recorded, the entrainment estimate was the second lowest recorded. Only the 1977 entrainment estimate was lower but the cooling water volume during the 1977 larval season was only 40% of the 1993 volume. The winter flounder entrainment estimate was low due to the low density of larvae.

TABLE 5. Taxonomic composition of ichthyoplankton collected at EN (as a percentage of the total) from June 1976 through May 1993 for larvae and April 1979 through September 1992 for eggs.

Taxon	Larvae	Eggs
<i>Anchoa</i> spp.	55.2	6.6
<i>Pleuronectes americanus</i>	12.4	0.0
<i>Ammodytes</i> spp.	7.9	0.0
<i>Myoxocephalus aeneus</i>	4.5	0.0
<i>Brevoortia tyrannus</i>	3.7	0.0
<i>Tautoglabrus adspersus</i>	2.4	55.6
<i>Tautoga onitis</i>	2.1	30.7
<i>Photichthys gunnellus</i>	2.0	0.0
<i>Enchelyopus cimbrius</i>	1.7	0.9
<i>Ulvaxia subbifurcata</i>	1.3	0.0
<i>Liparis</i> spp.	1.2	0.0
<i>Syngnathus fuscus</i>	1.0	0.0
<i>Scophthalmus aquosus</i>	0.8	2.4
<i>Peprilus triacanthus</i>	0.8	0.1
Gobiidae	0.5	0.0
<i>Clupea harengus</i>	0.4	0.0
<i>Prionotus</i> spp.	0.3	1.4
<i>Myoxocephalus octodecemspinosus</i>	0.2	0.0
<i>Stenotomus chrysops</i>	0.2	0.7
<i>Paralichthys oblongus</i>	0.2	0.0
<i>Menidia</i> spp.	0.1	0.0
<i>Cynoscion regalis</i>	0.1	0.1
<i>Scomber scombrus</i>	0.1	0.1
<i>Anguilla rostrata</i>	0.1	0.0
<i>Alosa</i> spp.	0.1	0.5
Clupeidae	0.1	0.0
<i>Etropus microstomus</i>	0.1	0.0
<i>Sphaeroides maculatus</i>	0.1	0.0
<i>Urophycis</i> spp.	0.0	0.4
Labridae	0.0	0.2
<i>Alosa pseudoharengus</i>	0.0	0.1

TABLE 6. Estimated number of cunner, tautog, and anchovy eggs entrained each year at MNPS and the volume of cooling water on which the entrainment estimates were based (two-unit operational period:1976-85; three-unit operational period: 1986-92).

Year	Cunner		Tautog		Anchovy	
	No. entrained (x10 <sup>6</sup> )	Volume (m <sup>3</sup> ) <sup>a</sup> (x 10 <sup>6</sup> )	No. entrained (x10 <sup>6</sup> )	Volume (m <sup>3</sup> ) <sup>a</sup> (x 10 <sup>6</sup> )	No. entrained (x10 <sup>6</sup> )	Volume (m <sup>3</sup> ) <sup>a</sup> (x 10 <sup>6</sup> )
1979	1,534	728	705	728	215	711
1980	2,302	806	1,273	806	91	795
1981	1,736	816	1,735	816	172	799
1982	2,726	853	1,486	853	234	843
1983	2,631	798	1,180	798	618	786
1984	2,031	827	1,369	827	652	812
1985	2,802	831	1,784	831	20	825
1986	2,932	1,870	3,907	1,870	517	1,846
1987	4,533	1,784	3,740	1,784	37	1,752
1988	4,386	1,953	2,813	1,953	16	1,920
1989	3,885	1,643	3,094	1,643	5	1,611
1990	3,651	1,823	2,185	1,823	28	1,795
1991	4,758	1,265	1,589	1,265	147	1,247
1992	2,736	1,022	1,271	1,022	17	1,022

\* Volume was determined from the condenser cooling water flow at MNPS during the season of occurrence for each taxa.

TABLE 7. Estimated number of anchovy, winter flounder, American sand lance and grubby larvae entrained each year at MNPS and the volume of cooling water on which the entrainment estimates were based (two-unit operational period:1976-85; three-unit operational period:1986-93).

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

\* Volume was determined from the condenser cooling water flow at MNPS during the season of occurrence for each taxa.

<sup>b</sup> Not calculated because larvae occur after end of report period (May 1993).

Additional data analyses were completed for six taxa that were identified as having a potential for impact based on their prevalence in entrainment samples or susceptibility to thermal impacts. Winter flounder is discussed in a separate section (see Winter Flounder Studies), and is not included among these fishes. The  $\Delta$ -mean densities (no./500m<sup>3</sup>) for eggs and larvae,  $\Delta$ -mean catches (no./0.69 km) for trawl catches and (no./30 m) for seine monitoring programs were calculated for abundant life stages of American sand lance, anchovies, silversides, grubbies, cunner and tautog.

### American sand lance

American sand lance are caught primarily as larvae in the winter and spring and are seldom caught by

trawl or seine. Larval catches are variable and annual entrainment estimates ranged from 5 to 90 million larvae. Their abundance has varied over two orders of magnitude during the past 17 years (Table 8). Larval densities during two- and three- unit operational periods were compared to assess annual variation (Fig. 3). Because sand lance larvae were so abundant from 1976-77 through 1980-81, larval densities in the three-unit period have been lower than during the two-unit period. The 1992-93 densities of sand lance larvae were within the range of densities from recent years. Given the large abundance changes of this species along the Atlantic coast (Monteleone et al. 1987), effects of MNPS operation on sand lance are difficult to ascertain.

TABLE 8. The  $\Delta$ -mean\* density (no./500 m<sup>3</sup>) and 95% confidence interval for American sand lance larvae collected at EN during each report year from June 1976 through May 1993 (two-unit operational period: 1976-77 through 1984-85; three-unit operational period: 1985-86 through 1992-93).

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

\* Data seasonally restricted to December - May.

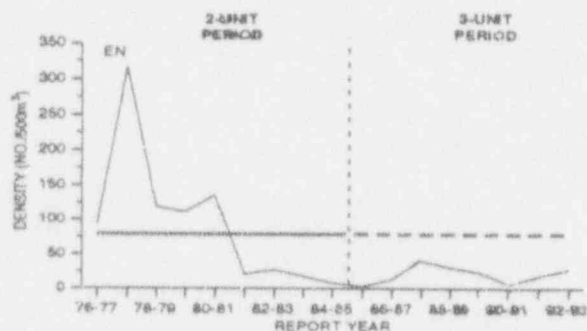


Fig. 3. The annual (—)  $\Delta$ -mean densities (no./500 m<sup>3</sup>) of American sand lance larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

### Anchovies

The bay anchovy is the most common fish along the Atlantic coast and the most abundant ichthyoplankton member within its range (McHugh 1977; Leak and Houde 1987). Larval anchovies dominate the plankton collections and anchovy eggs were ranked third in abundance. The entrainment estimates of eggs and larvae for 1992 were within the historic range (Table 6 and 7). The 1992 anchovy egg density was within the range of densities since 1984 (Table 9; Fig. 4). All egg densities during the three-unit operational period were below the two-unit average because after 1984 (before the three-unit period) densities declined by one order of magnitude and remained low. Larval densities in 1992 were the second lowest, with only the 1987 densities lower (Table 9). All three-

TABLE 9. The  $\Delta$ -mean\* density (no./500 m<sup>3</sup>) and 95% confidence interval for anchovy eggs and larvae collected at EN and larvae collected at NB during each report year from June 1976 through May 1993 (two-unit operational period: 1976-85; three-unit operational period: 1986-92).

Year	EGGS	LARVAE
	EN	EN
1976		1,152 $\pm$ 419
1977		931 $\pm$ 408
1978		483 $\pm$ 206
1979	1,447 $\pm$ 336	2,168 $\pm$ 908
1980	1,245 $\pm$ 597	2,430 $\pm$ 1,249
1981	1,080 $\pm$ 264	5,768 $\pm$ 3,326
1982	765 $\pm$ 228	816 $\pm$ 240
1983	2,257 $\pm$ 1,076	1,421 $\pm$ 530
1984	4,880 $\pm$ 3,680	302 $\pm$ 165
1985	145 $\pm$ 75	1,102 $\pm$ 453
1986	910 $\pm$ 547	1,244 $\pm$ 893
1987	89 $\pm$ 46	126 $\pm$ 69
1988	37 $\pm$ 33	359 $\pm$ 216
1989	54 $\pm$ 47	619 $\pm$ 416
1990	127 $\pm$ 117	1,122 $\pm$ 853
1991	476 $\pm$ 526	799 $\pm$ 801
1992	107 $\pm$ 112	178 $\pm$ 85

\* Data seasonally restricted to June 15 - August 5 for eggs and July - September for larvae.

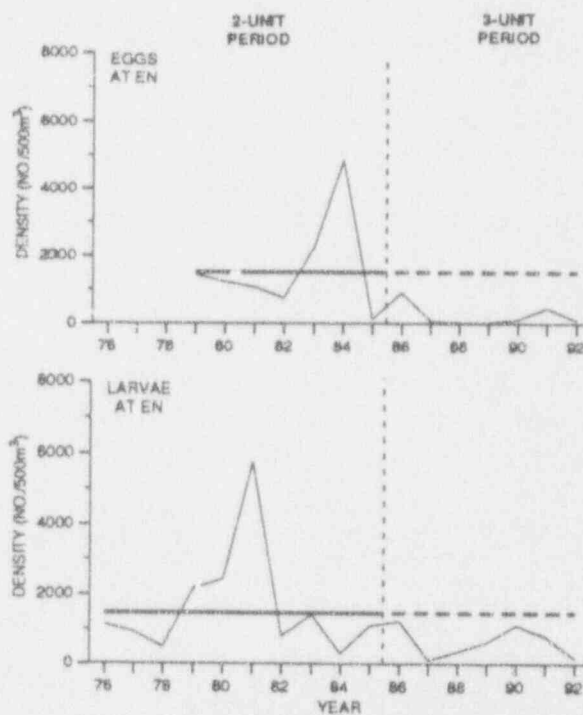


Fig. 4. The annual (—)  $\Delta$ -mean densities (no./500 m<sup>3</sup>) of anchovy eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

unit larval  $\Delta$ -mean densities were below the two-unit  $\Delta$ -mean average (Fig. 4). Juvenile anchovies resulting from the summer spawn are typically captured by trawl sampling from August through

October, predominantly in Niantic Bay. Even though anchovies rank fifth among fish caught by trawl,  $\Delta$ -mean CPUE could not be calculated because anchovies catches were highly variable. More than 70% of all the anchovies caught during the past 17 years were caught in only 2 years, 1985-86 and 1986-77, (Appendix II). Anchovies mature within a few months of hatching and live only 1 or 2 years; such short-lived species usually exhibit large oscillations in abundance.

### Silversides

Along the Connecticut coast, the Atlantic silverside and the inland silverside are the most common shore-zone species. The Atlantic silverside is the most dominant. Essentially, all the silversides caught by trawl were the Atlantic silverside; less than 0.1% caught in the past 17 years were the inland silverside. Historically, more than 80% of the silversides collected by seine were Atlantic silversides, although this proportion has varied from year to year. Both species are so abundant in the shore-zone that they can be analyzed separately. Trawl and seine catches are highly variable and annual catch indices ranged over two orders of magnitude, which is typical of short lived species such as these. The  $\Delta$ -mean trawl CPUE for Atlantic silversides was at a 17 year (1976-77 through 1992-93) high at the two Niantic Bay Stations (NB and IN) (Table 10). This was the first time in the three-unit operational period these two stations were above the two-unit average (Fig. 5). All 1992-93  $\Delta$ -mean trawl catches were above the

two-unit  $\Delta$ -mean average (Fig. 5). All three-unit annual  $\Delta$ -means were above two-unit average at NR. The  $\Delta$ -mean catches of Atlantic and inland silversides caught in seines were all within historic ranges (Fig. 6; Table 11) and all 1992  $\Delta$ -means were above the

TABLE 10. The  $\Delta$ -mean<sup>a</sup> catch (no./0.69 km) and 95% confidence interval for Atlantic silverside collected by trawl at selected stations during each report year from June 1976 through May 1993 (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1992-93).

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

<sup>a</sup> Data seasonally restricted to November-February at IN, NB and NR, and October - January at JC.

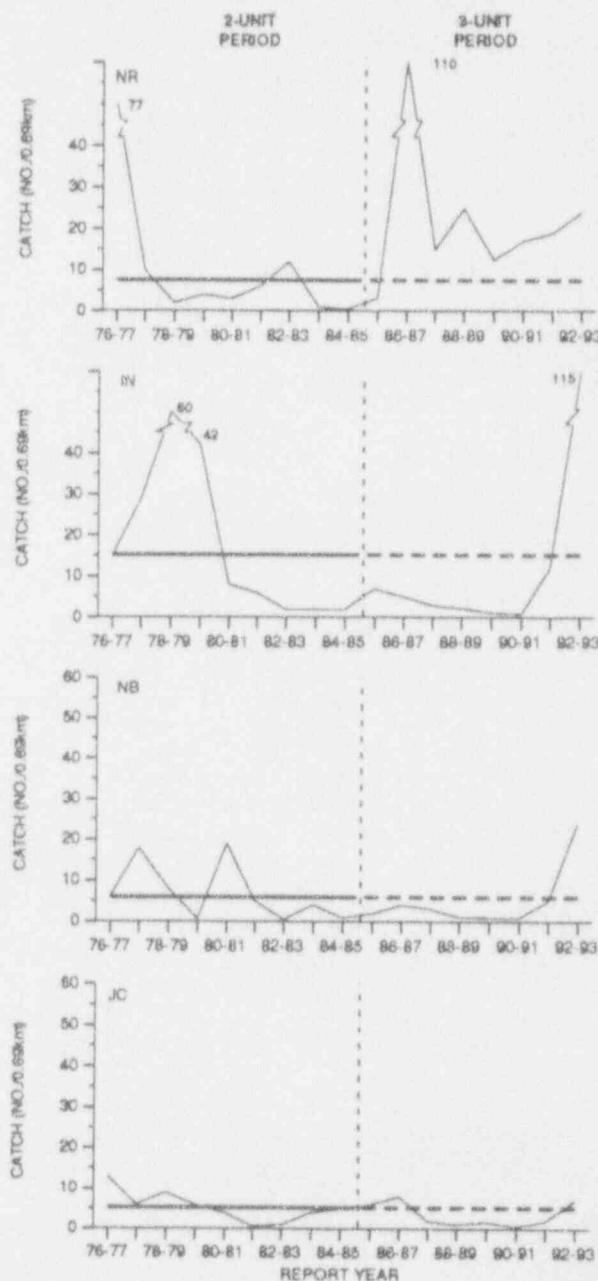


Fig. 5. The annual (—)  $\Delta$ -mean densities (no./0.69 km) of Atlantic silverside taken by trawl at NR, IN, NB and JC during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

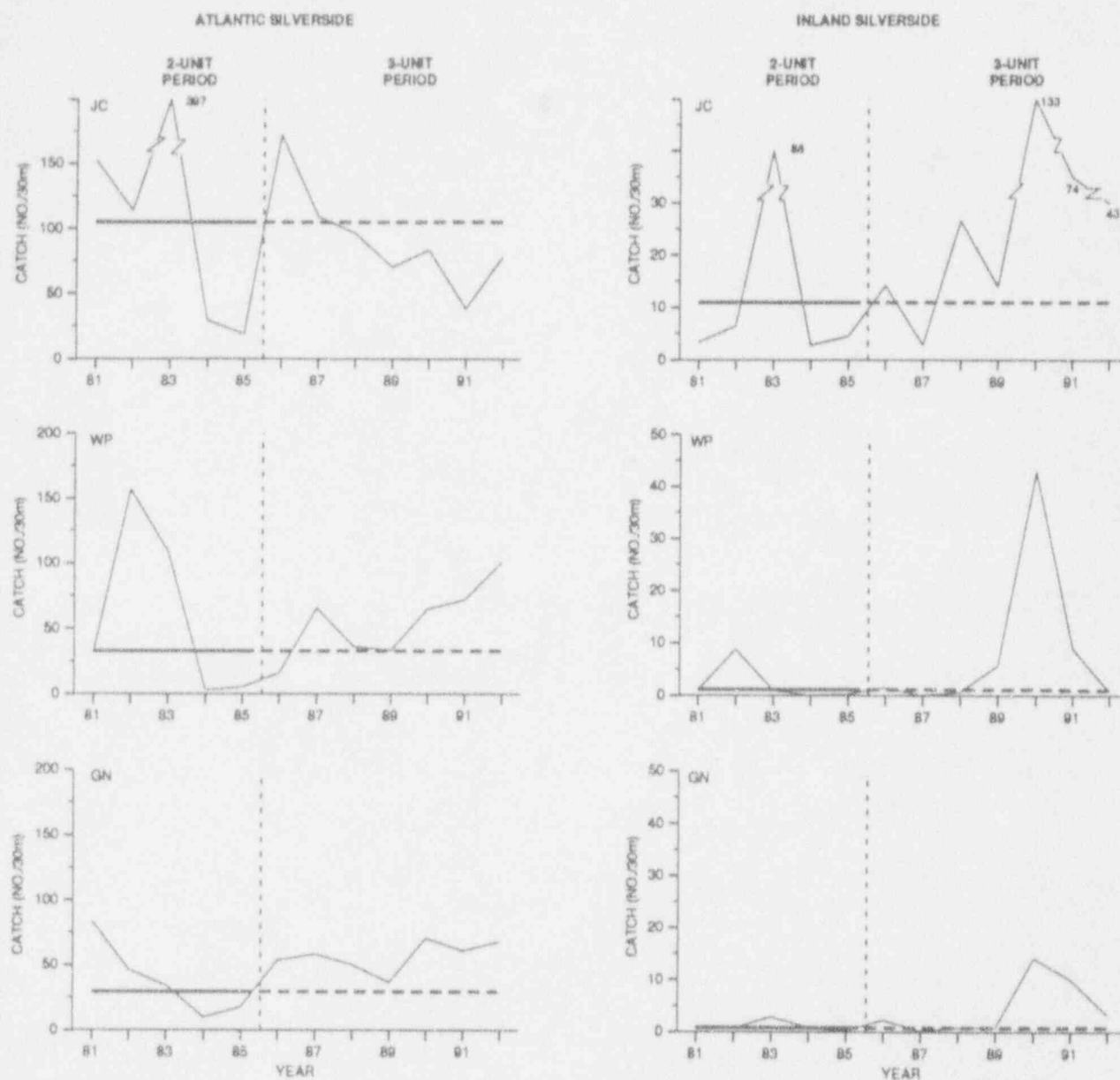


Fig. 6. The annual (—)  $\Delta$ -mean densities (no./30 m) for Atlantic silverside and inland silverside taken by seine at JC, WP and GN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (— — —) which is extended over the 3-unit operational period as a reference level.

two-unit average, except for Atlantic silversides at JC which was typical for all three-unit operational  $\Delta$ -means. Historically, Atlantic silversides were more abundant than inland silversides, however during 1990 and 1991 inland silversides were more abundant than Atlantic silversides at JC (Fig. 7). In 1992, Atlantic silversides again dominated the catch at JC but catches of both species were within historic values.

To determine if a change in length-frequency distributions occurred after Unit 3 became operational, the length frequencies (expressed as percentages) were examined for the periods before and after three-unit operation and for the 1992-93 study period. The length-frequency distribution for silversides collected by seines and trawls remained similar during these two operational periods (Fig. 8).



TABLE 11. The  $\Delta$ -mean\* catch (no./30 m) and 95% confidence interval for Atlantic silverside and inland silverside collected by seine during each report year from June 1981 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

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

Year	Inland silverside		
	JC	GN	WP
1981	3 $\pm$ 3	1 $\pm$ 1	1 $\pm$ 3
1982	6 $\pm$ 16	1 $\pm$ 2	9 $\pm$ 44
1983	88 $\pm$ 243	3 $\pm$ 5	1 $\pm$ 3
1984	3 $\pm$ 2	1 $\pm$ 1	0 $\pm$ 0
1985	4 $\pm$ 8	0 $\pm$ 0	0 $\pm$ 0
1986	14 $\pm$ 21	2 $\pm$ 2	1 $\pm$ 5
1987	3 $\pm$ 2	1 $\pm$ 1	0 $\pm$ 0
1988	27 $\pm$ 54	1 $\pm$ 1	1 $\pm$ 0.5
1989	14 $\pm$ 16	1 $\pm$ 2	6 $\pm$ 28
1990	133 $\pm$ 234	14 $\pm$ 24	43 $\pm$ 148
1991	74 $\pm$ 37	10 $\pm$ 9	9 $\pm$ 7
1992	43 $\pm$ 27	3 $\pm$ 1	1 $\pm$ 0.3

\* Data seasonally restricted to June-November at all stations.

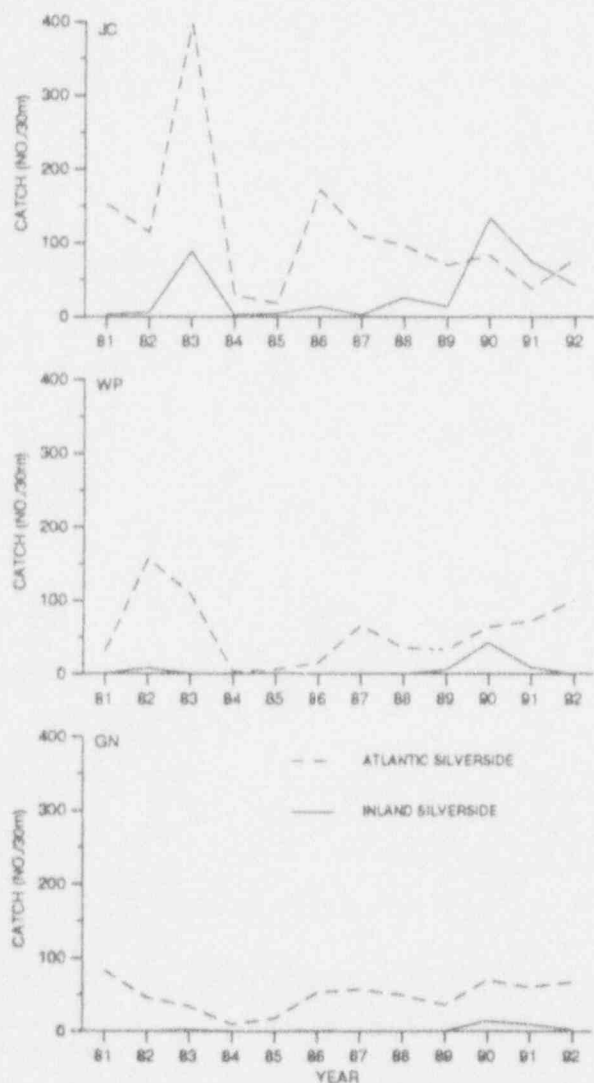


Fig. 7. The annual  $\Delta$ -mean catch (no./30 m) for the Atlantic silverside and inland silverside taken by seine at JC, WP and GN from 1981 through 1990.

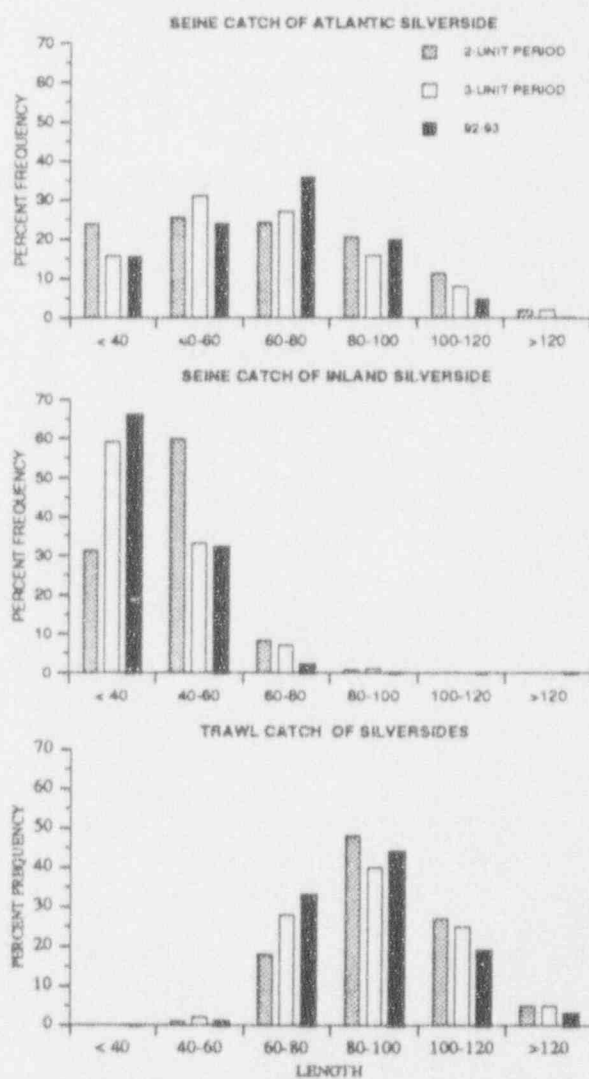


Fig. 8. Frequency distribution, by 20-mm length intervals, for the Atlantic silverside and inland silverside taken by seine and trawl during two-unit (1976-85) and three-unit (1986-92) operational periods and the 1992-93 report-year.

## Grubby

The grubby is the fourth most abundant larval taxon entrained, accounting for 4.5% of all larvae collected at EN from June 1976 through May 1993. Entrainment estimates ranged from 11 million in 1978 to 124 million in 1988. An estimated 54 million larvae were entrained in 1993, which was in the range of previous estimates. The  $\Delta$ -mean larval density was low in 1993, but was within the range of historic data (Table 12). Three-unit operational annual  $\Delta$ -mean larval densities fluctuated around the two-unit average (Fig. 9). The grubby was the seventh-most abundant taxon taken by trawls, accounting for more than 2% of the catch at all stations over the past 17 years. The 1992-93 catches were all above the two-unit  $\Delta$ -mean average (Table 13; Fig. 10), and all were within the range of historic values.

TABLE 12. The  $\Delta$ -mean\* density (no./500 m<sup>3</sup>) and 95% confidence interval for grubby larvae collected at EN during each report year from June 1976 through May 1993 (two-unit operational period: 1976-85; three-unit operational period: 1986-93).

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

\* Data seasonally restricted to February - May.

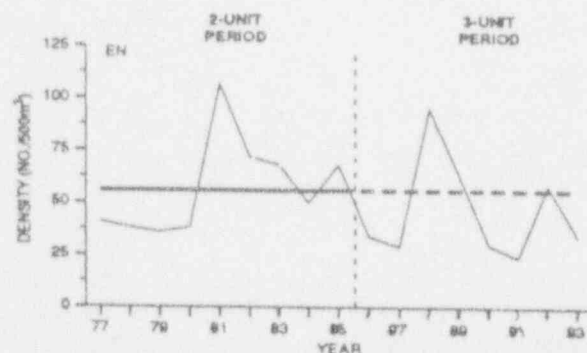


Fig. 9. The annual (—)  $\Delta$ -mean densities (no./500 m<sup>3</sup>) of grubby larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

TABLE 13. The  $\Delta$ -mean\* catch (no./0.69 km) and 95% confidence interval for grubby collected by trawl at selected stations during each report year from June 1976 through May 1993 (two-unit operational period: 1976-77 through 1985-86; three-unit operational period: 1986-87 through 1992-93).

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

\* Data seasonally restricted to December-June at IN, and year-round at JC and NR (June-May).

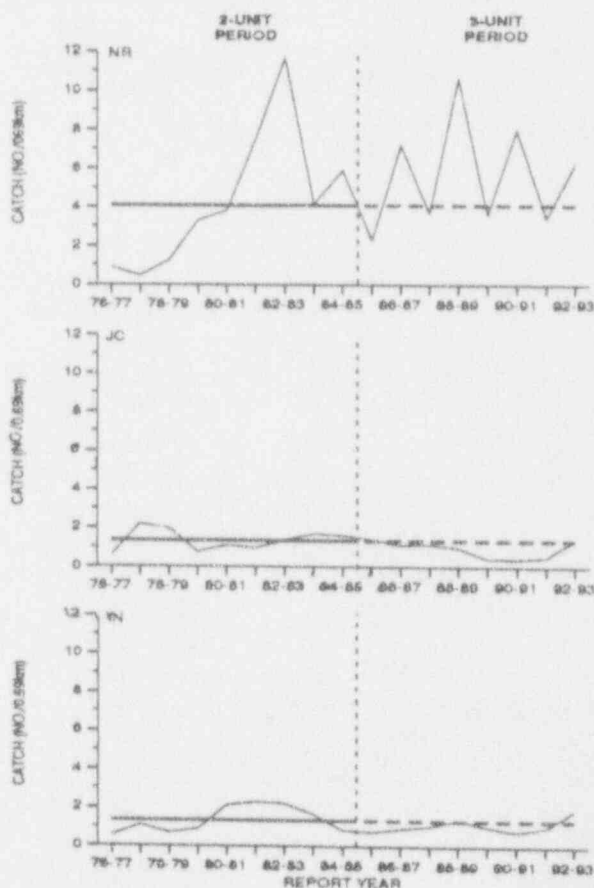


Fig. 10. The annual (—)  $\Delta$ -mean densities (no./0.69 km) of grubby taken by trawl at NR, JC and IN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

Similar to the larval abundance indices, the three-unit operational annual  $\Delta$ -mean indices of grubbies taken by trawl fluctuated around the two-unit average (Fig. 10). The normalized (each period equals 100%) trawl length frequency distributions of grubby were similar before and after three-unit operation (Fig. 11) although the 1992-93 distribution had higher frequencies for smaller fish.

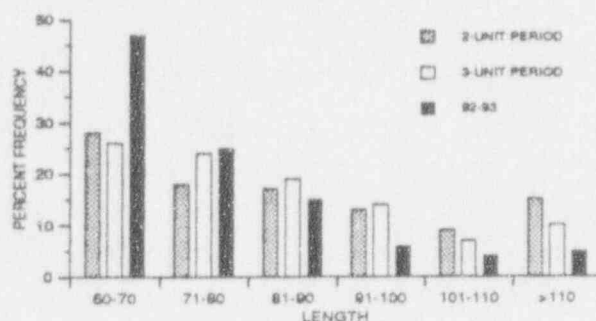


Fig. 11. Frequency distribution, by 10-mm length intervals, of grubby taken by trawl during two-unit (1976-85) and three-unit (1986-92) operational periods and the 1992-93 report-year.

### Tautog

The tautog is the second-most abundant egg taxon entrained and has accounted for more than 30% of the total eggs collected since 1979. Entrainment ranged from 705 million in 1979 to 4 billion in 1986 and was estimated at 1.3 billion in 1992 (Table 6). The 1992  $\Delta$ -mean egg density was the second lowest since sampling began (Table 14). During the four years immediately after three-unit operation began the annual  $\Delta$ -mean densities of tautog eggs were above the two-unit average, but during the past three years the annual  $\Delta$ -mean densities have been declining (Fig. 12). Tautog larvae accounted for 2.1% of all fish larvae caught at EN. The 1992 larval densities were within the range of historic catches (Table 14) but were lower than the two-unit average (Fig. 12). Annual  $\Delta$ -mean larval densities during the three-unit operational period had trends which were almost the opposite of the annual trends for egg densities, except for 1992. During the four years immediately after three-unit operation began, the annual  $\Delta$ -mean larval densities were below the two-unit average. The 1990 and 1991 densities were above the two-unit average, however, 1992 density was below the two-unit average (Fig. 12).

TABLE 14. The  $\Delta$ -mean\* density (no./500 m<sup>3</sup>) and 95% confidence interval for tautog eggs and larvae collected at EN during each report year from June 1976 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

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

\* Data seasonally restricted to May 23 - August 20 for eggs and June - August for larvae.

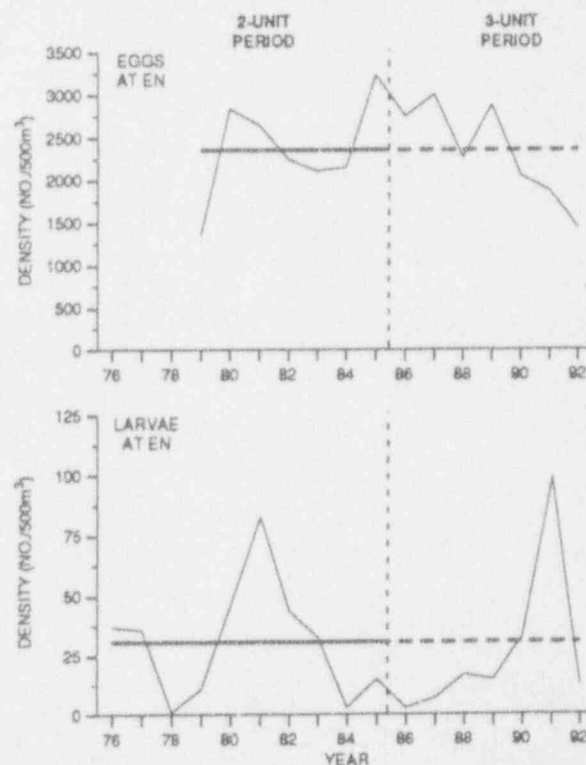


Fig. 12. The annual (—)  $\Delta$ -mean densities (no./500 m<sup>3</sup>) of tautog eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (— — —) which is extended over the 3-unit operational period as a reference level.

(NOTE: different vertical scale used for eggs and larvae.)

Tautog catches in trawls have always been relatively low. The 1992-93 catch was within the range of previous catches (Appendix II). Because tautog catches were low and the data contained many zeroes, annual  $\Delta$ -mean could not be calculated. Tautog collected both before and after three-unit operation were assigned an age based on their length according to recent age-length work in LIS reported by Simpson (1989). Young-of-the-year tautog have accounted for a high proportion of the fish caught since three-unit operation began (Fig. 13).

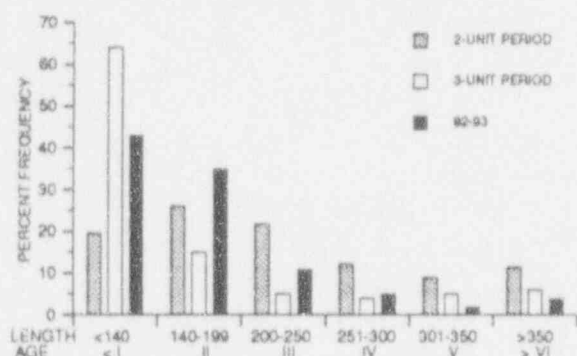


Fig. 13. Frequency distribution by length (mm) and age (determined from age-length key of Simpson 1989) for tautog taken by trawl during two-unit (1976-85) and three-unit (1986-92) operational periods and the 1992-93 report-year.

### Cunner

The  $\Delta$ -mean density of cunner eggs in 1992 was the second lowest in the 14 year data series; only the 1986 density was lower (Fig. 14). Because of the low densities, the 1992 entrainment estimate was the lowest since the three-unit operational period began. The  $\Delta$ -mean densities of larvae were also low but were within the range of previous values (Table 15). The 1992  $\Delta$ -mean densities of both eggs and larvae were below the two-unit operational  $\Delta$ -mean because of these low densities (Fig. 14). Most (5 of 7 years) of the annual three-unit operational  $\Delta$ -mean indices were below the two-unit average (Fig. 14).

The trawl catch of cunner has been declining since 1979 (NUSCO 1993). This trend continued at IN and JC and the 1992 catch at these two stations was below the two-unit operational  $\Delta$ -mean (Table 16; Fig. 15). The  $\Delta$ -mean catch at NB was above the two-unit operational  $\Delta$ -mean. This was the first time since three units began operating that any annual  $\Delta$ -means were above the two-unit operational  $\Delta$ -mean (Fig. 15). To determine an age-frequency distribution, ages were assigned based on an age-length key

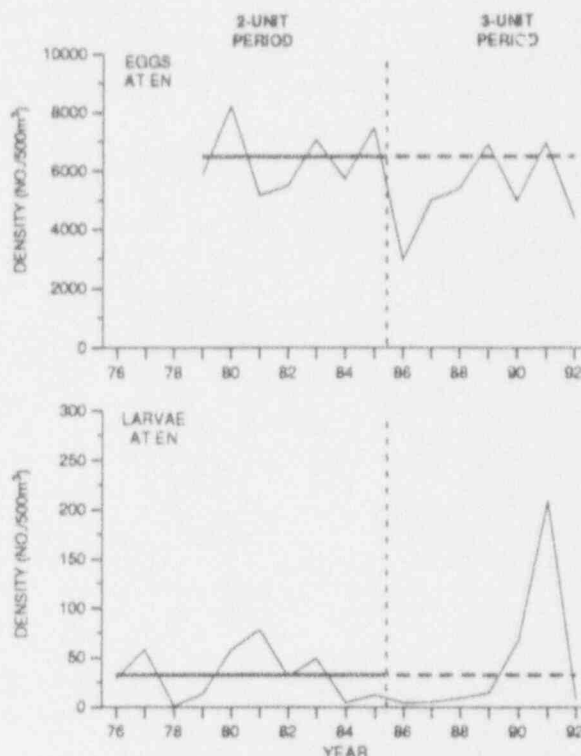


Fig. 14. The annual (—)  $\Delta$ -mean densities (no./500 m<sup>3</sup>) of cunner eggs and larvae at EN during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (— — —) which is extended over the 3-unit operational period as a reference level. (NOTE: different vertical scale used for eggs and larvae.)

TABLE 15. The  $\Delta$ -mean\* density (no./500 m<sup>3</sup>) and 95% confidence interval for cunner eggs and larvae collected at EN and larvae collected at NB during each report year from June 1976 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

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

\* Data seasonally restricted to May 22 - July 23 for eggs, and June - August for larvae.



TABLE 16. The  $\Delta$ -mean\* catch (no./0.69 km) and 95% confidence interval for cunner collected by trawl at selected stations during each report year from June 1976 through May 1993 (two-unit operational period:1976-85; three-unit operational period:1986-93).

Year	IN	JC	NB
1976	26.0 $\pm$ 19.0	4.0 $\pm$ 2.6	1.0 $\pm$ 0.7
1977	24.0 $\pm$ 23.0	3.0 $\pm$ 1.0	1.0 $\pm$ 0.6
1978	6.0 $\pm$ 3.7	3.0 $\pm$ 1.4	0.7 $\pm$ 0.3
1979	29.0 $\pm$ 23.0	9.0 $\pm$ 5.0	2.0 $\pm$ 1.0
1980	23.0 $\pm$ 16.0	6.0 $\pm$ 2.0	3.0 $\pm$ 1.2
1981	12.0 $\pm$ 10.0	5.0 $\pm$ 2.2	3.0 $\pm$ 0.9
1982	5.0 $\pm$ 3.0	4.0 $\pm$ 2.0	2.0 $\pm$ 0.9
1983	3.0 $\pm$ 1.3	4.0 $\pm$ 2.0	1.0 $\pm$ 0.6
1984	2.0 $\pm$ 1.0	2.0 $\pm$ 1.0	0.4 $\pm$ 0.2
1985	1.0 $\pm$ 0.6	1.0 $\pm$ 0.5	0.4 $\pm$ 0.7
1986	0.1 $\pm$ 0.2	0.5 $\pm$ 0.4	0.1 $\pm$ 0.1
1987	0.2 $\pm$ 0.2	0.4 $\pm$ 0.2	0.0 $\pm$ 0.0
1988	0.3 $\pm$ 0.1	3.0 $\pm$ 3.4	0.2 $\pm$ 0.1
1989	0.9 $\pm$ 0.4	0.8 $\pm$ 0.4	0.2 $\pm$ 0.2
1990	0.4 $\pm$ 0.1	0.9 $\pm$ 0.2	0.1 $\pm$ 0.1
1991	0.4 $\pm$ 0.1	2.3 $\pm$ 0.7	0.0 $\pm$ 0.0
1992	1.01 $\pm$ 0.7	1.4 $\pm$ 0.5	3.8 $\pm$ 1.0

\* Data seasonally restricted to May-August at IN, May-September at JC, and April-November at NB.

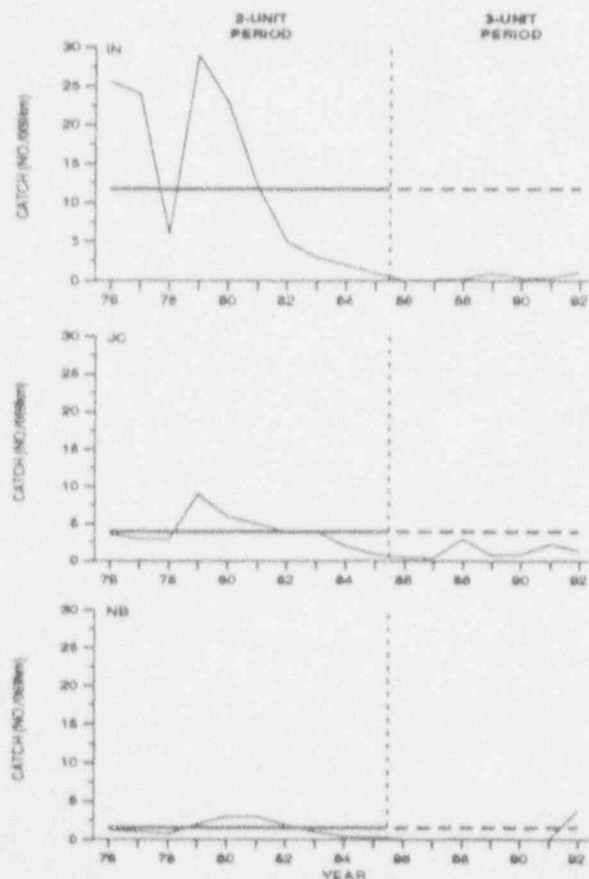


Fig. 15. The annual (—)  $\Delta$ -mean densities (no./0.69 km) of cunner taken by trawl at IN, JC and NB during two-unit and three-unit operational periods. The two-unit operational period (1976-1985)  $\Delta$ -mean density is represented by the flat line (---) which is extended over the 3-unit operational period as a reference level.

provided by Serchuk (1972). A normalized frequency distribution was calculated for both the two and three-unit periods and the 1992-93 report-year. The distributions for these three periods appeared quite different and over 50% of the cunner caught during three-unit operation were young-of-the-year (Fig. 16).

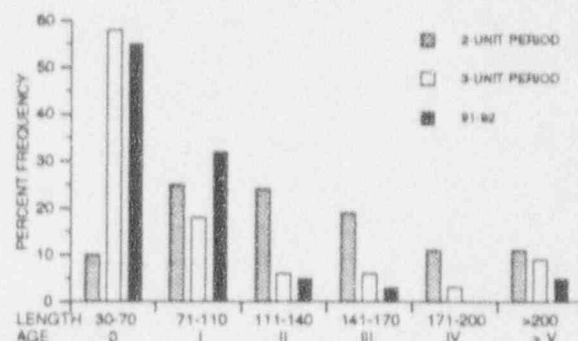


Fig. 16. Frequency distribution by length (mm) and age (determined from age-length key of Serchuk 1972) for cunner taken by trawl during two-unit (1976-85) and three-unit (1986-92) operational periods and the 1992-93 report-year.

## Entrainment mortality studies

Fish eggs entrained through the MNPS cooling-water system are at risk of suffering high mortality because they are exposed to elevated water temperatures, mechanical stresses and intermittent chlorination. Mortality caused by entrainment could effect local fish stocks because early life mortality rates influence adult abundance (Cushing and Harris 1973; Cushing 1974; DeAngelis et al. 1977). Because over 85% of the eggs entrained at MNPS were wrasse (tautog and cunner) eggs, studies were conducted in 1990, 1991 and 1993 to determine the entrainment mortality of wrasse eggs. Results of entrainment mortality studies conducted in 1990 and 1991 (NUSCO 1991, 1992) indicated that the wrasse egg natural hatching rates and entrainment survival were higher than previously believed (NUSCO 1992). However, in 1991 a comparison of wrasse abundance at EN and QC (located at the second Millstone quarry cut, see Fig. 2 in the Introduction section) indicated that about 56% fewer eggs were collected at QC. This information lead us to believe that wrasse eggs that suffered entrainment mortality died and settled out of the water column and were not available for capture at QC. This would result in a disproportionate number of eggs collected at QC hatching, thus producing an artificially low mortality rate. To



correct for this problem, samples were collected at station EN during 1993 using the Pitot tube sampler. Average wrasse egg hatching rates at EN decreased to 4% in 1993 compared to 41% for 1990 and 1991 (Fig. 17). Some of this decreased hatching may have been due to sampling, since the average hatching rate for eggs collected using the Pitot tube sampler at IN were only 59% while previously calculated hatching rates using the bongo sampler averaged 91%. Because of these discrepancies, entrainment mortality of wrasse eggs will be assumed to be 100% until more reliable estimates can be obtained.

Since there was such a large difference between the abundances of wrasse eggs collected during the day and at night, three 24-hour studies were conducted to investigate daily fluctuations in abundance. Samples collected every 2 hours for a 24-hour period showed very inconsistent results in the three 24-hour studies (Fig. 18). Based on the 24-hour geometric means, the average daily abundances were different but their 95% confidence intervals overlapped (Table 17). In general, abundances were lower prior to midnight and remained reasonably constant until late afternoon. A rapid increase in abundance occurred in the evening

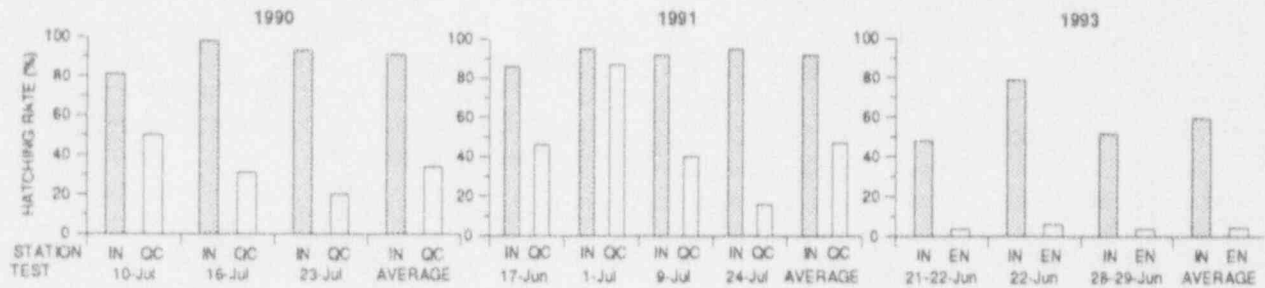


Fig. 17. Hatching rate of wrasse (tautog and cunner) eggs collected before (IN) and after (QC or EN) passing through MNPS during entrainment mortality studies conducted in 1990, 1991 and 1993.

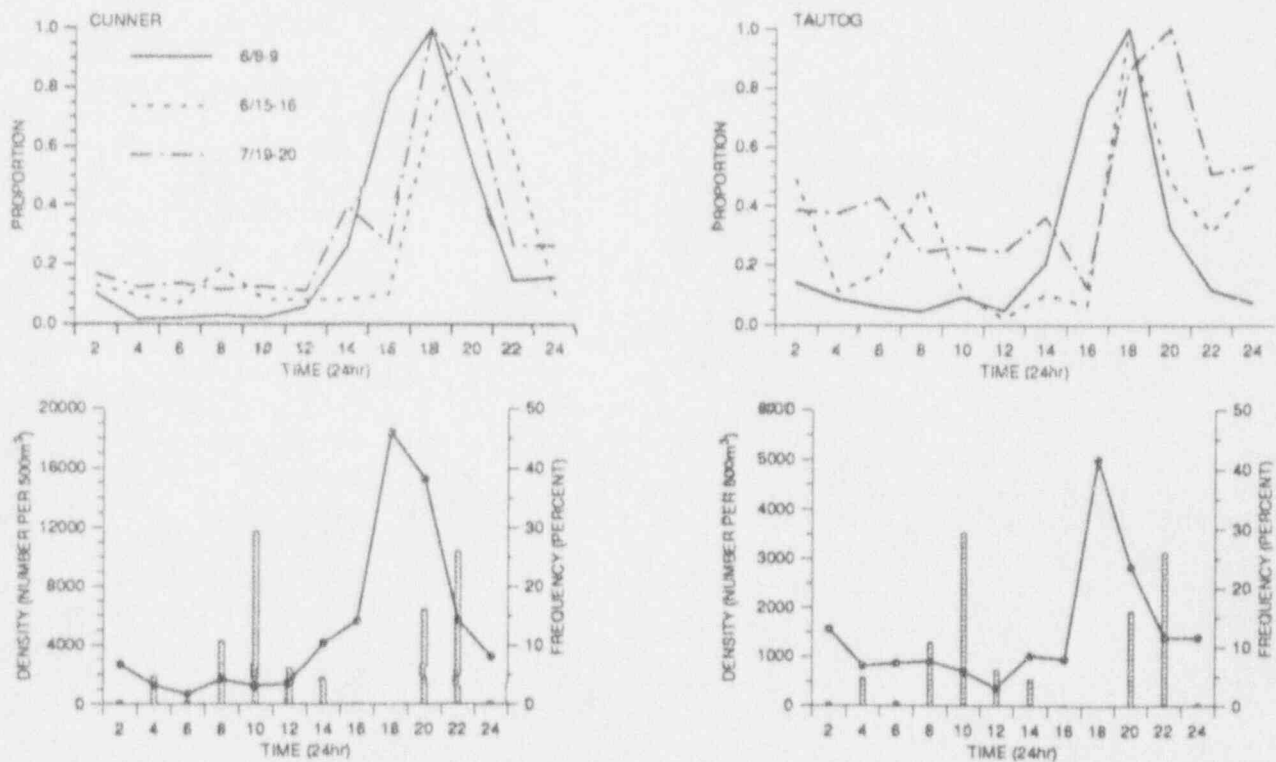


Fig. 18. Daily proportional abundance (top) (sample density/maximum samples density for each study) of cunner and tautog eggs for the three 24-hour studies at station EN and the geometric mean density of the three 24-hour studies combined (bottom) compared to frequency of sampling (percent in 2 hour increments) for entrainment collections taken during June through August from 1983 through 1992.

TABLE 17. Geometric mean (no. per 500 m<sup>3</sup> and 95% confidence intervals (CI) for cunner and tautog eggs collected at station EN during three 24-hour studies in 1993.

Date	Cunner (95% CI)	Tautog (95% CI)
June 8-9	1,345 (562-3,216)	386 (205-725)
June 15-16	4,671 (2,343-9,314)	1,167 (547-2,491)
July 19-20	4,120 (2,596-6,540)	2,639 (1,871-3,723)

followed by an even faster decrease, suggesting a short daily period of spawning for both cunner and tautog. The timing of peak abundance could not be related to tidal stage, because sampling for the June 8-9 and June 15-16 studies occurred during opposite tidal stages. Examination of the geometric mean for each 2-hour sampling period for the sampling dates combined showed that, on the average, daily peak spawning for cunner and tautog occurred at about 1800 hours (Fig. 18). The rapid decline in abundance from 1800 hours to 2400 hours for cunner and to 2200 hours for tautog cannot be attributed to hatching since egg incubation takes longer than a day. Therefore, this decline could be caused by high natural mortality, such as predation, suggested by Williams et al. (1973). Estimated mortality during this rapid decline period was 44% per hour for cunner and 47% per hour for tautog. Very little mortality apparently occurs during the remaining of the 24-hour period. These high egg mortality rates may account for the low number of cunner and tautog larvae collected compared to the large number of eggs.

Because of the daily cycle in wrasse egg abundance, the time of day when entrainment samples were collected was examined to determine if it could have resulted in biased estimates of average daily abundance. The period examined was 1983-92, during the months when wrasse eggs were abundant (June through August). The collection frequency distributions were grouped into the 2-hour intervals used for the 24-hour studies. The geometric mean density of all samples collected during the three 24-hour studies was 2,958 per 500 m<sup>3</sup> (95% CI of 1,971-4,440) for cunner and 1,059 (705-1,591) for tautog. The geometric mean density weighted by the historical sampling frequency distribution using the 2-hour average abundance was 2,974 for cunner and 1,000 for tautog. The similarity between these two abundance estimates for both cunner and tautog suggested that the data used for estimating annual entrainment had not been biased by the daily cycle of wrasse egg abundance.

## Conclusions

Abundance estimates were calculated for various life stages of fish in the vicinity of MNPS to help assess the effects of station operations. Impacts (entrainment, impingement and thermal changes) affecting fish populations are easily measured but quantifying the long-term effects of these impacts is far more difficult. The numbers of fish eggs and larvae entrained through the MNPS cooling-water system have been reliably estimated. How this loss actually affects local populations is influenced by many mechanisms such as compensatory mortality, density-dependent growth, fecundity of individual species, age structure of the population and life history strategies. Impingement of fish at the MNPS intakes can also be measured but, as in the case of eggs and larvae, the implications of removing these juvenile and adult fish from local populations are difficult to ascertain. In addition to impingement, fish populations are also affected by natural and fishing mortality and, furthermore, fish that only inhabit the area seasonally can be greatly affected by events that occur outside the MNPS area. Changes in the thermal regime of local waters are easy to measure and are well-documented. If water temperatures exceed the fish's tolerance levels, they can move out of the area. The loss of available habitat may alter local populations, especially if the area the fish vacates is a major spawning or nursery ground. Trawl, seine and ichthyoplankton monitoring programs have successfully measured impacts from MNPS on local fish populations, provided a good basis for identifying which taxa could be potentially impacted, as well as long-term abundance trends which can be used to assess changes in local populations.

Six species (American sand lance, anchovies, grubby, silversides, tautog and cunner) have been identified as having the potential to be impacted by MNPS, either by entrainment or by exposure to elevated seawater temperatures. Annual abundance estimates, length, and age based on length were analyzed to determine if changes have occurred during MNPS operation. Downward abundance trends were found for some life stages in four (American sand lance and anchovy larvae, and cunner and tautog adults) of the six species. Both American sand lance and bay anchovy inhabit the area for short periods of time, exhibit large year to year fluctuations, and are probably more affected by events elsewhere. Cunner and tautog adults have declined in trawl catches

concurrently with a shift to juveniles accounting for a high proportion of the catch. Entrainment of cunner and tautog eggs was identified as the primary potential impact to these fish because more than 85% of the eggs entrained at MNPS were of these two species. Studies conducted in 1993 suggested that the mortality of these entrained eggs was over 90%. The shift in age structure from older to younger could not be attributed to entrainment, because entrainment loss would be expected to shift the age distribution towards larger, older fish because of the loss of new recruits.

### References Cited

- Botelho, V.M., and G.T. Donnelly. 1978. A statistical analysis of the performance of the Bourne plankton splitter, based on test observations. NMFS unpub. ms.
- Cushing, D.H. 1974. The possible density-dependence of larval mortality and adult mortality in fishes. Pages 103-111 in J.H.S. Blaxter, ed. The early life history of fish. Springer-Verlag, New York.
- Cushing, D.H., and J.G.K. Harris. 1973. Stock and recruitment and the problem of density dependence. Rapp. P.-v. Cons. int. Explor. Mer 164:142-155.
- DeAngelis, D.L., S.W. Christensen, and A.G. Clark. 1977. Response of a fish population model to young-of-the-year mortality. Oak Ridge Nat. Lab. Publ. No. 1065.
- Draper, N., and H. Smith. 1981. Applied regression analysis. John Wiley and Sons, New York 709pp.
- Hennemuth, R.C., J.E. Palmer, and B.E. Brown. 1980. A statistical description of recruitment in eighteen selected fish stocks. J. Northwest Atl. Fish. 1:101-111.
- Leak, J.C., and E.D. Houde. 1987. Cohort growth and survival of bay anchovy, *Anchoa mitchelli*, larvae in Biscayne Bay, Florida. Mar. Ecol. 37:109-122.
- McHugh, J. L. 1977. Fisheries and fishery resources of New York Bight. NOAA Tech. Rep. NMFS Circ. 401. 51 pp.
- Monteleone, D. M., W.T. Peterson, and G.C. Williams. 1987. Interannual fluctuations in the density of sand lance, *Ammodytes americanus*, larvae in Long Island Sound, 1951-1983. Estuaries 10, 246-254.
- Northeast Utilities Service Company (NUSCO). 1988. Delta distribution. Pages 311-320 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Ct. Ann. Rpt, 1987.
- NUSCO. 1990. Fish ecology. Pages 81-118 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Ct. Ann. Rpt, 1989.
- NUSCO. 1991. Fish ecology. Pages 89-125 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Ct. Ann. Rpt, 1990.
- NUSCO. 1992. Fish ecology. Pages 111-156 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Ct. Ann. Rpt, 1991.
- NUSCO. 1993. Fish ecology. Pages 153-180 in Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Ct. Ann. Rpt, 1992.
- Pennington, M. 1983. Efficient estimators of abundance for fish plankton surveys. Biometrics 39:281-286.
- Pennington, M. 1986. Some statistical techniques for estimating abundance indices from trawl surveys. Fish. Bull., U.S. 84:519-525.
- SAS Institute, Inc. 1985. SAS user's guide: Statistics. Version 5 ed. SAS Institute Inc. Cary, N.C. 956 pp.
- Serchuk, F.M. 1972. The ecology of the cunner, *Tautoglabrus adspersus* (Walbaum) (Pisces: Labridae), in the Wewantic River Estuary, Wareham, Massachusetts. M.S. Thesis, Univ. Massachusetts, Amherst, MA. 111 pp.
- Simpson, D.G. 1989. Population dynamics of the tautog, *Tautoga onitis*, in Long Island Sound. M.S. Thesis, So. Conn. State Univ., New Haven, Ct. 65 pp.
- Williams, G.C. 1967. Identification and seasonal size changes of eggs of the labrid fishes, *Tautoglabrus adspersus* and *Tautog onitis*, of Long Island Sound. Copeia 1967:452-453.
- Williams, G.C., D.C. Williams, and R.J. Miller. 1973. Mortality rates of planktonic eggs of the cunner, *Tautoglabrus adspersus* (Walbaum), in Long Island Sound. Pages 181-195 in A. Pacheco, ed. Proceeding of a workshop on egg, larval and juvenile stages of fish in Atlantic Coast estuaries. Nat. Mar. Fish. Ser., Middle Atl. Coast. Fish. Ctr. Tech. Publ. No. 1.

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

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



## APPENDIX I. (continued).

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



APPENDIX II. Total number of samples collected and number of fish caught by trawl for each report year (two-unit operational period:1976-77 through 1985-86; three-unit operational period:1986-87 through 1992-93).

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
Number of samples	468	468	468	468	468	467	474	480	468	468	468	465	468	468	468	468	468
Taxon <sup>a</sup>																	
<i>P. americanus</i>	7,415	6,045	7,236	11,442	13,296	10,749	19,201	12,560	13,260	9,849	9,321	8,877	13,440	8,690	9,378	8,511	9,828
<i>S. chrysops</i>	1,918	4,040	2,556	4,094	3,844	3,403	4,896	5,268	4,206	2,640	5,205	3,632	3,294	2,869	10,497	25,287	9,710
<i>S. aquosus</i>	1,480	1,296	875	1,508	2,016	1,518	3,517	2,475	2,199	2,483	1,655	1,966	2,399	2,735	1,656	876	1,519
<i>Raja</i> spp.	661	579	362	402	954	696	2,797	2,493	1,583	3,801	2,207	2,183	2,864	2,437	2,858	2,872	1,892
<i>Anchoa</i> spp.	979	580	2,226	16	109	578	38	109	157	10,003	8,038	292	496	1,241	31	1,557	80
<i>Menidia</i> spp.	2,152	1,647	1,463	1,340	882	501	518	583	322	519	3,438	698	982	485	474	1,346	3,567
<i>M. aeneus</i>	266	636	297	342	632	870	996	672	477	341	727	434	989	615	640	451	857
Gadidae	112	326	230	211	3,296	1,424	476	481	562	630	168	593	88	84	121	106	207
<i>T. adspersus</i>	838	875	400	1,399	940	840	611	362	248	119	147	63	205	109	103	141	401
<i>Prionotus</i> spp.	338	322	138	313	405	661	1,059	422	371	395	436	159	356	1,277	363	435	327
<i>P. triacanthus</i>	37	44	407	174	44	69	182	244	19	135	132	111	1,831	179	1,878	426	1,302
<i>P. dentatus</i>	286	141	92	75	122	240	250	269	1,937	281	653	617	360	80	393	403	634
<i>Urophycis</i> spp.	99	87	103	69	163	313	615	286	251	272	286	164	174	141	335	91	964
<i>M. bilinearis</i>	425	163	69	134	558	220	382	147	100	175	197	118	73	321	124	179	337
<i>G. aculeatus</i>	30	12	47	77	206	103	63	218	1102	116	354	405	94	10	15	447	172
<i>E. microstomus</i>	43	7	0	3	31	91	94	56	85	218	640	190	359	62	492	394	694
<i>T. onitis</i>	229	283	263	270	146	228	239	140	119	134	215	87	162	85	185	111	131
<i>P. gunnellus</i>	85	106	99	65	251	273	302	145	127	151	186	203	407	189	155	126	152
<i>S. fuscus</i>	43	54	49	88	151	264	232	202	254	196	207	275	321	85	154	134	175
<i>O. mordax</i>	111	286	90	5	123	63	89	26	227	391	257	249	152	26	48	35	334
<i>H. americanus</i>	34	48	39	148	278	410	557	377	125	41	45	11	3	7	12	38	1
<i>A. quadracus</i>	10	6	24	27	194	765	76	11	112	130	107	52	31	11	18	100	69
<i>B. tyrannus</i>	1	14	11	1	1	1	0	1	0	34	10	4	1	1,320	5	205	64
<i>C. striata</i>	33	9	3	4	10	63	23	38	30	80	412	16	53	69	130	94	60
<i>P. oblongus</i>	31	7	21	11	51	32	138	34	81	66	72	28	123	155	92	28	55
<i>M. octodecemspinosus</i>	11	10	97	40	30	145	172	51	20	13	12	5	12	18	56	222	9
<i>A. pseudoharengus</i>	11	272	13	17	4	15	5	26	4	16	208	1	4	3	14	41	35
<i>O. iau</i>	98	21	7	18	31	35	25	23	24	32	56	51	58	30	55	17	5
<i>A. americanus</i>	5	59	128	36	117	14	19	11	19	6	11	29	1	1	1	1	2
<i>A. rostrata</i>	19	16	8	5	10	37	29	24	22	34	28	22	20	5	15	8	2
<i>C. lumpus</i>	19	11	28	58	11	0	14	1	29	1	1	44	6	1	7	6	21
<i>C. regalis</i>	9	21	4	2	2	45	7	0	1	5	36	5	14	9	6	1	5
<i>Liparis</i> spp.	9	27	10	10	18	33	15	16	11	3	18	8	12	22	2	3	22
<i>S. maculatus</i>	16	10	1	0	9	14	16	15	7	7	3	1	9	14	26	50	26
<i>A. sapidissima</i>	33	6	1	5	40	12	0	29	0	0	1	1	9	5	3	7	1
<i>C. harengus</i>	1	9	13	0	0	1	0	2	9	63	10	2	1	2	1	10	19
<i>Alosa</i> spp.	0	0	0	0	0	0	0	0	0	0	0	4	11	26	26	34	52
Clupeidae	2	1	0	0	0	0	0	0	0	110	0	0	0	0	0	0	0
<i>M. canis</i>	2	5	45	11	1	5	4	6	0	2	2	1	2	2	4	2	0
<i>Hippocampus erectus</i>	0	0	0	0	0	0	0	1	4	7	20	12	6	4	17	67	4
<i>P. ferrugineus</i>	7	5	5	2	3	15	6	0	4	0	0	23	0	3	0	3	0
<i>M. hispidus</i>	3	6	8	4	0	0	8	1	8	9	2	2	2	11	9	5	1
<i>A. aestivialis</i>	3	11	8	12	4	1	1	17	5	2	4	2	2	0	0	10	4
Gobiidae	3	0	0	0	4	0	0	3	9	7	2	5	10	2	23	17	1
<i>M. americana</i>	8	17	3	5	8	2	1	0	0	0	0	0	5	11	0	3	3
<i>F. labacaria</i>	2	3	0	0	3	0	1	0	8	1	2	0	1	3	22	9	5
<i>M. americanus</i>	5	7	9	2	2	2	2	2	3	1	0	6	2	2	0	10	1
<i>S. setipinnis</i>	0	0	0	0	0	0	0	0	1	0	0	0	30	0	0	0	1
<i>A. schoepfi</i>	0	2	2	1	1	0	0	1	1	2	2	0	3	4	6	1	1
<i>D. volitans</i>	3	0	0	0	0	1	3	1	0	1	3	4	1	2	4	3	3
<i>L. xanthurus</i>	5	6	0	0	0	0	2	0	0	3	1	0	5	0	0	0	0
<i>O. marginatum</i>	0	0	0	0	0	0	0	0	1	2	4	4	4	0	4	3	2
<i>P. cruentatus</i>	0	0	0	0	0	1	0	2	3	1	1	1	1	0	0	0	0
<i>P. salatrix</i>	1	1	0	2	1	2	3	3	0	0	0	2	1	0	1	11	0
<i>Men. saxatilis</i>	0	1	0	1	0	3	1	0	0	0	4	2	1	4	3	0	1
<i>S. vomer</i>	1	2	0	0	0	0	0	0	0	0	1	1	11	1	0	0	0

## APPENDIX II (continued).

Year	76-77	77-78	78-79	79-80	80-81	81-82	82-83	83-84	84-85	85-86	86-87	87-88	88-89	89-90	90-91	91-92	92-93
Taxon <sup>a</sup>																	
<i>P. pungitius</i>	0	0	0	0	1	2	0	0	5	1	5	0	0	0	0	0	0
Gasterosteidae	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Fundulus</i> spp.	0	0	0	0	0	5	2	0	0	2	1	0	1	0	0	0	0
<i>L. americanus</i>	2	0	0	0	1	0	1	1	0	0	1	4	0	0	2	3	0
<i>U. subbifurcata</i>	0	2	0	0	1	1	0	0	0	1	1	1	4	0	1	2	3
<i>Mor. saxatilis</i>	0	0	2	1	0	1	1	0	0	1	0	0	0	0	0	0	0
<i>S. borealis</i>	0	0	0	0	0	0	0	1	1	2	0	1	6	0	0	0	0
<i>S. foetens</i>	0	1	4	0	0	3	1	0	0	0	0	0	0	0	0	0	0
<i>C. oceanicus</i>	1	0	0	0	1	0	0	0	2	0	1	1	1	3	0	1	1
<i>M. auranus</i>	0	0	1	0	0	0	2	0	0	1	0	0	4	0	0	0	1
<i>P. arenatus</i>	0	0	0	0	0	0	0	0	2	1	0	0	0	1	3	1	0
<i>T. lathamii</i>	0	0	0	4	0	0	0	0	0	0	0	0	4	0	0	0	0
<i>T. maculatus</i>	3	1	0	0	0	0	0	0	0	1	2	1	0	0	0	0	0
<i>C. ocellatus</i>	0	0	0	0	1	0	0	1	0	0	1	1	1	0	3	0	0
<i>G. wheatlani</i>	0	0	0	0	0	1	1	1	0	1	2	0	1	0	0	1	1
<i>Lactophrys</i> spp.	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	3	0
<i>P. alba</i>	0	0	0	0	0	1	0	0	2	1	1	0	0	0	0	0	0
<i>C. crysos</i>	0	0	0	0	1	0	1	0	1	2	0	0	0	0	0	0	0
<i>C. hippos</i>	0	0	0	0	0	0	1	0	0	1	0	0	0	2	0	0	0
<i>E. cimbrius</i>	0	0	0	0	0	1	0	0	0	0	1	1	1	0	0	1	3
<i>M. cephalus</i>	0	0	0	0	0	0	1	0	0	2	0	0	0	2	0	0	0
<i>L. parva</i>	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
<i>S. scombrus</i>	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	1	0
<i>S. acanthias</i>	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	4	0
<i>A. mediocris</i>	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>C. variegatus</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
<i>D. macarellus</i>	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
<i>M. aeglefinus</i>	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0
<i>A. oxyrinchus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>A. maculatus</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>B. chrysoura</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>B. brasme</i>	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>D. centroura</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Monacanthus</i> spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
<i>M. fremivillei</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Myoxocephalus</i> spp.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Ophidiidae	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>S. trutta</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>S. retifer</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>S. crumenophthalmus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>S. marina</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>T. myops</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>T. falcatus</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>U. parvus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	3	0
Total	17,941	18,147	17,497	22,469	29,010	24,773	37,699	27,860	28,169	33,546	35,566	21,674	29,529	23,477	30,473	44,957	33,768

<sup>a</sup> Fish identified to the lowest practical taxon.

APPENDIX III. Total number of samples collected and number of fish caught by trawl at each station (June 1976-May 1993).

Station	JC	NR	NB	TT	BR	IN	TOTAL
Number of samples:	1,328	1,327	1,329	1,329	1,329	1,329	7,502
Taxon <sup>a</sup>							
<i>P. americanus</i>	15,247	66,574	21,665	25,605	21,506	28,501	179,098
<i>S. chrysops</i>	9,030	566	34,696	15,751	10,101	27,215	97,359
<i>S. aquosus</i>	1,701	3,315	2,823	4,258	14,327	5,749	32,173
<i>Raja</i> spp.	1,867	39	4,918	8,296	11,985	4,536	31,641
<i>Anchoa</i> spp.	3,342	1,177	18,223	339	20	3,429	26,530
<i>Menidia</i> spp.	4,529	6,356	2,176	1,644	457	5,755	20,917
<i>M. aeneus</i>	1,375	5,040	557	590	860	1,820	10,242
Gadidae	2,084	834	2,840	1,039	279	2,039	9,115
<i>Prionotus</i> sp.	118	1,013	684	1,341	3,666	955	7,777
<i>T. adspersus</i>	1,906	386	631	283	527	4,066	7,799
<i>P. dentatus</i>	1,032	1,825	932	1,794	276	974	6,833
<i>P. triacanthus</i>	73	13	1,001	1854	2,858	1,415	7,214
<i>Urophycis</i> spp.	530	82	448	405	2,384	564	4,413
<i>M. bilinearis</i>	199	10	532	669	1,558	754	3,722
<i>G. aculeatus</i>	2,076	1,349	17	12	6	11	3,471
<i>T. onitis</i>	745	813	305	206	271	684	3,024
<i>P. gunnellus</i>	1,551	515	322	230	39	365	3,022
<i>E. microstomus</i>	247	35	717	327	1,574	561	3,461
<i>S. fuscus</i>	844	1,458	163	100	116	203	2,884
<i>O. mordax</i>	1,421	300	245	174	96	276	2,512
<i>H. americanus</i>	444	82	405	298	540	405	2,174
<i>A. quadracus</i>	181	1,557	1	1	1	2	1,743
<i>B. tyrannus</i>	532	1,056	40	6	1	32	1,673
<i>C. striata</i>	94	212	74	53	74	620	1,127
<i>P. oblongus</i>	0	12	73	12	900	28	1,025
<i>M. octodecemspinosus</i>	3	0	20	53	831	16	923
<i>A. pseudoharengus</i>	10	67	57	30	274	251	689
<i>O. tau</i>	9	565	0	0	0	12	586
<i>A. americanus</i>	20	96	6	29	300	9	460
<i>A. rostrata</i>	43	230	0	20	3	8	304
<i>C. lumpus</i>	161	9	21	8	2	57	258
<i>Liparis</i> spp.	26	11	38	40	97	27	234
<i>S. maculatus</i>	19	101	17	9	18	60	221
<i>C. regalis</i>	22	1	27	11	77	34	172
<i>A. sapidissima</i>	8	17	56	12	38	22	153
<i>Hippocampus erectus</i>	53	63	9	3	2	12	142
<i>C. harengus</i>	69	4	27	12	31	5	148
Clupeidae	0	1	0	1	0	111	113
<i>Alosa</i> spp.	7	31	11	21	73	10	153
<i>M. canis</i>	10	1	40	3	36	4	94
Gobiidae	4	78	2	0	0	2	86
<i>A. aestivalis</i>	1	29	17	11	14	14	86
<i>M. hispidus</i>	20	2	11	10	22	14	79
<i>P. ferrugineum</i>	0	2	0	8	66	0	76
<i>M. americana</i>	8	23	6	2	6	22	67
<i>M. americanus</i>	0	0	0	1	55	2	58
<i>F. tabacaria</i>	38	13	1	0	0	8	60
<i>S. setapinnis</i>	16	0	8	2	0	6	32
<i>P. saltatrix</i>	3	5	6	8	5	1	28
<i>A. schoepfi</i>	10	0	4	2	4	7	27
<i>D. volitans</i>	2	13	0	0	0	14	29
<i>L. xanthurus</i>	4	0	8	0	4	6	22
<i>O. marginatum</i>	5	6	2	2	8	1	24
<i>P. cruentatus</i>	2	0	2	5	2	8	19
<i>S. vomer</i>	1	1	14	0	0	1	17

## APPENDIX III (continued).

Station	JC	NR	NB	TT	BR	IN	TOTAL
Taxon *							
<i>Men. saxatilis</i>	1	3	4	5	1	3	17
<i>L. americanus</i>	1	0	2	2	9	1	15
<i>P. pungitius</i>	10	3	0	0	0	1	14
<i>U. subbifurcata</i>	3	0	1	1	11	1	17
Gasterosteidae	2	11	0	0	0	0	13
<i>Fundulus</i> spp.	1	11	0	0	0	0	12
<i>C. oceanicus</i>	1	4	3	2	2	0	12
<i>Mor. saxatilis</i>	0	11	0	0	0	0	11
<i>S. borealis</i>	5	6	0	0	0	0	11
<i>S. foetens</i>	0	3	0	2	5	1	11
<i>Lactophrys</i> spp.	6	2	0	0	0	1	9
<i>P. arenatus</i>	0	1	1	0	2	5	9
<i>G. wheatlandi</i>	7	2	0	0	0	0	9
<i>M. auratus</i>	1	0	0	1	1	6	9
<i>T. lathami</i>	4	0	3	0	0	1	8
<i>T. maculatus</i>	5	2	0	0	0	1	8
<i>C. ocellatus</i>	2	2	1	0	0	2	7
<i>S. acanthias</i>	0	0	1	0	6	0	7
<i>P. alba</i>	3	0	0	1	1	1	6
<i>C. crysos</i>	0	0	2	0	1	2	5
<i>E. cimbrius</i>	2	0	0	0	3	2	7
<i>C. hippos</i>	0	0	0	0	0	4	4
<i>M. cephalus</i>	1	2	1	0	0	0	4
<i>S. scombrus</i>	0	0	1	1	1	1	4
<i>L. parva</i>	0	0	0	0	3	0	3
<i>A. mediocris</i>	1	0	0	0	1	0	2
<i>C. variegatus</i>	0	1	0	1	0	0	2
<i>D. macarellus</i>	1	0	1	0	0	0	2
<i>M. aeglefinus</i>	1	0	0	1	0	0	2
<i>A. oxyrhynchus</i>	0	0	1	0	0	0	1
<i>A. maculatus</i>	1	0	0	0	0	0	1
<i>B. chrysoura</i>	0	0	0	0	1	0	1
<i>B. brasme</i>	0	0	0	1	0	0	1
<i>D. centoura</i>	1	0	0	0	0	0	1
<i>Monocanthus</i> spp.	0	1	0	0	0	0	1
<i>M. fremantillei</i>	0	0	1	0	0	0	1
<i>Myoxocephalus</i> spp.	0	1	0	0	0	0	1
Ophidiidae	0	0	0	0	1	0	1
<i>P. marinus</i>	0	0	0	1	0	0	1
<i>S. trutta</i>	0	1	0	0	0	0	1
<i>S. retifer</i>	0	1	0	0	0	0	1
<i>S. crumenophthalmus</i>	0	0	0	1	0	0	1
<i>S. marina</i>	0	0	0	0	1	0	1
<i>T. myops</i>	1	0	0	0	0	0	1
<i>T. falcatus</i>	0	0	0	0	0	0	1
<i>U. parvus</i>	1	0	0	0	0	0	1
Total	51,805	96,057	94,921	65,610	76,451	91,737	476,581

\* Fish identified to the lowest practical taxon.

APPENDIX IV. Total number of samples collected and number of fish caught by seine for each report year (two-unit operational period:1976-77 through 1985-86; three-unit operational period:1986-87 through 1992-93).

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
Number of samples	66	72	72	72	72	72	98	120	174	156	156	156	156	180	180	180	135
Taxon *																	
<i>Menidia</i> spp.	40,619	18,194	1,335	1,062	7,996	3,186	5,413	9,807	1,538	1,375	5,441	8,542	6,107	5,044	11,191	8,596	9,503
<i>Fundulus</i> spp.	1,695	1,199	815	659	952	613	915	1,081	1,463	906	111	432	3,142	831	859	1,224	513
<i>A. quadricus</i>	464	603	258	266	49	94	89	1,827	167	106	297	98	152	302	123	1,078	55
<i>C. variegatus</i>	48	673	39	30	10	352	146	50	29	28	2	2	21	3	30	1,170	76
<i>A. americanus</i>	6	520	16	51	10	318	82	30	21	0	7	1	4	0	47	156	27
<i>P. saltairex</i>	1	0	1	6	0	2	135	4	19	35	12	12	5	6	825	14	0
<i>B. tyrannus</i>	0	0	17	0	4	0	7	1	0	8	6	6	3		521	2,652	2
<i>P. pungitius</i>	5	1	28	2	5	2	10	321	8	11	8	4	30	24	3	6	0
<i>S. fuscus</i>	9	3	9	108	6	8	21	12	35	30	33	19	74	11	17	40	11
<i>G. aculeatus</i>	9	154	27	5	3	2	5	53	6	6	19	15	38	8	0	3	0
<i>P. americanus</i>	4	6	4	1	6	5	2	3	17	40	18	17	16	48	9	10	2
<i>M. cephalus</i>	0	4	3	23	41	1	4	4	1	0	38	4	46	0	0	1	1
<i>A. pseudoharengus</i>	0	0	0	0	0	0	0	1	93	0	0	4	0	6	0	0	0
Gadidae	2	0	9	2	20	16	11	8	11	11	8	0	2	2	0	0	1
<i>G. wheatlandi</i>	0	0	0	0	0	8	6	6	19	12	9	22	9	8	0	1	0
<i>M. curema</i>	0	0	0	0	0	0	0	1	9	0	0	0	43	3	22	1	0
<i>C. harengus</i>	0	0	0	0	0	0	2	0	0	0	30	0	6	1	0	0	0
<i>L. parva</i>	1	2	0	0	0	0	0	2	0	1	0	16	14	2	1	32	7
<i>A. rostrata</i>	10	5	12	3	2	0	1	1	0	0	3	0	0	0	0	0	0
<i>T. falcatus</i>	0	0	1	0	3	0	0	0	0	0	0	0	22	7	0	0	0
<i>M. aeneus</i>	3	2	1	2	0	0	3	1	3	3	3	2	4	0	0	0	3
<i>O. mordax</i>	0	0	0	0	0	0	0	0	0	2	0	0	18	0	0	0	0
<i>Anchoa</i> spp.	0	0	0	0	2	0	7	2	1	0	0	0	0	0	4	0	0
<i>T. onitis</i>	0	0	0	0	0	0	4	0	0	0	0	0	2	0	7	1	1
<i>A. aestivus</i>	2	6	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0
<i>Gasterosteus</i> spp.	0	0	0	0	0	0	0	0	0	0	12	0	0	0	0	0	1
<i>C. hippos</i>	0	0	1	0	0	1	0	0	0	1	0	0	4	1	3	0	0
<i>S. maculatus</i>	0	0	0	1	0	0	1	0	0	3	3	0	1	0	1	0	0
<i>T. atropisus</i>	0	0	2	0	0	0	3	0	1	0	0	0	0	0	1	0	0
<i>A. sapidissima</i>	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Men. saxatilis</i>	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>P. triacanthus</i>	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>P. gunnellus</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0
<i>S. aquaticus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	0
<i>S. marina</i>	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	1
Clupeidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>C. regalis</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>S. setapinnis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>S. vomer</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1
<i>Prionotus</i> spp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Urophycis</i> spp.	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Total	42,881	21,372	2,579	2,221	9,109	4,609	6,869	13,215	3,444	2,582	6,061	9,196	9,763	6,336	13,667	15,012	10,204

\* Fish identified to the lowest practical taxon.



APPENDIX V. Total number of samples collected and number of fish caught by seine at each station (June 1976-May 1993).

Year	JC	GN	WP	TOTAL
Number of samples	693	740	747	2,180
Taxon <sup>a</sup>				
<i>Menidia</i> spp.	99,702	24,200	21,236	145,138
<i>Fundulus</i> spp.	13,413	2,141	1,876	17,430
<i>A. quadracus</i>	5,992	19	21	6,032
<i>B. tyrannus</i>	715	13	2,524	3,252
<i>C. variegatus</i>	1,831	845	34	2,710
<i>A. americanus</i>	4	213	1,080	1,297
<i>P. saltatrix</i>	945	50	82	1,077
<i>P. pungitius</i>	357	102	10	469
<i>S. fuscus</i>	94	64	288	446
<i>G. aculeatus</i>	277	29	47	353
<i>P. americanus</i>	43	12	154	209
<i>M. cephalus</i>	99	43	29	171
<i>A. pseudoharengus</i>	8	96	0	104
Gadidae	66	31	6	103
<i>G. wheatlandi</i>	36	26	38	100
<i>M. curema</i>	63	14	2	79
<i>L. parva</i>	66	7	5	78
<i>Anchoa</i> spp.	16	2	24	42
<i>C. harengus</i>	39	0	0	39
<i>A. rostrata</i>	31	2	4	37
<i>T. falcaus</i>	30	3	0	33
<i>M. aeneus</i>	8	13	9	30
<i>O. mordax</i>	18	0	2	20
<i>T. onitis</i>	12	2	1	15
<i>A. aestivalis</i>	3	6	3	12
<i>Gasterosteus</i> spp.	0	1	11	12
<i>C. hippas</i>	10	0	2	12
<i>S. maculatus</i>	0	2	8	10
<i>T. adspersus</i>	6	1	0	7
<i>S. aquosus</i>	0	0	3	3
<i>A. sapidissima</i>	0	0	2	2
<i>Men. saxatilis</i>	1	0	1	2
<i>P. triacanthus</i>	0	1	1	2
<i>P. gunnellus</i>	0	0	2	2
<i>S. marina</i>	2	0	0	2
Clupeidae	0	1	0	1
<i>C. regalis</i>	1	0	0	1
<i>Prionotus</i> spp.	0	1	0	1
<i>S. setapinnis</i>	0	1	0	1
<i>S. vomer</i>	1	0	0	1
<i>Urophycis</i> spp.	0	1	0	1
Total	123,890	27,942	27,505	179,337

<sup>a</sup> Fish identified to the lowest practical taxon.



## Winter Flounder Studies

Introduction .....	143
Materials and Methods .....	144
Sampling programs .....	144
Adult winter flounder sampling .....	144
Larval winter flounder sampling .....	145
Juvenile winter flounder sampling .....	147
Indices of abundance .....	148
Relative annual abundance of adults .....	148
Absolute abundance estimates of adults .....	149
Adult spawning stock size and egg production .....	149
Development and growth, abundance, and mortality of larvae .....	149
Abundance, growth, and mortality of juveniles in summer .....	151
Abundance of juveniles during fall and winter .....	151
Stock and recruitment relationship .....	151
Assessment of MNPS operation on Niantic River winter flounder .....	153
Estimates of larval entrainment at MNPS .....	154
Mass-balance calculations .....	154
Stochastic simulation of winter flounder stock dynamics .....	156
Results and Discussion .....	164
Seawater temperature .....	164
Adult winter flounder .....	164
Relative annual abundance .....	164
Absolute abundance estimates .....	167
Spawning stock size and egg production .....	171
Larval winter flounder .....	172
Abundance and distribution .....	172
Development and growth .....	178
Mortality .....	183
Juvenile winter flounder .....	185
Age-0 juveniles (summer) .....	185
Age-0 juveniles (late fall and early winter) .....	196
Age-1 juveniles (late winter) .....	198
Comparisons among life-stages of winter flounder year-classes .....	200
Stock-recruitment relationship (SRR) .....	202
MNPS impact assessment .....	207
Larval entrainment .....	207
Stochastic simulation of the Niantic River winter flounder stock .....	214
Conclusions .....	220
References Cited .....	222



# Winter Flounder Studies

## Introduction

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

summarized by Klein-MacPhee (1978). Because the early life history of the congeneric European plaice (*Pleuronectes platessa*) has many similarities to that of the winter flounder, relevant literature was also reviewed to gain further insights into winter flounder population dynamics.

MNPS operation results in the impingement of juvenile and adult winter flounder on the traveling screens of the cooling-water intakes and the entrainment of larvae through the condenser cooling-water system. Studies of the winter flounder found near MNPS started in 1973 to obtain data needed to assess power plant effects on the Niantic River stock. Although knowledge of annual variability is important for assessing short-term impacts, the most significant changes in fisheries tend to occur on longer time scales (Cushing 1977; Steele et al. 1980). Therefore, the development of a long-term assessment capability was the ultimate research goal of the NUSCO winter flounder studies. A combination of sampling programs and analytical methods is presently used to examine the current abundance of the Niantic River population for annual assessments of the spawning stock. A computer population simulation model is used for assessing long-term effects of MNPS operation. The impact of fish impingement at MNPS has been largely mitigated by the installation and operation of fish return sluiceways at MNPS Units 1 and 3. The mortality of entrained larvae potentially has greater significance as the winter flounder, unlike many marine fishes, is a product of local spawning with geographically isolated stocks associated with individual estuaries or specific coastal areas (Lobell 1939; Perlmuter 1947; Saita 1961). In particular, the population of winter flounder spawning in the nearby Niantic River has been studied in detail to assess the long-term effect of larval entrainment through the MNPS cooling-water system.

This report section summarizes data collected during 1992 and updates results reported previously (NUSCO 1993). The 1993 spawning season was the eighth year in which winter flounder have experienced impact from the operation of all three MNPS units. The NUSCO winter flounder stochastic population dynamics model (SPDM) was used to simulate the long-term effects of historical and projected rates of fishing mortality and simultaneous plant operation, resulting in annual mortalities from impingement of juveniles



and adults and the entrainment of larvae through the MNPS cooling-water system. The rate of annual production loss due to entrainment was determined from updated mass-balance calculations, which were first presented in NUSCO (1991b). The effect of impingement was included in the simulations as an equivalent annual instantaneous mortality rate added to fishing and was based on an analysis given in NUSCO (1992a).

## Materials and Methods

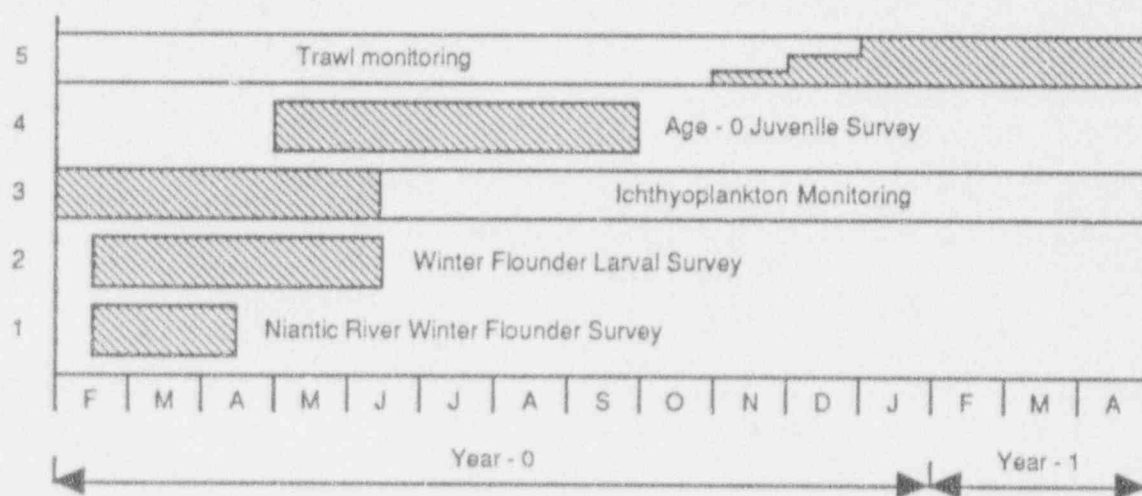
### Sampling programs

Data needed to assess MNPS impact on the winter flounder come from several biological sampling programs. Some programs (e.g., Niantic River adult and larval surveys, age-0 survey) were designed to investigate specific life history stages of winter flounder. Other information comes from year-round sampling of the entire local fish community, such as the trawl monitoring program (TMP) and ichthyoplankton monitoring programs at MNPS and in Niantic Bay. Additional information used in various assessments was presented in NUSCO (1987), which summarized

various life history studies of the winter flounder prior to the operation of Unit 3. Ongoing sampling programs that contributed data to the Niantic River winter flounder studies are shown in Figure 1, which includes the seasonal duration of sampling and timing relative to the annual life-cycle of Niantic River winter flounder. Brief descriptions of field methodologies used in these programs are given below. Information on water temperature was obtained from continuous temperature recorders at the intakes of MNPS Units 1 and 2; daily mean temperatures were determined from available records of 15-minute average temperatures. Monthly, seasonal, or annual means were calculated using daily means.

### Adult winter flounder sampling

Sampling methodology for the adult winter flounder spawning surveys in the Niantic River has remained basically unchanged since 1982 (NUSCO 1987). Each survey since 1982 started in February or early March, after most ice cover disappeared from the river, and continued through early April. Surveys ceased when the proportion of reproductively active females decreased to less than 10% of all females examined for



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

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

2 consecutive weeks, an indication of completion of most spawning. In each survey, the Niantic River was divided into a number of sampling areas, referred to as stations (Fig. 2). Since 1979 no samples have been taken outside of the navigational channel in the lower portion of the river because of an agreement

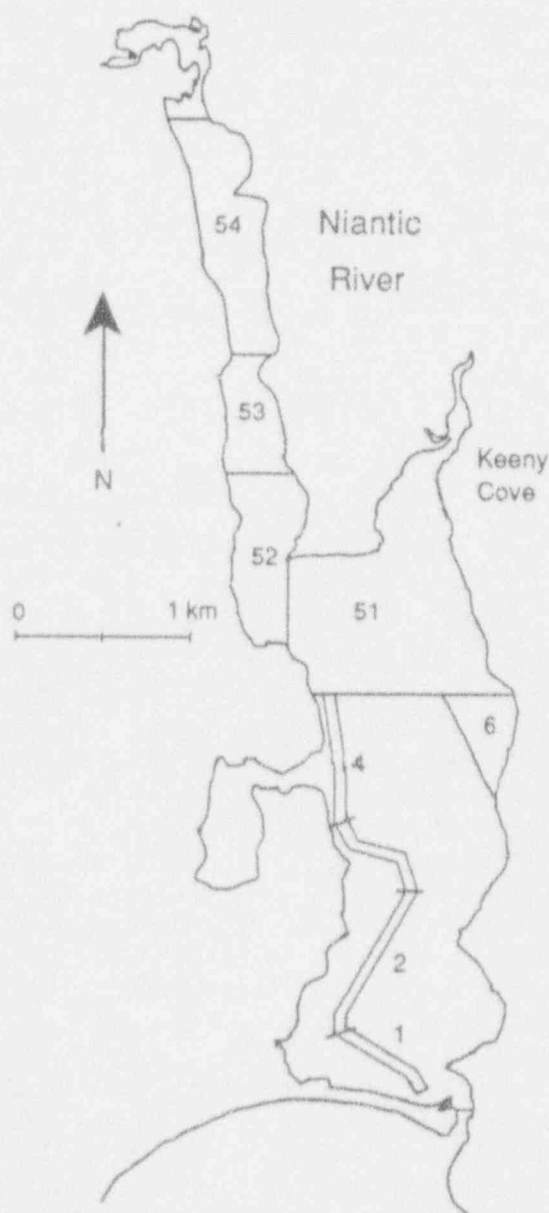


Fig. 2. Location of stations sampled for adult winter flounder during the spawning season in the Niantic River in 1993.

made with the East Lyme-Waterford Shellfish Commission to protect habitat of the bay scallop (*Argopecten irradians*). Winter flounder were collected on at least 2 days of each survey week using a 9.1-m otter trawl with a 6.4-mm bar mesh codend liner. Fish caught in each tow were held in water-filled containers aboard the survey vessel before processing. Since 1983, each fish larger than 20 cm was measured to the nearest mm in total length and its gender ascertained. Before 1983, at least 200 randomly selected winter flounder were measured during each week of sampling. Those fish not measured were classified into various length and gender groupings; at minimum, all winter flounder examined were classified as smaller or larger than 15 cm. The gender and reproductive condition of larger winter flounder was determined by either observing eggs or milt or, as suggested by Smigielski (1975), noting the presence (males) or absence (females) of ctenii on left-side caudal peduncle scales. Before release, healthy fish larger than 15 cm (1977-82) or 20 cm (1983 and after) were marked in a specific location with a number or letter made by a brass brand cooled in liquid nitrogen. Marks and brand location were varied in a manner such that the year of marking would be apparent in future collections.

#### Larval winter flounder sampling

Winter flounder larvae entrained through the MNPS cooling-water system were sampled at the MNPS discharges (station EN, Fig. 3) since 1976. Collections usually alternated between the discharges of Units 1 and 2, depending upon plant operation and water flow. Larvae were collected in a 1.0 x 3.6-m plankton net of 333- $\mu$ m mesh deployed from a gantry system. Four General Oceanic (GO) Model 2030 flowmeters were positioned in the net mouth to account for horizontal and vertical flow variation; sample volume was determined by averaging the four volume estimates from the flowmeters. Starting in 1993, the net was deployed for 3 to 4 minutes (filtering about 200 m<sup>3</sup>), but this varied depending upon the number of circulating pumps in operation and tidal stage. In previous years, sampling time was longer and filtered about 400 m<sup>3</sup>. Sampling frequencies have varied since 1976 (NUSCO 1987). In 1993, samples were collected once per week during both day and night in February and June. During March through May, samples were taken on three days and nights per week. This was one day and one night sample less per week than during the same months from 1983 through

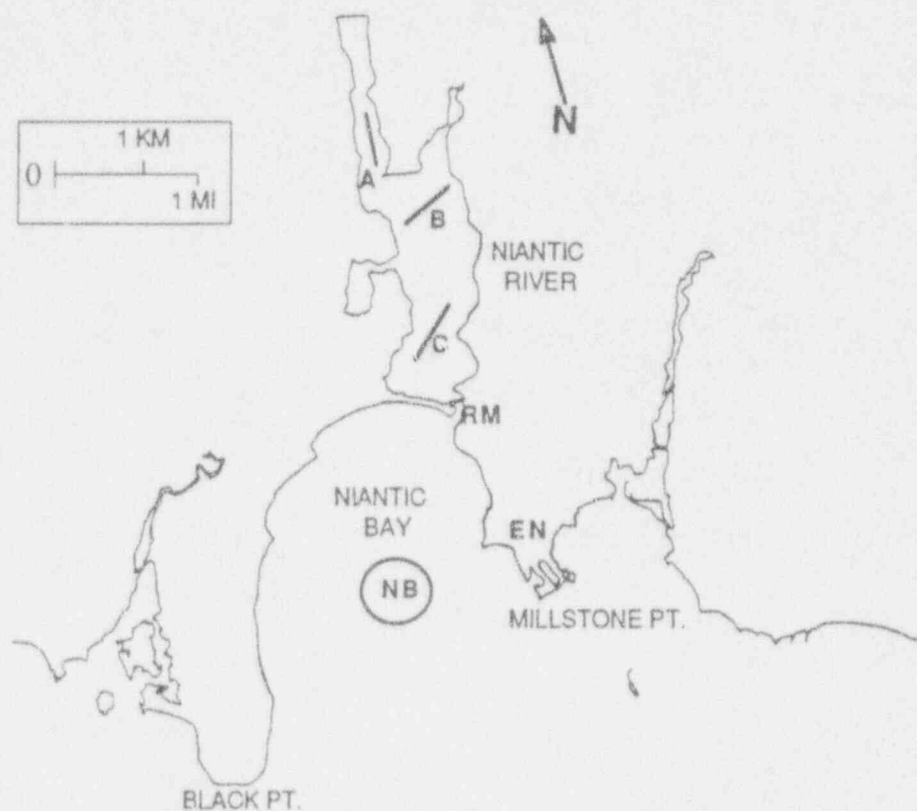


Fig. 3. Location of stations sampled for larval winter flounder during 1993.

1992. All ichthyoplankton samples, including those described below, were preserved with 10% formalin.

Winter flounder larvae have been collected in Niantic Bay at station NB since 1979 and in the Niantic River at stations A, B, and C since 1983 (Fig. 3). A 60-cm bongo plankton sampler was weighted with a 28.2-kg oceanographic depressor and fitted with 3.3-m long nets with mesh size of 202  $\mu\text{m}$  during February and March and 333  $\mu\text{m}$  during the remainder of the season. Volume of water filtered was determined using a single GO flowmeter mounted in the center of each bongo opening. The sampler was towed at approximately 2 knots using a stepwise oblique tow pattern, with equal sampling time at surface, mid-depth, and near bottom. The length of tow line necessary to sample the mid-water and bottom strata was determined by water depth and tow-line angle measured with an inclinometer. Nets were towed for 6 minutes (filtering about 120  $\text{m}^3$ ). One duplicate sample from the bongo sampler was retained for laboratory processing. The larval winter flounder sampling schedule for Niantic River and Bay was

based on knowledge gained during previous years and was designed to increase data collection efficiency while minimizing sampling biases (NUSCO 1987). Larval sampling at the three Niantic River stations usually started in mid-February. From then through the end of March, daytime tows were conducted within 1 hour of low slack tide. During the remainder of the season, until the disappearance of larvae at each station, tows were made at night during the second half of a flood tide. From 1983 through 1990, sampling was conducted 2 days a week. Starting in 1991, sampling was reduced to 1 day a week (NUSCO 1991a). Through 1992, station NB was sampled day and night every two weeks during February and at least once a week from March through the end of the larval winter flounder season. Beginning in 1993, station NB was sampled during the day from the start of the larval season through March and at night from April through the remainder of the larval season. In 1991-93, an additional station (RM) was sampled from March through May at the mouth of the Niantic River (Fig. 3) and collections at stations NB and RM

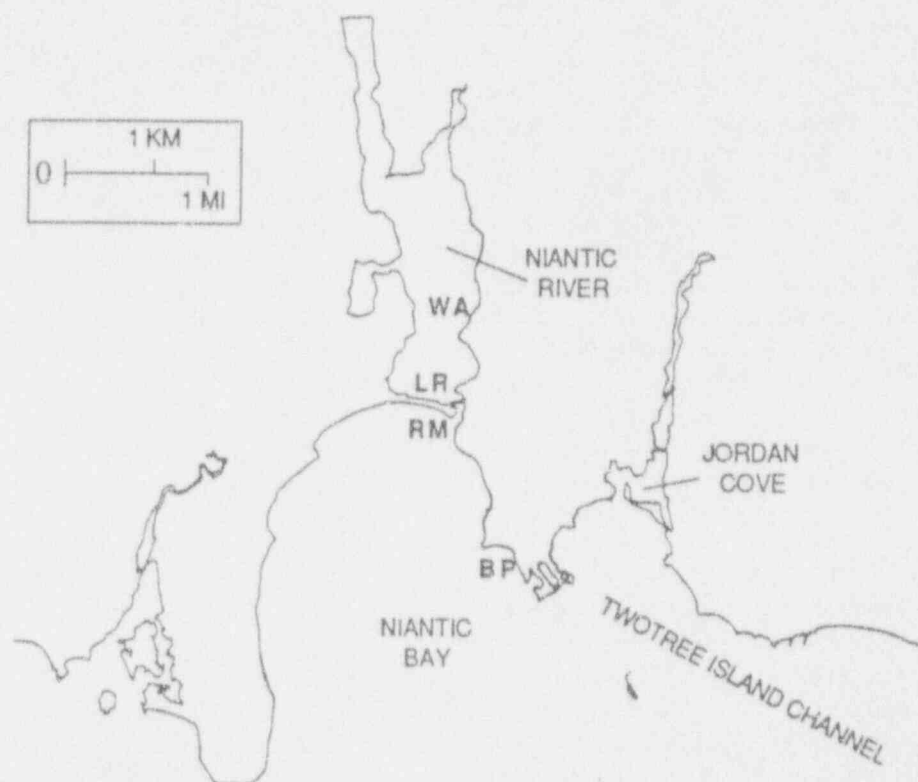


Fig. 4. Location of stations sampled biweekly by 1-m beam trawl for age-0 winter flounder from late May through September of 1993.

were taken consecutively near the time of maximum flood tidal currents. Jellyfish medusae at the three river stations were removed (1-cm mesh sieve) from the samples and measured volumetrically to the nearest 100 mL.

#### *Juvenile winter flounder sampling*

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

larval young in summer and during the subsequent fall and winter by the TMP. These fish became age-1 when taken during the February-April adult spawning surveys.

The abundance of post-larval age-0 winter flounder has been monitored at stations in the Niantic River since 1983 (LR) or late 1984 (WA), and in Niantic Bay (RM and BP) since 1988 (Fig. 4). Through 1992, collections were made weekly, but in 1993 sampling frequency was reduced to biweekly. River stations were sampled during daylight from about 2 hours before to 1 hour after high tide. Depending upon the time of high tide in relation to sunrise or sunset, stations located in the bay were sampled before or after those in the river, and, thus, were occupied at various tidal stages. During most years, sampling began during the third week of May. Monitoring continued through the end of September, unless no or few ( $< 1\text{-m}^{-2}$ ) young were taken during 2 consecutive sam-



pling periods, in which case the station was dropped.

Young winter flounder were sampled using a 1-m beam trawl having two tickler chains and with interchangeable nets of 0.8-, 1.6-, 3.2-, and 6.4-mm bar mesh. In 1983, triplicate tows were made at LR using nets of increasing larger mesh as the season progressed. Beginning in 1984, two nets of successively larger mesh were used during each sampling trip; nets were deployed in a random order. A change to the next larger mesh in the four-net sequence was made when fish had grown enough to become retained by it. Use of larger meshes also reduced the amount of detritus and algae collected. At each station, four replicate tows were made, two each with the two nets in use. Rarely, because of bad weather or damage to the net, only three tows were taken at a station. Tow distance was estimated by letting out a measured line attached to a lead weight as the net was hauled at about 25 m·min<sup>-1</sup>. 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.

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

### Indices of abundance

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

### *Relative annual abundance of adults*

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

Duration of tows varied and was usually longer in the lower river than in the upper river because of differences in tidal currents and amounts of extraneous material collected in the trawl. To lessen error in the calculation of CPUE, data from exceptionally long or brief tows made prior to 1983 were excluded from the analyses. Catches of winter flounder larger than 15 cm in tows made throughout the spawning surveys were standardized to either 15-minute tows at stations 1 and 2 or 12-minute tows at all other stations. The minimum length of 15 cm used for CPUE calculation was smaller than the 20 cm used for mark and recapture estimates described below because of data limitations from the 1977-82 surveys. Effort was standardized within each year by replicating the median CPUE estimate in a given week as needed so that effort (number of tows) was the same for each week sampled. A 95% confidence interval (CI) was calculated for each annual median CPUE using a distribution-free method based on order statistics (Snedecor and Cochran 1967).

A second relative index of abundance used the gender and size distribution of the fish from adult spawning survey catches standardized by variable weekly and yearly effort. Adjustments to the catches were made using sampling effort to insure that each size and sex



group of fish was given equal weight within each week of work, among weeks in each survey year, and to adjust for varying effort among years. Detailed methods of calculating these values were given in NUSCO (1989). To avoid confusion with the CPUE index, this measure is referred to as "annual standardized catch" throughout the remainder of this report. The annual standardized catch was the basis for the calculation of annual recruitment and egg production described below.

### *Absolute abundance estimates of adults*

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

### *Adult spawning stock size and egg production*

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

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

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

Absolute estimates of spawning females and corresponding annual egg production estimates for 1977 through 1993 were determined by assuming that the relative values represented 3.5% of the absolute values (see Absolute abundance estimates in Results and Discussion for how this fraction was determined). Annual estimates of the number of female spawners were also used in the derivation of a relationship between stock and recruitment for Niantic River winter flounder.

### *Development and growth, abundance, and mortality of larvae*

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

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

Larval data analyses were based on standardized densities (number/500m<sup>3</sup> of water sampled). A geometric mean of weekly densities was used in analyses because the data generally followed a lognormal distribution (McConnaughey and Conquest 1993) and weekly sampling frequencies varied among some stations. Because older larvae apparently remained near the bottom during the day and were not as susceptible to entrainment or the bongo sampler, data

from daylight samples collected after March at stations EN and NB were excluded from abundance calculations, except for estimating entrainment at MNPS.

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

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

where  $C_t$  = cumulative density at time  $t$

$t$  = time in days after February 15

$\alpha$  = total or asymptotic cumulative density

$p$  = inflection point scaled in days since February 15

$\kappa$  = shape parameter

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

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

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

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

Larval mortality rates were estimated from data collected at the three Niantic River stations; data from 1983 were excluded as smaller larvae were under-sampled because of net extrusion (NUSCO 1987). The abundance of 3-mm and smaller larvae was used to calculate an index of newly-hatched larvae because 3

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

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

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

where  $L$  = annual larval abundance of larvae 7-mm and larger at EN as estimated by  $\alpha$  (see Eq. 2)

$E$  = annual estimate of egg production in the Niantic River

$a$  = intercept

$b$  = slope or index of mortality dependence upon annual egg abundance

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

Regression analyses were used to examine possible relationships between variables and, at times, to make predictions. Ordinary least-squares linear regression was used when the independent variable was assumed to be measured without error (e.g., water temperature).

The test of a relationship was based on the slope being significantly ( $p \leq 0.05$ ) different from zero. Functional regression methods developed by Ricker (1973, 1984) were used in the cases where the independent variable was measured with error (e.g., abundance indices). For functional regressions, the probability that  $r$  (correlation coefficient) was significantly ( $p \leq 0.05$ ) different from zero was the criterion used to decide whether a valid relationship existed prior to determining the slope and the 95% confidence interval for the slope.

### *Abundance, growth, and mortality of juveniles in summer*

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

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

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

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

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

### *Abundance of juveniles during fall and winter*

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

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

### *Stock and recruitment relationship*

A stock-recruitment relationship (SRR) described by Ricker (1954, 1975) is the basis of the life-cycle algorithm that drives the population dynamics simulation model of Niantic River winter flounder. Application of this SRR to MNPS winter flounder stock assessment was described in detail in NUSCO (1989, 1990). The stock and recruitment data for determining the SRR were derived from the catch-at-age of female winter flounder during the Niantic River spawning survey. Because the spawning stock is made up of many year-classes, the true recruitment



consists of the total reproductive contribution over the life of each individual in a given year-class (Garrod and Jones 1974; Cushing and Horwood 1977). Therefore, the index of annual parental stock size was based on derived egg production and the index of recruits or year-class size was based on calculated egg production accumulated over the life-time of the recruits. This method accounted for variations in year-class strength and in fecundity by size and age. The assumptions and methods used to age Niantic River winter flounder and to calculate a recruitment index expressed as equivalent numbers of female spawners were described in detail in NUSCO (1989, 1990) and summarized below.

**Stock and recruitment indices.** Methods used to calculate the annual standardized catch index and total egg production of the parental stock were given previously. The recruitment index was determined by applying an age-length key described in NUSCO (1989) to the annual standardized catches of females partitioned into length categories. A common age-length key was used over all years because Witherell and Burnett (1993) reported that no trends were observed in mean length-at-age during 1983-91 for Massachusetts winter flounder despite a 50% reduction in biomass over that period. Aging the females allowed for the determination of their numbers by year-class present at ages 3, 4, 5, and 6+ during successive spawning seasons. The age-6+ group was further subdivided into the numbers of fish expected to survive to a terminal age of 15 by assuming various annual instantaneous mortality rates as fishing pressure increased from the 1970s into the 1990s. To follow each year-class from 1977 through 1989 to its terminal age (e.g., 2004 for the 1989 year-class), values of  $Z (= F + M)$  were used that represented estimates of current and anticipated annual instantaneous rate of fishing ( $F$ ) as provided by the Connecticut Department of Environmental Protection (CT DEP). The instantaneous natural mortality rate ( $M$ ) was assumed constant at 0.35 over all years. These were the same mortality rates used in the stochastic population dynamics model, discussed below. From observations made of abundance and age over the years, a large fraction of age-3 females, considerable numbers of age-4 fish, and even some age-5 females were apparently immature and not present in the Niantic River during the spawning season (NUSCO 1989). Thus, the total number of females was reduced to spawning females using length-specific proportions of mature fish estimated from annual catches in the

Niantic River for fish age-3 to 5; all females age-6 and older were assumed to be mature. Because the estimates of age-3 fish were thought to be unreliable, this estimation process was only carried through the 1989 year-class (age-4 females taken in 1993). The adjusted numbers of mature fish provided an index of the fully recruited year-class expressed as the aggregated number of female spawners passing through each age-class. An implied assumption was that catches in the Niantic River were representative of the population, with the exception of immature fish that did not enter the river until fully recruited. Although this recruitment index could be used together with the annual number of female spawners to derive an SRR, this would ignore size composition differences that affected annual egg production. Therefore, the above index was adjusted for differences in fecundity among fish using the length-fecundity relationship for Niantic River winter flounder given above (Eq. 1). Finally, annual egg production was summed up over the life-time of each year-class to determine the recruitment index as eggs and, then, converted to equivalent female spawners at the rate of one female spawner for each 561,000 eggs (i.e., the mean fecundity).

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

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

where  $R_t$  is the recruitment index for the progeny of the winter flounder spawning stock  $P_t$  in year  $t$  and  $\alpha$  and  $\beta$  are parameters estimated from the data. The  $\alpha$  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  $\beta$  parameter is the instantaneous rate at which recruitment declines at large stock sizes due to some form of density-dependent mortality. The natural logarithm of winter flounder recruitment was found correlated with mean water temperature during February at the intakes of MNPS, which is when most spawning and early larval development occurs (NUSCO 1988a, 1989). Therefore, the parameters  $\alpha$  and  $\beta$  were estimated initially by fitting Equation 6 to

the data and then re-estimated under the assumption that there was a significant temperature effect; this was accomplished by adding a temperature-effect component to Equation 6. Following Lorda and Crecco (1987) and Gibson (1987), annual mean water temperatures for a particular period were used as an explanatory variable to adjust the two-parameter SRR for temperature effects, which served to reduce recruitment variability and obtain more reliable parameter estimates for the SRR. The temperature-dependent SRR had the form:

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

where the second exponential describes the effect of February water temperature on recruitment and the new parameter  $\phi$  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 (1977-89) average of February water temperatures. When the February mean water temperature is equal to the long-term average, the deviation ( $T_{Feb}$ ) in Equation 7 becomes zero and the exponential term equals unity (i.e., no temperature effect). Thus, Equation 7 reduces to its initial form (Eq. 6) under average temperature conditions. Nonlinear regression methods (SAS Institute Inc. 1985) were used for estimating the parameters in the above equations.

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

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

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

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

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

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

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

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

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

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

### Assessment of MNPS operation on Niantic River winter flounder

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



found when applying deterministic models (i.e., assuming steady-state conditions) to fish stocks whose exploitation rates are not stable, especially when such stocks increase in abundance, as in the case of the winter flounder during the late 1970s and early 1980s (Smith et al. 1989). Environmental variability also results in year-to-year variation of natural mortality rates, which further weakens the results of deterministic assessments.

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

#### *Estimates of larval entrainment at MNPS*

The estimated number of larvae entrained in the MNPS condenser cooling water system each year is a

direct measure of impact on the local winter flounder stock. Annual estimates were determined using larval densities at station EN (Fig. 3) and the volume of cooling water used by MNPS. The Gompertz density function (Eq. 3) was fitted to larval data and daily densities (number/500m<sup>3</sup>) were calculated. Daily entrainment estimates were determined after adjusting for the daily condenser cooling-water volume and an annual estimate was determined by summing all daily estimates during the larval season.

#### *Mass-balance calculations*

The number of winter flounder larvae entrained depends upon larval densities in Niantic Bay. Potential impact to the Niantic River stock from larval entrainment should be related to the number of larvae in Niantic Bay that originated from the river. Mass-balance calculations were used to investigate whether the number of winter flounder larvae entering Niantic Bay from the Niantic River could sustain the number of larvae observed in the bay during the winter flounder larval season each year (1984-93). Three potential larval inputs to Niantic Bay include eggs hatching in the bay, larvae flushed from the Niantic River, and larvae entering the bay from LIS across the boundary between Millstone Point and Black Point (Fig. 3). The few yolk-sac larvae collected annually in Niantic Bay suggested that minimal spawning and subsequent hatching occurred in the bay, which was therefore considered a negligible source of larvae. Larvae were known to be flushed from the river into the bay and this input to the bay was estimated from available data. The number of larvae entering Niantic Bay from LIS was unknown. Four ways in which larvae may leave Niantic Bay include natural mortality, entering the Niantic River during a flood tide, being entrained at MNPS, and flushing from the bay into LIS. Estimates could be made for the number of larvae lost through natural mortality, entering the Niantic River, and entrained at MNPS, but little was known about the number of larvae flushed into LIS. The numbers of larvae flushed to and from LIS were combined as the unknown (*Source or Sink*) in the mass-balance calculations. Thus, the form of the mass-balance equation was:

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

where  $t$  = time in days

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

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

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

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

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

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

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

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

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

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

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

Thus:

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

Daily abundance estimates were derived from the Gompertz density equation (Eq. 3) and the daily densities for Niantic Bay at two points in time ( $NB_t$  and  $NB_{t+5}$ ) for each 5-day period were calculated from data collected at stations NB and EN combined. These densities, adjusted for the volume of Niantic Bay (about  $50 \times 10^6\ m^3$ ; 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  $5\text{-day change}$  in Equation 15. The selection of 5 days as the period of change was arbitrary and a cursory examination of results based on 10-day periods showed that the same conclusions were reached with either 5- or 10-day periods.

Daily entrainment estimates were based on data collected at station EN and the actual daily volume of condenser cooling water used at MNPS. The daily entrainment estimates were summed over each 5-day period ( $Ent$ ). Annual stage-specific mortality rates for 1984-89 were determined by Crecco and Howell (1990), for 1990 (V. Crecco, DEP Division of Marine Fisheries, Old Lyme, CT, pers. comm.), and for 1991

through 1993 by NUSCO staff. Mortality was partitioned among developmental stages by comparing the rates of decline of predominant size-classes for each stage. Each developmental stage was assigned a portion of the total annual larval mortality rate ( $Z$ ); similar mortality rates were assumed for Stages 3 and 4. Although estimating stage-specific mortality in this manner was not precise, sensitivity analysis on the mass-balance calculations (NUSCO 1991b) indicated that larval mortality was the least sensitive parameter. These annual rates were modified to daily stage-specific mortality rates by assuming 10-day stage durations for Stages 1, 3, and 4 larvae, and 20 days for Stage 2 larvae. The proportion of each stage collected at station EN during each 5-day period was applied to the daily standing stock for Niantic Bay ( $NB_t$ ) to estimate the number of larvae in each developmental stage for stage-specific mortality calculations. The daily loss due to natural mortality was summed for each 5-day period ( $Mort$ ).

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

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

The 95% confidence interval for the slope was 0.387 - 0.579. The estimated average density, the average tidal prism of  $2.7 \times 10^6\ m^3$  (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 for  $ToNR$  (NUSCO 1992a). In 1992 and 1993, sampling again was conducted at RM during a flood tide, but the collections were made by

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

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

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

### *Stochastic simulation of winter flounder stock dynamics*

**Modeling strategy and background.** The stochastic population dynamics model (SPDM) developed for the Niantic River winter flounder stock was based on the Ricker SRR fitted to the data, even though Equation 7 does not explicitly appear in the model formulation. The mechanisms underlying the Ricker form of recruitment are incorporated in the set of equations that the model uses to calculate mortality through the first year of life. Beyond that point (i.e.,

age-1) in the life-cycle simulation, the population model simply describes the annual reduction of each year-class through natural mortality and fishing, together with aging and reproduction. This process occurs at the beginning of each model time-step of length equal to 1 year. The projection of adult fish populations over time has been implemented in many models by means of Leslie matrix equations (e.g., Hess et al. 1975; Vaughan 1981; Spaulding et al. 1983; Reed et al. 1984; Goodyear and Christensen 1984). In the SPDM, adult winter flounder were projected over time by grouping fish into distinct age-classes and by carrying out the computations needed (mostly additions and multiplications) iteratively over the age index (1 through 15) and over the number of years specified for each simulation. This approach was algebraically identical to the Leslie matrix formulation, facilitated the understanding of how the model works, and simplified the computer code when describing the fish population either as biomass (allowing for size variation within each age-class) or numbers of fish. A similar implementation of an adult fish population dynamics simulation was used by Crecco and Savoy (1987) in their model of Connecticut River American shad (*Alosa sapidissima*).

**Model components.** Figure 5 illustrates the components of the computer program used for the SPDM. Components depicted by solid-line boxes constitute the model presently in use and one depicted by a box with dashed lines illustrates a part of the model that was not used in the present application, but could be used in future applications. The functionality of most model components should be clear from the flow chart and no further details will be provided. Some critical components, such as the one labeled age-1 cohort and the two random input boxes are described below. A list of the actual input data used in the application of the model to the Niantic River winter flounder stock is also given.

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



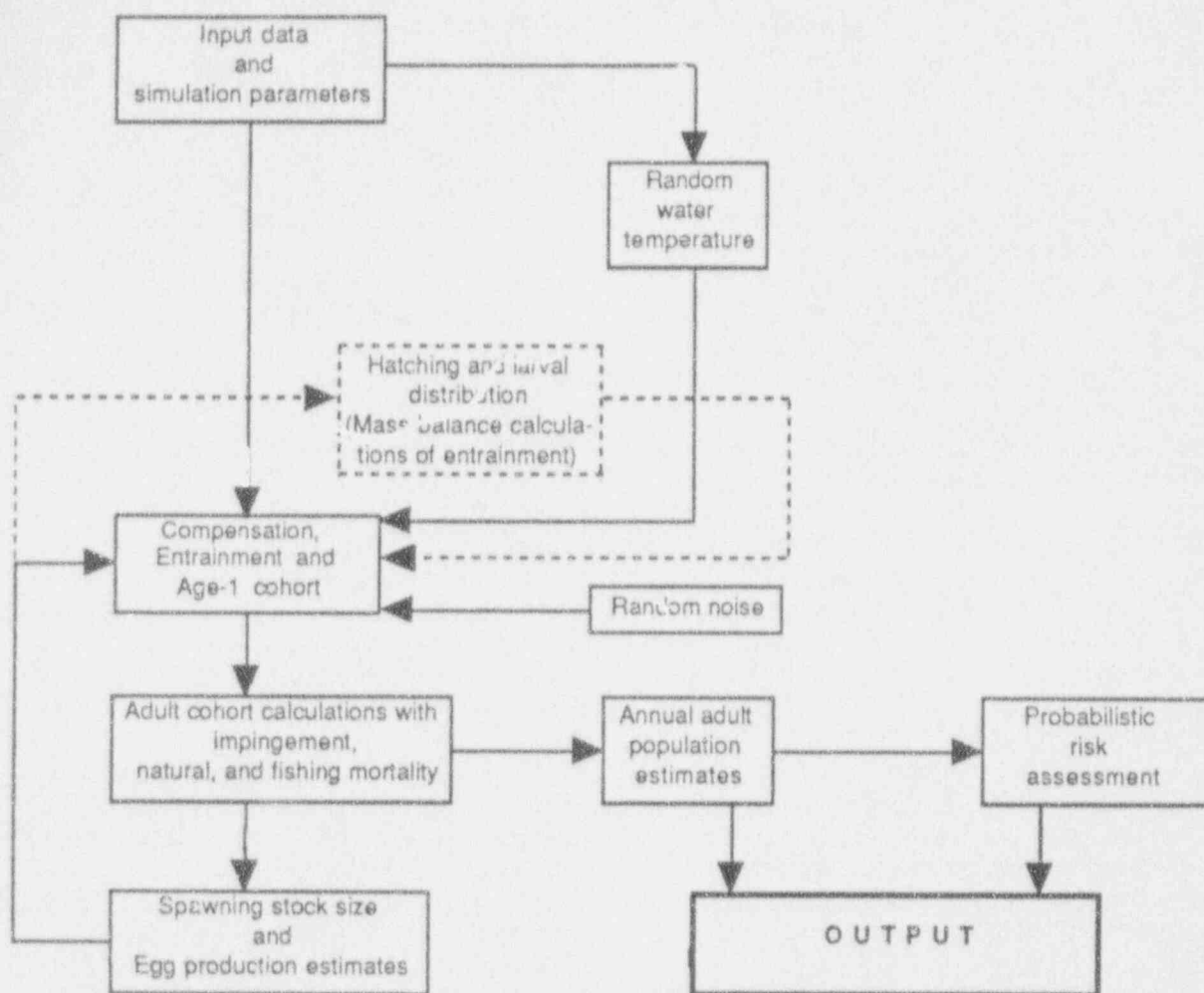


Fig. 5. Diagram of NUSCO stochastic population dynamics computer model for assessing the long-term effect of larval winter flounder entrainment at MNPS. Dashed boxes and arrows refer to components and calculations which are not an integral part of the model.

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

where the subscript  $t$  denotes the time-step (each time-step represents a year) and non-subscripted terms remain constant from year to year;  $\alpha$ ,  $\beta$ , and  $\phi$  are the parameters of the SR function (Eq. 7); FEC is the mean fecundity of the stock expressed as the number of female eggs produced per female spawner; ASF is a scaling factor to adjust  $\alpha$  for the effect of a multi-age spawning stock;  $n_t$  and  $\text{WT}_t$  are independent random variates from two specified normal distributions described below;  $Z_{1,2}$  is the instantaneous mortality through the immature age-classes; and the last term ( $\beta \cdot P_t$ ) is the feed-back mechanism simulating stock-dependent compensatory mortality, which varies

according to the size of the annual spawning stock  $P_t$ . The complete derivation of the above equation was given in NUSCO (1990: appendix to the winter flounder section). The scaling factor ASF is a multiplier that converts age-3 female recruits into the total spawning potential of the year-class. This spawning potential is defined as the cumulative number of mature females from the same year-class that survive to spawn year after year during the lifetime of the fish. The algebraic form of this multiplier is identical to the numerator of Equation A-4 in Christensen and Goodyear (1988).

Stochasticity in the winter flounder model (Fig. 5) has two annual components: a random term that represents uncertainties associated with the estimate of

Ricker's  $\alpha$  parameter and annual environmental variability in the form of random deviations from the long-term mean February water temperature. These two components of annual variability are incorporated into the calculation of each new year-class via the mortality from egg to age-1 (Eq. 18). The term  $n_t$  in Equation 18 (random noise) is simulated as independent random variates from a normal distribution with zero mean and variance equal to  $\sigma^2$ . The value of  $\sigma$  is estimated during the model calibration runs as the amount of variance required to generate  $\alpha$  values within the 95% confidence interval of the estimate of  $\alpha$  used in the model (NUSCO 1990). Similarly, the term  $\phi \cdot WT_t$  in Equation 18 represents the effect of annual environmental variability of February water temperatures on larval survival. This effect becomes random when the February water temperatures are themselves simulated as independent random variates from a normal distribution with mean and variance equal to the mean and variance of February water temperatures at the MNPS intakes from 1977 through 1989.

The stochastic simulation of fish population dynamics provides a framework for probabilistic risk assessment methodology. This type of assessment is based on Monte Carlo methods (Rubinstein 1981), where many independent random replicates of the time-series are generated so that the mean of the series and its standard error can be estimated. Monte Carlo replications can be used to derive the sample distribution function (Stuart and Ord 1987) without assuming a known statistical distribution. This methodology was used to assess the risk of stock reduction resulting from the effects of entrainment and impingement at MNPS. The probabilities of stock reductions were empirically derived from Monte Carlo replicates of the time-series of impacted stocks.

**SPDM assumptions and limitations.** Major assumptions relate to the underlying form of the SRR used and the reliability of the SRR parameter estimates. Because the SPDM incorporated the

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

**Model input data.** The dynamics of the Niantic River winter flounder stock were simulated using the SPDM under a credible real-time scenario running from 1960, well before operation of Unit 1, to 2060, long after the projected shutdown date for Unit 3 in 2025 (Table 1). The scenarios used power plant effects based on actual generating units in operation each year, concurrently with estimates of  $F$  that were based on historic and projected rates of commercial exploitation and sport fishing for winter flounder in Connecticut. Parameters used in the SPDM included:  $F$ , with some additional mortality equivalent to losses from impingement (IMP); conditional mortality rates (i.e., fraction of the annual production of winter

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

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



flounder removed as a result of power plant operation) determined for larval entrainment (ENT); a schedule of changes when any of these rates was not assumed constant; and the length of the time-series in years. The combined mortality of  $F + IMP$  was used only during the simulation period (1971-2025) that corresponded to MNPS operation (Table 1).

Because the ability of a fish stock to withstand additional stress is reduced by fishing mortality (Good-year 1980), simulations of the long-term entrainment of winter flounder larvae also included effects due to the substantial exploitation of the stock. The annual schedule of nominal fishing rates was determined from

recent DEP estimates (V. Crecco, C. DEP, Old Lyme, CT, pers. comm.). These exploitation rates took into account length-limit regulations in effect from 1982-93 and from changes in regulations proposed by the DEP to reduce fishing mortality in Connecticut waters (Tables 2 and 3). Vulnerability factors for age-classes 1 through 5+ were calculated for the commercial fishery (60% of the total winter flounder catch) that were based on: actual or proposed changes in length limits and minimum commercial trawl fishery codend sizes; the size-at-age of female Niantic River winter flounder at mid-year (age + 0.5) determined using the von Bertalanffy growth equation

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

Period	Length limit in inches		Length limit in mm		Seasonal closure
	Commercial fishery	Sport fishery	Commercial fishery	Sport fishery	
1982 <sup>a</sup>	8	8	203	203	None
1983 (Jan-May)	8	8	203	203	None
1983 (Jun-Dec)	11	8	279	203	None
1984 (Jan-Aug)	11	8	279	203	None
1984 (Sep-Dec)	10	8	254	203	None
1985-1986	10	10	254	254	None
1987 (Jan-Aug)	10	10	254	254	Dec 1 - Mar 31 (within Niantic River)
1987 (Sep-Dec)	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
1988-1993 <sup>b</sup>	11	10	279	254	Dec 1 - Mar 31 (within Niantic River)
≥ 1994 <sup>c</sup>	12	11	305	279	Nov 14 - Apr 15 (in all state waters)

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

<sup>b</sup> Minimum trawl mesh codend size also increased from 4.5 to 5.0 inches.

<sup>c</sup> At the time of preparation of this report, these changes were proposed by the DEP (P. Howell, CT DEP, Old Lyme, CT, pers. comm.) for implementation (also includes an increase in the minimum trawl mesh codend size to 5.5 inches).

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

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

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

<sup>b</sup> The notation 5+ refers to fish that are age-5 and older.

<sup>c</sup> Based on regulations proposed for implementation by the DEP at the time of preparation of this report.

(NUSCO 1987); selection curves for 114-mm (4.5-in), 127-mm (5-in), and 140-mm (5.5-in) trawl mesh codends provided by the DEP; and a discard mortality rate of 50% for undersized fish. The sport fishery was estimated to take 40% of the total catch, having a discard mortality rate of 15%. The values of  $F$  used in the simulations were stepped up from 0.40 in the 1960s to a peak of 1.30 in 1991 (Fig. 6), which reflected the recent historical increase in fishing and the current high exploitation of winter flounder. The value of  $F$  was subsequently reduced to meet a targeted rate of 0.50 by 2001. Although the Atlantic States Marine Fisheries Commission management plan for inshore stocks of winter flounder (Howell et al. 1992) calls for a further reduction in  $F$  to about 0.43, the perhaps more realistically attainable value of 0.50 was used for all remaining years after consulting with DEP staff (V. Crecco and P. Howell, DEP Division of Marine Fisheries, Old Lyme, CT, pers. comm.). The effect of the changing fishing rates on partially vulnerable fish is seen in Figure 7. As a result of more

protective regulations, the effect of commercial fishing on ages-1 and 2 has been or will be greatly diminished and many age-3 and 4 fish should be protected as well. The derivation of the equivalent mortality rate IMP was given in NUSCO (1992a) and is an additional small (0.01) component of mortality added to  $F$  during the years of MNPS operation. Other data, rates, and inputs to the SPDM are summarized on Table 4 and include the number of age-classes, age-specific rates of maturation, natural mortality, average weight and fecundity at age, the three-parameter SRR estimates, February water temperature statistics, and other specific factors for each simulation.

Conditional mortality rates for larval entrainment (ENT) from 1984 through 1993 used in SPDM simulations under actual operating conditions were estimated directly using the mass-balance calculations described above. Values of ENT determined for other years were varied stochastically. An annual value of ENT was chosen from the range of values determined from the mass-balance calculations for full MNPS three-unit

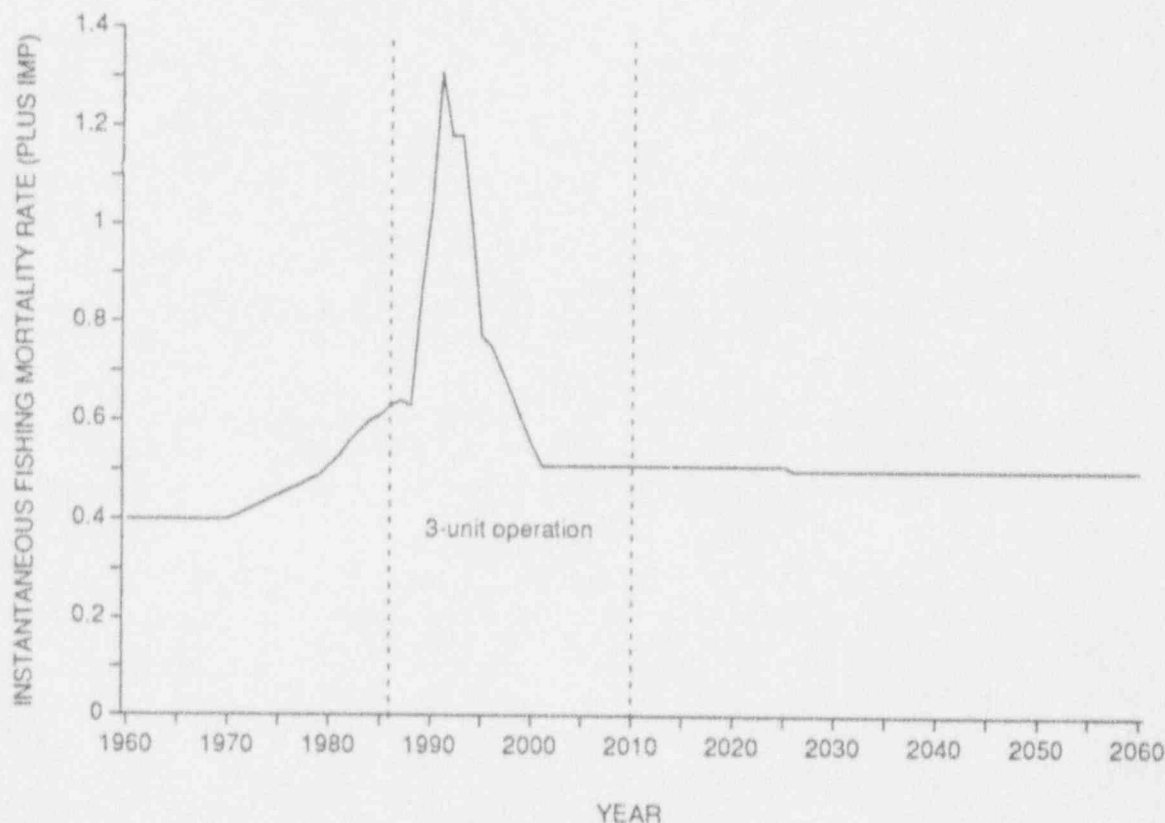


Fig. 6. Historic and projected annual mortality rate due to fishing ( $F$ ), as determined in consultation with the CT DEP, plus a small (0.01) component accounting for impingement mortality (IMP) at MNPS as implemented in the SPDM simulations.

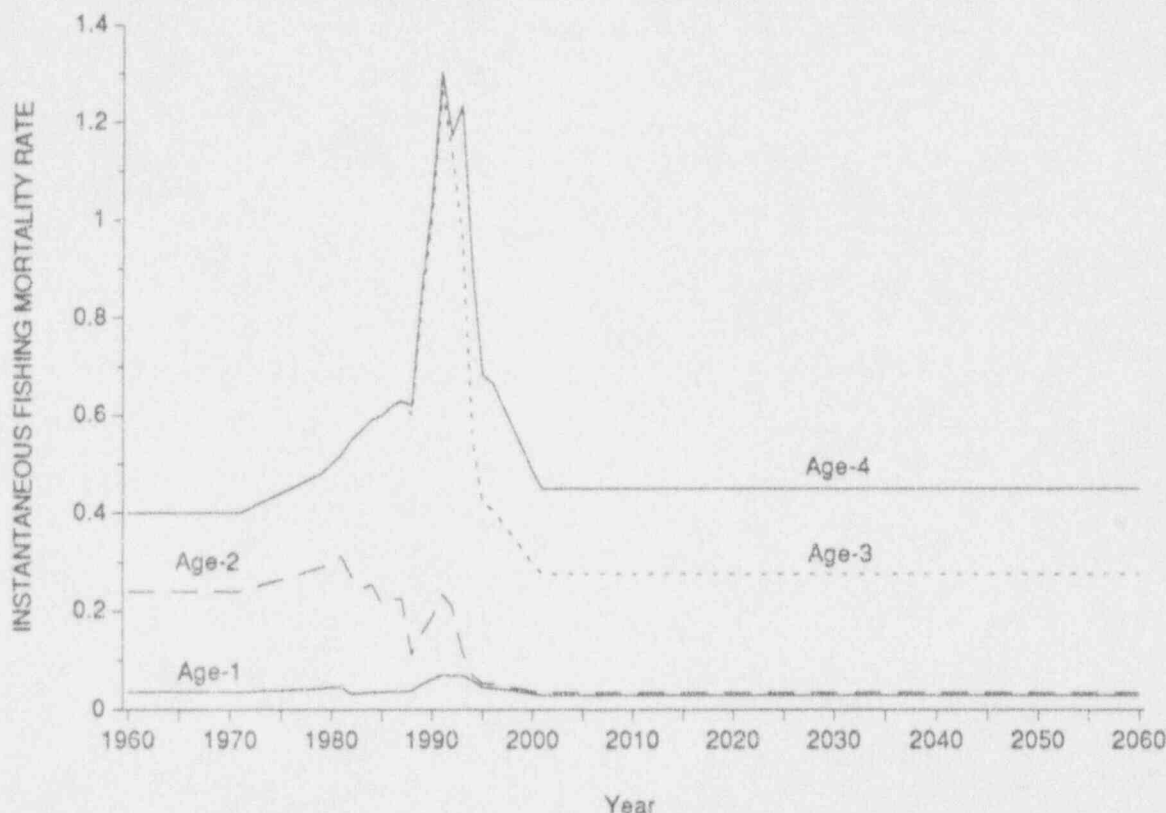


Fig. 7. Estimated reductions in  $F$  (including discard mortality) for age-1 through 4 winter flounder as a result of actual or planned regulations imposed by the CT DEP on the winter flounder commercial and sport fisheries.

operation. This was done by resampling with replacement using uniform probabilities to randomize the process. These estimates were calculated under the assumption that all three units used cooling water pumped at maximum capacity ( $11.1 \times 10^6 \text{ m}^3 \cdot \text{day}^{-1}$ ). The selected value of ENT was then scaled by both the number of units in operation in a particular year (Table 1) and the fractions of cooling-water flow actually used during the annual March-May larval winter flounder season (Table 5). MNPS cooling-water use was known for 1976 through 1983 and actual flow values were used to scale the randomly selected value of ENT. Because no data were available during 1971-75 for Unit 1, flow values for these years were estimated from net electrical generation records. Estimates for 1972 and 1975, years in which the unit apparently operated near maximum capacity, were normalized to the value for 1987, the year of maximum flow for the Unit 1 time-series; other years were scaled accordingly. Since the simulation time-series extended to 2060 (including a recovery period following the end of MNPS operation), historic cooling-

water flow rates calculated for 1971-93 were re-used to predict entrainment for 1994-2025 by resampling the historic flows with replacement using uniform probabilities to randomize the process. This approach assumed that the existing 23-year record of MNPS operation adequately described the operational variability expected at the station in the future. Except for those cases where randomly chosen values for a year had all three units operating near 100% capacity, annual values of ENT used in the simulations were less than the theoretical maximum under full three-unit operation.

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

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

Model input	Value used or available		
Number of age-classes in population	15		
Earliest age at which all females are mature	6		
Fraction mature, mean wt (lbs), and mean fecundity by age:			
Age-1 females	0	0.011	0
Age-2 females	0	0.125	0
Age-3 females	0.08	0.554	223,735
Age-4 females	0.36	0.811	378,584
Age-5 females	0.92	1.089	568,243
Age-6 females	1.00	1.377	785,897
Age-7 females	1.00	1.645	1,004,776
Age-8 females	1.00	1.873	1,201,125
Age-9 females	1.00	2.057	1,366,951
Age-10 females	1.00	2.203	1,502,557
Age-11 females	1.00	2.304	1,598,597
Age-12 females	1.00	2.390	1,682,208
Age-13 females	1.00	2.461	1,754,800
Age-14 females	1.00	2.516	1,805,000
Age-15 females	1.00	2.552	1,845,800
Age after which annual mortality is constant	3		
Instantaneous mortality rates M and F at age-1	0.50	0 <sup>a</sup>	
Instantaneous mortality rates M and F at age-2	0.35	0	
Instantaneous mortality rates M and F at age-3+	0.35	0	
Initial number of female spawners	66,988 <sup>b</sup>		
Mean fecundity of the stock (eggs per female spawner)	871,000 <sup>b</sup>		
$\alpha$ from the three-parameter SRR for the virgin (F = 0) stock (numbers of fish)	5.42 <sup>c</sup>		
$\beta$ from the three-parameter SRR	2.523 X 10 <sup>-5</sup>		
$\phi$ from the three-parameter SRR	0.412		
Mean February (1977-89) water temperature (°C)	2.48		
standard deviation	1.09		
minimum temperature	0.36		
maximum temperature	4.02		
Number of spawning cycles (years) to simulate	100		
Number of simulation replicates per run	100		
Fraction of age-0 group entrained at MNPS (i.e., impact)	0.00 <sup>d</sup>		

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

<sup>b</sup> Corresponds to the unfished stock at equilibrium (see Table 32 in Results and Discussion).

<sup>c</sup> Indirectly calculated from life history parameters (see Stock-recruitment relationship in Results and Discussion).

<sup>d</sup> A zero simulates a non-impacted stock; otherwise the conditional mortality due to entrainment is used.

previously concluded that 100 replicates were sufficient, given the amount of variability present in SPDM simulations (NUSCO 1990). Thus, the Monte Carlo sample size was set to 100 and the

geometric mean of the replicates was computed. All stock projections are given in units of spawning biomass (lbs) because overfishing criteria often rely on measurements of biomass and assessments based



TABLE 5. Annual average cooling-water flow and percent of nominal maximum flow at MNPS Units 1 through 3 during the March-May winter flounder larval entrainment season from 1971 through 1993.

	Unit 1		Unit 2		Unit 3	
Nominal flow at 100% capacity:	29.18 m <sup>3</sup> ·sec <sup>-1</sup>		37.62 m <sup>3</sup> ·sec <sup>-1</sup>		61.91 m <sup>3</sup> ·sec <sup>-1</sup>	
Fraction of total MNPS flow:	0.227		0.292		0.481	
Year <sup>a</sup>	March-May average flow in m <sup>3</sup> ·sec <sup>-1</sup>	% of nominal maximum	March-May average flow in m <sup>3</sup> ·sec <sup>-1</sup>	% of nominal maximum	March-May average flow in m <sup>3</sup> ·sec <sup>-1</sup>	% of nominal maximum
1971	-	67.41	-	-	-	-
1972	-	99.64	-	-	-	-
1973	-	33.81	-	-	-	-
1974	-	83.50	-	-	-	-
1975	-	99.64	-	-	-	-
1976	25.39	90.80	29.16	80.83	-	-
1977	27.61	98.73	24.61	68.20	-	-
1978	17.48	62.53	18.91	52.41	-	-
1979	17.18	61.44	21.48	59.53	-	-
1980	27.60	98.70	31.75	88.01	-	-
1981	1.52	5.43	33.98	94.18	-	-
1982	27.60	98.70	32.33	89.51	-	-
1983	26.79	95.83	30.90	85.63	-	-
1984	13.88	49.61	35.83	99.31	-	-
1985	27.86	99.64	16.40	45.45	-	-
1986	27.21	93.25	36.89	98.07	49.82	80.48
1987	29.01	99.40	36.99	98.32	47.12	76.12
1988	28.14	98.81	32.83	87.27	55.58	89.78
1989	13.85	47.46	24.72	65.72	51.33	82.91
1990	27.55	94.39	33.28	88.48	48.71	78.68
1991	10.79	36.98	32.29	85.83	38.65	62.44
1992	25.11	86.06	28.50	75.75	51.10	82.55
1993	27.78	95.21	33.52	89.10	58.82	95.00

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

on biomass tend to be more conservative than those based on fish numbers. Furthermore, larval entrainment effects result in long-term stock reductions which can be quite different depending on whether the stock is expressed as fish numbers or as biomass.

A complete simulation of MNPS impact consisted of three model runs, which provided a set of time-series generated under the same scenario, but with different combinations of F (plus IMP) and ENT. These model runs were designed to simulate the natural variability of the theoretical unfished stock (i.e., with no fishing or plant operational effects); the reduced stock biomass when subjected to fishing mortality (i.e., the baseline time-series without MNPS effects); and the expected biomass when all

three types of anthropogenic mortality (F, IMP, and ENT) occurred. The first time-series with no fishing or plant effects was the reference series against which the potential for recruitment failure was evaluated when the largest reductions of stock biomass occurred during any of the other simulations. The second time-series represented the most likely trajectory of the exploited stock without MNPS operation. The third time-series was the expected stock trajectory when the conditional mortality rates corresponding to ENT and IMP were added to the fishing mortality simulated for the baseline. This last time-series was the basis for quantitatively assessing MNPS impact on the Niantic River winter flounder population.



## Results and Discussion

### Seawater temperature

On the basis of the coefficient of variation (CV), monthly mean seawater temperatures recorded at MNPS were most variable from January through March (CV = 32-47%; Table 6). Winter flounder spawning and early larval development occur during these months. Temperatures were most stable (CV = 4-6%) during summer, when collections of winter flounder were dominated by young and other immature fish. The annual mean temperature for 1993 was 11.69°C, slightly warmer than the overall average of 11.50°C since 1976. Mean water temperature during winter (3.79°C; Table 7) was the coldest since 1989, reflecting episodes of inclement weather discussed below in conjunction with the adult spawning surveys. However, mean temperatures during spring (11.03°C) and summer (19.91°C), the periods for late larval development, metamorphosis, and development of young, were warmer than the respective 17-year averages of 10.89°C and 19.61°C.

### Adult winter flounder

#### *Relative annual abundance*

The 1993 adult winter flounder survey in the Niantic River began on February 16 (Table 8). Ice conditions and storms in 1993 were among the worst of the 18-year history of this program and resulted in reduced sampling during 3 of the survey weeks. Only 21 tows were completed during the second week of sampling (February 22-26), 3 during the third week (March 1-5), and 25 during the fifth week (March 15-19); the weekly average for the other 5 weeks of sampling was 48 tows per week. The total of 288 tows in 1993 was the smallest since 277 were completed in 1987 (Table 9). As found during the past few years, most adults were concentrated in several areas, including the northern section of the upper river arm and in Keeny Cove (Fig. 2). Fewer females spawned earlier in 1993 compared to recent years, when proportionately more females were spent, probably because of the colder weather this year. Nevertheless, spawning was completed by mid-March as the percentage of egg-bearing females larger than 26 cm declined from about one-third of the females examined during the first few weeks of sampling to about 5%

TABLE 6. Monthly and annual mean seawater temperature (°C) from January 1976 through December 1993 as calculated from mean daily water temperatures recorded continuously at the intakes of MNPS Units 1 and 2.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual mean
1976	3.65	3.31	4.81	7.55	10.75	15.11	18.29	19.60	18.93	15.04	9.28	4.73	10.90
1977	0.67	0.36	2.85	5.66	10.72	14.92	19.08	20.33	19.41	15.58	12.18	6.72	10.84
1978	3.01	1.09	1.67	4.85	9.10	14.24	17.68	19.82	19.24	16.14	12.47	7.74	10.64
1979	4.53	1.48	3.35	5.93	10.50	15.57	18.84	20.91	20.05	15.99	12.41	8.60	11.57
1980	5.16	2.38	2.80	6.38	10.44	14.76	18.44	20.23	20.16	16.07	10.25	5.73	11.10
1981	1.06	2.63	3.36	6.40	10.19	15.48	19.51	20.86	19.94	14.75	11.07	6.29	11.01
1982	2.20	1.56	3.04	5.41	10.06	14.16	17.98	21.10	20.01	15.95	12.47	8.97	11.13
1983	5.58	3.74	4.55	7.07	10.50	15.05	19.10	19.17	20.57	17.37	12.57	7.90	11.98
1984	4.84	4.02	3.98	6.58	10.84	15.53	18.90	20.60	19.52	16.41	13.04	9.07	11.97
1985	4.56	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
Overall mean	3.99	2.85	3.86	6.63	10.72	15.24	18.76	20.40	19.67	16.06	11.84	7.45	11.50
CV (%)	47	46	32	20	15	10	6	4	5	10	15	27	5

TABLE 7. Seasonal<sup>a</sup> mean seawater temperature (°C) for 1976 through 1993 as calculated from mean daily water temperatures determined by continuous recorders at the intakes of MNPS Units 1 and 2.

Year	Winter	Spring	Summer	Fall
1976	3.94	11.14	18.94	9.69
1977	1.32	10.72	19.61	11.49
1978	1.95	9.40	18.91	12.11
1979	3.17	10.67	19.93	12.33
1980	3.47	10.53	19.61	10.69
1981	2.34	10.69	20.11	10.70
1982	2.29	9.88	19.69	12.46
1983	4.65	10.87	19.61	12.61
1984	4.29	10.99	19.68	12.84
1985	3.67	10.98	20.22	12.85
1986	4.06	11.52	19.43	12.38
1987	4.40	11.56	19.66	11.32
1988	3.28	10.79	19.16	11.17
1989	3.82	10.68	19.72	10.97
1990	4.28	10.83	19.89	13.16
1991	5.38	12.32	20.00	12.48
1992	4.45	10.98	19.08	11.19
1993	3.79	11.03	19.91	11.85
Overall mean	3.59	10.89	19.61	11.79
CV (%)	44	35	6	34

<sup>a</sup> Winter is January through March, spring is April through June, summer is July through September, and fall is October through December.

TABLE 8. Annual Niantic River winter flounder<sup>a</sup> population surveys during the spawning season from 1976 through 1993.

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 <sup>b</sup>
1990	February 20 - April 4	7
1991	February 13 - March 20	6
1992	February 18 - March 31	7
1993	February 16 - April 7	8 <sup>c</sup>

<sup>a</sup> Minimum size for marking was 15 cm during 1976-82 and 20 cm thereafter.

<sup>b</sup> Limited sampling during week 2 because of ice formation.

<sup>c</sup> Almost no sampling during week 3 and limited sampling during weeks 2 and 5 because of ice and weather conditions.

by the end of March and early April (Fig. 8). As a result, the spawning survey ended on April 7. The median CPUE in 1993 for winter flounder larger than 15 cm was 1.9 (Table 9; Fig. 9). This value is only about 30% of the CPUE of 6.2 for 1992, which had been the smallest CPUE on record. In fact, in 17 tows during 1993 (about 6% of the total), no winter flounder of any size were taken. The winter flounder taken during 1993 were, on the average, larger than those collected during the past 3 years (Fig. 10), or during any previous survey. Most fish taken during 1993 were larger than 32 cm and, thus, more vulnerable to continued high rates of fishing. The peaks seen in the annual length-frequency distributions from 1990 through 1993 were probably fish from the relatively strong 1988 year-class, since recruitment has been poor in more recent years. This can be better illustrated by comparing the annual standardized catch of females for 1990 through 1993, which shows relative abundance from year to year by size (Fig. 11). The decline in winter flounder abundance during most recent years, particularly among fish smaller than 32 cm, was even more pronounced when catches from 1993 were compared to those for 1982 (near historical peak abundance for this series), 1985 (after winter

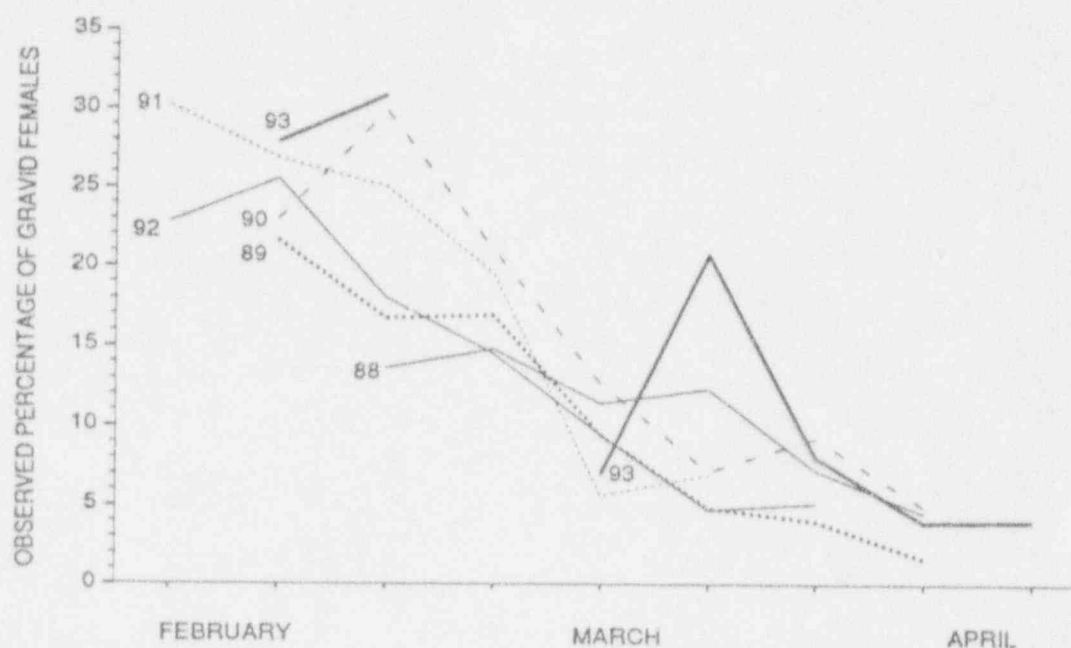


Fig. 8. Weekly percentage of Niantic River female winter flounder larger than 26 cm that were gravid during the 1987 through 1993 adult population abundance surveys. Data from weeks in 1993 during which few or no tows were taken were not included.

TABLE 9. Annual 9.1-m otter trawl adjusted median CPUE<sup>a</sup> of winter flounder larger than 15 cm<sup>b</sup> taken throughout the Niantic River during the 1976 through 1993 adult population abundance surveys.

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

<sup>a</sup> Catch per standardized tow (see Materials and Methods).

<sup>b</sup> Mostly age-2 and older fish.

<sup>c</sup> Only tows of standard time or distance were considered and effort equalized among weeks.

<sup>d</sup> Zero for symmetrically distributed data.

<sup>e</sup> Because of low effort, data from the third week of sampling not used for the computation of CPUE.

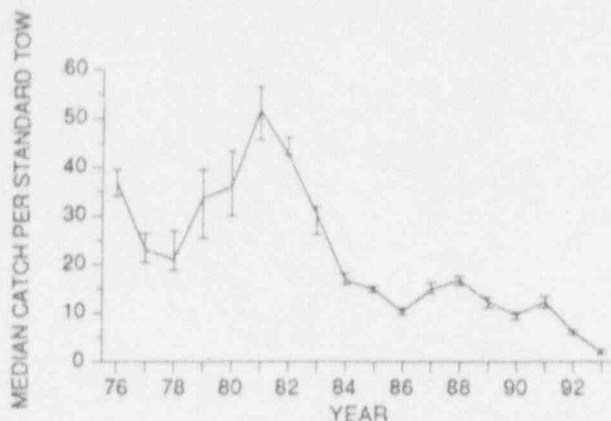


Fig. 9. Annual median CPUE ( $\pm 2$  standard errors) for Niantic River winter flounder larger than 15 cm from 1976 through 1993.

flounder had decreased from peaks observed during the early 1980s), and 1990 (Fig. 12). NMFS (1993) reported that combined sport and commercial fishery landings for the Southern New England-Middle Atlantic stocks of winter flounder declined 35% from 1991 to 1992 and were at a 14-year low. NMFS research

trawl survey indices also indicated very low levels of abundance in recent years.

#### *Absolute abundance estimates*

Mark and recapture data were used with the Jolly (1965) model to determine absolute abundance estimates of fish larger than 20 cm ( $N$ ); estimates of survival ( $\Phi$ ), recruitment ( $B$ ), and sampling intensity ( $p$ ) were also generated. Because of the decline in population size, only 972 winter flounder 20 cm and larger were marked with a freeze brand this year, which was well below the total branded in any other year since 1983 (Table 10). The total of 154 previously marked fish that were recaptured in 1993, however, was not disproportionately low. Most ( $n = 109$ ; 71%) of the recaptured fish had been marked in 1992. Addition of recapture data from 1993 resulted in an increase for the 1991 estimate of  $N = 59,165$  reported in NUSCO (1993) to 62,743 (Table 11). The initial abundance estimate for 1992 of about 12,178 winter flounder was only 19% of the 1991 population estimate, although this value will likely increase

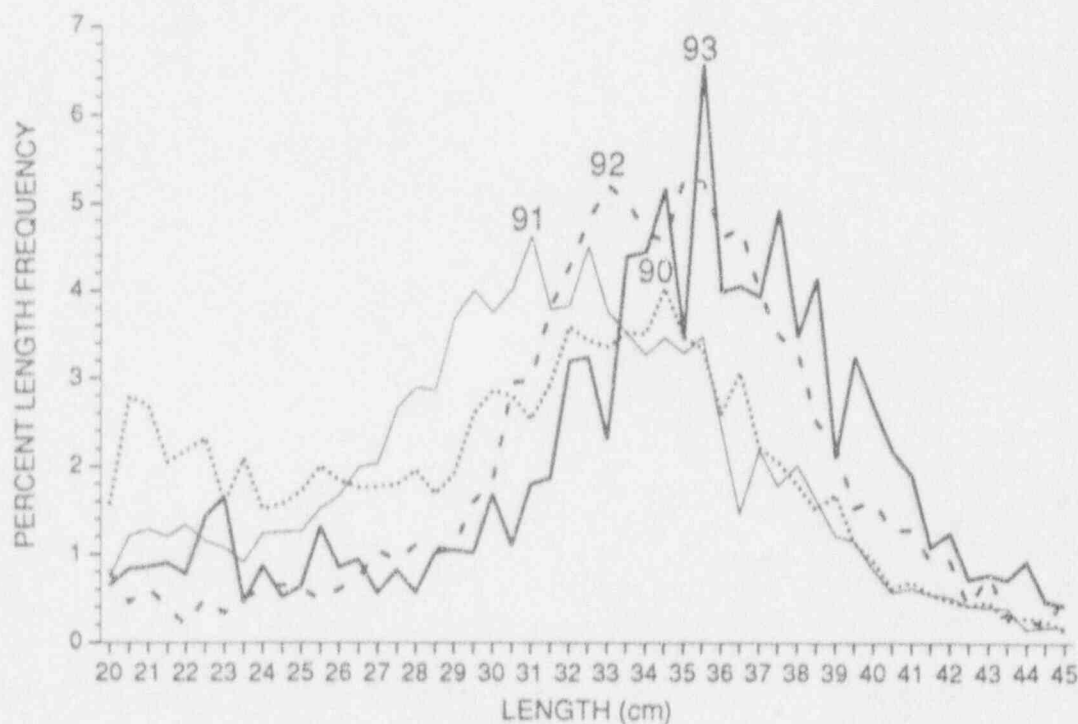


Fig. 10. Comparison of percent length-frequency distributions of winter flounder 20 cm and larger taken in the Niantic River during the spawning season from 1990 through 1993.

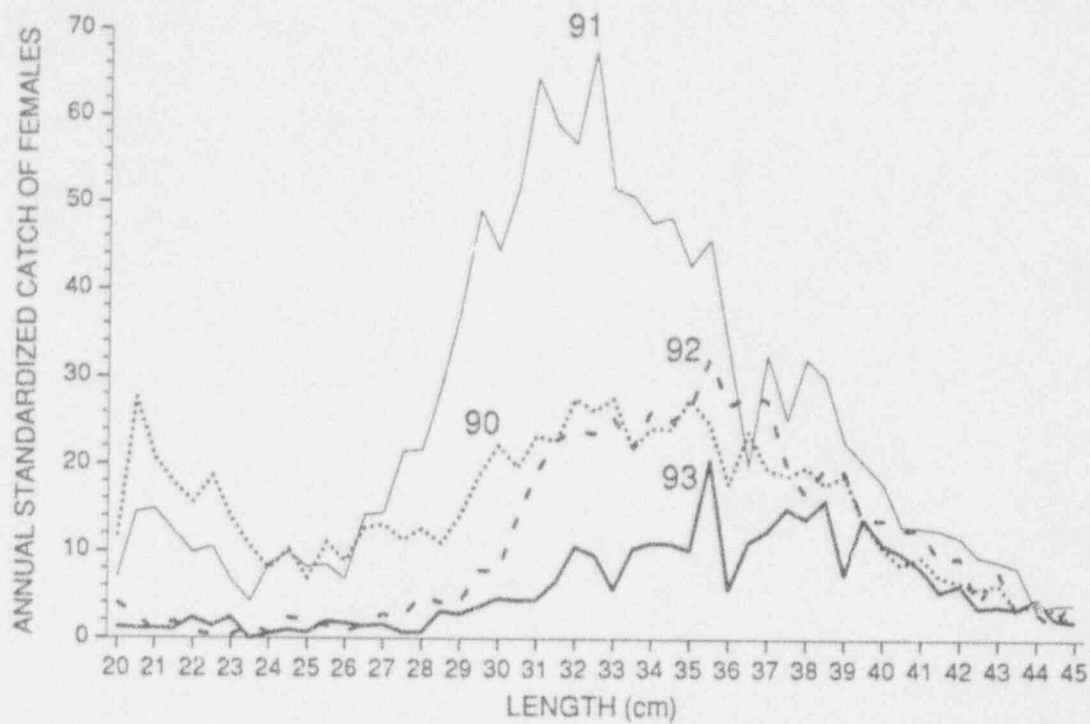


Fig. 11. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season from 1990 through 1993.

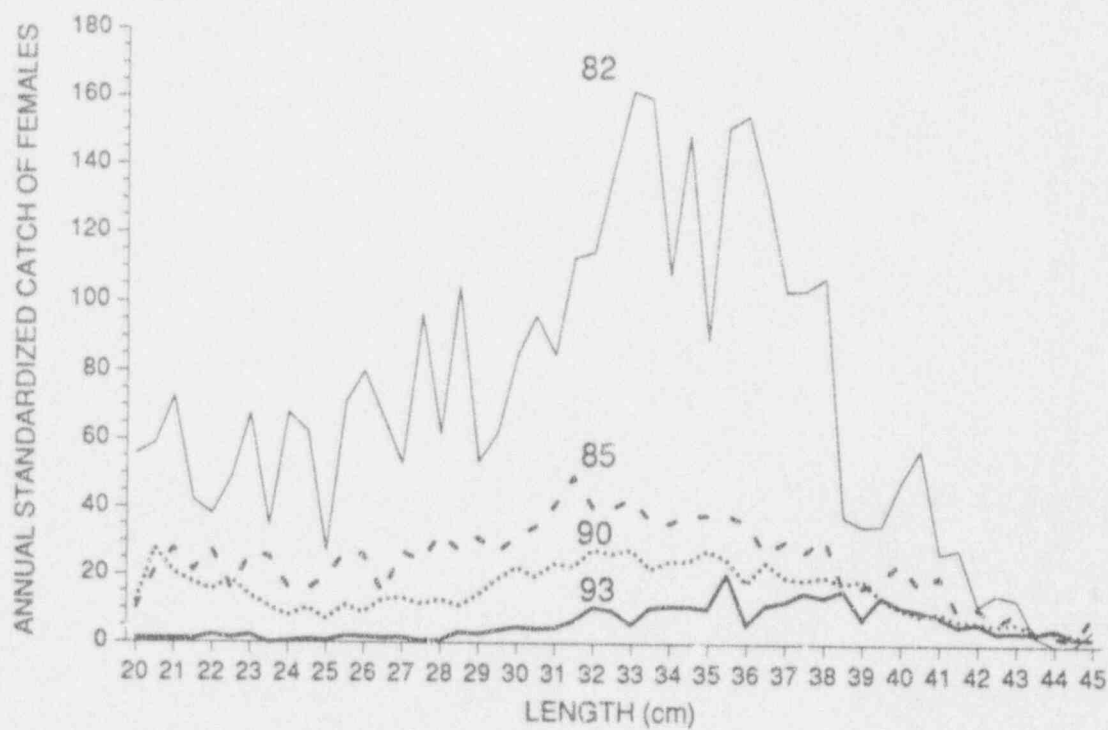


Fig. 12. Comparison of annual standardized catch by length of female winter flounder 20 cm and larger taken in the Niantic River during the spawning season in 1982, 1985, 1990, and 1993.



TABLE 10. Mark and recapture data from 1983 through 1993 used for estimating abundance of winter flounder larger than 20 cm in the Niantic River during the spawning season.

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

TABLE 11. Estimated abundance\* of winter flounder larger than 20 cm taken during the spawning season in the Niantic River from 1984 through 1992 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 ( $\Phi$ )	Standard error of $\Phi$	95% CI for $\Phi$
1983				0.328	0.040	0.251 - 0.405
1984	57,706	8,370	41,300 - 74,112	0.558	0.065	0.430 - 0.686
1985	79,607	10,851	58,338 - 100,876	0.361	0.041	0.281 - 0.442
1986	49,287	6,229	37,078 - 61,497	0.650	0.067	0.518 - 0.781
1987	75,592	9,746	56,490 - 94,695	0.596	0.062	0.474 - 0.719
1988	66,607	7,247	52,404 - 80,011	0.462	0.049	0.366 - 0.558
1989	42,534	4,865	32,998 - 52,069	0.390	0.042	0.308 - 0.472
1990	33,243	3,861	25,675 - 40,810	0.863	0.105	0.657 - 1.068
1991	62,743	7,872	47,314 - 78,171	0.149	0.023	0.104 - 0.194
1992	12,178	1,844	8,565 - 15,791			
Mean	53,277	2,730	48,515 - 58,040	0.484	0.013	0.459 - 0.509

Year	Sampling intensity (p)	Standard error of p	95% CI for p	Annual recruitment (B)	Standard error of B	95% CI for B
1984	0.071	0.0103	0.050 - 0.091	47,428	9,083	29,626 - 65,231
1985	0.044	0.0060	0.032 - 0.055	20,550	5,226	10,307 - 30,794
1986	0.061	0.0078	0.046 - 0.076	43,585	8,464	26,995 - 60,176
1987	0.034	0.0044	0.025 - 0.043	21,525	6,371	9,038 - 34,011
1988	0.065	0.0071	0.051 - 0.079	11,762	3,739	4,434 - 19,091
1989	0.066	0.0076	0.051 - 0.081	16,695	3,114	10,592 - 22,798
1990	0.069	0.0081	0.053 - 0.085	34,087	5,772	22,774 - 45,400
1991	0.099	0.0087	0.052 - 0.086	2,839	1,123	638 - 5,040
1992	0.192	0.0292	0.135 - 0.249			
Mean	0.074	0.0040	0.066 - 0.082	24,809	1,175	22,507 - 27,111

\* Estimates may vary from those reported in NUSCO (1993) because of mark and recapture data added from the 1993 adult winter flounder population survey.

somewhat as additional marked fish are recaptured in future surveys. The standard errors of  $N$  given in Table 11 are correlated with  $N$  because of the particular form of Jolly's variance formula. Therefore, the 95% confidence intervals computed are generally considered unreliable as a measure of sampling error, except at very high sampling intensities (Manly 1971; Roff 1973; Pollock et al. 1990). Sampling intensity ( $p$ ), or the probability that a fish will be captured, was relatively high (0.192) during 1992 in comparison to other years (0.034 - 0.071). This was probably the result of intensified sampling on a smaller population of fish that were concentrated in relatively few areas of the Niantic River, which may have introduced some bias into the estimates if interannual catchability changed. Sampling intensities of about 0.10 are recommended to obtain reliable and precise estimates of population size and survival rates with the Jolly model (Bishop and Sheppard 1973; Nichols et al. 1981). Hightower and Gilbert (1984) found that low sampling effort may give acceptable estimates if population size is relatively large and the number of marked animals is also relatively high. However, Gilbert (1973) and Carothers (1973) reported that  $N$  was underestimated and had low accuracy when sampling intensities were low (5-9%), regardless of population size or number of fish marked. Loss of marks because tags were not observed, or from mortality, also requires increased sampling effort. Other sampling errors, model assumptions, and biases inherent in the Jolly model that could have affected these estimates are discussed in NUSCO (1989) and Pollock et al. (1990). Nevertheless, when set to a similar scale, CPUE and Jolly abundance estimates appeared to track each other well (Fig. 13), even though the latter may be subject to considerable error. The two abundance indices are significantly correlated (Pearson correlation coefficient = 0.85;  $p = 0.004$ ). Thus, based on a median CPUE of 1.9, absolute abundance of winter flounder in 1993 was likely less than 10,000 fish. In contrast, by extrapolation the abundance in 1981 could have been greater than 200,000 winter flounder.

The estimate for survival ( $\Phi$ ) of fish from 1991 to 1992 was particularly low (0.149; Table 11), as was the estimate of recruitment ( $B = 2.839$ ). Although reflecting apparent trends for the Niantic River population of winter flounder, estimates of these parameters are considered to be less reliable than those of abundance when using the Jolly model (Bishop and Sheppard 1973; Arnason and Mills 1981; Hightower

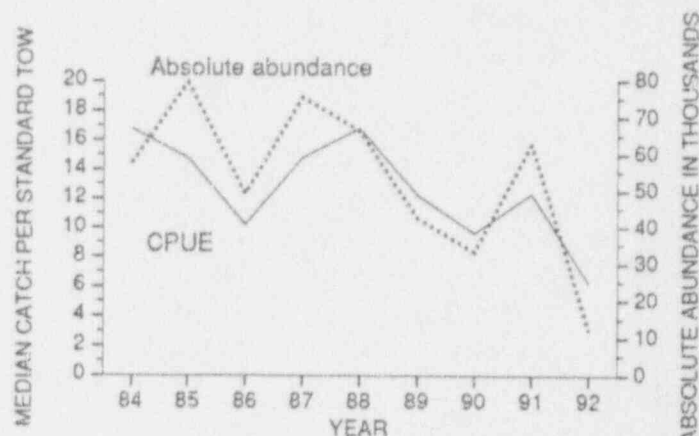


Fig. 13. Comparison between the estimates of absolute abundance in thousands of winter flounder larger than 20 cm in the Niantic River during the spawning season (dashed line) and the corresponding median CPUE (solid line) from 1984 through 1992.

and Gilbert 1984). If the estimate of  $\Phi$  was correct, it would imply an annual fishing rate equal to about 1.5, assuming a natural mortality rate of 0.35. As for other values based on only 1 year of recapture information, estimates of  $\Phi$  and  $B$  may change considerably with the addition of data from the next winter flounder survey. Nevertheless, the low estimates of  $B$  appeared to accurately reflect low recruitment corresponding to weak year-classes that have been produced in recent years.

Because of the reasonable correspondence between the median trawl CPUE index and the Jolly abundance estimate, the annual standardized catches of all fish larger than 20 cm for 1984-92 were compared to the total abundance estimates from the Jolly model. In previous reports, the relative numbers of females and eggs produced each year were assumed to represent, conservatively, about 3.5% of the absolute values (range of 2.7 - 4.5%) and a multiplier of 28.571 (the ratio of 100 to 3.5) was used to scale abundance indices to absolute numbers. The 1992 annual standardized catch index was 8.4% of the corresponding total population estimate, increasing the geometric mean of the series to 3.8%. Because of uncertainties associated with the latest population estimate, the 3.5% value continued to be used in scaling absolute abundance indices. This adjustment also assumed that the ratios of annual standardized catch to absolute abundance for 1977 through 1983 would have been similar to those for 1984-92, had estimates of absolute abundance been available for the earlier period.

### Spawning stock size and egg production

The size of the Niantic River winter flounder female spawning stock is used in various assessments of MNPS impact. The annual standardized catch of female spawners (an index of spawning stock size) and the production of eggs were determined from available data on sex ratios, sexual maturity, and fish length-frequencies. The sex ratio of winter flounder larger than 20 cm during the 1993 spawning season in the Niantic River was 1.47 females for each male (Table 12), the largest ratio found since 1988 and somewhat higher than the 17-year average of 1.34. Ratios of 1.50 to 2.33 in favor of females have been reported by Saila (1962a, 1962b) and by Howe and Coates (1975) for other winter flounder populations in southern New England. Witherell and Burnett (1993) also reported greater proportions of female winter flounder in Massachusetts waters, particularly in older age-classes. They believed that males likely have a higher natural mortality rate, based on evidence of earlier ages of senescence reported for males by Burton and Idler (1984).

TABLE 12. Female to male sex ratios of winter flounder taken during the spawning season in the Niantic River from 1977 through 1993.

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
Geometric mean	1.33	1.34

The rate of spawning was determined by observing weekly changes in the percentage of gravid females larger than 26 cm, the size at which about half of all

observed females were mature (NUSCO 1988a). This is comparable to  $L_{50}$  estimates of 28.3 and 27.6 cm reported for Massachusetts waters by Witherell and Burnett (1993) and O'Brien et al. (1993), respectively. In recent years, most spawning in the Niantic River was completed by late March or early April as relatively few gravid females were found afterwards (Fig. 8). During most years, ice in the upper river prevented the start of field work in January or early February, so approximately two-thirds of the females examined during late February and early March had spawned before sampling began. Apparently, most females spawned relatively early in recent years, as few gravid fish were taken, particularly after mid-March. Spawning was likely correlated with water temperature. In relatively cold years (e.g., 1977 and 1978) proportionately fewer females spawned during the earlier portion of the survey, compared to warmer years (e.g., 1989 and 1992) when more fish were spent at the beginning of sampling.

For each year, the proportion of females considered to be mature for each 0.5-cm length increment was used with the annual standardized catch of females to obtain relative annual abundance indices for female spawners. Mature females comprised approximately one-third to one-half of each yearly total, with relative numbers of female spawners ranging from a low of 274 in 1993 to 2,752 in 1982 (Table 13). Varying sex ratios and differences in percent maturity due to changes in length-frequency distributions somewhat affected average fecundity, which was low during the late 1970s when smaller fish were more abundant, but increased during recent years because of increasing proportions of older and larger fish. The relative index of total egg production reflected female stock abundance and length distribution and was greatest from 1981 through 1983 because of peak population abundance and moderate average fecundity. Average fecundity estimates for 1992 and 1993 were the highest of all years, because of an older age structure and relatively poor recruitment from incoming year-classes that would have comprised the majority of the spawners.

Absolute estimates of spawning females and associated egg production were generated by multiplying corresponding relative numbers by 28,571 (see Absolute abundance estimates, above). Female stock size was between approximately 7,821 and 78,629 fish, while estimates of annual egg production ranged from about 6.4 to 45.6 billion (Table 13). The total number of female spawners was used as an estimate of parental stock size for the SRR, which is discussed

TABLE 13. Relative and absolute annual standardized catch of female winter flounder spawners and corresponding egg production in the Niantic River from 1977 through 1993.

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

<sup>a</sup> Based on proportion of the relative annual standardized catches of winter flounder that were mature females.

<sup>b</sup> As a proportion of all winter flounder 20 cm or larger.

<sup>c</sup> Total egg production divided by the number of spawning females.

<sup>d</sup> A relative index for year-to-year comparisons and not an absolute estimate of production.

<sup>e</sup> Calculated on the assumption that the relative annual standardized catches were approximately 3.5% of absolute values.

below. Egg production was greatest in the early 1980s, but estimates were relatively high in 1988 and 1989 as proportionally older and larger females dominated the moderately-sized reproductive stock. Egg production decreased to about 10.6 billion in 1990 because of a decline in female abundance and in their average size, increased by about 50% to 18.3 billion in 1991 as the number of spawners increased, but decreased again to 11.2 billion in 1992 and to a series low of 6.4 in 1993.

Comparatively little is known about the egg stage of winter flounder. Buckley et al. (1991) noted that female size and time of spawning affected various reproductive parameters, including egg size, fecundity, and viability. Embryos produced earlier in the season appeared to have a survival advantage, particularly over those from smaller fish late in the season. Egg deposition apparently takes place on gravel bars, algal mats, eelgrass beds, and near freshwater springs in Rhode Island salt ponds (Crawford 1990). Based on estimated egg production and abundance of Stage 1 larvae, egg mortality may be considerable. DeBlois and Leggett (1991) found that the amphipod *Callinectes laevisculus* preyed heavily upon demersal

capelin (*Mallotus villosus*) eggs, removing up to 39% of the production. They suggested that invertebrate predation on demersal fish eggs may be an important regulatory mechanism for population size in marine fishes having demersal eggs. Morrison et al. (1991) reported high mortality of demersal Atlantic herring (*Clupea harengus*) eggs in the Firth of Clyde, Scotland because of heavy deposition of organic matter resulting from a bloom of the diatom *Skeletonema costatum*. The decomposing material caused a depletion of oxygen and egg death due to anoxia. *Skeletonema costatum* was one of the most abundant of the phytoplankton collected at MNPS during entrainment sampling from 1977 through 1980 (NUSCO 1981). However, highest densities occurred in summer, after the winter flounder spawning season.

### Larval winter flounder

#### Abundance and distribution

The  $\alpha$  parameter of the Gompertz function (Eq. 2) was used as an index for temporal (year to year) and spatial (Niantic River and Bay) abundances of winter



flounder larvae. Based on the 95% confidence interval around the  $\alpha$  parameter estimate, larval abundance during 1993 in the river (stations A, B, and C combined) and the bay (stations EN and NB combined) was significantly lower than any other in the 11-year series (Table 14). In general, annual abundances in the bay varied less than in the river. In 1985, 1988, and 1989, larval abundance in the river was much greater than in the bay. Although 1993 had the lowest abundance in both the river and the bay, no consistent relationship was found between the annual abundances between the two areas (Spearman's rank-order correlation coefficient  $\rho = 0.455$ ;  $p = 0.160$ ) during the 11-year period. This lack of relationship may have at least two possible causes. First, if many of the larvae in the bay came from the river, then highly variable annual larval mortality rates occurred prior to the period when larvae were flushed from the river to the bay. Secondly, the Niantic River may not be the only source of larvae entering the bay and this possibility was discussed in detail in NUSCO (1992a, 1992b, 1993) and will be addressed again later in this section. Larval abundance in the bay since 1976 appeared to reflect regional-wide trends as annual abundance ( $\alpha$  parameter) at EN since 1976 was highly correlated (Spearman's rank-order correlation coefficient  $\rho = 0.651$ ;  $p = 0.003$ ) with annual abundance indices in Mount Hope Bay, MA and RI (Marine Research, Inc. 1992; M. Scherer, Marine Research, Inc., Falmouth, MA., pers comm.). However, no relationship was found between the abundances in the Niantic River (1984-93) and Mount Hope Bay (Spearman's rank-order correlation coefficient  $\rho = 0.091$ ;  $p = 0.803$ ).

Annual spatial abundances of the first four larval developmental stages were based on cumulative weekly geometric means (Figs. 14 and 15). The abundance distribution of Stage 5 was not examined because so few were collected. In instances where this function did not satisfactorily fit the data, cumulative density data (the running sum of the weekly geometric means) was used to compare abundances as a surrogate for the  $\alpha$  parameter from the Gompertz function. 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  $\alpha$  parameters were found to be highly correlated (Spearman's rank-order correlation coefficient  $\rho = 0.999$ ;  $p < 0.001$ ) in a previous study (NUSCO 1989), suggesting that cumulative weekly geometric means could be used as an alternative index of larval abundance in some cases.

Stage 1 abundance during 1993 in the river was one of the lowest during the 11-year period of sampling at all three stations. A comparison of annual Stage 1 abundance among years showed a similar relative ranking at the three stations, with 1988 and 1989 ranked the highest and 1983, 1986, and 1993 the lowest. Except for a slightly greater abundance at station A in some years, annual abundances at the three river stations have been similar. This indicated a somewhat homogeneous distribution of Stage 1 larvae throughout the river. Because winter flounder eggs are demersal and adhesive and the duration of Stage 1 is short (about 10 days), the homogenous distribution suggested that spawning was not restricted to any specific area of the river or, conversely, that the river is well-mixed. Low abundance in 1983 was partly

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

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



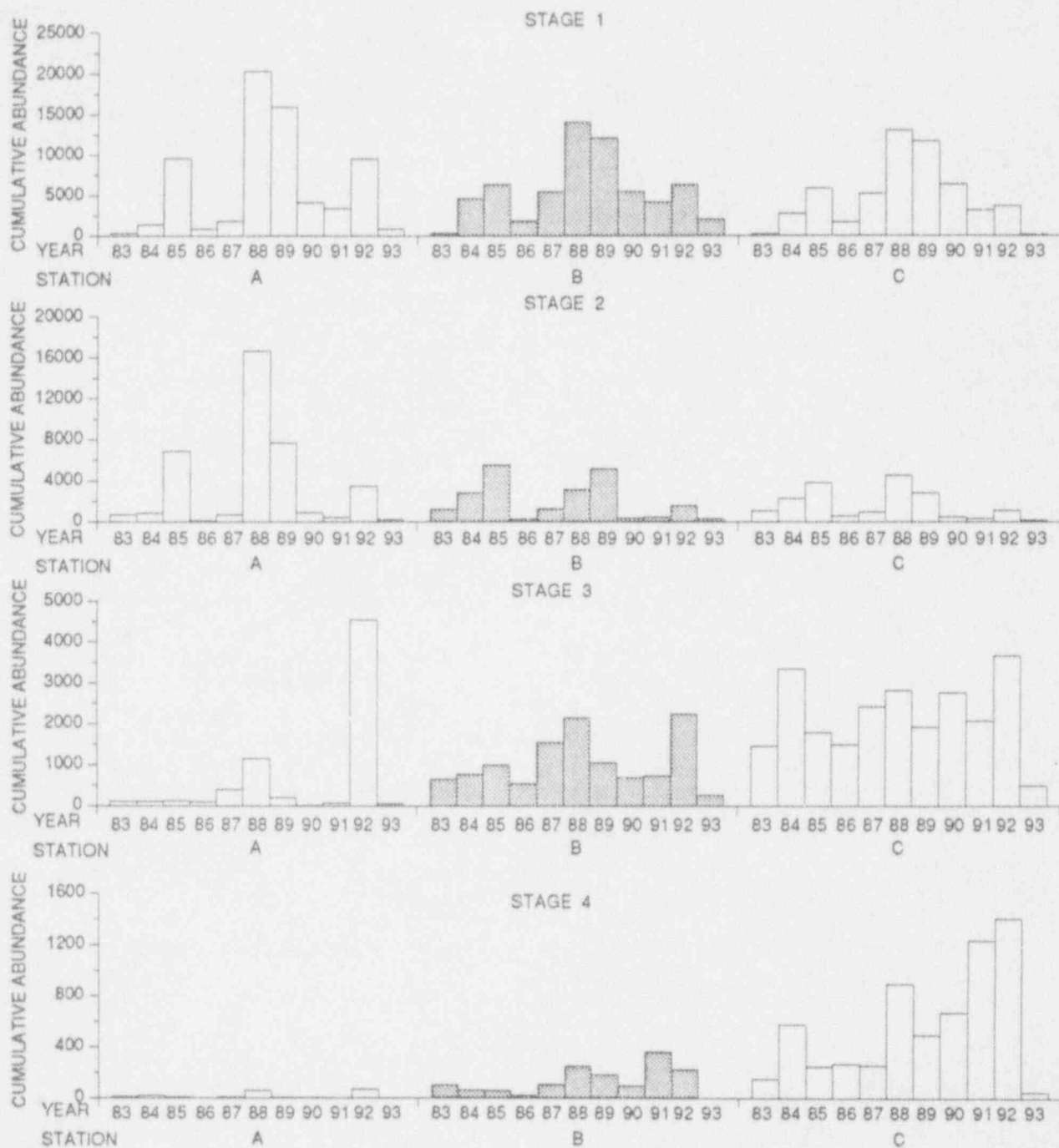


Fig. 14. Cumulative density of each developmental stage of larval winter flounder at the Niantic River stations A, B and C from 1983 through 1993. (Note that the vertical scales differ among the graphs).

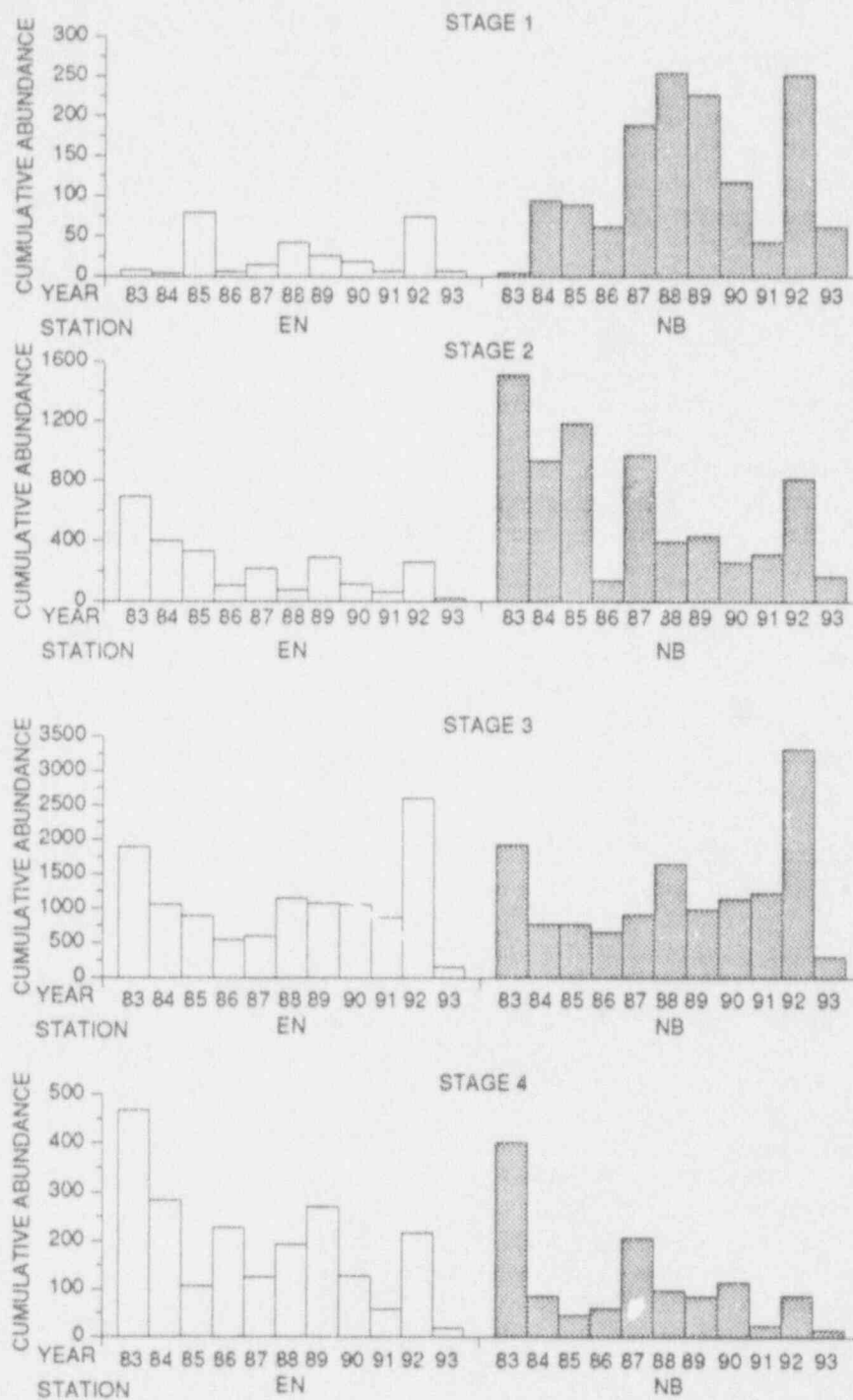


Fig. 15. Cumulative density of each developmental stage of larval winter flounder at the Niantic Bay stations EN and NB from 1983 through 1993. (Note that the vertical scales differ among the graphs).

attributed to undersampling because of net extrusion (NUSCO 1987). However, this was rectified in 1984 when a net with smaller mesh (202  $\mu\text{m}$ ) was used during the early portion of the larval season. Abundance of Stage 1 larvae at the two Niantic Bay stations was low in comparison to the river, indicating that little, if any, spawning occurred in the bay. Abundances at station NB were consistently greater than at EN, possibly because NB was located closer to the river mouth, the likely source of Stage 1 larvae, or because undersampling occurred at EN as a result of extrusion through the 333- $\mu\text{m}$  mesh net used there. At station NB, ranks of annual abundance indices were similar to those of the river stations and this suggested that most Stage 1 larvae collected in the bay probably originated from the Niantic River. There was a significant ( $p \leq 0.05$ ) positive correlation among Stage 1 annual abundances at all stations, except for stations EN and NB (Table 15).

Stage 2 abundance in 1993 at the three river stations was also among the lowest in the 11-year series. In general, annual ranks of Stage 2 abundance at the three river stations were similar to those of Stage 1. This implied a similar annual rate of larval loss (mortality and flushing) during larval development from Stage 1 to 2. Annual abundances at stations B and C were almost identical. Stage 2 larvae occurred predominantly in the river, but were more prevalent in the bay compared to Stage 1. As with Stage 1, Stage 2 abundance at station NB was greater than at EN, which suggested that either station NB was closer to the source of newly hatched larvae or smaller Stage 2 larvae were extruded through the 333- $\mu\text{m}$  mesh net used at EN. There was a significant ( $p \leq 0.05$ ) positive correlation among all river stations and between stations EN and NB (Table 15).

Stages 3 and 4 larvae were generally most abundant at station C and their abundance at the two bay stations (EN and NB) increased to levels similar to or greater than at stations A and B. In 1993, Stage 3 larval abundance was the lowest recorded at stations B, C, EN, and NB and low abundance was also evident for Stage 4 larvae. In previous years, later developmental stages of winter flounder larvae were not found to be homogeneously distributed throughout the river, but due to low abundance in 1993 this was not as evident. The larval abundance decline at the upper river stations (A and B) as development progressed may have represented a gradual flushing to the lower portion of the river and into the bay. Stage 3 annual abundances were similar at the two bay stations and

were highly correlated (Table 15).

During 1993, the abundance of all developmental stages at all stations was consistently among the lowest recorded in the 11-year period. Similar annual abundance of Stage 3 larvae at EN and NB suggested that the different sampling techniques at the two stations were comparable, at least for Stage 3 larvae (Fig. 15). Generally, Stage 4 larvae were slightly less abundant at NB than at EN, suggesting that this developmental stage was somewhat more susceptible to entrainment than to capture by the bongo sampler.

Annual abundance of newly hatched winter flounder larvae should be related to adult reproductive capacity (egg production) and the fraction of eggs that hatch. To examine this relationship, the annual egg production estimates (Table 13) were compared to the annual abundance of Stage 1 larvae. The index of Stage 1 larval abundance was the  $\alpha$  parameter from the Gompertz function (Eq. 2) for the Niantic River (stations A, B, and C combined). A functional regression indicated a strong positive relationship ( $r = 0.795$ ;  $p = 0.006$ ) between egg production and Stage 1 abundance (Fig. 16). The abundance of newly hatched larvae was directly related to the adult reproductive capacity under the assumption that egg hatchability was similar among years. The consistency of this relationship implied that precision in the sampling of Stage 1 larvae and, additionally, that egg production estimates were a reasonable index of annual reproductive capacity.

The 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

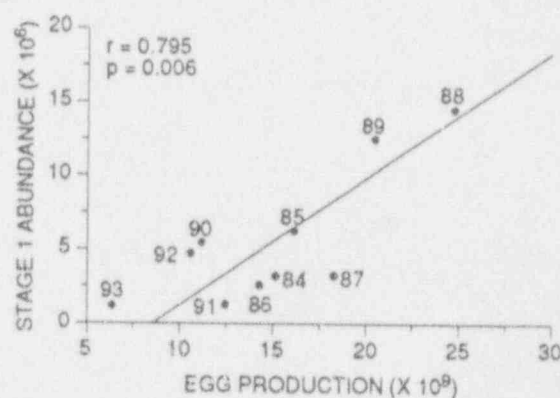


Fig. 16. The relationship (functional regression) between annual Stage 1 abundance in the Niantic River (estimated from the  $\alpha$  parameter of the Gompertz function) and egg production from 1984 through 1993.

TABLE 15. Matrix of Spearman's rank-order correlations among stations for the annual cumulative abundance of each developmental stage of larval winter flounder from 1983 through 1993.

Stage	Station	B	C	EN	NB
1	A	0.9636 0.0001 **	0.9273 0.0001 **	0.7654 0.0060 **	0.7636 0.0062 **
	B		0.9182 0.0001 **	0.7791 0.0047 **	0.8455 0.0010 **
	C			0.7016 0.0161 *	0.7727 0.0053 **
	EN				0.5786 0.0622 NS
2	A	0.8546 0.0008 **	0.8273 0.0017 **	0.3182 0.3403 NS	0.3636 0.2716 NS
	B		0.9000 0.0002 **	0.5364 0.0890 NS	0.6455 0.0320 *
	C			0.5455 0.0827 NS	0.5455 0.0827 NS
	EN				0.8182 0.0021 **
3	A	0.9000 0.0002 **	0.5091 0.1097 NS	0.5091 0.1097 NS	0.3546 0.2847 NS
	B		0.7364 0.0098 **	0.5455 0.0827 NS	0.4364 0.1797 NS
	C			0.4909 0.1252 NS	0.6909 0.0186 *
	EN				0.7636 0.0062 **
4	A	0.3853 0.2419 NS	0.3119 0.3504 NS	0.4771 0.1379 NS	0.5885 0.0568 NS
	B		0.7182 0.0128 *	0.0546 0.8734 NS	0.3098 0.3539 NS
	C			0.0636 0.8525 NS	0.0775 0.8209 NS
	EN				0.5239 0.0981 NS

\* 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  $p \leq 0.01$ ).

river (station A, B, and C combined) and bay (stations EN and NB combined) for each developmental stage (Table 16). Dates of peak abundance for Stage 1 larvae could not be estimated for the bay stations

because this larval stage was rarely collected there and, similarly, for Stage 4 in the river during 1993 because of low abundance. In 1993, the dates of peak abundance for the first three developmental stages of larvae

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

Year	Stage 1	Stage 2	Stage 3	Stage 4
<u>Niantic River</u>				
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)	May 16 (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)	*
<u>Niantic Bay</u>				
1983	-	April 7 (2)	April 23 (1)	May 10 (4)
1984	-	April 8 (2)	May 4 (3)	May 25 (8)
1985	-	April 1 (4)	April 29 (6)	May 18 (3)
1986	-	April 5 (30)	April 28 (3)	May 11 (2)
1987	-	April 6 (6)	April 28 (2)	May 16 (4)
1988	-	March 24 (3)	April 22 (2)	May 9 (5)
1989	-	April 13 (1)	April 23 (2)	May 17 (3)
1990	-	April 3 (8)	April 23 (2)	May 7 (5)
1991	-	March 28 (5)	April 11 (3)	April 29 (4)
1992	-	April 15 (4)	April 30 (2)	May 7 (4)
1993	-	April 3 (44)	May 6 (8)	May 23 (11)

\* Due to low abundance during the 1993 sampling, the Gompertz function could not be fitted to the data.

in the river were within the range of those from the previous 10-year period. This was also evident for Stage 2 larvae in the bay. Stage 1 larvae in the river generally peaked in late February to early March. Based on water temperatures of 2 to 3°C during February and egg incubation times reported by Buckley (1982), peak spawning generally occurred in early to mid-February. Buckley et al. (1990) reported that egg incubation time was inversely related to water temperature during oocyte maturation and egg incubation. But a comparison between the 1983-93 February water temperatures (Table 6) and the annual dates of Stage 1 peak abundance in the river did not show a significant relationship (Spearman's rank-order correlation coefficient  $\rho = -0.478$ ;  $p = 0.137$ ). The dates of peak abundance for Stage 3 and 4 larvae in the bay were the latest of the 11-year period and they had the largest 95% confidence intervals.

### *Development and growth*

The length-frequency distribution of each larval stage has remained fairly consistent since developmental stage determination began in 1983 (NUSCO 1987, 1988a, 1989, 1990, 1991b, 1992a, 1993). Stage-specific length-frequency distributions by 0.5-mm size-classes in 1993 showed a separation in predominant size-classes for each developmental stage (Fig. 17). Stage 1 larvae were primarily (96%) in the 2.5 to 3.5-mm size-classes, 87% of Stage 2 were 3.0 to 4.0 mm, 85% of Stage 3 were 4.5 to 8.0 mm, and 79% of Stage 4 were 7.5 to 8.5 mm. Larger Stage 3 larvae apparently overlapped smaller Stage 4 fish in total length. These consistent results from year to year indicated that developmental stage and length of larval winter flounder were closely related. These data agreed with laboratory studies on larval winter flounder, which showed that there were positive correlations



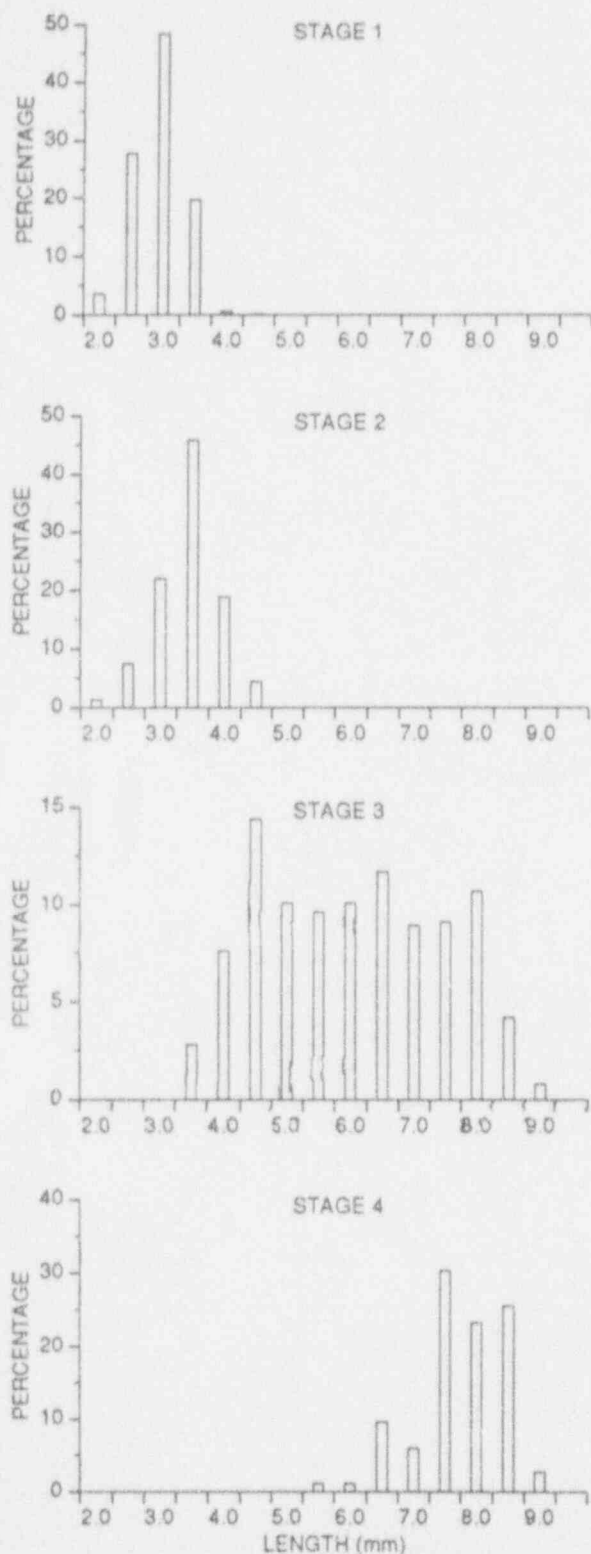


Fig. 17. Length-frequency distribution of larval winter flounder by developmental stage for all stations combined in the Niantic River and Bay during 1993. (Note that the vertical scales differ among the graphs).

between growth and developmental rates (Chambers and Leggett 1987; Chambers et al. 1988). This relationship was the basis for the estimation of developmental stage from length-frequency data.

The length-frequency distributions of larvae (all stages combined) collected in the Niantic River (stations A, B, and C combined) were quite different from those obtained for Niantic Bay (stations EN and NB combined) in 1993 (Fig. 18). Differences in size-class distribution between the two areas were consistent with previous findings (NUSCO 1987, 1988a, 1989, 1990, 1991b, 1992a, 1993) and the pattern seen in spatial distribution by developmental stage, where Stage 1 and 2 larvae were more abundant in the river than in the bay (see Figs. 14 and 15). Smaller size-classes predominated in the river, which had about 82% of the larvae in the 3.5-mm and smaller size-classes. By contrast, more than 76% of the larvae in the bay during 1993 were in the 4.0-mm and larger size-classes. A slight increase in frequency of larger size-classes has been apparent in some previous years in the river (NUSCO 1987, 1988a, 1989, 1991b,

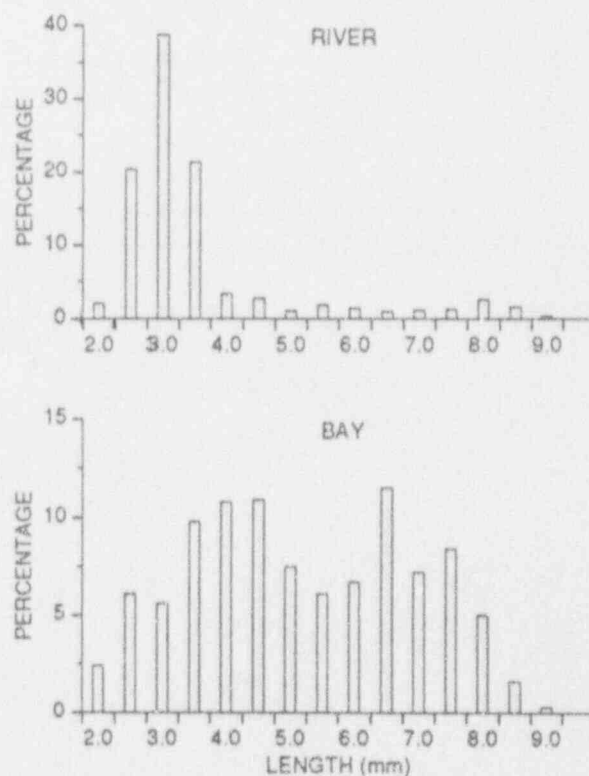


Fig. 18. Length-frequency distribution of larval winter flounder in the Niantic River and Bay during 1993. (Note that the vertical scales differ between the graphs).

1992a, 1993), suggesting that some older larvae were imported to the river. This import of larger size-classes was apparent in the length-frequency distribution at the river mouth (station RM) in samples collected near maximum flood current during 1991-93 (Fig. 19).

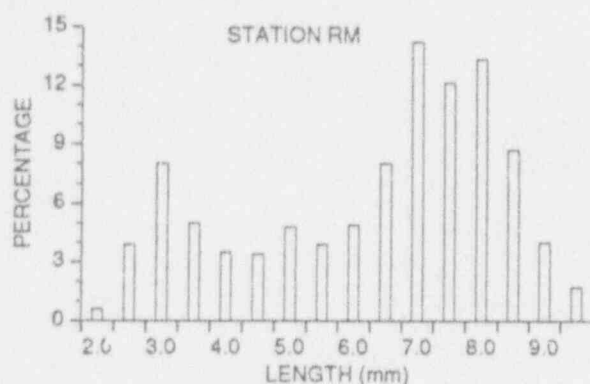


Fig. 19. Length-frequency distribution of larval winter flounder collected at the Niantic River mouth near maximum flood current during 1991-93.

Length-frequency data from entrainment collections taken from 1976 through 1993 (station EN) were used

to estimate larval winter flounder growth rates for Niantic Bay. Weekly mean lengths during a season formed a sigmoid-shaped curve (NUSCO 1988a). The linear portion of the sigmoid curve usually occurred during the middle of the larval season and growth rates were estimated by fitting a linear model to individual larval length measurements during that period. This model adequately described growth and all slopes (growth rate as  $\text{mm} \cdot \text{day}^{-1}$ ) were significantly ( $p \leq 0.001$ ) different from zero (Table 17). In addition, most intercepts of the linear regression were about 2, the approximate size of winter flounder larvae at hatching. Annual growth rates for station EN were variable and ranged from 0.048 to  $0.100 \text{ mm} \cdot \text{day}^{-1}$ , with 1993 having the lowest value. To validate this estimation technique, growth rates were estimated from length data collected at station NB from 1979 through 1989 (NUSCO 1990); annual growth rates were highly correlated ( $r = 0.89$ ;  $p \leq 0.001$ ) with those from station EN.

In laboratory studies, water temperature affected the growth rate of winter flounder larvae (Laurence 1975; NUSCO 1988a). To examine the effect of temperature on estimated annual growth rates, mean water temperatures for Niantic Bay, determined using data collected from continuous recorders in the intakes of Units 1 and 2, were calculated for a 40-day period

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

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

<sup>a</sup> Time period of the weekly mean lengths used to estimate growth rate.

<sup>b</sup> 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.

starting at the beginning of the week when the first larval length measurements were used to estimate the annual growth rate (Table 17). A positive exponential relationship was found between growth rate and water temperature (Fig. 20). A similar exponential relationship of temperature to larval growth was reported for plaice by Hovenkamp and White (1991). If temperature affects growth rate, then the length of a larva at a specific time during the season should be related to water temperatures to which it has been exposed. Therefore, the mean length of larvae collected at station EN during the period of April 1-15 for each year was compared to the mean March water temperatures (Fig. 21). Again, there was a positive relationship with larger mean lengths associated with warmer March temperatures. As concluded previously from comparisons of annual length-frequency distribution and developmental stages, growth and larval development were found to be closely related. If water temperature affects growth rates, then it should also affect larval developmental time. The timing of peak larval abundance should therefore be related to the rates of recruitment and loss (including mortality and juvenile metamorphosis), which, in turn, would be affected by larval development. Annual dates of peak abundance of larval winter flounder collected at EN were negatively correlated to the mean water temperature in March and April (Fig. 22). Earlier dates of peak abundance were associated with warmer mean water temperatures. This agreed with the results of Laurence (1975), who found that winter flounder larvae metamorphosed 31 days earlier at 8°C than at 5°C. Annual dates of peak abundance varied by 41 days during the 17-year period, possibly because of a 3.6°C difference in the March-April water temperature between the earliest (April 13, 1991) and the latest (May 23, 1978) dates of peak abundance. 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 consistent with laboratory observations for larval winter flounder as Chambers et al. (1988) found that, at metamorphosis, age was more variable than length and larval age and length were independent.

Growth rates were also estimated for Niantic River larvae using length data from station C with the methods given above. Station C was selected for this analysis because all developmental stages were collected there in abundance (Fig. 14). Estimated growth rates for larvae from the river were generally greater than for larvae from the bay and the rate for 1993 for

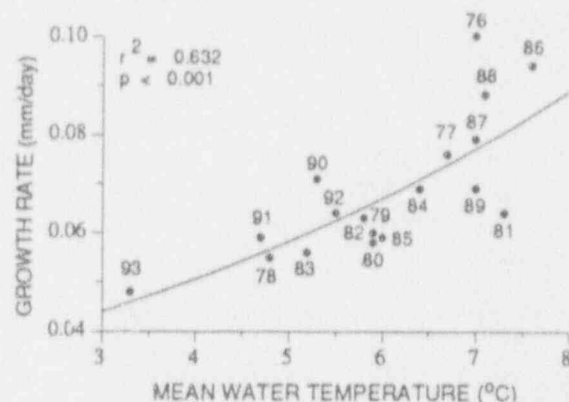


Fig. 20. The exponential relationship between mean water temperature  $T$  (°C) and the estimated growth rate  $G$  (mm per day) of winter flounder larvae at station EN from 1976 through 1993 ( $G = 0.029 \cdot e^{0.140T}$ ).

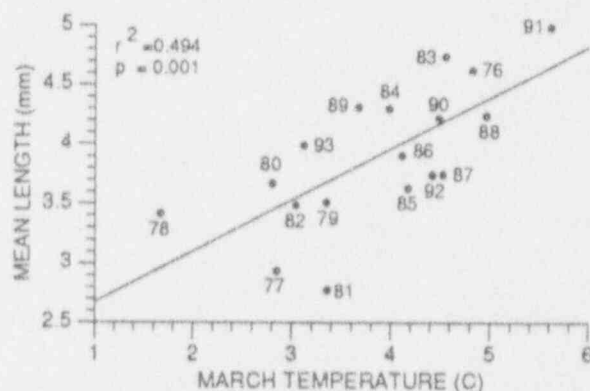


Fig. 21. The relationship between annual mean March water temperature (°C) and the mean length during April 1-15 at station EN for 1976 through 1993.

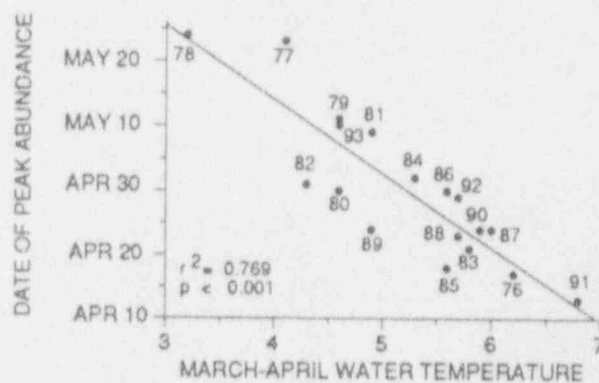


Fig. 22. The relationship between March-April mean water temperature (°C) and the annual date of peak abundance (estimated from the Gompertz function) of winter flounder larvae at station EN from 1976 through 1993.

the river was the lowest during the 11-year period (Table 18). Again, a linear model provided a good fit and slopes (growth rates as  $\text{mm}\cdot\text{day}^{-1}$ ) were highly significantly ( $p \leq 0.001$ ) different from zero. Growth of larvae in the river was similar to laboratory growth rates of 0.104 and 0.101  $\text{mm}\cdot\text{day}^{-1}$  at mean water temperatures of 6.9 and 7.5°C, respectively (NUSCO 1988a). An annual mean water temperature was determined from surface and bottom temperatures measured at the time of sample collection during a 6-week period starting the same week from which the first length measurements were used in the growth rate calculation. Previously, there was no apparent relationship between growth rates in the river and water temperature, but a negative relationship was found between growth and the abundance of Stage 2 larvae, suggesting density-dependent growth (NUSCO 1990, 1991b, 1992a, 1993). The abundance of Stage 2 larvae was examined because it is during this developmental stage that larvae begin to feed. With the addition of 1993 data this density-dependent growth relationship was no longer significant ( $p = 0.322$ ) when tested with a functional regression. Because there was a strong relationship between growth and water temperature in the bay, both Stage 2 abundance ( $\alpha$  parameter; Eq. 2) and water temperature were used as independent variables in a multiple regression model to examine growth rates. Prior to conducting multiple regression analysis it was determined that the two independent variables were not correlated (Spearman's rank-order correlation coefficient  $\rho = 0.145$ ;  $p = 0.670$ ). The multiple regression was

significant ( $p = 0.035$ ;  $r^2 = 0.567$ ), with the coefficient for temperature positive and the one for Stage 2 abundance negative, although the Stage 2 coefficient was not significant ( $p = 0.055$ ). These results suggested that winter flounder larval growth in the Niantic River may be a function of both water temperature and larval density. A laboratory growth study of larval winter flounder held at 8°C showed a decrease in growth rate as prey densities decreased (Laurence 1977). This study, along with the apparent density-dependent growth in the Niantic River, suggested that as the number of feeding larvae increased, the numbers of available prey declined to levels less than optimum for larval growth.

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

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

Year	Time period included <sup>a</sup>	Growth rate ( $\text{mm}\cdot\text{day}^{-1}$ )	95% confidence interval	Mean water temperature (°C) <sup>b</sup>
1983	March 20 - May 1	0.100	0.096 - 0.104	6.1
1984	March 25 - May 6	0.100	0.094 - 0.105	6.4
1985	March 31 - May 26	0.084	0.080 - 0.088	7.7
1986	March 23 - May 4	0.109	0.103 - 0.115	8.0
1987	March 22 - May 10	0.099	0.095 - 0.103	7.2
1988	March 20 - May 21	0.099	0.094 - 0.104	6.8
1989	March 26 - May 21	0.087	0.082 - 0.092	7.4
1990	March 25 - May 13	0.106	0.099 - 0.113	7.5
1991	March 10 - April 28	0.123	0.114 - 0.132	6.9
1992	March 15 - May 17	0.088	0.083 - 0.093	5.7
1993	March 7 - May 16	0.070	0.065 - 0.075	4.1

<sup>a</sup> Time period of the weekly mean lengths used to estimate growth rate.

<sup>b</sup> Mean during a 6-week period starting the week of the first weekly mean length used in estimating growth rate.



## Mortality

Based on length-frequency distributions in the river during 1993 (Fig. 18) and previous years, most winter flounder larval mortality occurred between the 3.0- to 4.0-mm size-classes. About a 90% decline in frequency occurred in 1993 between these two size-classes, which included yolk-sac (Stage 1) and first-feeding Stage 2 larvae. This initial large decline was followed by smaller decreases to the 5.0-mm size-class, indicating a reduction in the mortality rate. Pearcy (1962) reported a greater mortality for young winter flounder larvae ( $20.7\% \cdot \text{day}^{-1}$ ) compared to older individuals ( $9.1\% \cdot \text{day}^{-1}$ ). In a laboratory study on winter flounder larvae, Chambers et al. (1988) reported that larval mortality was concentrated during the first 2 weeks after hatching. Based on the estimated growth rate in the river for 1983 of  $0.077 \text{ mm} \cdot \text{day}^{-1}$  (Table 18), it would require about 13 days to grow from 3 to 4 mm. The above 90% decline between these size-classes would be equivalent to a mortality of about  $16\% \cdot \text{day}^{-1}$ , similar to that reported by Pearcy for young winter flounder larvae in the Mystic River. Laurence (1977) found that winter flounder larvae had a low energy conversion efficiency at first feeding (i.e., Stage 2) compared to later developmental stages, and that it was probably a "critical period" in larval development. Hjørleifsson (1989) showed that the ratio between RNA and DNA, an index of condition and growth rate, was lowest at the time of first feeding of winter flounder (about 4 mm) and that these ratios were affected by food availability. The "critical period" concept, hypothesized by Hjort (1926), was discussed by May (1974) for marine fishes. In many cases, the strength of a year-class is thought to be determined by the availability of sufficient food after completion of yolk absorption.

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

contact predators, including carnivorous copepods and amphipods, cnidarians, and ctenophores.

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

A possible predator of winter flounder larvae in the Niantic River was medusae of the lion's mane jellyfish (*Cyanea* sp.), which was prevalent in the upper portion of the river at station A. Marshall and Hicks (1962) also reported that jellyfish were abundant in the upper river. A laboratory study showed that winter flounder larvae contacting the tentacles of the lion's mane jellyfish were stunned and ultimately died, even if not consumed by the medusa (NUSCO 1988a). During 5 of the 11 years (1983, 1984, 1986, 1989, and 1990) that larvae were sampled at station A, weekly mean larval abundance was negatively correlated ( $p \leq 0.05$ ; Spearman's rank-order correlation coefficient  $\rho$  range of  $-0.736$  to  $-0.927$ ) to weekly mean jellyfish volume during the period when medusae were collected. In 1993, jellyfish abundance was bimodal with a smaller peak in mid-March and the larger in late April (Fig. 23). During the time of both peaks, jellyfish abundance was greater than the previous 10-year average. Although the 1993 annual larval abundance was among the lowest at station A (Fig. 13) and coincident with higher than average jellyfish abundance, larval abundance was also low at the remaining river stations where jellyfish were rarely collected. Therefore, the low abundance of larvae in the river could not be directly attributed to jellyfish



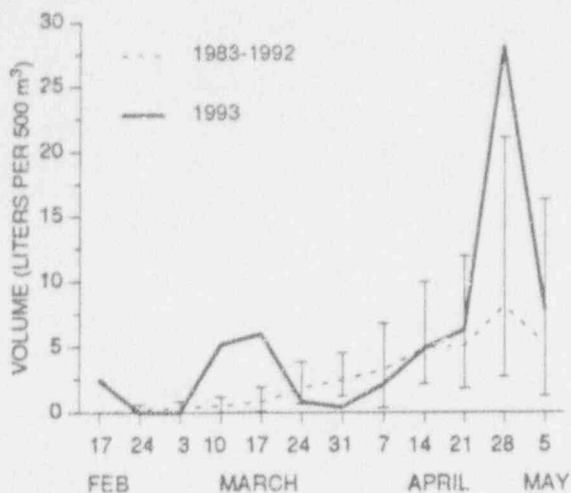


Fig. 23. Comparison of *Cyanea* sp. weekly mean volumes collected in 1983-92 (with 95% confidence intervals) to weekly volumes in 1993. Collections were made at station A in the Niantic River.

predation in the upper portion of the river. Although during some years there appeared to be a relationship between the temporal decline of winter flounder larval abundance at station A in the Niantic River with the occurrence of lion's mane jellyfish medusae, for other years there was no relationship between annual larval abundance at station A and annual mean jellyfish volume. The decline in larval abundance at station A may also be attributed to a gradual flushing of larvae out of the upper portion of the river.

The possibility of density-dependent mortality of winter flounder larvae was examined using a function (Eq. 4) provided by Ricker (1975) that requires estimates of annual spawning stock size and larval recruitment. The annual egg production estimate in the Niantic River (Table 13) was used as a measure of spawning stock size. The  $\alpha$  parameter from the Gompertz function fit to the abundance of 7-mm and larger larvae collected from 1976 through 1993 at station EN was selected as a measure of larval recruitment. Larvae in the 7-mm and larger size-classes were used because they would soon metamorphose into juveniles. A larval recruitment index was calculated by taking the logarithm of the ratio of the  $\alpha$  parameter for 7-mm and larger larvae to the egg production estimates. This value was plotted against egg production estimates and the slope determined with functional regression (Fig. 24). Although there was some scatter around the relationship, a significant ( $r = -0.579$ ;  $p = 0.015$ ) negative relationship was found, indicating that compensatory mortality occurred during

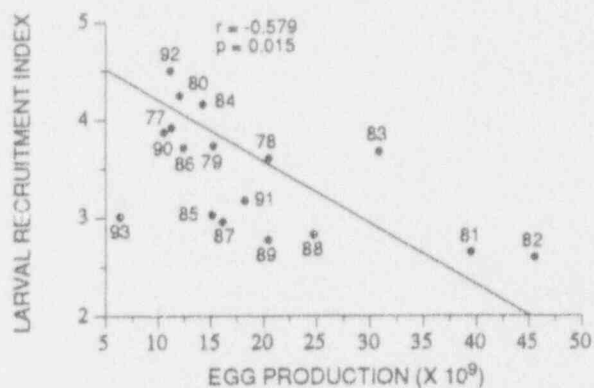


Fig. 24. The relationship (functional regression) between the annual winter flounder egg production in the Niantic River and the larval recruitment index (logarithm of the ratio of the annual abundance of 7 mm and larger larvae to the egg production) at station EN from 1976 through 1993.

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

Because the egg production estimate was used in calculating the larval recruitment index above, a possibility existed of introducing correlation between the independent (egg production) and dependent (recruitment index) variables. Therefore, another approach for detecting the presence of density-dependent larval mortality for the Niantic River stock was used, where annual larval mortality rates from the river were compared to estimates of river spawning stock size (i.e., egg production). Total larval mortality in the river for 1984-93 ranged from 82.4 to 97.9%, with a mean instantaneous rate  $Z$  of 2.77 (Table 19). To determine if density-dependent mortality could be identified in the larval stage, the values of  $Z$  were compared to egg production estimates using functional regression. Previously, a significant ( $p \leq 0.05$ ) relationship was apparent, such that when egg production increased, larval mortality also increased (NUSCO 1991b). But with the addition of 1991-93 data, this relationship was no longer significant ( $p = 0.204$ ). The mortality rate for 1991 was lower and the rates for 1992 and 1993 were higher than expected, based on total egg production estimates (Fig. 25). Therefore, it was no longer evident that density-dependent larval mortality was occurring in the Niantic River.

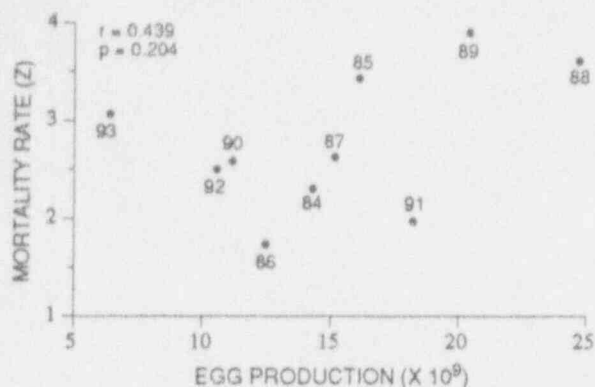


Fig. 25. Lack of relationship between annual winter flounder egg production and instantaneous larval mortality rate in the Niantic River from 1984 through 1993.

## Juvenile winter flounder

### Age-0 juveniles (summer)

**Abundance.** Although beam trawls are much more efficient than small otter trawls for collecting juvenile flatfish (Kuipers et al. 1992), the densities reported below should be regarded as minimum estimates because of inefficiencies in the collection process. For example, using a beam trawl Berghahn (1986) caught more young plaice at night in comparison to samples taken during the day and Rogers and Lockwood (1989) showed that replacing tickler chains normally used with even heavier, spiked chains nearly doubled catches. The efficiency of the NUSCO 1-m beam trawl was discussed in NUSCO (1990). In late June of 1983, two tickler chains added to the beam trawl considerably increased catch efficiency as older and larger fish apparently were able to avoid the net without them in place (NUSCO 1987).

Compared to previous years of sampling, the initial numbers of newly metamorphosed young-of-the-year winter flounder were relatively low at all stations in 1993, particularly in the Niantic River, where abundance was only 10-20 fish·100m<sup>-2</sup> (Fig. 26). Although densities in the bay were greater than in the river, considerably fewer fish were present than in 1991 or 1992. As in most other years of sampling, numbers of young declined quickly at the bay stations and none were collected by late summer. Similar to observations made in several other years, a small increase in abundance occurred at the river stations during the last day of sampling in September, perhaps

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

Year	Abundance index		Mortality (%)	Instantaneous mortality rate
	Newly hatched	7-mm size-class		
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

mean = 2.77

the result of additional fish entering shallow water once water temperatures began to decrease from late summer peaks. Saucerman and Deegan (1991) also found that young winter flounder responded to warm water temperatures during late August in Waquoit Bay, MA by moving into deeper water and returning to the shallows after those areas became cooler. Catches during the previous two sampling dates in September may also have been negatively affected at LR by large mats of the alga *Enteromorpha clathrata*, which likely reduced sampling efficiency of the 1-m beam trawl, and by a boat moored at WA, which caused a deviation in the normal tow path to somewhat shallower or deeper water. Bagge and Nielsen (1988) noted reductions in age-0 plaice abundance in Denmark when filamentous algae covered large parts of their shallow nursery area. Pihl and van der Veer (1992) also reported similar density reductions of young plaice in a Swedish bay when about half of the bottom became covered by *Enteromorpha flexuosa* and the organic content of the sediments nearly doubled.

The median CPUE for the first half of the season at station BP was 22.5, the lowest since 1989, and the CPUE at RM was 8.3, which was the smallest of the 5-year time series (Table 20). Early season densities in the Niantic River were also historic lows (10.6 for LR and 5.0 for WA; Table 21). Densities for the second half of the summer at LR (5.0) and WA (5.5) were less than or equal to the lowest values found during 11 years of sampling in the river. A plot of moving average densities for each station illustrated the rapid decline of young at the bay stations during all years of sampling, regardless of initial numbers

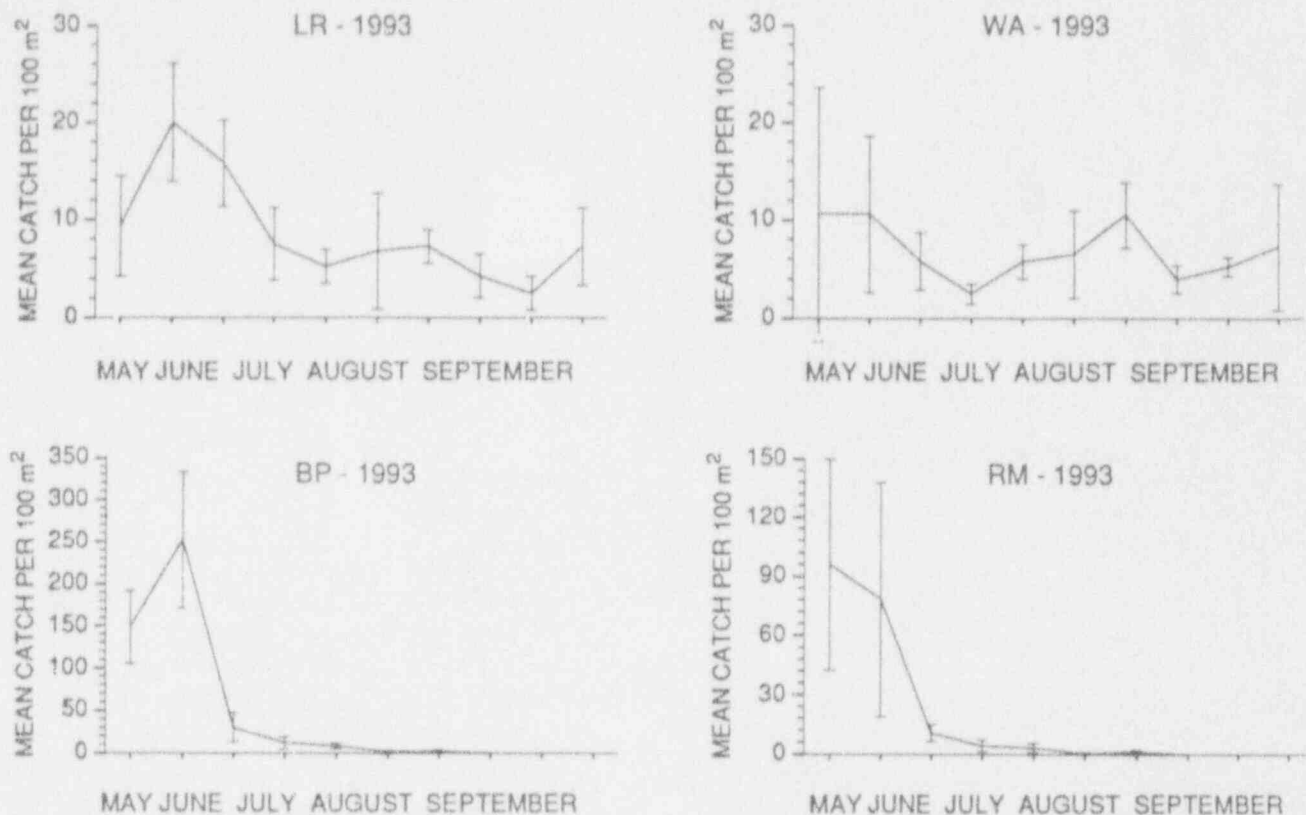


Fig. 26. Weekly mean CPUE ( $\pm 2$  standard errors) of age-0 winter flounder taken in the Niantic River and Bay during 1993. (Note that the vertical scales differ among the graphs).

present (Fig. 27). During many years, a slight increase in density usually occurred following the first one or two weeks of sampling in the Niantic River (Figs. 28 and 29). As summer progressed, this was followed by much smaller decreases. The average abundance of the 1993 year-class was somewhat less than that of the 1989 year-class, which previously had been the lowest of the 11-year time series.

**Growth.** Changes of weekly mean lengths were used to express growth of age-0 winter flounder. A consistent, relatively rapid increase in weekly mean lengths was observed in 1993 for fish in the Niantic River from May through the beginning of July (Fig. 30), which was typical for other years sampled. The rate of increase then decreased through the end of sampling in September. Fast growth after settlement followed by a rapid decline in growth rate was also reported for young winter flounder in New Jersey bays by Sogard and Able (1992), who reported nearly imperceptible growth by the time young reached 50 mm in length. Growth of age-0 winter flounder near

MNPS was less variable than abundance as the weekly means had relatively small confidence intervals, although variability increased as sample sizes became smaller. Mean length of young at LR during late summer (July through September) was 61 mm in 1993, which was exceeded only by the mean of 66 mm for 1983 (Table 22). The mean of 51 mm for WA was also among the highest values observed at that station. This 10-mm difference in mean length between the two river stations was also the largest difference observed since 1985, when it was 15 mm. The relatively large mean lengths in association with low abundance this year and others (e.g., 1983, 1984, 1989) may have been indicative of density-dependent growth. However, this has not been consistent for all years of study, as other environmental factors undoubtedly influenced the growth of young winter flounder.

Almost all fish collected in the bay were smaller than those taken in the Niantic River, especially early in the season (Fig. 30). Growth of individuals in both areas was likely affected by water temperature.

TABLE 20. Seasonal 1-m beam trawl median CPUE (number/100m<sup>2</sup>) of age-0<sup>a</sup> winter flounder at two stations in Niantic Bay (RM and BP) from 1988 through 1993.

Survey year <sup>a</sup>	Station	Season <sup>b</sup>	Tows used for CPUE	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness <sup>c</sup>
1988	RM	Early	39	47.5	30.0 - 72.5	3.68
	RM	Late	36	7.0	6.0 - 8.0	0.20
	BP	Early	40	71.3	32.5 - 107.5	1.17
	BP	Late	32	1.6	1.0 - 3.0	0.99
1989	RM	Early	40	50.8	26.7 - 75.0	0.64
	RM	Late	32	0.0	0.0 - 1.3	1.79
	BP	Early	39	20.0	6.3 - 32.5	1.78
	BP	Late	12	0.0	0.0 - 0.0	3.46
1990	RM	Early	40	40.0	17.5 - 55.0	0.81
	RM	Late	24	0.0	0.0 - 1.0	2.40
	BP	Early	40	32.5	12.5 - 50.0	0.89
	BP	Late	24	1.0	0.0 - 2.0	0.89
1991	RM	Early	44	46.3	15.0 - 65.0	0.63
	RM	Late	0	-	-	-
	BP	Early	44	30.0	5.0 - 107.5	2.29
	BP	Late	8	0.5	0.0 - 1.0	0.00
1992	RM	Early	39	15.0	3.0 - 60.0	1.80
	RM	Late	0	-	-	-
	BP	Early	38	110.0	3.0 - 262.5	1.34
	BP	Late	0	-	-	-
1993	RM	Early	19	8.3	4.0 - 67.5	1.69
	RM	Late	12	0.0	0.0 - 1.0	0.80
	BP	Early	19	22.5	8.8 - 172.5	1.21
	BP	Late	12	0.0	0.0 - 2.0	1.44

<sup>a</sup> For age-0 fish, the year-class is the same as the survey year.

<sup>b</sup> Early season corresponds to late May through July and late to August through September.

<sup>c</sup> Zero for symmetrically distributed data.

Faster growth occurs in warmer waters unless optimal temperatures for growth are exceeded (Sogard and Able 1992). Water temperature was warmer at the river stations, particularly in spring and early summer when growth was most rapid and probably accounted for some of the differences noted. Bergman et al. (1988) and van der Veer et al. (1990) noted that growth of young plaice in northwestern Europe was not food-limited, but was related to prevailing water temperatures and the length of the growing season in different nursery areas. Furthermore, fish grew more rapidly on the warmer nursery grounds in embayments than did fish settling on beaches in the cooler North Sea. Bergman et al. (1988) re-examined reports by Steele and Edwards (1970) and Lockwood (1972) of density-

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

Other factors found to affect growth of young winter flounder include physical location and specific habitat (Sogard 1990; Sogard and Able 1992). Benthic food

TABLE 21. Seasonal 1-m beam trawl median CPUE (number/100m<sup>2</sup>) of age-0<sup>a</sup> winter flounder at two stations in the lower Niantic River (LR and WA) from 1983 through 1993.

Survey year <sup>a</sup>	Station	Season <sup>b</sup>	Tows used for CPUE	Median CPUE estimate	95% confidence interval for median CPUE	Coefficient of skewness <sup>c</sup>
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

<sup>a</sup> For age-0 fish, the year-class is the same as the survey year.

<sup>b</sup> Early season corresponds to late May through July and late to August through September.

<sup>c</sup> Zero for symmetrically distributed data.



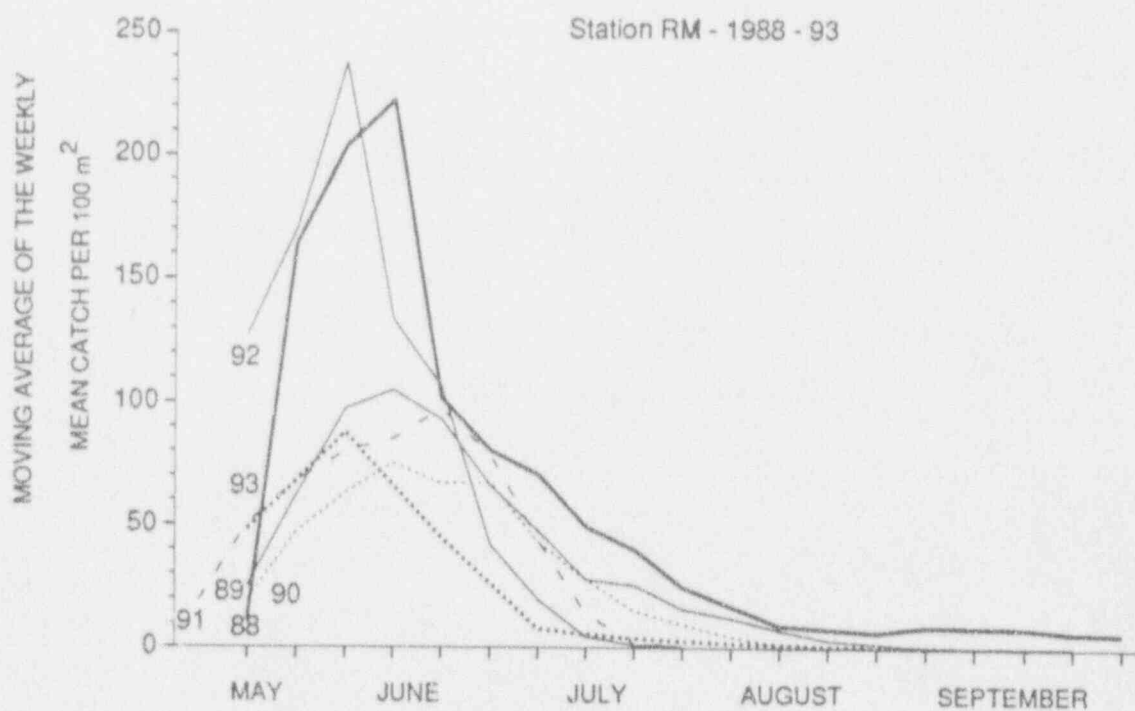
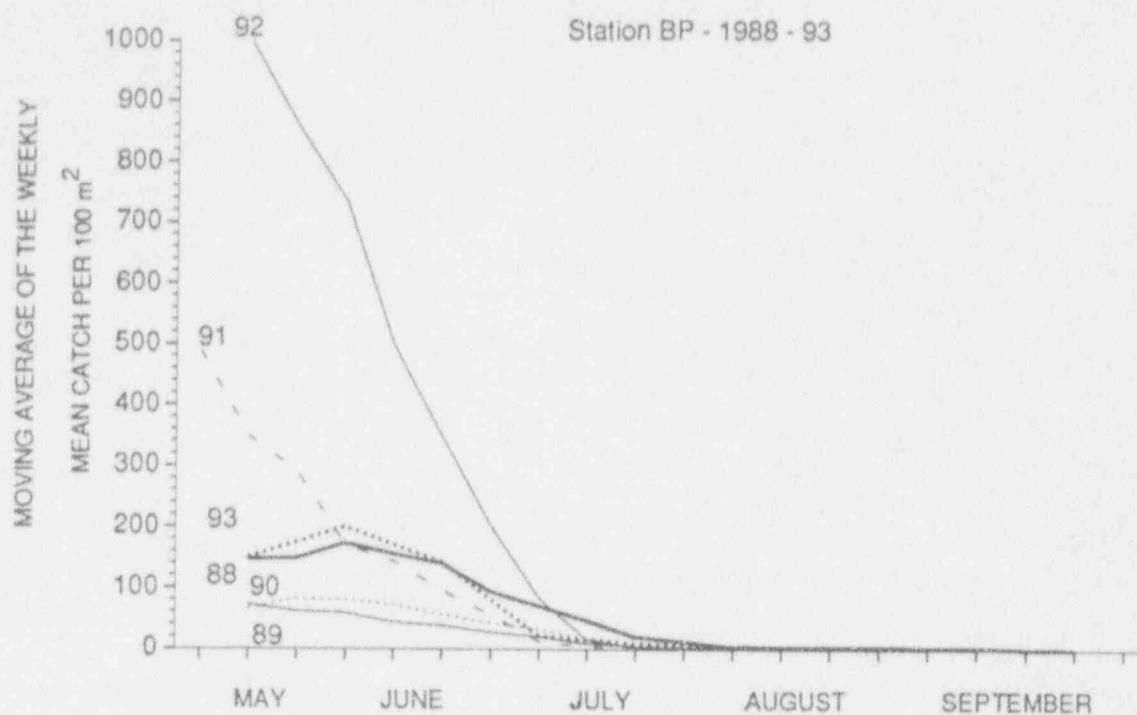


Fig. 27. Moving average of weekly mean CPUE of age-0 winter flounder taken at stations BP and RM in Niantic Bay from 1988 through 1993. (Note that the vertical scales differ between the graphs).

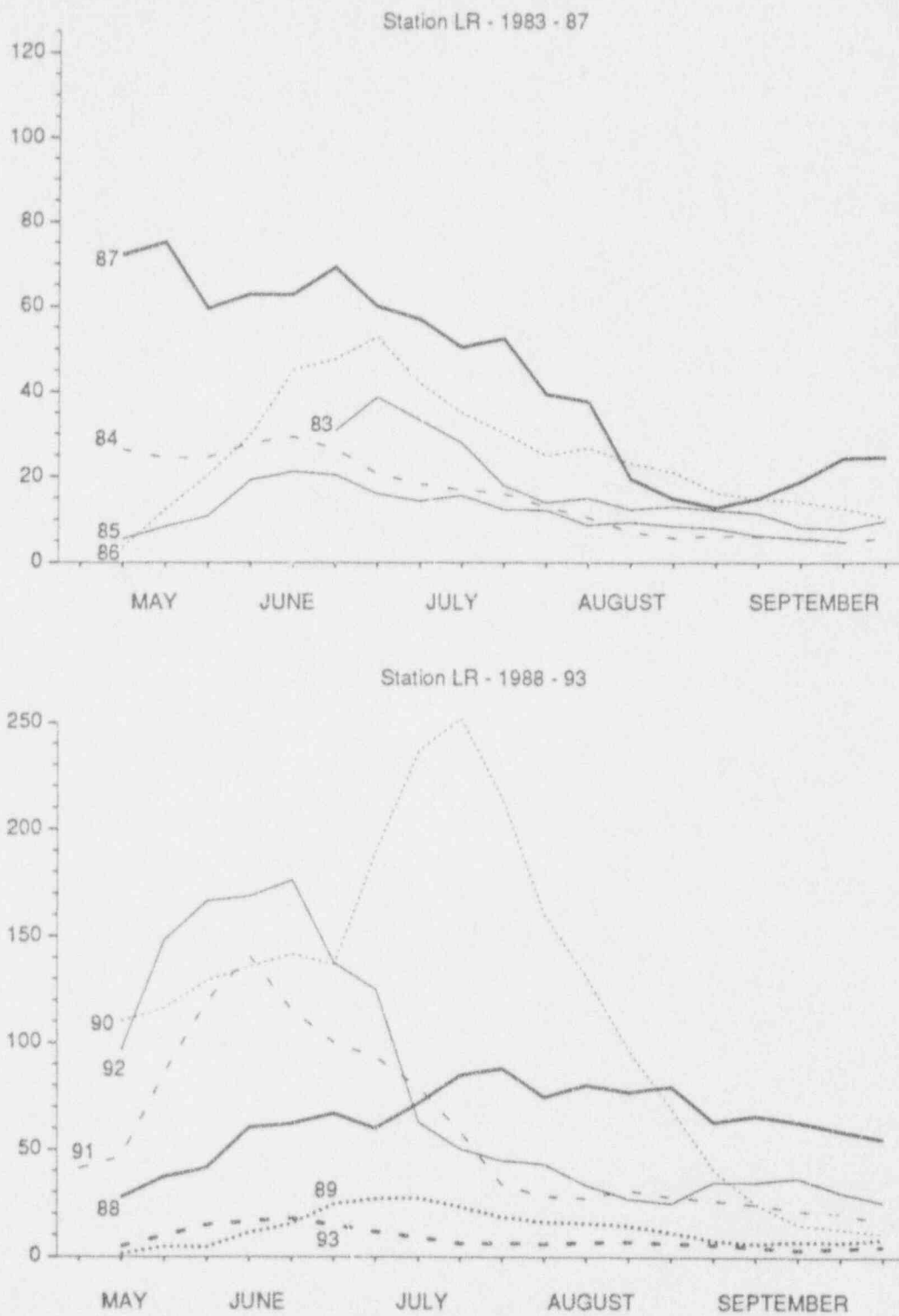


Fig. 28. Moving average of weekly mean CPUE of age-0 winter flounder taken at station LR in the Niantic River from 1983 through 1987 and 1988 through 1993. (Note that the vertical scales differ between the graphs).

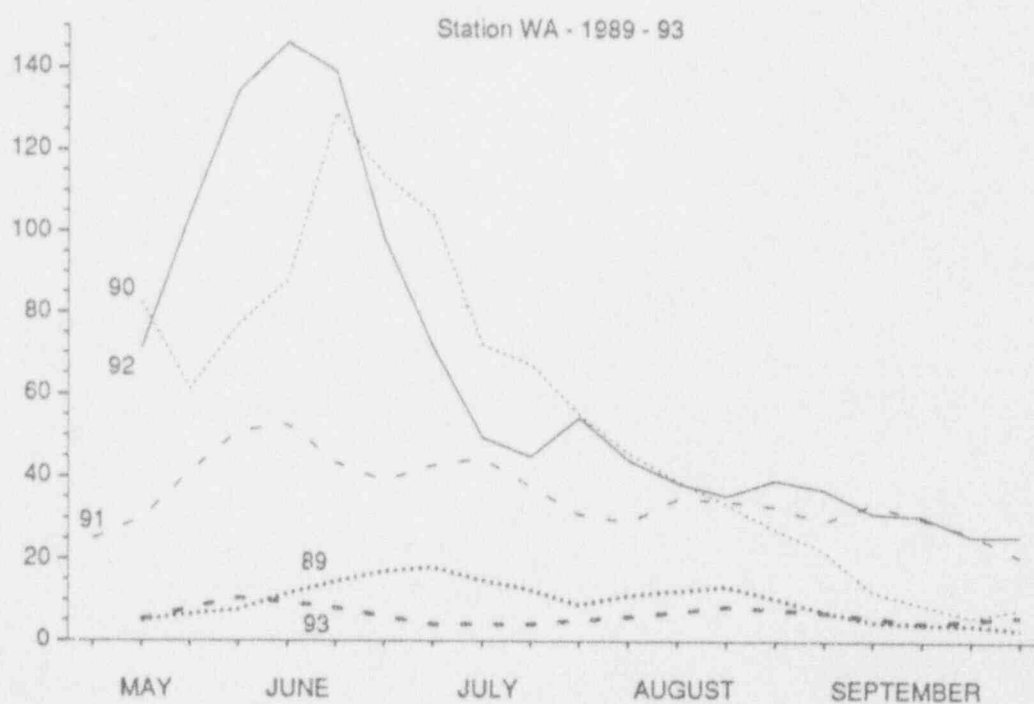
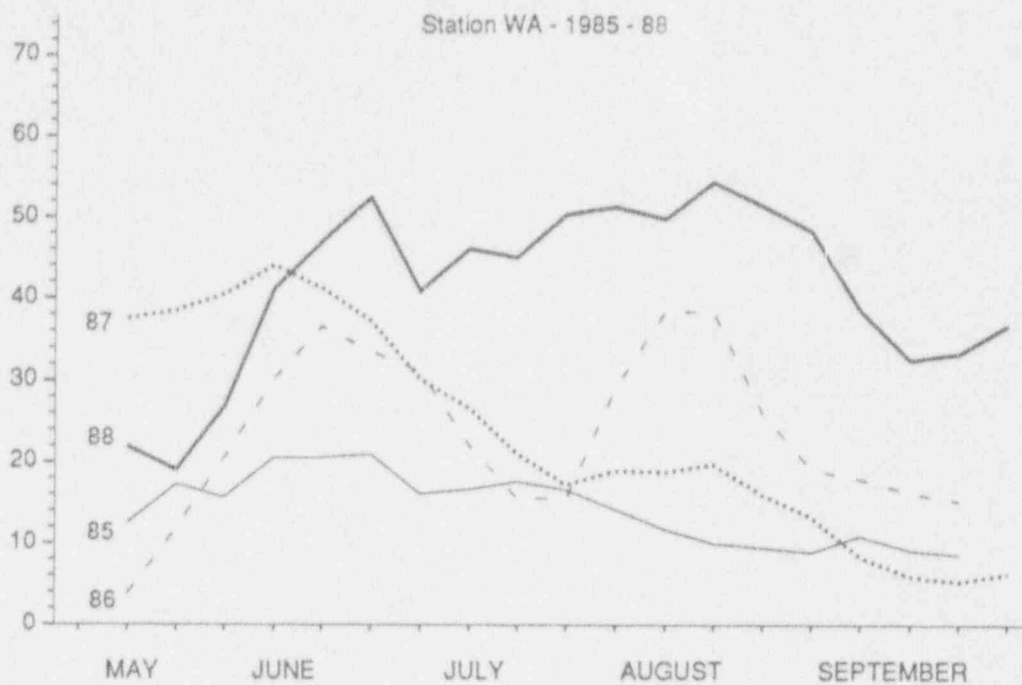


Fig. 29. Moving average of weekly mean CPUE of age-0 winter flounder taken at station WA in the Niantic River from 1985 through 1988 and 1989 through 1993. (Note that the vertical scales differ between the graphs).

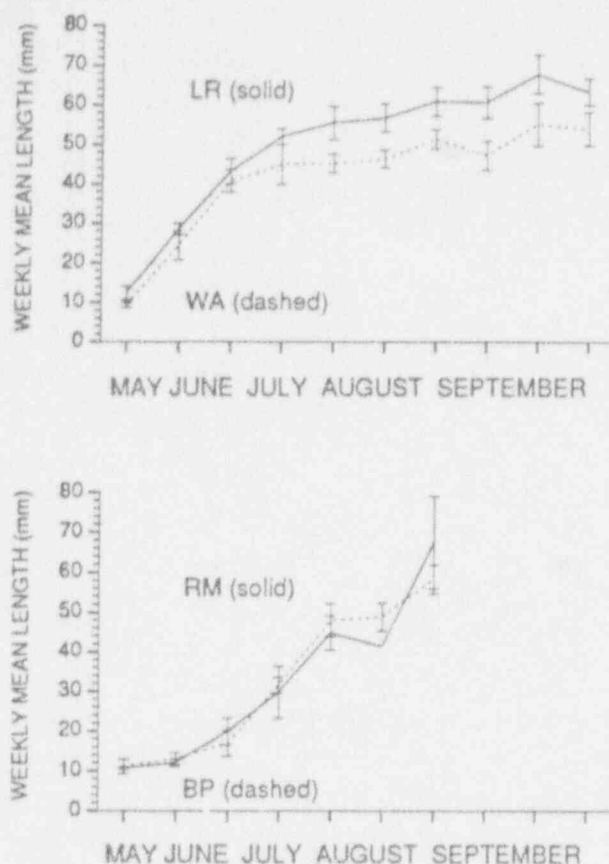


Fig. 30. Weekly mean length ( $\pm 2$  standard errors) of age-0 winter flounder taken in Niantic River and Bay during 1993.

production and its availability also likely differed among areas in the Niantic River and Bay and from year to year. Karakiri et al. (1989) reported differences in the size of young plaice of similar age between Wadden Sea estuarine nursery grounds (larger fish) and coastal waters off Germany (smaller fish). They suggested that the differences were due to lower water temperature, food limitation, or wave action in the waters outside of the Wadden Sea. Al-Hossaini et al. (1989) reported greater growth for cohorts of plaice that settled relatively early in Wales, but these fish also had higher mortality. Conversely, growth was slower for late-settling cohorts, but survival was higher.

**Mortality.** Catch curves constructed from weekly abundance data by year and station were used to obtain estimates of monthly instantaneous mortality rate ( $Z_{mo}$ ); this method assumed that young comprised a single-age cohort throughout the season. With some exceptions, the catch curves generally fit the data well with relatively high  $r^2$  values (Table 23). No estimates could be determined for LR and WA during the high abundance year of 1988 as slopes of these catch curves were not significantly different from zero and for WA in 1986 and 1993 because of considerable variation in weekly abundance during those years. The  $Z_{mo}$  estimate for station LR in 1993 was 0.377 (equivalent to a survival of 68.6%) and, except for 1988, was the lowest observed since 1983. Long-term averages of  $Z_{mo}$  at LR and WA were 0.634 and 0.553, respectively. Mortality estimates for Niantic

TABLE 22. Comparison of the mean lengths of age-0 winter flounder taken at stations LR and WA in the Niantic River during late July through September of 1983 through 1993.

Mean length* in mm for station and year:																				
66	61	<u>59</u>	<u>58</u>	<u>57</u>	<u>56</u>	55	<u>51</u>	<u>51</u>	<u>51</u>	<u>50</u>	48	<u>47</u>	<u>46</u>	45	<u>44</u>	<u>43</u>	<u>43</u>	<u>42</u>	<u>42</u>	<u>42</u>
LR	LR	LR	LR	LR	LR	WA	WA	LR	WA	WA	LR	WA	LR	LR	WA	LR	WA	WA	WA	WA
83	93	84	89	85	91	91	93	88	88	89	92	87	86	87	92	90	84	85	90	86
Difference between the late seasonal mean at LR as compared to that for WA:																				
Year		84		85		86		87		88		89		90		91		92		93
Difference in mm		16		15		4		-2		0		8		1		1		4		10

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

TABLE 23. Monthly instantaneous total mortality rate (Z) estimates as determined from catch curves for age-0 winter flounder taken at two stations in the Niantic River (LR and WA) from 1983 through 1993 and two stations in Niantic Bay (RM and BP) from 1988 through 1993.

Year	Station	Niantic River				Station	Niantic Bay			
		n <sup>a</sup>	slope <sup>b</sup>	Standard error	r <sup>2</sup>		n <sup>a</sup>	slope <sup>b</sup>	Standard error	r <sup>2</sup>
1984	LR	16	-0.129 **	0.017	0.80	BP	-	-	-	-
1985		15	-0.118 **	0.015	0.82		-	-	-	-
1986		15	-0.127 **	0.012	0.89		-	-	-	-
1987		15	-0.108 **	0.021	0.67		-	-	-	-
1988		19	NS	-	-		16	-0.405 **	0.031	0.92
1989		12	-0.154 **	0.022	0.84		11	-0.485 **	0.044	0.93
1990		13	-0.322 **	0.028	0.92		15	-0.412 **	0.035	0.92
1991		18	-0.140 **	0.016	0.82		12	-0.667 **	0.069	0.90
1992		18	-0.129 **	0.019	0.74		10	-1.046 **	0.103	0.93
1993		9	-0.087 *	0.028	0.57		7	-0.438 **	0.066	0.90
1985	WA <sup>c</sup>	16	-0.084 **	0.023	0.51	RM	-	-	-	-
1987		16	-0.139 **	0.016	0.84		-	-	-	-
1988		19	NS	-	-		18	-0.235 **	0.025	0.85
1989		13	-0.145 **	0.028	0.71		15	-0.413 **	0.035	0.91
1990		15	-0.235 **	0.028	0.84		13	-0.523 **	0.051	0.90
1991		18	-0.049 **	0.011	0.54		10	-0.557 **	0.131	0.69
1992		16	-0.112 **	0.009	0.91		9	-0.871 **	0.078	0.95
1993		10	NS	-	-		8	-0.382 **	0.079	0.80
<hr/>										
		Mortality (Z <sub>mo</sub> )		Survival (S <sub>mo</sub> )				Mortality (Z <sub>mo</sub> )		Survival (S <sub>mo</sub> )
1984	LR	0.560		57.1%		BP	-	-	-	
1985		0.512		59.9%			-	-	-	
1986		0.552		57.6%			-	-	-	
1987		0.469		62.6%			-	-	-	
1988		-		-			1.759		17.2%	
1989		0.669		51.2%			2.106		12.2%	
1990		1.398		24.7%			1.789		16.7%	
1991		0.608		54.4%			2.879		5.5%	
1992		0.560		57.1%			4.543		1.1%	
1993		0.377		68.6%			1.902		14.9%	
	Mean	0.634		53.1%		Mean	2.496		8.2%	
	SD	0.298				SD	1.085			
	CV	47%				CV	43%			
1985	WA <sup>c</sup>	0.363		69.9%		RM	-		-	
1987		0.604		54.7%			-		-	
1988		-		-			1.021		36.0%	
1989		0.630		53.3%			1.794		16.6%	
1990		1.021		36.0%			2.271		10.3%	
1991		0.213		80.8%			2.419		8.9%	
1992		0.486		61.5%			3.781		2.3%	
1993		-		-			1.660		19.0%	
	Mean	0.553		57.5%		Mean	2.158		11.6%	
	SD	0.277				SD	0.937			
	CV	50%				CV	43%			

<sup>a</sup> Weekly sampling during 1984-92 and biweekly sampling in 1993.

<sup>b</sup> Probability level that the slope of the catch curve differs from zero is shown:

NS - not significant ( $p > 0.05$ ), \* - significant at  $p \leq 0.05$ , \*\* - significant at  $p \leq 0.01$ .

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



River winter flounder were usually greater than the equivalent  $Z_{mo}$  value of 0.371 reported by Percy (1962) for the Mystic River, CT estuary, but were similar to various estimates (0.563 - 0.693-month<sup>-1</sup>) made for young plaice in British coastal embayments (Lockwood 1980; Poxton et al. 1982; Poxton and Nasir 1985; Al-Hossaini et al. 1989). Mortality of young was much greater in the bay than in the river. The 1993 estimates for  $Z$  of 1.902 at BP and 1.660 at RM were also below average (2.496, 2.158). However, except for RM in 1988, no fish have ever been found at the two bay stations by the end of summer. Even in 1988, densities at RM in late summer were only 10-15% of those in the river.

Because the mortality estimates were based on the disappearance of young at a station, it is possible that some of the decrease could have been due to off-station emigration as well as mortality (i.e., fish movements would have been indistinguishable from natural mortality). However, Saucerman and Deegan (1991) noted only limited (< 100 m) movements of marked young winter flounder from various areas of Waquoit Bay, MA, particularly during the first few months after settlement. Also using marked fish, Howell and Molnar (1993) reported only one fish that moved among three different sampling sites within New Haven Harbor, CT. Riley (1973) noted that young plaice moved very little from their location in shallow water during summer and individuals even returned to previous positions if displaced laterally or to deeper water. As noted above, sizes of age-0 winter flounder found in the river and bay differ noticeably. It was therefore assumed that an influx of many smaller fish from the bay likely would have decreased weekly means and increased the variability observed in the length-frequency distributions at the river stations, unless extraordinarily fast growth of emigrating fish occurred. This characteristic may be used to determine if there was any emigration of young from the bay into the river during summer, which could have accounted for a fraction of the observed loss of fish. Because weekly mean lengths of young have usually been somewhat larger at LR than at WA and at RM than at BP, lengths of fish at stations WA in the river and RM in the bay should show the smallest differences when the two areas are compared. At the start of sampling in May, fish in the river were about twice the size of those in the bay (Fig. 31). This large size differential was maintained between the two areas throughout most of the season. It was not until nearly the end of the sampling season at RM that

lengths of fish in the bay approach those at WA. However, the last few weekly means at RM were based on small sample sizes as only a few fish were found in the bay by then. The 95% confidence intervals for weekly means at each station were relatively small and did not show changes expected if many smaller fish joined the larger young already present at WA. Based on the length and abundance data, it is likely that few or no fish emigrate from the bay to the river during summer. In contrast, based on increases in density generally noted from late May through early June (Figs. 28 and 29), some recruitment to LR and WA may have occurred early in the season from other nearby areas of the river having similar potential for growth.

If no substantial emigration into the river took place, lower densities at the bay stations also could have resulted from movements into other areas of the bay. However, few young have been taken since 1976 by otter trawl or seine sampling in Niantic Bay or in other nearby offshore waters until late fall or winter (see following section), when many young winter flounder withdrew from the river in response to decreasing water temperature. At this time, winter flounder as small as 30 mm became available for capture by otter trawl sampling in LIS. Small juveniles are also caught from February through early April in the Niantic River during the adult population surveys (see Age-1 juveniles [late winter], below). If young were present at deeper stations in the bay during mid-summer, they should have been collected during TMP sampling. Thus, high natural mortality of young winter flounder in Niantic Bay is likely the reason for the decline in density following larval metamorphosis and settlement to the bottom and it is likely that few older juveniles are produced in the bay.

The reasons for high mortality of juvenile winter flounder after settlement have not been investigated. Predation by caridean shrimp (*Crangon* spp.) has been suggested as the cause of high mortality after settling for both winter flounder (Witting and Able 1993) and plaice (Lockwood 1980; van der Veer and Bergman 1987; Pihl 1990; van der Veer et al. 1990; Pihl and van der Veer 1992). Van der Veer et al. (1990) speculated that, in general, predation by crustaceans on juvenile flatfishes may be a common regulatory process. Witting and Able (1993) found that the size of age-0 winter flounder significantly affected their probability of predation by sevenspine bay shrimp (*Crangon septemspinosa*), with greatest risk found for smallest fish. Juveniles apparently outgrew predation

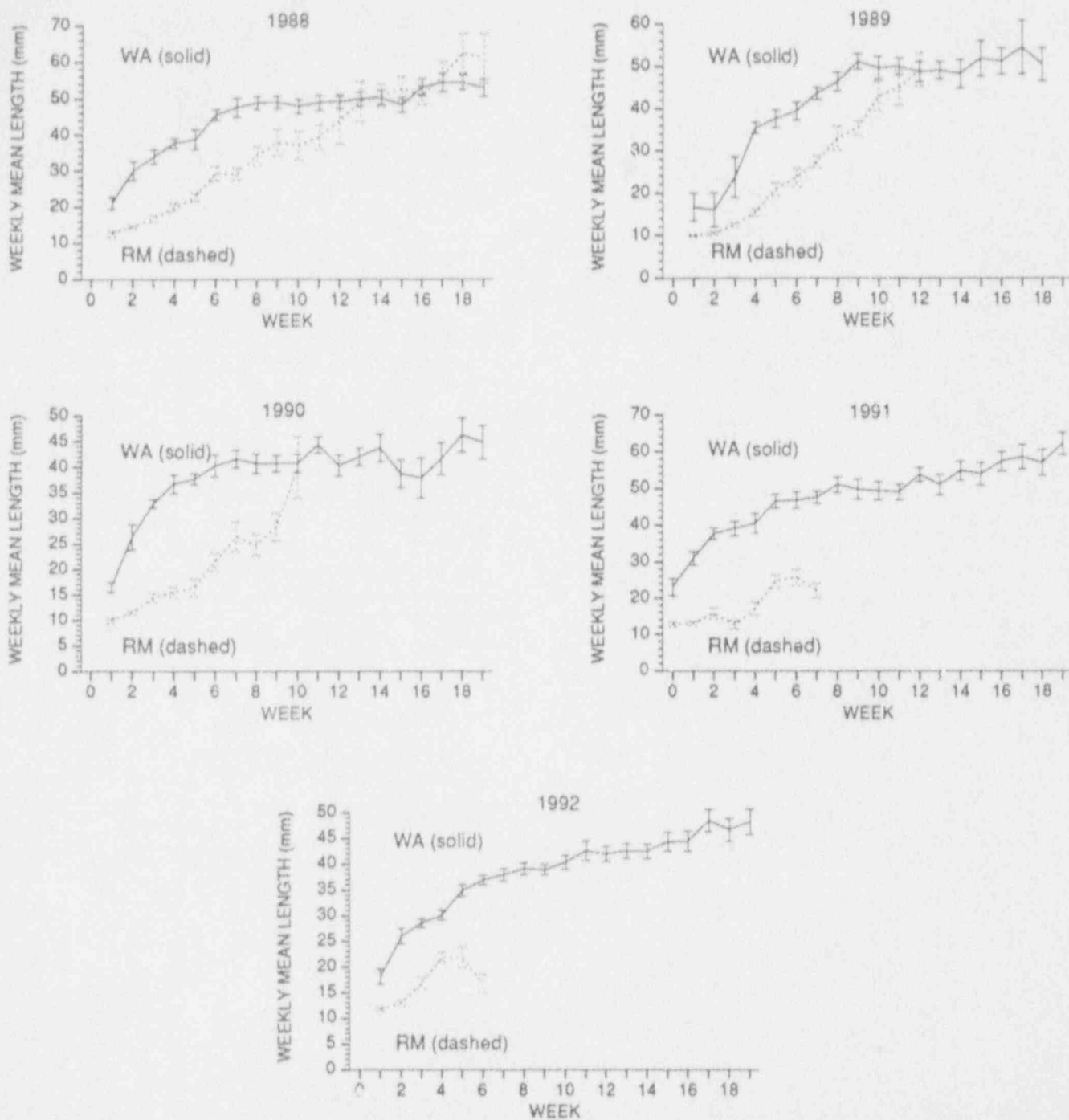


Fig. 31. Comparison of weekly mean length and 95% confidence interval of age-0 winter flounder taken at stations WA in the Niantic River and RM in Niantic Bay from 1988 through 1992. (Note differing scales among the graphs).

by shrimp when they reached 17 mm in length, which meant that they have to double in size after settlement before attaining a size refuge from shrimp predation. Therefore, the duration of time spent in a vulnerable size range, which is related to growth rate, affects the

vulnerability of young winter flounder to predation by shrimp and other organisms. Variation in growth, which can depend upon specific location of settling, specific habitat within a location, or temperature (Sogard 1990; Sogard and Able 1992) may have

significant implications for young winter flounder survival after metamorphosis. Greater mortality in the bay may be related to the smaller size of young and the greater numbers of fish available to prey upon them in these deeper water areas. This is analogous to reports of higher mortality for age-0 plaice in Europe, where survival in the shallow inland Wadden Sea of The Netherlands was greater than that of plaice inhabiting the more open and deeper British bays, which had more resident fish predators (Bergman et al. 1988).

Adult recruitment for many fishes is greatly affected by density-dependent processes occurring during the first year of life following the larval stage (Bannister et al. 1974; Cushing 1974; Sissenwine 1984; Anderson 1988; Houde 1989; Myers and Cadigan 1993a, 1993b). Bannister et al. (1974), Lockwood (1980), and van der Veer (1986) all reported density-dependent mortality for young plaice, although examination of their findings indicated that greatest rates of mortality occurred only when extremely large year-classes of plaice were produced (three to more than five times larger than average). A similar situation may have existed in the Niantic River as the peak ( $> 1\text{-m}^{-2}$ ) densities reached during early summer in 1990 and 1992 (Figs. 28 and 29) resulted in greater rates of decline than during other years. Although apparent survival rates were highest in 1988, when densities

were well above average, observed densities that year never approached  $1\text{-m}^{-2}$ .

### *Age-0 juveniles (late fall and early winter)*

Young winter flounder disperse from shallow waters near the shoreline to deeper waters as water temperatures decrease in fall and become available for sampling by the otter trawl used in the year-round TMP. Young are first regularly captured by trawl at the two shallower inshore stations (NR and JC) adjacent to inshore nursery grounds in November, the near-shore Niantic Bay stations (IN and NB) in December, and at the deeper-water stations in LIS (TT and BR) in January. A  $\Delta$ -mean (NUSCO 1988b) index of relative abundance was developed for these age-0 fish using TMP catch data, beginning with these months and continuing through the end of February. The November-February period was a transitional period following the beam trawl sampling of young in summer and preceding the catch of this cohort of fish as age-1 juveniles during the intensive adult winter flounder surveys that take place in the Niantic River from late February through early April. The most recent abundance index given in this report is for the 1992 year-class; the  $\Delta$ -mean for 1992-93 was 31.1 (Table 24).

TABLE 24. The late fall-early winter seasonal<sup>a</sup>  $\Delta$ -mean CPUE<sup>b</sup> of age-0<sup>c</sup> winter flounder taken at the six trawl monitoring stations in the vicinity of MNPS from 1976-77 through 1992-93.

Survey year <sup>c</sup>	Number of samples	Non-zero observations	$\Delta$ -mean <sup>b</sup>	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

<sup>a</sup> Data seasonally restricted to November-February for NR and JC, December-February for IN and NB, and January-February for TT and BR. A few indices differ from those given in NUSCO (1993) because the number of samples analyzed per year was standardized at 42.

<sup>b</sup> Catch per standardized tow of 0.69 km (see Materials and Methods of Fish Ecology section).

<sup>c</sup> For age-0 fish, the year-class is the same as the first year given.

Although having a wide confidence interval associated with it, this value is the largest of the 17-year series and is indicative of the strength of the 1992 year-class. Even though winter flounder abundance generally has been decreasing, the  $\Delta$ -means for the last 5 years were among the largest values recorded.

Since 1983, when data were first available from the beam trawl sampling, the fall-early winter  $\Delta$ -means were compared to 1-m beam trawl densities determined for late summer at stations LR and WA in the Niantic River (Fig. 32). These abundance indices were significantly correlated (Spearman's rank-order correlation coefficient  $\rho = 0.85$ ;  $p = 0.004$ ). However, there was no apparent relationship found between the median CPUE of winter flounder smaller than 15 cm taken in the Niantic River during the subsequent (late February-early April) adult winter flounder survey and TMP  $\Delta$ -means (Fig. 33). For the 1985 and earlier year-classes, more juvenile winter flounder were taken in the river than at the six TMP stations (five of which are outside of the Niantic River). But since the 1988 year-class was produced, consistently more fish have been taken by the TMP.

Relationships among abundance indices of juvenile winter flounder may have been obscured by differences in sampling gear and variations in fish behavior. Major biases in abundance estimation can arise from size selectivity of the gear, spatial distribution of individuals in relation to the gear, and behavior of fish

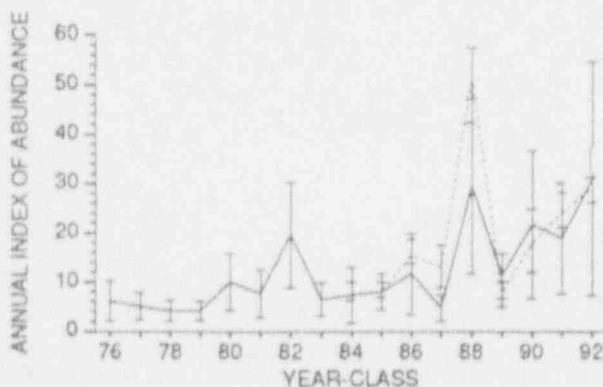


Fig. 32. Comparison between the late fall-early winter seasonal  $\Delta$ -mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the 1983-92 late summer Niantic River (stations LR and WA combined) age-0 1-m beam trawl median CPUE (dashed line) for the 1976-92 year-classes.

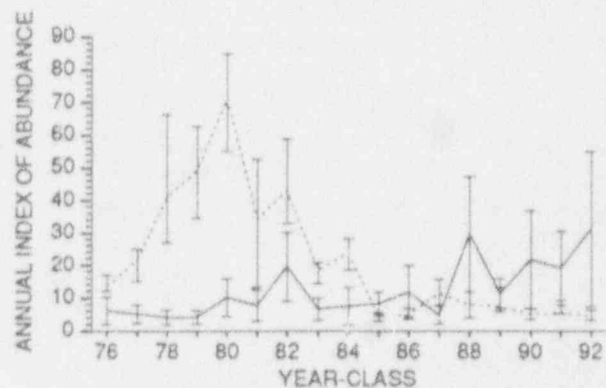


Fig. 33. Comparison between the late fall-early winter seasonal  $\Delta$ -mean CPUE (solid line) of age-0 winter flounder (all trawl monitoring program stations) and the Niantic River (stations 1 and 2) spawning survey median CPUE (dashed line) of winter flounder smaller than 15 cm for the 1976-92 year-classes.

in the vicinity of the gear (Parrish 1963). Mean lengths of age-0 winter flounder taken by otter trawl in fall were about 15 to 25 mm larger than those taken during the immediately preceding months by 1-m beam trawl. This size difference was greater than would have been expected from growth alone and suggests that CPUE indices were biased because smaller individuals were excluded from the catch. Differences in mean length by year (Table 22) may also have differentially biased the trawl sampling. The fixed locations of the otter trawl sampling stations in relation to the habitat available to juveniles also may have affected catches. Movements of small juveniles were probably influenced by factors such as water temperature and tide. Moreover, their availability to sampling gear in fall and winter appeared to have varied from week to week and year to year. Relatively large confidence intervals around the  $\Delta$ -means were probably a consequence of this variation. In contrast, variation was less in data collected during summer by the relatively efficient beam trawl. Furthermore, sampling in summer occurred weekly 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 source areas most likely occurs throughout LIS during the winter, and could also have influenced measures of abundance because they are dependent upon unknown contributions from different stocks.



### Age-1 juveniles (late winter)

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

The 1993 median CPUE for age-1 juveniles taken in the navigational channel of the lower Niantic River was 4.3, which was a decrease from the CPUE of 5.6 for 1992, but similar to the value of 4.9 for 1991 (Table 25). When tows from throughout the river were considered for the calculation, the median CPUE

was 1.9, marking the fourth straight year that relatively low values were recorded for this time-series (Table 26). Behavior of juvenile winter flounder largely influences their availability to sampling and apparently varies from year to year as a result of changing environmental conditions. Differential distribution (and, therefore, relative abundance) of small winter flounder has been noted to occur between Niantic Bay and River from year to year (NUSCO 1993). Trawl monitoring program (TMP) data from this spring, however, are not yet available for determining abundance of winter flounder in Niantic Bay during the 1993 adult spawning season. A  $\Delta$ -mean CPUE was computed for winter flounder smaller than 15 cm taken by the TMP from January through April at stations outside of the Niantic River. This time span overlapped the spawning period and also served to increase sample size. Values of this index were then compared to the CPUE median for fish found within the river during the spawning season (Fig. 34). Generally, the catch of age-1 winter flounder in the

TABLE 25. Annual 9.1-m otter trawl adjusted median CPUE<sup>a</sup> of winter flounder smaller than 15 cm<sup>b</sup> taken in the navigational channel of the lower Niantic River during the 1976 through 1993 adult population abundance surveys.

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

<sup>a</sup> Catch per standardized tow (see Materials and Methods).

<sup>b</sup> Mostly age-1 fish, so predominant age-class was produced 1 year before the survey year.

<sup>c</sup> Only tows of standard time or distance made in the navigational channel of the lower river were considered.

<sup>d</sup> Effort equalized among weeks.

<sup>e</sup> Zero for symmetrically distributed data.

<sup>f</sup> Because of low effort, data from the third week of sampling not used for the computation of CPUE.



TABLE 26. Comparison of annual 9.1-m otter trawl adjusted median CPUE\* of winter flounder smaller than 15 cm<sup>b</sup> taken in the navigational channel of the lower Niantic River with those caught throughout the entire sampling area of the river during the 1976 through 1993 adult population abundance surveys.

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

\* Catch per standardized tow (see Materials and Methods).

<sup>b</sup> Mostly age-1 fish, so predominant age-class was produced 1 year before the survey year.

<sup>c</sup> Effort equalized among weeks.

<sup>d</sup> Zero for symmetrically distributed data.

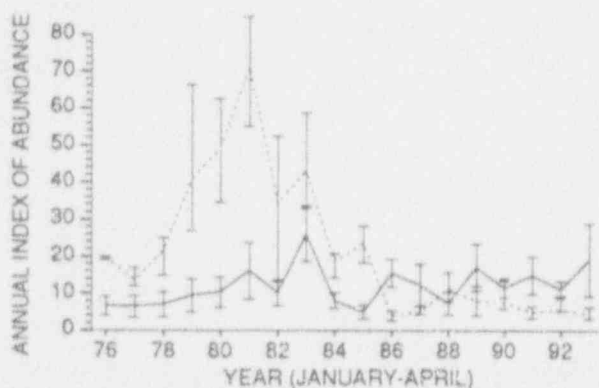


Fig. 34. Comparison between the annual January-April  $\Delta$ -mean CPUE (solid line) for all trawl monitoring program stations except NR and the Niantic River (stations 1 and 2) spawning survey median CPUE (dashed line) for winter flounder smaller than 15 cm from 1976 through 1992.

winter and early spring fluctuated less outside than inside the Niantic River. In 1988 catch within the

river was relatively high while catch outside the river was low. This trend reversed in 1989 as catch in the bay was the second highest of that series and showed that most juveniles did not remain within or re-enter the Niantic River. As the number of small fish in the river declined to low levels in recent years, the relative number outside the river increased. The CPUE of fish found in the bay remained greater than that for fish taken in the river from 1990 through 1993. A small CPUE index in the river may not necessarily indicate a continued decline in abundance, because even a relatively small increase in catch from the much larger geographical area of Niantic Bay and nearby LIS waters could account for a low abundance index in the river. As a result of the differential distribution and abundance of age-1 juveniles, perhaps as a result of variable environmental conditions influencing their behavior and availability to sampling, their abundance indices remain generally unreliable predictors of future adult population size.

## Comparisons among life-stages of winter flounder year-classes

Abundance indices for various life-stages of the 1976 through 1993 year-classes of winter flounder discussed throughout this report are summarized in Table 27. Coefficients of variation (CV) were used to examine annual variability in these abundance indices (Table 28). The CVs of all Niantic River winter flounder abundance indices increased over those given in NUSCO (1993). This was likely the result of near or all-time low abundance indices for spawners, egg production, larvae, and juveniles, and a series high for the fall-winter  $\Delta$ -mean for age-0 winter flounder. Variability was lowest (CV = 60%) for the number of females spawning in the Niantic River and for associated egg production (56%). For the first three adult female age-classes, variability decreased from age-3 (95%) through age-5 (74%). This likely reflected variation in recruitment of year-classes as well as the varying numbers of immature 3- and 4-year old fish present in the river each year. Miller et al. (1991) noted that interannual variability of many flatfishes appeared to decrease with age. Over all years, variability among larval stages was greatest for Stage 2 larvae

(107%). This was expected because much of the compensatory mortality is believed to occur during this stage of development. Late-age Niantic River winter flounder larvae and juveniles had CVs of 83 to 91%. Variability decreased to 71% during fall and early winter after young left shallow inshore waters. An increase in CV to 91% for age-1 juveniles in the Niantic River during the adult surveys was probably related to the previously discussed annual differences in distribution related to their behavior as well as from actual variation in year-class strength. Rothschild and DiNardo (1987) reported a median CV for recruitment indices of various marine fishes of 70%, although various flatfishes had CV values mostly less than 75%. The CV for abundance of European flounder decreased from 172% (n = 9) in the larval stage to 99% (n = 8) for newly settled young to 80% (n = 8, 12) for both young in September and at age-1 (van der Veer et al. 1991). As summarized by van der Veer (1986), the highest CV for yearly abundance estimates of different life stages of plaice in The Netherlands occurred during larval development in late winter (n = 4, CV = 95%) and at first settlement of pelagic juveniles in spring following larval metamorphosis and settling (9, 62%). Less variation was found for post-

TABLE 27. Comparison of indices of abundance for various life-stages of winter flounder for the 1976 through the 1993 year-classes.

Year-class	Adult indices		Larval indices				MNPS (EN)	Lower river (May-Jul)	Juvenile indices		Age-1 CPUE (Feb-Apr)
	Female spawners (Feb-Apr)	Annual egg production	Niantic River stations (Feb-Jun)						Lower river (Aug-Sep)	River/bay Δ-mean (Nov-Feb)	
			Stage 1 (3 mm)	Stage 2 (3.5 mm)	Stage 3 (6 mm)	Stage 4 (7.5 mm)					
76	-	-	-	-	-	-	854	-	-	6.1	13.5
77	884	394.6	-	-	-	-	567	-	-	5.1	21.6
78	1,412	717.5	-	-	-	-	754	-	-	4.2	41.0
79	1,120	535.3	-	-	-	-	641	-	-	4.2	49.3
80	903	424.3	-	-	-	-	845	-	-	10.1	71.1
81	2,669	1,383.1	-	-	-	-	561	-	-	7.7	4.4
82	2,752	1,596.8	-	-	-	-	610	-	-	19.6	43.0
83	1,869	1,082.0	-	749	408	56	1,215	32.7	10.0	6.6	18.5
84	871	501.6	2,601	1,501	573	67	917	18.8	6.3	7.4	23.6
85	928	565.2	6,260	4,676	584	35	312	13.3	7.0	8.1	4.1
86	655	436.7	1,279	176	301	24	510	33.8	13.8	11.7	5.0
87	852	531.6	3,218	829	1,036	48	315	59.2	17.9	4.8	11.2
88	1,279	866.9	14,491	4,469	1,531	210	419	61.3	60.0	29.6	7.9
89	984	716.2	12,453	3,976	589	73	327	17.5	8.8	11.3	7.4
90	579	370.4	4,728	355	258	57	508	156.3	20.0	21.7	4.9
91	1,061	639.6	3,248	252	343	112	439	77.5	21.7	19.0	5.6
92	534	391.1	5,476	1,367	2,339	195	1,003	90.0	28.1	31.1	4.3
93	274	223.6	1,117	133	111	6 <sup>a</sup>	130	10.6	5.0	-	-

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

TABLE 28. Coefficients of variation (CV) for annual abundance indices\* of various life stages of Niantic River winter flounder.

Life stage	Abundance index used	Number of observations	CV
Female spawners	Annual standardized catch	17	60%
Age-3 females	Annual standardized catch	15	95%
Age-4 females	Annual standardized catch	14	74%
Age-5 females	Annual standardized catch	13	67%
Eggs	Egg production index	17	56%
Stage 1 larvae	$\alpha$ parameter of Gompertz function	10	83%
Stage 2 larvae	$\alpha$ parameter of Gompertz function	11	107%
Stage 3 larvae	$\alpha$ parameter of Gompertz function	11	91%
Stage 4 larvae	$\alpha$ parameter of Gompertz function	11	83%
Age-0 young	Median CPUE at station LR (May-July)	11	85%
Age-0 young	Median CPUE at station LR (August-Sept)	11	87%
Age-0 young	Fall-winter $\Delta$ -mean at trawl stations	17	71%
Age-1 juveniles	Median CPUE of fish < 15 cm in Niantic River	17	91%

\* Indices used correspond to those given on Table 27, except for age-3 through age-5 females.

larval young during mid-summer (9, 30%) and for age-2 recruits (9, 35%). He attributed the decline in variation of abundance of older juveniles to a density-dependent regulatory mechanism that operated during and shortly after larval settlement. Both van der Veer (1986) and Bergman et al. (1988) noted that recruitment variability in plaice in The Netherlands was stabilized between years as a result of density-dependent regulatory processes (i.e., shrimp predation) on newly metamorphosed fish. The CVs for year-class strength of plaice in Swedish bays varied to a greater degree (67-118%), which was thought related to temperature effects during the larval stage and more variable crustacean predation on newly metamorphosed young in northern waters than in The Netherlands (Pihl 1990; Pihl and van der Veer 1992).

Relationships among abundance indices of winter flounder for the same year-class are of interest for impact assessment. Knowledge of the earliest possible measure of relative year-class strength is desirable because it enables predictions of future recruitment to the adult stock, thus providing an early warning of decreases in stock abundance. If indices for all life-stages are assumed to be accurately and precisely measured each year, they should be correlated after applying appropriate time lags, except when processes such as density-dependent mortality or size-selective fishing result in a lack of colinearity between two consecutive life-stages. With the addition of data from 1993, several life-stage abundance indices of Niantic River winter flounder became significantly correlated (Spearman's rank-order correlation) this year and

relationships between previously correlated indices were strengthened (Tables 29 and 30). Among these were female spawners and egg production, which was expected because calculation of the latter included female spawner abundance as part of the methodology of estimation. A significant positive correlation was also found this year between calculated egg production and yolk-sac (Stage 1) larvae. Significant correlations were found among most larval stage indices. The abundance of larvae 7 mm and larger collected at MNPS (station EN), however, was not significantly correlated with any of the juvenile abundance indices. Age-0 juvenile abundance during summer and late fall-early winter were also correlated (see Fig. 32). This year the late fall-early winter  $\Delta$ -mean abundance index was significantly negatively correlated with the abundance of age-1 juveniles taken in the Niantic River during the adult winter flounder surveys. The relationship between these two indices was shown previously (Fig. 33). The reasons for this inverse relationship and the change in patterns of abundance that occurred during the mid-1980s are unknown.

If catch indices were assumed to be representative of annual relative abundances, Niantic River winter flounder were found to be fully recruited only at about age-5 (NUSCO 1990). Thus, age-3 or age-4 fish probably should not be used as an index of year-class strength because it is likely that only a fraction of these fish occur on the spawning grounds each year. Furthermore, this fraction may vary from year to year. A few correlation coefficients between the abundance of female spawners at ages-3, 4, and 5 and those for

TABLE 29. Matrix of Spearman's rank-order correlations among various winter flounder spawning stock and larval abundance indices. Except for larvae 7 mm and larger taken at the MNPS intakes, all other indices refer to adults or larvae collected in the Niantic River.

Index*	Adult egg production	Stage 1 larvae	Stage 2 larvae	Stage 3 larvae	Stage 4 larvae	Larvae at MNPS intakes (≥7 mm)
Female spawners	0.9387 <sup>b</sup> 0.0001 ** 17	0.6000 0.0667 NS 10	0.4364 0.1797 NS 11	0.2727 0.4171 NS 11	0.4000 0.2229 NS 11	0.2378 0.3582 NS 17
Adult egg production		0.6727 0.0330 * 10	0.5000 0.1173 NS 11	0.4182 0.2006 NS 11	0.4000 0.2229 NS 11	0.0980 0.7082 NS 17
Stage 1 larvae			0.8061 0.0049 ** 10	0.6606 0.0376 * 10	0.6849 0.0289 * 10	-0.0303 0.9338 NS 10
Stage 2 larvae				0.7636 0.0062 ** 11	0.4546 0.1601 NS 11	-0.0909 0.7904 NS 11
Stage 3 larvae					0.6091 0.0467 * 11	0.0818 0.8110 NS 11
Stage 4 larvae						0.3636 0.2716 NS 11

\* Indices used correspond to those given on Table 27.

<sup>b</sup> The three statistics shown in each correlation matrix element are:

correlation coefficient (*r*),

probability of a larger *r* (NS - not significant [ $p > 0.05$ ], \* - significant at  $p \leq 0.05$ , \*\* - significant at  $p \leq 0.01$ ), and

number of annual observations (sample size).

7-mm and larger larvae at MNPS and some juvenile indices were significant (Table 31). The CPUE of age-1 fish taken in the river during the adult spawning surveys was significantly correlated with age-3 female spawners. However, the correlations were not significant for age-4 and 5 females. Significant negative correlations were found between both age-3 and age-4 females and the age-0 fall-winter  $\Delta$ -mean CPUE, but the weakest of the negative correlations (not significant) was found between age-5 spawners and the fall-early winter juveniles, which should have been one of the most reliable as females are fully recruited by this age. If negative correlations persist in future years, they could be interpreted as an indication of unknown processes operating after winter flounder become age-1 that result in fewer adults being recruited in spite of larger numbers of juveniles. Possibilities include variable discard mortality of juveniles in the commercial fishery; high rates of fishing; and non-random

fishing effort, which may occur in overfished stocks. Meanwhile, none of these life-stage indices can presently be used as a reliable measure of year-class strength.

### Stock-recruitment relationship (SRR)

**Sampling-based estimates.** Egg production estimates from annual spawning were used to determine recruitment because the abundance of the other early life-stages have not been reliably correlated with adult spawners. Both recruitment and the parental spawning stock indices were scaled to absolute population size as described previously (see Absolute abundance estimates, above). The resulting annual values were used with the Ricker SRR model as estimates of adult female spawning stock and potential female recruitment (Table 32). The addition of new catch data from the 1993 adult winter flounder survey and

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

Index <sup>a</sup>	Niantic River Stage 4 larvae	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles
MNPS intake larvae (≥7 mm)	0.3636 <sup>b</sup> 0.2716 NS 11	0.4727 0.1420 NS 11	0.3091 0.3550 NS 11	-0.1778 0.4948 NS 17	0.4485 0.0709 NS 17
Niantic River Stage 4 larvae		0.5727 0.0655 NS 11	0.6818 0.0208 * 11	0.6242 0.0537 NS 10	0.1030 0.7770 NS 10
Lower river early age-0 juveniles			0.8909 0.0002 ** 11	0.6485 0.0425 * 10	-0.2242 0.5334 NS 10
Lower river late age-0 juveniles				0.7212 0.0186 * 10	-0.2364 0.5109 NS 10
Fall-early winter river-bay age-0 juveniles					-0.5199 0.0324 * 17

<sup>a</sup> Indices used correspond to those given on Table 27.<sup>b</sup> 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).

TABLE 31. Matrix of Spearman's rank-order correlations among various winter flounder larval and female spawner abundance indices.

Index <sup>a</sup>	MNPS intake larvae (≥7 mm)	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles
Age-3 female spawners <sup>b</sup>	0.3607 <sup>c</sup> 0.1866 NS 15	0.0000 1.0000 NS 8	0.1191 0.7789 NS 8	-0.5952 0.0192 * 15	0.7124 0.0028 ** 15
Age-4 female spawners <sup>b</sup>	0.3758 0.1854 NS 14	0.1071 0.8152 NS 7	-0.1429 0.7599 NS 7	-0.8889 0.0001 ** 14	0.5121 0.0612 NS 14
Age-5 female spawners <sup>b</sup>	0.6154 0.0252 * 13	-0.2571 0.6228 NS 6	-0.3714 0.4685 NS 6	-0.4319 0.1405 NS 13	0.1813 0.5533 NS 13

<sup>a</sup> Early life history indices used correspond to those given on Table 27.<sup>b</sup> Determined by applying an age-length key (NUSCO 1989) to the length distribution of annual standardized female abundances.<sup>c</sup> 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).



TABLE 32. Annual Niantic River winter flounder stock-recruitment data based on indices of egg production for the 1977 through the 1989 year-classes with mean February water temperature and deviations ( $T_{Feb}$ ) from the mean.

Year-class	Index of female spawners (P)*	Index of female recruits (R)*	R/P ratio	Mean February water temperature ( $^{\circ}$ C)	Deviation from mean February water temperature ( $T_{Feb}$ )
1977	20,097	65,125	3.24	0.36	-2.12
1978	36,544	45,750	1.25	1.09	-1.39
1979	27,262	36,757	1.35	1.48	-1.00
1980	21,608	28,861	1.34	2.38	-0.10
1981	70,441	28,101	0.40	2.63	0.15
1982	81,324	32,708	0.40	1.56	-0.92
1983	55,104	35,874	0.65	3.74	1.26
1984	25,544	27,175	1.06	4.02	1.54
1985	28,789	25,666	0.89	2.36	-0.12
1986	22,241	24,062	1.08	3.38	0.90
1987	27,076	20,912	0.77	3.28	0.80
1988	44,153	8,812	0.19	2.67	0.19
1989	36,478	3,194	0.09	3.24	0.76
Mean	38,205	29,415	0.77	2.48	
CV	51%	53%		44%	

\* Scaled number of female spawners and recruits from expected egg production; scaling factors used were 561,000 eggs per females and a multiplier of 28.571 to convert relative abundance to an absolute population size. Indices of female spawners and recruits differ from those reported in NUSCO (1993) because of data added from the 1993 adult winter flounder population survey and changes in the values of Z used in the calculations.

changes in the values of Z used in the calculation of potential recruitment resulted in some differences between present estimates and those reported in NUSCO (1993).

The two-parameter SRR model (Eq. 6) was initially fitted to the spawner and recruit data. The stock growth potential parameter  $\alpha$  (scaled as numbers of fish) was estimated as 2.329, having a standard error of 0.964 (41% of the parameter value). The two-parameter model estimates were used as initial values for fitting the three-parameter SRR model with temperature effects (Eq. 7). This second fit produced an estimate for  $\alpha$  of 1.977, with a standard error of 0.566, which is 29% of the parameter value (Table 33). For the three-parameter model, the parameter  $\phi$ , which was an estimate of the effect of February temperature deviations ( $T_{Feb}$ ) from the 1977-89 mean of  $2.48^{\circ}$ C, was -0.412, an increase from the value of -0.357 reported in NUSCO (1993). The three-parameter SRR explained 51% of the variability associated with the recruitment index. Relationships resulting from fits of both Ricker models are shown in the central portion of Figure 35 as follows: the unadjusted SRR (two-parameter model; Eq. 7) is shown as the broken-line curve and the three-parameter curve (SRR adjusted

for the deviations from the 1977-89 mean February temperature of  $2.48^{\circ}$ C) is the solid-line curve. The outermost two dashed-line curves describe low recruitment in the warmest year (1984;  $T_{Feb} = +1.54$ ) and high recruitment in the coldest year (1977;  $T_{Feb} = -2.12$ ).

The  $\alpha$  values determined this year from both the two- and three-parameter models decreased from corresponding values reported during the past 4 years in NUSCO (1990, 1991b, 1992a, 1993). This variation could be caused by increasing fishing mortality rates on winter flounder in addition to the inherent instability of parameter estimates fitted to small data sets. In particular, the influence of the 1988 and 1989 data points on the estimate of  $\alpha$  were illustrative of higher recent exploitation (Fig. 35). Because of relatively high abundance of juvenile winter flounder from the 1988 year-class, numbers of females were expected to increase in 1992 and 1993 and form the bulk of the spawning population. However, these recruitment indices were much below expected values, likely the result of high fishing mortality rates in recent years.

The estimate of Ricker's  $\beta$  parameter, which describes the annual rate of compensatory mortality as a function of the stock size is important in SPDM

TABLE 33. Parameters of the Ricker stock-recruitment model fitted to data for Niantic River female winter flounder spawners from 1977 through 1989 and some derived points of reference.

Model parameters and reference points			
Model parameters (determined from numbers of fish):	Estimated value	Standard error	t <sup>a</sup>
$\alpha_0$ (compensatory reserve for unfished stock)	5.42	-	-
$\alpha$ (current compensatory reserve)	1.977 <sup>b</sup>	0.566	3.49 **
$\beta$ (stock-dependent compensatory rate)	$2.523 \times 10^{-5}$	$6.42 \times 10^{-6}$	3.93 **
$\phi$ (environmental [temperature] effect)	-0.412	0.1075	3.83 **
Derived points of reference:		Numbers of fish	Biomass (lbs)
Unfished stock equilibrium size ( $P_{\infty}$ ; called maximum spawning potential by Howell et al. 1992)		66,988 <sup>c</sup>	97,132
Present (through 1989) equilibrium size ( $P_{E(89)}$ )		27,015 <sup>c</sup>	27,015
F for $P_{E(89)}$ = 26,526 female spawners		1.01	-
Estimate of critical stock size (25% of maximum spawning potential)		-	24,283

<sup>a</sup> t-statistic for parameter estimate  $\neq 0$  with d.f. =  $n-3 = 10$ .

<sup>b</sup> Includes the effects of recent exploitation rates.

<sup>c</sup> Average weight of female spawner for unfished stock is 1.45 lbs and for current exploited stock is 1 lb.

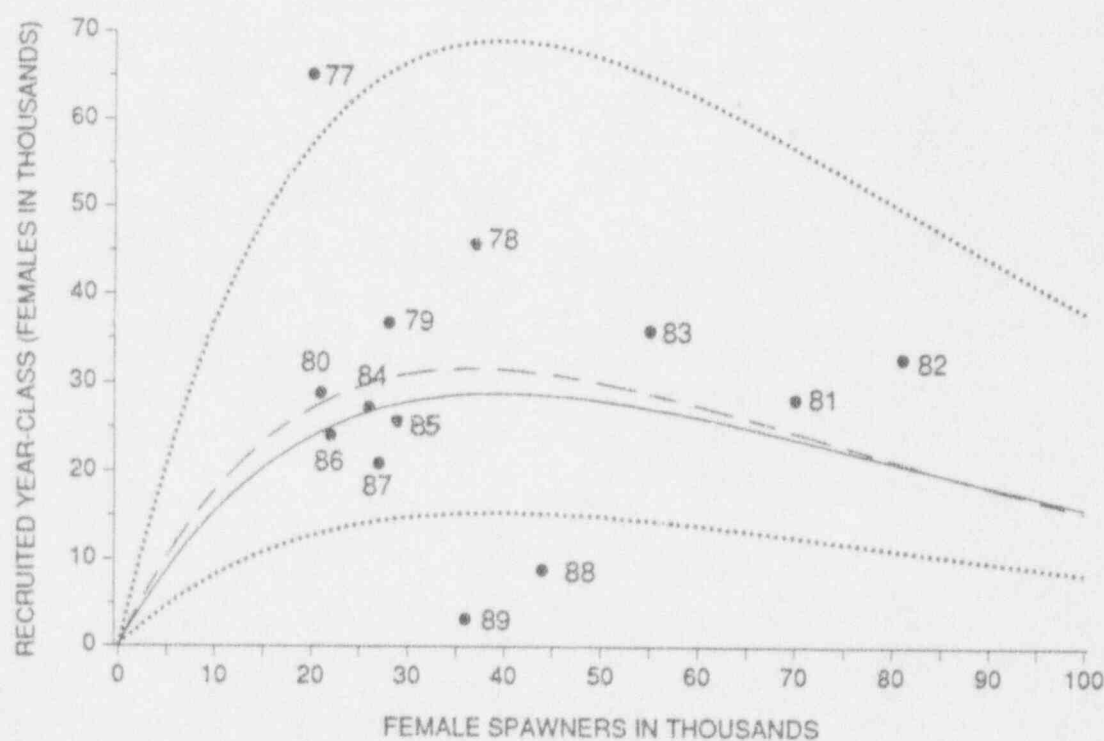


Fig. 35. The Ricker SRRs for Niantic River winter flounder (see text for explanation of the four curves plotted). Calculated recruitment indices for the 1977 through the 1989 year-classes are shown.

simulations. The parameter estimate of 0.0000252 has remained fairly consistent since 1988. The value for  $\phi$  (-0.412) was negative and although the relationship between winter flounder recruitment and February temperatures remain unknown, February coincides with most spawning, egg incubation, and hatching. These processes, as well as larval growth, are all temperature-dependent. Buckley et al. (1990) noted that the winter flounder reproductive process appears to have been optimized for cold winter temperatures followed by gradual spring warming. Adult acclimation temperatures and egg and larval incubation temperature affected larval size and biochemical composition. Cold winters and warm springs produced the largest larvae in the best condition at first feeding. This favored good survival and may partly explain the observed correlation between cold years and strong year-classes of winter flounder. Townsend and Cammen (1988) noted that the metabolic rates of pelagic consumers are more sensitive to lower temperature than rates of photosynthesis by phytoplankton, which bloom more in response to the amount of solar radiation received. Therefore, an earlier bloom in a cold year has the possibility of lasting longer before being grazed down by zooplankton. This allows for a greater contribution of organic matter to the benthos than in other years, benefiting juvenile demersal fishes that metamorphose just after the spring bloom of phytoplankton and have to outgrow various predators. As noted previously, the effect of temperature on potential prey or predators of larvae and newly metamorphosed juveniles, such as the sevenspine bay shrimp, may be an additional means for control of population abundance. Strong year-classes of plaice were also associated with cold winters, likely because the predatory brown shrimp (*Crangon crangon*) suffers high mortality or migrates out of plaice nursery areas (Zijlstra and Witte 1985; van der Veer 1988; Pihl 1990; Pihl and van der Veer 1992).

In addition to the above SRR parameters directly estimated from stock-recruitment data through 1988 (Fig. 35), Table 33 includes four derived biological reference points. Ricker's stock-at-replacement, or  $P_{rep}$  (Eq. 11), was estimated at 66,988 female spawners and is the unfished equilibrium spawning stock size, also known as the maximum spawning potential (MSP). This reference point, expressed in units of biomass as 97,132 lbs, is the basis for the critical stock size (25% of MSP) below which the stock is considered overfished (Howell et al. 1992). The present equilibrium size  $P_{E(F)}$  (Eq. 9) of 27,015

spawners refers to the sustainable or equilibrium size to which the stock could converge if present (through 1989) exploitation and other conditions remained unchanged. The calculated (Eq. 10) value of  $F$  that would achieve equilibrium stock size was 1.01, which is much higher than the DEP estimates of  $F$  for those years. This difference can mostly be due to the lack of age structure in Ricker's model, which causes fishing mortality to be concentrated in a single year for each year-class; winter flounder year-classes are exploited during many years. As mentioned previously in the Materials and Methods section, these reference points derived from fishery data are only deterministic approximations useful for comparative purposes across stocks and, in this study, to compare to the corresponding and more realistic values derived through simulation using SPDM.

#### Estimation of $\alpha$ for SPDM simulations.

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

on the basis of the following relationship:

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

where the mean weight was calculated for a population at equilibrium and one for which only natural mortality was assumed to have occurred (i.e., the unfished population). A mean weight of 1.45 lbs per female spawner for the Niantic River unfished winter flounder stock was calculated (Table 34) using population data previously reported (NUSCO 1990). Using this mean weight, the re-scaled  $\alpha$  parameter for this study was obtained as:

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

This parameter describes the inherent potential of a stock for increase because the natural logarithm of  $\alpha$  is the slope of the SRR at the origin for the unfished stock (Ricker 1954) and that slope, in turn, corresponds to the intrinsic rate of natural increase of the population (Roughgarden 1979). Since the slope of the SRR at the origin decreases with increasing exploitation rates, it is useful to think of  $\alpha$  as the "remaining growth potential" or "growth reserve" of the stock. Consequently, the large difference between the derived value of  $\alpha$  (5.42) and the direct regression estimate of  $\alpha$  (1.977; Table 33) reflects the different growth reserves of unfished and highly exploited stocks of winter flounder, respectively. Using an

unfished stock as a starting point for simulations also has other advantages, depending upon the particular scenario selected. For example, simulation in this report includes initially moderate fishing rates that are much lower than those affecting the data on which the regression estimate of  $\alpha$  was based. The data-based estimates of the other two SRR parameters ( $\beta$  and  $\phi$ ), which do not depend upon fishing and entrainment rates, were used in the population simulations as given in Table 33.

## MNPS impact assessment

### Larval entrainment

**Estimates of larval entrainment at MNPS.** The number of winter flounder larvae entrained in the condenser cooling water of MNPS is the most direct measure of potential impact on the Niantic River winter flounder stock. Annual totals of entrained larvae were related to larval densities in Niantic Bay and plant operations (cooling-water volume). Nearly all winter flounder larvae were collected at station EN from February through June, with most (> 90%) found in April and May. The entrainment estimate for 1993 (45.1 million) was the smallest after 1986, when three-unit operation began, and was among the smallest of the 18-year period since 1976 (Table 35).

TABLE 34. Biomass calculations for the Niantic River winter flounder female spawning stock at equilibrium based on an instantaneous natural mortality rate of  $M = 0.35$  and an instantaneous fishing mortality rate of  $F = 0$  (virgin stock).

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

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



TABLE 35. Annual abundance index ( $\alpha$  parameter of the Gompertz function) with 95% confidence interval of winter flounder larvae in entrainment samples and total annual entrainment estimates during the larval season of occurrence, and the volume of seawater entrained at MNPS each year from 1976 through 1993 during an 136-day period from February 15 through June 30.

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

Even though the seawater volume for 1993 was the largest reported during larval winter flounder occurrence, the low larval abundance in Niantic Bay accounted for this reduced entrainment estimate. As in previous years, Stage 3 larvae predominated in entrainment collections. In 1993, the percentages of each developmental stage entrained were 5% for Stage 1, 16% for Stage 2, 70% for Stage 3, and 9% for Stage 4, and were similar to previous years. Overall percentages for 1983-92 were 3% for Stage 1, 22% for Stage 2, 64% for Stage 3, and 11% for Stage 4 of development.

**Effect of entrainment on a year-class.** To determine the effect of winter flounder entrainment on a year-class, the relationship between entrainment estimates and various indices of juvenile abundance were examined. Annual entrainment estimates were significantly correlated with two abundance indices of juvenile winter flounder (Table 36). These were for age-0 fish taken in late summer at station LR and during late fall-early winter at the TMP stations. However, the abundance of young at LR in early summer, which immediately follows the larval entrainment season, was not significantly correlated with estimated entrainment. Although significant, the form of the relationships between the entrainment estimates and these two age-0 abundance indices was not obvious (Fig. 36). No significant correlations

were found between the density of winter flounder larvae 7 mm and larger taken at EN and the age-0 abundance indices (Table 30). Furthermore, the significant correlation coefficients were positive, implying no apparent entrainment effect. In general, even negative correlations between annual entrainment and abundance of early life history stages do not necessarily imply an entrainment impact unless positive correlations can be found between those early life history stages and mature female fish.

**Mass-balance calculations.** The magnitude of the impact of entrainment on the Niantic River winter flounder stock depends upon how many of the entrained larvae originated from this stock. Hydrodynamic modeling (NUSCO 1976) and current drogue studies (NUSCO 1992b) showed that much of the condenser cooling-water used by MNPS enters Niantic Bay from LIS. Other stocks are known to spawn both to the east and west of the bay and results from tidal studies also indicated that a number of winter flounder larvae enter Niantic Bay from LIS (NUSCO 1992a, 1992b). To determine if the number of winter flounder larvae leaving the Niantic River could support the number of larvae observed in Niantic Bay each year, mass-balance calculations were made for 1984 through 1993; eight of these years (1986-93) occurred during three-unit operation. The results determined for each 5-day period in 1993 are provided as an example of



TABLE 36. Spearman's rank-order correlations between the annual estimates of larval winter flounder entrainment at MNPS and the abundance indices of several post-entrainment early life history stages.

Index <sup>a</sup>	Lower river early age-0 juveniles	Lower river late age-0 juveniles	Fall-early winter river-bay juveniles	Niantic River winter-spring age-1 juveniles	Apparent larval survival rate
Annual estimate of entrainment	0.5273 <sup>b</sup> 0.0956 NS 11	0.6455 0.0320 * 11	0.5675 0.0175 * 17	-0.2514 0.3304 NS 17	-0.3237 0.2050 NS 17

<sup>a</sup> Indices used correspond to those given on Table 27, except for the apparent survival rate, which is the age-1 index divided by the index of 7 mm and larger larvae in Niantic Bay.

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

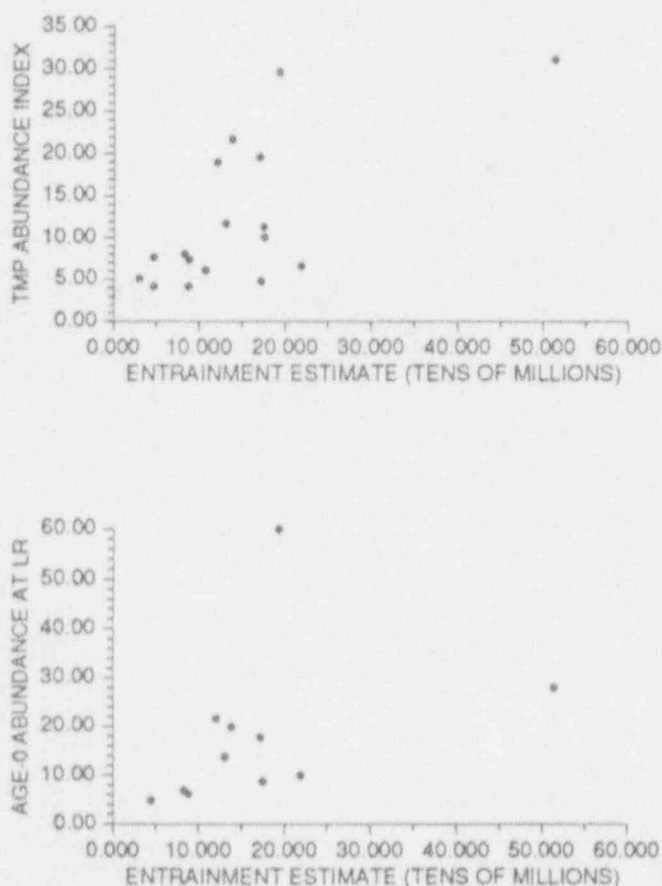


Fig. 36. Relationship between the annual entrainment estimate of winter flounder larvae at MNPS and the late fall-early winter seasonal  $\Delta$ -mean CPUE of age-0 winter flounder from all trawl monitoring program stations (TMP) for the 1976-92 year-classes, and between the entrainment estimate and the median CPUE of age-0 winter flounder taken at station LR in the Niantic River during late summer for the 1983-92 year-classes.

these calculations (Table 37). Results for other years were provided in NUSCO (1993).

During the 1993 larval season, the sign of the term *5-day change* shifted from positive to negative when the estimated number of larvae in Niantic Bay started to decline during a 5-day period beginning on May 6 (Table 37). Also, in late February the sign of the *Source or Sink* term changed from negative to positive. A negative *Source/Sink* term indicated a net loss of larvae from Niantic Bay during the first part of the larval season. During the 5-day period starting on about February 25, the *Source or Sink* term became positive, an indication that larvae from other sources (i.e., LIS) were required to support the change in larval abundance and balance the equation. The timing of this change in the *Source or Sink* term in 1993 was earlier than in previous years (NUSCO 1993) and may be due to the low larval abundance in the Niantic River (Table 14), which serves a source of larvae in the mass-balance calculation. During peak entrainment (April and May), fewer larvae were entrained than were imported from LIS, indicating that sources other than the Niantic River provided larvae found in Niantic Bay.

During each 5-day period the proportion of entrained larvae from the Niantic River was estimated from the ratio of larvae entering the bay from the river (*FromNR*) to the total input from both sources (*FromNR + Source or Sink*). This proportion was applied to the total number entrained in that 5-day period to estimate the number entrained from the Niantic River. During 5-day periods when there was a net loss (negative *Source or Sink* term) or when the proportion from the river was greater than one, all larvae entrained were assumed to have originated from

TABLE 37. Results of mass-balance calculations for each 5-day period in 1993.

Start of 5-day period	5-day change (X 10 <sup>6</sup> )	Number entrained ( <i>Ent</i> ) (X 10 <sup>6</sup> )	Loss due to mortality ( <i>Mor</i> ) (X 10 <sup>6</sup> )	Number from the Niantic River ( <i>FromNR</i> ) (X 10 <sup>6</sup> )	Number to the Niantic River ( <i>ToNR</i> ) (X 10 <sup>6</sup> )	Source or Sink (X 10 <sup>6</sup> )
2-15	0.0*	0.0	0.0	2.7	0.0	-2.7
2-20	0.0	0.0	0.0	4.3	0.0	-4.3
2-25	0.0	0.0	0.0	5.8	10.4	4.7
3-02	0.0	0.0	0.0	6.9	10.4	3.6
3-07	0.1	0.0	0.0	7.5	10.5	3.1
3-12	0.2	0.0	0.1	7.6	10.6	3.3
3-17	0.3	0.1	0.1	7.2	10.8	4.1
3-22	0.4	0.3	0.2	6.5	11.2	5.5
3-27	0.5	0.6	0.4	5.7	11.6	7.3
4-01	0.5	1.0	0.4	4.9	12.2	9.2
4-06	0.5	1.6	0.5	4.1	12.7	11.2
4-11	0.5	2.4	0.7	3.4	13.2	13.4
4-16	0.4	3.0	0.8	2.8	13.7	15.0
4-21	0.3	3.5	1.0	2.4	14.1	16.4
4-26	0.2	3.3	1.1	2.0	14.3	16.9
5-01	0.0	3.7	0.9	1.6	14.4	17.3
5-06	-0.1	3.8	0.7	1.4	14.4	17.4
5-11	-0.2	3.6	0.9	1.2	14.3	17.4
5-16	-0.2	3.3	0.7	1.0	14.1	16.8
5-21	-0.3	2.9	0.7	0.9	13.9	16.2
5-26	-0.3	2.6	0.7	0.8	13.6	15.7
5-31	-0.3	2.3	0.5	0.8	13.3	15.1
6-05	-0.3	2.0	0.5	0.7	13.0	14.5
6-10	-0.3	1.7	0.4	0.7	12.7	13.9
6-15	-0.3	1.4	0.4	0.6	12.5	13.4
6-20	-0.2	1.1	0.3	0.6	12.2	12.8
6-25	-0.2	1.0	0.3	0.6	12.0	12.5

\* Due to rounding, any zero value represents less than 50,000 larvae.

the Niantic River. This estimate was conservative, because the results of a dye study and larval dispersal modelling (Dimou and Adams 1989) showed that only about 20% of the water discharged from the Niantic River passed through MNPS during full three-unit operation. Estimates of annual total entrainment and the annual number entrained from the Niantic River were determined by summing over all 5-day periods. Based on mass-balance calculations for data collected in 1984-93, about 11 to 35% of winter flounder larvae entrained by MNPS originated from the Niantic River (Table 38). For 1993, the estimated number of larvae entrained that originated from the river was much lower than previously reported (NUSCO 1993), probably due to low abundance in the river and the low total entrainment estimate (Table 35).

The potential impact of larval entrainment to the population depends upon the age of each larva at the time it is entrained. Older individuals have a greater probability to contribute to year-class strength than younger ones. Therefore, the estimated number of each developmental stage entrained during each 5-day period was based on the proportion of each stage collected at station EN. By applying the proportion of entrainment attributed to the Niantic River ( $FromNR / [FromNR + Source\ or\ Sink]$ ), the number of larvae in each stage was allocated to each of the two sources (Niantic River or LIS) for every 5-day period. The annual number of each larval stage entrained from each source was estimated by summing over all 5-day periods (Fig. 37). Most of the Stage 3 larvae (the predominant stage entrained) originated from sources

TABLE 38. Estimates of total number of larval winter flounder entrained, number of larvae entrained from the Niantic River, and the percentage of total entrainment attributed to the Niantic River for 1984-93.

Year	Total entrainment ( $\times 10^6$ )	Niantic River larval entrainment ( $\times 10^6$ )	% entrainment attributed to the Niantic River
1984	88.1	31.0	35.2
1985	83.3	27.7	33.3
1986	130.6	25.5	19.5
1987	172.0	39.9	23.2
1988	193.3	39.0	20.2
1989	175.0	33.0	18.9
1990	138.8	36.3	26.2
1991	121.3	33.1	27.3
1992	513.9	79.1	15.4
1993	45.1	4.8	10.6

other than the Niantic River. As mentioned previously, some of the larger larvae from other areas may have entered the Niantic River during flood tides and caused the increased frequency noted in the larger size-classes (Fig. 18). Results from a special bay-wide sampling during 1991 (NUSCO 1992a) showed that in April and May, when about 75% of Stage 3 larvae were entrained, more larvae entered Niantic Bay from LIS east of Millstone Point and passed by the MNPS intakes during a flood tide than were flushed out of the bay to LIS during an ebb tide. Therefore, greater densities of Stage 3 larvae were expected at station EN during a flood tide than during an ebb tide. This was confirmed in NUSCO (1993), where significantly ( $p \leq 0.05$ ) greater Stage 3 densities found in April and May from 1983 through 1992 at station EN were from collections made during flood tides as compared to ebb tides.

**Estimated production loss from the Niantic River stock.** Estimates of larvae entrained by stage from the river were compared to annual abundance estimates for each larval stage in the Niantic River to determine the percentage of production loss from the Niantic River stock (Table 39). Estimates of Niantic River Stage 1 larvae entrained were calculated from daily abundance estimates (Eq. 3) at station C, following an evaluation presented in NUSCO (1993). This study indicated that entrainment sampling may underestimate Stage 1 larval abundance because of net extrusion. Based on dye studies (Dimou and Adams 1989), 20% of the Niantic River discharge passes through MNPS during full

three-unit operation. Therefore, for full three-unit operation, 20% of the daily density of Stage 1 larvae at station C were used as an estimate of Stage 1 larval entrainment from the Niantic River. During periods of reduced plant operation, estimates were proportionally reduced based on daily water volume use. Entrainment estimates for Niantic River Stages 2, 3, and 4 larvae were from the results of mass-balance calculations, which used entrainment sampling densities. The estimated percentage of the Niantic River winter flounder production entrained annually since 1984 ranged from about 4 to 21.1% and had a geometric mean of 8.5%. Thus, based on several special studies (NUSCO 1992a, 1992b) and the empirical mass-balance calculations, a large number of larvae entrained at MNPS likely came from areas other than the Niantic River.

The above mass-balance calculations were based on actual daily condenser cooling-water volumes. To determine the production loss for projected full (100% capacity) three-unit operations, these calculations were recomputed based on a maximum daily condenser cooling-water volume of  $11.1 \text{ million m}^3 \cdot \text{day}^{-1}$  (Table 39). To increase the time-series, three-unit operation was simulated to include 1984 and 1985, prior to Unit 3 start-up. Estimated annual percentages of the Niantic River winter flounder production that would have been entrained since 1984 under simulated three-unit operation ranged from 5.2 to 23.9% (geometric mean = 11.5%). These estimated annual reductions in year-class strength were used in impact assessment simulations with the SPDM as described below.

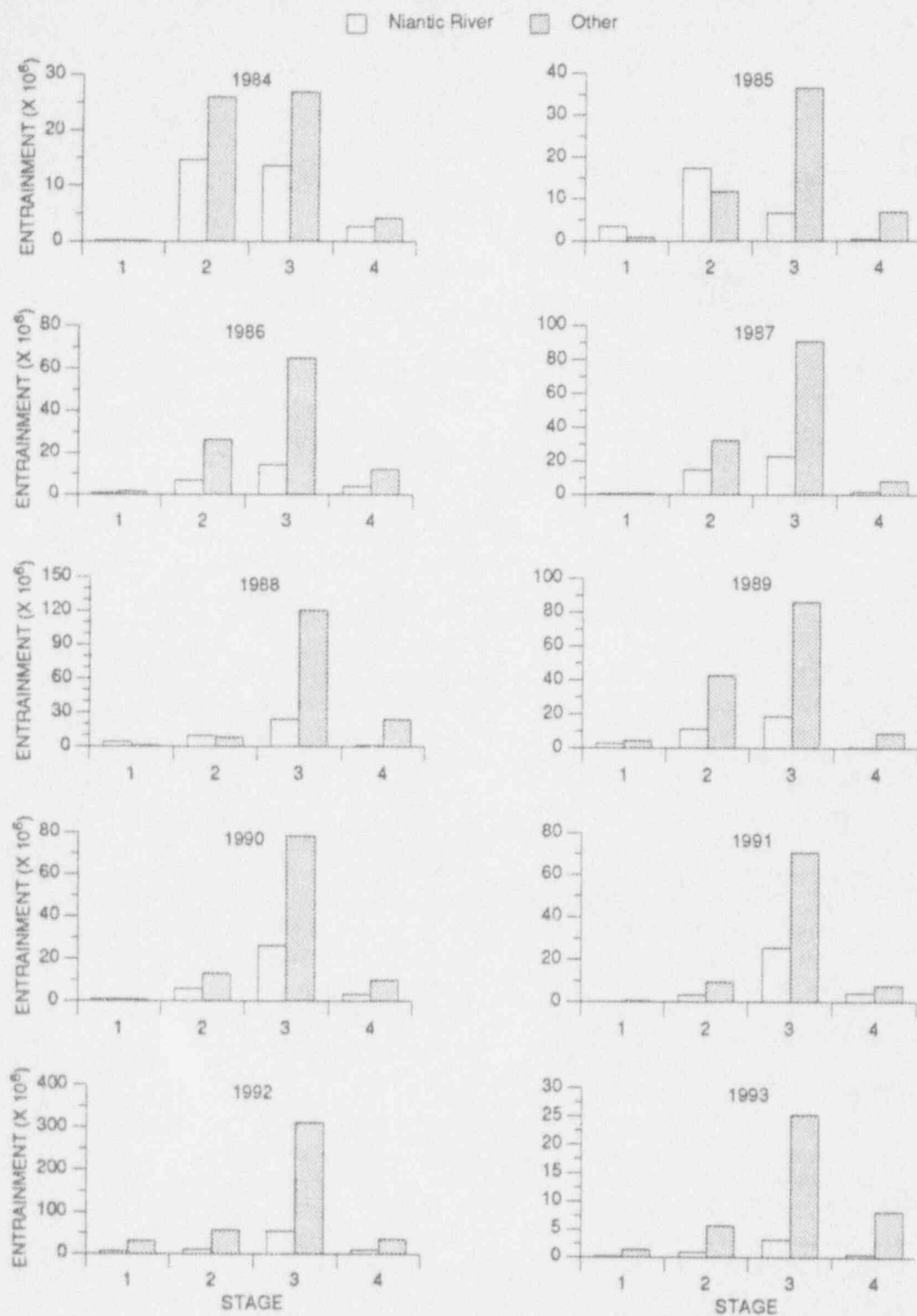


Fig. 37. Estimated number of winter flounder larvae entrained at MNPS by developmental stage from the Niantic River and other sources, based on mass-balance calculations for 1984 through 1993. (Note that the vertical scales differ among the graphs).

TABLE 39. Estimated abundance of winter flounder larvae in the Niantic River and the number and percentage of the production entrained from the Niantic River by developmental stage for 1984-93. Numbers of larvae from the Niantic River were based on the most recent mass-balance calculations.

Year	Stage of development	Niantic River abundance* (X 10 <sup>6</sup> )	Actual MNPS operating conditions:		Projected full MNPS three-unit operating conditions:	
			Entrainment from the Niantic River <sup>b</sup> (X 10 <sup>6</sup> )	% of the production	Entrainment from the Niantic River (X 10 <sup>6</sup> )	% of the production
1984 <sup>c</sup>	Stage 1	2864	10.3	0.4	22.6	0.8
	Stage 2	685	14.6	2.1	31.5	4.6
	Stage 3	337	13.5	4.0	33.2	9.9
	Stage 4	235	2.7	1.1	7.8	3.3
	Total		41.1	7.6	95.1	18.6
1985 <sup>c</sup>	Stage 1	3228	15.6	0.5	44.2	1.4
	Stage 2	773	17.3	2.2	43.2	5.6
	Stage 3	380	6.6	1.7	14.4	3.8
	Stage 4	265	0.3	0.1	8.8	0.3
	Total		39.8	4.6	102.6	11.1
1986	Stage 1	2494	11.6	0.5	14.4	0.6
	Stage 2	700	6.9	1.0	7.6	1.1
	Stage 3	366	14.2	3.9	14.8	4.0
	Stage 4	255	3.8	1.5	4.0	1.6
	Total		36.5	6.8	40.8	7.3
1987	Stage 1	3036	34.4	1.1	39.8	1.3
	Stage 2	853	14.7	1.7	17.1	2.0
	Stage 3	445	22.8	5.1	23.8	5.3
	Stage 4	311	1.7	0.5	1.8	0.6
	Total		73.6	8.5	82.5	9.2
1988	Stage 1	4951	83.7	1.7	92.1	1.9
	Stage 2	741	9.5	1.3	9.9	1.3
	Stage 3	267	24.1	9.0	25.9	9.7
	Stage 4	192	1.4	0.7	1.4	0.7
	Total		52.8	12.7	129.3	13.6
1989	Stage 1	4091	66.5	1.6	84.3	2.1
	Stage 2	570	11.0	1.9	13.9	2.4
	Stage 3	188	18.7	9.9	23.1	12.3
	Stage 4	126	0.4	0.3	0.6	0.5
	Total		96.6	13.8	121.9	17.3
1990	Stage 1	2115	33.2	1.6	36.7	1.7
	Stage 2	869	5.9	0.7	7.1	0.8
	Stage 3	239	26.2	11.0	29.8	12.5
	Stage 4	206	3.3	1.6	3.8	1.8
	Total		68.6	14.8	77.4	16.9
1991	Stage 1	3653	8.0	0.2	13.0	0.4
	Stage 2	2549	3.4	0.1	4.8	0.2
	Stage 3	775	25.3	3.3	33.4	4.3
	Stage 4	628	4.2	0.7	5.5	0.9
	Total		40.9	4.3	56.7	5.7



TABLE 39. (cont.).

Year	Stage of development	Niantic River abundance <sup>a</sup> (X 10 <sup>6</sup> )	Actual MNPS operating conditions:		Projected full MNPS three-unit operating conditions:	
			Entrainment from the Niantic River <sup>b</sup> (X 10 <sup>6</sup> )	% of the production	Entrainment from the Niantic River (X 10 <sup>6</sup> )	% of the production
1992	Stage 1	2234	23.0	1.0	28.6	1.3
	Stage 2	936	10.3	1.1	11.6	1.2
	Stage 3	344	54.7	15.9	61.5	17.9
	Stage 4	276	8.5	3.1	9.8	3.5
	Total		96.5	21.1	111.5	23.9
1993	Stage 1	1277	11.7	0.9	13.3	1.0
	Stage 2	660	0.9	0.1	1.2	0.2
	Stage 3	119	3.2	2.7	3.9	3.3
	Stage 4	83	0.5	0.6	0.6	0.7
	Total		16.3	4.3	19.0	5.2
Geometric mean				8.5		11.5

<sup>a</sup> Abundance estimates for 1984-89 were from Crecco and Howell (1990) and those for 1990-93 were calculated by NUSCO staff.

<sup>b</sup> Entrainment estimates attributed to the Niantic River are higher than those in Table 38 due to adjustments made for Stage 1 entrainment.

<sup>c</sup> Although only MNPS Units 1 and 2 operated in 1984 and 1985, the projected values assume full three-unit operation for all years.

### *Stochastic simulation of the Niantic River winter flounder stock*

**Model simulation of MNPS impact.** The initial input data used to run the SPDM were described in the Materials and Methods section (Tables 1-5; Figs. 5-7). The model accessed a secondary input file, which included fishing (plus impingement) rates and the larval entrainment losses assumed for each year of the simulation (Table 40). The combined mortality of fishing (F) and impingement (IMP) was used only during the simulation period (1971-2025) that corresponded to MNPS operation. Rates of larval entrainment (ENT) for 1971-93 were based on actual MNPS cooling-water flow and entrainment rates for Niantic River winter flounder larvae during three-unit operation as derived from the mass-balance calculations discussed above. Entrainment rates shown in Table 40 for 1994 through 2025 were based on randomized values of ENT and nominal plant cooling-water flow, which also depended upon the number of units remaining in operation in a particular year. The values used in simulation years after 1993 were generated by randomly re-sampling those calculated through 1993 and adjusting them to account for varying fractional flows for each unit and season. For this process, it was assumed that MNPS would operate during

a larval winter flounder season as it had in the past. Expected changes in the values of F over time were determined after consultation with DEP Marine Fisheries (V. Crecco and P. Howell, CT DEP, Old Lyme, CT, pers. comm.) and reflect recent changes in regulations to considerably reduce F (see Table 2).

**Simulation results.** The stochastic baseline generated for impact assessment purposes describes the female spawning stock sizes and annual variability of Niantic River winter flounder since 1960 and includes only the effect of fishing. Because the baseline stock projection included no power plant effects, it was used as the reference against which the impacted stock projection was compared. Therefore, the baseline needed to be a fair representation of past and projected trends of the local winter flounder abundance. According to the simulation schedule (Table 40), nominal fishing rates started at  $F = 0.40$ , remained unchanged through the 1960s, increased gradually to 0.62 in 1988, and thereafter increased more rapidly to a maximum rate of 1.31 in 1991 (see also Fig. 6). Note that the tabled rates included an additional mortality equal to 0.01 that accounted for fish impingement during years of MNPS operation. After 1991 and as a result of proposed regulatory changes to the commercial fishery, F was projected to decrease fairly substantially through the late 1990s, reach a low of 0.50 in 2001,

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

Time step	Simulation year	% of year-class reduction based on calculated or simulated levels of ENT <sup>a</sup>	Nominal F (plus IMP) <sup>b</sup>	Age-1	Fractional fishing discard F for:			
					Age-2	Age-3	Age-4	
0	1960	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
1	1961	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
2	1962	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
3	1963	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
4	1964	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
5	1965	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
6	1966	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
7	1967	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
8	1968	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
9	1969	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
10	1970	0.0	0.40	0.0360	0.2400	0.4000	0.4000	
11	1971	0.1530 X ENT value	0.41	0.0360	0.2400	0.4000	0.4000	
12	1972	0.2262 X ENT value	0.42	0.0369	0.2460	0.4100	0.4100	
13	1973	0.0767 X ENT value	0.43	0.0378	0.2520	0.4200	0.4200	
14	1974	0.1895 X ENT value	0.44	0.0387	0.2580	0.4300	0.4300	
15	1975	0.2262 X ENT value	0.45	0.0396	0.2640	0.4400	0.4400	
16	1976	0.4421 X ENT value	0.46	0.0405	0.2700	0.4500	0.4500	
17	1977	0.4232 X ENT value	0.47	0.0414	0.2760	0.4600	0.4600	
18	1978	0.3018 X ENT value	0.48	0.0423	0.2820	0.4700	0.4700	
19	1979	0.3133 X ENT value	0.49	0.0432	0.2880	0.4800	0.4800	
20	1980	0.4810 X ENT value	0.51	0.0450	0.3000	0.5000	0.5000	
21	1981	0.2873 X ENT value	0.53	0.0468	0.3120	0.5200	0.5200	
22	1982	0.4857 X ENT value	0.56	0.0330	0.2695	0.5500	0.5500	
23	1983	0.4675 X ENT value	0.58	0.0342	0.2451	0.5700	0.5700	
24	1984	7.6	0.60	0.0354	0.2537	0.5900	0.5900	
25	1985	4.6	0.61	0.0360	0.2160	0.6000	0.6000	
26	1986	6.8	0.63	0.0372	0.2232	0.6200	0.6200	
27	1987	8.5	0.64	0.0378	0.2268	0.6300	0.6300	
28	1988	12.7	0.63	0.0372	0.1116	0.6014	0.6200	
29	1989	13.8	0.86	0.0510	0.1530	0.8245	0.8500	
30	1990	14.8	1.04	0.0618	0.1854	0.9991	1.0300	
31	1991	4.3	1.31	0.0780	0.2340	1.2610	1.3000	
32	1992	21.1	1.18	0.0702	0.2106	1.1349	1.1700	
33	1993	4.3	1.18	0.0702	0.1170	0.9594	1.1232	
34	1994	U1, U2, U3 flow X ENT value	1.00	0.0594	0.0693	0.5445	0.8910	
35	1995	U1, U2, U3 flow X ENT value	0.77	0.0456	0.0532	0.4180	0.6840	
36	1996	U1, U2, U3 flow X ENT value	0.75	0.0444	0.0518	0.4070	0.6660	
37	1997	U1, U2, U3 flow X ENT value	0.70	0.0414	0.0483	0.3795	0.6210	
38	1998	U1, U2, U3 flow X ENT value	0.65	0.0384	0.0448	0.3520	0.5760	
39	1999	U1, U2, U3 flow X ENT value	0.60	0.0354	0.0413	0.3245	0.5310	
40	2000	U1, U2, U3 flow X ENT value	0.55	0.0324	0.0378	0.2970	0.4860	
41	2001	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
42	2002	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
43	2003	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
44	2004	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
45	2005	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
46	2006	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
47	2007	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
48	2008	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
49	2009	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
50	2010	U1, U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
51	2011	U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	
52	2012	U2, U3 flow X ENT value	0.51	0.0300	0.0350	0.2750	0.4500	

TABLE 40. (continued).

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

<sup>a</sup> For 1971-83 and 1994-2025, ENT values were randomly selected from projected rates determined from mass-balance calculations for full three-unit operation during 1984-93 (Table 39). Actual MNPS flow values were used for 1971-83 and randomly selected values from Table 5 were used for 1994-2025. ENT values for 1984-93 were estimates made under actual MNPS operating conditions as shown on Table 39.

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

and remain unchanged throughout the rest of the simulation time period. The unfished stock size used initially in the simulation was 97,075 lbs (equivalent to approximately 66,950 female spawners). Based on the age and size structure of female winter flounder used in the SPDM, this value was similar to the deterministic estimate of  $P_{rep} = 97,132$  lbs given in Table 33. The initial stock size represents the MSP for the unfished Niantic River female spawning stock. However, the stochastic mean size of the exploited stock by 1970 (under the starting nominal fishing rate of  $F = 0.40$ ) was reduced to 48,271 lbs. The simulated baseline responded as expected to the high rates of fishing through 1991 as the stock steadily declined to 12,312 lbs in 1993 (the solid line in Figs. 38A and 39). This biomass is about half of the critical stock size (defined as a stock biomass equal to 25% of the MSP) of 24,269 lbs, shown as the dashed line in Figures 38A and B. This reference stock size will be discussed in greater detail below. Even allowing for natural variation in the simulation, maximum replicate values of stock sizes for several years were below 25% of the MSP and minimum values were as small as 8.4% of MSP. The simulation illustrated that the baseline population could fall below the critical stock size at any time from 1985 through 1999. However, if reductions in  $F$  work as planned the stock should recover rapidly following its nadir in 1993.

To determine the effect of MNPS on the Niantic River female spawning stock, the baseline time-series may be compared to the impacted time-series, shown in Figure 38B and as the dashed line (ENT + IMP) in Figure 39. The impacted series corresponds to projections of the same initial stock as the baseline, but with additional annual losses due to MNPS operation. In this impacted population projection, the stock did not respond to larval losses due to entrainment until 1974 (the fourth year of Unit 1 operation), when biomass began to decline below baseline levels (Fig. 39). The lowest projected stock biomass (10,562 lbs) was reached in 1993, whereas the greatest absolute decline relative to the baseline occurred in 2001 (a difference of 7,840 lbs), when the effects of reductions in  $F$  since 1994 were propagated through the spawning population. From this point on, biomass of the impacted stock generally paralleled that of the baseline and began to approach it as MNPS units went offline. The impacted stock moved to within about 600 lbs of the baseline in 2031, 6 years after the end of Unit 3 operation in 2025, and became virtually identical to it by 2033.

The different nature of stock reductions caused directly by fishing and impingement and those result-

ing from larval losses through entrainment at MNPS is related to the age structure of the spawning stock. Fishing reduces biomass of the stock at a greater rate than it reduces the number of spawners because it tends to select for larger fish and, thus, reduces the average weight of the spawners remaining in the population. However, the most important difference between fishing (with an added component accounting for impingement) and larval entrainment is that while the former process removes fish from each year-class every year for as long as any fish remain, the latter causes a reduction only once in the lifetime of each generation and, then, very early in the life history of a species. The relative effects of stock reductions due to fishing and MNPS impact can be assessed by comparing the unfished stock projection line to those for the fished stock with and without plant effects (Fig. 40). The majority of biomass reductions are due to fishing. Therefore, as fishing mortality is reduced throughout the 1990s and the baseline biomass increases rapidly, the absolute annual losses due to MNPS impact (i.e., the gap between the baseline and impacted losses in Fig. 39) become larger. However, these losses represent increasingly smaller fractions of a much larger baseline.

Stock sizes projected for each simulation scenario at seven selected points in time are given in Table 41; losses relative to the theoretical unfished stock for each particular year are shown as percentages. Minimum and maximum stock sizes representing the range of stock sizes for the 100 Monte Carlo replicates generated for each year (shown on Fig. 38) are also given. The theoretical unfished stock in each of the 7 years shown varied little and averaged about 102,000 lbs for each year. Prior to MNPS operation in 1970, the baseline and the impacted stocks were identical (48,271 lbs) and made up about 47% of the unfished stock. By 1990, winter flounder spawning stocks under full MNPS three-unit operation declined by about 50% relative to 1970. However, this was mostly the result of increased fishing as the impacted stock was only about 1,900 lbs less than the baseline. Smallest stock sizes were found in 1993 as a result of high rates of exploitation during the early 1990s. The baseline and impacted stocks were only 12.3% and 10.5%, respectively, of the unfished stock. By 2000, the baseline stock had responded more rapidly to decreased fishing than the impacted stock. The gap between the two began to narrow in 2010 and 2020 and, as stated previously, were nearly indistinguishable by 2031, when the stock can be considered as fully recovered.



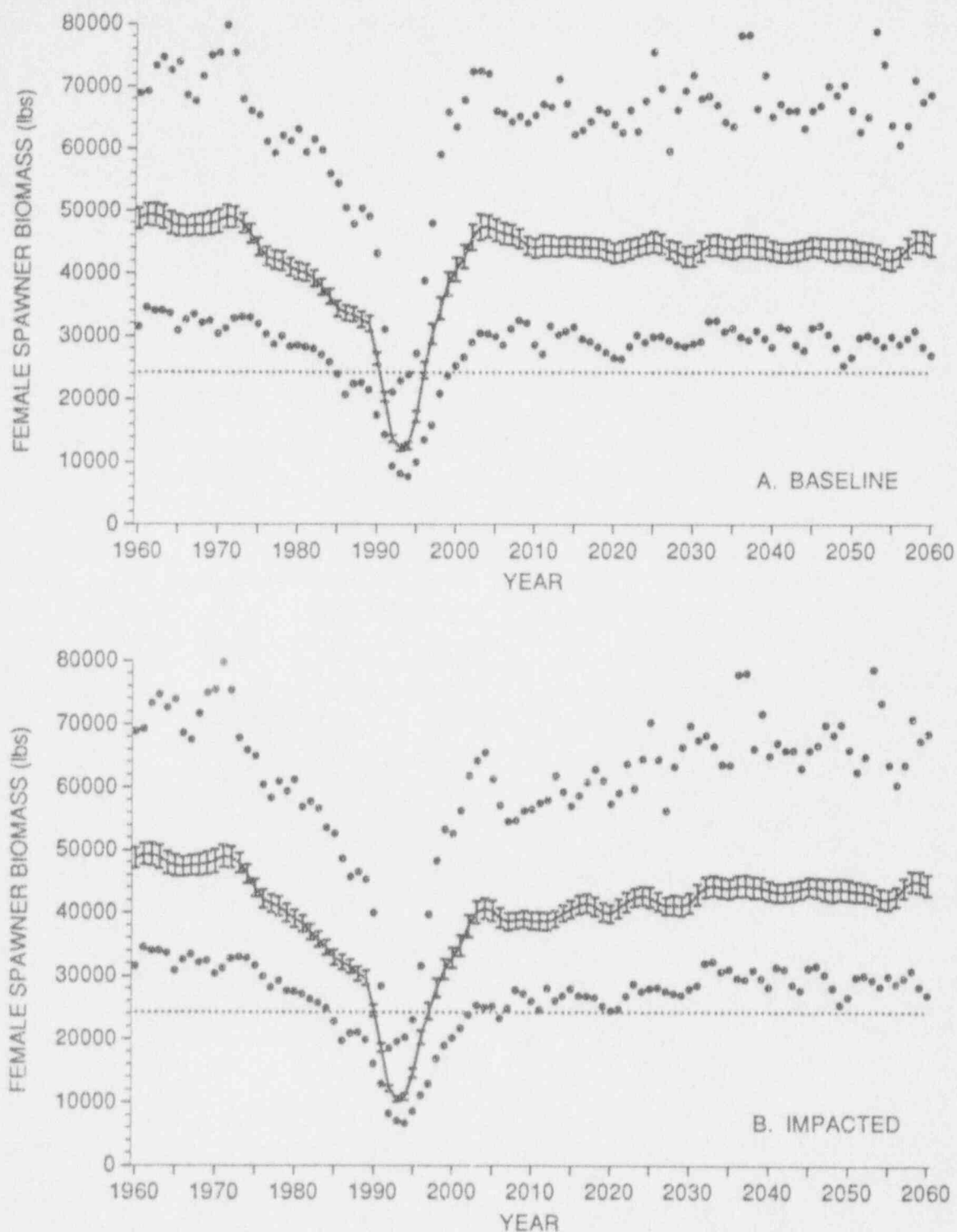


Fig. 38. Stochastic variability associated with the projected Niantic River female winter flounder stock expressed as biomass (lbs) for: A. The baseline stock with simulated sport and commercial fishing rates, but no effects from MNPS operation, and B. The impacted stock with both fishing effects and MNPS impact. The solid lines are the geometric means and 95% confidence interval (100 Monte Carlo replications) of each stock size trajectory and are equal to the baseline and impacted stocks illustrated on Figure 39. The symbols above and below the line correspond to the largest and smallest stocks among the 100 replicates generated for each year. The dashed line represents the critical stock size (here, 24,269 lbs), defined as when stock biomass has been reduced to 25% of the maximum spawning potential (Howell et al. 1992).



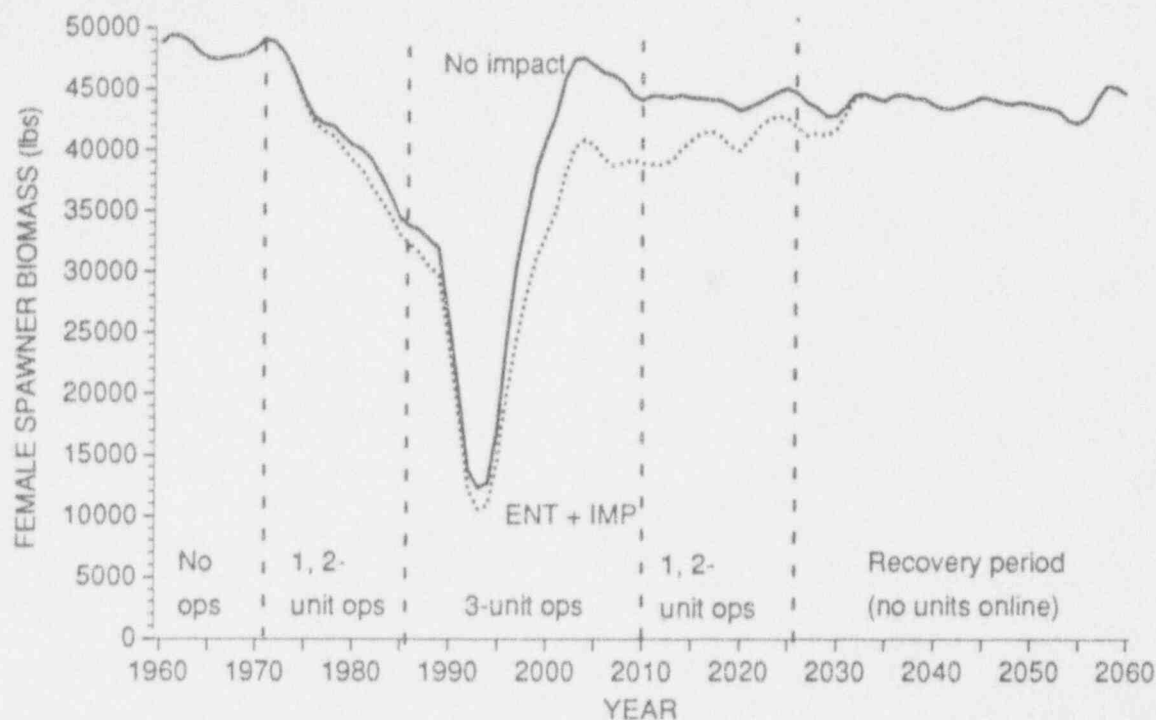


Fig. 39. Results of the SPDM simulation showing the combined effects of fishing and calculated larval entrainment rate and fish impingement rates (dashed line labeled ENT + IMP) on the biomass of the Niantic River female winter flounder spawning stock. Entrainment rates changed annually according to the number of MNPS units in operation and fishing rates were also variable (see text and Table 40 for details). The solid line labeled "No impact" is the baseline with fishing effects only. All stock sizes are averages of 100 Monte Carlo replicates.

**Probabilistic assessment of MNPS effects.** The stochastic variability associated with stock projections for combined fishing, impingement, and larval entrainment illustrated in Figure 38B formed the basis for probabilistic analyses. These analyses took into account not only the mean stock biomass predicted for each year, but also the frequency distribution of 100 replicate predictions both smaller and larger than the mean. To assess effects of MNPS operation, the probabilities that the Niantic River female winter flounder spawning stock would fall below selected reference sizes were determined. These reference sizes were percentages (25, 30, and 40%) of the biomass of spawning females for the unfished stock (i.e., the MSP) and were suggested in the Atlantic States Marine Fisheries Commission management plan for inshore stocks of winter flounder (Howell et al. 1992). In eastern LIS, values of  $F$  ranging from 0.37 to 0.68 would be necessary to achieve maximum yield, depending upon various combinations of length (10, 11, or 12 inches) and trawl codend mesh (3.5, 4.5, 5.0, or 5.5 inches) restrictions imposed on the

commercial fishery. A stock that has been reduced to less than 25% of the MSP is considered overfished and its continued maintenance is questionable. Furthermore, spawner abundance may decline to even lower levels. Alternatively, fishing rates that preserve 40% of MSP allow for the preservation of the stock and maximize yield to the fisheries. This level may be viewed as conservative because the simulations suggested that even under moderate exploitation in the 1960s and 1970s, a number of simulated projections were below 40% of MSP.

Probabilities that the projected baseline and impacted stocks of Niantic River female spawner biomass would fall below 25, 30, and 40% of MSP were determined (Table 42). In 1970, the stocks were likely ( $p \geq 0.87$ ) larger than 40% of the MSP. However, by 1980 both the baseline and impacted stocks had increased probabilities (0.44, 0.53) of falling below 40% of MSP. In 1990, the baseline and impacted stock sizes were almost certainly less than 40% of MSP and likely ( $p \geq 0.73, 0.80$ ) less than 30% of MSP. The impacted stock also had a probability of

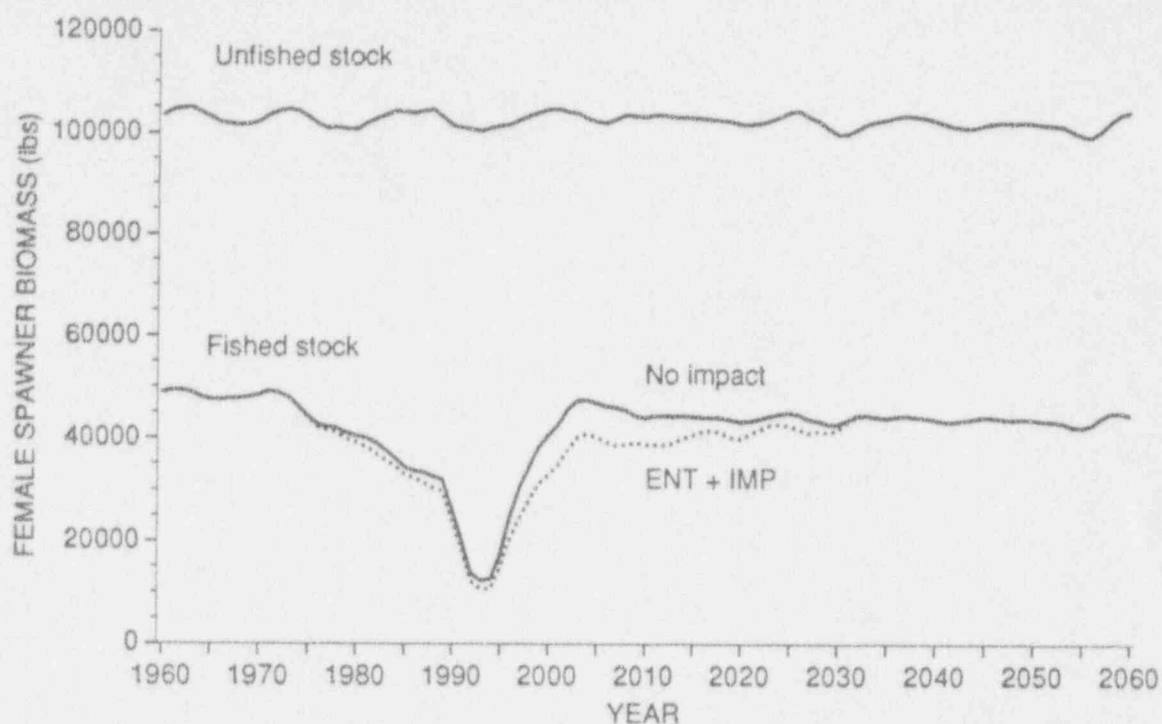


Fig. 40. Illustration of the effects of fishing (solid line labeled "No impact") and MNPS operation under calculated larval entrainment and impingement rates (dashed line labeled "ENT + IMP") relative to the theoretical (SRR-based estimate) unfished stock expressed as female spawner biomass in lbs. All stock sizes are averages of 100 Monte Carlo replicates.

0.45 of falling below 25% of MSP. At the lowest point of both stock projections in 1993, all replicates were below 25% of MSP. Reductions in fishing rates in the late 1990s allowed for a rapid increase in spawning biomass above this critical level to more optimal stock sizes by 2000. Spawning biomass was then likely greater than 25% of MSP and the impacted stock had a one in four chance of being greater than 30% of MSP. By 2010 and 2020, the stocks had high likelihood of being above 30% of MSP and the impacted stock had a better than even probability of being greater than 40% of MSP. Note that these probabilities may be viewed as conservative because due to the dynamics of the simulated stock, these years happened to represent local minima in the stock size time-series (see Fig. 38). For the stock to reach a more desirable size, which according to Howell et al. (1992) is greater than 40% of MSP, fishing mortality would have to be further reduced. The stock stabilized at a biomass of about 43 to 44 thousand lbs following the shutdown of MNPS in 2025 and probabilities of biomass being smaller than this reference size were about 30 to 40% annually.

## Conclusions

The number of winter flounder spawning in the Niantic River further decreased in 1993 and adult abundance indices were the lowest since sampling began in 1976. The low level of abundance mirrored regional trends reported by the National Marine Fisheries Service for recent years. Furthermore, the large size of spawners and lack of smaller adults observed during the Niantic River surveys may be a forewarning of still more declines in spawner biomass during the next few years. The relatively small spawning stock produced the smallest densities of larvae yet found in Niantic River and Bay. Even with three-unit operation, the number of larvae entrained at MNPS was among the lowest in the 17-year series of annual estimates. Juvenile winter flounder had somewhat better than average survival in 1993, but because relatively few fish settled in the Niantic river, the 1993 year-class strength appears to be poor. This was in contrast to 1992 when, despite very low abundance of spawners, larval abundance was the highest in 18 years and juvenile abundance indices indicated a

TABLE 41. Expected biomass in pounds of female winter flounder spawners at seven selected points in time during SPDM simulations of the Niantic River population (see Figures 38 and 39). Expected mean stock sizes are geometric means of 100 Monte Carlo replicates and the minimum and maximum stock sizes represent the range for the 100 replicates of each year.

Type of population simulated	1970	1980	1990	1993	2000	2010	2020
<b>Theoretical unfished stock<sup>a</sup></b>							
Mean	102,145	100,645	101,306	100,388	104,605	103,016	101,573
Minimum	75,843	76,274	77,871	71,561	79,448	81,889	69,904
Maximum	141,890	135,589	135,781	137,863	146,075	132,545	130,058
<b>Baseline<sup>b</sup></b>							
Mean	48,271	40,352	26,413	12,312	40,571	43,988	43,160
Minimum	30,405	28,520	17,487	8,185	25,315	28,575	26,547
Maximum	75,404	62,991	43,190	23,005	63,357	65,211	63,667
% of the theoretical unfished stock	47.3%	40.1%	26.1%	12.3%	38.8%	42.7%	42.5%
<b>Impact (ENT + IMP)<sup>c</sup></b>							
Mean	48,271	39,094	24,531	10,562	32,890	38,767	39,836
Minimum	30,405	27,552	16,072	7,116	20,160	26,061	24,502
Maximum	75,405	61,171	40,040	19,598	52,666	56,634	57,448
% of the theoretical unfished stock	47.3%	38.8%	24.2%	10.5%	31.4%	37.6%	39.2%

<sup>a</sup> No fishing or MNPS effects.

<sup>b</sup> Fishing effects, but no MNPS impact.

<sup>c</sup> Combined effects of entrainment and impingement (ENT + IMP) at MNPS in addition to fishing.

TABLE 42. Probabilities of Niantic River female spawning stock biomass falling below three selected reference sizes at seven selected points in time. Postulated reductions are relative to the maximum spawning potential (MSP) of 97,075 lbs for the theoretical unfished stock ( $F = 0$ ). Probabilities were based on 100 Monte Carlo replications of the simulations.

Type of population simulated	Reference stock size <sup>a</sup>	1970	1980	1990	1993	2000	2010	2020
Baseline <sup>b</sup>	25% of MSP	0.00	0.00	0.33	1.00	0.00	0.00	0.00
Impacted <sup>c</sup>	25% of MSP	0.00	0.00	0.45	1.00	0.13	0.00	0.00
Baseline	30% of MSP	0.00	0.01	0.73	1.00	0.09	0.01	0.02
Impacted	30% of MSP	0.00	0.01	0.80	1.00	0.24	0.03	0.02
Baseline	40% of MSP	0.13	0.44	0.99	1.00	0.41	0.21	0.26
Impacted	40% of MSP	0.13	0.53	0.99	1.00	0.78	0.52	0.44

<sup>a</sup> Corresponds to reference stock sizes given in Howell et al. (1992) of 25%, 30%, and 40% of the MSP (24,269, 29,123, and 38,830 lbs, respectively).

<sup>b</sup> Fishing effects, but no MNPS impact.

<sup>c</sup> Combined effects of entrainment and impingement (ENT + IMP) at MNPS in addition to fishing.

relatively strong year-class. A number of environmental and biological factors interact to determine reproductive success within a particular year. The low abundance of adult spawners in 1993 combined with low larval production will retard the recovery of the Niantic River winter flounder stock. However, newly implemented fishing regulations may protect the relatively strong 1992 year-class, helping increase the numbers of spawning fish in subsequent years.

A long-term assessment using the SPDM indicated that fishing alone reduced Niantic River spawner biomass from somewhat more than 100,000 lbs for the theoretical unfished stock to about half that amount by the 1960s and 1970s. Large increases in fishing mortality since then reduced baseline (fishing effects only) stock biomass rather rapidly to about 12,000 lbs by 1993. Adding the effects of MNPS operation (primarily larval entrainment) further reduced stock size by an additional 1,500 lbs. These stock sizes are considerably below the critical size (about 24,000 lbs), defined by the Atlantic States Marine Fisheries Commission as 25% of the maximum spawning potential (i.e., spawner biomass in the absence of fishing). However, the simulation of substantial reductions in fishing mortality expected to occur in the near future as a result of new regulations showed that the stock could recover quickly. By 2000, the baseline and MNPS-impacted stocks had high probability of being larger than 25% of MSP and by 2010 of being greater than 30% of MSP. As stock abundance increased, the effect of MNPS became larger in terms of absolute losses of stock biomass. However, the two biomass time-series became identical within a few years after the cessation of MNPS operation in 2025 and the stock could be considered as fully recovered. This scenario, however, will require that the planned changes in fishing regulations be implemented as scheduled and that they achieve the expected reductions in fishing mortality.

### References Cited

- Al-Hossaini, M., Q. Liu, and T.J. Pitcher. 1989. Otolith microstructure indicating growth and mortality among plaice, *Pleuronectes platessa* L., post-larval sub-cohorts. *J. Fish Biol.* 35 (Suppl. A):81-90.
- Anderson, J.T. 1988. A review of size dependent survival during pre-recruit stages of fishes in relation to recruitment. *J. Northw. Atl. Fish. Sci.* 8:55-66.
- Arai, M.N., and D.E. Hay. 1982. Predation by medusae on Pacific herring (*Clupea harengus*) larvae. *Can. J. Fish. Aquat. Sci.* 39:1537-1540.
- Arnason, A.N., and K.H. Mills. 1981. Bias and loss of precision due to tag loss in Jolly-Seber estimates for mark-recapture experiments. *Can. J. Fish. Aquat. Sci.* 38:1077-1095.
- Bagge, O., and E. Nielsen. 1988. The change in abundance and growth of plaice and dab in Subdivision 22, 1965-1985. ICES Bal/No. 27. (Not seen, cited by Pihl and van der Veer 1992).
- Bailey, K.M., and R.S. Batty. 1984. Laboratory study of predation by *Aurelia aurelia* on larvae of cod, flounder, plaice and herring: development and vulnerability to capture. *Mar. Biol.* 83:287-291.
- Bannister, R.C.A., D. Harding, and S.J. Lockwood. 1974. Larval mortality and subsequent year-class strength in the plaice (*Pleuronectes platessa* L.). Pages 21-38 in J.H.S. Blaxter, ed. *The early life history of fish*. Springer-Verlag, New York.
- Begon, M. 1979. Investigating animal abundance: capture-recapture for biologists. University Park Press, Baltimore. 97 pp.
- Berghahn, R. 1986. Determining abundance, distribution, and mortality of 0-group plaice (*Pleuronectes platessa* L.) in the Wadden Sea. *J. Appl. Ichthyol.* 2: 11-22.
- Berghahn, R. 1987. Effects of tidal migration on growth of 0-group plaice (*Pleuronectes platessa* L.) in the North Frisian Wadden Sea. *Meeresforsch.* 31:209-226. (Not seen, cited by Karakiri et al. 1989).
- Bergman, M.J.N., H.W. van der Veer, and J.J. Zijlstra. 1988. Plaice nurseries: effects on recruitment. *J. Fish Biol.* 33 (Suppl. A): 210-218.
- Bishop, J.A., and P.M. Sheppard. 1973. An evaluation of two capture-recapture models using the technique of computer simulation. Pages 235-253 in M.S. Bartlett and R.W. Hiorns, eds. *The mathematical theory of the dynamics of biological populations*. Academic Press, London.
- Boudreau, P.R., and L.M. Dickie. 1989. Biological model of production based on physiological and ecological scaling of body size. *Can. J. Fish. Aquat. Sci.* 46:614-623.
- Buckley, L.J. 1980. Changes in ribonucleic acid, deoxyribonucleic acid, and protein content during ontogenesis in winter flounder, *Pseudopleuronectes americanus*, and effect of starvation. *Fish. Bull., U.S.* 77:703-708.



- Buckley, L.J. 1982. Effects of temperature on growth and biochemical composition of larval winter flounder *Pseudopleuronectes americanus*. Mar. Ecol. Prog. Ser. 8:181-186.
- Buckley, L.J., A.S. Smigielski, T.A. Halavik, and G.C. Laurence. 1990. Effects of water temperature on size and biochemical composition of winter flounder *Pseudopleuronectes americanus* at hatching and feeding initiation. Fish. Bull., U.S. 88:419-428.
- Buckley, L.J., A.S. Smigielski, T.A. Halavik, E.M. Caldarone, B.R. Burns, and G.C. Laurence. 1991. Winter flounder *Pseudopleuronectes americanus* reproductive success. II. Effects of spawning time and female size on size, composition and viability of eggs and larvae. Mar. Ecol. Prog. Ser. 74:125-135.
- Burton, M.P., and D.R. Idler. 1984. The reproductive cycle in winter flounder *Pseudopleuronectes americanus* (Walbaum). Can. J. Zool. 62:2563-2567.
- Carothers, A.D. 1973. The effects of unequal catchability on Jolly-Seber estimates. Biometrics 29:79-100.
- Chambers, R.C., and W.C. Leggett. 1987. Size and age at metamorphosis in marine fishes: an analysis of laboratory-reared winter flounder (*Pseudopleuronectes americanus*) with a review of variation in other species. Can. J. Fish. Aquat. Sci. 44:1936-1947.
- Chambers, R.C., W.C. Leggett, and J.A. Brown. 1988. Variation in and among early life history traits of laboratory-reared winter flounder *Pseudopleuronectes americanus*. Mar. Ecol. Prog. Ser. 47:1-15.
- Christensen, S.W., and C.P. Goodyear. 1988. Testing the validity of stock-recruitment curve fits. Am. Fish. Soc. Monogr. 4:219-231.
- Cormack, R.M. 1968. The statistics of mark-recapture methods. Oceanogr. Mar. Biol. Ann. Rev. 6:455-506.
- Crawford, R.E. 1990. Winter flounder in Rhode Island coastal ponds. Rhode Island Sea Grant, Univ. of Rhode Island, Narragansett, RI. RIU-G-90-001. 24 pp.
- Crawford, R.E., and C.G. Carey. 1985. Retention of winter flounder larvae within a Rhode Island salt pond. Estuaries 8:217-227.
- Crecco, V.A., and P. Howell. 1990. Potential effects of current larval entrainment mortality from the Millstone Nuclear Power Station on the winter flounder, *Pseudopleuronectes americanus*, spawning population in the Niantic River. Connecticut Dept. Envir. Prot., Bu. Fish., Spec. Pub. 37 pp.
- Crecco, V.A., and T. Savoy. 1987. Fishery management plan for the American shad in the Connecticut River. Connecticut Dept. Envir. Prot., Bu. Fish., Spec. Pub. 140 pp.
- Cushing, D.H. 1971. The dependence of recruitment on parent stock in different groups of fish. J. Cons. int. Explor. Mer 33:340-362.
- Cushing, D.H. 1974. The possible density-dependence of larval mortality and adult mortality in fishes. Pages 103-111 in J.H.S. Blaxter, ed. The early life history of fish. Springer-Verlag, New York.
- Cushing, D.H. 1977. The problems of stock and recruitment. Pages 116-133 in J.A. Gulland, ed. Fish population dynamics. John Wiley and Sons, New York.
- Cushing, D.H., and J.G.K. Harris. 1973. Stock and recruitment and the problem of density dependence. Rappr. P.-v. Reun. Cons. int. Explor. Mer 164:142-155.
- Cushing, D.H., and J.W. Horwood. 1977. Development of a model of stock and recruitment. Pages 21-35 in J.H. Steele, ed. Fisheries mathematics. Academic Press, New York.
- DeBlois, E.M., and W.C. Leggett. 1991. Functional response and potential impact of invertebrate predators on benthic fish eggs: analysis of the *Callinectes laevis*-capelin (*Mallotus villosus*) predator-prey system. Mar. Ecol. Prog. Ser. 69:205-216.
- Dimou, N.K., and E.E. Adams. 1989. Application of a 2-D particle tracking model to simulate entrainment of winter flounder larvae at the Millstone Nuclear Power Station. Energy Laboratory Report No. MIT-EL 89-002. Massachusetts Institute of Technology, Cambridge, MA. 73 pp.
- Draper, N., and H. Smith. 1981. Applied regression analysis. John Wiley and Sons, New York. 709 pp.
- Dunn, R.S. 1970. Further evidence for a three-year oocyte maturation time in the winter flounder (*Pseudopleuronectes americanus*). J. Fish. Res. Board Can. 27:957-960.
- Dunn, R.S., and A.V. Tyler. 1969. Aspects of the anatomy of the winter flounder ovary with hypotheses on oocyte maturation time. J. Fish. Res. Board Can. 26:1943-1947.



- Garrod, D.J., and B.W. Jones. 1974. Stock and recruitment relationships in the Northeast Arctic cod stock and the implications for the management of the stock. *J. Cons. int. Explor. Mer* 36:35-41.
- Gendron, L. 1989. Seasonal growth of the kelp *Laminaria loniceris* in Baie des Chaleurs, Quebec, in relation to nutrient and light availability. *Bot. Mar.* 32:345-354.
- Gibson, M.R. 1987. Preliminary assessment of winter flounder (*Pseudopleuronectes americanus*) stocks in Rhode Island waters. Rhode Island Div. Fish Wildl., Res. Ref. Doc. 87/7. 51 pp.
- Gibson, M.R. 1989. Stock-recruitment relationships for winter flounder in the S. New England area and revised fishery reference points. Rhode Island Div. Fish Wildl., Res. Ref. Doc. 89/9. 10 pp + 5 fig.
- Gilbert, R.O. 1973. Approximation of the bias in the Jolly-Seber capture-recapture model. *Biometrics* 29:501-526.
- Goodyear, C.P. 1977. Assessing the impact of power plant mortality on the compensatory reserve of fish populations. Pages 186-195 in W. Van Winkle, ed. *Proceedings of the conference on assessing the effects of power plant-induced mortality on fish populations*. Pergamon Press, New York.
- Goodyear, C.P. 1980. Compensation in fish populations. Pages 253-280 in C.H. Hocutt and J.R. Stauffer, eds. *Biological monitoring of fish*. Lexington Books, Lexington, MA.
- Goodyear, C.P., and S.W. Christensen. 1984. Bias-elimination in fish population models with stochastic variation in survival of the young. *Trans. Am. Fish. Soc.* 113:627-632.
- Hennemuth, R.C., J.E. Palmer, and B.E. Brown. 1980. A statistical description of recruitment in eighteen selected fish stocks. *J. Northwest Atl. Fish.* 1:101-111.
- Hess, K.W., M.P. Sissenwine, and S.B. Saila. 1975. Simulating the impact of entrainment of winter flounder larvae. Pages 1-30 in S.B. Saila, ed. *Fisheries and energy production: a symposium*. D.C. Heath and Co., Lexington, MA.
- Hightower, J.E., and R.J. Gilbert. 1984. Using the Jolly-Seber model to estimate population size, mortality, and recruitment for a reservoir fish population. *Trans. Am. Fish. Soc.* 113:633-641.
- Hjorleifsson, E. 1989. (Abstr.). Condition of winter flounder larvae in Narragansett Bay as measured by RNA/DNA ratio. Workshop on winter flounder biology, Mystic, CT, December 5-6, 1989.
- Hjort, J. 1926. Fluctuations in the year classes of important food fishes. *J. Cons. int. Explor. Mer* 1:5-38. (not seen, cited by May 1974).
- Hoenig, J.M., D.M. Heisey, W.D. Lawing, and H.D. Schupp. 1987. An indirect rapid methods approach to assessment. *Can. J. Fish. Aquat. Sci.* 44 (Suppl. 2):324-338.
- Houde, E.D. 1987. Fish early life history dynamics and recruitment variability. *Am. Fish. Soc. Symposium* 2:17-29.
- Houde, E.D. 1989. Subtleties and episodes in the early life history of fishes. *J. Fish Biol.* 35 (Suppl. A):29-38.
- Hovenkamp, F., and J.I.J. Witte. 1991. Growth, otolith growth and RNA/TNA ratios of larval plaice *Pleuronectes platessa* in the North Sea 1987 to 1989. *Mar. Ecol. Prog. Ser.* 70:105-116.
- Howe, A.B., and P.G. Coates. 1975. Winter flounder movements, growth and mortality off Massachusetts. *Trans. Am. Fish. Soc.* 104:13-29.
- Howell, P., A. Howe, M. Gibson, and S. Ayvazian. 1992. Fishery management plan for inshore stocks of winter flounder. Fisheries management rep. no. 21 of the Atlantic States Marine Fisheries Commission. 138 pp.
- Howell, P.T., and D.R. Molnar. 1993. Job 3. In-shore survey of juvenile winter flounder. Pages 49-66 in *A study of marine recreational fisheries in Connecticut. Federal aid to sport fish restoration F54R annual performance report*. State of Conn., Dept. Envir. Prot., Bu. Nat. Res., Fish. Div.
- Howell, W.H., and R. Langan. 1987. Commercial trawler discards of four flounder species in the Gulf of Maine. *N. Am. J. Fish. Man.* 7:6-17.
- Howell, W.H., and R. Langan. 1992. Discarding of commercial groundfish species in the Gulf of Maine shrimp fishery. *N. Am. J. Fish. Man.* 12:568-580.
- Jolly, G.M. 1965. Explicit estimates from capture-recapture data with death and immigration stochastic model. *Biometrika* 52:225-247.
- Karakiri, M., R. Berghahn, and H. von Westernhagen. 1989. Growth differences in 0-group plaice *Pleuronectes platessa* as revealed by otolith microstructure analysis. *Mar. Ecol. Prog. Ser.* 55:15-22.
- Klein-MacPhee, G. 1978. Synopsis of biological data for the winter flounder, *Pseudopleuronectes americanus* (Walbaum). NOAA Tech. Rep. NMFS Circ. 414. 43 pp.
- Kollmeyer, R.C. 1972. A study of the Niantic River estuary, Niantic, Connecticut. Final report phases

- I and II, physical aspects of the Niantic River estuary. Rep. No. RDCGA 18. U.S. Coast Guard Academy, New London, CT. 78 pp.
- Kuipers, B., B. MacCurrin, J.M. Miller, H.W. van der Veer, and J. IJ. Witte. 1992. Small trawls in juvenile flatfish research: their development and efficiency. *Neth. J. Sea Res.* 29:109-117.
- Laurence, G.C. 1975. Laboratory growth and metabolism of the winter flounder *Pseudopleuronectes americanus* from hatching through metamorphosis at three temperatures. *Mar. Biol.* 32:223-229.
- Laurence, G.C. 1977. A bioenergetic model for the analysis of feeding and survival potential of winter flounder, *Pseudopleuronectes americanus*, larvae during the period from hatching through metamorphosis. *Fish. Bull., U.S.* 75:529-546.
- Lobell, M.J. 1939. A biological survey of the salt waters of Long Island, 1938. Report on certain fishes. Winter flounder (*Pseudopleuronectes americanus*). Suppl. 28th Ann. Rep., N.Y. Cons. Dep., Pt. I:63-96.
- Lockwood, S.J. 1972. The settlement, distribution and movements of 0-group plaice (*Pleuronectes platessa* L.) in Filey Bay, Yorkshire. *J. Fish. Biol.* 6:465-477.
- Lockwood, S.J. 1980. Density-dependent mortality in 0-group plaice (*Pleuronectes platessa* L.) populations. *J. Cons. int. Explor. Mer* 39:148-153.
- Longhurst, A. 1983. Benthic-pelagic coupling and export of organic carbon from a tropical Atlantic continental shelf, Sierra Leone. *Est. Coast. Shelf Sci.* 17:261-285.
- Lorda, E.C., and V.A. Crecco. 1987. Stock-recruitment relationship and compensatory mortality of American shad in the Connecticut River. *Am. Fish. Soc. Symposium* 1:469-482.
- Manly, B.J.F. 1971. A simulation of Jolly's method for analysing capture-recapture data. *Biometrics* 27:415-424.
- Marine Research, Inc. 1992. Brayton Point investigations semi-annual report. January-June 1992. Submitted to New England Power Co.
- Marshall, N., and S.D. Hicks. 1962. Drift of medusae and their distribution in relation to the hydrography of the Niantic River, Connecticut. *Limnol. Oceanogr.* 7:268-269.
- May, R.C. 1974. Larval mortality in marine fishes and the critical period concept. Pages 3-20 in J.H.S. Blaxter, ed. *The early life history of fish*. Springer-Verlag, New York.
- McConnaughey, R.A., and L.L. Conquest. 1993. Trawl survey estimation using a comparative approach based on lognormal theory. *Fish. Bull., U.S.* 91:107-118.
- McCracken, F.D. 1963. Seasonal movements of the winter flounder, *Pseudopleuronectes americanus* (Walbaum), on the Atlantic coast. *J. Fish. Res. Board Can.* 20:551-586.
- Miller, J.M., J.S. Burke, and G.R. Fitzhugh. 1991. Early life history patterns of Atlantic North American flatfish: likely (and unlikely) factors controlling recruitment. *Neth. J. Sea Res.* 27:261-275.
- Moller, H. 1984. Reduction of a larval herring population by jellyfish predator. *Science* 224:621-622.
- Morrison, J.A., I.R. Napier, and J.C. Gamble. 1991. Mass mortality of herring eggs associated with a sedimenting diatom bloom. *ICES J. Mar. Sci.* 48:237-245.
- Myers, R.A., and N.G. Cadigan. 1993a. Density-dependent juvenile mortality in marine demersal fish. *Can. J. Fish. Aquat. Sci.* 50:1576-1590.
- Myers, R.A., and N.G. Cadigan. 1993b. Is juvenile mortality in marine demersal fish variable? *Can. J. Fish. Aquat. Sci.* 50:1591-1598.
- Nichols, J.D., B.R. Noon, S.L. Stokes, and J.E. Hines. 1981. Remarks on the use of capture-recapture methodology in estimating avian population size. *Studies in Avian Biol.* 6:121-136. (not seen, cited by Hightower and Gilbert 1984).
- NMFS (National Marine Fisheries Service). 1993. Status of fishery resources off the northeastern United States for 1993. NOAA Tech. Mem. NMFS-F/NEC-101. 140 pp.
- NUSCO (Northeast Utilities Service Company). 1976. Environmental assessment of the condenser cooling water intake structures (316b demonstration). Vol. 1 and 2.
- NUSCO. 1981. Plankton ecology. In *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Annual report, 1980. 40 pp.
- NUSCO. 1985. Winter flounder studies. In *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Annual report, 1984. 74 pp.
- NUSCO. 1986. Winter flounder studies. In *Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut*. Annual report, 1985. 69 pp.

- NUSCO. 1987. Winter flounder studies. *In* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Summary of studies prior to Unit 3 operation. 151 pp.
- NUSCO. 1988a. Winter flounder studies. Pages 149-224 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-1987.
- NUSCO. 1988b. The usage and estimation of DELTA means. Pages 311-320 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Three-unit operational studies, 1986-1987.
- NUSCO. 1989. Winter flounder studies. Pages 239-316 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1988.
- NUSCO. 1990. Winter flounder studies. Pages 9-77 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1989.
- NUSCO. 1991a. Evaluation of the larval winter flounder sampling program in the Niantic River. Enclosure to letter D04343 dated January 23, 1991 from E.J. Mroczka, NUSCO, to L. Carothers, Commissioner, CT DEP.
- NUSCO. 1991b. Winter flounder studies. Pages 9-86 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1990.
- NUSCO. 1992a. Winter flounder studies. Pages 7-109 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1991.
- NUSCO. 1992b. Niantic Bay current studies. Pages 317-331 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1991.
- NUSCO. 1993. Winter flounder studies. Pages 191-269 *in* Monitoring the marine environment of Long Island Sound at Millstone Nuclear Power Station, Waterford, Connecticut. Annual report 1992.
- O'Brien, L., J. Burnett, and R.K. Mayo. 1993. Maturation of nineteen species of finfish off the northeast coast of the United States, 1985-1990. NOAA Tech. Rep. NMFS 113. 66 pp.
- Olla, B.L., R. Wicklund, and S. Wilk. 1969. Behavior of winter flounder in a natural habitat. *Trans. Am. Fish. Soc.* 98:717-720.
- Parrish, B.B. 1963. Some remarks on the selection processes in fishing operations. *Int. Comm. Northwest Atl. Fish. Spec. Pub.* 5:166-170.
- Pearcy, W.G. 1962. Ecology of an estuarine population of winter flounder *Pseudopleuronectes americanus* (Walbaum). *Bull. Bingham Oceanogr. Coll.* 18:1-78.
- Pennington, M. 1983. Efficient estimators of abundance for fish plankton surveys. *Biometrics* 39:281-286.
- Pennington, M. 1986. Some statistical techniques for estimating abundance indices from trawl surveys. *Fish. Bull., U.S.* 84:519-525.
- Perlmutter, A. 1947. The blackback flounder and its fishery in New England and New York. *Bull. Bingham Oceanogr. Coll.* 11:1-92.
- Pihl, L. 1990. Year-class strength regulation in plaice (*Pleuronectes platessa* L.) on the Swedish west coast. *Hydrobiologia* 195:79-88.
- Pihl, L., and H.W. van der Veer. 1992. Importance of exposure and habitat structure for the population density of 0-group plaice, *Pleuronectes platessa* L., in coastal nursery areas. *Neth. J. Sea Res.* 29:145-152.
- Pollock, K.H., J.D. Nichols, C. Brownie, and J.E. Hines. 1990. Statistical inference for capture-recapture experiments. *Wildl. Monogr.* 107. 97 pp.
- Poxton, M.G., A. Eleftheriou, and A.D. McIntyre. 1982. The population dynamics of 0-group flatfish in the Clyde Sea area. *Est. Coast. Shelf Sci.* 14:265-282.
- Poxton, M.G., and N.A. Nasir. 1985. The distribution and population dynamics of 0-group plaice (*Pleuronectes platessa* L.) on nursery grounds in the Firth of Forth. *Est. Coast. Shelf Sci.* 21:845-857.
- Reed, M., M.L. Spaulding, E. Lorda, H. Walker, and S.B. Saila. 1984. Oil spill fishery impact assessment modeling: the fisheries recruitment problem. *Est. Coast. Shelf Sci.* 19:591-610.
- Ricker, W.E. 1954. Stock and recruitment. *J. Fish. Res. Board Can.* 11:559-623.
- Ricker, W.E. 1973. Linear regressions in fishery research. *J. Fish. Res. Board Can.* 30:409-434.
- Ricker, W.E. 1975. Computation and interpretation



- of biological statistics of fish populations. Bull. Fish. Res. Board Can. 191. 382 pp.
- Ricker, W.E. 1984. Computation and uses of central trend lines. Can. J. Zool. 62:1897-1905.
- Riley, J.D. 1973. Movements of 0-group plaice *Pleuronectes platessa* L. as shown by latex tagging. J. Fish Biol. 5:323-343.
- Roff, D.A. 1973. On the accuracy of some mark-recapture estimators. Oecologia 12:15-34.
- Rogers, S.L., and S.J. Lockwood. 1989. Observations on the capture efficiency of a two-metre beam trawl for juvenile flatfish. Neth. J. Sea Res. 23:347-352.
- Rothschild, B.J., and G.T. DiNardo. 1987. Comparison of recruitment variability and life history data among marine and anadromous fishes. Pages 531-546 in M.J. Dadswell, R.J. Klauda, C.M. Moffitt, R.L. Saunders, R.A. Rulifson, and J.E. Cooper, eds. Common strategies of anadromous and catadromous fishes. Am. Fish. Soc. Sym. 1.
- Roughgarden, J. 1979. Evolutionary ecology of single populations. Pages 295-408 in The theory of population genetics and evolutionary ecology: an introduction. MacMillan Publishing Company, Inc., New York.
- Rubinstein, R.Y. 1981. Simulation and the Monte Carlo method. John Wiley and Sons, New York. 278 pp.
- Saila, S.B. 1961. A study of winter flounder movements. Limnol. Oceanogr. 6:292-298.
- Saila, S.B. 1962a. The contribution of estuaries to the offshore winter flounder fishery in Rhode Island. Proc. Gulf Caribb. Fish. Inst. 14th Ann. Sess. 1961:95-109.
- Saila, S.B. 1962b. Proposed hurricane barriers related to winter flounder movements in Narragansett Bay. Trans. Am. Fish. Soc. 91:189-195.
- SAS Institute Inc. 1985. SAS user's guide: statistics. Version 5 edition. SAS Institute Inc., Cary, NC. 956 pp.
- Saucerman, S.E., and L.A. Deegan. 1991. Lateral and cross-channel movement of young-of-the-year winter flounder (*Pseudopleuronectes americanus*) in Waquoit Bay, Massachusetts. Estuaries 14:440-446.
- Scott, W.B., and M.G. Scott. 1988. Atlantic fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219. 731 pp.
- Simpson, D.G. 1989. Codend selection of winter flounder *Pseudopleuronectes americanus*. NOAA Tech. Rep. NMFS 75. 10 pp.
- Sissenwine, M.B. 1984. Why do fish populations vary? Pages 59-94 in R.M. May, ed. Exploitation of marine communities. Springer-Verlag, New York.
- Smigielski, A.S. 1975. Hormonal-induced ovulation of the winter flounder, *Pseudopleuronectes americanus*. Fish. Bull., U.S. 73:431-438.
- Smith, E.M., E.C. Mariani, A.P. Petrillo, L.A. Gunn, and M.S. Alexander. 1989. Principal fisheries of Long Island Sound, 1961-1985. Connecticut Dept. Envir. Prot., Bu. Fish., Mar. Fish. Program. 47 pp. + app.
- Smith, T.D. 1988. Stock assessment methods: the first fifty years. Pages 1-33 in J.A. Gulland, ed. Fish population dynamics (second ed.). John Wiley and Sons, New York.
- Smith, W.G., J.D. Sibunka, and A. Wells. 1975. Seasonal distributions of larval flatfishes (Pleuronectiformes) on the continental shelf between Cape Cod, Massachusetts and Cape Lookout, North Carolina, 1965-1966. NOAA Tech. Rep. NMFS SSRF-691.
- Snedecor, G.W., and W.G. Cochran. 1967. Statistical methods. The Iowa State University Press, Ames, IA. 593 pp.
- Sogard, S.M. 1990. Parameters of habitat quality for epibenthic fishes and decapod crustaceans in New Jersey estuaries. Ph.D. dissertation, Rutgers University, New Brunswick, NJ. 195 pp. (Not seen, cited by Sogard and Able 1992).
- Sogard, S.M., and K.W. Able. 1992. Growth variation of newly settled winter flounder (*Pseudopleuronectes americanus*) in New Jersey estuaries as determined by otolith microstructure. Neth. J. Sea Res. 29:163-172.
- Southwood, T.R.E. 1978. Ecological methods. Halstead Press, New York. 523 pp.
- Spaulding, M.L., S.B. Saila, E. Lorda, H. Walker, E. Anderson, and J.C. Swanson. 1983. Oil-spill fishery impact assessment model: application to selected Georges Bank fish species. Est. Coast. Shelf Sci. 16:511-541.
- Steele, J., C. Clark, P. Larkin, R. Lasker, R. May, B. Rothschild, E. Ursin, J. Walsh, and W. Wooster. 1980. Fisheries ecology: some constraints that impede our understanding. Ocean Science Board, National Academy of Science, Washington, D.C.

- Sieele, J., and R.R.C. Edwards. 1970. The ecology of 0-group plaice and common dabs in Loch Ewe. IV. Dynamics of the plaice and dab populations. *J. Exp. Mar. Biol.* 4:174-187.
- Stuart, A., and J.K. Ord. 1987. Kendall's advanced theory of statistics. Vol. I. Distribution theory. Oxford University Press, New York. 604 pp.
- Townsend, D.W., and L.M. Cammen. 1988. Potential importance of the timing of spring plankton blooms to benthic-pelagic coupling and recruitment of juvenile demersal fishes. *Biol. Oceanogr.* 5:215-229.
- Vaughan, D.S. 1981. An age structure model of yellow perch in western Lake Erie. Pages 189-216 in D.G. Chapman and V.F. Gallucci, eds. Quantitative population dynamics. International Co-operative Publishing House, Fairland, MD.
- Veer, H.W. van der. 1985. Impact of coelenterate predation on larval plaice *Pleuronectes platessa* and flounder *Platichthys flesus* stock in the western Wadden Sea. *Mar. Ecol. Prog. Ser.* 25:229-238.
- Veer, H.W. van der. 1986. Immigration, settlement, and density-dependent mortality of a larval and early postlarval 0-group plaice (*Pleuronectes platessa*) population in the western Wadden Sea. *Mar. Ecol. Prog. Ser.* 29:223-236.
- Veer, H.W. van der, and M.J.N. Bergman. 1987. Predation by crustaceans on a newly settled 0-group plaice *Pleuronectes platessa* population in the western Wadden Sea. *Mar. Ecol. Prog. Ser.* 35:203-215.
- Veer, H.W. van der, M.J.N. Bergman, R. Dapper, and J. IJ. Witte. 1991. Population dynamics of an intertidal 0-group flounder *Platichthys flesus* population in the western Dutch Wadden Sea. *Mar. Ecol. Prog. Ser.* 73:141-148.
- Veer, H.W. van der, L. Pihl, and M.J.N. Bergman. 1990. Recruitment mechanisms in North Sea plaice *Pleuronectes platessa*. *Mar. Ecol. Prog. Ser.* 64:1-12.
- Wigley, S.E., and W.L. Gabriel. 1991. Distribution of sexually immature components of 10 northwest Atlantic groundfish species based on Northeast Fisheries Center bottom trawl surveys 1968-86. NOAA Tech. Mem. NMFS-F/NEC-80. 17 pp.
- Williams, P.J., and J.A. Brown. 1992. Development changes in the escape response of larval winter flounder *Pleuronectes americanus* from hatch through metamorphosis. *Mar. Ecol. Prog. Ser.* 88:185-193.
- Witherell, D.B., and J. Burnett. 1993. Growth and maturation of winter flounder, *Pleuronectes americanus*, in Massachusetts. *Fish. Bull., U.S.* 91:816-820.
- Witting, D.A., and K.W. Able. 1993. Effects of body size on probability of predation for juvenile summer and winter flounder based on laboratory experiments. *Fish. Bull., U.S.* 91:577-581.
- Zijlstra, J.J., R. Dapper, and J. IJ. Witte. 1982. Settlement, growth and mortality of post-larval plaice (*Pleuronectes platessa* L.) in the western Wadden Sea. *Neth. J. Sea Res.* 15:250-272.