Massachusetts Bay plankton communities: characterization and discussion of issues relative to MWRA's outfall relocation

Massachusetts Water Resources Authority

Environmental Quality Department
Report ENQUAD 98-08

Massachusetts Bay Plankton Communities:
Characterization and Discussion of Issues Relative to MWRA's Outfall Relocation

Task 337
MWRA Harbor and Outfall Monitoring Project

submitted to

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July 24, 1998
Reprinted April 1999

Citation:

Cibik SJ, Lemieux KB, Davis CS, Anderson DM. 1998. Massachusetts Bay plankton communities: characterization and discussion of issues relative to MWRA's outfall relocation. Boston: Massachusetts Water Resources Authority. Report ENQUAD 98-08. 140 p.

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EXECUTIVE SUMMARY

The principal goal of this Plankton Issues Report was to characterize the baseline of the phytoplankton and zooplankton communities in the Massachusetts Bay system prior to relocation of the Massachusetts Water Resources Authority (MWRA) effluent discharge approximately 15km offshore. Available plankton data, from both the literature and those generated by the MWRA baseline monitoring program (1992 to 1995), were examined to identify seasonal and interannual differences in plankton assemblages. The physical and chemical dynamics of the system were also examined to support the interpretation of these differences.

Additionally, the ecological processes governing plankton growth and succession were reviewed in support of a second goal, which was to consider the means to assess potential responses of phytoplankton and zooplankton to the post-relocation environment. The findings of this endeavor are summarized below.

Physical and Chemical Dynamics. Massachusetts Bay is a complex and highly variable system. Much of what occurs in the system is controlled by its seasonal physical characteristics. The system undergoes an annual progression from a vertically mixed water mass from late fall to spring, to a strongly stratified system in summer (from June to October). Stratification in the spring is due to freshwater inputs to the system, primarily from coastal river plumes advecting from the north. Temperature is primarily responsible for the summer stratification. The system is strongly affected by wind forcing, which causes upwelling and intrusion of offshore water.

The existing MWRA discharge into Boston Harbor, in conjunction with other coastal nutrient sources, creates an inshore-offshore nutrient gradient. Relocation and discharge through a submarine diffuser will alter the existing nutrient loading scheme. The system's nutrient dynamics are strongly associated with the annual phytoplankton cycle. Coastal influences also produce an inshore-offshore gradient in light penetration into the water column due to higher turbidity and suspended particulate matter inshore. The relative influence of temperature, nutrients and light change with season and distance offshore, and will be modified in the post-relocation scenario.

Phytoplankton. Water column nutrient concentrations reach annual maxima in the winter. As light increases during late winter (February-March), a strong seasonal phytoplankton bloom typically develops which occasionally depletes nutrients throughout the water column. This late winter bloom may be followed by a second event in late April, particularly if the first bloom is not strong and sufficient nutrients remain. The MWRA baseline data indicate that nitrogen is typically the limiting macronutrient in the system, with indications that silicate may become limiting during strong diatom blooms.

As the seasonal thermocline sets up, a strong density barrier is formed and nutrient concentrations in the surface layer diminish due to phytoplankton uptake. However, both productivity and plankton measurements during the baseline period have also shown the summer stratified period to be highly productive. It is a time when phytoplankton (and zooplankton) populations reach their maximum annual numerical densities, suggesting that rapid nutrient recycling is occurring in the surface mixed layer.

Nutrient concentrations in the bottom layer increase throughout the stratified period, largely due to remineralization of the spring bloom's sedimented organic material. The depth of the thermocline typically reaches 15 to 20 meters in the nearfield late in the season. Periodic upwelling and mixing events which occur during the stratified period transport nutrients from below the pycnocline into the surface waters, enhancing summertime phytoplankton productivity. As the surface layer begins to cool and sink in the fall, and the magnitude and frequency of seasonal storms increase, the water column mixes and nutrients trapped in the bottom layer are released. This nutrient release typically produces fall phytoplankton blooms that often exceed the spring event in terms of chlorophyll biomass and productivity.

Although a general overall trend of decreasing productivity and plankton abundance has been documented with distance from shore, peak activity has been frequently documented in the nearfield region. This has been attributed to a transitional ("fulcrum") zone in the nearfield where light (which limits productivity inshore) becomes more available, while nutrients (which limit productivity offshore) are still available in sufficient quantities due to coastal loading.

In terms of species composition, microflagellates are typically the numerical dominant class of phytoplankton in the nearfield throughout most of the year. However, the major blooms in the spring and summer periods are driven by centric diatoms, whereas the fall bloom is additionally dominated by the pennate diatom *Asterionellopsis glacialis*. The spring bloom is largely driven by *Thalassiosira* and *Chaetoceros* species. The summer period includes *Rhizosolenia fragilissima*, *Leptocylindrus danicus* and *L. minimus*, *Chaetoceros* spp., and *Skeletonema costatum*. In addition to *A. glacialis*, the fall bloom includes contribution from centric diatoms such as *R. delicatula* and *Thalassiosira* species. Dinoflagellates also have periodic blooms, including *Gymnodinium* spp. (spring), *Ceratium longipes* (summer), and *Ceratium tripos* (fall).

In addition to the observed gradient in abundance with distance offshore, there is a high degree of spatial and interannual variability in biomass (chlorophyll concentration). Interannual variability in the abundance of seasonally dominant taxa is often two to three orders of magnitude, while variability in major classes (e.g., centric diatoms, dinoflagellates) is around one order of magnitude. However, seasonally dominant species were typically found in each region, although the harbor assemblage did have some species present which were more characteristic of shallow inshore environments (e.g. several pennate diatom taxa, increased densities of cyanobacteria). The harbor and near-coastal regions had more pronounced summertime maxima in terms of chlorophyll and cell densities. Differences in Cape Cod Bay included indications of an earlier winter/spring bloom cycle, and a summer assemblage dominated to a greater degree by flagellated species and a smaller contribution by centric diatoms.

Several nuisance species have been targeted for close scrutiny by the MWRA monitoring program. These include *Phaeocystis pouchetii*, *Alexandrium tamarense*, and *Pseudo-nitzschia multiseries*. Each of these species, which occur in late winter, spring, and fall/winter, respectively, present unique challenges with regard to characterizing historical abundance and distribution as well as assessing their potential response to the outfall relocation.

Zooplankton. Zooplankton abundance and species composition also exhibit strong seasonal and interannual variation. Pronounced shifts in species composition were found with respect to both region and season, with copepods the dominant group of zooplankton. Boreal species were most abundant during the first half of the year, while warmer water species dominated during the latter half.

Regionally, the coastal species assemblage was dominated by *Acartia* spp., while offshore species composition was characterized by *Calanus* and *Pseudocalanus* in winter/spring and *Centropages typicus* and *Paracalanus parvus* in late summer and fall. The ubiquitous cyclopoid copepod *Oithona similis* was dominant away from the coast in both seasons and had highest abundance in the second half of the year. Barnacle nauplii were a dominant group during spring in coastal waters. Although interannual variations were observed in both environmental conditions and zooplankton abundance, the connections between them were not obvious.

The seasonal patterns for most dominant taxa are similar to those found in shelf regions along the eastern U.S. The nearshore/offshore affinities of the various species are also common to other eastern shelf areas. Published data suggest that *Acartia* is food limited in nature and thus is restricted to eutrophic nearshore environments. It is reasonable to hypothesize then that were the discharge of effluent in the nearfield region to cause local eutrophication then it would be manifest as a shift in the zooplankton species composition towards an *Acartia* spp. dominated system.

Recommendations. The MWRA monitoring program has produced an invaluable baseline data set which has contributed immensely to the understanding of plankton dynamics in Massachusetts Bay. The basic structure of the plankton monitoring should be continued prior to outfall relocation to further define the extent of spatial and temporal variability. Post-relocation monitoring should continue this record such that potential changes in community structure can be identified.

Since the highest potential for significant change will be in the nearfield, consideration should be given to increasing the number of stations occupied there. Given the high degree of interannual variability, post-discharge comparisons of phytoplankton data should focus on abundance of major classes or groups (i.e., centric diatoms, dinoflagellates, etc.) rather than individual species. Particular attention should also be paid to the potential changes which occur at or below the pycnocline during the stratified period.

For the nuisance species *A. tamarense*, changes in numerical abundance due to the outfall will be difficult to detect because this species is not well sampled by the monitoring program design. We thus recommend that a statistical treatment of weekly shellfish toxicity data collected at multiple stations within the bay since 1972 be undertaken in collaboration with personnel from the Massachusetts Division of Marine Fisheries. These historical data can provide a baseline against which future levels of toxicity can be compared. In this way, criteria can be established which indicate whether toxicity, and thus *A. tamarense* abundance, have exceeded historical averages sufficiently to indicate outfall-related enhancement of abundance.

A phased domoic acid monitoring program is recommended if future monitoring efforts detect large numbers of *Pseudo-nitzschia*. This phased program would initially entail analyses of particulate carbon samples from the nearfield for domoic acid if increased densities of *Pseudo-nitzschia* are encountered. Once the presence of toxic forms are confirmed, shellfish testing for domoic acid would be warranted. It may be appropriate to perform pilot assays during the fall of 1997 to establish baseline data on the presence of toxic forms, either using electron microscopy or new antibody and RNA-probe technologies to investigate the abundance of toxic *Pseudo-nitzschia* strains and species.

The use of moored sensors in the nearfield is recommended to further define the magnitude and duration of plankton response to the new nutrient loading regime. This approach is warranted given the dynamic nature of the nearfield, particularly during the stratified period. Hopefully the existing WetLabs instrumentation for continuous chlorophyll sensing will become more reliable, otherwise other means of obtaining these data should be investigated. MWRA should confirm the utility and sensitivity of the statistical approach for evaluating chlorophyll thresholds.

Bay-wide high-resolution sampling is recommended as part of farfield monitoring surveys to tie together the scales of variability and provide an understanding of how prevalent small scale features such as copepod patches are in the Bay and how they may be affected (if at all) by the outfall. High-resolution bay-wide towyo-type sampling provide insights into the dominant scales of variability as well as how the zooplankton species are related spatially to each other and to their associated environmental variables. In addition, the continuous data approach could be employed through the use of a moored vertically-profiling VPR now under development. This instrument would provide high resolution temporal data on plankton species abundance and associated environmental variables.

A thorough review and analysis of historical zooplankton data sets, analogous to the Cura study for phytoplankton, should be undertaken to help establish the baseline for zooplankton.

Finally, to determine potential impacts of natural and outfall-induced perturbations on the dominance of zooplankton species, coupled population/physical modeling is recommended. Population models of species dominance should be combined with the three-dimensional circulation model of R. Signell to determine the effects of different forcing scenarios on plankton population abundance patterns.

1.0 INTRODUCTION

1.1 Background

The Massachusetts Water Resources Authority (MWRA) is currently in the process of relocating its treated wastewater discharge from its present location in Boston Harbor to a site approximately 15 km offshore in Massachusetts Bay. Beginning in late 1998, effluent will be dispersed at the new location through a series of 55 submarine diffusers. Together with other engineering improvements in both the wastewater treatment process and in the control of combined sewer discharges to Boston Harbor, the existing configuration of nutrient loading to adjacent offshore waters will change (Galya *et al.*, 1996).

The environmental planning process associated with the permitting of this new outfall required baseline monitoring of the Massachusetts Bay ecosystem, which will continue as post-discharge monitoring once the new outfall is commissioned. In conjunction with effluent monitoring required by the MWRA's discharge permit, the post-discharge monitoring will provide the basis to evaluate threshold criteria developed for contingency planning purposes (MWRA, 1997).

One issue subject to the contingency review will be assessment of the potential for eutrophication in the nearfield water column resulting from the change in nutrient loading. The seasonal physical structure of the water column (summer stratification, seasonal overturn and mixing) influences the manner in which eutrophication might be manifest. Both annual and seasonal chlorophyll concentrations will be used to assess whether an increase in biomass has occurred relative to the baseline data record (MWRA, 1997). However, changes in the phytoplankton community may potentially occur without a significant change in chlorophyll biomass (see Cura, 1991). Such a scenario would entail a change in species composition and successional patterns which might alter the trophic structure in the water column.

A second impact might be an increase in the frequency, magnitude, duration, or location of nuisance algal blooms. Nuisance blooms include the seasonal appearance of red tides, which are caused by a diverse group of dinoflagellates, some of which produce toxic agents which may be accumulated by shellfish or other marine life. Some diatom species also produce toxins. Other nuisance species may cause clogging of fish nets and create aesthetic problems. In general, the potential changes in nuisance species due to the outfall, as for phytoplankton in general, would be earlier occurrence or prolonged persistence beyond typical patterns, or appreciable increases in abundance.

Changes in the phytoplankton assemblage (either biomass or species succession change) may be accompanied by shifts in the zooplankton populations which graze on them, with the further potential for alterations in higher trophic levels. It should be recognized that increased primary and secondary production is not necessarily detrimental to the ecosystem. However, changes in carbon production and utilization patterns could result in dissolved oxygen deficits should increased production be under-utilized and fuel increased microbial respiration. In order to assess the potential for post-operational impacts, adequate characterization of the existing phytoplankton and zooplankton communities in the ecosystem is required.

1.2 Objectives

The principal goal of this Plankton Issues Report is to characterize the baseline of the plankton communities in the Massachusetts Bay system prior to relocation of the MWRA effluent discharge into offshore waters. To achieve this, plankton data from the MWRA Harbor and Outfall Monitoring (HOM) Program, as well as the literature, were reviewed to identify seasonal plankton assemblages in the future outfall area and in adjacent regions of the Bay. Interannual differences were then examined to establish the degree of variability exhibited by these communities. Supporting physical and chemical data from the water column were used, to the extent possible, to explain the observed temporal and spatial variability.

A second goal of this undertaking was to provide the means to assess potential post-discharge effects in the nearfield. By considering the ecological processes which govern plankton abundance and distribution, the baseline was used to develop conceptual models for the projected post-discharge environment. Potential hypotheses for evaluating post-discharge effects on each community were considered. Finally, recommendations to strengthen the plankton monitoring program were developed.

1.3 Report Organization

Following this introductory section, Section 2 presents the overview of the plankton record in the Massachusetts Bay system, including the MWRA HOM data as well as identification of other relevant data sets. Section 3 discusses generic factors affecting plankton abundance and distribution. Section 4 presents MWRA data from 1992-1995, and then compares the MWRA results with the literature.

Section 5 examines the potential control of the observed physical and chemical characteristics of the system (temperature, salinity, nutrients, and fluorescence) on the MWRA plankton record. In Section 6, the conceptual models of post-relocation scenarios are developed to evaluate potential effects on the plankton communities associated with the outfall relocation. Finally, recommendations are given in Section 7 for post-discharge monitoring.

2.0 REVIEW OF PLANKTON LITERATURE

2.1 Phytoplankton

A comprehensive review of the literature for phytoplankton studies in Massachusetts Bay prior to the initiation of the MWRA HOM Program was performed by Cura (1991). This review included results from 80 years of studies in Massachusetts Bay, the adjacent waters of the Gulf of Maine, as well on those generated within Boston Harbor and Cape Cod Bay (the reader is referred to the Cura review for specifics on study dates and sampling locations).

Subsequent to the Cura review, phytoplankton sampling was included in work performed in northern Massachusetts Bay for MWRA by Bigelow Laboratories (Townsend *et al.*, 1990a; 1990b; 1990c; 1991). This study included one survey during October of 1989 and five in 1990 (performed during the months of February, March, April, June, and August). The Cura review, as well as information from Townsend *et al.*, are summarized below.

MWRA has included phytoplankton sampling in its HOM Program since 1992. The phytoplankton results from the MWRA program are summarized in a series of Annual Technical Reports (Kelly *et al.*, 1993; Kelly and Turner, 1995a; 1995b; Cibik et al., 1997). Data from the MWRA program are presented in Section 4 of this report.

2.1.1 Species Succession

Cura (1991) summarized the 80-year record of numerically-dominant net phytoplankton (>10µm) in a series of tables which presented "an idealized floristic year". In these studies, taxonomic characterization of the nannoplankton (<10µm) was seldom attempted. Seasonal assemblages were developed for the regions by including taxa which were dominant in at least one of the 13 studies. This approach was necessary as only one of the studies (Parker, 1980) included monthly sampling, and few included routine sampling during each season.

The seasonal distribution of net phytoplankton based on the Cura review, and whole-water plus net phytoplankton from Townsend et al. (1990a; 1990b; 1990c; 1991), are summarized in Table 2-1. The distribution of dominant taxa in Massachusetts Bay, denoted as region 1 in Table 2-1, is listed by month. Data from Boston Harbor are denoted as region 2, while Cape Cod Bay data are denoted as region 3. Data specific to the Townsend study are marked with an asterisk.

The net phytoplankton are predominately centric diatoms, which in Massachusetts Bay appear to have distinctive seasonal assemblages. The late winter/early spring assemblage is dominated by *Coscinodiscus* and *Thalassiosira*, with the latter comprised of *T. decipiens* and *T. nordenskioldii*. By March, *Thalassiosira* co-dominates with several species of *Chaetoceros*, including *C. debilis*, *C. compressus*, and *C. socialis*. The nuisance alga *Phaeocystis pouchetii* also occurs during the spring. Cura cited a study by Parker (1980) which identified 96 percent of the biomass during this period as being due to the diatoms *Porosira glacialis*, *Detonula confervacea*, *T. nordenskioldii*, *C. debilis*, and other species of *Chaetoceros*.

By May, these dominant diatom species are replaced in Massachusetts Bay by *Skeletonema costatum*, *Rhizosolenia delicatula*, and *Leptocylindrus minimus*. These centric diatoms are joined by peridinian dinoflagellates. By August, *R. delicatula* is succeeded by *R. fragilissima*, and *L. minimus* by *L. danicus*, while *S. costatum* remains as a co-dominant. The dominant dinoflagellates include *Ceratium* and *Gonyaulax*. By October, the summertime assemblage appears to give way to a reappearance of the winter forms *Coscinodiscus* and *Thalassiosira*, along with *Guinardia flaccida*.

Table 2-1 also provides information comparing dominant forms in Boston Harbor (region 2) and Cape Cod Bay (region 3) with that described above for Massachusetts Bay. However, only five studies reported data from Boston Harbor, and three from Cape Cod Bay. Temporal coverage was better than spatial coverage, however, as data from Cape Cod Bay consisted of only a few stations which were weighted toward the Plymouth/Duxbury area.

The results indicate that the Boston Harbor assemblage is generally similar to Massachusetts Bay in its seasonal dominants. Exceptions included the spring occurrence of the pennate diatom *Asterionella glacialis* and the summer occurrence of the centric diatom *Lithodesmium undulatum*. The presence of the dinoflagellate *Katodinium rotundatum* was noted during the summer, a species which often forms non-toxic red tides in coastal estuaries.

Successional patterns in Cape Cod Bay appear to differ from Massachusetts Bay and the harbor largely in the timing of the late winter/early spring assemblage (e.g. *Thalassiosira* and *Chaetoceros* in Table 2-1) and the composition of the summer assemblage. Limited sample coverage prevents more definitive conclusions. Noteworthy is the almost year-round presence of *Skeletonema costatum*.

2.1.2 Phytoplankton Abundance

In terms of phytoplankton biomass as measured by chlorophyll concentration, the Cura review indicated that the phytoplankton community in Massachusetts Bay undergoes a bimodal annual cycle typical of temperate coastal waters. This cycle is characterized by large increases in abundance (blooms) which occur in late winter/early spring and again during the fall. The spring bloom is produced by increasing light levels and ample nutrients following winter mixing, while the fall bloom results from the breakdown in water column stratification and transport of nutrients from the bottom water to the seasonally-depleted photic zone. Low abundances in summer are interspersed with short-lived blooms.

Quantitative data for the seasonal dominants described in the previous section are presented in context with the MWRA results in Section 4. It should be noted that abundances for individual taxa vary widely by location, month/season, and study. Where practical, seasonal densities from the literature are compared with comparable periods from the MWRA program.

One general observation from the review is that the nannoplankton fraction of the phytoplankton in Massachusetts Bay typically comprises over 90 percent of the number of cells. For example, spring assemblages yielded maximum netphytoplankton plankton densities between 10^5 and 10^6 cells/L, while nannoplankton densities were often greater than 10^7 cells/L. Summertime netphytoplankton densities between 10^4 to 10^6 cells/L were associated with nanoplankton counts of 10^5 to 10^8 cells/L, although occasional results (often in July) yielded almost equivalent proportions. Absolute numbers can be misleading, however, as nanoplankton cell volume (and biomass) can be several orders of magnitude lower than netphytoplankton.

The studies reviewed by Cura indicated that the relative dominance by nannoplankton was more pronounced in inshore than in offshore waters. Another generality that emerged is that total abundances of all phytoplankton decreased with distance offshore, with highest densities in the harbor, followed by coastal stations, and lowest densities offshore. Harbor densities were generally two to five times those found in Massachusetts Bay.

Townsend *et al.* (1991) described a transitional area ("fulcrum point") in the vicinity of the future outfall where total cell densities were enhanced by increased light availability (resulting from decreased turbidity from coastal sources) and sufficient nutrient residual from inshore areas. Townsend also attributed differences in species composition to light quantity and quality, as well as nutrient regimes, along the inshore-offshore gradient.

2.1.3 Nuisance Species

Three nuisance species have been identified for scrutiny in post-discharge monitoring: *Alexandrium tamarense, Pseudo-nitzschia multiseries*, and *Phaeocystis pouchetii*. While other species typically found in the seasonal phytoplankton assemblages in Massachusetts Bay have been associated with harmful bloom conditions in other regions (e.g., *Ceratium* spp. and *Dinophysis* spp.), only the former three species have historically been shown to be of local concern.

Alexandrium tamarense (and the closely related species *A. fundyense*) is a dinoflagellate which can cause paralytic shellfish poisoning (PSP) when shellfish consume the algae and concentrate its toxins. In Massachusetts Bay it is associated with the spring assemblage, typically occurring between April and June (Anderson and Keafer, 1995). It has formed blooms off the north shore of Massachusetts ever since a "massive" bloom occurred in 1972 in the coastal waters off Maine, New Hampshire, and Massachusetts (Anderson *et al.*, 1982; Franks and Anderson, 1992). The transport of this species into Massachusetts Bay (see below) can bring it directly over the future outfall site. A related concern is that *A. tamarense* undergoes diel vertical migration, giving it access to nutrients below the pycnocline where the outfall effluent will be discharged.

Its seasonal presence may arise from two sources: locally from in situ germination of benthic cysts, and from advective transport of established blooms in a buoyant plume from river discharge in the north (Anderson and Keafer, 1992; Franks and Anderson, 1992). Once established, its abundance and distribution (and potential for producing PSP outbreaks) are governed by nutrient availability, wind patterns, and coastal current behavior. These physical controlling factors may alternatively disperse the population offshore or lead to conditions of increased residence time which allow the species to persist and proliferate. Densities vary accordingly, with low densities (i.e., less than 100 cells/L) in one year followed by high densities (>1,000 cells/L) the next.

Another toxic taxon with documented occurrences in this system (Villareal *et al.*, 1994) is the pennate diatom *Pseudo-nitzschia multiseries* (formerly *Nitzschia pungens forma multiseries*, then *Pseudonitzschia pungens f. multiseries*, Hasle, 1994; 1995). This species has been implicated in outbreaks of amnesic shellfish poisoning (ASP) in coastal waters off eastern Canada (Bates *et al.*, 1989) and along the west coast of the U.S., and was potentially responsible for elevated levels of the toxic agent (domoic acid) which were detected in monitoring of Nantucket shellfish in January-February of 1991 (Nassif and Timperi, 1991).

Its seasonal distribution in local waters appears from the literature to be from November to March, but this is highly uncertain. The difficulty with evaluating its distribution and abundance, both from the literature and the existing monitoring program, is that *Pseudo-nitzschia pungens* co-occurs with *P. multiseries* and is similar in appearance under light microscopy at even the highest magnifications. Relative proportions of the two can vary dramatically over time (Smith *et al.*, 1990). Taxonomic separation of the two species requires either examination of the cell structure via electron microscopy, or the use of antibody or nucleic acid probes (Anderson, 1995; Bates *et al.*, 1993), neither of which is feasible in a large monitoring program.

P. pungens from the northeastern U.S. has not been shown to produce domoic acid. To complicate matters further, *P. multiseries* has non-toxic strains, and greater scrutiny of other species of *Pseudo-nitzschia* is revealing domoic acid production in some strains (e.g., *P. seriata* in Europe by Lundholm *et al.*, 1994, *P. pungens* in New Zealand by Rhodes *et al.*, 1996). For this reason, routine phytoplankton monitoring using light microscopy can only report the potential presence of toxic forms of this genus.

Phaeocystis pouchetii can be present either as a single cell or can form visible colonies (around 1mm diameter) in which the single cells are imbedded in a gelatinous material. When present in its colonial form, it can cause aesthetic problems such as thick deposits of foam on beaches, and can clog fishing nets. It is typically reported during the months of February through April (Table 2-1) in densities reaching a few million cells per liter. Phaeocystis is best known for the substantial problems it causes along the beaches of western Europe. It goes through cycles of increased abundance and seems to thrive when nutrient inputs from anthropogenic sources are high. Anecdotal evidence (Mayo, pers. comm.) suggests that large *Phaeocystis* blooms may affect the feeding behavior of the Northern Right whale, and may even cause the whales to leave Cape Cod Bay earlier than in years without blooms.

2.2 Zooplankton

Several studies of zooplankton in the Bays region have been conducted, beginning with Bigelow's cruises from 1912-1924 (Bigelow, 1926). Early surveys were non-quantitative since no flow meters were used in the nets. Subsequent studies have included the Continuous Plankton Recorder (CPR) Surveys which were conducted bimonthly from 1961-present, and include two processed samples in the Bays region per month during this period (Jossi and Goulet, 1990). These data have not been analyzed.

A zooplankton survey was conducted in eastern Cape Cod Bay as part of an impact assessment for the siting of the Pilgrim Nuclear Power Plant in Plymouth, Massachusetts (Toner *et al.*, 1984; Horst *et al.*, 1984; Bridges *et al.*, 1984). The survey found that *Acartia tonsa* and *A. clausii* (now *A. hudsonica*) were dominant inshore and that bivalve larvae were often dominant offshore. They found a seasonal trend with higher concentrations of zooplankton in the warmer season (August) but a minor peak in May following the spring phytoplankton bloom.

The zooplankton results from the MWRA HOM Program are summarized in a series of Annual Technical Reports (Kelly *et al.*, 1993; Kelly and Turner, 1995a; 1995b; Cibik *et al.*, 1997). Turner *et al.* (1994) described the MWRA 1992 zooplankton data and pointed out that the small cyclopoid copepod *Oithona similis* was the most abundant zooplankton species, with the large *Calanus finmarchicus* having a relatively insignificant abundance. As will be shown in this review, however, *Calanus finmarchicus* dominates the zooplankton biomass at nearly all locations in the Bay during the first half of the year due to its large size.

A few short-term focused studies have been done in the Bays region as well. These include studies of zooplankton lipid composition (Jefferies, 1987), grazing on red tide phytoplankton by larval polychaetes and tintinnids (Watras *et al.*, 1985), foraging by the North Atlantic right whale on copepod patches in Cape Cod Bay (Mayo and Marx, 1990), and the effect of internal wave packets on patchiness in zooplankton over Stellwagen Bank (Haury *et al.* 1983). In general, only a limited amount of information has been published on the zooplankton of the Bays region and this information is largely restricted to such focused or short-term studies. Data from the previous surveys remain largely unsynthesized. One of the recommendations of this report is for a thorough review and analysis of historical data sets.

TABLE 2-1
DOMINANT PHYTOPLANKTON SPECIES

	J	F	M	A	M	J	J	A	S	O	N	D
CENTRIC DIATOMS												
Coscinodiscus spp.	2	1, 2			2	2	2	2	2	2	1, 2	1, 2
Thalassiosira spp.			1, 2	1, 2				1*			1	3
decipiens	1	1	1, 2	1, 2								
nordenskioldii		1	1, 2	1, 2								
pseudonana		1*	1*, 2*									
Chaetoceros spp.			1	1, 3	3	1*	3	1*		1		
debilis			1	1		1*						
decipiens		1*,2*	1*, 2*									
compressus				1		1*						
socialis				1		1*						
Porosira glacialis			1	1								
Detonula confervacea		1*, 2*	1, 2*									
Cerataulina pelagica								1				
Skeletonema costatum	3	3	2, 3	1, 2, 3	1, 3	1	1, 3	1, 3	1	1, 2, 3	2, 3	2, 3
Leptocylindus spp.												
minimus					1, 2	1*2,3	1, 2	1*		2, 3	3	
danicus							1, 2	1, 2, 3	1, 2			
Rhizosolenia spp.	2											
delicatula						1	1			1		
fragillissima					2	2	1, 2	1, 2	1, 2	1		
alata		3						1, 3			1, 3	3
setigera										1		
stolterfothii								1*		1*		
Lithodesmium undulatum							2					
Guinardia flaccida										1*	1	1
PENNATE DIATOMS												
Asterionellopsis glacialis			2	2								
Cylindrotheca closterium	1	1	1*				3					
Thalassionema nitzschoides						1*				1*		
Nitzschia spp.						1*		1*		1*		
Thalassiothrix spp.		1*	1*									
DINOFLAGELLATES												
Peridinium spp.					3	1, 3	1				1	
Ceratium spp.								1		1		
Gonyaulax spp.									1			
Katodinium rotundatum					2	1*, 2	2	1*, 2	2			
Prorocentrum spp.						1*				1*		
Gymnodinium spp.		1*	1*	1*		1*		1*				
OTHERS												
Cryptophytes			1*, 2*	1*, 2*				1*		1*		
Phaeocystis pouchetii		1*	1, 2*	1, 3								
Emeliania huxleyi										2	2	2
Nannochloris atomus										2,3	2,3	2,3
Phytoflagellates*		1*, 2*	1*, 2*	1*, 2*				1*		1*, 2*		
Dictyocha fibula										1*		

Notes: 1 = Massachusetts Bay
2 = Boston Harbor
3 = Cape Cod Bay

* = Occurrences exclusive to Townsend et al., 1990a,b,c

3.0 FACTORS CONTROLLING PLANKTON COMMUNITIES

This section provides an overview of the physical, chemical, and biological factors controlling the plankton populations in the system. The MWRA monitoring results for physical and chemical parameters are presented for the 1992-1995 period to provide context for the plankton results presented in Section 4.

3.1 Phytoplankton

The abundance and species composition of the phytoplankton community result from the opposing interactions of cellular growth and division versus population loss rates. The former are determined by the complex interplay between organism physiology and environmental conditions such as temperature, light, salinity, nutrients, and turbulence, and the latter by loss processes such as advective transport, grazing, and pathogen-induced mortality. Community composition is the end result of the differential growth and loss rates of individual species within the plankton assemblage. The diversity of species, each with its own unique set of environmental tolerances and requirements, and the diversity of grazers and grazing preferences combine to make a system which is exceedingly complex and difficult to parameterize and model.

Light. Phytoplankton must grow within the illuminated surface waters of the ocean, so light can be considered the ultimate limiting factor for cellular growth. Although many phytoplankton species can photosynthesize at relatively low light intensities, primary production is ultimately limited by depth, given the attenuation of light in the sea. Of particular importance in this regard are the relative depths of the euphotic zone and the mixed layer. When the euphotic zone reaches below the pycnocline, as occurs sometimes in Massachusetts Bay for example (Smayda, 1992), nutrients in deeper layers (such as those which will be supplied by the new outfall) can be utilized by phytoplankton and benthic algae. The implications of this will be discussed below.

It is of obvious adaptive significance for phytoplankton to modify their photosynthetic response to ambient light intensity so as to maximize their growth potential under various light conditions. This adaptation may be accomplished by modifying the light and/or dark reactions of photosynthesis. Thus a cell which finds itself near the bottom of the euphotic zone will grow slowly due to light limited conditions, but will make the most of what light energy is available by producing a more efficient light-capturing system. Typical adaptation observed in cultures involves a two- to ten-fold increase in chlorophyll content per cell (Smayda, 1980). Some species are notably better at this type of low-light adaptation than are others.

At high light intensities, cell growth is limited by the rate at which carbon is fixed, which in turn depends on cell metabolism. Thus cells in bright light devote less energy and nutrients to chlorophyll synthesis and more to the synthesis of enzymes that can increase carbon fixation. Here again, species differ dramatically in their tolerances and adaptation to high light intensities.

Various wavelengths of the visible spectrum of light are differentially transmitted and absorbed in water. The rapid loss of the longer wavelengths of radiant energy entering the sea is accommodated by the accessory pigments of phytoplankton which permit the use of those wavelengths which are not absorbed. The differential ability of various species to utilize particular wavelengths is another determinant of the phytoplankton community composition. Light is thus a fundamental controlling factor in phytoplankton ecology, affecting the rate of carbon fixation and cell growth, the species composition, and the spatial structure of that community.

Temperature. The second major environmental factor that affects cell and population growth is temperature. Since the temperature of relatively large bodies of water is relatively constant on a diel basis, phytoplankton ecologists are more concerned with seasonal and latitudinal impact of temperature rather than with daily temperature fluctuations. One of the most important effects of temperature on phytoplankton is indirect through its impact on water column stability. Temperature is an extremely important variable in mixing processes because of its well known affect on the density of water and therefore on the stability of the water column.

During the winter, temperature (and therefore density) differences between surface and deeper waters usually are not sufficient to prevent complete mixing by the wind. As surface waters are warmed in spring and early summer, they become less dense and less likely to be mixed with deeper, colder and denser water. Eventually, density differences result in stratification, which severely restricts the mixing of the deeper, nutrient-rich water with the surface mixed layer where phytoplankton growth typically has stripped the water of nutrients. In that situation, two important requirements for algal growth - light and nutrients - are most abundant in separate water masses. Stratification breaks down in the autumn when surface waters cool and deep mixing resumes.

Phytoplankton are directly affected by temperature in that growth rates typically increase with increasing temperature, up to some optimum after which growth rate declines, often abruptly (Eppley, 1972). The temperature optima for growth of many phytoplankton species differ, some being exceptionally tolerant to a wide range of temperatures, whereas others are restricted to a narrow temperature "window" (Smayda 1980). The diatom *Skeletonema costatum*, for example, is eurythermal and capable of growth between 0° and 30°C, whereas *Detonula confervacea* ceases growth above 15°C. Such species differences are clearly a major factor in phytoplankton species succession.

One feature that reflects the interaction between light and temperature is the so-called "chlorophyll maximum" - a layer of chlorophyll containing material at the base of the euphotic zone or pycnocline. The subsurface maximum, often comprised of shade-adapted cells with high chlorophyll content, can utilize the high nutrient concentrations in the colder layer below, but this depends on the rate of vertical diffusion. The chlorophyll maximum can also result from the accumulation of sedimented material, including dead or dying cells, which are retained at that point due to density differences of the water. In that instance, there can be high chlorophyll levels, but low productivity.

Salinity. Experimental and field observations demonstrate that phytoplankton species collectively grow well over a wide range in salinity (e.g., McLachlan, 1961). Given this euryhaline nature of the phytoplankton and the relatively modest seasonal fluctuations in salinity in open coastal waters, salinity is not often a regulatory factor in species succession in such areas. Notable exceptions to this generalization occur in regions subject to extensive seasonal inputs of freshwater in the form of runoff and snowmelt. For example, the toxic dinoflagellate *Alexandrium tamarense* is known to thrive in the low-salinity plume of the southwestern Maine coastal current, which enters Massachusetts Bay. Although the salinity of that water mass is only one or two psu lower than ambient Gulf of Maine waters, the plume's buoyancy and its chemical composition (including its salinity) create a suitable growth environment for this dinoflagellate (Franks and Anderson, 1992). *Alexandrium* will grow at the salinities of the open Gulf of Maine and Massachusetts Bay, but does prefer the slight dilution of the coastal current.

Nutrients. Phytoplankton must obtain a range of substances from their environment in order to sustain growth and division. Some of the universally required elements (carbon and ions such as sodium, potassium, and magnesium) are available in abundance relative to what is needed. In addition to these abundant constituents, phytoplankton require a variety of elements which are present at much lower concentrations in seawater. The most important of these are the macronutrients nitrogen, phosphorous, and silicate. Nitrogen and phosphorous are essential for all algae. The requirement for silicate is limited to those groups which form siliceous skeletons such as the diatoms and silicoflagellates.

Certain other trace elements are essential for phytoplankton growth as well, including the metals Fe, Mn, Zn, Cu, and Co which are incorporated into essential organic molecules, particularly a variety of enzyme cofactors involved in photosynthetic reactions. Requirements for trace metals are generally very low, and despite the low concentrations of many in seawater, they will normally be present in considerable excess of the amounts needed by the phytoplankton. Iron, however, is present largely in forms which are not readily available, and thus may be limiting under certain situations. Little is know about the complex relationships between chemical speciation of metals and their biological availability. It is known that certain organic molecules complex (chelate) metals and thus influence their availability. These organic materials can be produced by the algae themselves or can be supplied by anthropogenic sources such as pollution or terrestrial runoff.

Nitrogen is considered the most important of the phytoplankton nutrients in coastal waters, though there are situations in which phosphorous can be limiting instead. Massachusetts Bay is considered to be nitrogen-limited. Most phytoplankton take up the inorganic forms of nitrogen (nitrate, nitrite, and ammonium), although there is growing evidence that many species are also able to utilize organic forms such as urea and amino acids.

There are advantages in terms of physiological energetics when nitrogen is taken up as ammonium, since this form can be used directly in the synthesis of amino acids whereas nitrate must be reduced. However, an extensive comparison by Berland *et al.*, (1976) of the growth of 26 species of marine phytoplankton on various nitrogen sources showed that all grew well on both ammonium and nitrate. Guillard (1963) observed that growth on nitrate can be reduced by the presence of ammonium, an effect explained by the repression of the nitrate reductase system when high concentrations of ammonium are present. This effect is unlikely to be significant unless concentrations of ammonium are higher than 1μM. It is thus clear that many algae are able to use a variety of nitrogen sources at low environmental concentrations. Variations in the relative proportions of different forms of nitrogen between environments or at different times in a particular environment may be critical in influencing the species composition and succession of a community.

Assimilation of phosphorous by phytoplankton generally involves the uptake of inorganic orthophosphate, but again it appears that organic phosphorous compounds can be utilized. Silicon, required predominantly by diatoms, is present in seawater as silicate in utilizable form. The drawdown of silicon during blooms can result in conditions which are limiting to diatoms but not to other species which have no requirement for this element.

One aspect of this differential utilization of nutrients that is important in the context of Massachusetts Bay and the new outfall is the concept of nutrient ratios. Smayda (1989) has argued that one reason for an apparent proliferation of harmful algal species throughout the world relates to the anthropogenic loading of coastal waters with effluents which contain different ratios of the major nutrients (nitrogen, phosphorous and silicon) relative to the receiving waters. This can create situations in which diatoms (which in most cases are harmless) run out of silicon and cease growth, whereas other species, typically flagellates, are able to utilize the available nitrogen and phosphorous and thus can dominate the community. Since there are more harmful species within this latter group of phytoplankton, changes in nutrient ratios can potentially be linked to the proliferation of red tides and harmful algal blooms. Variations in N:P ratios due to differential nutrient loadings from land, vertical mixing of nutrients, and utilization of these elements by phytoplankton can thus have a dramatic impact on the physiology and species composition of the phytoplankton.

The mathematical treatment of phytoplankton growth in terms of nutrient uptake kinetics gives a basis for understanding the competition between species for nutrient resources that is needed to explain species diversity and succession. This is a complex topic far beyond the scope of this review, but a few generalities are possible. Species vary dramatically in their requirements and uptake capabilities for the major nutrients. Some, such as the harmful species *Phaeocystis pouchetii*, thrive when nutrients are high. High nutrient concentrations offset the inefficient uptake of nutrients characteristic of the gelatinous colonial life stage. At lower nutrient concentrations, the free-living forms of *Phaeocystis* are typically observed. Since it is only the large colonies which cause harm to coastal ecosystems and fisheries, nutrient loadings can have a dramatic effect on the impact of this species.

The key issue here is a species' "affinity" for nutrients. Models of nutrient uptake by phytoplankton have followed the well known Michaelis Menten Equation (Eppley *et al.*, 1969) in which one of the parameters, Ks (the half-saturation or affinity coefficient) is considered an index of a species' potential competitive ability at low nutrient concentrations. The argument has been that species with high Ks coefficients will dominate in high nutrient regimes. Smayda (in press) has tabulated values for various phytoplankton classes and argues that diatoms generally have lower Ks coefficients than dinoflagellates for example, suggesting that diatoms would be most competitive at low nutrient environments. In fact, the opposite is often observed, as diatoms frequently occur in nutrient rich waters such as during the spring bloom following winter overturn. Clearly, other factors (such as stratification and sinking) can override the role of nutrients in species succession.

These inconsistencies highlight the difficulty of addressing nutrient dynamics of phytoplankton on this theoretical basis. Although many species have been characterized with respect to their uptake capacity through laboratory experiments, it is still not possible to predict competitive outcome on the basis of this information. One would like to take the expected changes in Massachusetts Bay nutrient levels due to the outfall and predict the associated changes in phytoplankton species composition or abundance, but this is simply not possible given present capabilities. We are thus left with empirical generalizations such as those of Margalef (1978) who argued that diatoms would dominate turbulent, high nutrient regimes, and dinoflagellates and other flagellates would dominate under stratified conditions. After decades of work on this problem, Margalef and his colleagues have concluded that the most we can hope for is to understand or predict the succession of phytoplankton classes, i.e., that predictions of the dominance of individual species are unlikely.

Turbulence. Numerous field observations have established that flagellate species tend to predominate in stratified water and diatoms are characteristic of more turbulent waters. Both regionally and seasonally, turbulence can thus be a major factor in determining the structure, organization and succession of phytoplankton communities (Margalef, 1978). A particular effect of turbulence is not through its influence on the mean light intensity received by the vertically mixed cells, but rather the influence of hydrodynamic factors, such as shear, on cellular integrity, metabolism, and nutrient uptake. Forced motions by turbulence may lead to cellular damage including cell rupture and loss of flagella, as is observed for many sensitive dinoflagellates (Berdalet, 1993). White (1976) has shown that cell division in laboratory cultures of the toxic dinoflagellate *Alexandrium* is influenced by the frequency and intensity of agitation. The extent to which turbulence influences species succession has not been investigated experimentally. The degree of turbulence is known to influence diatom colony size and viability in natural populations (Schone, 1970).

Another mechanism by which turbulence can affect phytoplankton is through the enhancement of vertical diffusion of nutrients from deeper waters to the impoverished surface layer. Geyer (unpub. data) has shown that in Massachusetts Bay, stratification profoundly affects vertical exchange of nutrients. Internal mixing is slow within the Bay, occurring on time scales of 70 days, which is comparable to the transit time of water through the Bay. Wind-driven mixing and turbulence will thus dominate the exchange of nutrients from deeper layers to the surface. Of particular importance in this regard are upwelling conditions which will bring nutrients to the surface in episodic pulses. These may be responsible for the summer "bloomlets" which are often observed in Massachusetts Bay.

With respect to population dynamics of the phytoplankton, upwelling events will provide pulses of nutrients which can stimulate growth in an episodic fashion, whereas deep nutrients are accessible only to vertically migrating phytoplankton, to those capable of accumulating at the pycnocline, and to those existing below the pycnocline at times when light levels penetrate well into the water column. Smayda has emphasized this latter possibility as an important and generally unanticipated outcome of the release of outfall nutrients into deeper waters. An assessment of the probability of this occurring, or of the magnitude of the growth and carbon deposition that might occur as a result, are not possible in this analyses. Here we only emphasize the potential for these interactions between outfall nutrients and the phytoplankton and identify areas that are suitable for further study.

3.2 Zooplankton

Zooplankton abundance in any given location is determined by a combination of in situ population growth and immigration/emigration via lateral exchange. Population growth is determined by development rate, somatic growth, egg production, and mortality. Development, growth and fertility are all affected by temperature and food availability. Mortality is caused largely by predation, which in turn is affected by predator abundance and consumption rates (also temperature-dependent). Immigration/emigration is determined by physical transport of the animals in and out of the area by advection and diffusion.

Each of these factors is influenced by variability in environmental forcing. Changes in temperature can have both direct and indirect effects on zooplankton population abundance. Direct effects include limitations in the rates of egg production and survival. Indirect effects include limitations to growth and development rates which in turn affect final adult size and generation time. Adult size is related to egg production rate and generation time determines the turnover rate of the population. Colder temperatures will lead to slower growth of the population but will also slow predatory consumption rate.

Seasonal changes in temperature are very pronounced in the Massachusetts Bay region due to the continental climatic conditions. The changes in sea surface temperature (0-20°C) are very large compared to other regions of the world ocean (e.g., Davis, 1987). These large seasonal changes cause pronounced shifts in the species composition of the zooplankton, from boreal species during winter/spring to warm water species in summer/fall. The seasonal succession of offshore species composition mirrors that found on Georges Bank (Davis, 1987).

Unlike temperature, changes in salinity, within the tolerance range of a given species, typically have little effect on their reproduction or growth rates. Likewise the amount of ambient light has little direct impact on zooplankton vital rates. These factors, however, can be associated indirectly with patterns of zooplankton abundance. Temperature and light can affect the vertical distribution of zooplankton which in turn can affect feeding rate and advective transport. During the stratified period, a subsurface phytoplankton maximum occurs, within which turbulence is low and zooplankton species can migrate to feed at elevated rates. This food-rich environment enhances growth and reproductive rates. Ambient light levels and cycles serve as cues for vertically migrating zooplankton. Vertical migration may be diel, ontogenic, or seasonal. Some species stay near the surface or in the chlorophyll maximum both day and night, while others migrate between different depths to maximize food intake but avoid visual predators.

Food availability can limit the overall distribution of certain species of zooplankton such as Centropages typicus (Davis and Alatalo, 1992), whereas within certain areas such as Georges Bank, food may not be limiting (e.g. *Pseudocalanus* sp., Davis, 1984a,b). The role of food limitation versus predation in controlling zooplankton populations is an active area of research in marine zooplankton ecology (Kleppel *et al.*, 1996). It is generally known that protozoans are the dominant grazers of phytoplankton in the sea, but that at certain times such as during the spring diatom bloom, copepods (the dominant zooplankton taxa) are the dominant grazers. It is also well established that copepods feed on protozoans as well as on phytoplankton, and that protozoans provide an important nutritional source for copepods. The traditional view that temperate marine zooplankton production is tied to the spring diatom bloom no longer holds, rather the highest production occurs during the warmer months when turnover rates are higher even though zooplankton biomass is lower.

Episodic events such as upwelling and freshwater plumes can lead to phytoplankton blooms that provide food-rich environments for the zooplankton. Such events would be most beneficial to the zooplankton during the warmer stratified period when nutrients are limiting. In the nearfield region of the future outfall site, nutrients are supplied to the surface waters by lateral advection from Boston Harbor. Once the outfall goes on line, it is possible that the surface waters in the nearfield region will have lower nutrients during the stratified period. There may be a stronger subsurface chlorophyll maximum due to the larger nutrient gradient at the pycnocline. Possible changes in phytoplankton cell size distribution due to changes in the nutrient regime are not likely to affect zooplankton production, since zooplankton are generalist feeders and can consume a wide range of cell shapes and sizes. Zooplankton production would be reduced if the nutritional quality of the food species becomes poor.

Food availability is actually determined by the encounter rate of an organism with its prey. Encounter rate is a function of prey and predator concentrations as well as their swimming speeds and the microscale turbulent motions that bring them together. At low turbulence levels, encounter rates can be high since micropatches of food are able to form, and the consumer can forage within these patches. At intermediate levels of turbulence, the patches are dissipated and encounter rate is reduced. At higher levels of turbulence the encounter rate may increase again due to the rapid rate at which the physical motions bring predator and prey together. If turbulence is too high, feeding behavior may be disrupted.

Strong wind events during the stratified period may thus have several effects. Existing micropatchiness would be destroyed and encounter rate would first decrease. This rate would then increase with wind speed and possibly decrease again if the winds become too strong. Vertical mixing of the water column, or the possible occurrence of upwelling, would result in nutrient enrichment of the surface layer. Following the wind event, stratification would return and the subsurface phytoplankton maximum would reform, again creating a food-rich environment for the zooplankton. Enhancement of these mixing events by increased bottom water nutrients from the future outfall could result in pronounced increases in phytoplankton biomass and creating an even richer feeding environment for the zooplankton. Opportunistic food-limited zooplankton species, such as *Acartia* spp., may benefit greatly from such events.

Physical transport is a dominant factor affecting zooplankton abundance. The generation times of zooplankton are long enough to be significantly affected by physical transport through the Massachusetts Bay region. During winter/spring for example, the generation time of the dominant copepod *Calanus finmarchicus* is about two months. Since the transit time for particles through the Bays region is of similar order, the distribution of this species is clearly affected by advection. During the warmer months the generation times of the warmer water species is on the order of two weeks, so that in situ growth in the region may be relatively more important for these species. Wind-forced intrusions of Gulf of Maine water into Massachusetts Bay, as well as coastal freshwater plumes moving into the region, are expected to be important sources of zooplankton to the Massachusetts Bay standing stock. Further biological/physical modeling work is needed to determine the relative importance of lateral exchange versus in situ growth in determining the size of zooplankton populations in the Bay.

3.3 Overview of Massachusetts Bay Physical and Chemical Data

MWRA physical and chemical monitoring data were reviewed to provide context in which the subsequent plankton data analysis can be interpreted. Nearfield data, collected in higher frequency than farfield data, are summarized below to support the interpretation of the nearfield plankton cycle. Data from the six combined nearfield/farfield surveys in each year are summarized to support the regional comparisons of plankton populations.

3.3.1 Nearfield

Massachusetts Bay is a seasonally stratified coastal water body which is influenced by the greater Gulf of Maine circulation (Geyer *et al.*, 1992; Signell, 1997). Stratification is driven by salinity in the spring and temperature during the summer (Figure 3-1). Freshwater input to the system occurs largely from the coastal plume which originates in the discharge of large rivers to the north. This coastal plume is a source of advective transport of nutrients and plankton into the system. Surface salinities in the nearfield are typically greater than 29 psu, whereas bottom salinities (average nearfield depth around 30m) exceed 31 psu (Figure 3-1a). Salinities in the harbor are similar due to its high degree of flushing and relatively low discharge rates from its tributaries.

Salinity-driven stratification predominates until the surface water begins to warm in April. Minimum surface temperatures in the nearfield (ca. 1-2°C) typically occur in late February or March, whereas maximum surface temperatures (ca. 20°C) were recorded in August (Figure 3-1b). Bottom temperatures continue to rise until turnover, reaching maximum temperatures of around 12°C. Thermal stratification is typically strong from mid- to late-May until the fall turnover in October or November, after which the water column remains mixed until the following spring (Figure 3-1c).

The seasonal stratification cycle contributes to the presence of a recurrent nutrient cycle, illustrated in 1992-1995 results for nitrate-nitrite (Figure 3-2), dissolved inorganic nitrogen (DIN, Figure 3-3), phosphate (Figure 3-4), and silicate (Figure 3-5). This cycle is characterized by:

- relatively high winter nutrient concentrations which are uniform with respect to depth;
- reduction in nutrient concentrations in late winter (February) prior to stratification, which sometimes reaches the bottom, due to algal activity;
- a rebound in concentration (typically during April, but note exception in 1992); and
- low surface concentrations from May to October, followed by complete water column mixing.

The relative availability of nitrogen to phosphorus is illustrated by plotting the DIN:PO4 ratio (Figure 3-6). Nitrogen was shown to be typically lower than the Redfield ratio of 16:1 (Redfield, 1958), ranging from around 10:1 in the winter to as low as 2:1 during the summer. Nitrogen limitation was also evident during the late winter nutrient drawdown. Potential growth limitation of diatoms by low silicate concentrations may also occur at various times throughout the year (e.g., Figure 3-5, late winter and summer 1994; late summer/fall, 1993).

The observed depletion of nutrients in the surface mixed layer during the summer relative to the bottom waters has implications in the relocation of the MWRA discharge. The existing nutrient loading from Boston Harbor produces a diminishing nutrient gradient (from inshore to offshore) which is largely a surface effect. The post-relocation nutrient loading is expected to be trapped below the pycnocline during the stratified period, in effect reducing nutrient concentrations in the surface mixed layer and increasing the relatively nutrient-rich bottom water layer.

Another aspect of the seasonal water column structure is the potential for release of bottom water nutrients to the surface during upwelling events and during temporary mixing events caused by storms. Persistent westerly winds have been shown to produce coastal upwelling along the entire length of Massachusetts and Cape Cod Bays (Geyer *et al.*, 1992), which may result in enhanced localized phytoplankton productivity, including in the nearfield (Cibik *et al.*, 1997). Persistence of undisturbed water column stratification may therefore be useful in explaining the occurrence of short-term summer blooms. From this perspective, 1994, followed by 1995, would appear to have lower potential for summertime blooms than the previous two years (Figure 3-1c).

3.3.2 Farfield

Monitoring of Massachusetts Bay regions beyond the nearfield during the 1992-1995 period revealed several notable spatial and temporal differences in water properties. These are summarized below for the time frames where combined nearfield/farfield surveys were conducted.

February. Surface temperature was colder in 1994 than in other years by more than 2°C. Temperatures throughout most of the region were less than 2°C, with the 2.0°C isotherm well offshore in 1994 but near the coast in the other years. Satellite derived sea surface temperature (SST) images also revealed colder water extending further offshore in 1994 relative to other years. Warmer Gulf of Maine water was further offshore during 1994 than other years with only the boundary station F27 having a temperature greater than 3°C.

Temperature was highest in 1995 with the values over 4°C offshore and a strong gradient from the coast where temperatures were near 1.5-2.0°C. Coastal waters were warmest in 1992 with temperatures near 2.5°C. This pattern was evident in both the surface temperatures as well as in the "Boston - Nearfield" (BN) transect vertical sections (F30, F23, F24, N20, N16, F19; see Figure 4-1).

Salinity gradients mirrored those for temperature. The BN vertical sections again indicated that the warmer saltier water was well offshore in 1994. Fluorescence was higher in 1992 than in the other years by over a factor of four (see Section 4.1 for further discussion of chlorophyll data), and lowest in 1993 during this period. The transect data revealed higher fluorescence offshore in 1994 than in 1993.

March. Temperatures remained coldest in 1994 with >2.0°C surface water found only at two boundary stations. In 1992 and 1995, >2.0°C water was present close to shore and in 1993, this "warmer" water was present well into Massachusetts Bay. The 2.0°C isotherm was well offshore in 1994 but inshore in other years. Temperatures in Cape Cod Bay waters were >2.0°C in 1992 and 1995, near 1.5°C in 1993, and -0.8°C in 1994.

Surface salinity had no obvious trends except that 1994 was the only year with <32.0 psu in Cape Cod Bay. The BN transects show that 1994 was well-mixed relative to 1993. Fluorescence was higher in 1994, with subsurface levels of 4.0 μg chl/L near the outer nearfield. This higher fluorescence in the nearfield in 1994 was associated with low silicate and nitrate, suggesting an early diatom bloom (Cibik *et al.*, 1997).

April. Surface temperatures were similar between years, but transect data revealed stronger stratification with cold bottom water in 1994 and 1993. Coldest surface temperatures occurred in 1993 with temperatures within all regions <4.0°C, while temperatures in other years averaged about 4.0°C. BN transect data showed that bottom temperatures were 1.0°C lower in 1994 than in other years.

A pronounced low salinity plume was observed in 1993 from Cape Ann into the nearfield region. In 1992, 1994, and 1995, a tongue of saltier offshore water (>32.0 psu) was observed penetrating across Stellwagen Bank from the Gulf of Maine, but this offshore water was not observed in 1993. BN transect data showed that the distribution of bottom water salinity was similar in all years.

Surface fluorescence values ranged from 1-2 μg chl/L in all years except for a denser patch (2-4 μg chl/L) in the northern nearfield in 1994. Subsurface maxima were observed in all years with fluorescence level of 3-5 μg chl/L.

June. Surface temperature was patchy and had little overall pattern in any given year. Surface temperatures ranged from 10-17°C. The range in 1995 was 14-17°C. Nearfield temperatures were colder in 1993, possibly due to local upwelling. In all years, there was a strong west to east gradient across Cape Cod Bay from Plymouth to Cape Cod with colder temperatures to the west. This gradient may be due to upwelling given the prevailing southwest winds.

BN transect data had similar patterns in temperature in all years with strong stratification beginning to develop. Salinity was lowest in 1993, possibly the result of the freshwater plume observed in April and further runoff in May and June. All surface salinity values in 1993 were <30.8 psu, except at the boundary stations. In all other years, all surface salinities were >30.8 psu, except 2 stations near Plymouth in 1992. BN transect data revealed that the 30.8 psu isopleth extended well offshore and was deeper in 1993 relative to other years. Thus, it appears that salinity was substantially lower in 1993 throughout the region than in other years.

Surface fluorescence was low in all years (<1.0 μg chl/L), but the transects revealed subsurface maxima which deepened offshore in all years. The concentrations in 1993 were about twice that of other years (7 vs 3.5 μg chl/L, respectively).

August. Surface temperatures were relatively low in 1994 (<18°C), intermediate in 1992-1993 (18-19°C), and high in 1995 (18-21°C). Cape Cod Bay surface temperatures were slightly lower in 1993 than in other years (18.0 vs >18.0°C, respectively). Surface temperature had no well defined patterns in any year, and there was no obvious surface intrusion of offshore water. BN transect data revealed that the subsurface 9.0°C isotherm was further offshore in 1994 than in other years indicating colder overall subsurface temperatures.

Surface salinity was higher in 1994 and 1995 (>31.5 psu at most stations) than in 1992 and 1993 (<31.5 psu all stations). BN transect data showed saltier subsurface water further inshore in 1994 than in other years, the opposite situation from the temperature section. High inshore fluorescence patches (5-10 μg chl/L) were observed in all years. Time series plots of fluorescence in the nearfield region show highest maximum values in 1993 (Cibik *et al.*, 1997), agreeing with the transect data which had dense maxima near 8 μg chl/L in 1993 and 2-2.5 μg chl/L in 1992 and 1994.

October. No inshore/offshore gradients in surface temperature were observed in any year except 1995, in which a small gradient was found, perhaps due to a weak Gulf of Maine intrusion. There was considerable interannual variability in surface temperature however, with the following ranges being observed: 1992, 12-13°C; 1993, 10-12°C; 1994, 13-15°C; and 1995, 14-16°C.

BN transect data showed that the region was freshest in 1993 with all water <32.0 psu unlike the other years in which the 32 psu water was present along the bottom half of these sections. Fluorescence was markedly higher in 1993 than in other years, with values >10 μg chl/L (range 3-20 μg chl/L). Subsurface maxima in other years were around 5.0 μg chl/L.

3.3.3 Summary of MWRA Baseline Environmental Conditions

For the MWRA monitoring period encompassing 1992 to 1995, the data revealed the following distinctive environmental characteristics for the Massachusetts Bay system:

- seasonal stratification is the dominant physical characteristic of the system, with the vertical structure controlling nutrient availability;

- nitrogen alone or in combination with silicate limits seasonal blooms;

- the nearfield water column was least disturbed in 1994 and 1995 compared 1992 and 1993;

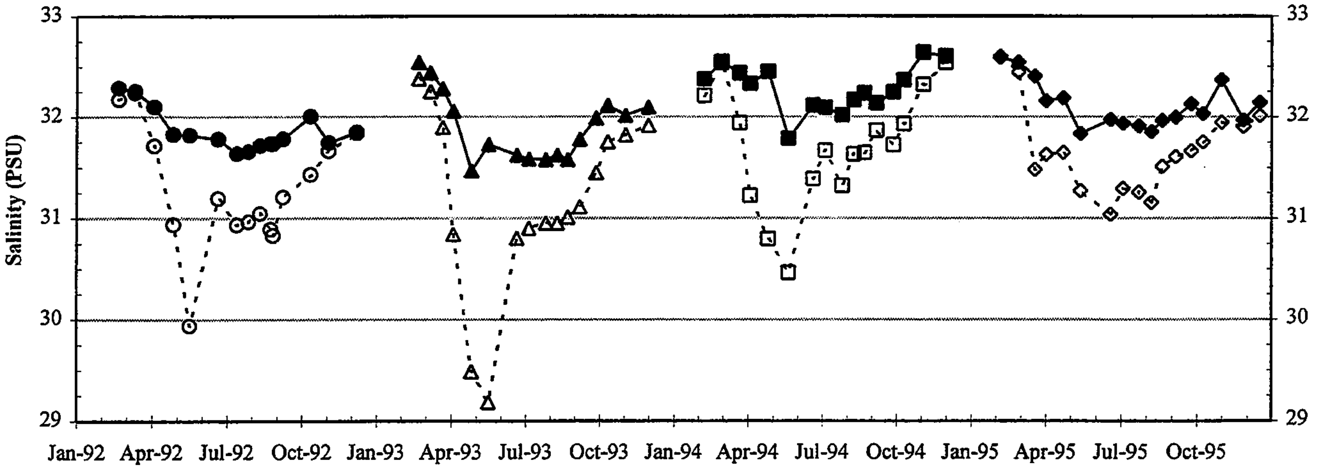
- temperatures in the bay were colder in 1994 than in other years for the months of January to April, and in August;

- temperatures in the bay were higher in 1995 than in other years;

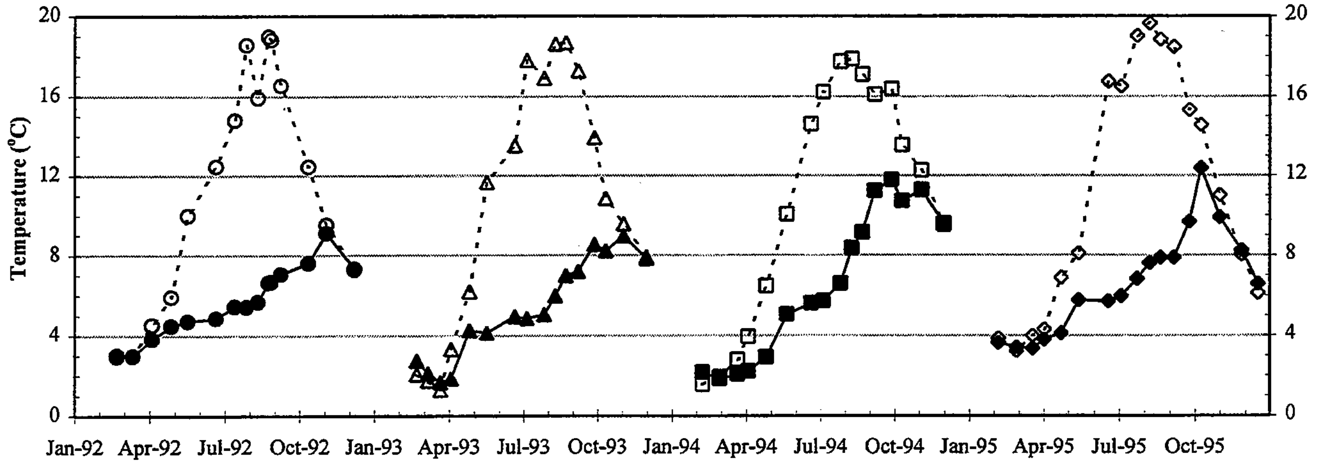
- a pronounced coastal freshwater plume was evident in 1993, possibly due to water entering the bays region from Cape Ann in April, leading to lower salinities in the region relative to other years.

Further in depth analysis of available satellite imagery from 1992 to the present is recommended, as is analysis of surface wind data and riverine discharge.

(a) Salinity



(b) Temperature



(c) Density

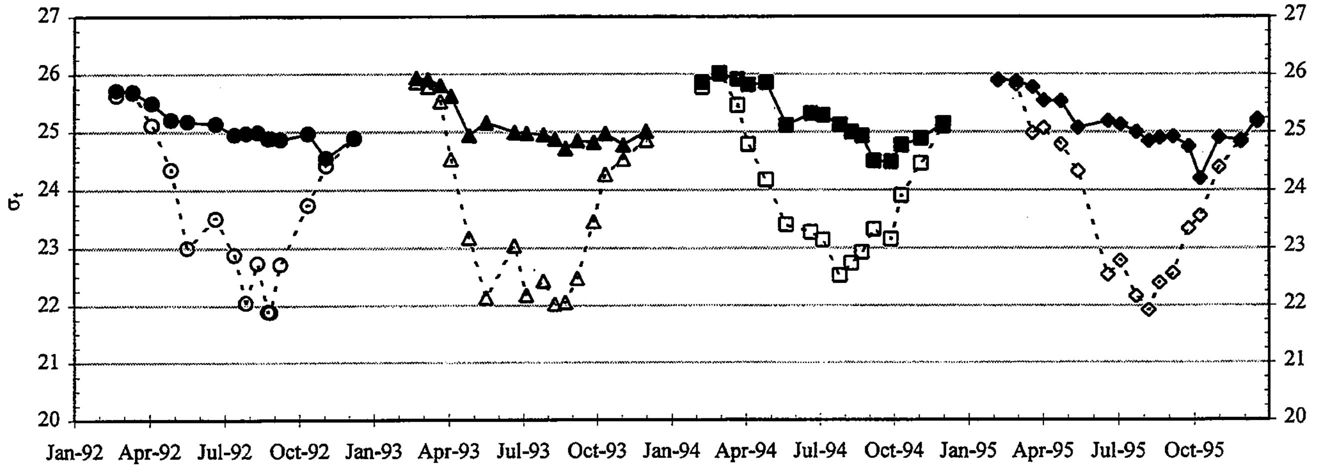


FIGURE 3-1

Average Nearfield Salinity, Temperature, and Density Profiles

Average data for stations N04, N07, N16, N20.

Open symbols = surface. Closed symbols = bottom.

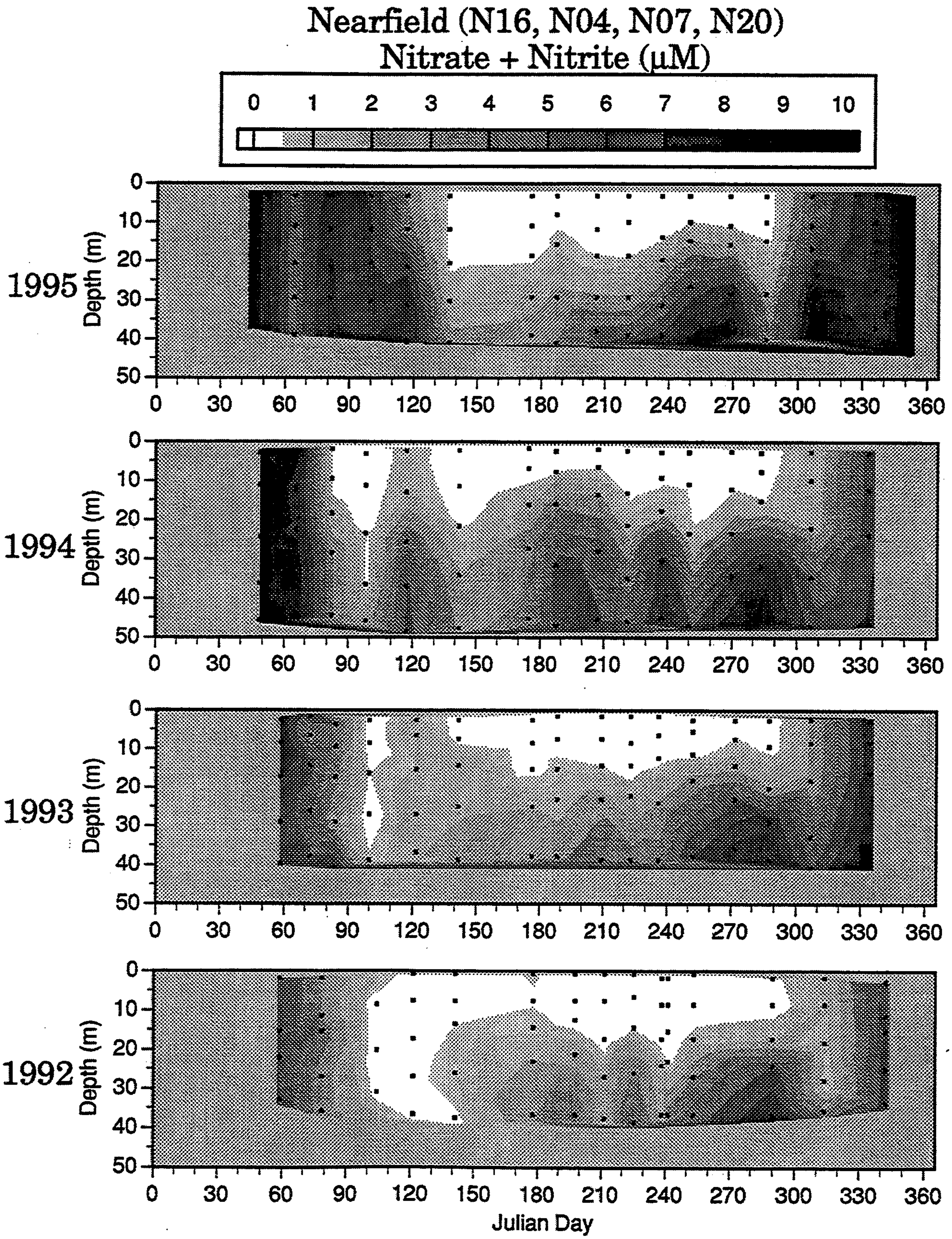


FIGURE 3-2 Nearfield nitrate plus nitrite concentrations for 1992-1995.

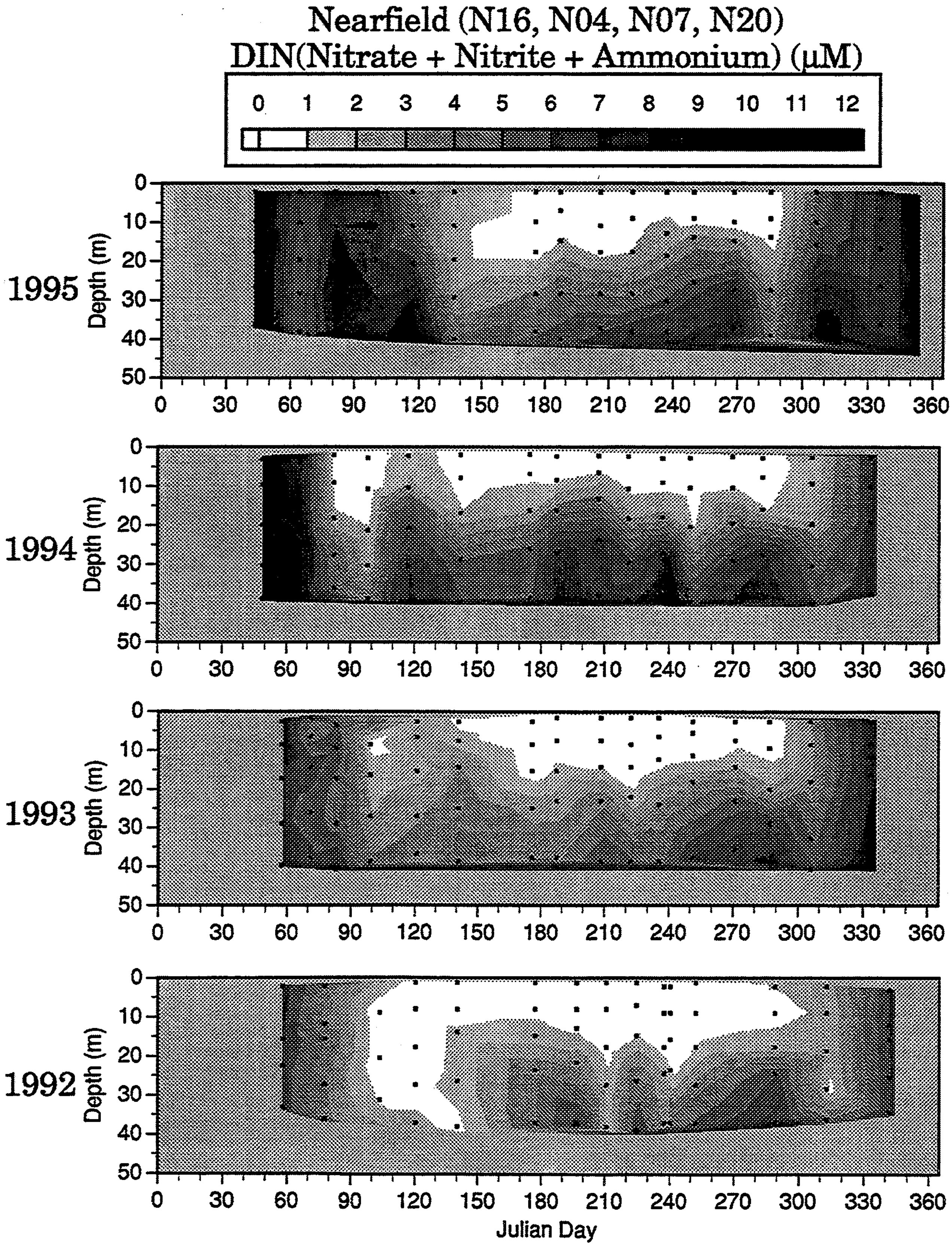


FIGURE 3-3 Nearfield DIN concentrations for 1992-1995.

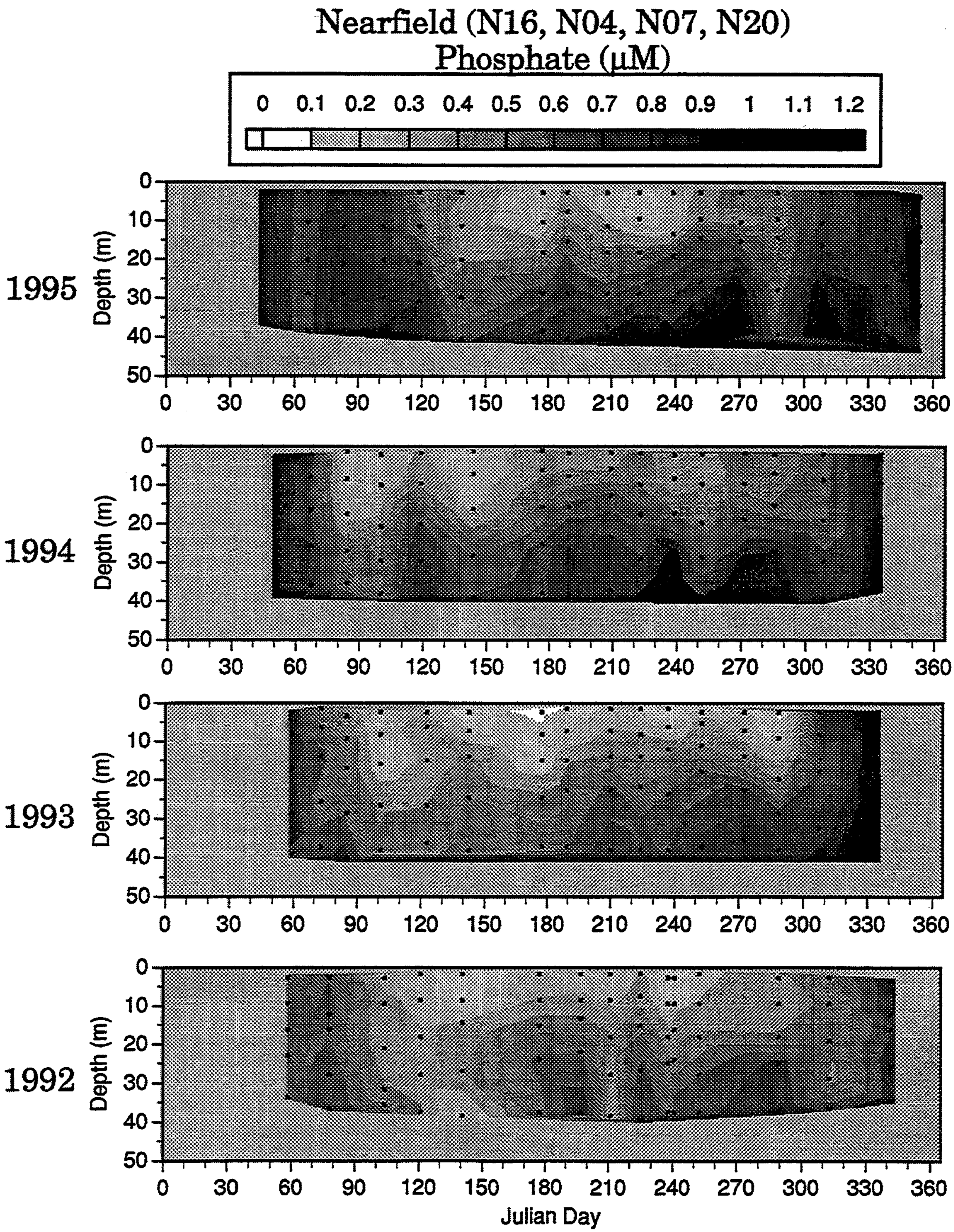


FIGURE 3-4 Nearfield phosphate concentrations for 1992-1995.

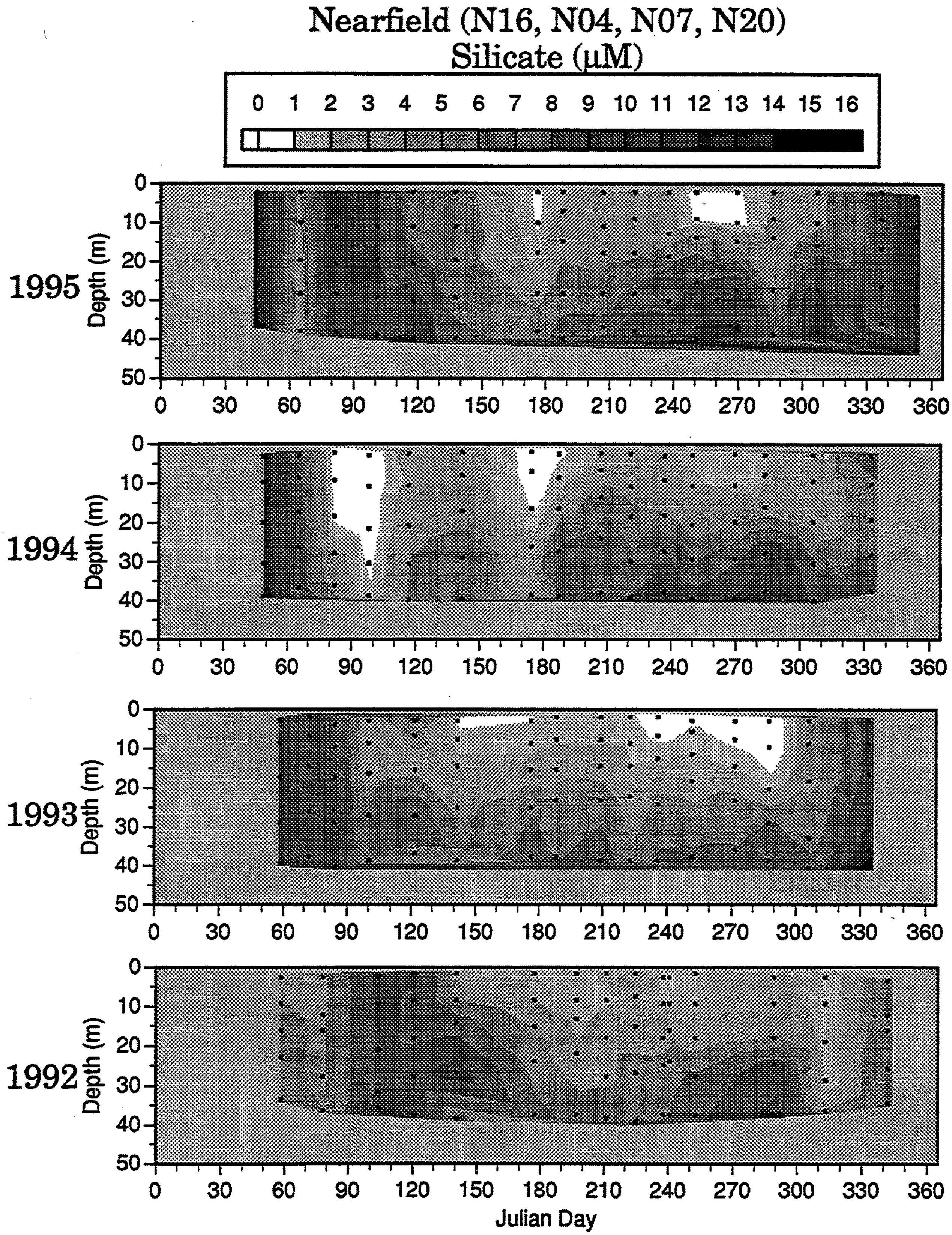


FIGURE 3-5 Nearfield silicate concentrations for 1992-1995.

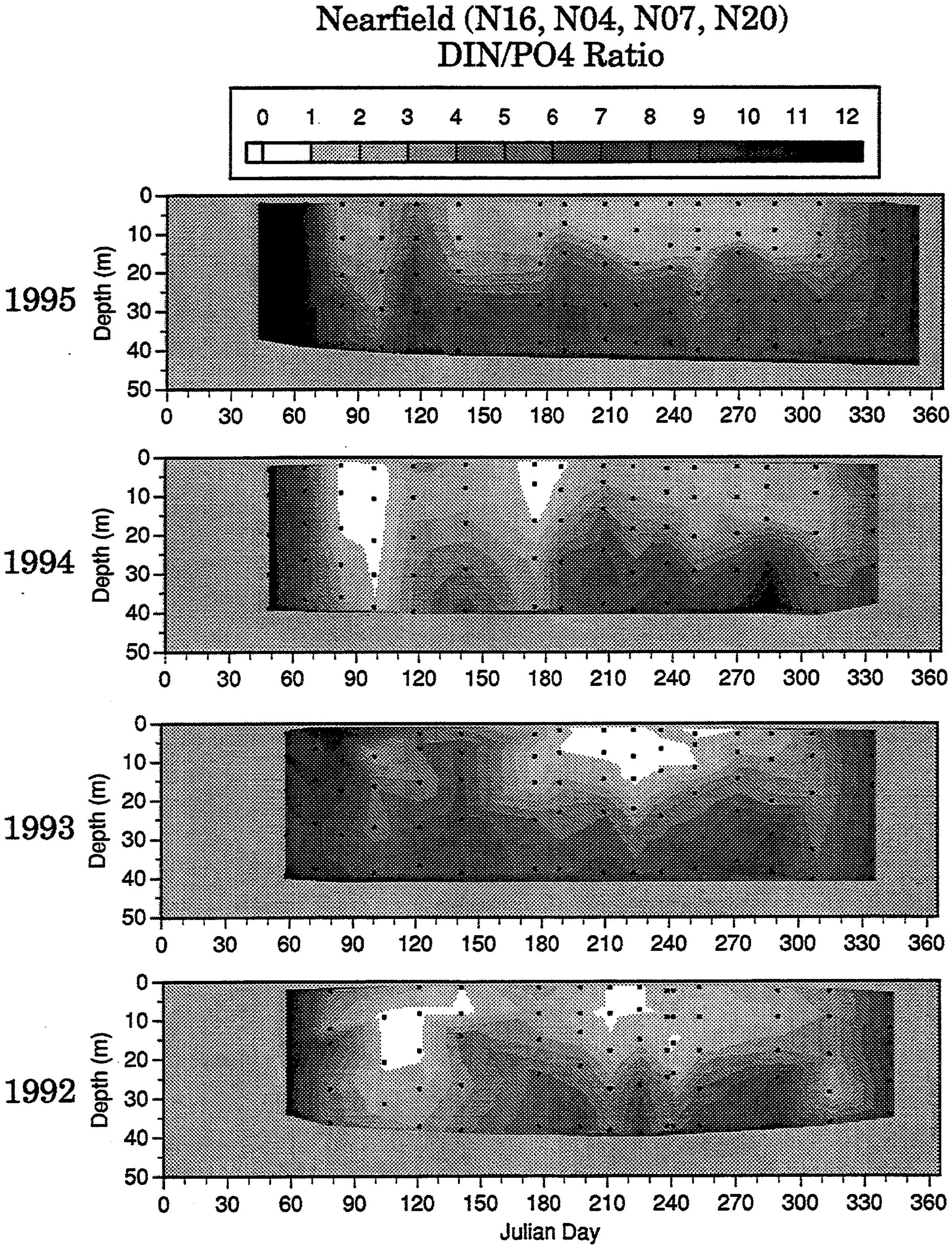


FIGURE 3-6 Nearfield DIN/PO4 ratios for 1992-1995.

4.0 MWRA PLANKTON RECORD

This section presents a review of the MWRA record for phytoplankton and zooplankton data in Massachusetts Bay. The presentation focuses on spatial and temporal patterns in total abundances and dominant taxa. Together with the literature review conducted in Section 2 and the summary of environmental effects discussed in Section 3, this characterization of the four-year baseline record will support development of conceptual models to evaluate potential post-discharge effects (Section 6).

4.1 Summary of Methodologies

Plankton sampling was conducted as part of the MWRA monitoring program at 10 to 12 stations in Massachusetts and Cape Cod Bays from 1992 to present (Figure 4-1). Station locations were changed in 1995 to include more stations in the farfield and fewer stations in the nearfield (Table 4-1). Six plankton sampling stations were common between all years. From 1992-1994, plankton sampling was conducted during the combined nearfield/farfield surveys which occurred six times per year, although N10 was sampled during all nearfield surveys for surface water phytoplankton samples. In 1995, both phytoplankton and zooplankton sampling was conducted at stations N10 and N16 during each of the nearfield surveys, providing greater temporal coverage (11 times per year, Table 4-1).

Phytoplankton samples were collected using two methods. Whole-water grabs were taken at the surface and at mid-depth, with the latter often selected to coincide with the presence of a sub-surface chlorophyll maximum (as determined by in-vivo fluorometry). Screened phytoplankton samples (2 to 4 liters of seawater concentrated using 20μm Nitex) were also collected at the same two depths to enumerate rarer dinoflagellates and potential nuisance taxa.

Zooplankton samples were collected at each station using a 0.5 m diameter net (102-μm mesh opening) equipped with a flowmeter. The net was hauled obliquely from the bottom to the surface (or from 50 m to the surface if the bottom was deeper than 50 m). On shipboard, samples were rinsed into glass jars and preserved in 5-10% formalin seawater.

A complete description of methodologies for sample collection and analysis are given in the Combined Work/Quality Assurance Project Plan (CW/QAPP) for water column monitoring (Bowen *et al.*, 1997). In addition, an inter-laboratory Quality Assurance Program was initiated with the change in subcontractors in 1995. Selected samples from each survey in 1995 were analyzed by both laboratories to determine if quantitative or taxonomic differences could be discerned and corrected.

For phytoplankton, quantitative discrepancies were found to have resulted from the use of increased magnification beginning in 1995. The increase in magnification resulted in higher total cell counts due to increased scrutiny of nannoplankton (<10 μm) taxa. Coupled with the minimum cell count criteria used, it also resulted in lower precision for less numerous larger taxa. As a result, new counting criteria were implemented in 1996 to yield comparable precision in larger taxa while retaining the desirable inclusion of nannoplankton forms. These discrepancies were taken into consideration when making interannual comparisons across years.

For zooplankton, quantitative differences were small. However, two taxonomic issues were identified. *Pseudocalanus newmani* was apparently identified incorrectly as in *Paracalanus parvus* 1992-1994, and *Acartia hudsonica* was identified incorrectly as *Acartia tonsa* (during late summer and fall) in 1992-1994. These discrepancies in results were taken into consideration with respect to interannual evaluations of zooplankton data in this report by combining these taxa at the generic level.

4.2 Phytoplankton

The MWRA monitoring data were reviewed to characterize the annual phytoplankton cycle in the nearfield, and to relate these characteristics to other regions included in the sampling program. The emphasis of the review was to identify important events in the annual cycle, identify the important taxa which contribute to these events, and to ultimately establish the pre-discharge baseline for the phytoplankton community. An important focus was to understand the annual variability in the occurrence and magnitude of these events and the presence of the principal species. As a first step, monitoring data for chlorophyll were reviewed to identify seasonal and regional patterns in phytoplankton biomass.

4.2.1 MWRA Chlorophyll Data

Nearfield data from 1992 to 1995 confirm the annual occurrence of spring and fall blooms described in the literature (Figure 4-2). The chlorophyll data also reveal a considerable amount of annual variability in the cycle. The timing of an event such as the fall bloom (e.g., October 1993, with an average nearfield chlorophyll concentration of almost 10 μg/L) may shift by several weeks from year to year, and may vary several-fold in magnitude. Contrary to previous data, both chlorophyll and productivity measurements from the MWRA program have revealed that the productivity associated with the fall bloom may actually exceed that of the spring bloom event.

These data also demonstrate that the annual cycle typically includes smaller blooms which may occur at various times throughout the summer (e.g., Julian days 150 to 270 in Figure 4-2). These smaller blooms have been linked to short-term alterations in water column stratification which release bottom water nutrients to the nutrient-impoverished surface layer (Cibik *et al.*, 1997).

A large degree of spatial variability can also be seen in the nearfield chlorophyll data (Figure 4-3). For example, summertime blooms are often more pronounced in the inshore areas of the nearfield (e.g. July and

September), and often may be associated with blooms which occur in the harbor. There is also evidence that the fall bloom may be more intense at the more seaward stations (e.g., 1995).

Regional chlorophyll data also document recurrent annual patterns which exhibit substantial temporal and spatial variability in phytoplankton biomass (Figure 4-4). Notable are the generally higher concentrations in the harbor and coastal stations, and the relatively large late winter blooms which have been observed in Cape Cod Bay.

Given these patterns which emerged from the MWRA chlorophyll data, the phytoplankton data were examined to investigate their seasonal patterns, and to identify which taxa were responsible for the seasonal and regional bloom events. Additional discussion of nearfield chlorophyll concentrations relative to contingency planning is included in Section 6.

4.2.2 Total Phytoplankton and Major Groups

Abundance data for phytoplankton were examined by plotting results for the six stations which were sampled within each of the baseline monitoring years (stations N10 and N16 in the nearfield, harbor station F23, coastal station F13, and Cape Cod Bay stations F01 and F02; Figure 4-1).

Although the chlorophyll data indicated phytoplankton biomass peaks occur in spring and fall, total phytoplankton densities in Massachusetts Bay generally increase from February through August (Figure 4-5). This is primarily due to the increasing densities of microflagellates throughout this time frame (Figure 4-6), but is also true for other major phytoplankton groups (see below). As noted in Section 4-1, increased scrutiny of microflagellates during 1995 likely resulted in the observed increases in microflagellate abundance relative to previous years, especially during the summer of 1995.

The sustained peak in microflagellate abundance throughout the summer in did not produce concurrent peaks in chlorophyll or estimated phytoplankton carbon (Figure 4-7), attesting to their small size and relative chlorophyll content. Epifluorescence examinations conducted by Townsend *et al.* (1990c) during June and August of 1990 indicated that up to 45 percent (by number) of the total assemblage were heterotrophic, which would also explain the relative lack of chlorophyll during peak periods of abundance.

In addition to this loosely defined microflagellate group, numerically dominant taxonomic groups reported from the 1992-1995 database included centric diatoms, pennate diatoms, cryptophytes, and dinoflagellates (Figure 4-7). For 1995 results, these five groups also comprised over 90 percent of estimated cell carbon for all surveys, and typically comprised greater than 95 percent The one exception to this was in December of 1995, where three silicoflagellate species (*Dictyocha fibula*, *Distephanus speculum*, and *Ebria tripartita*) combined to comprise about 10 percent (surface) to 20 percent (mid-depth) of the estimated phytoplankton carbon.

Less dominant taxa were found which included cyanophytes, chrysophytes (including the previously mentioned silicoflagellates), xanthophytes, haptophytes, chlorophytes, prasinophytes, and euglenophytes. A complete list of taxa generated by the MWRA program is provided in Appendix A. Results for the major taxonomic groups are presented below.

Centric Diatoms. The results from 1992-1995 indicate that centric diatoms were present throughout the year in all regions, typically at densities of over 104 cells/L (Figure 4-8). Maximum densities at stations other than Cape Cod Bay were often highest during the summer, reaching over 5 million cells/L (station N10). Cape Cod Bay data (stations F01 and F02) indicated that summer densities are typically lower than spring and fall, but that densities early in the year were often higher than other regions.

Pennate Diatoms. Pennate diatoms showed an overall trend of increasing densities during the year (Figure 4-9), although some years did have somewhat diminished densities during the middle of the year (e.g., 1992 in Cape Cod Bay, 1993 in the nearfield). Typically densities ranged from 10^4 to 10^6 cells/L, with a maximum density of nearly 10 million cells/L (Fall of 1993). Pennates can thus dominate the fall assemblage numerically, and can comprise as much as half the estimated phytoplankton carbon (Figure 4-7).

Cryptophytes. Cryptophytes were also found throughout the year in densities between 10^4 to 10^6 cells/L (Figure 4-10). Densities exceeding 10^6 cells/L only occurred at station N10 (1995) and F02 (1992). Inshore stations appeared to have summertime maxima, whereas the more seaward nearfield station (N16) and Cape Cod Bay stations showed relatively constant densities throughout the year.

Dinoflagellates. Screened sample results for dinoflagellates showed fairly uniform densities throughout the year which typically ranged from 10^1 to 10^4 cells/L (Figure 4-11). Cape Cod Bay yielded higher densities in June of 1992, exceeding 10^5 cells/L, and in February of 1995, reaching around 2.5×10^4 cells/L.

The results presented above represent the seasonal contributions of numerous genera and species. These observations will be further examined with respect to dominant taxa in Section 4.2.3.

4.2.3 Dominant Taxa

The 1992-1995 data were evaluated to identify numerically dominant taxa for each of the major taxonomic groups (centric diatoms, pennate diatoms, and dinoflagellates). Based on the carbon equivalence estimates begun in 1995, dominant taxa in terms of estimated biomass were also reviewed. The seasonal distribution of these dominant taxa are depicted in Figures 4-12 and 4-13. Quantitative results are presented in the following sections, which include the presence of each taxon in either the surface or mid-depth samples.

4.2.3.1 Centric Diatoms

Thalassiosira. Species of the genus *Thalassiosira* are present throughout the year in Massachusetts Bay, but are more common during the fall and early spring (Figure 4-14). Densities (exceeding 10^5 cells/L) were regionally similar early in the year, and typically lower in Cape Cod Bay later in the season. The harbor station F23 yielded relatively high densities during 1992, with respect to both regional and interannual comparisons.

Three species of this genus which contribute to the winter/spring assemblage in all sampling regions are *T. nordenskioldii* (Figure 4-15), *T. gravida*, and *T. rotula* (plotted together in Figure 4-16). These species are present primarily from February to April in densities between 10^3 and 10^5 cells/L in all regions, with Cape Cod Bay and the harbor occasionally higher (3×10^5 cells/L) for *T. nordenskioldii*. Abundances for *T. nordenskioldii* were higher in harbor and coastal waters in 1992. *T. gravida* and *T. rotula* appear to persist later in the year in the harbor and coastal waters (Figure 4-16).

Peak densities of *T. nordenskioldii* reported in the literature review by Cura (1991, see Section 2.1) were 1.1×10^5 cells/L in samples taken in March. For *T. gravida*, peak densities in the literature were 2×10^5 cells/L. These results were from early September, an order of magnitude higher than documented during the 1992-1995 baseline during that time of the year. *T. rotula* had maximum densities of around 3×10^5 cells/L.

Chaetoceros. Numerous species of *Chaetoceros* have been reported in the MWRA monitoring samples. They are present in all regions throughout the year in densities ranging from 10^3 to 10^6 cells/L (Figure 4-17). Cape Cod Bay results exhibited a spring maxima, followed by smaller numbers into the fall. Harbor, coastal, and inshore nearfield stations appeared to peak during the summer.

One dominant species which co-occurs with *Thalassiosira* in the spring is *Chaetoceros debilis* (Figure 4-18). It is present in similar densities (reaching 3×10^5 cells/L) in all regions, and often persists into the summer months.

March samples in the studies reviewed by Cura (1991) yielded densities approaching 7×10^5. Other dominants in the MWRA record include *C. socialis*, *C. compressus*, *C. didymus*, and *C. decipiens*. Literature values for *C. compressus* yielded a maximum density of nearly 5×10^5 in an early April sample.

Skeletonema costatum. *Skeletonema costatum* is considered "one of the most common and abundant neritic and estuarine species" (Marshall, 1986). Its distribution in Massachusetts Bay is widespread both temporally and regionally (Figure 4-19). Maximum densities (2×10^6 cells/L) occur in mid- to late summer in harbor and near-coastal waters. It is also a component of the spring and fall maxima in each region. As seen with other diatom taxa present throughout the year, minimum densities appear to occur in Cape Cod

Bay during the summer months. A maximum density for *S. costatum* of $5x10^5$ was reported in the literature during August.

Rhizosolenia. Two species of *Rhizosolenia, R. delicatula* and *R. fragilissima,* were dominant taxa in Massachusetts Bay during the summer and fall (Figures 4-20 and 4-21). *R. delicatula* reached peak abundance between August and October in all regions at densities approaching one million cells/L. Minimum densities found during this period exceeded 10^3 cells/L. It was also reported in each region during late winter at densities of 10^3 to 10^4 cells/L, approaching 10^5 cells/L in Cape Cod Bay (Figure 4-20). *R. fragilissima* exhibited a similar pattern, and was found to reach densities approaching $5x10^6$ cells/L in September at the nearfield station N10 (Figure 4-21).

Maximum densities reported by Cura (1991) for *R. delicatula* exceeded $1.1x10^6$ in July samples. Results for *R. fragilissima* often exceeded 1 million cells/L throughout August and September, with a maximum density of $3.1x10^6$ cells/L.

Leptocylindrus. Massachusetts Bay also has two dominant species of *Leptocylindrus: L. danicus* and *L. minimus* (Figures 4-22 and 4-23). These taxa exhibited a similar distribution in the nearfield and coastal waters as that seen for the two *Rhizosolenia* species in that they occurred in relatively low numbers in the late winter/spring period, and subsequently peaked during the summer and fall. Maximum densities of *L. danicus* (July-August) exceeded one million cells/L in the near-coastal areas (Figure 4-22), whereas similarly high maxima of *L. minimus* were found to occur in Cape Cod Bay in both March and October (Figure 4-23).

Of the two species, *L. danicus* appeared to peak earlier in the season. Densities of *L. danicus* early in the year in Cape Cod Bay also appeared to be an order of magnitude higher than other regions, whereas Cape Cod densities of *L. minimus* during this period exceeded the other regions by two orders of magnitude.

Results for *L. minimus* reported in the literature indicated higher densities and an earlier season for this species, with results in northern Massachusetts Bay (off Salem) exceeding $6x10^6$ cell/L in May/June samples (Cura, 1991). *L. danicus* reached almost $2x10^6$ cells/L in July samples.

4.2.3.2 Pennate Diatoms

Asterionellopsis glacialis. The MWRA phytoplankton data indicate that *A. glacialis* is a dominant contributor to the fall bloom, which typically is observed during October or early November (Figures 4-2 and 4-24). The highest fall chlorophyll concentration seen in 1993 coincided with maximum cell densities of *A. glacialis* (around 5 million cells/L). Densities of greater than 1 million cells/L were seen in all regions in 1993 except Cape Cod Bay, which nonetheless showed increased abundance during the period. Minimum densities throughout the four year period were around 10^3 cells/L.

Pseudo-nitzschia. The genus *Pseudo-nitzschia* is well represented in the MWRA data record, with both *P. pungens* and *P. seriata* routinely reported (Figures 4-25 and 4-26, respectively). Both species are present in all regions in the summer and fall. *P. seriata* is also present in late winter/spring, whereas *P. pungens* was less prevalent early in the year. Fall densities range from 10^3 to 10^5 cells/L, whereas springtime densities typically ranged from 10^3 to 10^4 cells/L.

As discussed in Section 2.1.3, *P. pungens* is indistinguishable under light microscopy from the toxin-producing *P. multiseries*. As well, domoic acid production by *P. seriata* has been demonstrated in one study in Europe (Lundholm *et al.*, 1994). Bates (pers. comm.) has indicated that ambient densities of 10^5 cells/L of the toxic form of *P. multiseries* have produced shellfish tissue concentrations equivalent to the Canadian standard of 20 μg/g. The four-year MWRA record indicated that densities of either species are typically less than 10^5 cells/L.

4.2.3.3 Dinoflagellates

Blooms of dinoflagellate species, which are the causative agent for the phenomenon known as red tides, have been associated with water quality problems in other regions of the U.S. and in other countries but not in Massachusetts Bay. These problems include anoxic events due to the decay of unused plankton biomass (e.g. the 1976 bloom of *Ceratium tripos* in the New York Bight, Swanson and Sinderman, 1979). Toxic species of dinoflagellates can also form blooms. *Alexandrium tamarense* is the only indigenous species in Massachusetts Bay which has resulted in episodes of toxicity. The distribution of these species and other dominant genera are described below.

Ceratium. Several species of the genus *Ceratium* have been identified in the Massachusetts Bay samples. The overall distribution of the genus indicates peak abundances during the summer (Figure 4-27), which were caused by *C. longipes*. Highest densities (approaching 10^5 cells/L) occurred in Cape Cod Bay during June of 1992, with occasional abundances in the nearfield during 1993 approaching 10^4 cells/L. Most results during this peak month were less than 10^3 cells/L, which included densities of a co-dominant species (*C. lineatum*). A secondary peak was often observed in each region during October, which was caused by *C. fusus*. Seasonal densities of this species occasionally exceeded 10^3 cell/L. *C. tripos* was a co-dominant during October, present in each region at densities of around 700 cells/L.

Protoperidinium. *Protoperidinium* spp. were shown to have a year-round presence in Massachusetts Bay (Figure 4-28). Densities rarely exceeded 10^3 cells/L except in Cape Cod Bay. The dominant species was *P. claudicans*, present in Cape Cod Bay in June of 1992 at densities of up to 3×10^3 cells/L.

Gymnodinium. *Gymnodinium* spp. were more prevalent in 1995 results than in previous years (Figure 4-29), which may be attributable to the change in taxonomists beginning in 1995. Results from the interlab

QA study suggested that there may have been discrepancies in taxonomy between *Gymnodinium* and *Gyrodinium* spp. Dodge (1985) considered the latter "clearly an artificial genus", citing studies where cell division produced one daughter cell with characteristics of *Gymnodinium* and the other *Gyrodinium*. Both species were therefore reviewed in the MWRA data base.

The reported distribution of *Gyrodinium* in 1992 to 1994 indicated relatively low densities, typically $<10^2$ cells/L (April to July), much less than the 1995 results which included a late winter bloom (February) of *Gymnodinium* in Cape Cod Bay (densities of 2.4×10^4 cells/L). This spring bloom was also seen in the other regions, with densities closer to 10^3 cells/L. *Gymnodinium* was also present in summer and fall at densities less than 10^3 cell/L.

Two studies reviewed by Cura (1991) reported related taxa only to the family level (Gymnodinaceae), which would include the genera *Amphidinium, Cochlodinium, Gymnodinium, Gyrodinium, Hemidinium, Ptychodiscus*, and *Torodinium* (Dodge, 1985). These studies reported taxa from this family in all surveys (April to September) in densities which ranged from 2×10^3 to 2×10^4 cells/L. Townsend *et al.* (1990a,b,c; 1991) reported *Gymnodinium* spp. in densities exceeding 10^4 cells per liter near station N16 and in the boundary region during February, June, and August of 1990, and densities exceeding 10^3 during March, April, and October. Baseline densities in the nearfield for this genus would therefore appear to range from 10^3 to just over 10^4 cells/L.

Dinophysis. *Dinophysis norvegica* is the principal species of this genus present in Massachusetts Bay (not shown). It is typically present in samples from June surveys in all regions, but seldom exceeds 1,000 cells/L. Maximum densities occurred in June of 1992 when Cape Cod Bay saw densities of 6,239 cells/L.

Alexandrium tamarense. The MWRA sampling program design is not suitable to adequately document the distribution and abundance of the toxic dinoflagellate *Alexandrium tamarense* in Massachusetts Bay because of sample timing and location. Paralytic Shellfish Poisoning (PSP) from this species generally occurs in the bay in May and June, and occurs most frequently along Boston's South Shore, and virtually never along the south shore of Cape Ann or in Boston Harbor. The MWRA monitoring program generally misses this organism, evidenced by the rare occurrences in which it was observed in the samples between 1992 and 1995 (Figure 4-30). Only in 1993 were *A. tamarense* cells seen to any appreciable extent, and then only a few samples. That year, very low concentrations were seen at three far-field stations in March and April, and higher concentrations at a nearfield station in May and June. These observations likely reflect the intrusion of low salinity water and cells from the north (Franks and Anderson 1992) via the western Maine coastal current.

Despite the potential shortcomings of the screened sampling methodology for definitively addressing issues relative to *Alexandrium*, sampling should be continued as it also provides the basis to assess potential changes in other dinoflagellate taxa. Though species such as *Dinophysis, Prorocentrum*, and *Ceratium* are not currently considered a nuisance in the Massachusetts Bay ecosystem, they are part of the natural assemblage and their abundance and distribution could potentially be affected by the relocation.

Information on *Alexandrium* obtained by the MWRA data collections can be supplemented with observations made by D.M. Anderson and co-workers as part of Sea Grant/MWRA sampling programs focused on *A. tamarense* in Massachusetts Bay in 1993-95. Results from those studies are summarized as follows (Anderson, unpub. data).

In 1993, PSP levels were very high and extended beyond the Cape Cod Canal and into Cape Cod Bay, whereas in 1994 and 1995, no PSP was detected anywhere in the bay. Cell count data, the patterns of PSP progression, and a suite of hydrographic measurements all indicate that *Alexandrium* was introduced into Massachusetts Bay in April or early May, several weeks after toxicity was first detected in western Maine. The distribution of *Alexandrium* was patchy during that time, with concentrations of 100-900 cells/liter immediately adjacent to the South Shore, and much lower abundances further offshore. Changes in wind direction and a lack of rainfall in subsequent weeks reduced the north-to-south driving force that normally flushes the cells and freshwater from the bay. Residence time thus increased, allowing the *Alexandrium* cells to persist and multiply, spreading southward along the shore without any significant loss of cells via transport out of the bay. Shellfish toxicity eventually spread beyond the Cape Cod Canal to Sandwich. This persistence of toxicity is unusual based on records of past years, and its extension beyond the Canal was unprecedented. The high cell distribution observed along the South Shore in 1993 (with very few cells offshore) may reflect growth stimulation by nutrients leaving Boston Harbor from the existing outfall, or other local sources of nutrients along the South Shore.

In both 1994 and 1995, *Alexandrium* cells and PSP were detected north of Massachusetts in April and May, just as occurred in 1993, but the transport of those cells into Massachusetts Bay was limited due to persistent southwest winds which forced the coastal current and its associated cells offshore. With little input from the north and apparently little or no in situ germination and growth within the bay, *A. tamarense* cell abundances never exceeded 100 cells per liter in either year and PSP remained undetectable.

The differences between these three years emphasize the critical importance of the timing of winds relative to the patches of *Alexandrium* cells being transported in the coastal current from the north. In those years when the cells are near Cape Ann and there are northeast winds, toxicity develops in the bay. If the cells are not present when those winds blow, or if the cells are present but the winds are from other directions, the water and the cells bypass the bay and travel along Stellwagen Bank towards the Great South Channel and Georges Bank. Stimulation of *Alexandrium* cells by nutrients within the bay seems likely but the evidence remains circumstantial.

4.2.3.4 Other Species of Interest

Phaeocystis pouchetii. Phaeocystis pouchetii is often associated with massive, harmful blooms in other parts of the world. The species occurs throughout the Gulf of Maine and Massachusetts Bay, but blooms are sporadic and at often at relatively low densities. The colonial form of this species, which can include hundreds to thousands of individual cells in a gelatinous matrix, was observed bay-wide in April 1992, reaching densities of almost 10 million cells/L (Figure 4-31). Blooms of this magnitude can foul fishermen's nets, and under the right circumstances when the blooms get close to shore, can result in long lines of foam on beaches.

In 1993, no colonial *Phaeocystis* were seen at any station. In April 1994, another bloom occurred, less extensive than the outbreak in 1992, but reaching 2-3 million cells/L at the two Cape Cod Bay stations. The species was not observed in 1995 samples at any station, but has once again bloomed in 1996, predominantly in Cape Cod Bay.

4.2.4 Summary of Massachusetts Bay Phytoplankton Record

Major observations which emerged from the MWRA baseline data for Massachusetts Bay are summarized below. Comparisons are also made with the historical data reviewed in Section 2.

4.2.4.1 Biomass and Abundance

1) A pattern of successive blooms occurs every year in Massachusetts Bay consisting of a late-winter/early spring bloom and a fall bloom. This pattern is evident in all regions except for the harbor, which typically increases toward a single peak in mid-year.

2) The fall bloom in the nearfield is perhaps more important with respect to productivity potential than previously thought.

3) Peak phytoplankton abundance for most major taxonomic groups in the nearfield occurs during the summer, with production apparently driven by rapid nutrient recycling.

4) Episodic upwelling and mixing events in the nearfield during the summer appear to produce blooms which enhance summertime production. The persistence or regularity of these summer blooms may be important in maintaining a continuous zooplankton community.

5) Cape Cod Bay typically exhibits an earlier winter/spring bloom with higher densities relative to other regions. Cape Cod Bay has somewhat diminished abundances during the summer for many taxa compared with the more northerly regions of Massachusetts Bay.

6) The future outfall site represents a transition zone where nutrients and light penetration are both high, yielding occasional localized chlorophyll maxima. The relative location of these maxima may shift along the existing inshore-offshore nutrient gradient as light availability changes seasonally.

7) Despite relatively high nutrients, measurable chlorophyll, and sufficient light penetration below the pycnocline in the nearfield, productivity in that layer is low.

4.2.4.2 Species Composition and Succession

1) Diatoms are an important component of the Massachusetts Bay phytoplankton community, and are largely responsible for the winter/spring and fall bloom events. A succession of dominant centric diatoms occurs throughout the year, including *Thalassiosira, Chaetoceros, Rhizosolenia, Leptocylindrus,* and *Skeletonema.* The fall bloom is often numerically dominated by the pennate diatom *Asterionellopsis glacialis*, a phenomenon not seen in the literature due to inadequate temporal sample coverage.

2) Boston Harbor assemblages are generally similar to Massachusetts Bay with respect to species composition and seasonal dominants. Overall phytoplankton abundance tends to decrease with distance offshore.

3) Several nuisance phytoplankton species do occur within Massachusetts Bay, but their abundance and distribution vary dramatically from year to year. Two of these species, *Phaeocystis pouchetii* and *Pseudo-nitzschia pungens* are difficult to quantify in the MWRA monitoring program because of the timing of their blooms or their lack of distinctive morphological features.

4.3 Zooplankton

4.3.1 Total Zooplankton and Major Groups

The strong seasonality of the zooplankton can be seen in the plots of total zooplankton abundance at the six stations common to all survey years, consisting of nearfield stations N10 and N16, and farfield stations F01, F02, F13, and F23 (Figure 4-1). Total abundance was highest in all areas in the second half of the year (Figure 4-32). In the nearfield, maximal abundance was observed from July through October, with lower densities from November through May. No obvious interannual differences were observed in total abundance (see statistical analysis below). Regionally and interannually, total zooplankton abundance appears similar, although the harbor station (F23) appears to have lower average abundance during spring than the other stations.

Abundance during 1992 appeared on average to be higher than in other years, especially during the latter half of the year (1992 was also the year of high *Phaeocystis* sp. abundance during the spring bloom). During spring in Cape Cod Bay (F01, F02), zooplankton concentrations were highest in 1994. That year also had the coldest seawater temperatures, and the waters in Cape Cod Bay had the lowest salinity of any year.

The major groups of zooplankton were copepod nauplii and copepodites as well as barnacle nauplii. Abundance patterns for copepod nauplii mirrored those for total zooplankton (Figure 4-33), the former comprising about 50% of the latter. These nauplii were more abundant during the second half of the year and occurred in highest numbers in 1992. Abundance in Cape Cod Bay during spring was highest in 1994. Also during spring, the harbor station (F23) had the lowest concentration of any region. At the two stations closest to the harbor (F23 and N10), abundance of copepod nauplii during spring was highest during 1995.

Abundance patterns of barnacle nauplii are indicative of their coastally-linked meroplanktonic life history (Figure 4-34). This taxon was present in the samples in Cape Cod Bay almost exclusively during the spring period, with highest densities from February to April. Their abundance was higher near the coast than further offshore (N16) or in Cape Cod Bay. No obvious interannual trends can be seen in the data.

4.3.2 Dominant Copepod Taxa - Regional and Seasonal Trends

Copepods were by far the dominant group of zooplankton, and they in turn were dominated by nine species. These species exhibited strong affinities relative to coastal versus offshore environments. During spring the dominant offshore species were *Calanus finmarchicus, Pseudocalanus newmani*, and *Oithona similis* while the dominant coastal species included *Acartia hudsonica*, and *Oithona similis* (Figures 4-35 and 4-36) in addition to *Eurytemora herdmani*. During the warm half of the year, the offshore assemblage was dominated by *Centropages typicus* and *Oithona similis*, while the nearshore assemblage comprised *Acartia tonsa, Centropages hamatus*, and *Oithona similis*.

Although *Oithona similis* is the numerical dominant at most stations at most times of the year, due to its small body size it contributes little to the zooplankton biomass (Figures 4-35 and 4-36). Note that the biomass approximations given here are the product of abundance and adult body weight obtained from the literature. For species where body weight was unknown, weights of similar sized species were used. Body weights used for the 10 species plotted (*Oithona similis* to *Paracalanus parvus*) were 1, 10, 10, 15, 10, 125, 10, 10, 1, and 5 μgC, respectively. The most important species in terms of biomass is the large copepod *Calanus finmarchicus*, which dominates the biomass at nearly every station during the winter/spring period (e.g., Figure 4-36). This species accounts for most of the zooplankton biomass in the water column during this period and is a primary food of many planktivorous organisms including the endangered North Atlantic Right Whale.

4.3.3 Dominant Copepod Taxa - Annual Cycles 1992-1995

Abundance of *Calanus finmarchicus* in Massachusetts Bay was typically higher during 1995 than in other years (Figure 4-37). This trend is most pronounced at the nearshore stations (F23 & N10), but was also apparent at stations further from the harbor (N16 & F13). In Cape Cod Bay, however, *Calanus* abundances were similar between years. Temperatures were higher in 1995 than in other years by 2° to 4°C, and the higher temperature could have caused more rapid growth of the *Calanus* population. Highest concentrations of this species (>2000 ind/m^3) were found at the station furthest offshore (N16) and in Cape Cod Bay (F01 & F02).

Oithona similis had the highest abundance of all species with peak concentrations between 10^4 and 10^5 ind/m^3 (Figure 4-38). Abundance was higher during the latter half of the year with the maxima occurring between July and October. Lowest abundance occurred at the nearshore stations (F23 and N10). Concentrations of *Oithona* were found to be highest during 1992 and lowest during 1995. Perhaps the *Phaeocystis* bloom in 1992 contributed to its high abundance in that year given that *Oithona* may be able to handle and ingest the larger *Phaeocystis* colonies.

Pseudocalanus newmani and *Paracalanus parvus* abundance is given as the total of the two species because of discrepancies in distinguishing between them (Figure 4-39) (see Section 4.1). *Pseudocalanus* is a boreal species and *Paracalanus* is a warm water species, so that the abundances for the first half of the year are expected to be nearly all *Pseudocalanus*, whereas abundances in the second half of the year are a mix of the two species. Total abundance is higher in the second half of the year with the maxima (>10^4 ind/m3) occurring during July and August. During spring, highest abundances were observed in 1992 at all stations except F02, and lowest abundances occurred in 1993. During summer and fall, abundances were highest in both 1992 and 1993, and lowest concentrations were found in 1994.

Centropages typicus is a dominant shelf species during the warmer months, and its abundance was higher during late summer and fall (Figure 4-40). Peak densities were near 10^4 ind/m^3. Springtime concentrations were highest in 1995. During fall, densities were similar between years showing no obvious pattern. Abundances were lower at the harbor station (F23), and the species was not present (in the subsamples counted) at this site on several occasions.

Centropages hamatus is a dominant shallow water species, and was most abundant during the late summer and fall and had highest concentrations (near 10^4 ind/m^3) at the nearshore stations (N10 and F23, Figure 4-41). Abundance was much lower during winter and spring (<100 ind/m^3). Densities were highest in 1995. The species is known to overwinter as bottom resting eggs that hatch when the water warms up during summer. Some proportion of the population may overwinter as adults since individuals are found in the water column year around.

Copepodite life stages of *Centropages typicus* and *C. hamatus* are difficult to distinguish below stage CIII, so the species were grouped for these younger stages. At the life stages of CIV and CV, *Centropages* can typically be identified to a species level. Abundance of *Centropages* copepodites reflected the adult abundance patterns, with highest concentrations occurring during summer and fall (Figure 4-42). No obvious interannual trends were observed and the regional differences were not seen since the species have opposite affinities for near- versus offshore regions.

Temora longicornis was a common species found at all stations (Figure 4-43). Its seasonal cycle was less well defined but it reached highest abundances (10^4 ind/m^3) between June and August with lower concentrations at other times of the year ($<10^3$ ind/m^3). This species had no apparent regional trend and was present at all stations in similar concentrations. Likewise it had no obvious interannual trend.

Eurytemora herdmani is a coastal species that was much more abundant at the two nearshore stations (N10 and F23, Figure 4-44). It occurred rarely and in low numbers at the other stations. Its presence in the samples was sporadic, but it tended to be more common during the latter half of the year. No obvious interannual pattern was observed, due in part to the sparsity of the data.

Acartia hudsonica and *Acartia tonsa* are both coastal species that were generally more abundant at the two nearshore stations (Figure 4-45). Due to discrepancies in taxonomic identification (see Section 4.1), these species are grouped as total *Acartia* for the purpose of the interannual comparison. *A. tonsa* is a warmer water species and is unlikely to be very abundant during the cold season, thus, abundances during the winter/spring period are likely due to *A. hudsonica*. During this cold season, abundance was highest during 1992, and the February 1992 concentrations of *Acartia* were higher in Cape Cod Bay than at the nearshore stations. It appears from these data that *A. hudsonica* may be less restricted to nearshore environments during the winter/spring period.

The discrepancy mentioned above arose because 1995 results indicated co-occurrence of *A. hudsonica* and *A. tonsa* during the summer and fall (after initial dominance by *A. hudsonica* early in the year), whereas the 1992-1994 data set indicated no overlap (sharp demarcation occurring in June). After confirmation that defining morphological features for both species were present in August 1995 samples, the literature was examined for studies on *Acartia* beyond Massachusetts Bay. Studies done south of Cape Cod Bay (Narragansett Bay and other southern New England estuaries, Long Island Sound and the Middle Atlantic states) support the seasonal succession of *A. hudsonica* and *A. tonsa* with little overlap between the two taxa (Deevey, 1947; Conover, 1956; Bowman, 1961; Jefferies, 1962; Durbin & Durbin, 1981; Sullivan and McManus, 1986).

Available studies of zooplankton distribution north of Cape Cod suggest that seasonal co-occurrence in northern waters does occur (Lee and McAlice, 1979; Toner *et al.*, 1984). Lee & McAlice documented that the well-defined successional pattern evidenced in studies south of Cape Cod did not occur in the

Damariscotta River Estuary, Maine. Instead, both species increased in their abundance at the same time on most occasions. Toner *et al.* found *A. hudsonica* to be abundant year around in western Cape Cod Bay except in November. *Acartia tonsa* was not found during the spring.

The overlap in species occurrence in these studies seems to indicate that colder summer temperatures north of Cape Cod may allow *A. hudsonica* to persist later in the year than is typical in the warmer waters south of the Cape.

4.3.4 Comparison with MARMAP data - Calanus

The Marine Monitoring Assessment and Prediction (MARMAP) program conducted plankton sampling along the continental shelf of the northeastern U. S. from 1977-1987. Two samples were collected in Massachusetts and Cape Cod Bays at bimonthly or interspersed intervals during this period, dependent upon the year sampled. The locations of all MARMAP samples collected in the Bays region are shown in Figure 4-46. The data for this region remain largely unanalyzed, and such analysis is beyond the scope of the present report. A comparison is presented, however, for one of the species, *Calanus finmarchicus*, showing how its interannual abundance cycle compares with that found by the MWRA monitoring program.

The annual cycle of *Calanus* was found to be repeatable from year to year during the MARMAP sampling period (Figure 4-47). A running average overlayed on the data depicts clearly this repeated annual cycle. When the data from 1992-1995 are overlaid on the MARMAP data, it can be seen that the general annual pattern is very similar during these two periods (Figure 4-48; MARMAP data are denoted by asterisks; symbols for MWRA data are the same as in prior plots). Note also that the 1995 data (closed circles) appear higher than other years during February and March. According to the temperature data, discussed above, this year was much warmer (2 to 4°C) than other MWRA monitoring years. The MARMAP data require further analysis and interpretation with regard to other species and hydrographic variables in order to compare it with the MWRA data. Moreover, further analysis of other data sets (e.g., Continuous Plankton Recorder data from 1961-present) is needed to properly interpret the MWRA monitoring results in a long-term interannual context.

4.3.5 Three-Factor Analysis of Variance

Description of Statistical Method. An investigation of the MWRA data was made to determine how the abundance (# ind/m^3) of 11 dominant taxa of zooplankton differs under three controlling factors. Each observation was made on a given year (1992-1995) on a given Julian day (1-365) at a given station (F01, F02, N10, F13, N16, F23) (Figure 4-1). Thus, the three factors, A, B, C are Year, Season, and Station, respectively. Factor A has four levels (1992 through 1995). Factor B was made into two levels by grouping all observations recorded from January 1 to July 1 into one level and the remainder into the second level. Factor C has six levels as stated above.

In total, over the four years, 173 independent samples were taken and the abundance of each of eleven taxa was analyzed. Table 4-2 shows the number of samples taken for combinations of factors, e.g. in 1995 (A4), in season 2 (B2) at station 16 (C5), 12 samples were taken.

Since an ANOVA assumes data are from a normally distributed population, the data were log transformed by taking the natural log of each value for use in the statistical evaluation. The 11 taxa analyzed were *Acartia* (*tonsa* and *hudsonica* combined), barnacle nauplii, *Calanus finmarchicus, Centropages* copepodites, *Centropages hamatus*, copepod nauplii, *Centropages typicus, Eurytemora herdmani, Oithona similis, Pseudocalanus* and *Paracalanus* (combined), and *Temora longicornis.* Analysis was also done on total zooplankton abundance. All taxa were analyzed, and many exhibited similar outcomes. Thus, a few representative taxa will be discussed here.

Case 1 - Variation in abundance due purely to main effects. In this case, the differences in abundance are due solely to the three factors acting independently of each other. Abundance patterns in *Calanus, Centropages* copepodites, and barnacle nauplii can be explained this way. The variation in abundance of *Calanus* and *Centropages* copepodites can be explained solely by factor B (season), i.e., the abundance depends only on a seasonal effect (created by grouping the data). The mean abundance of barnacle nauplii is due to all factors acting independently of each other (note that mean abundance is actually mean log abundance). It was found that barnacle nauplii abundance varies significantly from year to year, but *Centropages* copepodite abundance does not (Figure 4-49a). Note that an effect is significant if the mean abundance is not constant over the years.

It can be seen that the abundance of *Centropages* appears constant over the years whereas barnacle nauplii does not. In the analysis, this effect is justified by the p-values of 0.26602 and 0.00024, respectively. When the same two taxa were examined with respect to station, it was again found that *Centropages* spp. abundance was independent of station, but that barnacle nauplii had significant regional variation (Figure 4-49b; p=0.23683 and p=0.00010, respectively). Significant seasonal effects were found for both of these taxa (Figure 4-49c), with p-values recorded as 0.00000. As stated above, the results for *Calanus* were the same as for *Centropages* copepodites, i.e., only seasonal effects were found to be significant (but *Calanus* is most abundant in Season 1 and *Centropages* in Season 2).

Case 2 - Variation in abundance due to effects of first order interactions. In this case, the differences in abundance are due to a combination of two of the three factors being dependent on each other. The remaining eight taxa examined all had interaction effects between factors A (Year) and B (Season), and two of the coastal species, *Centropages hamatus* and *Eurytemora herdmani*, also had interaction effects between season and station. Interaction effects between A and B means that the mean abundance in a specific year is dependent on season, and vice versa.

For copepod nauplii, it is clear that the mean abundance differs over the years depending on which season is sampled (Figure 4-49d). Abundance was lower during the first half of the year relative to the second half for all years except 1994, when the reverse was true (p=0.00000). By comparison, mean abundance of copepod nauplii within a given season (all years combined) is not significantly different between stations (Figure 4-49e).

Other species had a similar pattern (Figure 4-50) except as noted above for *C. hamatus* and *E. herdmani*. The mean abundance of these species is significantly higher at the two nearshore stations and during the second half of the year (Figures 4-50f&g; station 3 = N10 and station 6 = F23). Significant station effects also were found for *Acartia*. Note that Case 2b was examined, in which all three factors are dependent upon each other, but no significant effects were found.

In summary, all species had significant interannual and seasonal variations in abundance except for *Calanus* and *Centropages* copepodites, whose abundances could be explained solely by seasonal effects. *Eurytemora* and *Centropages hamatus* also had significant interaction effects between station and season. *Calanus* interannual variation was apparently masked by its strong seasonal signal, so that the higher abundance in 1995 did not appear significant using this three-factor analysis. Further investigation of individual years regions and seasons is required to explain such apparent trends in the data.

The temporal and spatial resolution of the data collected by the monitoring program appears to be adequate for detecting significant interannual, seasonal, and regional patterns in the baseline case. This finding implies that significant shifts from the observed baseline patterns due to the outfall also will be detectable. Nonetheless, it will be difficult to determine the mechanisms causing the detected changes without higher resolution sampling in time and space.

4.3.6 Summary of Massachusetts Bay Zooplankton Record

Strong regional, seasonal, and interannual variation in zooplankton abundance was found. Pronounced shifts in species composition were found with region and season. Copepods were the dominant group of zooplankton in each region and season. Boreal species were most abundant during the first half of the year, while warmer water species dominated during the latter half.

Regionally, the coastal species assemblage was dominated by *Acartia* spp., while offshore species composition was characterized by *Calanus* and *Pseudocalanus* in winter/spring and *Centropages typicus* and *Paracalanus parvus* in late summer an fall. The ubiquitous cyclopoid copepod *Oithona similis* was dominant away from the coast in both seasons and had highest abundance in the second half of the year. Barnacle nauplii were a dominant group during spring in coastal waters.

TABLE 4-1

Plankton Stations Sampled from 1992-1995
by MWRA Monitoring Program

	1992		1993		1994		1995	
Month	Date	Sta*	Date	Sta*	Date	Sta*	Date	Sta*
Jan								
Feb	22-28	A	23-27	A	15-18	A	6-14	B
Mar	14-18	A	9-12	A	1-2 & 5-7	A	2/28-3/5 20-22	B C
Apr	7-14	A	6-10	A	5-10	A	3-10 24-27	B C
May							15-17	C
Jun	22-26	A	22-26	A	21-25	A	20-25	B
Jul							5-7 24-26	C C
Aug	25-29	A	24-28	A	25-29	A	8-10 21-26	C B
Sep							6-14 25-29	C C
Oct	13-17	A	12-16	A	11-15	A	9-13	B
Nov							11/27 & 12/5	C
Dec							17-19	C

*Station Numbers:
A = N1, N4, N7, N10, N16, N20, F1, F2, F13, F23
B = N10, N16, F01, F02, F06, F13, F23, F24, F25, F27, F30, F31
C = N10, N16

NOTE:

N10 surface phytoplankton samples on nearfield surveys were performed 1992-1994. Zooplankton samples were not performed during the nearfield 1992-1994.

Farfield stations were sampled during W9516 and W9517.

TABLE 4-2

Number of Samples used in 3-Factor ANOVA

	A_1		A_2		A_3		A_4	
	B_1	B_2	B_1	B_2	B_1	B_2	B_1	B_2
C_1	4	2	4	2	4	2	4	2
C_2	6	4	4	2	4	2	4	2
C_3	4	2	4	2	4	2	7	9
C_4	4	2	4	2	4	2	3	1
C_5	4	2	4	2	4	2	11	12
C_6	4	2	4	2	4		4	2

Notes: A = Year, B = Season, and C = Station

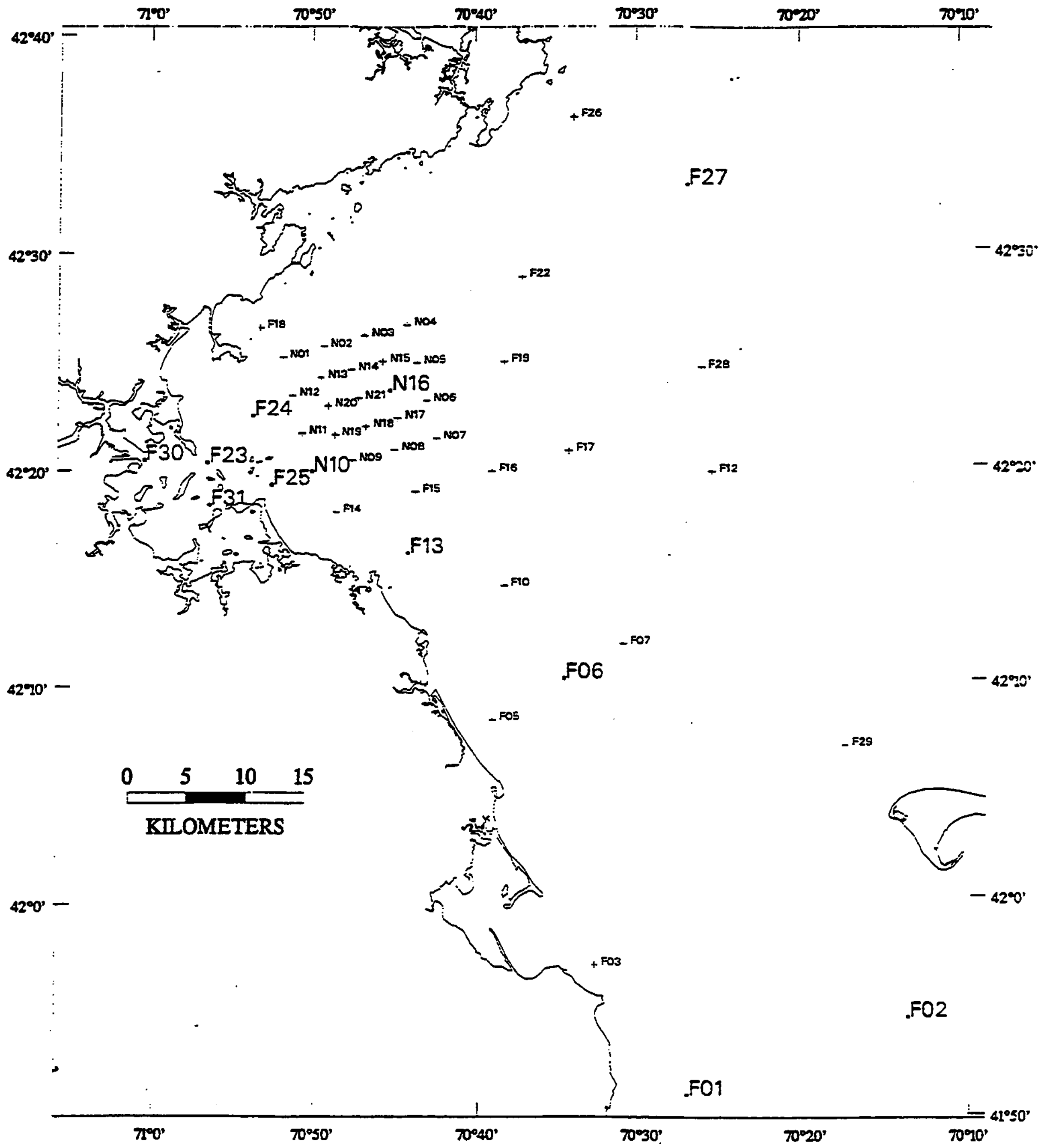


FIGURE 4-1
Plankton Station Locations [Enlarged Text]

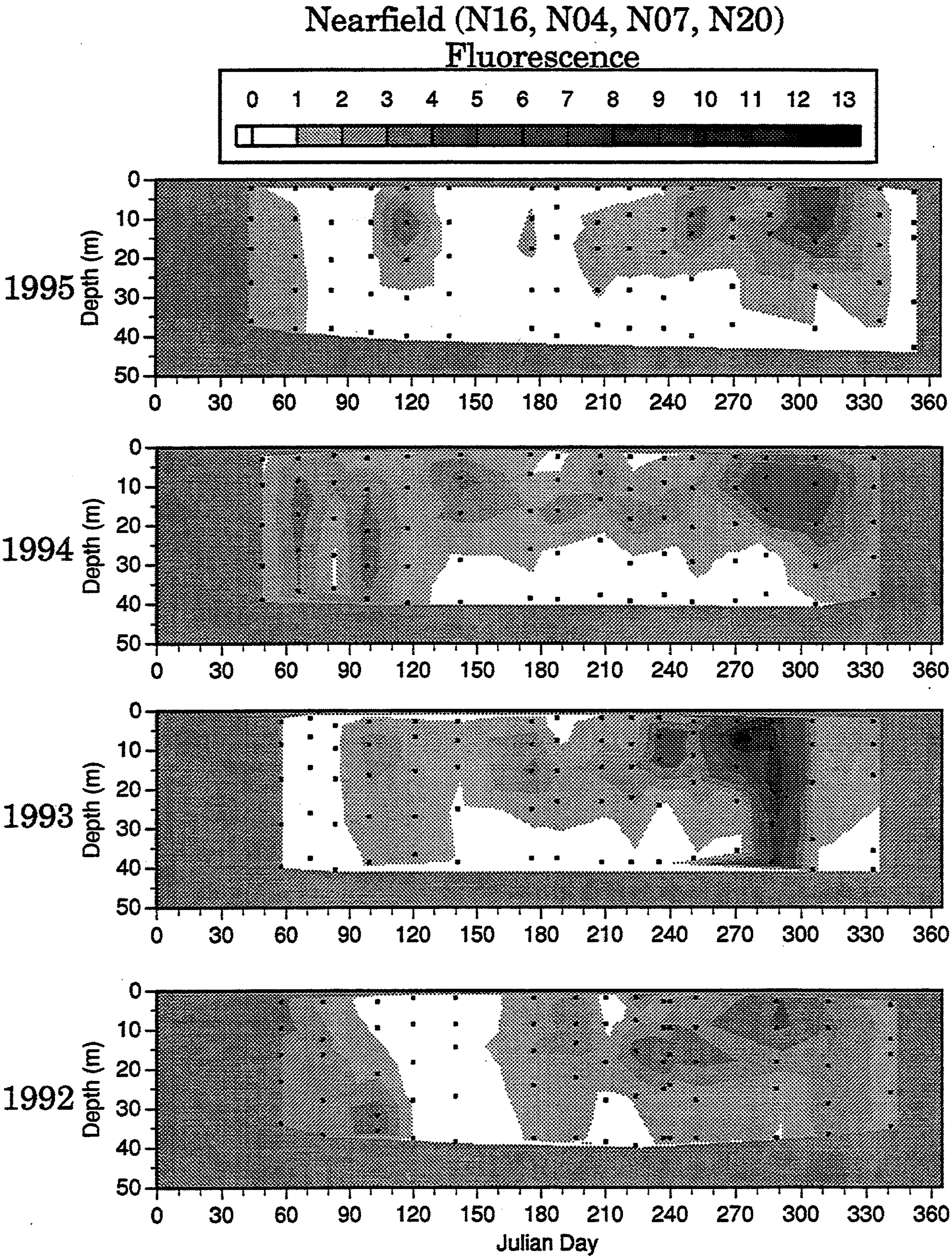
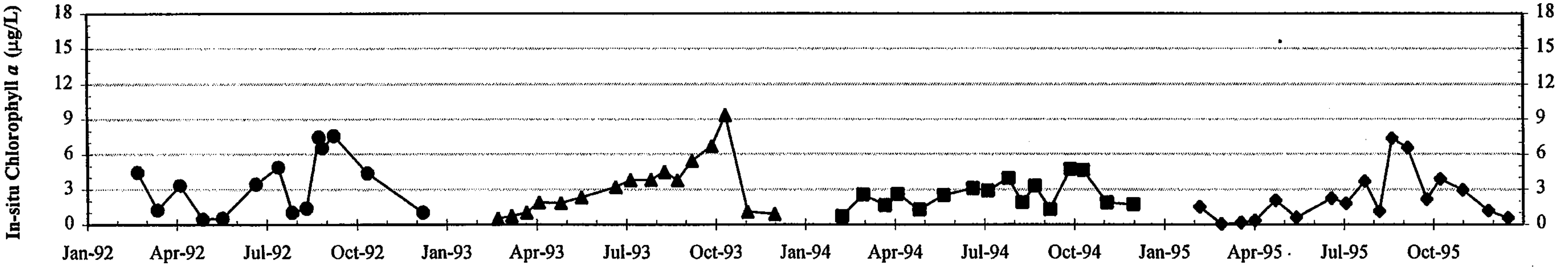
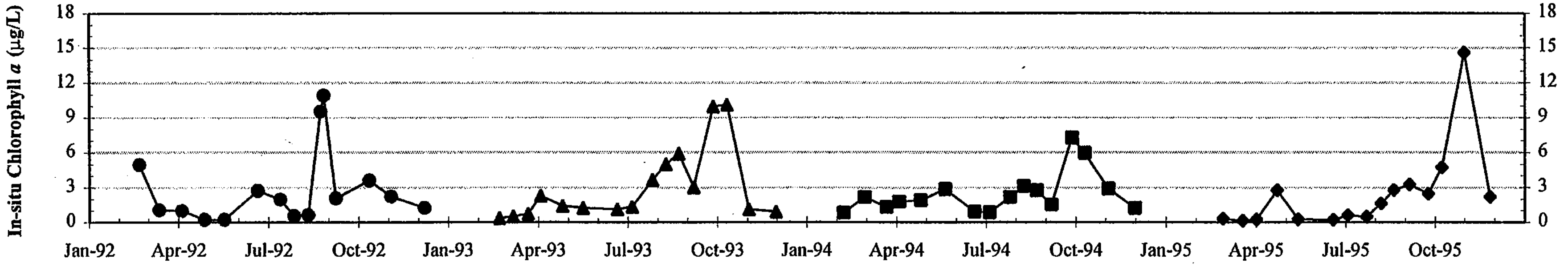


FIGURE 4-2 Depth-averaged fluorescence concentrations for 1992-1995.

(a) Inner Nearfield: N10, N11



(b) Broad Sound: N01



(c) Outer Nearfield: N04, N07, N16, N20

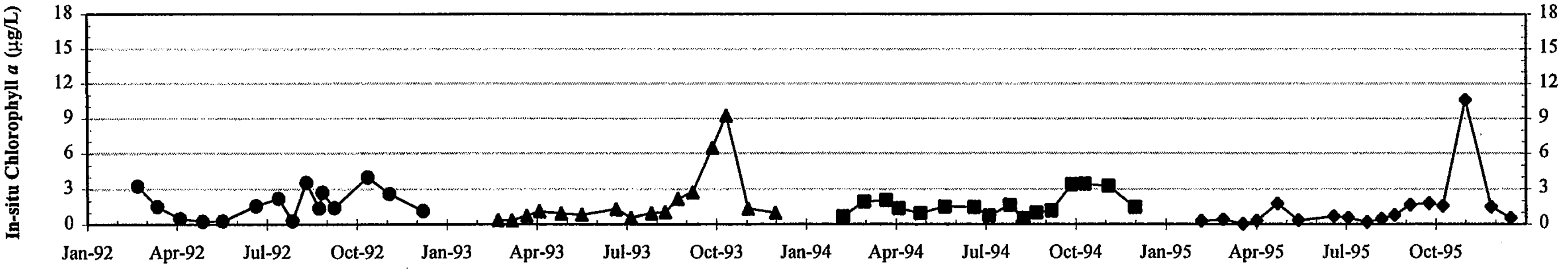
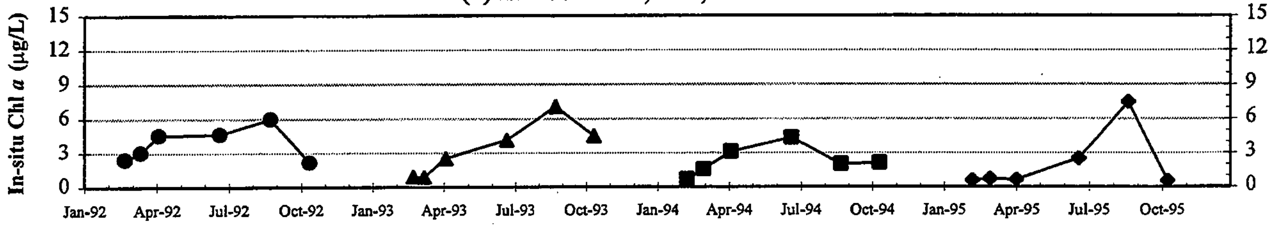


FIGURE 4-3

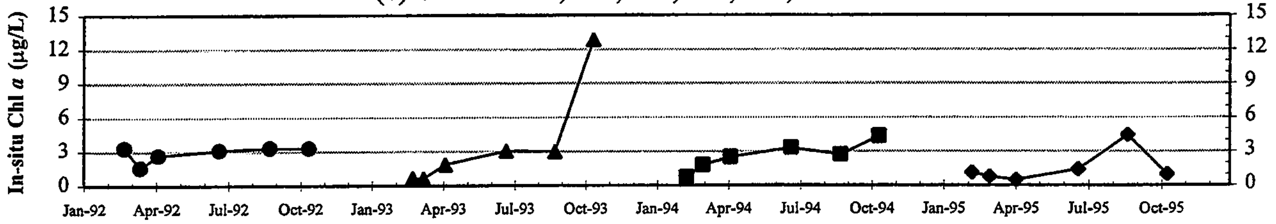
Interannual In-situ Chlorophyll a Patterns for Selected Areas in Nearfield

Symbols indicate surface (A depth) average.

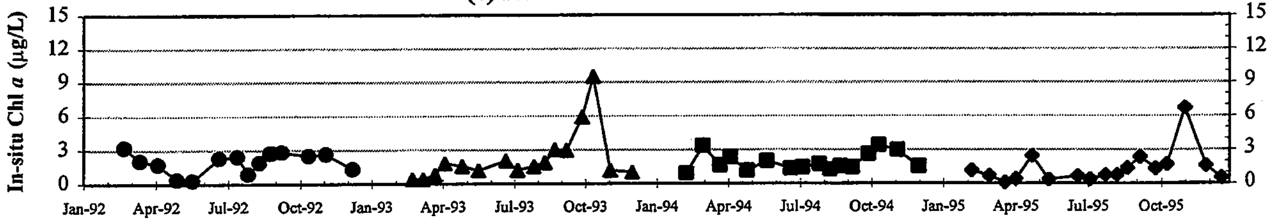
(a) Harbor: F23, F30, F31



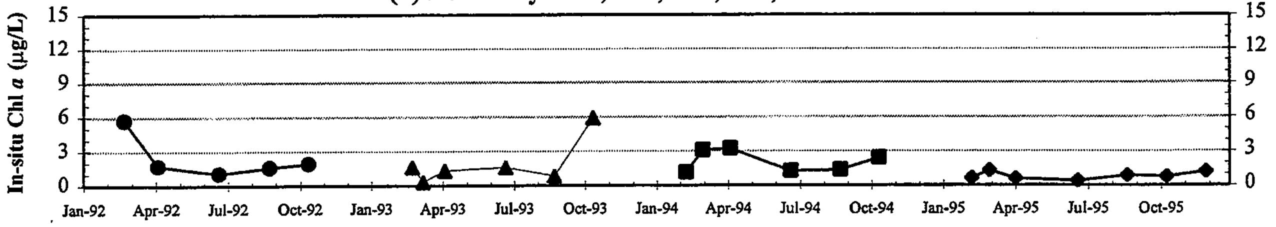
(b) Coastal: F05, F13, F14, F18, F24, F25



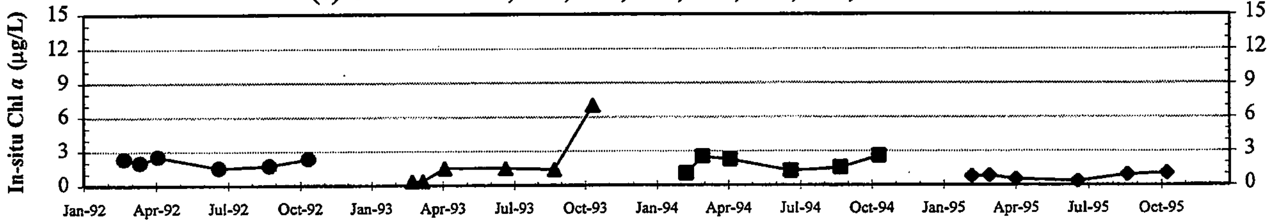
(c) Nearfield: 17 Stations



(d) Boundary: F12, F26, F27, F28, F29



(e) Offshore: F06, F07, F10, F15, F16, F17, F19, F22



(f) Cape Cod Bay: F01, F02, F03

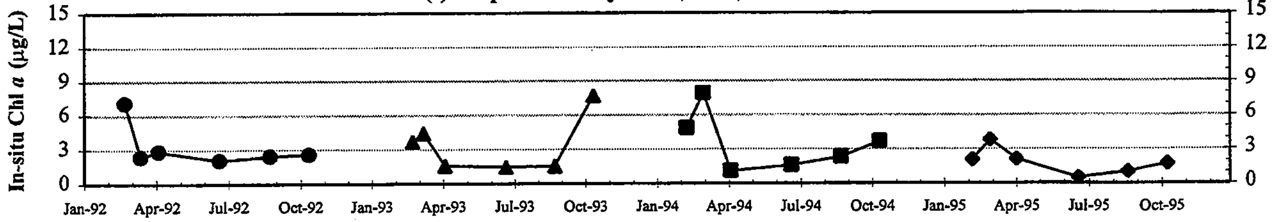


FIGURE 4-4

Regional In-situ Chlorophyll *a* Patterns 1992 - 1995

Average concentrations for listed stations, all depths.

FIGURE 4-5

Total Phytoplankton

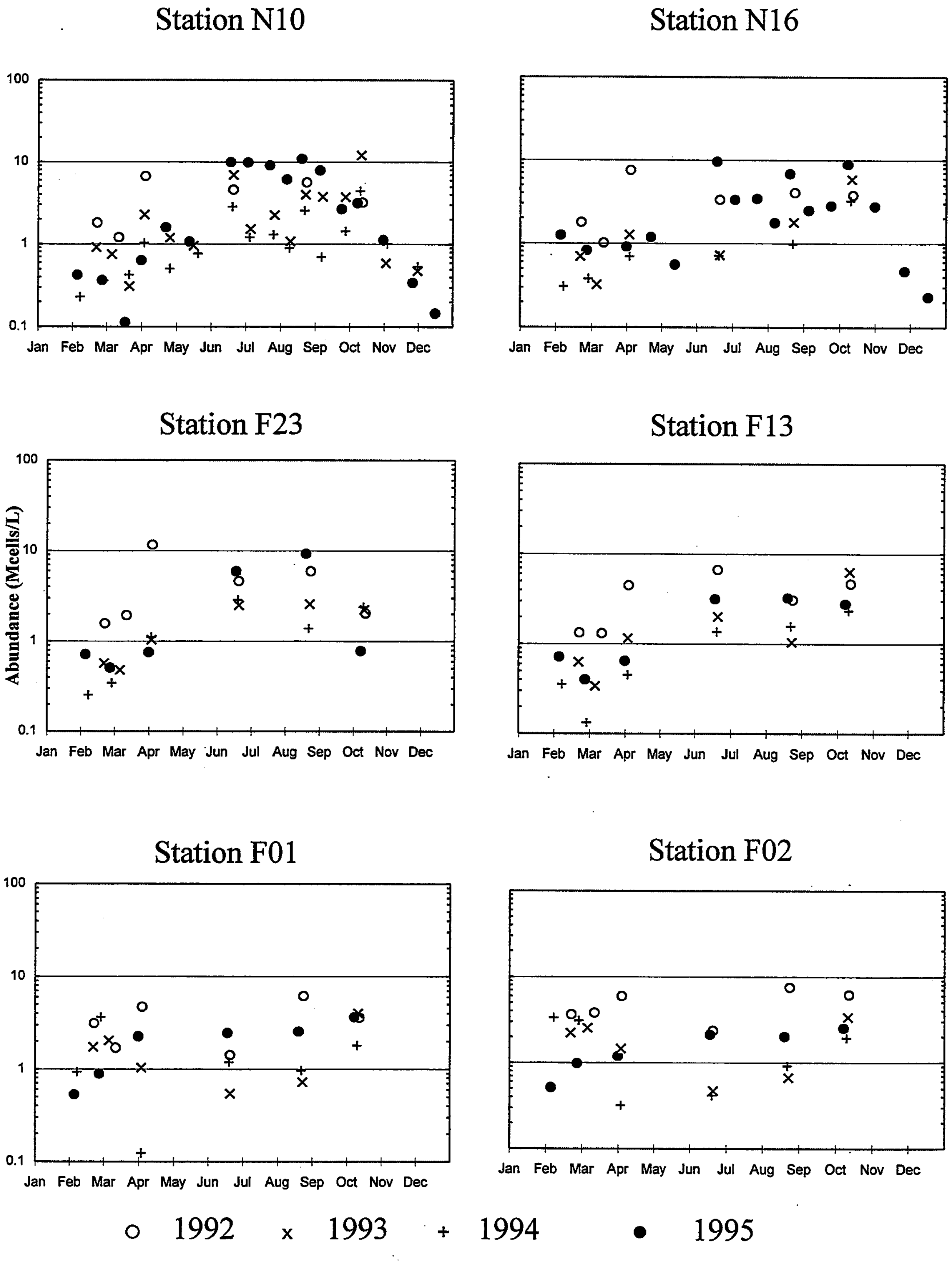
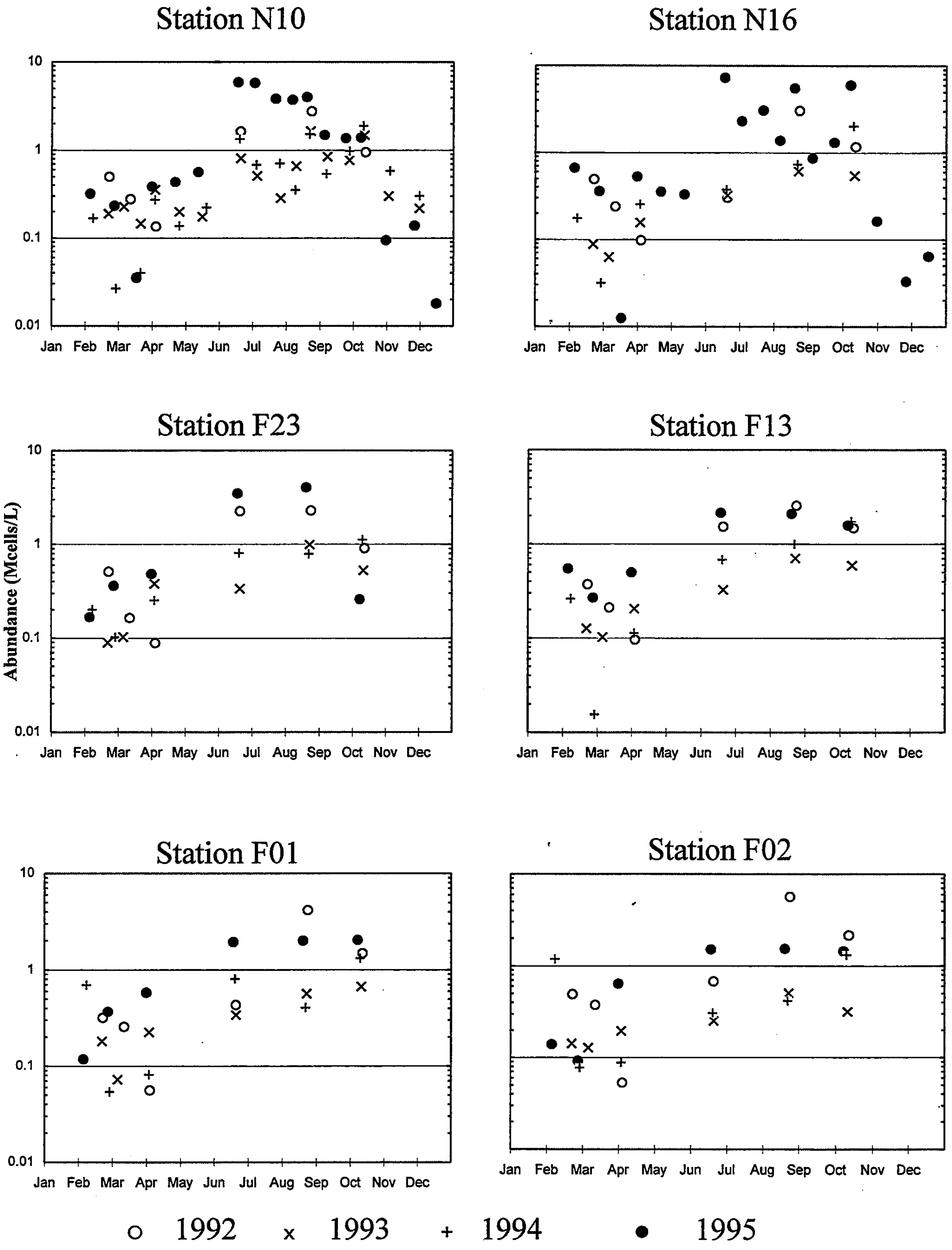


FIGURE 4-6

Microflagellates



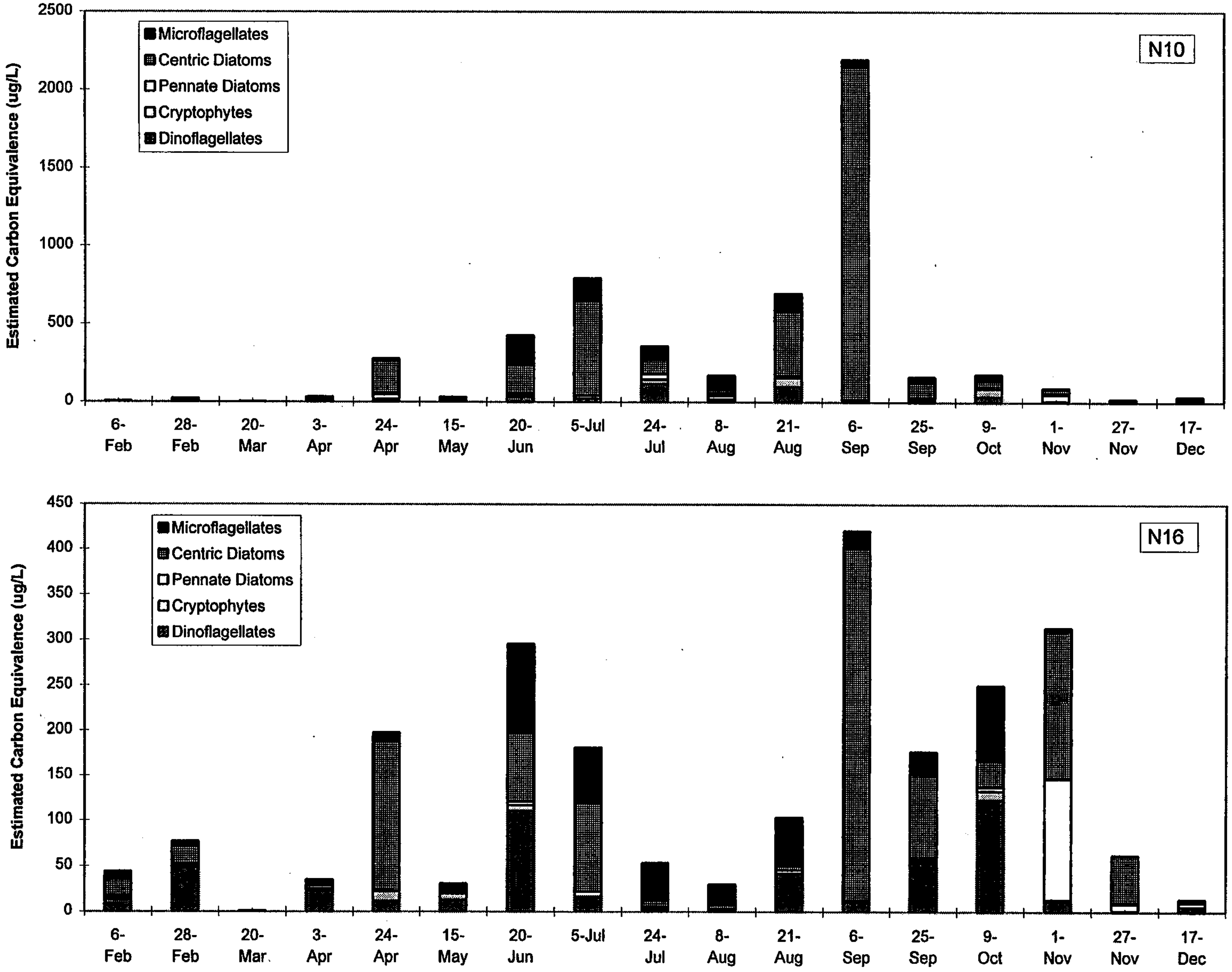


FIGURE 4-7

Distribution of Carbon by Major Taxonomic Groups in 1995 Surface Samples
Nearfield Stations N10 (Top) and N16
(Note difference in y-axis scale)

FIGURE 4-8

Centric Diatoms

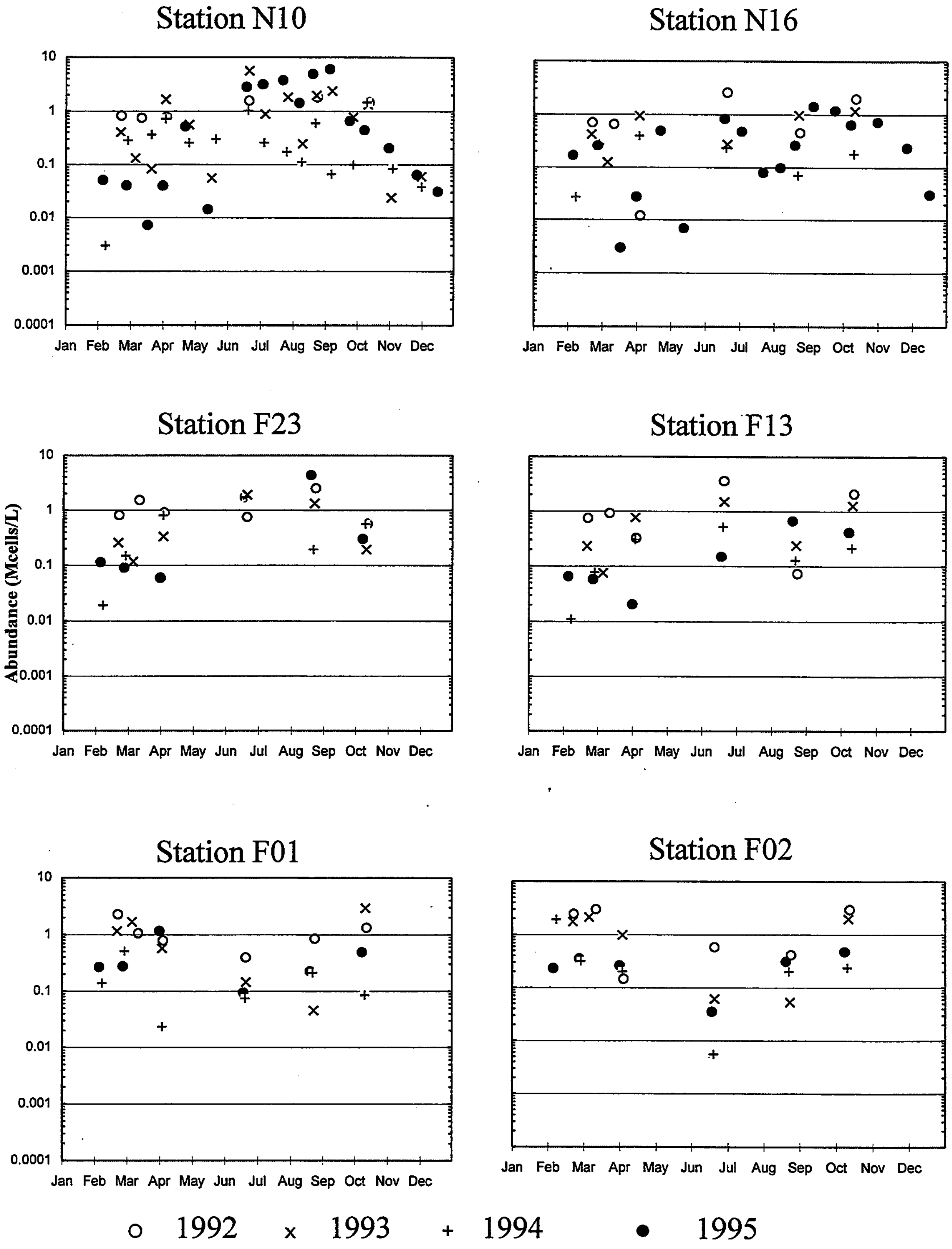


FIGURE 4-9

Pennate Diatoms

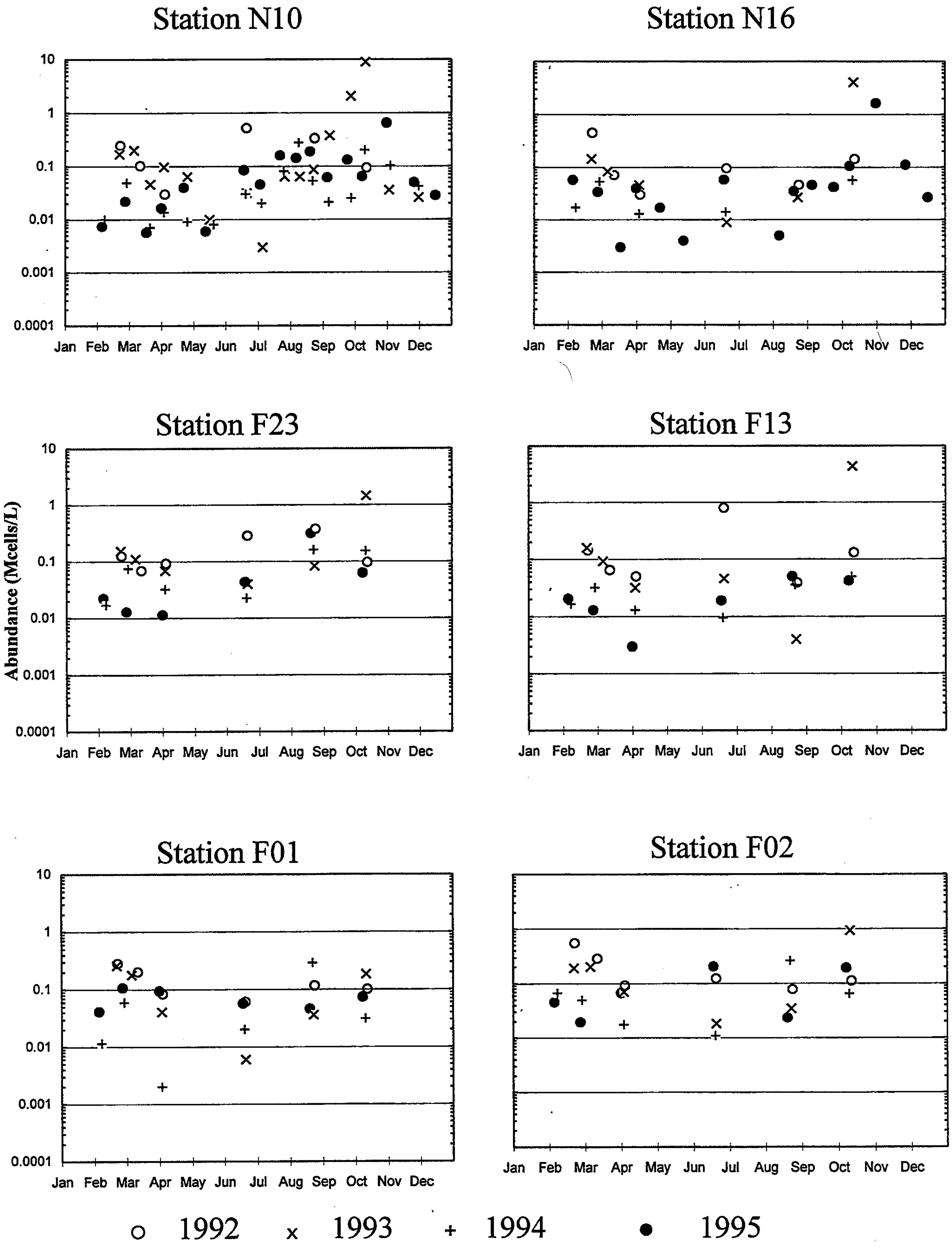


FIGURE 4-10

Cryptophytes

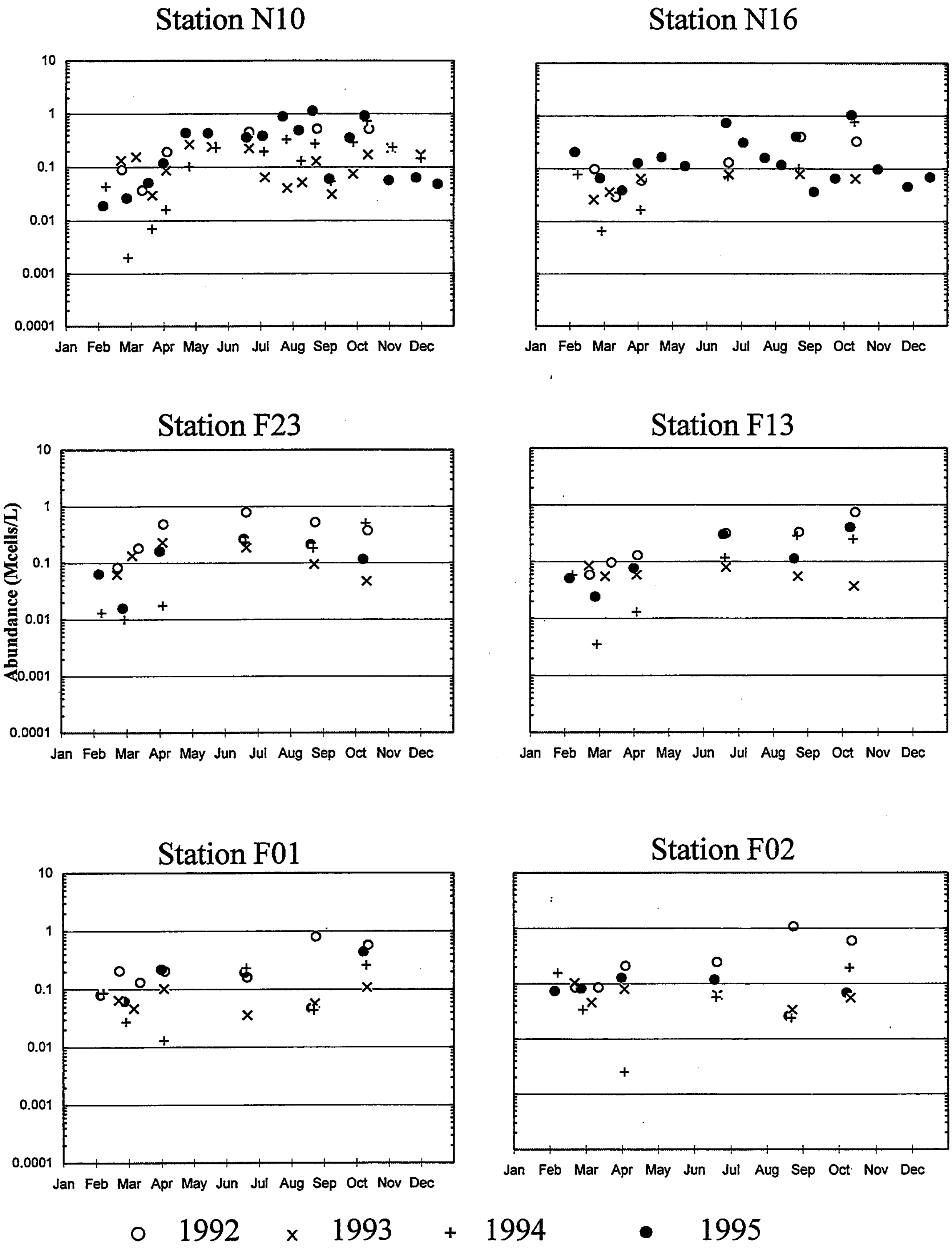
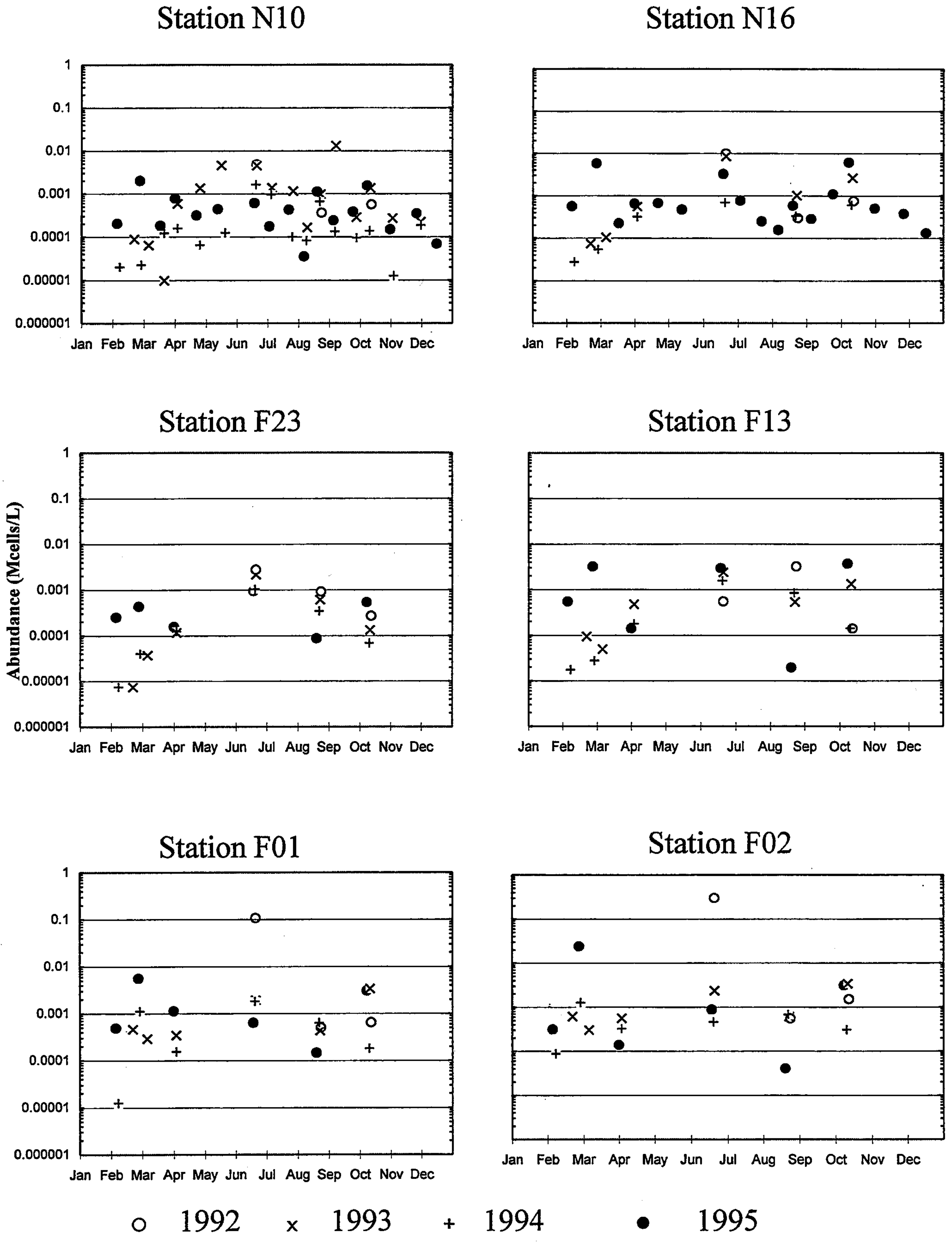


FIGURE 4-11

Dinoflagellates (screened samples)



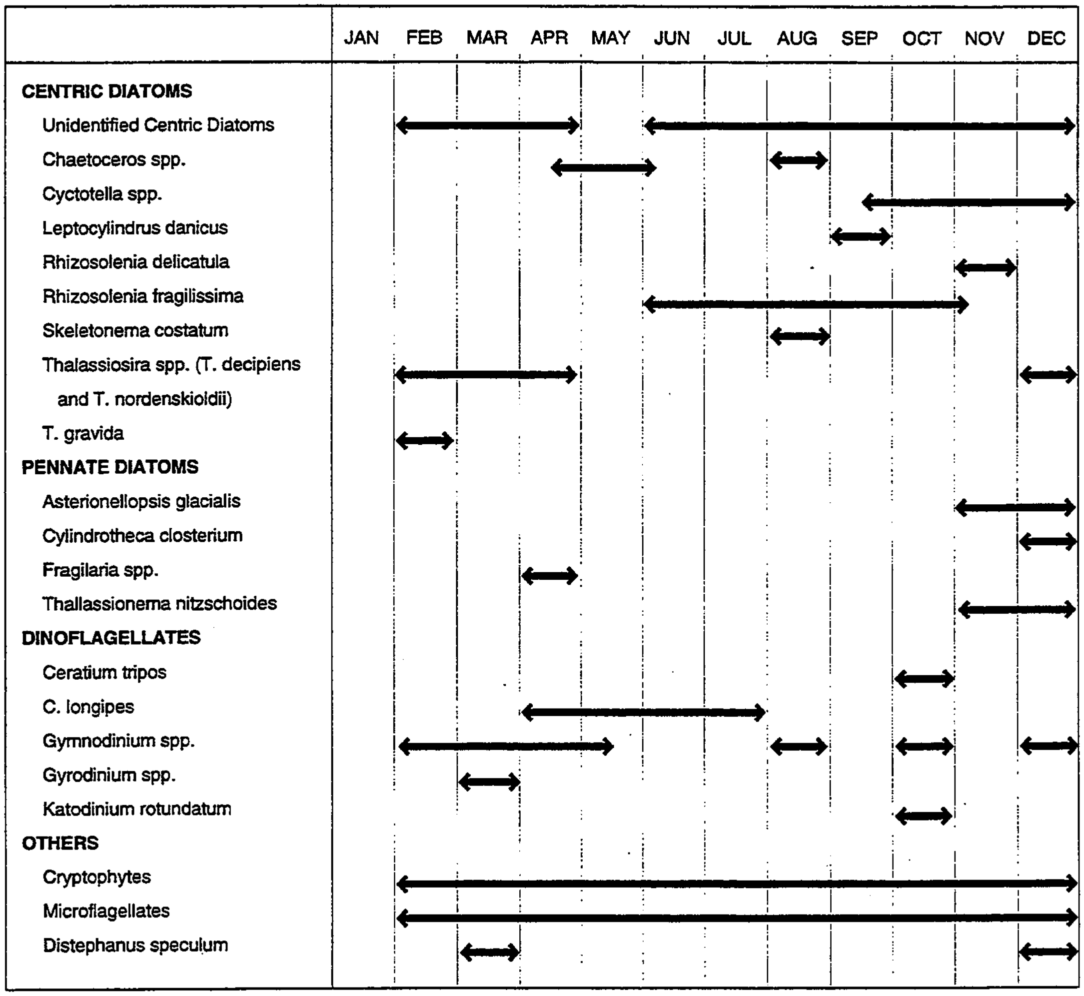


FIGURE 4-12
Dominant Phytoplankton Species (by Abundance) in Nearfield

M970099a

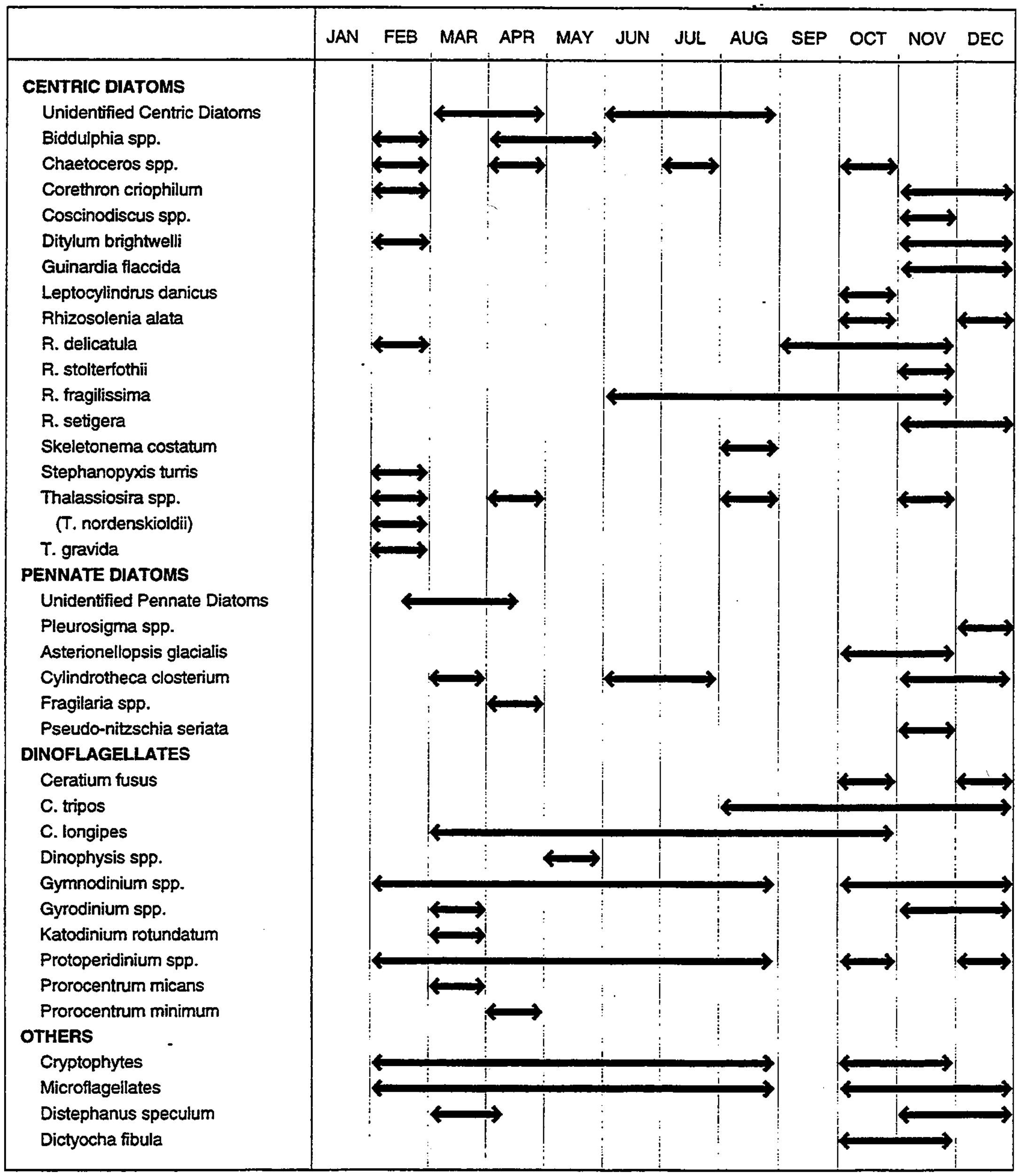


FIGURE 4-13
Dominant Phytoplankton Species (by Carbon) in Nearfield

M970165a

FIGURE 4-14

***Thalassiosira* spp.**

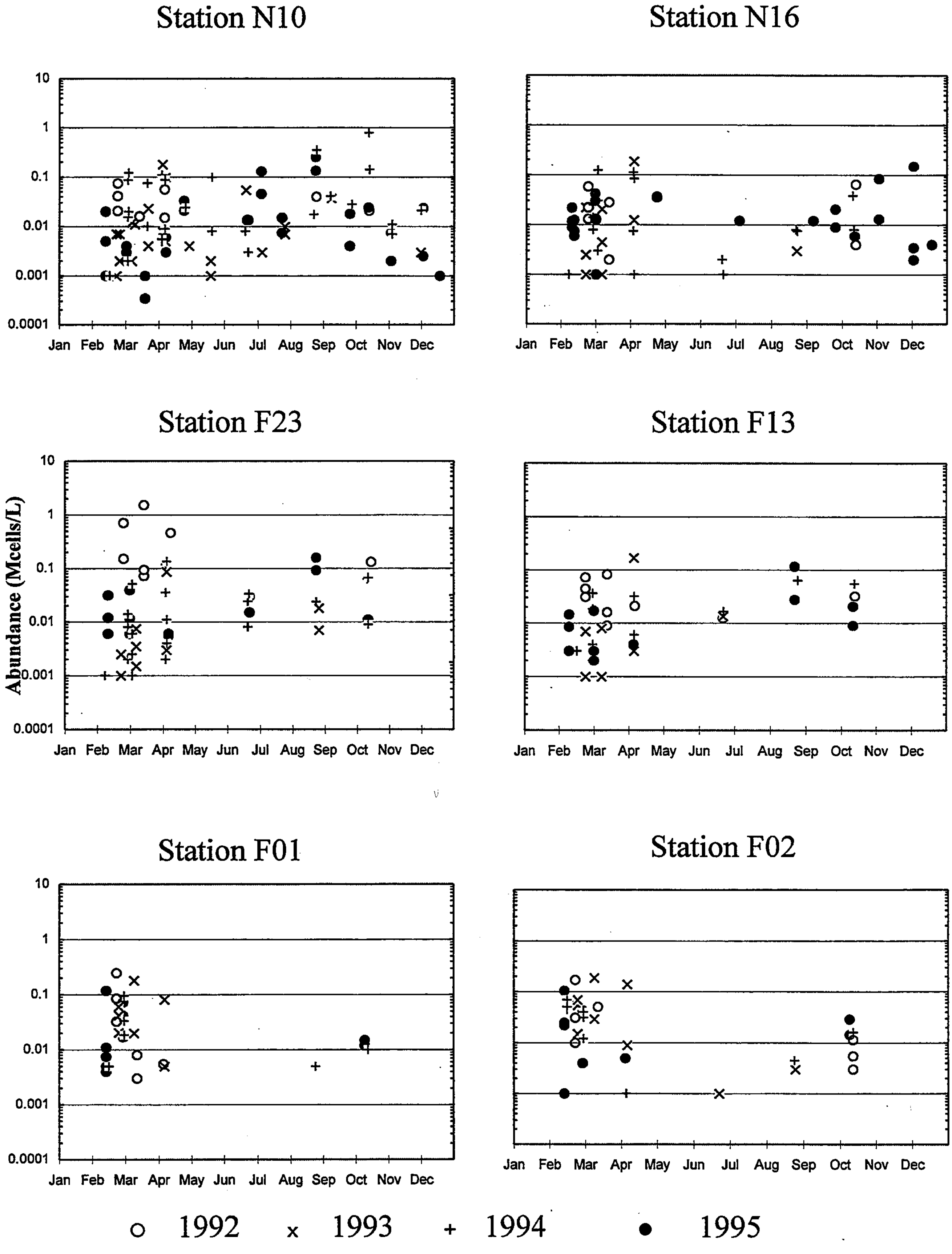


FIGURE 4-15

Thalassiosira nordenskioldii

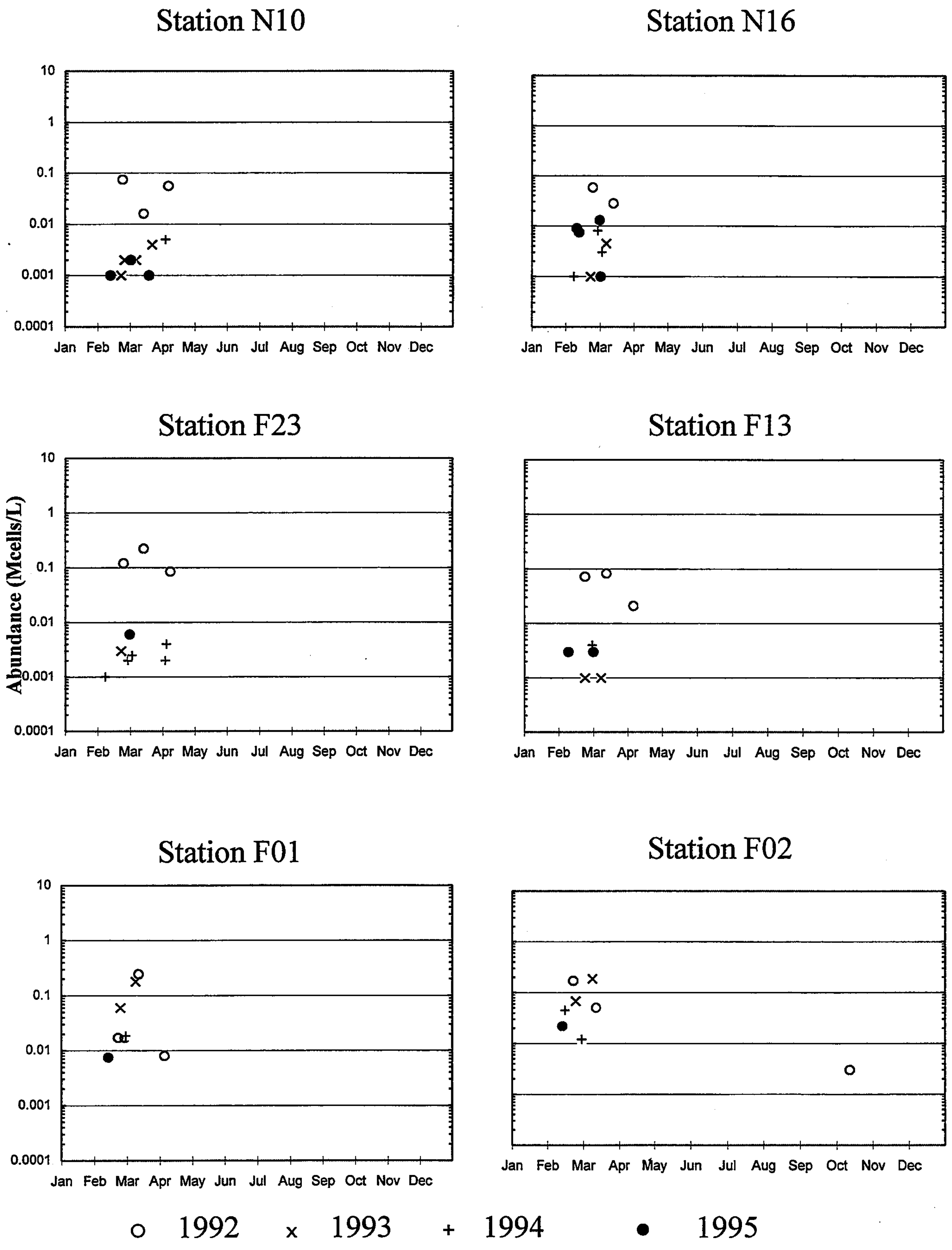


FIGURE 4-16

Thalassiosira gravida* and *rotula

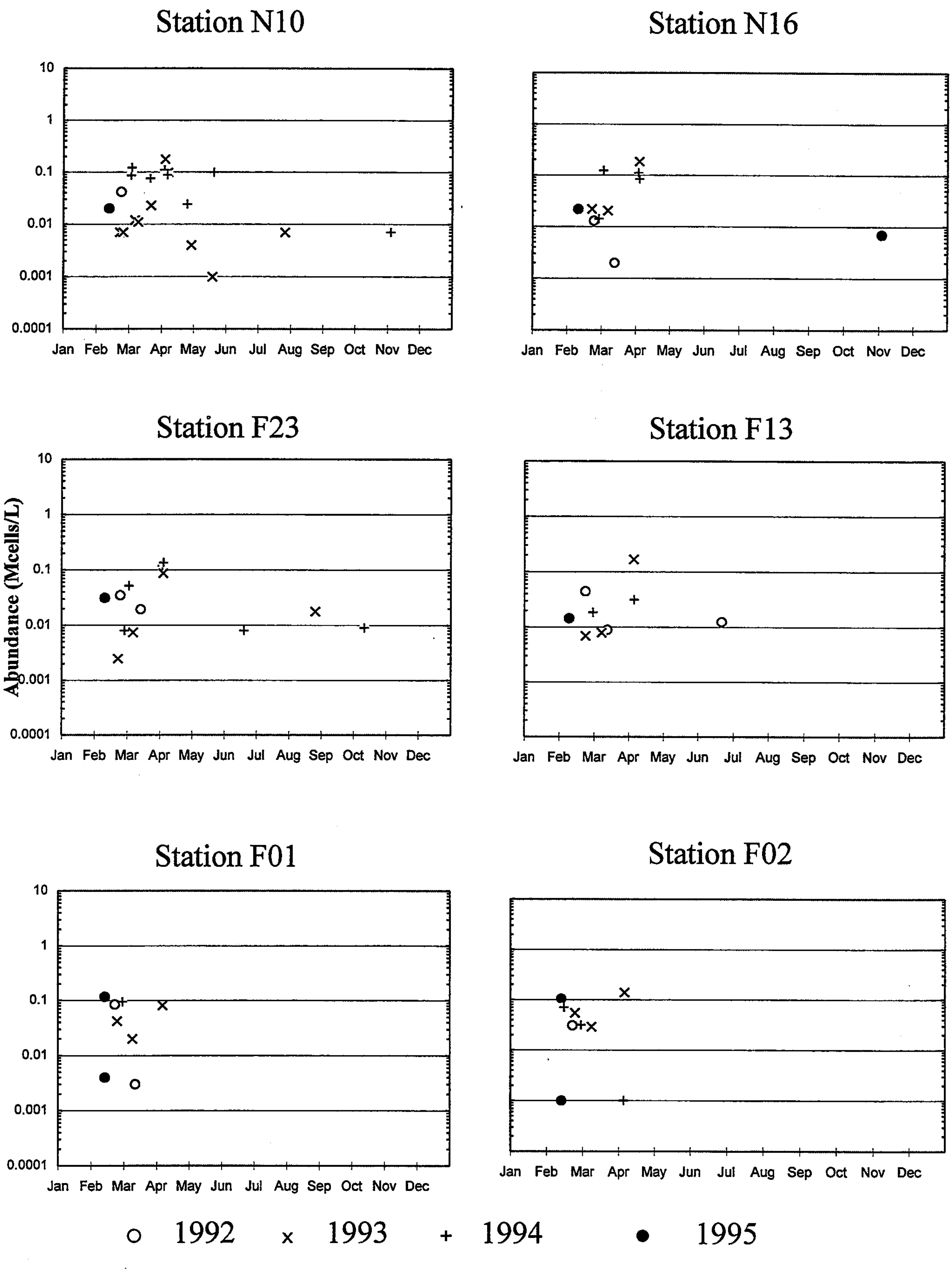
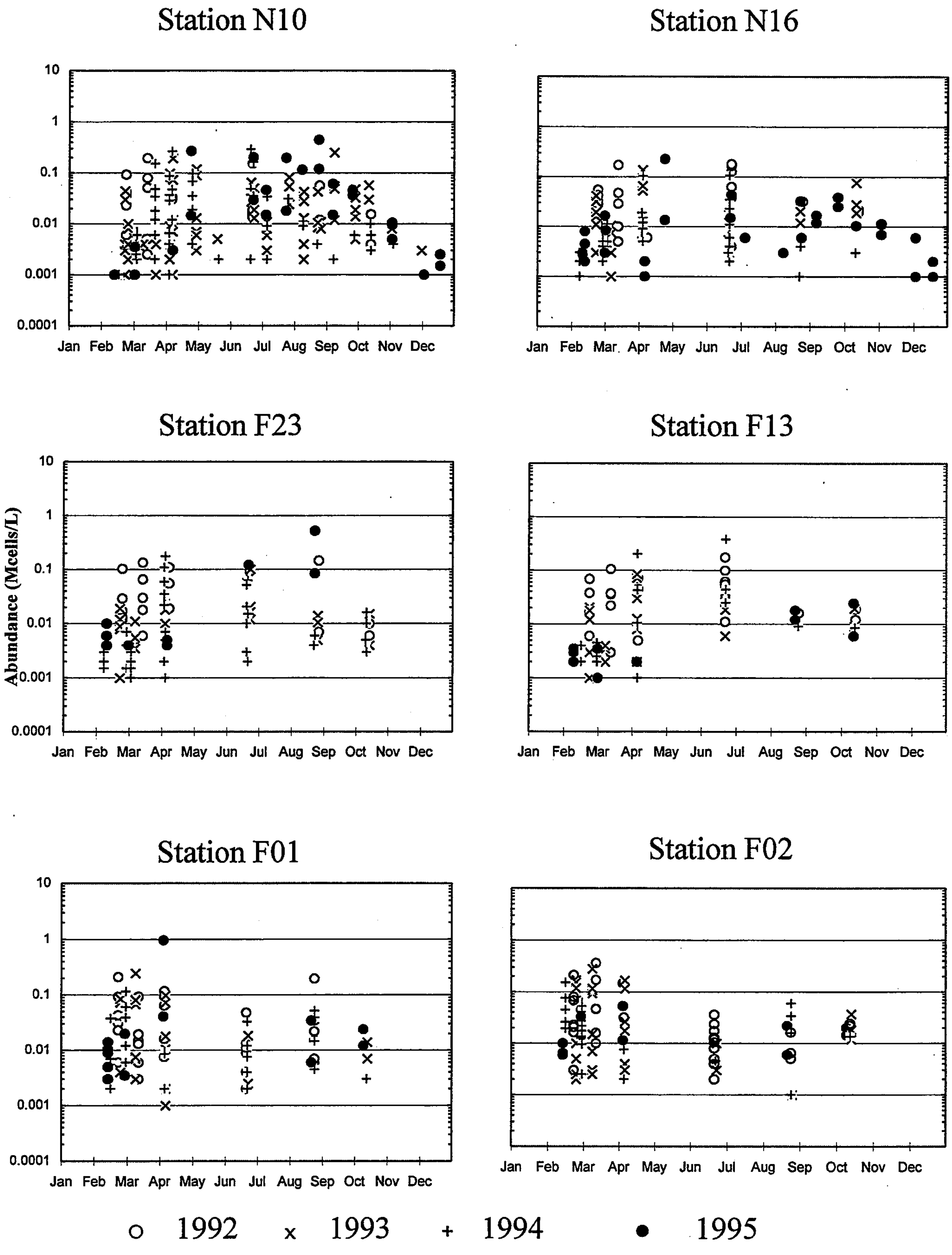


FIGURE 4-17

***Chaetoceros* spp.**


Station N10

Station N16

Station F23

Station F13

Station F01

Station F02

FIGURE 4-18

Chaetoceros debilis

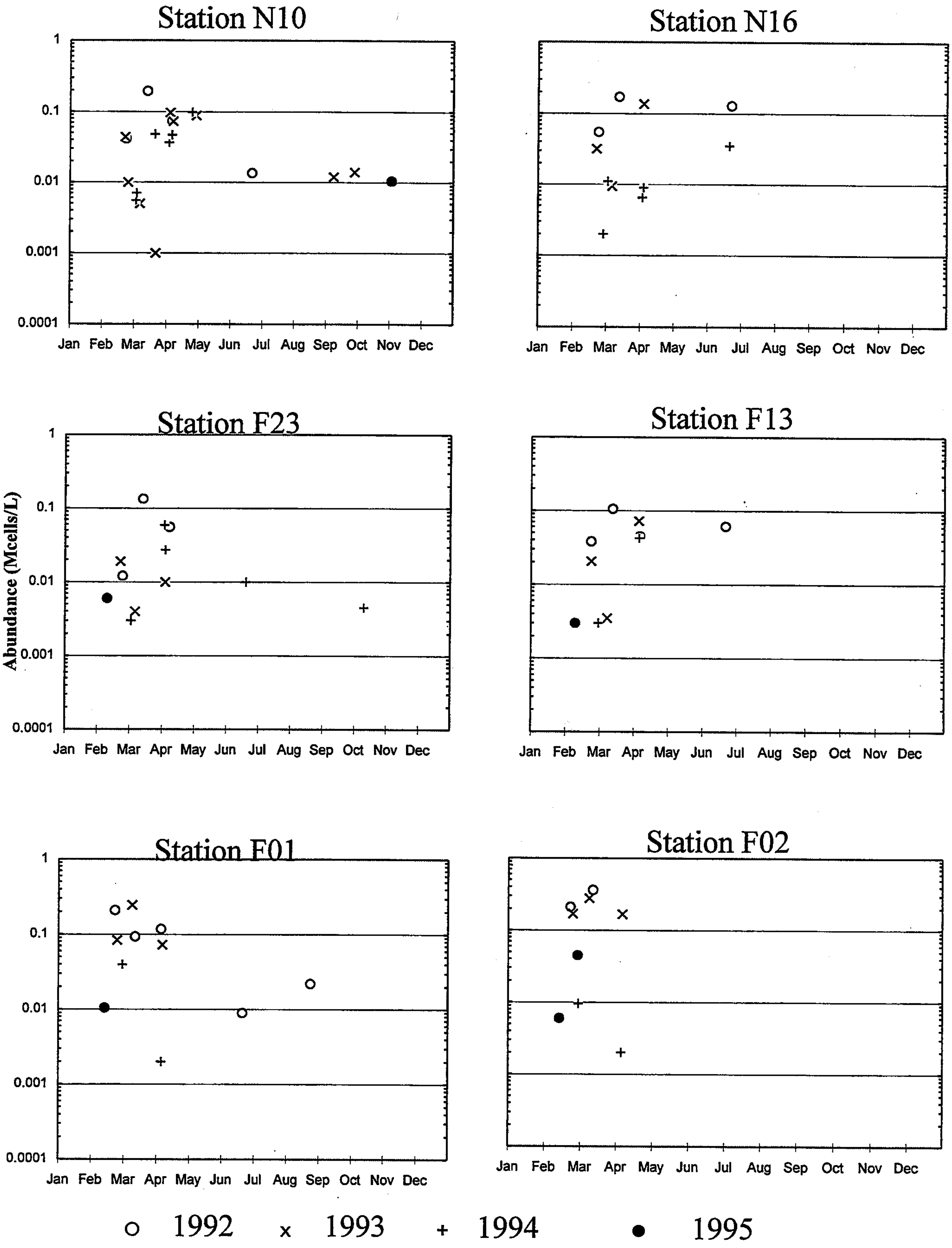


FIGURE 4-19

Skeletonema costatum

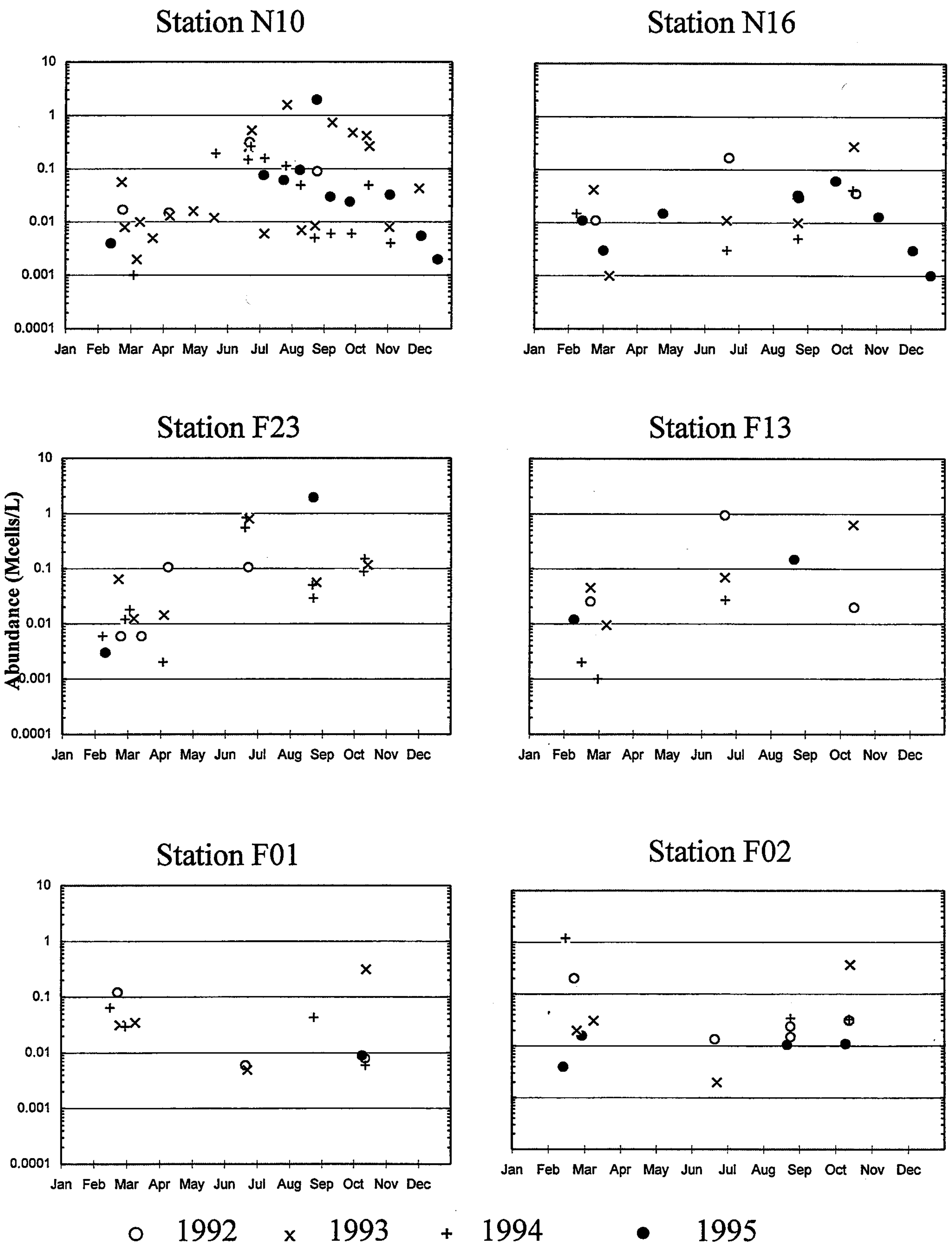


FIGURE 4-20

Rhizosolenia delicatula

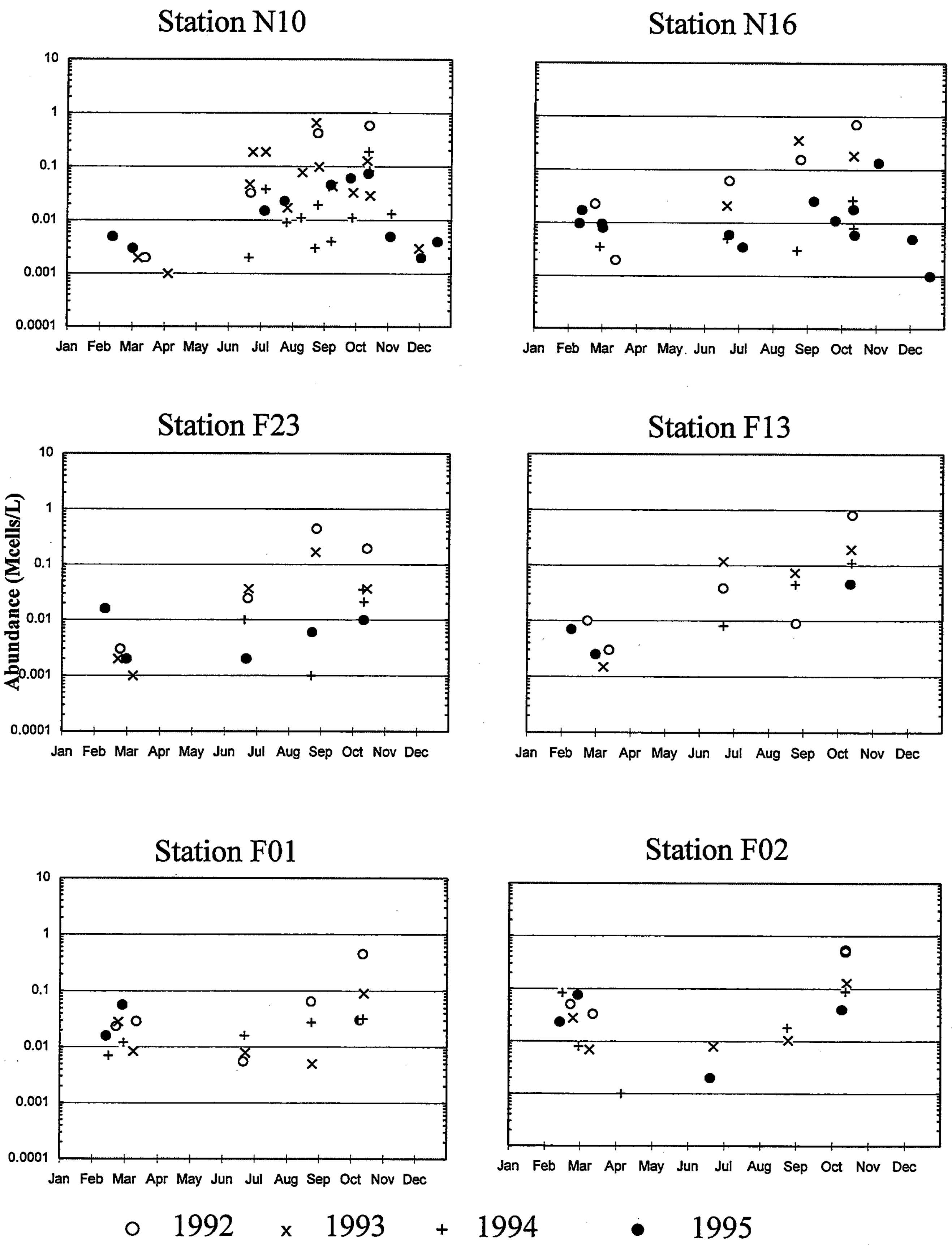


FIGURE 4-21

Rhizosolenia fragilissima

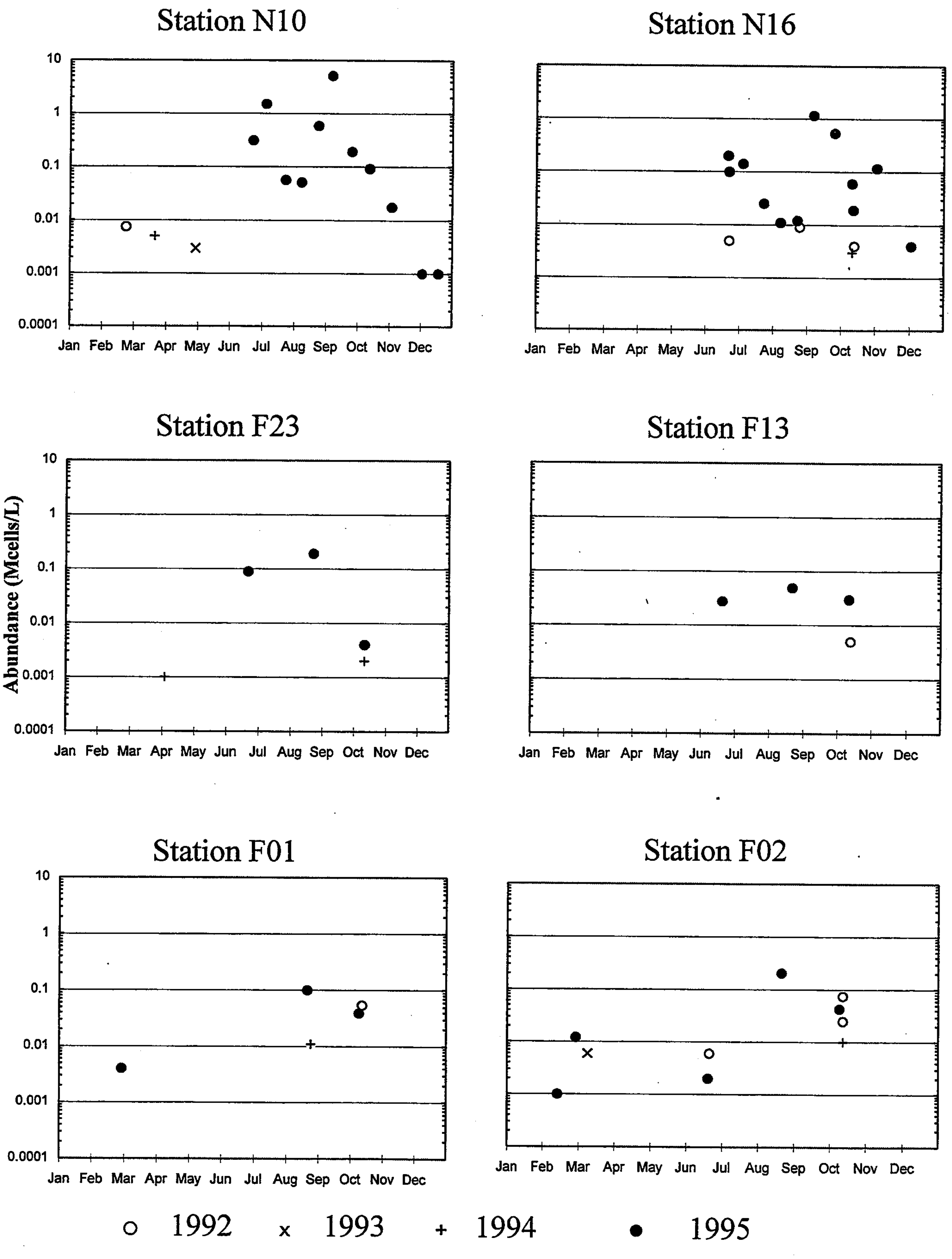


FIGURE 4-22

Leptocylindrus danicus

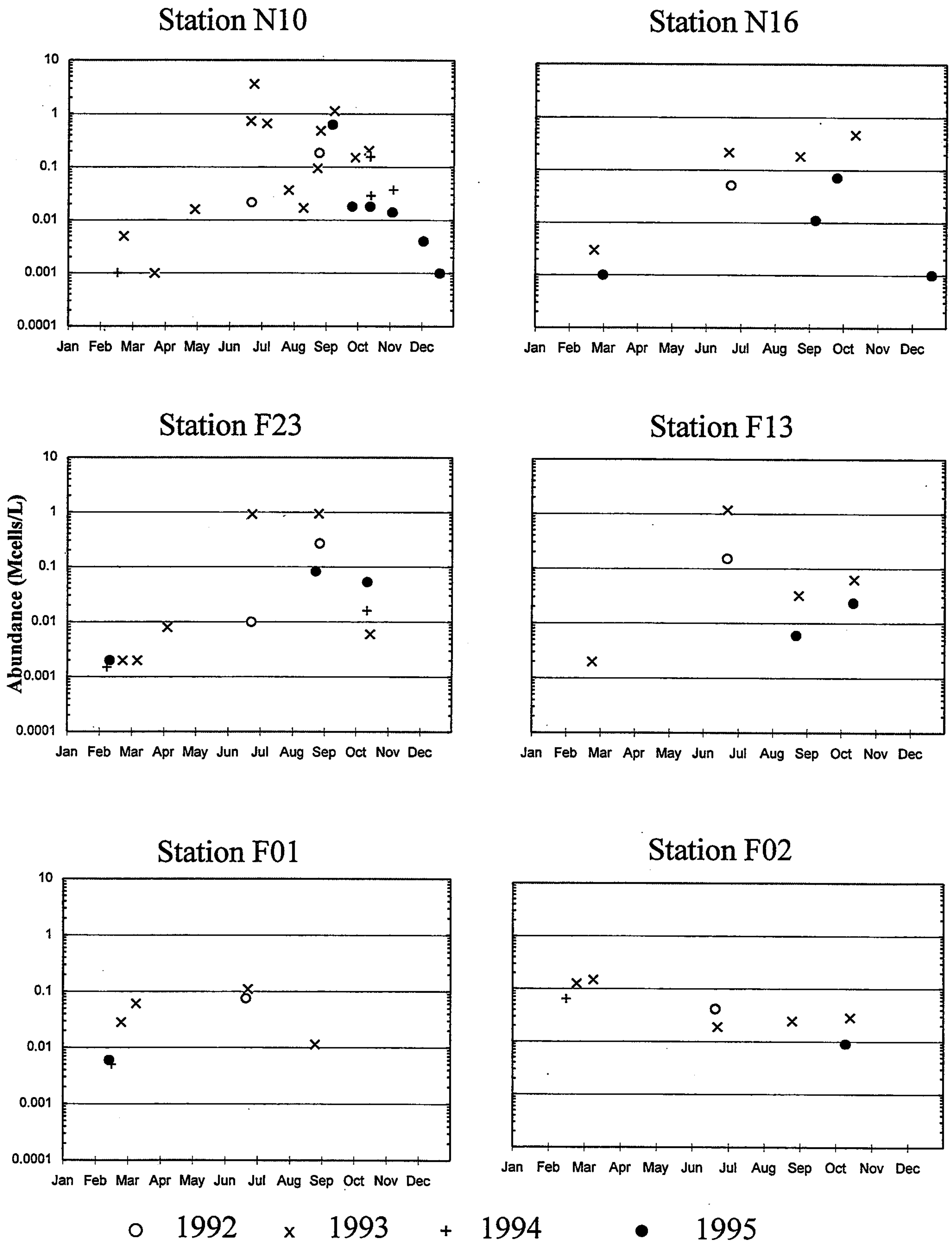
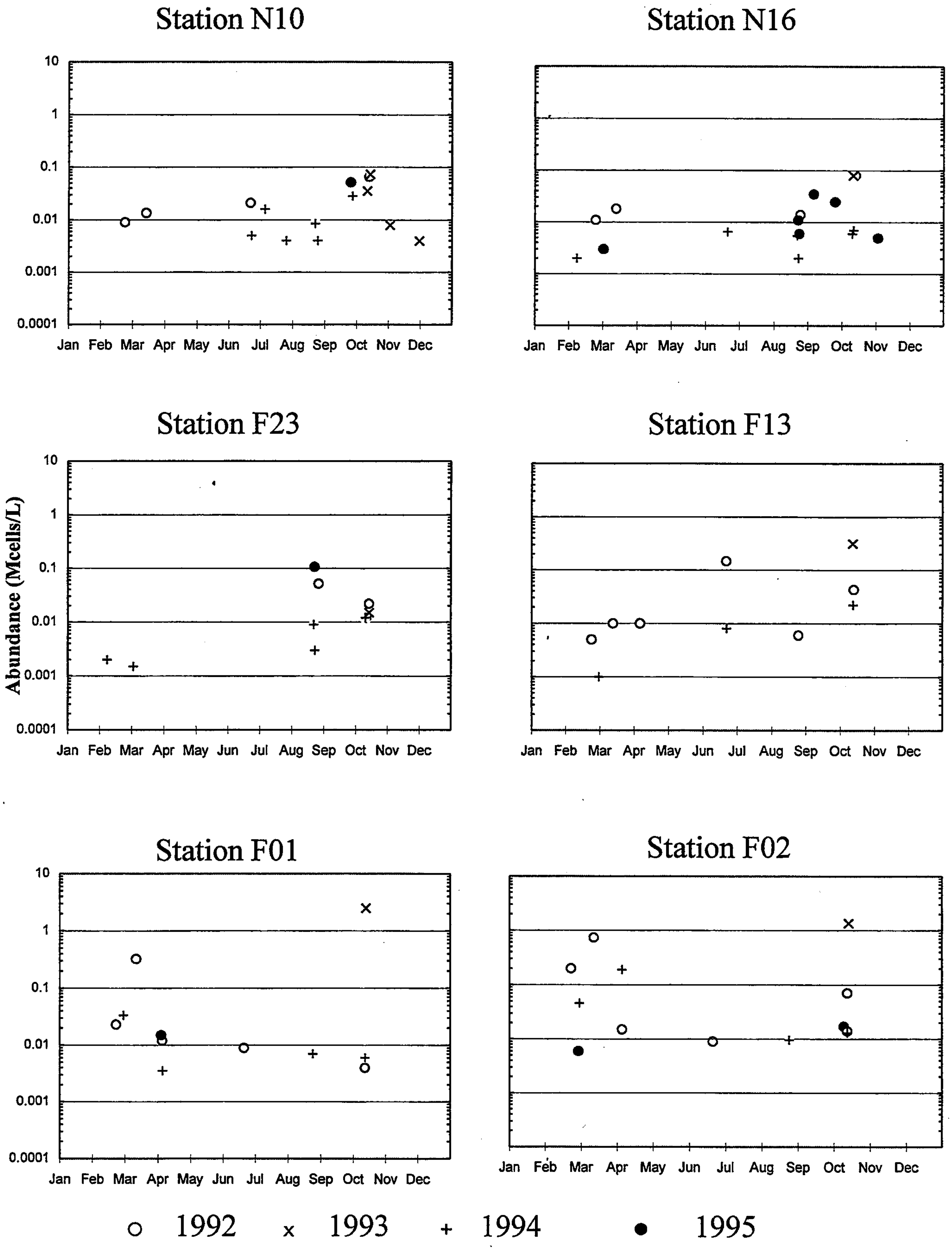


FIGURE 4-23

Leptocylindrus minimus



Station N10

Station N16

Station F23

Station F13

Station F01

Station F02

FIGURE 4-24

Asterionellopsis glacialis

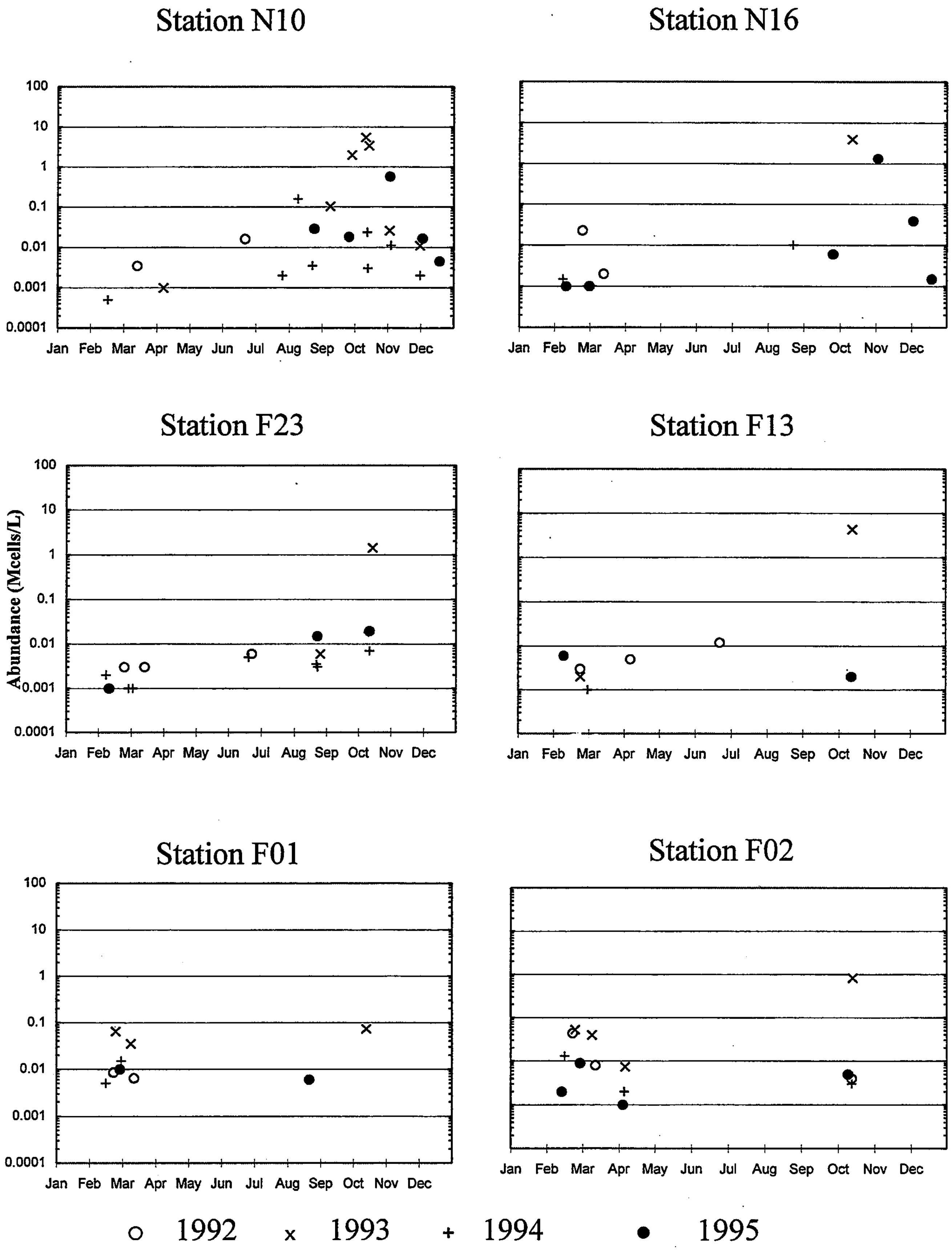


FIGURE 4-25
Pseudo-nitzschia pungens

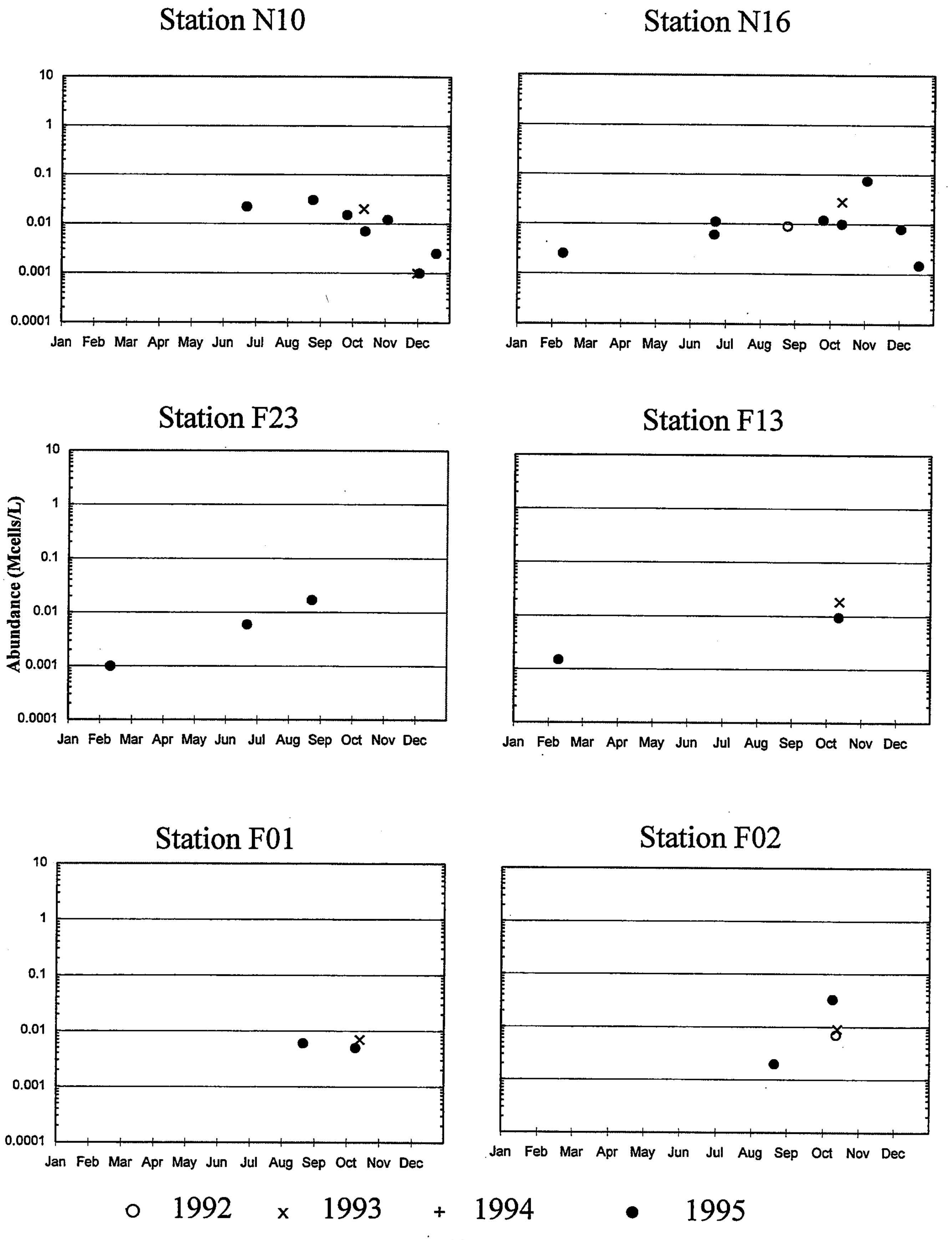
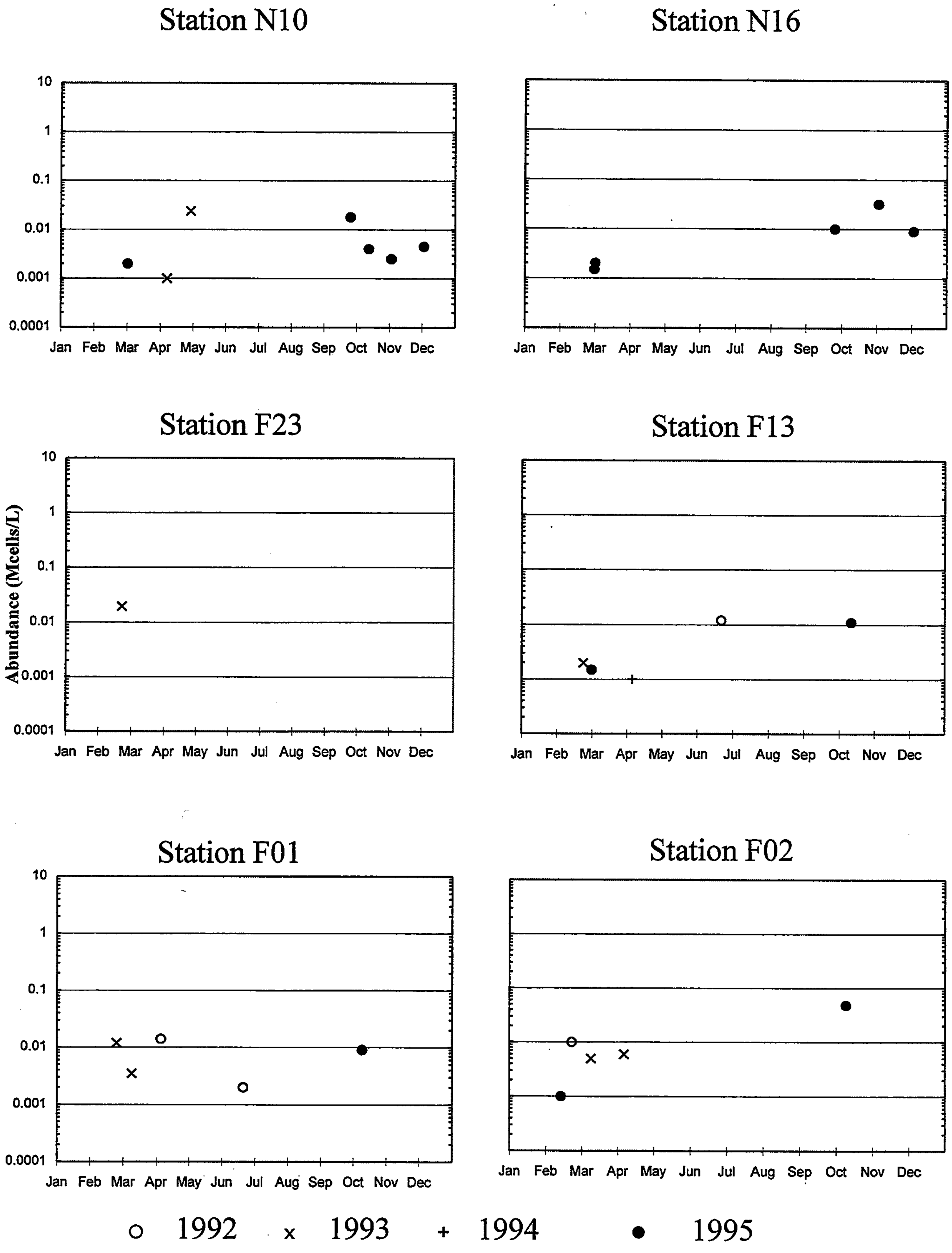


FIGURE 4-26

Pseudo-nitzschia seriata



Station N10

Station N16

Station F23

Station F13

Station F01

Station F02

FIGURE 4-27
Ceratium spp.

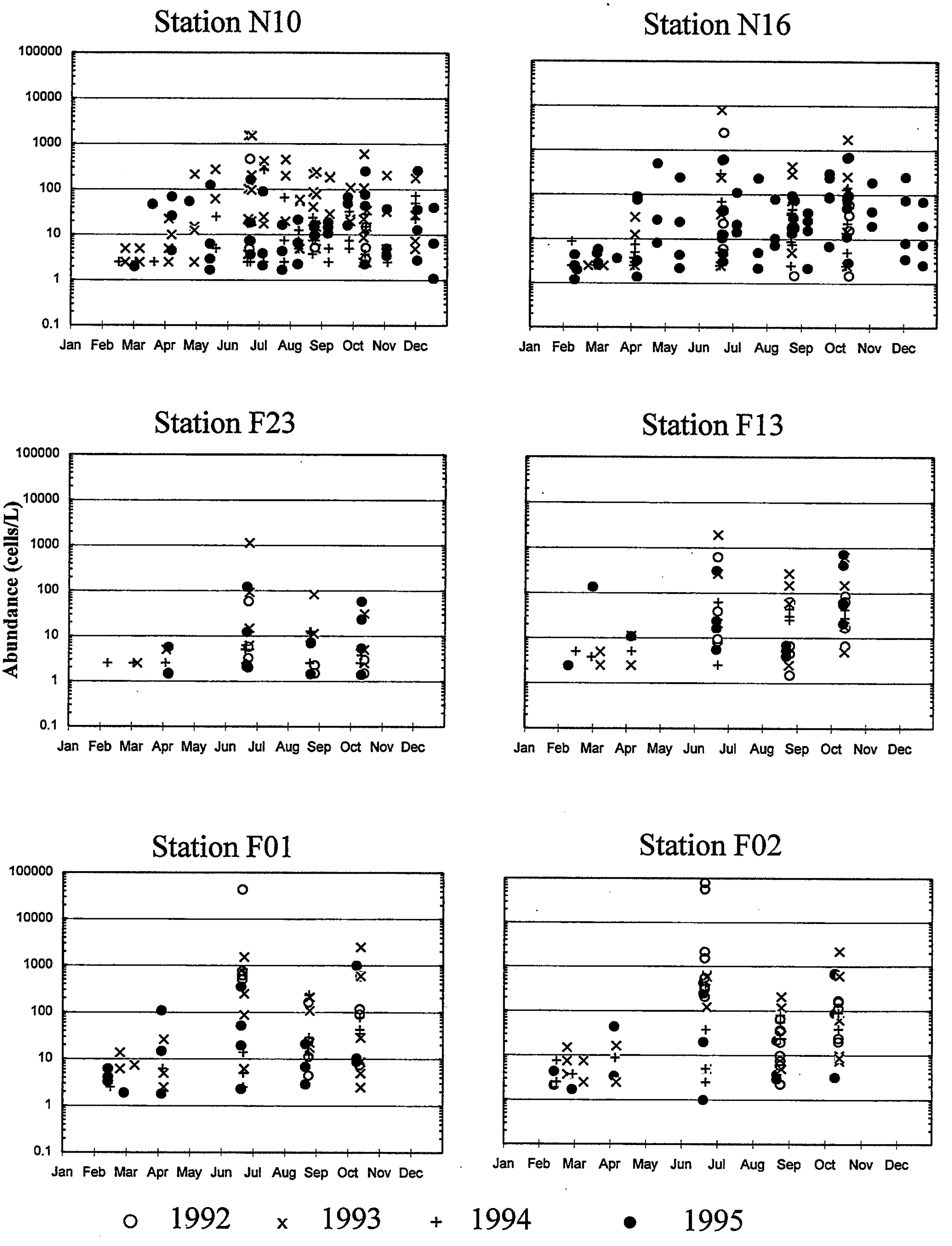


FIGURE 4-28
Protoperidinium spp.

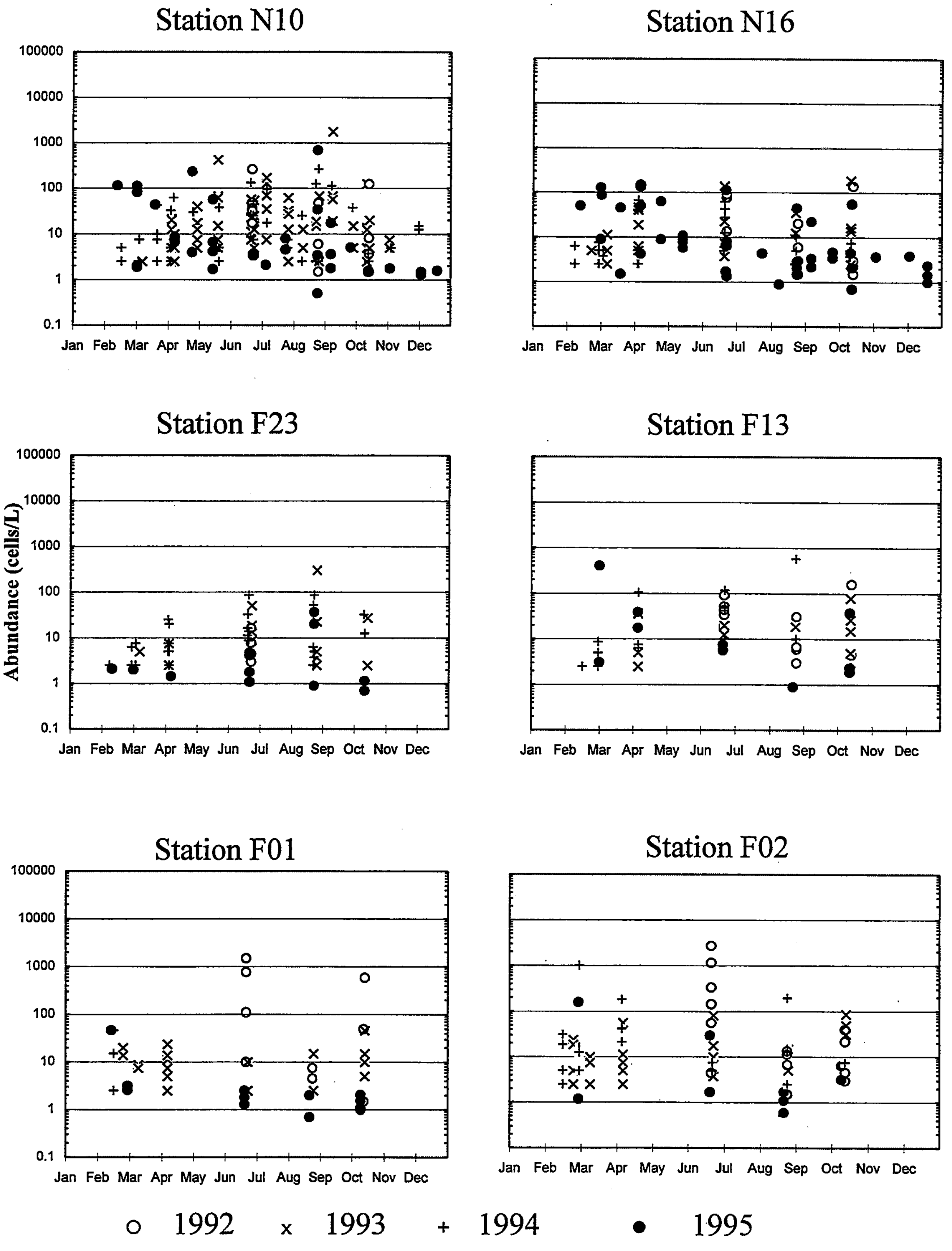


FIGURE 4-29
Gymnodinium spp.

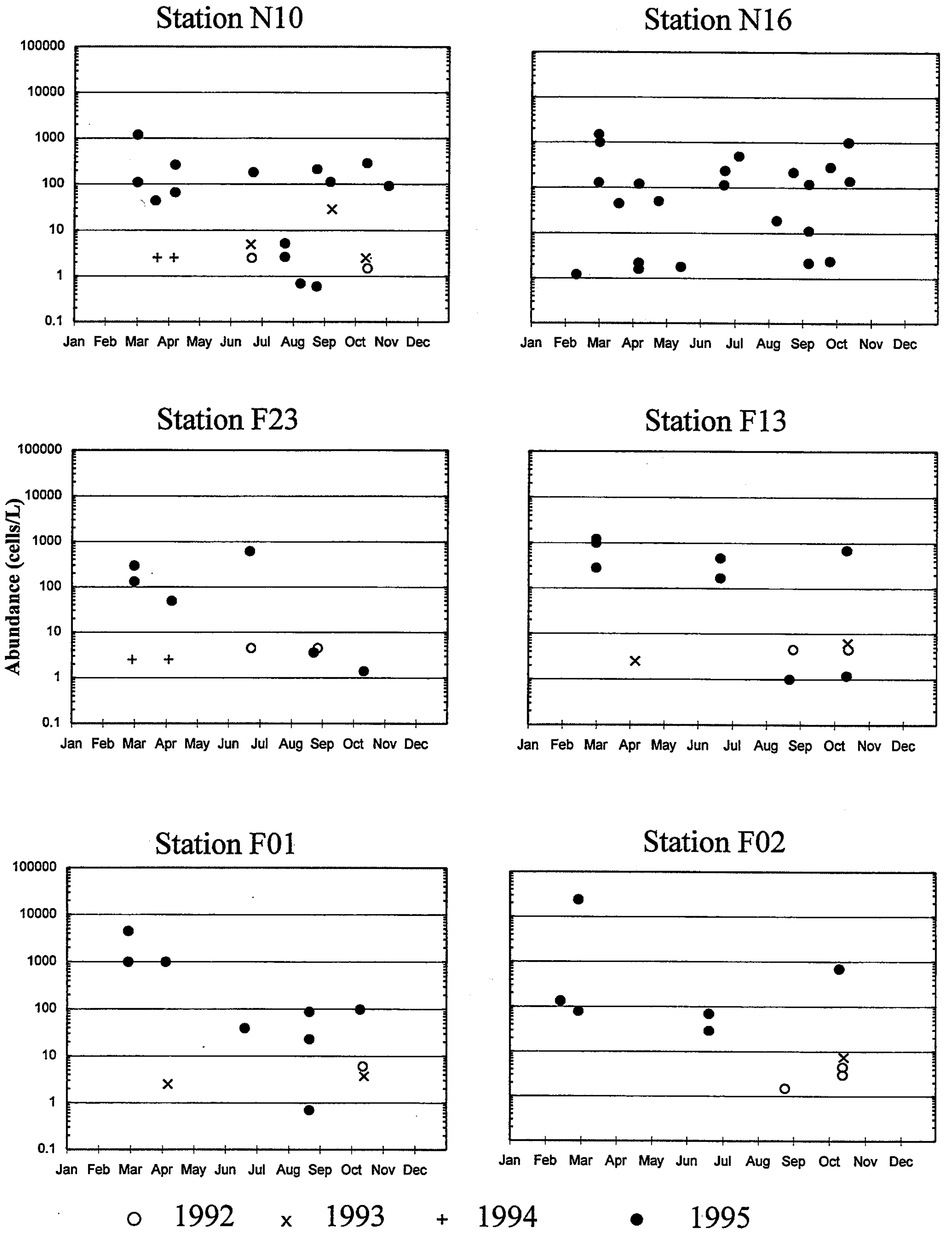


FIGURE 4-30

Alexandrium tamarense

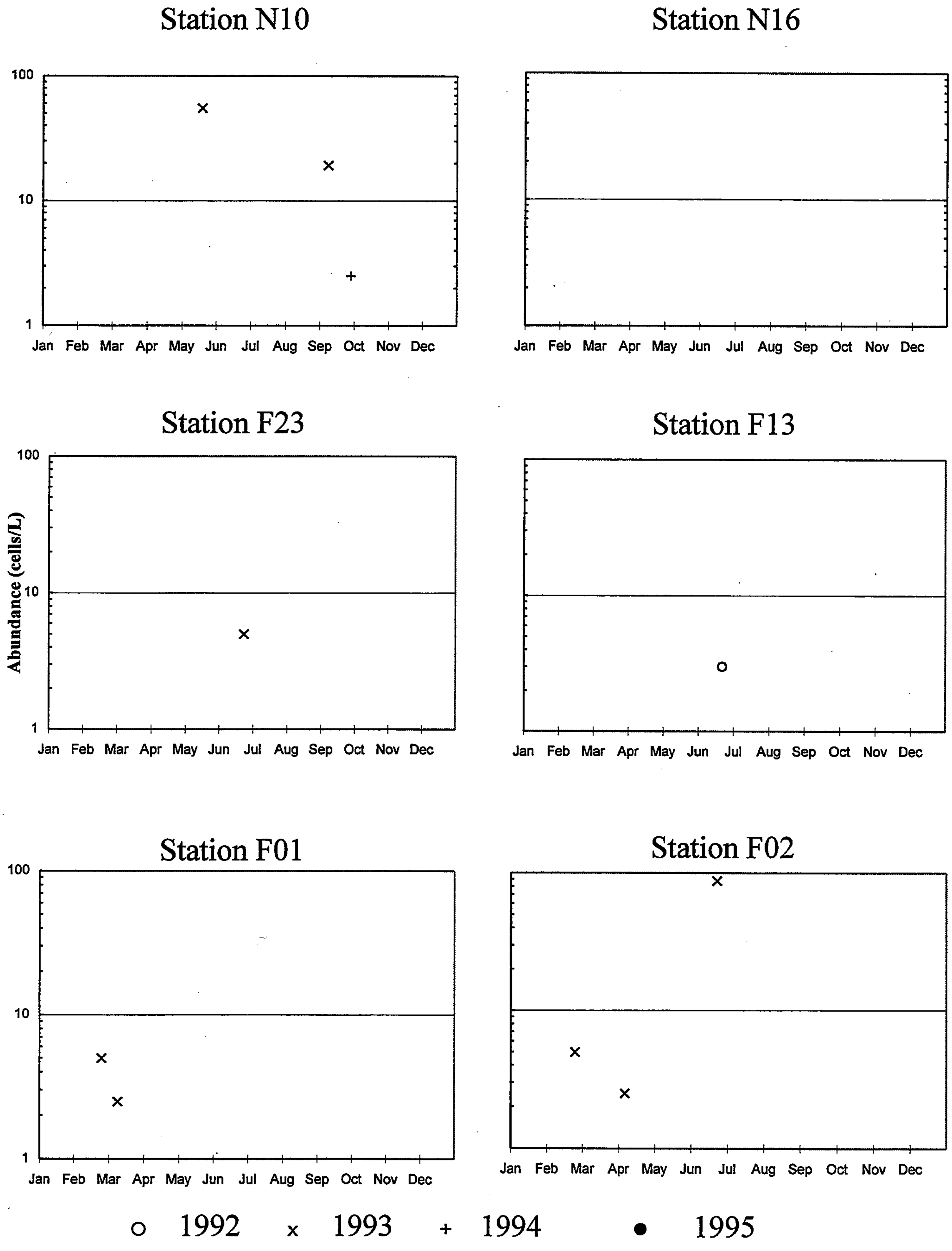


FIGURE 4-31

Phaeocystis pouchetii

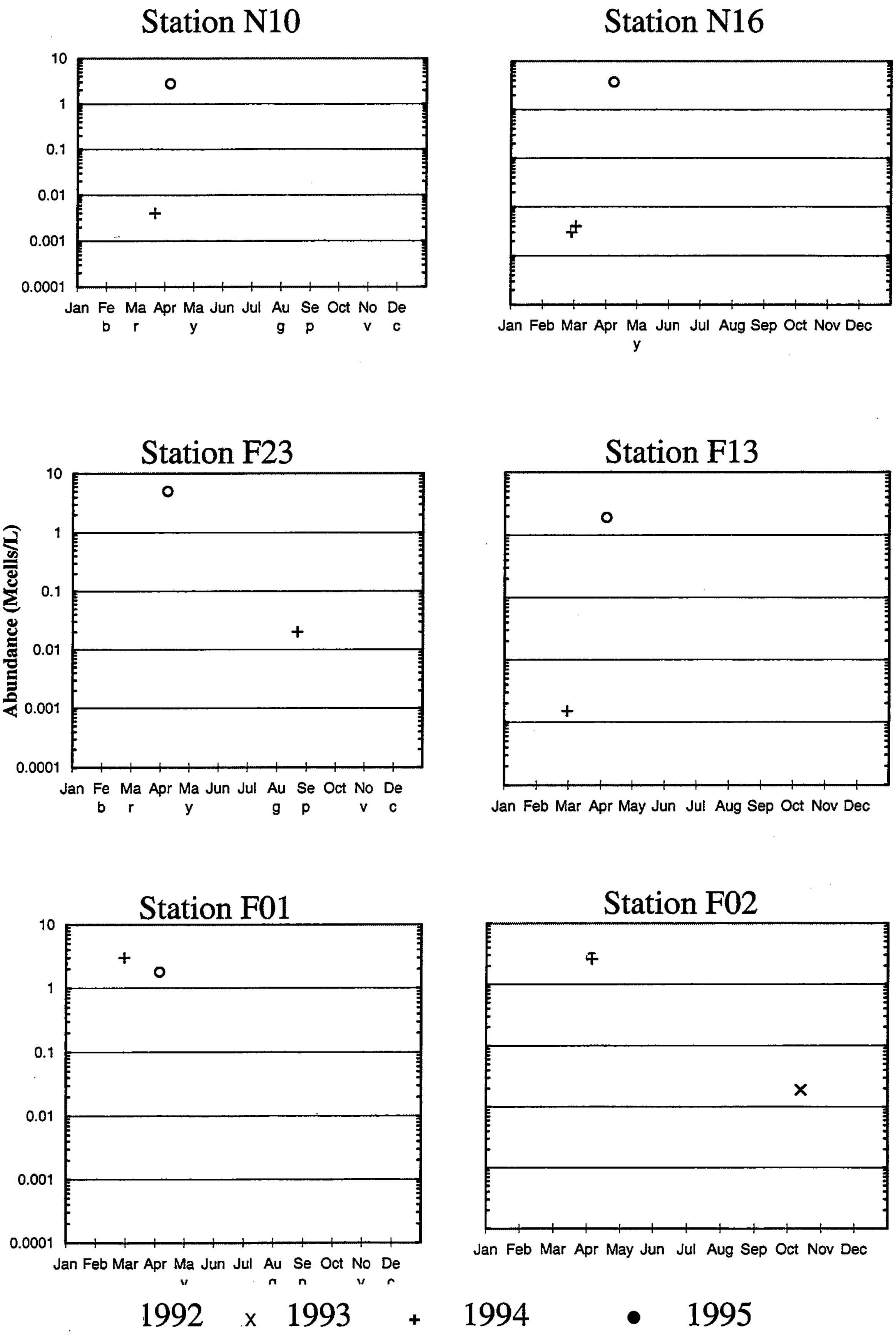


FIGURE 4-32

Total Zooplankton

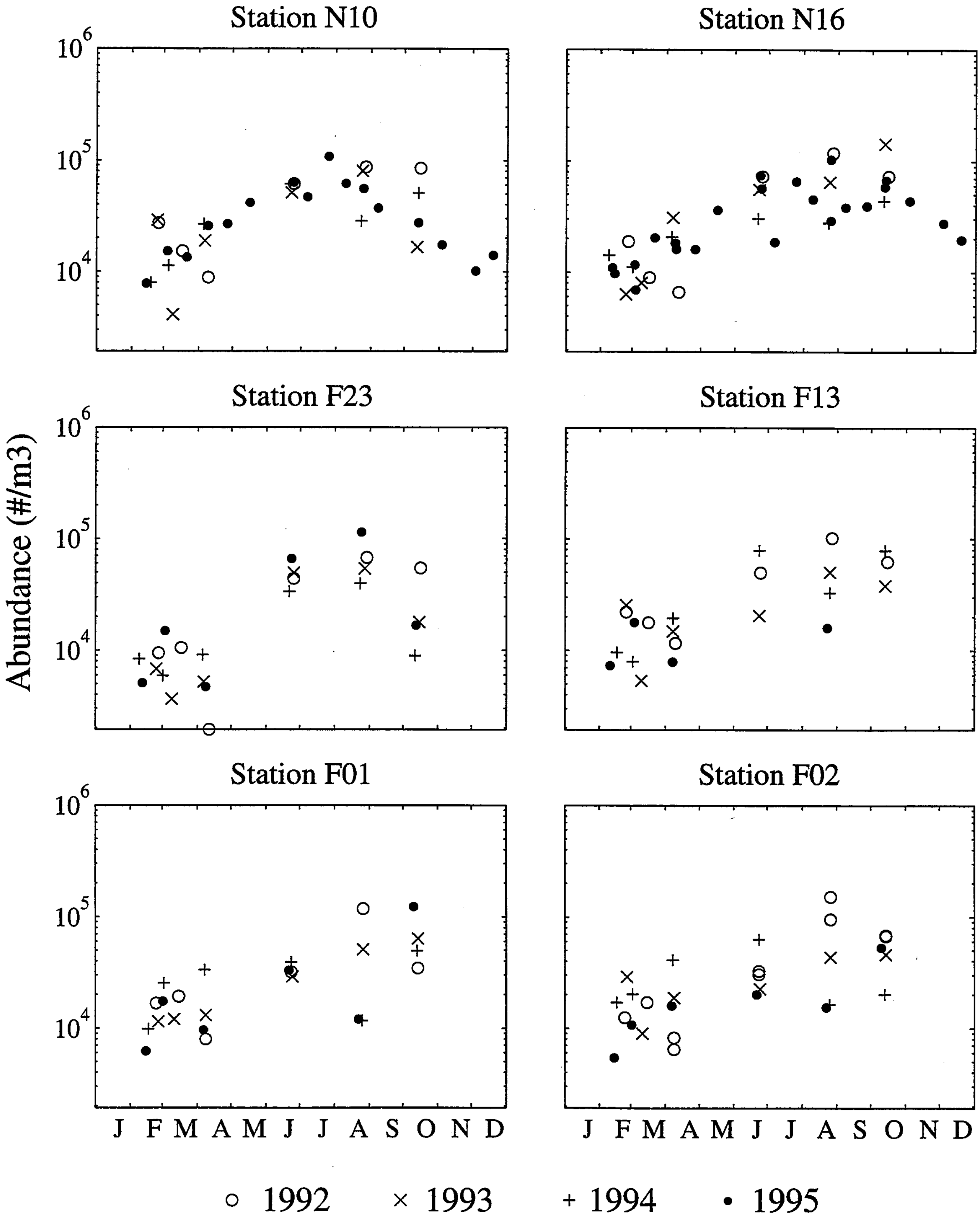


FIGURE 4-33

Copepod Nauplii

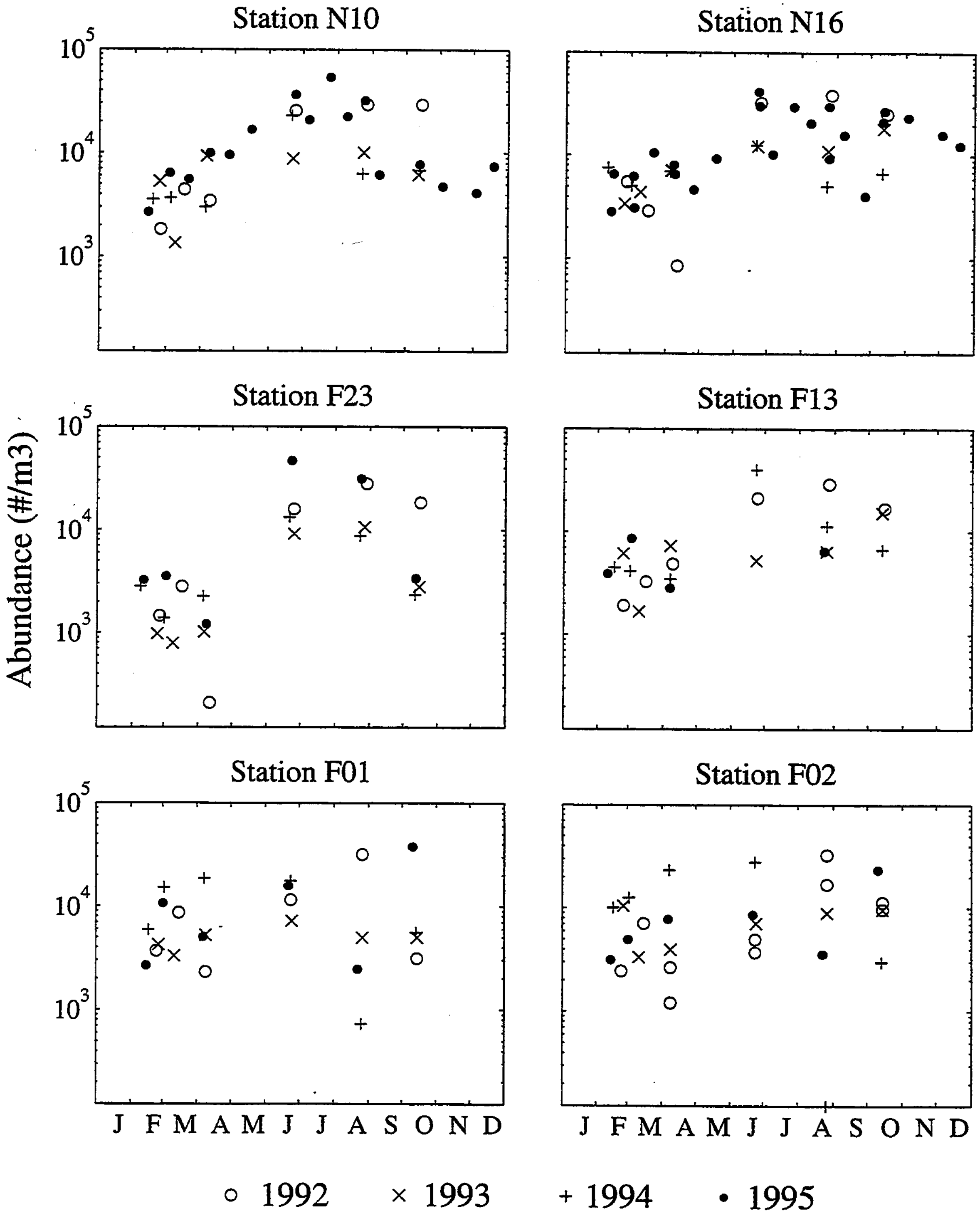
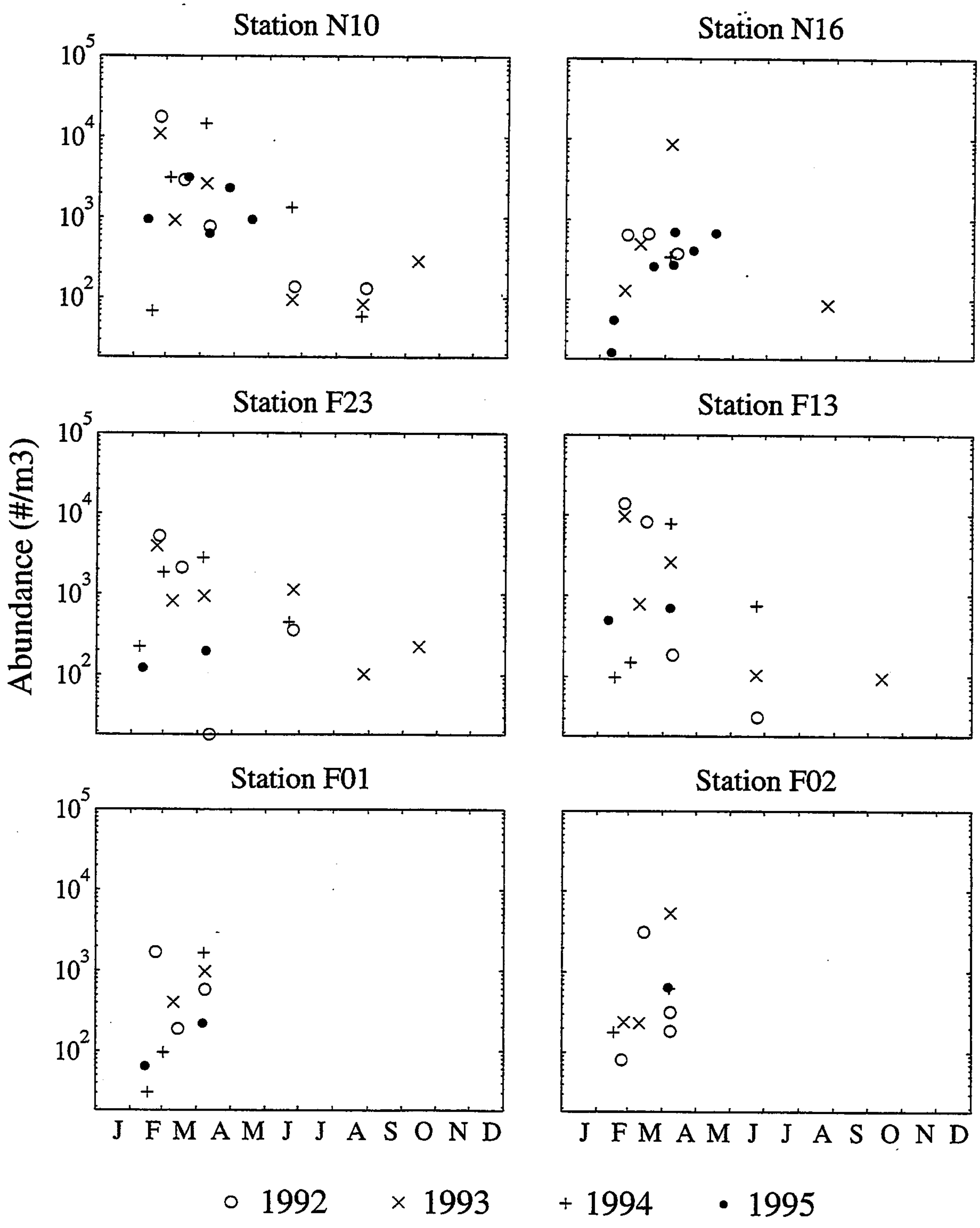
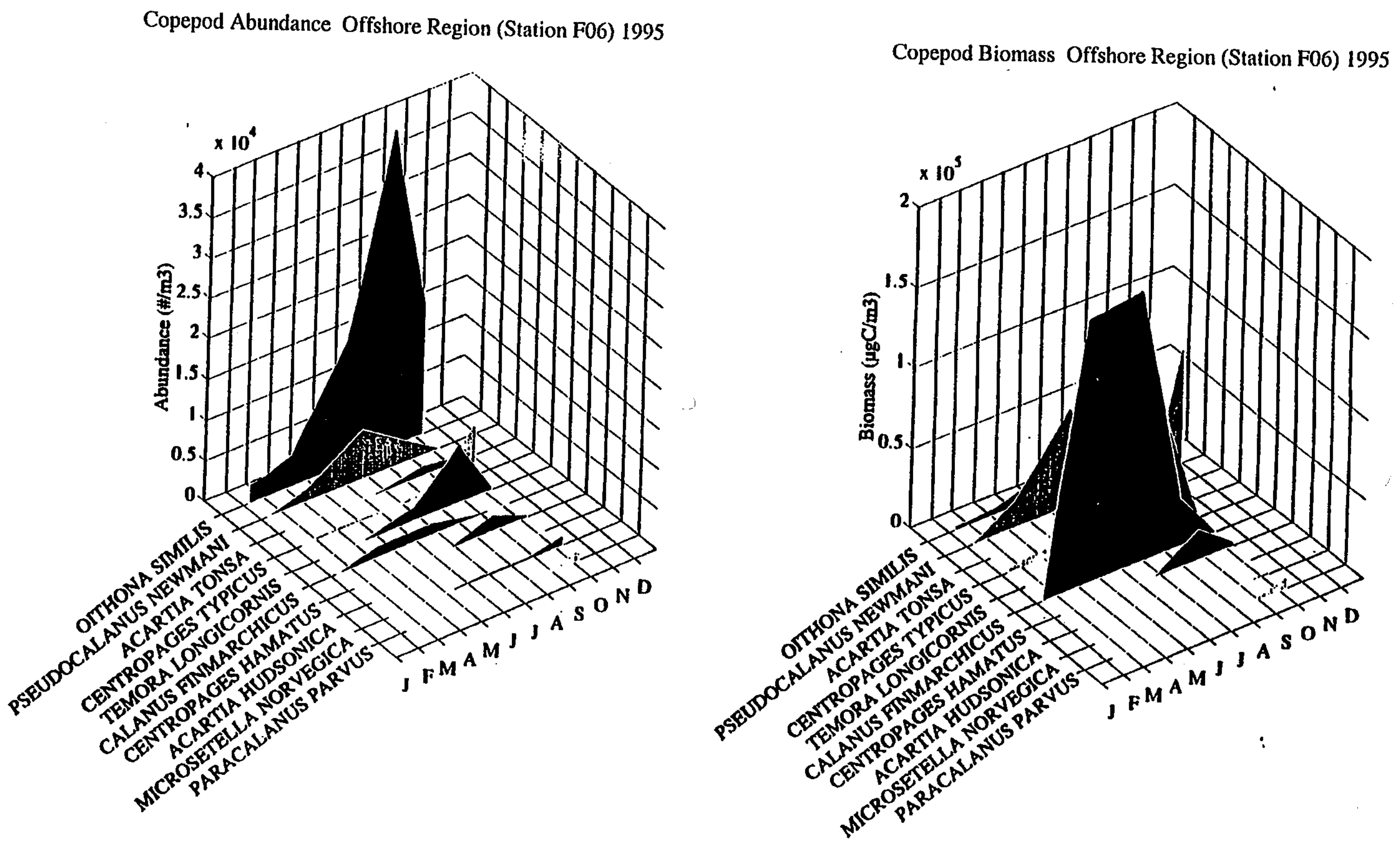


FIGURE 4-34
Barnacle Nauplii

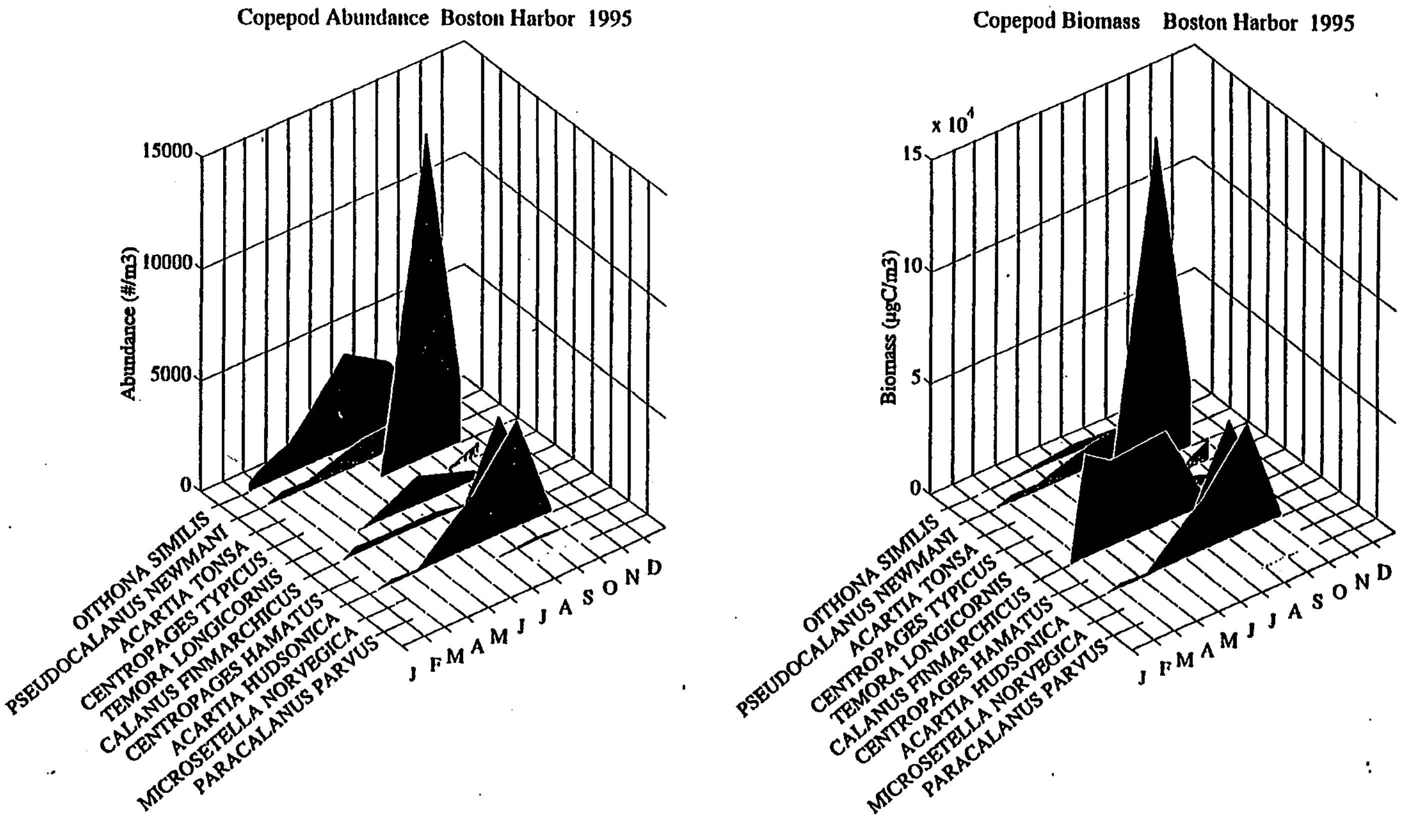




Copepod Abundance Offshore Region (Station F06) 1995

Copepod Biomass Offshore Region (Station F06) 1995

FIGURE 4-35
1995 Seasonal Abundance and Biomass
of Dominant Copepod Species at Offshore Stations



Copepod Abundance Boston Harbor 1995

Copepod Biomass Boston Harbor 1995

FIGURE 4-36

1995 Seasonal Abundance and Biomass of Dominant Copepod Species at Boston Harbor Stations

(F23, F30, and F31)

FIGURE 4-37
Calanus finmarchicus

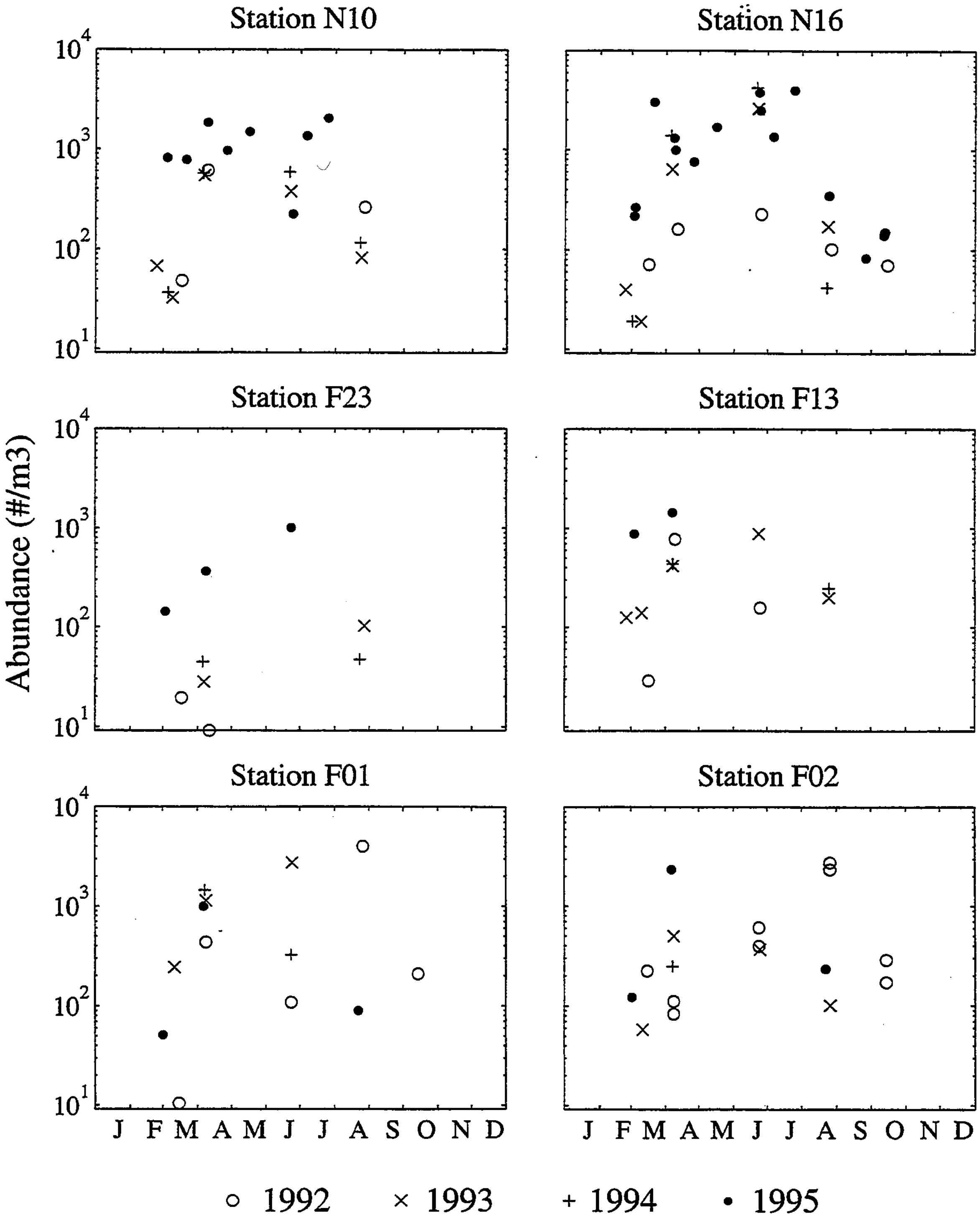


FIGURE 4-38

Oithona similis

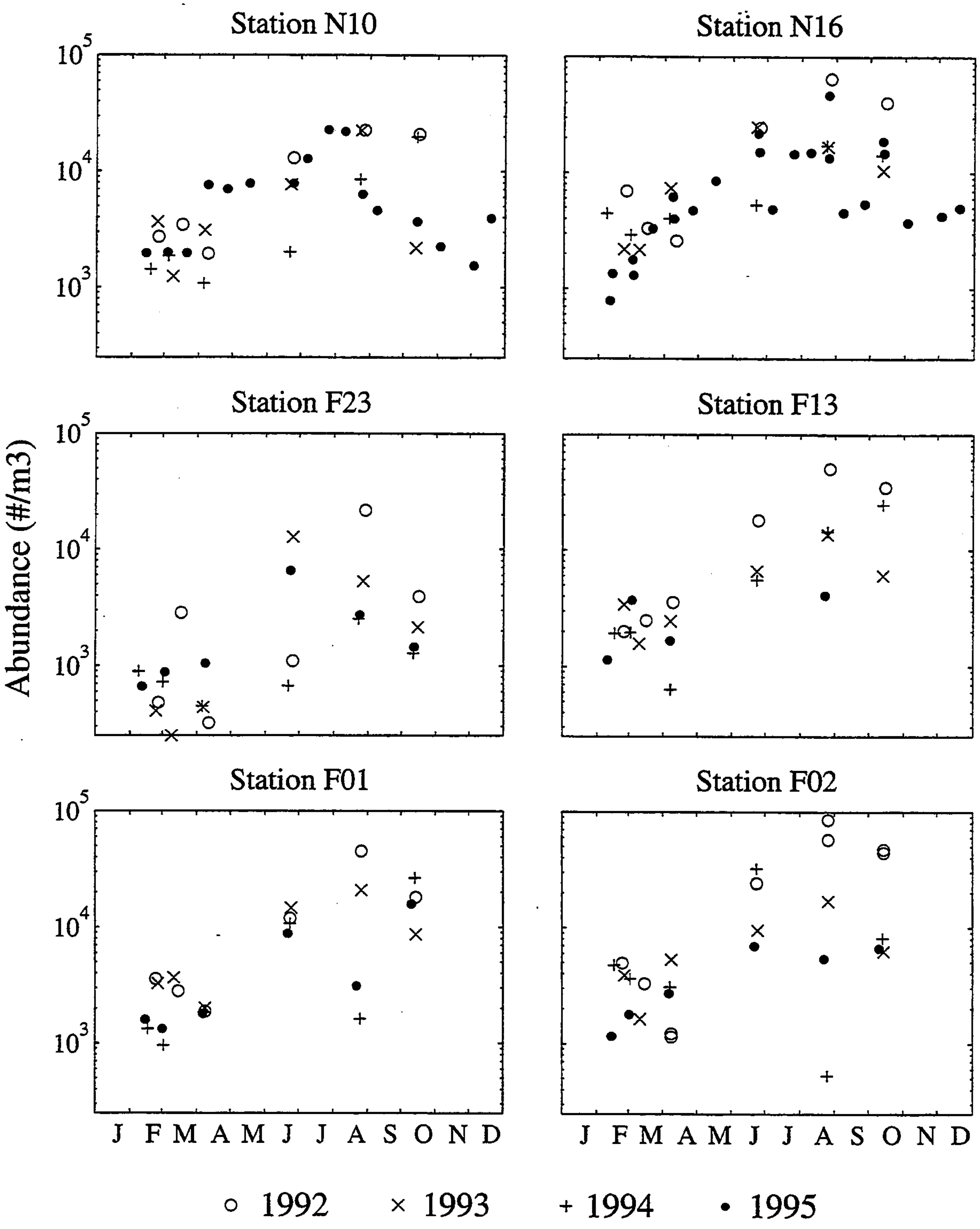


FIGURE 4-39

Pseudocalanus newmani + Paracalanus parvus

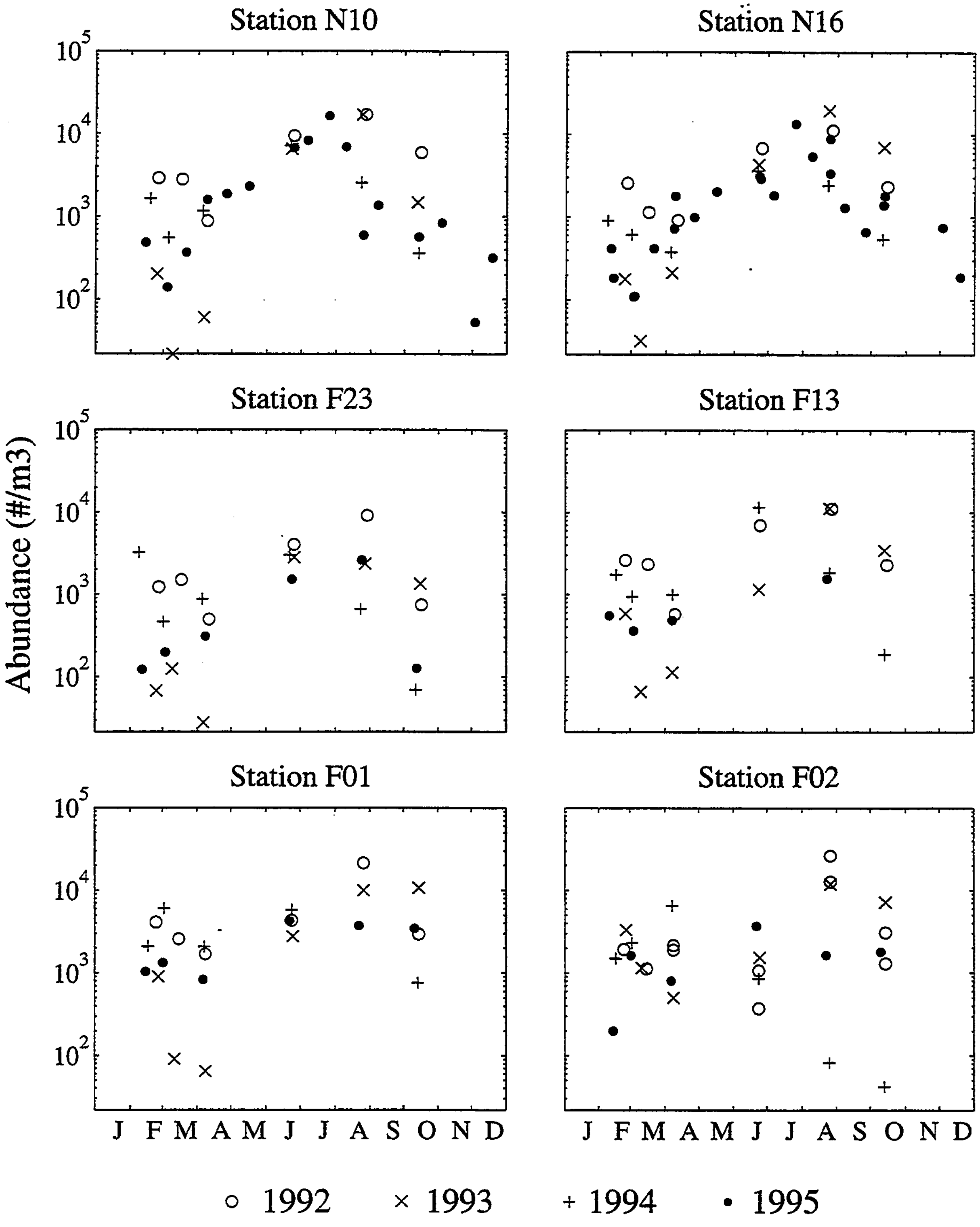


FIGURE 4-40
Centropages typicus

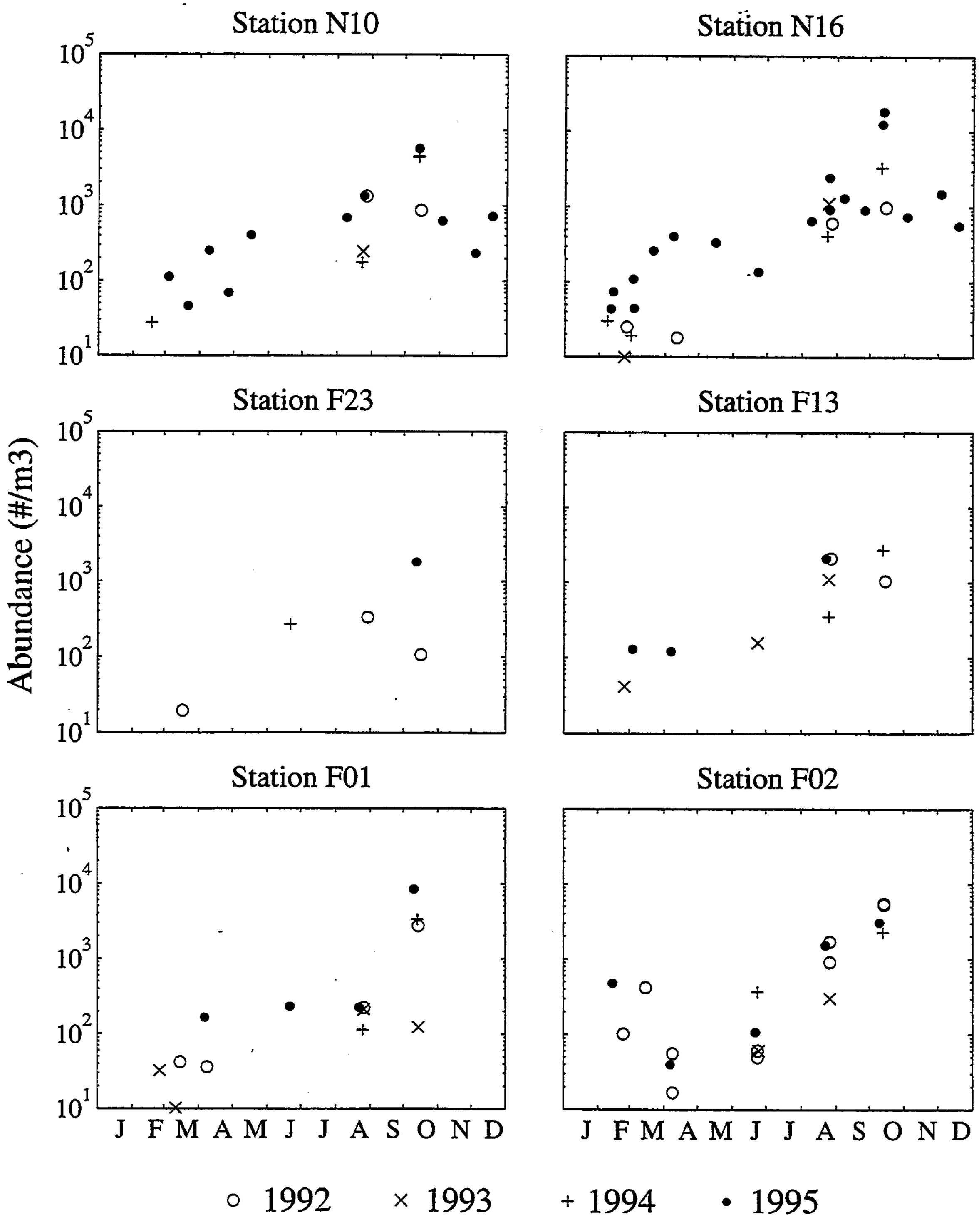


FIGURE 4-41

Centropages hamatus

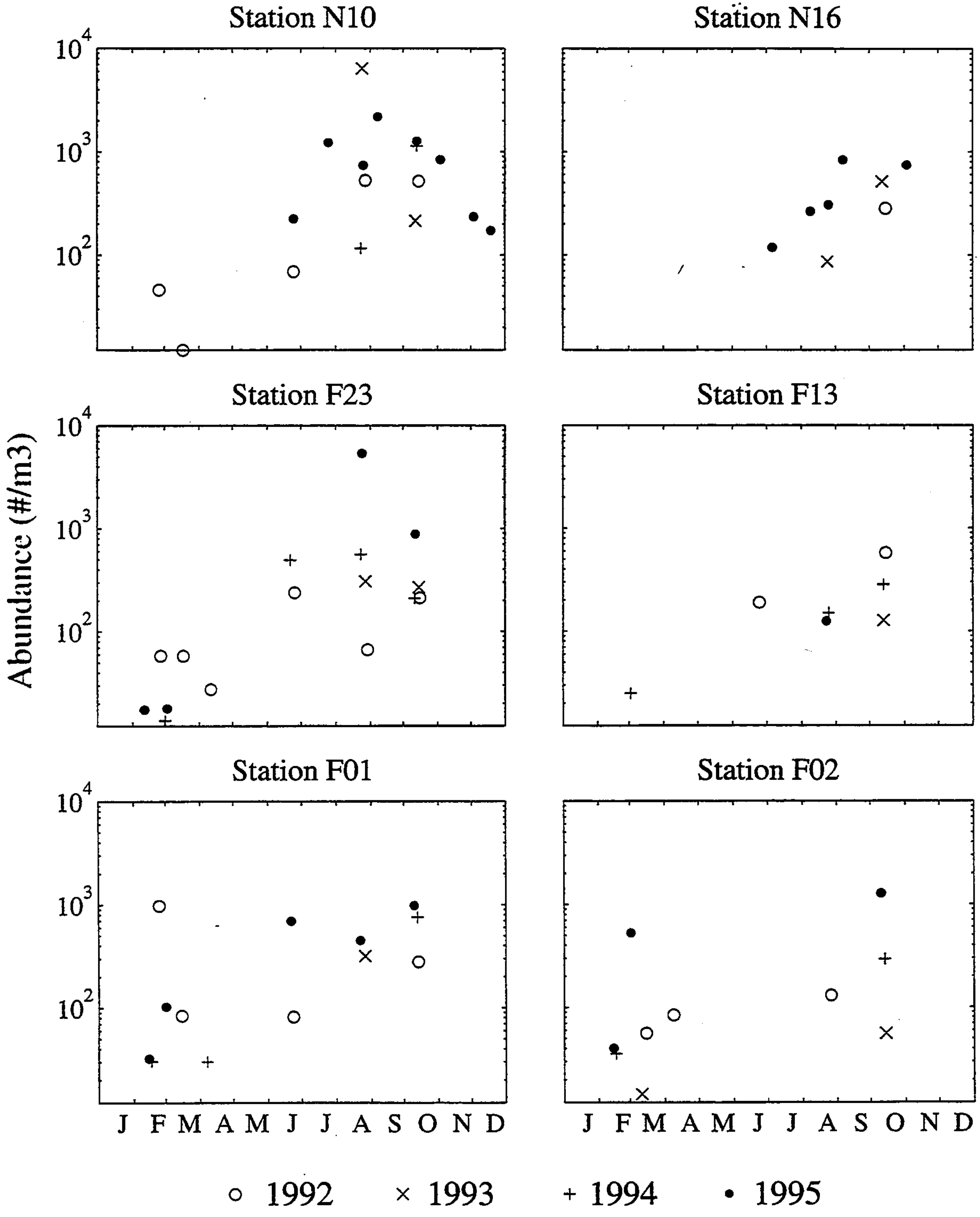


FIGURE 4-42

Centropages spp. Copepodites

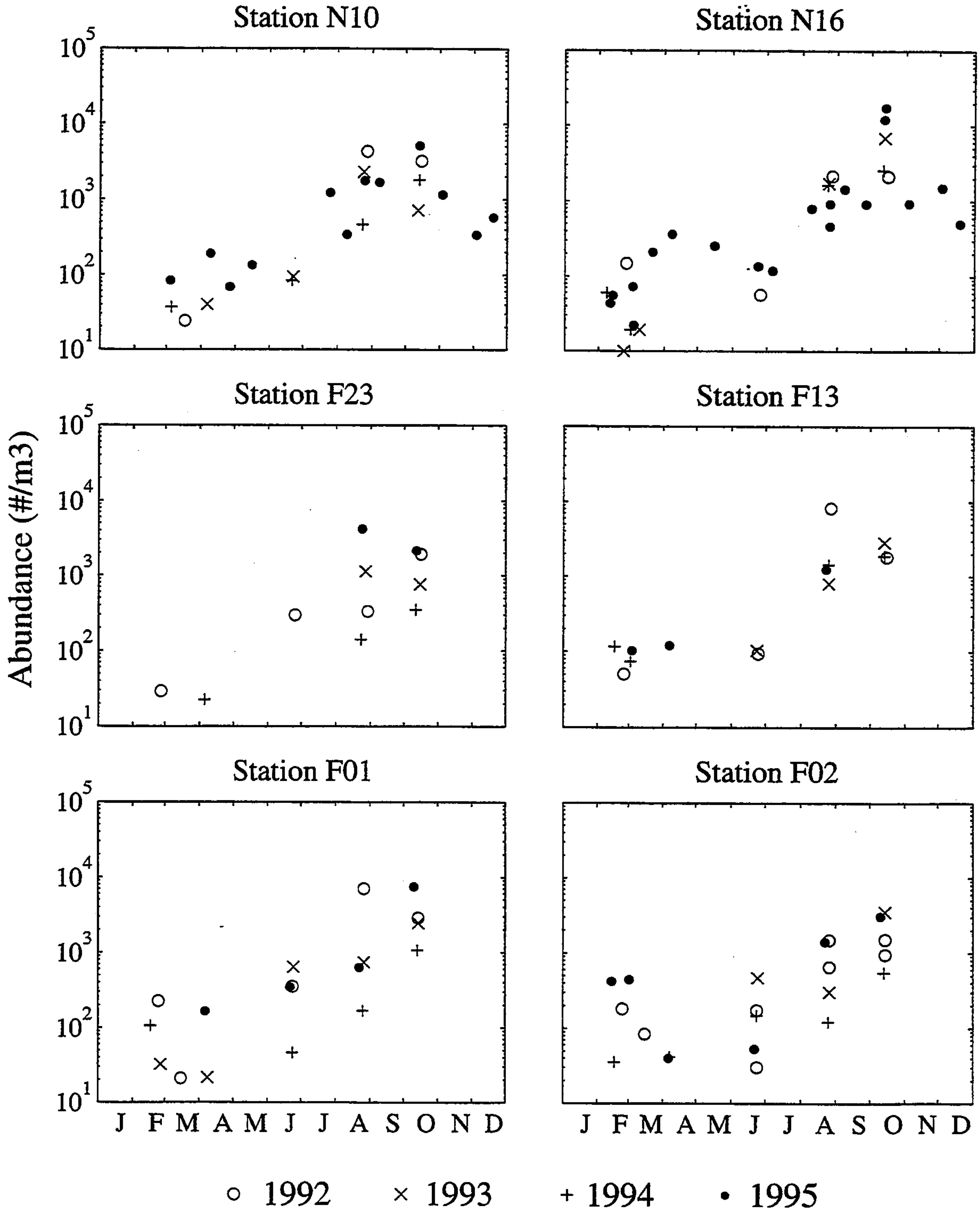


FIGURE 4-43

Temora longicornis

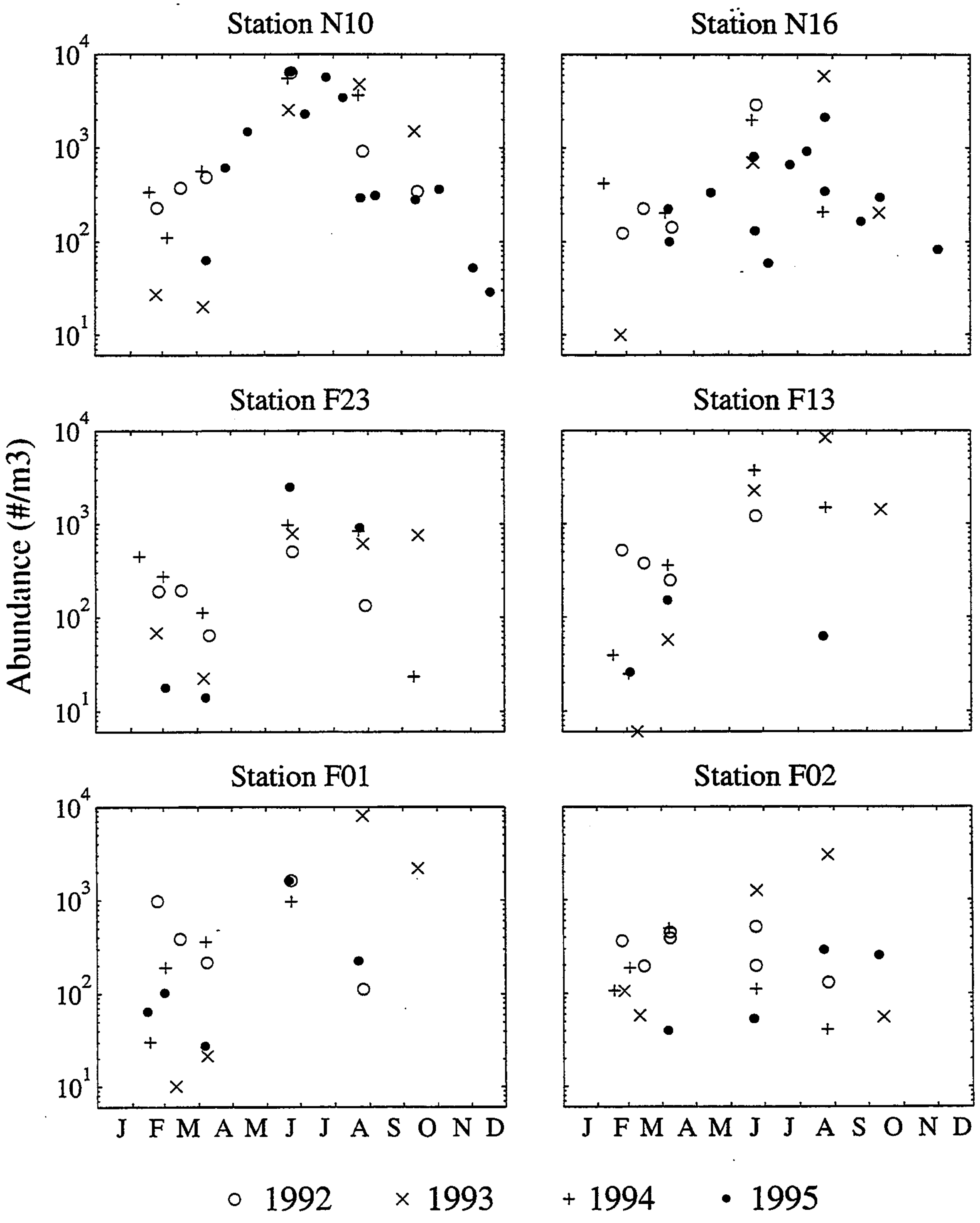


FIGURE 4-44

Eurytemora herdmani

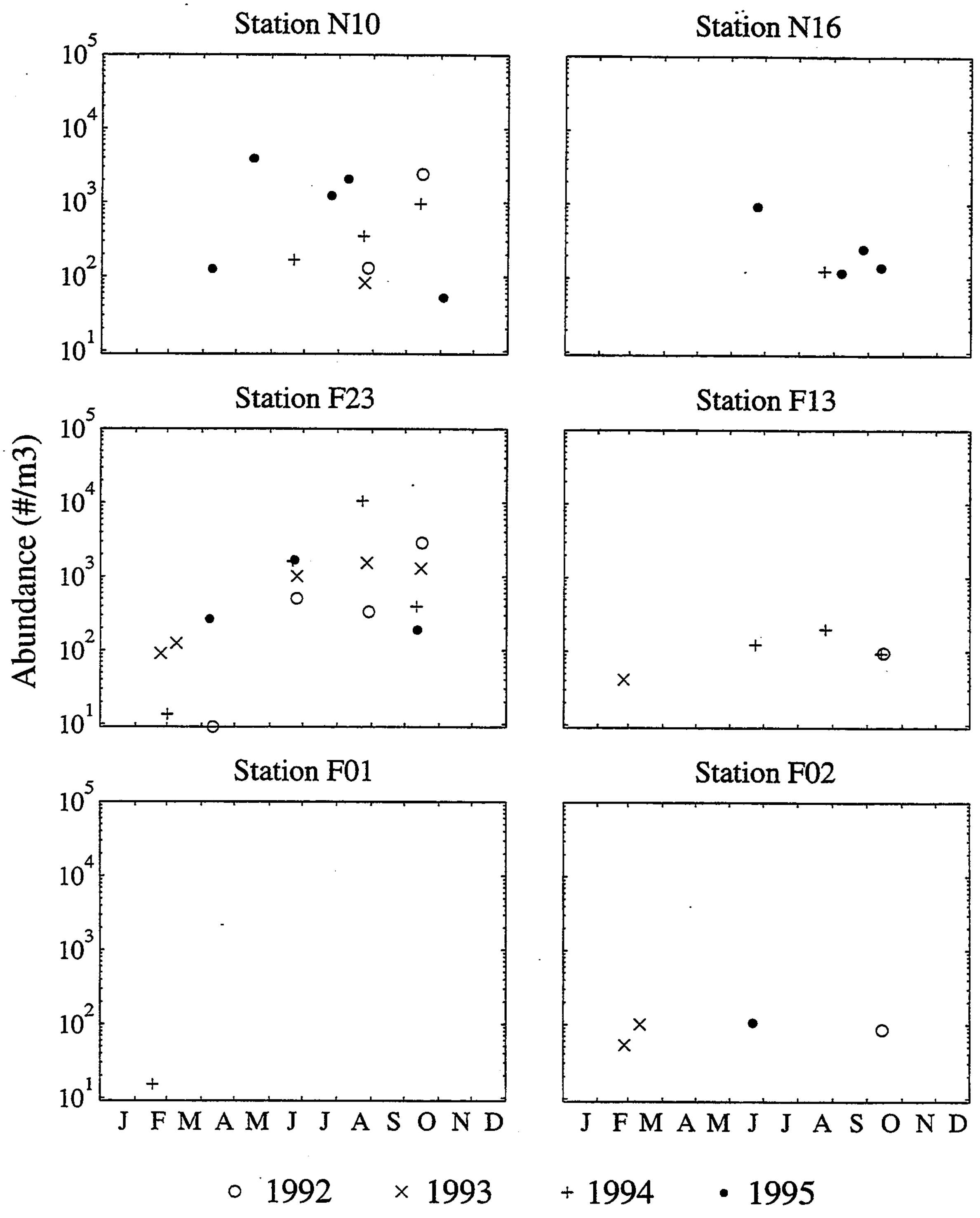


FIGURE 4-45

Acartia spp.

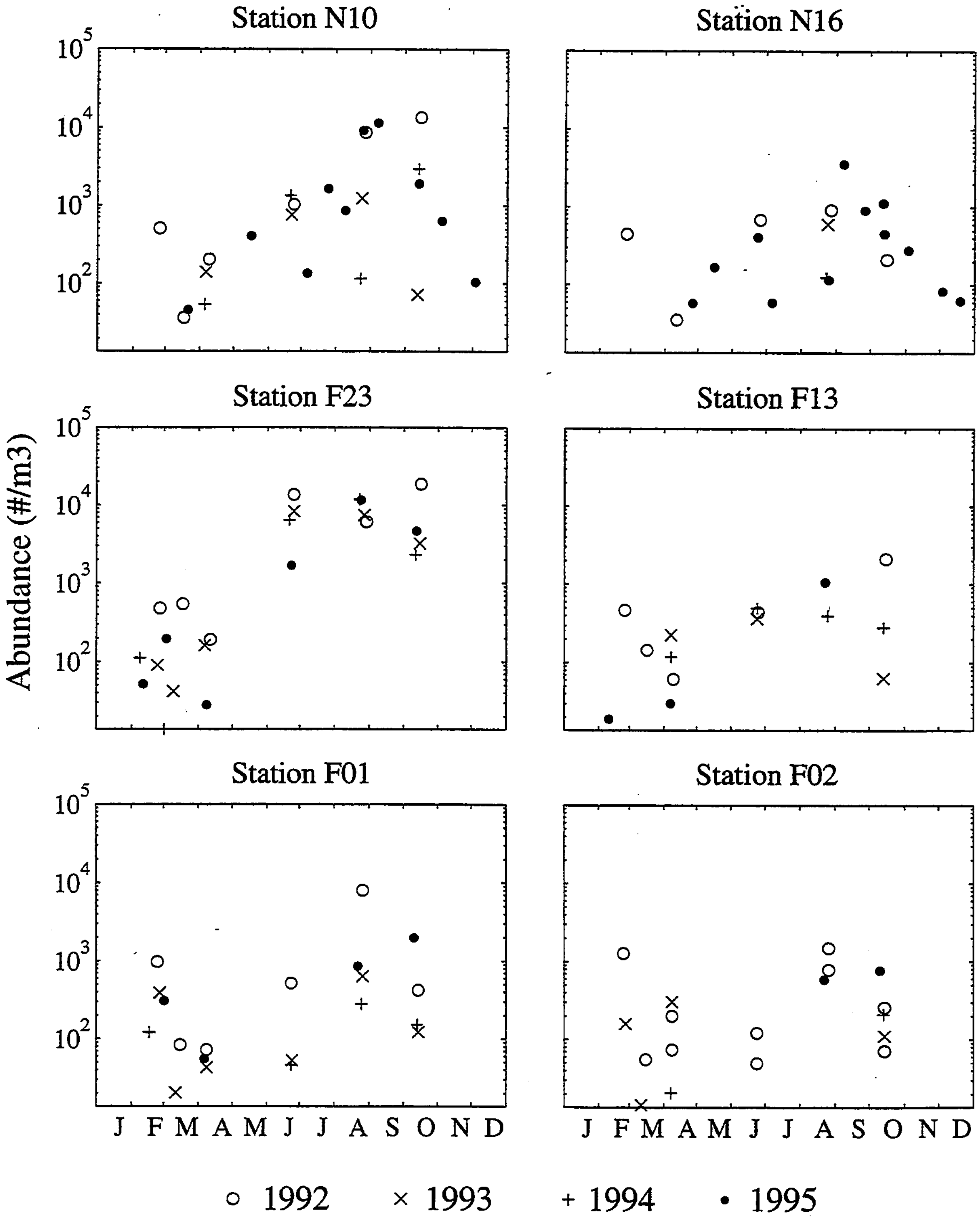


FIGURE 4-46

MARMAP Station Locations 1979–1987

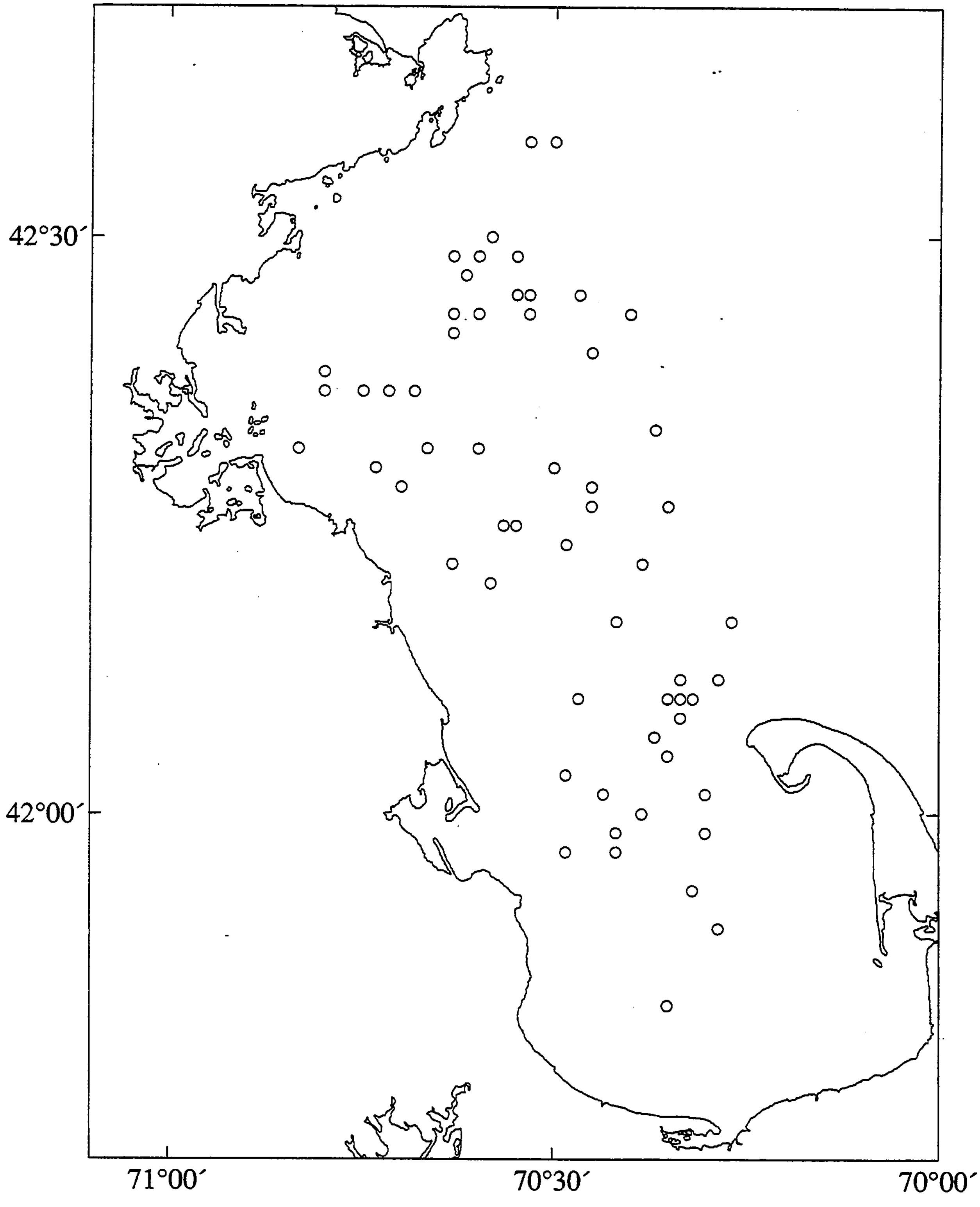


FIGURE 4-47

Calanus finmarchicus Abundance, MARMAP Data 1979–1987

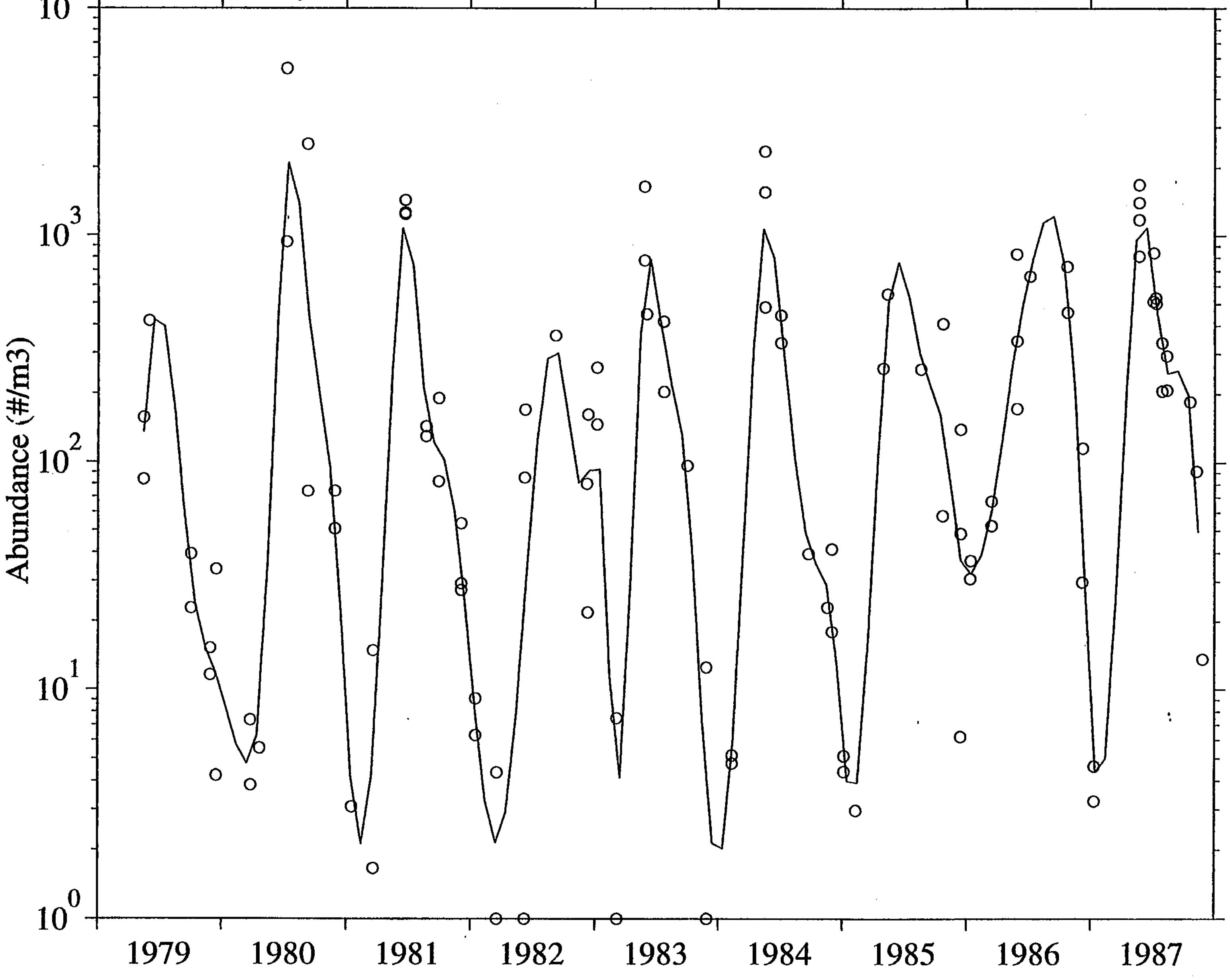
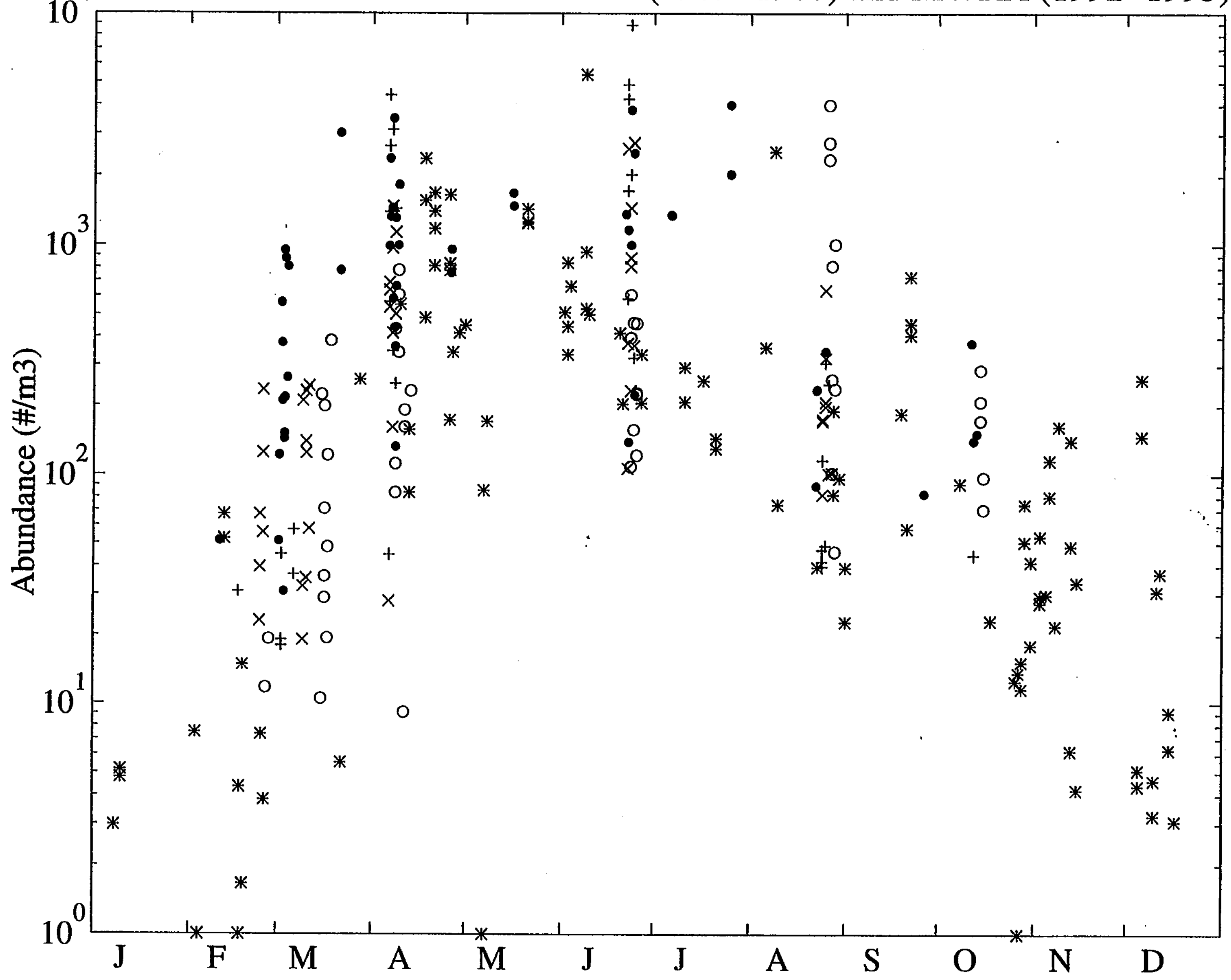


FIGURE 4-48

Calanus Abundance: MARMAP (1979–1987) and MWRA (1992–1995)



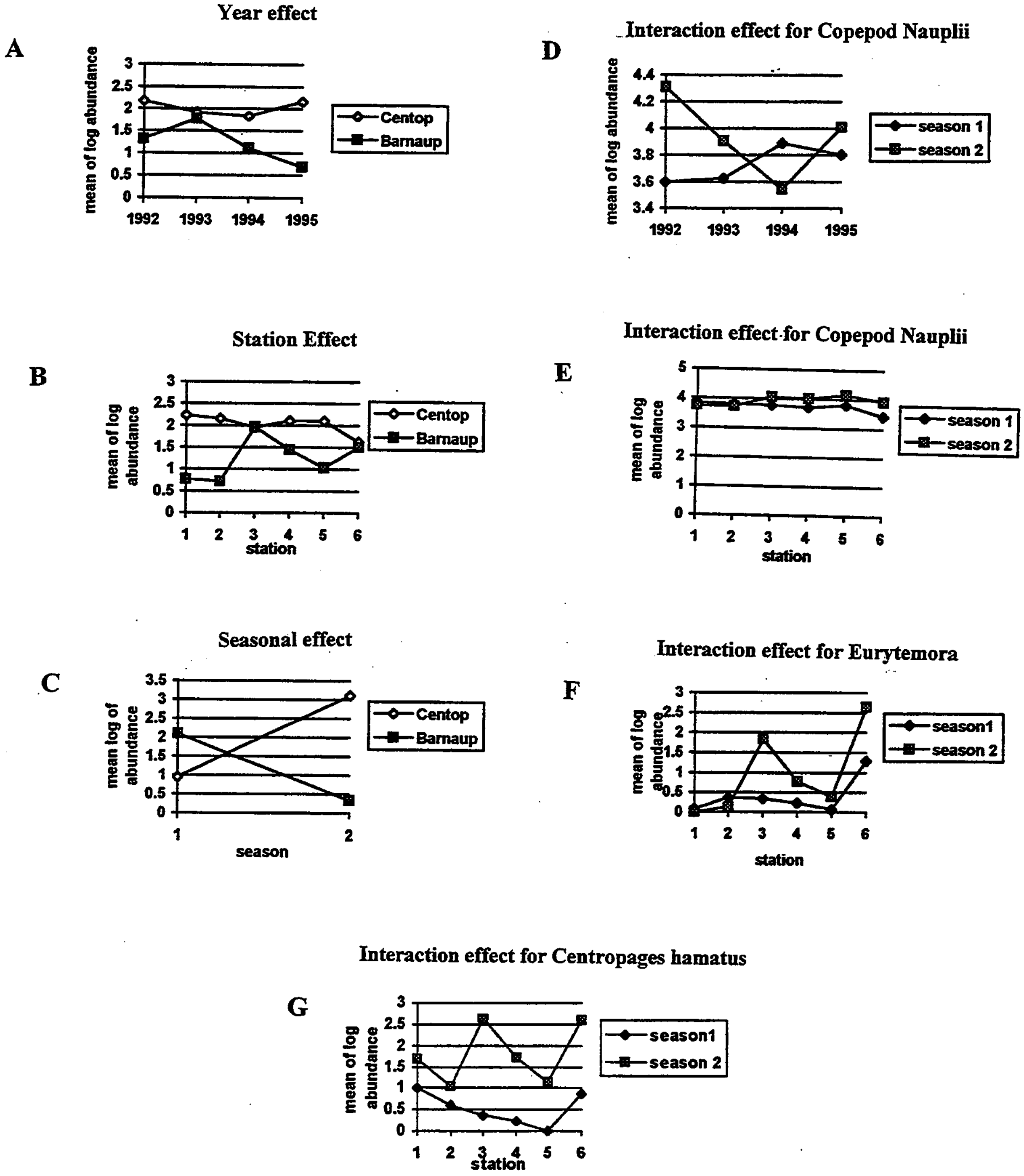
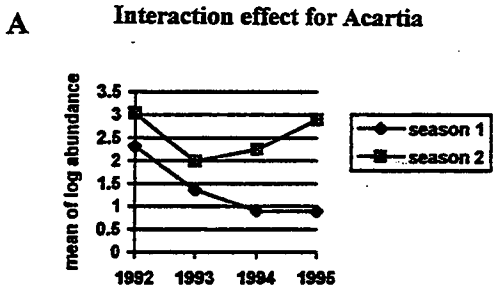
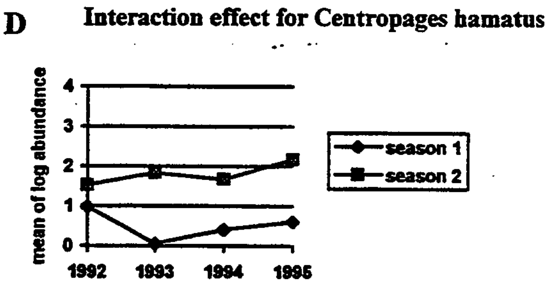
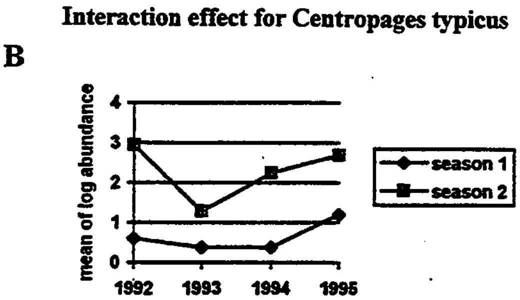
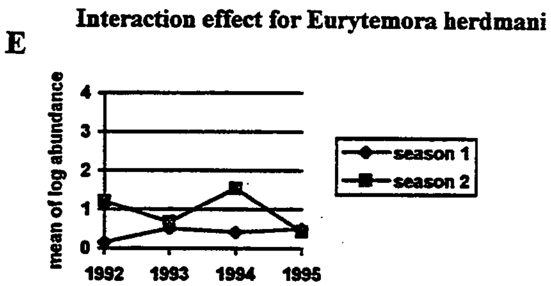


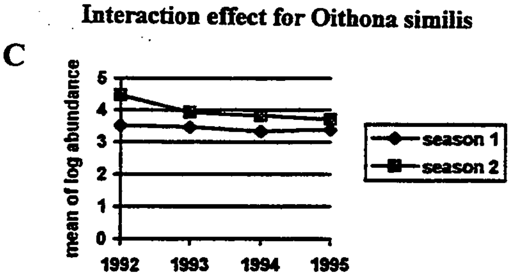
FIGURE 4-49
Output from Three-way ANOVA

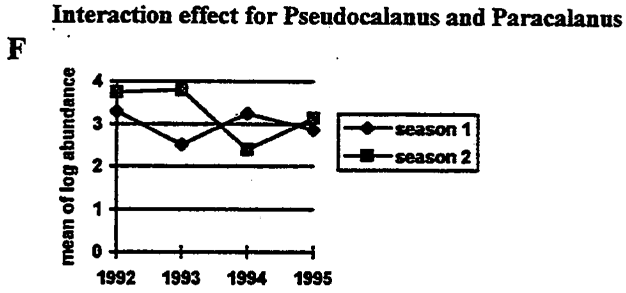


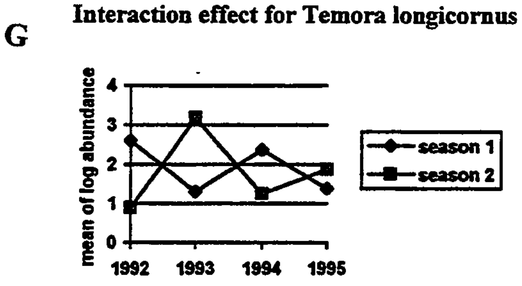












The significance levels for these interactions from top to bottomleft to right are : .0158, .0428, .0280, .0125, .0057, .0000, .0000.

FIGURE 4-50
Output from Three-way ANOVA

5.0 EVIDENCE OF ENVIRONMENTAL CONTROLS ON PLANKTON

5.1 Phytoplankton

As reviewed in Section 3.1, the principal physical and chemical controls on phytoplankton growth are light availability, temperature, turbulence, and nutrient concentrations. In Massachusetts Bay, these are related to season, region, and depth, and are subject to modification by other physical factors such as tides, currents, wind speed and direction, and water column stratification. Grazing (both zooplankton and planktivorous fish) is the principal biological control.

The MWRA data base documented increasing phytoplankton abundance for total densities and several major groups from late winter through summer, most noticeable in the harbor and coastal regions. For many taxa (with the notable exception of microflagellates), Cape Cod Bay often showed reduced densities during the summer period. The general increase in cell densities in the harbor and near-coastal areas throughout the spring and summer may be attributed to the seasonal increases in light availability and temperature, and the ample supply of nutrients.

The potential control of phytoplankton abundance, and to an extent the relative abundance of major taxonomic groups, by nutrients in the nearfield was investigated through a comparison of chlorophyll and phytoplankton results presented in Section 4.1 with seasonal nearfield nutrient concentrations presented in Section 3.1. The use of nearfield data was emphasized due to the nearfield's relatively high sampling frequency. Nutrient ratios were also examined to investigate potential control by the relative concentrations of major nutrients (e.g., N:P and N:Si). Comparisons of the abundance of total zooplankton densities were also made to further interpret phytoplankton standing stock as a resource for zooplankton grazing.

Phytoplankton data from station N16 were used for comparisons with the nutrient results from the more offshore region of the nearfield (stations N04, N07, N16, and N20 as presented in Figures 3-2 to 3-6). High total cell densities in late winter of 1992, 1993, and 1995 (Figure 4-5) appeared to coincide with an early reduction in nutrient concentrations, particularly nitrogen (Figure 3-2 and 3-3). The year with the lowest densities (1994) coincided with comparably high nutrient concentrations during the early part of the year. Lowest silicate concentrations (1992, Figure 3-5) also coincided with the highest diatom densities (Figures 4-8 and 4-9).

Cell densities in April 1992 were an order of magnitude higher than other years (primarily Phaeocystis pouchetii, see Figure 4-31), coinciding with almost complete depletion of nitrogen and, to a lesser extent, phosphate. Nitrogen and phosphate levels stayed low through the onset of stratification in 1992, whereas in 1995 they rebounded until the second bloom in April.

Unlike nitrogen and phosphate, however, silicate concentrations in 1992 rebounded after the first initial depletion to reach seasonal maxima (ca. 9 μM) in April, also evident to a degree in 1995. Coincidentally, these were also the two years of lowest centric diatom abundance in April (Figure 4-8). N:P ratios during April were quite low (<3) in most years (the exception being 1993, when they ranged from 3 to 9 due to abundant ammonium). While nitrogen limitation may be implicated in terminating the 1992 bloom based on this analysis, one must also consider studies which have shown that various micronutrients (e.g., iron or certain vitamins) often play a larger role in nutrient limitation than do macronutrients (Frey and Small, 1979).

Algal activity during the late winter and spring blooms typically results in nitrogen (except perhaps for ammonium) being stripped from the surface mixed layer as stratification begins. Reduced phosphate and silicate concentrations, as well as N:P ratios, are also typically evident. Despite this, both cell densities and estimated phytoplankton carbon biomass can reach annual maxima during the stratified period.

As such, high phytoplankton abundance and low nutrient concentrations during the stratified period indicate rapid uptake of nutrients advected into the nearfield, as well as a high degree of internal nutrient recycling in the upper mixed layer of the water column. In some cases these densities are enhanced by short-term upwelling events similar to that seen in 1995 (Cibik *et al.*, 1997). However, phytoplankton standing stocks during this period seem to sustain annual maxima for zooplankton densities (both total densities and those of several taxa, e.g., *Calanus finmarchicus*) on a recurrent annual basis, suggesting that high phytoplankton densities are fairly persistent.

Each year's data for dissolved inorganic nutrients during the stratified period exhibit a general trend of increasing concentrations in the bottom layer, with the rebound typically beginning around mid-May after the spring bloom is over (e.g., Figure 3-2). This observation can be attributed to remineralization of the spring bloom biomass and possibly to advective transport. While the nutrient concentrations in the bottom layer progressively increase throughout the stratified period, the depth of the pycnocline also increases (from around 5m in June to around 20m in September). Given that the euphotic zone encompasses this depth range, the addition of these nutrients to the surface layer nutrient pool may serve to enhance productivity during the late summer. The larger issue of how these baseline observations during the stratified period, as well as during the fall turnover, might be influenced by enrichment of sub-pycnocline waters by the discharge will be developed further in Section 6.

5.2 Zooplankton

Several specific trends were observed in the environmental data. First, relatively high chlorophyll concentrations were observed during the winter and spring of 1992, and the spring bloom that year included a large bloom of *Phaeocystis* in all regions (Figure 4-30). Second, a freshwater plume was observed in 1993 extending from Cape Ann in April, and subsequently the salinities in 1993 were lower than in other years surveyed (Figure 3-1a). Third, temperatures were lower in 1994 than in other years, and the low winter temperatures coincided with an early diatom bloom. Fourth, temperatures were higher in 1995 than in other years by several degrees. Thus, each of the four years surveyed was characterized by unique environmental features that could impact the zooplankton populations of the bays region.

This interannual environmental variation was accompanied by significant interannual variability in the abundance of dominant zooplankton taxa. Linkages between environmental variation and zooplankton abundance are difficult to determine given that the zooplankton are affected by multiple processes acting simultaneously and in a non-linear manner. Thus, a change in environmental conditions such as temperature can have multiple interacting effects. It is difficult, therefore, to construct scenarios that explain the relationships between observed environmental changes and variations in zooplankton populations.

Zooplankton abundance was generally higher in 1992 than in other years. This observation was true for total zooplankton as well as for copepod nauplii and the four numerically dominant copepod species, *Oithona similis, Pseudocalanus newmani, Paracalanus parvus, Acartia hudsonica*, and *Acartia tonsa*. The main environmental characteristic of 1992 was the relatively high fluorescence in February and the dominance of *Phaeocystis* during the spring bloom at all stations. The region was reported to appear partially eutrophic in 1992 (Kelly *et al.*, 1993). Such conditions would promote zooplankton growth and lead to higher population abundances. This scenario assumes that other factors such as predation and physical exchange remained fairly steady between years, an assumption for which there are little data (predator concentrations and ocean currents are not sampled adequately for such analysis). Moreover, other years had as high or higher fluorescence concentrations, and *Phaeocystis* may not be a particularly good food source for copepods. Thus, there are no obvious explanations for the higher abundance of zooplankton in 1992, based on observed trends in environmental conditions.

In contrast to 1992, the abundances of three species, *Calanus finmarchicus, Centropages typicus*, and *Centropages hamatus*, were highest in 1995. Likewise, the fall abundance of copepod nauplii at the nearshore station was highest in 1995. As discussed above, that year was much warmer than the other years by 2-4°C. Higher temperatures would in fact increase rates of development, growth, and reproduction (assuming food supply was not limiting), but it would also increase feeding rates of predators. It is not clear how these effects would interact in determining zooplankton abundance.

In Cape Cod Bay during spring, abundance of total zooplankton and copepod nauplii was highest in 1994. This year was much colder than the other years and an early diatom bloom was observed in the nearfield region (Kelly and Turner, 1995b). Colder water may have resulted in a population dominated by nauplii, or possibly slowed predation rates, either of which could have produced the observed results. It is unlikely that a large influx of copepods from outside the bay was responsible either, since the Gulf of Maine water is warmer and saltier in spring and was observed to be positioned well offshore.

In sum, although interannual variations were observed in both environmental conditions and zooplankton abundance, the connections between them is not obvious. Given the sparsity of the data and the large number of unmeasured parameters, the relationships are unresolved. It is recommended that population/physical modeling studies should be used to examine the array of potential interactions that could lead to the observed interannual variations.

Although the effect of interannual environmental forcing on zooplankton abundance is difficult to interpret, the trends in region and season are much stronger and are better understood. As discussed previously, the seasonal patterns of the dominant taxa are similar to those found in shelf regions along the eastern U.S. Likewise the nearshore/offshore affinities of the various species are common to other shelf areas.

6.0 CONCEPTUAL MODELS OF POST-DISCHARGE EFFECTS

6.1 Phytoplankton Community

One of the potential adverse impacts of the proposed outfall is that the nutrients in the effluent could alter the abundance and species composition of the phytoplankton community and that these changes could cause other changes at higher levels of the food chain. Cura (1991) summarizes several ways in which increased nutrient loading can affect phytoplankton communities. One is that increased nutrient concentrations will cause a proportional increase in growth rate, although the extent to which this occurs in nature remains unknown. Some (e.g. McCarthy and Goldman 1979) argue that phytoplankton are able to grow at or near maximal growth rates even at limiting nutrient concentrations, such that no change in rates would be expected with nutrient enhancement.

The most likely changes would be in the total biomass of phytoplankton and in the species composition. A change in species composition would result from the emergence of species which are more competitive at higher nutrient concentrations, whereas changes in biomass result from a general enhancement in population growth once the limiting nutrient is increased. There are numerous examples of nearshore systems which have experienced measurable phytoplankton biomass increases and changes in community composition following increased nutrient loading (Smayda 1989; Radach *et al.*, 1990).

As discussed in Section 3.1, phytoplankton growth and population development are affected by a variety of parameters such as temperature, light, salinity, turbulence, and nutrients. Given these multiple factors and the diverse requirements and tolerances of different phytoplankton species, predictions of the nature and magnitude of species or biomass changes within Massachusetts Bay following outfall relocation are simply not possible at this time. The best that can be done is to examine the new outfall in light of the general characteristics of different classes of phytoplankton, or when possible, for individual species of particular interest (e.g., nuisance species) to evaluate possible changes.

Central to this evaluation is an estimate of the extent to which nutrients released at depth will affect phytoplankton throughout the water column at different times of the year. In the winter, deep vertical mixing will take the outfall nutrients and combine them with ambient nutrients throughout the water column. With relatively little nutrient uptake by phytoplankton due to cold temperatures, low light, and deep mixing, the increase in nutrient concentrations in the nearfield prior to the onset of the spring bloom will be a function of the residence time of water in that area at that time. That increase is best estimated using modeling techniques such as the Bay Euthrophication Model (BEM), a linked hydrodynamic/water quality model which has been used to predict the potential effects of the outfall relocation on nutrients, chlorophyll, and dissolved oxygen. Model simulations of the outfall relocation performed to date have shown only a localized increase (around 30%) for dissolved inorganic nitrogen in the nearfield, and have illustrated the substantial reduction in nutrients expected in the harbor (HydroQual and Normandeau, 1995).

The increase in water column nutrients from the winter loadings may affect the magnitude of the bloom, and perhaps its species composition as well. The onset of the bloom is not likely to be affected by the increased nutrients, since it is most dependant on other factors such as the depth of mixing, the temperature, and the ambient light levels (Sverdrup, 1953). The most probable effect for that time of the year would be a change in algal biomass, as occurs in laboratory chemostats and mesocosm experiments when nutrient concentrations are increased. For a given growth (= dilution) rate, more biomass is produced when the nutrient concentration in the experimental enclosures is increased. However, BEM runs have indicated only a modest localized increase in chlorophyll biomass (HydroQual and Normandeau, 1995).

Some have argued that a continuous resupply of nutrients during the spring bloom may prolong that event, but changes in bloom duration may not occur every year. This is because the end of the bloom is sometimes determined by the "capping" of the deep nutrient layer by the onset of stratification, which will shut off the supply of both "natural" nutrients and those associated with the outfall, given that the diffusers have been designed to keep effluent below the pycnocline. In years when the bloom terminates because of the complete utilization of nutrients throughout the water column by the phytoplankton, the continuous supply of nutrients from the outfall should result in an increase in the duration of the bloom. Once stratification sets up, the gradient driving diffusive transport across the pycnocline will be stronger due to the elevated nutrient concentrations at depth. However, the nutrients crossing the interface will be small given the very slow rate of vertical diffusion (Geyer, unpublished data), and certainly small relative to the nutrient pool being rapidly recycled in the surface layer.

During the summer, potential phytoplankton production in the surface layer is limited because of the stratification and the removal of nutrients by the spring bloom. The outfall relocation will also eliminate a large fraction of the horizontal surface flux from the harbor, but will result in increased nutrient concentrations in bottom waters. As discussed above, there will be more nutrients diffusing across the pycnocline due to the stronger gradient from the elevated nutrients at depth. This again will be reflected in an increase in phytoplankton biomass, but may not offset the reduction in horizontal surface nutrient flux from the harbor. The BEM indicated a substantial decrease in surface layer chlorophyll biomass, but only negligible changes at mid-depth (HydroQual and Normandeau, 1995).

The second type of change will be seen when episodic upwelling events bring nutrients from the bottom layer up to the surface in the nearfield. Geyer (1992) argues that favorable conditions for upwelling occur about 50% of the time in the nearfield region during the summer months, but only several times per year are the winds strong enough for long enough to bring water from mid-thermocline depths (the upper limit of the effluent plume from the outfall) to the water surface. Since the bottom concentrations will be higher than they are at present, the biomass of the summer "bloomlets" resulting from those upwellings may be proportionally higher. The BEM simulations to date do not indicate increased biomass during such events (HydroQual and Normandeau, 1995), however, the model is not sensitive to short-term events of this nature (Fitzpatrick, pers. comm.).

The third potential change is at depth. Smayda (1992) has argued that since the euphotic zone sometimes extends to the bottom after the spring bloom has cleared the water, the outfall nutrients trapped below the pycnocline will be available for uptake and growth by the phytoplankton there. While the scenario of abundant growth at depth is plausible, it seems unlikely since productivity measurements during MWRA monitoring have consistently been low in deep layers when light penetration was high, despite the already elevated nutrient concentrations in those waters.

Changes in summer phytoplankton dynamics due to outfall operation are thus likely to include possible decreases in the surface layer cell abundance (due to the reduced horizontal influx of surface nutrients from the harbor), increases in algal biomass at and below the pycnocline, and higher biomass in the episodic, upwelling-driven "bloomlets". Since the nutrient alterations driving these changes are expected to be relatively small, it is difficult to speculate on whether they would also cause changes in species composition or result in significantly increased respiration and DO depression.

The fall bloom would also be likely to change in the nearfield, as once again, the breakdown of stratification would bring nutrient-enriched bottom waters to the surface. Since seasonally diminishing light availability and increasingly colder temperatures can be expected to limit the duration of the fall bloom, the effect of the release of outfall nutrients will likely be increased biomass during the normal bloom period (in proportion to the increase in nutrient concentrations), but not prolongation of the bloom. The BEM did not predict increased biomass during this period, rather, it suggested a modest decrease in both surface and mid-layer chlorophyll biomass (HydroQual and Normandeau, 1995).

Given the seasonal implications on eutrophic response by the phytoplankton community, which are largely controlled by the physical structure of the water column, MWRA has developed both annual and seasonal nearfield chlorophyll thresholds for contingency planning purposes (MWRA, 1997). The latter are based on the seasonal distribution of data from the multiple years comprising the baseline monitoring period and thus will continue to be revised with new data until the new outfall is commissioned in late 1998. A threshold is defined in this case as the level above which an event can be considered to be appreciably different from the baseline. If the seasonally averaged chlorophyll for one year (Table 6-1) exceeds the threshold established in the baseline monitoring, the change may be attributed to the outfall effluent (assuming that the baseline data sufficiently capture the system's natural variability). The seasons were defined based on the annual stratification cycle (Section 3.3), and are defined as:

- Spring - January through April
- Summer - May through August

- Fall - September through December

The seasonal thresholds utilize data from the 17 nearfield stations occupied during 17 surveys per year, and consist of in vivo sensor data calibrated with discrete chlorophyll-*a* analyses, and include all five depths sampled at each station. These thresholds comprise the 95th percentile of the distribution of seasonal average concentrations, calculated as the mean of the seasonal averages + 1.645 x the standard deviation (Table 6-1).

The 1992-1995 baseline results for spring and summer are essentially identical at concentrations of 2.38 and 2.37 μg/L, respectively. The threshold concentration for the fall season is 4.56 μg/L. Results from monitoring of plankton abundance and speciation will thus be used to interpret appreciable departures from these seasonal concentrations. Alternatively, plankton results will also be used to monitor for appreciable changes in abundance or species distribution which may not result in appreciable alterations in chlorophyll concentration.

The foregoing offers a view on the general phytoplankton community's response to the outfall nutrients. Another issue is the effect those nutrients will have on nuisance algae. As discussed earlier, one of the manifestations of increased nutrient loading and eutrophication of coastal waters is that algal biomass (the total abundance of all algal species) increases. This can be associated with an increase in the frequency and magnitude of red tides, as has occurred in polluted regions such as the industrialized areas of Japan, Korea, and other countries where water quality restrictions lagged industrial development (Anderson, 1989; Smayda, 1989). In the context of Massachusetts Bay, the species of concern are *Alexandrium tamarense, Phaeocystis pouchetii*, and *Pseudo-nitzschia multiseries*. Other harmful species may appear, but those listed are the ones that already occur in these waters.

There are several ways in which outfall nutrients can affect these organisms. When stratification is not present in the late fall and winter, outfall nutrients will be mixed to the surface, but that is the time when *A. tamarense* red tides do not occur in this region. Seasonal blooms of *Phaeocystis* and *Pseudo-nitzschia* do occur then, however (Figures 4-25, 4-26 and 4-31), and there is thus the possibility that growth stimulation of those species might occur.

In other parts of the world, especially western Europe, *Phaeocystis* blooms are associated with cultural eutrophication (Smayda 1989), since this species' large size and colonial morphology requires high nutrient concentrations. Increased abundance of *Phaeocystis* colonies might thus be an indicator of increased nutrient levels from the MWRA outfall. With elevated levels of nutrients following outfall discharges and deep mixing, this species might well find the high nutrient conditions suitable for blooming. The pattern has been one bloom every other year since 1992, so an increase in this frequency would be suggestive of a fundamental change in nutrient loadings.

Toxic *Pseudo-nitzschia* species often bloom in the late fall (Bates, 1989; Smith, 1990), thus release of elevated bottom water nutrients with the fall turnover could potentially enhance their abundance. These species also proliferate throughout the winter, so increased outfall nutrients in surface waters prior to the onset of stratification might result in enhanced abundance of toxic diatoms. There is no way at present to predict whether *Pseudo-nitzschia* species would be able to out-compete the typical members of the spring bloom diatom community such as *Thalassiosira* or *Chaetoceros* species.

During stratified periods in the late spring and summer, outfall nutrients can reach surface waters in the nearfield as a result of episodic upwelling. The extent to which this will affect nuisance species cannot be estimated without more understanding of the frequency and nature of the upwelling events, the spatial distribution of the nuisance species at the time of upwelling, and the response of those species to nutrient pulses.

For *Alexandrium*, past surveys indicate that in May and June, toxic cells are carried into Massachusetts Bay by the coastal current formed by the outflow of rivers to the north (Franks and Anderson 1992). Most often, this current passes to the east of the outfall site and proceeds either directly to the South Shore, or bypasses the bay entirely and travels seaward of Stellwagen Bank. On occasion, however, the cells and the plume will pass near enough to the outfall site to enter the region to the west and north of the outfall where exposure to upwelled nutrients may occur. However, since shellfish toxicity is extremely rare along the shore immediately to the north of the outfall site (one closure in over 20 years of monitoring), *A. tamarense* will have few opportunities to respond to the episodic pulses.

Under those same stratified conditions, outfall nutrients at depth would be available to species capable of migrating to deep layers or of accumulating at the pycnocline. *Alexandrium tamarense* is an example of the former, and *Pseudo-nitzschia* of the latter. Past studies (Townsend *et al.*, 1991; Section 3.3 of this report) indicate that when *A. tamarense* enters the bay in May and June, nutrients in the surface layers are very low. However, *A. tamarense* cells have an advantage over many other species of algae because they can swim vertically at a rate of up to 1 meter per hour. They are thus able to swim down to the lower, nutrient-rich water to obtain nutrients and then back to the surface to obtain light for photosynthesis, all in a 24 hour period.

The ability to vertically migrate has long been considered an important characteristic of red tide species. The outfall diffusers are designed so that the effluent will rise to, but not penetrate the upper layer of the stratified water column, and diffusion from the lower layer is likely to be small (Geyer, 1992). Nevertheless, the bulk of the nutrients in the effluent will be accessible to the toxic *A. tamarense* cells because they vertically migrate. The question becomes whether nutrients added by the outfall will stimulate *A. tamarense* in any way.

Estimates are that the outfall will increase dissolved inorganic nitrogen (nitrate + nitrite + ammonium) concentrations in the deeper water by 3 to 4 μM or less within the 85 km^2 zone where the effluent will be diluted by a factor of 400. Townsend *et al.* (1991) report total dissolved inorganic nitrogen concentrations of 3 to 11 μM or higher in the water below the thermocline during June. MWRA measurements show average DIN concentrations of 2 to 7 μM in May and June (Figure 4-3). In phytoplankton, the uptake rate of nutrients is a hyperbolic function of nutrient concentration. This means that uptake rate increases linearly with the amount of available nutrient up to a certain concentration, but the rate quickly reaches a maximum value (saturates) and does not increase substantially if more nutrient is made available.

For *A. tamarense*, the linear portion of the uptake curve ends at about 2-3 μM nitrate, and concentrations higher than this have little effect on nutrient uptake rate (MacIsaac *et al.*, 1979). More comprehensive experiments are clearly needed, given the importance of this issue, but the indication is nevertheless that the nutrient uptake kinetics of *A. tamarense* are such that nutrients are already sufficiently high in deeper waters near the outfall site relative to its uptake capability that the addition of effluent nutrients is not likely to have an impact. In other words, there may be sufficient nutrients there already to saturate the uptake requirements of this species, so adding more will not likely result in faster growth or a prolonged bloom.

Since the area of increased nutrients is relatively small due to dilution, the likelihood of an impact also depends on having the cells transported to that spot, which is not a common occurrence based on our knowledge of the movement of the coastal current and past shellfish toxicity records in the area. Enhancement of *A. tamarense* cell concentrations by outfall nutrients thus seems unlikely, but should be recognized as an issue worth monitoring carefully once the effluent discharges begin.

6.2 Zooplankton Community

The harbor and inner nearfield stations are dominated by nearshore species including *Acartia tonsa, Acartia hudsonica, Eurytemora herdmani*, and *Centropages hamatus*. The abundance of *Acartia* spp. falls off markedly away from the mouth of Boston Harbor and has a strong gradient across the nearfield region. Since *Acartia* spp. are known to require very high concentrations of food for maximal growth and egg production, the observed nearshore affinity for this species must be due in large part to food limitation. Thus, potential increases in phytoplankton densities due to the outfall may cause a shift in the distribution of this species further offshore.

Moreover, were eutrophication to increase in Massachusetts Bay on a regional scale due to the outfall relocation, one result would probably be increased abundance and dominance of *Acartia* spp., supplanting the offshore assemblage which currently dominates the region. A substantial increase in *Acartia* spp. abundance could serve as an index of eutrophication of the nearfield region. If the grazing pressure by *Acartia* spp. and microzooplankton (eg. protozoans) is strong, increases in the abundance of *Acartia* spp.

may serve as an important eutrophication index, since chlorophyll may be removed quickly from the water column.

The hypothesis that *Acartia* spp. are restricted to nearshore waters due to food-limitation is based on several previous studies of this genus. First, Paffenhofer and Stearns' 1988 paper entitled "Why is *Acartia tonsa* restricted to nearshore environments?" concluded that food-limitation is the main reason. Egg production and growth of *A. tonsa* are known to be limited at very high food levels (14 μg chl*a*/L or 1600 μgC/L, Durbin *et al.*, 1983). *A. hudsonica* is limited at lower food levels (250 μgC/L) but still is food limited in late spring in Narragansett Bay (Durbin *et al.*, 1992; Durbin and Durbin 1992). In February, *A. hudsonica* is not food limited in Narragansett Bay, but the food concentrations there (11-20 μg chl*a*/L) are well above those found in Massachusetts Bay at that time (1-2 μg chla/L).

Likewise, Peterson and Belatoni (1987) found increased egg production of *A. tonsa* above 4 μg chl/L. Fulton (1984) found that *Acartia tonsa* abundance increased with nutrient enrichment while *Paracalanus crassirostris* abundance decreased. Sullivan and Banzon (1990) found that *A. hudsonica* had higher egg production and densities in nutrient enriched mesocosms. Toner (1981) found a positive correlation between *A. hudsonica* abundance and phytoplankton abundance as influenced by nutrient concentrations in Mt. Hope Bay, Mass.

There also is direct evidence of *Acartia* dominance as a result of sewage pollution. Champalbert and Patriti (1982) and Arfi *et al.* (1981) found distinct shifts in zooplankton community composition along a transect through the domestic sewage discharge of Marseilles. Near the discharge they found a few "pollution tolerant" species including *Acartia clausi (hudsonica)*, while further away from the discharge they found *Paracalanus parvus* and *Centropages typicus* to be dominant. This situation is very similar to that existing in Boston Harbor and Massachusetts Bay.

Interestingly, there is a study which parallels the Boston Harbor one near Athens, Greece, in which Elefsis Bay is currently eutrophic due to sewage discharge and *Acartia* is dominant (Theodorou *et. al.*, 1994). Apparently the Athens sea outfall became operational in spring 1992 and has been pumping the effluent offshore since that time. It will be informative to follow the post-discharge changes in zooplankton community composition in that region in relation to other aspects of the environment.

Alternative factors which could contribute *Acartia's* nearshore affinity include limited salinity tolerances (Tester and Turner, 1991), omnivorous feeding characteristics (Johnsson and Tiselius, 1990), and predation. In addition to enhancing phytoplankton biomass, the outfall would create slightly less saline water (a reduction of 0.2 PSU over a few square miles) in the nearfield region, thus promoting *Acartia* growth. It grows in Narragansett Bay at about 30.0 psu and is routinely cultured at that salinity (R. Campbell, URI, pers. comm.). Since salinity in the Bays region (30-32 psu) is slightly above this value, a small reduction in salinity could conceivably promote *Acartia* growth, altering its distributional pattern.

Such a salinity dependence, however, was not evident in the 1992-1995 MWRA data. Highest abundance of *Acartia* in the offshore regions was found in 1992, whereas 1993 was the year with the lowest salinity (<30.8 psu everywhere in June). In February of 1992, highest concentrations of *Acartia* were found in Cape Cod Bay where salinities were near 32 psu. It appears therefore that *Acartia* is capable of growing quite well at the slightly higher salinities of the Bays region.

The idea that *Acartia* feeds heavily on protozoans and is restricted to nearshore environments because of higher protozoan densities (Johnsson and Tiselius, 1990), actually supports the present food-limitation hypothesis. Since *Acartia* is likely food-limited offshore, and it is known to grow and reproduce maximally on a diet of phytoplankton alone, then the increased offshore food supply from the outfall, whether it enhances protozoan growth or just phytoplankton growth, will lead to an richer food environment for *Acartia*. Such enhancement will promote its growth and shift its distribution further offshore.

It is unlikely that predation plays a role in the nearshore restriction of *Acartia*, since other copepods of the same body size (and vulnerability to predation) are dominant offshore. *Pseudocalanus, Paracalanus*, and *Centropages typicus* all have similar body sizes to *Acartia* spp., but all are more abundant offshore than nearshore. The distribution of planktivorous fish and invertebrates in Massachusetts Bay is not being measured by the monitoring program, and the relative abundance of these predators inshore versus offshore cannot easily be determined. It seems likely that predation pressure on *Acartia*-sized copepods would actually be higher in inshore regions, which serve as nursery grounds for many larval and juvenile stages that may prey on the copepods.

Nonetheless, higher abundance of the other *Acartia*-sized copepods offshore indicates that predation pressure is not a contributing factor. Thus, although many species of copepods can be regulated by predation (eg. Davis, 1984a,b), it appears that *Acartia's* nearshore affinity is controlled by food limitation. This hypothesis represents a case of bottom-up as opposed to top-down population control.

In sum, there is considerable evidence in the literature that *Acartia* spp. are food-limited in nature and thus are restricted to eutrophic nearshore environments. It is reasonable to hypothesize then, that were the discharge of effluent in the nearfield region to cause local eutrophication then it would be manifest as a shift in the zooplankton species composition towards an *Acartia* spp. dominated system.

Table 6-1

Mean MWRA Baseline Seasonal Chlorophyll Data

	Spring	Summer	Fall
1992	1.97	1.88	2.46
1993	1.01	1.81	4.42
1994	1.95	1.55	2.47
1995	1.02	0.73	2.58
Mean	1.49	1.49	2.98
StDev	0.54	0.53	0.96
N	4	4	4
Threshold Concentration	2.38	2.37	4.56

Threshold Concentration = Mean + (1.645)STDEV

7.0 CONCLUSIONS AND RECOMMENDATIONS

7.1 Phytoplankton

It will be extremely difficult to detect appreciable changes in the Massachusetts Bay phytoplankton community once the outfall begins operation. This is because of the high interannual and spatial variability in species abundance, and to a lesser degree chlorophyll, and the relatively short interval of baseline monitoring relative to the observed variability. For example, total or group abundances in a given sampling month can easily vary by an order of magnitude (e.g., Figures 4-5 and 4-8), while densities of dominant species can vary by as much as three orders of magnitude (e.g., Figure 4-24). Annual means of chlorophyll vary less but tell us little, and seasonal or monthly averages can vary several-fold.

The simplest and most robust indicator of changes in the phytoplankton biomass would be chlorophyll, but changes will only be meaningful if those levels are tabulated and compared using appropriate spatial and temporal scales. The calculation of seasonal thresholds (discussed in Section 6) is an appropriate step given the current understanding of the system's dynamics, although the seasonal timeframes may require reconsideration as more baseline data become available. We do recommend that careful consideration be given to the means for detecting change in the chlorophyll levels being measured by the monitoring program. Analysis of the data by a statistician is needed to determine if the capability exists to detect changes specified as caution and warning levels in the Contingency Plan (MWRA, 1997).

Given the potential for response to enrichment during episodic events such as upwelling or mixing, long-term/high frequency monitoring of in situ chlorophyll concentrations using moored sensors would provide additional data suitable for statistical comparisons. The summer "bloomlets" can thus be fully examined, as increased nutrient availability in the surface layer from upwelling or vertical mixing will likely enhance and perhaps prolong these short-lived summer bloom events. As this is also the period of maximum zooplankton abundance, surface layer primary production may well be transferred up the food web and be evidenced in changes there. Likewise, chlorophyll maxima and the durations of the spring and fall blooms can be better documented.

Changes in species composition during the spring, summer and fall will be even harder to document, as they are likely to be obscured by interannual variability that has already been documented by the baseline monitoring. When comparisons are made, it is wise to keep in mind the conclusion of Margalef and others that it will be possible to detect and explain meaningful changes in the dominance of different classes of phytoplankton, but not changes in individual species. To this end, we recommend that phytoplankton data be combined into appropriate classes before interannual comparisons are made. These groups include microflagellates, centric diatoms, pennate diatoms, cryptophytes, dinoflagellates, chlorophytes, chrysophytes, cyanophytes, euglenophytes, and haptophytes. Species-specific evaluations should also be undertaken, but only large-scale shifts in dominance or abundance will be discernible. In the future, it might also be useful to use HPLC to measure the relative abundance of different phytoplankton pigment classes, as this can provide a direct, integrated measure of the classes that are present.

Of greater uncertainty is the potential stimulation of phytoplankton production when light penetrates beneath the pycnocline. Increased nutrient availability in the bottom waters could lead to increased production and respiration of phytoplankton carbon, adding to the effluent-derived carbon trapped below the pycnocline. Increased phytoplankton production below the pycnocline may be unavailable for grazing by zooplankton which remain in the surface layer. MWRA data suggest that there is sufficient light penetration in certain months to permit this bottom layer production, that nutrients are high there, and that phytoplankton are present. Productivity measurements have yielded only marginal rates, however (Cibik *et al.* 1997), so it is not clear how serious this concern is. Nevertheless, given the potential for enhanced production in the bottom layer once the outfall is in operation and the significant impact on bottom oxygen levels if this occurs, we recommend that careful attention be given to the nearfield zone in this regard in future monitoring. A good dataset on productivity and chlorophyll exists for that layer in the nearfield, so it should be possible to determine if these parameters increase.

Nuisance species bloom sporadically and often vary substantially in abundance among years (e.g. Fig. 4-31), so for several Massachusetts Bay species, it will be very hard to link future fluctuations in abundance to outfall loadings. *Phaeocystis pouchetii* bloomed extensively in 1992 and 1994 (and again in 1996), but was either not observed at all or was in low concentrations in 1993 and 1995. In other parts of the world, especially western Europe, *Phaeocystis* blooms are associated with cultural eutrophication, since this species' colonial morphology requires high nutrient concentrations. Free-living *Phaeocystis* cells are associated with low nutrient waters. Increased abundance of *Phaeocystis* colonies might thus be an indicator of increased nutrient levels from the MWRA outfall.

Toxic and non-toxic *Pseudo-nitzschia* species are so similar morphologically that they are presently being combined in cell counts. It will thus not be possible to resolve whether toxic forms are being selectively stimulated without a change in analytical methodology. Only the general abundance of *P. pungens* and closely related species can thus be compared. There is no routine monitoring in Massachusetts waters for the domoic acid produced by those diatoms, so it will not be possible to obtain an indication of species abundance in that manner.

Given the serious nature of domoic acid poisoning (termed amnesic shellfish poisoning (ASP) in humans), it would be prudent to consider a phased domoic acid monitoring program if future monitoring efforts detect large numbers of *Pseudo-nitzschia*. This phased program could initially entail analyses of duplicate particulate carbon samples already collected from the nearfield for domoic acid if increased densities of *Pseudo-nitzschia* are encountered. Alternatively, antibody or RNA probes can be used to ascertain if toxic species or strains of *Pseudo-nitzschia* are present. Once the presence of toxic forms are confirmed, shellfish testing for domoic acid would be warranted. Here again, the lack of baseline data on the presence of toxic forms makes it impossible to determine if outfall nutrients are responsible for the outbreak, therefore it may be appropriate to perform pilot assays during the fall of 1997.

For *A. tamarense*, it will be possible to detect change, since shellfish toxicity has been monitored weekly at multiple stations within the bay since 1972. A sound statistical treatment of these historical data can provide a baseline against which future levels of toxicity can be compared. We therefore recommend that this retrospective statistical analysis be undertaken in collaboration with personnel from the Massachusetts Division of Marine Fisheries, and that criteria be established which indicate whether toxicity, and thus *A. tamarense* abundance, has exceeded historical averages sufficiently to indicate outfall-related enhancement of abundance.

7.2 Zooplankton

The present regional sampling of zooplankton as part of the monitoring program is limited to discrete sample collection at 12 stations, six times per year. However, the more frequent sampling in the nearfield gives further insight into seasonal succession in the zooplankton population. While this sampling strategy may be adequate for detection of dominant regional and seasonal patterns and large interannual changes in these patterns, the time-space scales sampled are not adequate to ascertain the causal mechanisms relating environmental forcing to fluctuations in zooplankton abundance. Data on the variability of zooplankton and environmental variables over a continuum of scales (from centimeters to tens of kilometers, the latter equivalent to bay-wide scale) is needed to provide a framework within which the coarse scale monitoring data can be interpreted. For example, concentration and composition of zooplankton present in intrusions of Gulf of Maine water caused by wind and river discharge events are not measured by the current monitoring program. Such fine-scale features may be a dominant source of zooplankton to the Bay but will remain undetected unless high resolution surveys are conducted.

Dense patches of copepods in eastern Cape Cod Bay are fed upon by the northern right whale, but the spatial structure of these patches and their relationship to the rest of the Bay remains unknown. Bay-wide high-resolution sampling is needed to tie together the scales of variability and provide an understanding of how prevalent small scale features such as these copepod patches are in the Bay and how they may be affected (if at all) by the outfall. Pilot studies are being conducted to examine the spatial structure of these patches and to obtain bay-wide high-resolution data for zooplankton abundance. Results of these studies will be available in future reports.

High resolution bay-wide towyo-type sampling will provide insights into the dominant scales of variability as well as how the zooplankton species are related spatially to each other and to their associated environmental variables. This information will improve understanding of how changes in environmental conditions affects zooplankton population distributions. Such information is critical for understanding how the outfall affects the zooplankton community in the Bay and what remediation scenarios are likely to work. High resolution surveys also will provide direct documentation of the boundaries of the area impacted by the discharge. It is likely that this area will be limited to the nearfield region, but without high-resolution surveys during the baseline and post-discharge years, the extent of the impact on the Bay's zooplankton community will remain unknown. The current monitoring is too sparse to provide this information, but, as discussed above, it is adequate to determine whether a major change in the zooplankton assemblage (outside the baseline interannual, seasonal, and regional patterns) has occurred.

In addition to the need for data over a range of spatial scales, a historical review and analysis of published and unpublished zooplankton data is needed for the Bay to provide a multi-decadal context within which the MWRA monitoring data can be interpreted. As discussed above, previous data were collected on a number of surveys including Bigelow's surveys from 1912-1926, Continuous Plankton Recorder Surveys from 1961 to present, Plymouth Power Plant surveys, NMFS larval herring surveys, and MARMAP surveys. Other surveys may have been conducted as well. A thorough literature review and possible re-analysis of published and unpublished data is needed. In particular, extensive data collected by the Center for Coastal Studies should be incorporated into the zooplankton data base for the Massachusetts Bay system.

The method of multifactor analysis of variance presented in Section 4 may be adequate for detecting significant changes in the zooplankton community from the baseline survey years. A major shift toward an *Acartia*-dominated community in the outer nearfield (e.g. station N16) or in Cape Cod Bay may be detectable using this method. Such a shift may or may not be associated with shifts in phytoplankton (chlorophyll) or protozoan biomass since grazing by the copepods may dampen or eliminate the enhancement of prey biomass. The proportion of *Acartia* spp. in the samples could be used as an index of eutrophication, with the end-point being a system dominated by *Acartia*, as currently exists at the harbor stations. A consistent dominance by *Acartia* at the offshore part of the nearfield (N16) should signal a warning and trigger contingency planning measures.

Beyond the detection of a major shift in coastal versus offshore assemblages, the current data are likely to be too sparse to provide insights into the role of episodic events and fine scale features that may link the discharge to the zooplankton community. Analysis of high resolution transects, mooring time series (see below), using statistical methods such as scale analysis, spatial autocorrelation, and principal component analysis of zooplankton abundance and associated environmental variables is recommended.

The suggested high resolution towyo sampling should be conducted in conjunction with the current combined nearfield/farfield surveys. An instrument such as the Video Plankton Recorder (VPR, Davis *et al.* 1992, 1996; Gallager *et al.*, 1996) towyoed between standard stations on these surveys would allow immediate visualization of the plankton species distributions in relation to environmental variables throughout the Bay. Comparison of these visualizations between pre- and post-discharge cases would provide a clear picture of the extent of the outfall's impact on the Bay's plankton communities.

The VPR provides data on the distribution of plankton species (zooplankton and large phytoplankton) and seston (e.g. marine snow) over scales of microns to hundreds of kilometers. The VPR provides data on important fragile forms that are destroyed by net sampling, including algal colonies, marine snow, and gelatinous species. Automated analysis of the data from the VPR system will allow real time mapping of plankton species composition. This system could potentially be towed at the cruising speed of the survey vessel and thus would not affect the normal completion time of the surveys. The net tows should be continued to provide continuity with the baseline monitoring.

Another potential monitoring method could be the use of a moored vertically profiling VPR which is now under development. This instrument would provide high resolution temporal data on plankton species abundance and associated environmental variables. Moored VPRs could be placed in the nearfield and farfield areas to capture event-scale forcing of the plankton populations such as upwelling events and offshore intrusions. Since the transport of the outfall discharge is intermittent due to tidal, wind, and buoyancy forcing, capturing the range of temporal scales involved in the transport of the discharge and the plankton species is important but is currently unknown.

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APPENDIX A

LIST OF PHYTOPLANKTON SPECIES FOUND IN THE 1992-1997 MWRA DATABASE

APPENDIX A. List of phytoplankton species found in the 1992-1997 MWRA database

Kingdom: Monera

Cyanophycota (Blue Green Algae)

Cyanophyceae

0301SPP	CYANOPHYCEAE SPP.
0301SPP4UM	CYANOPHYCEAE (NOSTOC-LIKE 4um diam)

Chroococcales

03020101SPP	AGMENELLUM SPP.
03020102SPP	ANACYSTIS SPP.
03020111SPP	MERISMOPEDIA SPP.
03020113SPP	MICROCYSTIS SPP.

Nostocales

03040105SG1	OSCILLATORIA CELLS #1 DIAM <5UM
03040105SPP	OSCILLATORIA SPP. (TRICHOME)
03040201SPP	ANABAENA SPP.
03SPP	BLUE GREEN SINGLE SPHERE SPP.
O3O4SPP	BLUE GREEN TRICHOME SPP.

Kingdom: Plantae

Chrysophyta (Golden-Brown Algae)

Chrysophyceae

0401SPP	SILICOFLAGELLATE SPP.
04020103SPP	OCHROMONAS SPP.

Ochromonadales

04020201SPP	DINOBRYON SPP.

Chromalinales

0405020101	CALYCOMONAS WULFFII
0405020102	CALYCOMONAS OVALIS
0405040202	PSEUDOPEDINELLA PYRIFORMIS
0405040301	APEDINELLA RADIANS

Dictyochales

0407010101	DICTYOCHA FIBULA
04070101SPEC	DICTYOCHA SPECULUM
0407010202	DISTEPHANUS SPECULUM
0407010301	EBRIA TRIPARTITA

Haptophyta

Prymnesiales

0603020101	PHAEOCYSTIS POUCHETII
06030401SPP	ACANTHOICA SPP.
06030404SPP	CALCIOSOLENIA SPP.
06030418SPP	SYRACOSPHAERA SPP.
0603050102	EMILIANIA HUXLEYI
0603050602	RHABDOSPHAERA CLAVIGER
0603050603	RHABDOSPHAERA HISPIDA
0603050604	RHABDOSPHAERA LONGISTYLIS
06030506SPP	RHABDOSPHAERA SPP.

Other Chrysophytes (Non-NODC Codes)

AAAAAAAA76	CHRYSOPHYTE SP. 1 (TURNER 1992)

Bacillariophyta (Diatoms)

Centrales

0702010103	MELOSIRA MONILIFORMIS	
0702010104	MELOSIRA NUMMULOIDES	
0702010107	AULACOSEIRA GRANULATA	
070201010702	AULACOSEIRA GRANULATA V. ANGUSTISSIMA	
07020101SG1	MELOSIRA SP#1 DIAM <20 MICRONS	
07020101SG2	MELOSIRA SP#2 DIAM >20 MICRONS	
07020101SPP	MELOSIRA SPP. *	
0702010201	PARALIA SULCATA	
07020102MARI	PARALIA MARINA	
0702010601	STEPHANOPYXIS PALMERIANA	
0702010602	STEPHANOPYXIS PALMERIANA	
0702010603	STEPHANOPYXIS TURRIS	
07020106SPP	STEPHANOPYXIS SPP.	
070201070201	ODONTELLA AURITA	
070201070201	ODONTELLA AURITA	BIDDULPHIA AURITA V. OBTUSA
07020110	CORETHRON SP.	
0702011001	CORETHRON CRIOPHILUM	
07020110SPP	CORETHRON SPP.	
0702011101	SCHRODERELLA DELICATULA	
0702011201	LEPTOCYLINDRUS DANICUS	
0702011202	LEPTOCYLINDRUS MINIMUS	
0702020101	SKELETONEMA COSTATUM	GREV+CLEVE
0702020113	SKELETONEMA POTOMOS	
07020203SG1	CYCLOTELLA SP#1 DIAM <10 MICRONS	
07020203SG2	CYCLOTELLA SP#2 DIAM 10-30 MICRONS	
07020203SG3	CYCLOTELLA SP#3 DIAM >30 MICRONS	

APPENDIX A. List of phytoplankton species found in the 1992-1997 MWRA database

Bacillariophyta (Diatoms)

Centrales (Continued)

0702020501	THALASSIOSIRA CF. AESTIVALIS	
0702020505	THALASSIOSIRA DECIPIENS	
0702020506	THALASSIOSIRA GRAVIDA	
0702020508	THALASSIOSIRA NORDENSKIOLDII	CLEVE
0702020510	THALASSIOSIRA ROTULA	
0702020511	THALASSIOSIRA SUBTILIS	
0702020517CF	THALASSIOSIRA CF. ECCENTRICA	
0702020521	THALASSIOSIRA PSEUDONANA	
07020205GRRO	THALASSIOSIRA CF. GRAVIDA/ROTULA	
07020205SG1	THALASSIOSIRA SP#1 DIAM <20 MICRONS	
07020205SG2	THALASSIOSIRA SP#2 DIAM >20 MICRONS	
07020205SP01	THALASSIOSIRA SP. 1 (GRAVIDA/ROTULA, TURNER 1992)	
07020205SPP	THALASSIOSIRA SPP.	
0702020701	DETONULA CONFERVACEA	
0702021001	COSCINOSIRA POLYCHORDA	
0702030104	COSCINODISCUS CENTRALIS	
0702030106	COSCINODISCUS EXCENTRICUS	
0702030112	COSCINODISCUS OCULUS-IRIDIS	EHRENB
0702030114	COSCINODISCUS RADIATUS	
0702030116	COSCINODISCUS WAILESII	
0702030116CF	COSCINODISCUS CF. WAILESII	
07020301SG1	COSCINODISCUS SP#1 DIAM <40 MICRONS	
07020301SG2	COSCINODISCUS SP#2 DIAM 40-100 MICRONS	
07020301SG3	COSCINODISCUS SP#3 DIAM >100 MICRONS	
07020301SPP	COSCINODISCUS SPP.	
0702040102	ACTINOPTYCHUS SENARIUS	
0702070101	BIDDULPHIA ALTERNANS	
0702070102	ODONTELLA AURITA	
0702070105	ODONTELLA LONGICRURIS	
0702070106	ODONTELLA MOBILENSIS	
0702070115	BIDDULPHIA SINENSIS	
07020701SPP	BIDDULPHIA SPP.	
0702071103	EUCAMPIA ZODIACUS	
07020712SPP	CLIMACODIUM SPP.	
07020714SPP	TRICERATIUM SPP.	
07020801SPP	HEMIAULUS SPP.	
0702080202	CERATAULINA PELAGICA	
0702090101	CHAETOCEROS AFFINIS	
0702090104	CHAETOCEROS ATLANTICUS	
0702090105	CHAETOCEROS BOREALIS	
0702090107CF	CHAETOCEROS CF. CERATOSPORUS	
0702090110	CHAETOCEROS COMPRESSUS	
0702090111	CHAETOCEROS CONCAVICORNIS	
0702090112	CHAETOCEROS CONSTRICTUS	
0702090116	CHAETOCEROS CRINITUS	C. CURVISETUM
0702090118	CHAETOCEROS DANICUS	
0702090119	CHAETOCEROS DEBILIS	
0702090120	CHAETOCEROS DECIPIENS	
0702090124	CHAETOCEROS DIDYMUS	
0702090124UN	CHAETOCEROS DIDYMUS (UNICELL FORM)	
0702090128	CHAETOCEROS EIBENII	
0702090130	CHAETOCEROS NEOGRACILIS	
0702090133	CHAETOCEROS LACINIOSUS	
0702090135	CHAETOCEROS LORENZIANUS	
0702090140	CHAETOCEROS PERUVIANUS	
0702090140CF	CHAETOCEROS CF. PERUVIANUS	
0702090142	CHAETOCEROS PENDULUS	
0702090148	CHAETOCEROS SEPTENTRIONALIS	
0702090150	CHAETOCEROS SOCIALIS	
0702090152	CHAETOCEROS SUBTILIS	
07020901CORO	CHAETOCEROS CORONATUS	
07020901RAUN	CHAETOCEROS RADICANS (1 CELL FORM)	
07020901SG1	CHAETOCEROS SP#1 DIAM <10 MICRONS	
07020901SG2	CHAETOCEROS SP#2 DIAM 10-30 MICRONS	
07020901SPGT	CHAETOCEROS SPP.(>10UM)	
07020901SPLG	CHAETOCEROS SPP. (10-20UM)	
07020901SPLT	CHAETOCEROS SPP.(<10UM)	
07020901SPP	CHAETOCEROS SPP.	
0702090202	BACTERIASTRUM DELICATULUM	
07020902SPP	BACTERIASTRUM SPP.	
0702100102	PROBOSCIA ALATA	
070210010202	RHIZOSOLENIA ALATA F. GRACILLIMA	
0702100104	RHIZOSOLENIA CALCAR-AVIS	
0702100107	RHIZOSOLENIA DELICATULA	
0702100108	RHIZOSOLENIA FRAGILISSIMA	
070210010902	RHIZOSOLENIA HEBETATA F. SEMISPINA	
0702100109XX	RHIZOSOLENIA SP. 1 (HEBETATA CF. SEMISPINA?)	TURNER 1992
0702100110	*RHIZOSOLENIA IMBRICATA*	
0702100113	RHIZOSOLENIA SETIGERA	
0702100114	RHIZOSOLENIA STOLTERFOTHII	
0702100115	RHIZOSOLENIA STYLIFORMIS	
0702100118	RHIZOSOLENIA SHRUBSOLEII	
07021001SPP	RHIZOSOLENIA SPP.	
0702100201	GUINARDIA FLACCIDA	

APPENDIX A. List of phytoplankton species found in the 1992-1997 MWRA database

Bacillariophyta (Diatoms)

Centrales (Continued)

0702100301	LITHODESMIUM UNDULATUM	
0702100301CF	LITHODESMIUM CF. UNDULATUM	
07021003SPP	LITHODESMIUM SPP.	
0702100401	DITYLUM BRIGHTWELLII	
07028888SPP	CENTRALES SPP.	
0702SG1	UNID. CENTRIC DIATOM DIAM <10 MICRONS	
0702SG2	UNID. CENTRIC DIATOM DIAM 10-30 MICRONS	
0702SG3	UNID. CENTRIC DIATOM DIAM 31-60 MICRONS	
0702SG4	UNID. CENTRIC DIATOM DIAM >60 MICRONS	

Pennales

0703010101	FRAGILARIA CROTONENSIS	
07030101SG1	FRAGILARIA SP#1 LENGTH <30 MICRONS	
07030101SG2	FRAGILARIA SP#2 LENGTH 30-60 MICRONS	
07030101SG3	FRAGILARIA SP#3 LENGTH >60 MICRONS	
07030103SPP	SYNEDRA SPP.	
0703010401	ASTERIONELLA GLACIALIS	
0703010403	ASTERIONELLA FORMOSA	
07030104GLAC	ASTERIONELLA GLACIALIS	
0703010501	THALASSIOTHRIX DELICATULA	
0703010502	THALASSIOTHRIX FRAUENFELDII	
07030105SPP	THALASSIOTHRIX SPP.	
0703010601	THALASSIONEMA NITZSCHIOIDES	
0703011010	LICMOPHORA ABBREVIATA	
07030110SPP	LICMOPHORA SPP.	
0703011302	STRIATELLA UNIPUNCTATA	
07030113SPP	STRIATELLA SPP.	
0703011905	GRAMMATOPHORA MARINA	
07030119SPP	GRAMMATOPHORA SPP.	
0703012003	RHABDONEMA MINUTUM	
0703012301	TABELLARIA FENESTRATA	
07030123SPP	TABELLARIA SPP.	
070301GLAC	ASTERIONELLOPSIS GLACIALIS	
07030401SPP	ACHNANTHES SPP.	
0703040203	COCCONEIS SCUTELLUM	EHRENB.
07030402SPP	COCCONEIS SPP.	
0703040401	RHOICOSPHENIA ABBREVIATA	
0703050109	NAVICULA DISTANS	
0703050109CF	NAVICULA CF. DISTANS	
0703050144CF	NAVICULA CF. FORCIPATA	
07030501LSPP	NAVICULOIDS (LYRATE)	
07030501SG1	NAVICULA SP#1 LENGTH <20 MICRONS	
07030501SG2	NAVICULA SP#2 LENGTH 20-60 MICRONS	
07030501SG3	NAVICULA SP#3 LENGTH >60 MICRONS	
07030501SPP	NAVICULOID DIATOMS	
0703050606	DIPLONEIS CRABRO	
07030509SPP	PINNULARIA SPP.	
0703051701	PLEUROSIGMA ANGULATUM	
0703051710	PLEUROSIGMA AESTUARII	
0703051710CF	PLEUROSIGMA CF. AESTUARII (TURNER 1992)	
07030517SPP	PLEUROSIGMA SPP.	
0703051802	GYROSIGMA FASCIOLA	
07030518SPP	GYROSIGMA SPP.	
07030520SPP	TROPIDONEIS SPP.	
07030522SPP	AMPHIPRORA SPP.	
070305GYPL	GYRO/PLEUROSIGMA SPP.	
07030701SPP	AMPHORA SPP.	
07030801SPP	GOMPHONEMA SPP.	
703100104CF	NITZSCHIA CF. DELICATISSIMA	
0703100104	NITZSCHIA DELICATISSIMA	
0703100107	NITZSCHIA LONGISSIMA	BREB ROLFS
0703100113	NITZSCHIA PUNGENS	
0703100113CF	NITZSCHIA CF. PUNGENS	
0703100114	NITZSCHIA SERIATA	
0703100134	NITZSCHIA SIGMA	
0703100151	NITZSCHIA CLOSTERIUM	W.SMITH
07031001SG1	NITZSCHIA SP#1 LENGTH <30 MICRONS	
07031001SG2	NITZSCHIA SP#2 LENGTH 30-70 MICRONS	
07031001SG3	NITZSCHIA SP#3 LENGTH >70 MICRONS	
07031001SP01	NITZSCHIA SP. 1 (DELICATISSIMA?),TURNER 1992	
07031001SPP	NITZSCHIA SPP.	
0703100201	BACILLARIA PAXILLIFER	
0703100402	CYLINDROTHECA CLOSTERIUM	
07031101SPP	SURIRELLA SPP.	
07038888SPP	PENNALES SPP.	
0703SG1	UNID. PENNATE DIATOM <10 MICRONS LENGTH	
0703SG2	UNID. PENNATE DIATOM 10-30UM LENGTH	
0703SG3	UNID. PENNATE DIATOM 31-60UM LENGTH	
0703SG4	UNID. PENNATE DIATOM 61-100UM LENGTH	
0703SG5	UNID. PENNATE DIATOM >100 MICRONS LENGTH	

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Chlorophycota (Green Algae)

Volvocales

08020204SPP	CHLAMYDOMONAS SPP.
08020211SPP	CHLOROGONIUM SPP.
08020302SPP	PANDORINA SPP.
08020305SPP	PLATYDORINA SPP.
08030501SPP	CHLORELLA SP.
08030503SPP	OOCYSTIS SPP.
08030507SPP	ANKISTRODESMUS SP.
0803050701	ANKISTRODESMUS FALCATUS
0803080102	PEDIASTRUM DUPLEX
080308010201	PEDIASTRUM DUPLEX V. CLATHRATUM
080308010205	PEDIASTRUM DUPLEX V. GRACILIMUM
0803080103	PEDIASTRUM SIMPLEX
08030801SP01	PEDIASTRUM SP. 1..TURNER 1992
08030801SPP	PEDIASTRUM SPP.
0803090203	SCENEDESMUS QUADRICAUDA
0803090205	SCENEDESMUS BIJUGA
08030902SPP	SCENEDESMUS SPP.
0803090302	CRUCIGENIA TETRAPEDIA
0803090304	CRUCIGENIA FENESTRATA
0803090309	CRUCIGENIA RECTANGULARIS
080309SPP	SCENEDESMUS SPP.
0803120201	MICRACTINIUM PUSILLUM
08032001SPP	GLOEOCYSTIS SPP.

Zygnematales

08100202SPP	STAURASTRUM SPP.

Prasinophyta (Green Flagellates)

Prasinophyceae

10020203SPP	PYRAMIMONAS SPP.
10020302SPP	TETRASELMIS SPP.
1002PYTESPP	PYRAMIMONAS/TETRASELMIS SPP.
1002SPP	PYRAMINONADALES SPP.

Euglenophycota (Euglenoids)

Eutreptiales

11020002SPP	EUTREPTIA SPP.
1102010102	EUTREPTIA LANOWII
1102010103	EUTREPTIA VIRIDIS
11020101SPP	EUTREPTIA SPP.

Euglenales

11030101SPP	EUGLENA SPP.
11030112SPP	PHACUS SPP.

Pyrrophycophyta (Dinoflagellates)

12	UNID. DINOFLAGELLATE
12SPP	UNID. DINOFLAGELLATE

Prorocentrales

1202010401	PROROCENTRUM MICANS
1202010402	PROROCENTRUM MINIMUM
1202010403	PROROCENTRUM TRIESTINUM
1202010407	PROROCENTRUM BALTICUM
1202010408	PROROCENTRUM COMPRESSUM
1202010413	PROROCENTRUM MAXIMUM
1202010414	PROROCENTRUM GRACILE
1202010420	PROROCENTRUM ROTUNDATUM
12020104SPP	PROROCENTRUM SPP.

Dinophysiales

1203020101	DINOPHYSIS ACUTA
1203020103	DINOPHYSIS NORVEGICA
1203020106	DINOPHYSIS OVUM
1203020107	DINOPHYSIS ACUMINATA
1203020108	DINOPHYSIS CAUDATA
1203020112	DINOPHYSIS FORTII
1203020116	DINOPHYSIS PUNCTATA
1203020118	DINOPHYSIS LACHMANNII
1203020144	DINOPHYSIS PULCHELLA
12030201SPP	DINOPHYSIS SPP.

Peridiniales

1204010108	AMPHIDINIUM SPHENOIDES
1204010109	AMPHIDINIUM LONGUM
1204010111	AMPHIDINIUM ACUTUM
1204010119	AMPHIDINIUM CRASSUM
12040101SPP	AMPHIDINIUM SPP.
12040101SPP	AMPHIDINIUM SPP. SYN. PHALACROMA SPP.
1204010306	GYMNODINIUM SPLENDENS
1204010327	GYMNODINIUM VERRUCULOSUM
12040103SG1	GYMNODINIUM SP.#1 5-20UM W 10-20UM L
12040103SG2	GYMNODINIUM SP.#2 21-40UM W 21-50UM L
12040103SG3	GYMNODINIUM SP.#3 41-70UM W 51-70UM L
12040103SG4	GYMNODINIUM SP.#4 71-100UM W 71-120UM L
12040103SP30	GYMNODINIUM SPP. (30UM)

Pyrrophycophyta (Dinoflagellates)

Peridiniales (Continued)

12040103SPP	GYMNODINIUM SPP.
1204010401CF	GYRODINIUM CF. AUREOLUM
1204010409	GYRODINIUM SPIRALE
1204010412	GYRODINIUM ESTUARIALE
1204010422	GYRODINIUM COMPLANATUM
12040104LACR	GYRODINIUM LACRYMA
12040104SG1	GYRODINIUM SP#1 5-20UM W 10-20UM L
12040104SG2	GYRODINIUM SP#2 21-40UM W 21-50UM L
12040104SG3	GYRODINIUM SP#3 41-70UM W 51-70UM L
12040104SG04	GYRODINIUM SP#4 71-100UM W 71-120UM L
12040104SPP	GYRODINIUM SPP.
1204010502	KATODINIUM ROTUNDATUM
12040105KATO	KATODINIUM SPP.
12040301	POLYKRIKOS SP.
1204090202	DIPLOPSALIS LENTICULA
12040902SP01	DIPLOPSALIS SP. 1 (LENTICULA?),,TURNER 1992
12040902SPP	DIPLOPSALIS SPP.
1204090305	GLENODINIUM ROTUNDUM
12040904OBLE	OBLEA SPP.
1204100101	HETEROCAPSA TRIQUETRA
1204100201	MINISCULA BIPES
1204100302	PROTOPERIDINIUM BREVIPES
1204100307	PROTOPERIDINIUM DEPRESSUM
1204100309	PROTOPERIDINIUM GRANII
1204100312	PROTOPERIDINIUM PALLIDUM
1204100313	PROTOPERIDINIUM PELLUCIDUM
1204100314	PROTOPERIDINIUM OVATUM
1204100331	PROTOPERIDINIUM PYRIFORME
1204100334	PROTOPERIDINIUM PENTAGONUM
1204100340	PROTOPERIDINIUM LEONIS
1204100354	PROTOPERIDINIUM DIABOLUM
1204100356	PROTOPERIDINIUM MINUTUM
1204100401	DISSODIUM ASYMMETRICUM
1204100509	PROTOPERIDINIUM DENTICULATUM
12041005ACIC	PROTOPERIDINIUM ACICULIFERA
12041005BIPE	PROTOPERIDINIUM BIPES
12041005BRCF	PROTOPERIDINIUM CF. BREVIPES
12041005BREE	PROTOPERIDINIUM BREVE
12041005BREV	PROTOPERIDINIUM BREVIPES
12041005CLAU	PROTOPERIDINIUM CLAUDICANS
12041005CONI	PROTOPERIDINIUM CONICUM
12041005DENT	PROTOPERIDINIUM DENTICULATUM
12041005DEPR	PROTOPERIDINIUM DEPRESSUM
12041005DIVE	PROTOPERIDINIUM DIVERGENS
12041005LEON	PROTOPERIDINIUM LEONIS
12041005MARI	PROTOPERIDINIUM MARIELEBOURAE
12041005MINU	PROTOPERIDINIUM MINUSCULLUM
12041005PALL	PROTOPERIDINIUM PALLIDUM
12041005PELL	PROTOPERIDINIUM PELLUCIDUM
12041005PENT	PROTOPERIDINIUM PENTAGONUM
12041005SG1	PROTOPERIDINIUM SP.#1 10-30W 10-40L
12041005SG2	PROTOPERIDINIUM SP.#2 31-75W 41-80L
12041005SG3	PROTOPERIDINIUM SP.#3 76-150W 81-150L
12041005SPP	PROTOPERIDINIUM SPP.
12041005STEI	PROTOPERIDINIUM STEINII
12041005TROC	PROTOPERIDINIUM TROCHOIDIUM
1204110202	GONYAULAX SPINIFERA
1204110205	GONYAULAX DIACANTHA
1204110207	GONYAULAX DIGITALIS
1204110212	GONYAULAX POLYGRAMMA
1204110214	AMYLAX TRIACANTHA
12041102SPP	GONYAULAX SPP.
1204140102	CERATIUM FURCA
1204140103	CERATIUM FUSUS
1204140104	CERATIUM LINEATUM
1204140105	CERATIUM LONGIPES
1204140106	CERATIUM MACROCEROS
1204140108	CERATIUM TRIPOS
1204140110	CERATIUM MASSILIENSE
12041401SPP	CERATIUM SPP.
12041602SPP	OXYTOXUM SPP.
12041701SPP	CLADOPYXIS SPP.
1204170201	CLADOPYXIS SETIFERA
1204270101	SCRIPPSIELLA TROCHOIDEA

Other Dinoflagellates (Non-NODC Codes)

61694901TAMA	ALEXANDRIUM TAMARENSE
AAAAAAAA77	THECATE DINOFLAGELLATE SPP.
AAAAAAAA78	THECATE FLAGELLATE SPP.
AAAAAAAA79	ATHECATE DINOFLAGELLATE
AAAAAAAA64	UNID. DINOFLAGELLATES
AAAAAAAA65	UNID. NAKED DINOFLAGELLATE
AAAAAAAA70	DISSODINIUM SPP.
AAAAAAAA72	HETEROSIGMA AKASHIWO
AAAAAAAA74	ATHECATE DINOFLAGELLATE SP. 1 (TURNER 1992)

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Microflagellates (Non-NODC Codes)		
	PHYTOFLAGSG1	UNID. MICRO-PHYTOFLAG LENGTH <10 MICRONS
	PHYTOFLAGSG2	UNID. MICRO-PHYTOFLAG LENGTH >10 MICRONS
	18888888SPP	MICROFLAGELLATES
	AAAAAAAA81	UNID. MICRO-PHYTOFLAG LENGTH <10 MICRONS
	AAAAAAAA82	UNID. MICRO-PHYTOFLAG LENGTH >10 MICRONS
Cryptophycophyta		
Cryptophyceae		
	14020303SG1	CRYPTOMONAS SP#1 LENGTH <10 MICRONS
	14020303SG2	CRYPTOMONAS SP#2 LENGTH >10 MICRONS
	140203SPP	CRYPTOMONADS
Kingdom: Animilia		
Protozoa		
Coccolithophorida		
	3409SPP	COCCOLITHOPHORE SPP.
	3409SPP	COCCOLITHOPHORID
Choanoflagellida		
	3423SPP	CHOANOFLAGELLATE SPP.
Rhabdophorina		
	3516010202	MESODINIUM RUBRUM
Cnidaria		
Hydrozoa		
	3701SPP	HYDROZOA SPP.
Platyhelminthes (Flatworms)		
Turbellaria (Planarians)		
	39SPP	PLATYHELMINTHES:TURBELLARIA
Nematoda (Nematodes)		
	47SPP	NEMATODA SPP.
Annelida		
Polychaeta		
	5001SPP	POLYCHAETE SPP.
Mollusca		
	5085SPP	GASTROPODA;MOLLUSCA
	5181SPP	OPISTHOBRANCHIA SPP.
Bivalvia (Bivalves)		
	55SP02	BIVALVE SP. B
	55SP03	BIVALVE SP. C
	55SP04	BIVALVE SP. D
	55SP05	BIVALVE SP. E
	55SPP	BIVALVIA SPP.
Crustacea		
	61	CRUSTACEA:UNIDED CRUSTACEAN
	61SPP	CRUSTACEA:UNIDED CRUSTACEAN
Eucladocera		
	6109010101	PENILIA AVIROSTRIS
	6109050101	EVADNE NORDMANNI
	61090501SPP	EVADNE SPP.
	6109050203	PODON POLYPHEMOIDES
	61090502SPP	PODON SPP.
Copepoda (Copepods)		
	6117CALA	COPEPODA CALANOIDA
	6117HARP	COPEPODA HARPACTICODA
	6117SPP	COPEPOD SPP.
Calanoida	6118010208	CALANUS FINMARCHICUS
	6118040101	PARACALANUS PARVUS
	6118040102	PARACALANUS CRASSIROSTRIS
	61180505NEWM	PSEUDOCALANUS NEWMANI
	6118160207	METRIDIA LUCENS
	6118170105SG1	CENTROPAGES HAMATUS-C4+C5
	6118170106	CENTROPAGES TYPICUS
	61181701SPP	CENTROPAGES SPP.
	6118190201	Pseudodiaptomus coronatus
	6118200203	EURYTEMORA HERDMANI
	6118200303	TEMORA LONGICORNIS
	6118290104	ACARTIA TONSA
	6118290113	ACARTIA HUDSONICA
	6118300101	TORTANUS DISCAUDATUS

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Copepoda (Copepods)			
Harpacticoida	6119090101	MICROSETELLA NORVEGICA	
	6119110102	ALTEUTHA DEPRESSA	
	6119140101	EUTERPINA ACUTIFRONS	
	6119SPP	HARPACTICOIDA SPP.	
Cyclopoida	61200602SPP	SAPHIRELLA SPP.	
	6120090103	OITHONA SIMILIS	CLAUS
	6120090123	OITHONA ATLANTICA	
Cirripedia (Barnacle)	6130SPP	CIRRIPEDE SPP.	
	6151SPP	MYSIDACEA SPP.	
	6168SPP	AMPHIPODA SPP.	
Decapoda (Crabs, Lobsters, Crayfishes, Prawns, Shrimp)	6175SPP	DECAPODA SPP.	
Arthropleona (Elongate Springtails)	62090108LOCF	ODONTELLA CF. LONGICRURIS (TURNER 1992)	
	62090108SICF	ODONTELLA CF. SINENSIS	
	62090108SINE	ODONTELLA SINENSIS	
Melooidea	631806SPP	EUGLENOID SPP.	
Bryozoa (Bryozoans)	78SPP	BRYOZOA SPP.	
Echinodermata (Echinoderms)	81PLUT	ECHINODERM PLUTEI	
Chaetognatha (Arrow Worms)	83SPP	CHAETOGNATHIA SPP.	
Tunicata (Sea Squirts, Tunicates)	8401SPP	ASCIDIAN SPP.	
Copelata	8413010101	OIKOPLEURA DIOICA	
Other Zooplankton (Non-NODC Codes)	AAAAAAAA60	MEDUSA	
	AAAAAAAA66	UNIDENTIFIED LARVAE	
	AAAAAAAA71	FISH SPP.	

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