Nutrients and Massachusetts Bay:

an update of eutrophication issues

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FINAL REPORT

NUTRIENTS AND MASSACHUSETTS BAY: AN UPDATE OF EUTROPHICATION ISSUES

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Executive Summary

This report updates a previous synthesis report on nutrients and eutrophication issues for Massachusetts Bay (Kelly, 1991). A considerable body of information gathered in the last several years has added significantly to understanding of the Bay's ecology. In particular, the water column monitoring program initiated for the MWRA by Battelle Ocean Sciences in 1992 has provided an extensive data set on nutrients, plankton, and water quality in Massachusetts and Cape Cod Bay. The data have been reported in three volumes for 1992 (Kelly *et al.*, 1992; Kelly *et al.*, 1993a,b) and as an annual report (Kelly *et al.*, 1993c). The 1992 data, along with additional special studies of water column and benthic processes sponsored by the MWRA and by the Massachusetts Bays Program during the 1991-1993 period, are synthesized in this report to explore three principal eutrophication issues that were identified and recognized in the development of the MWRA monitoring program.

Key parameters chosen as indicators for water column enrichment were nutrients, chlorophyll, plankton species, dissolved oxygen, and metabolism (MWRA, 1991). In this report, the monitoring data on these parameters are used to (1) confirm previously described gradients and patterns, (2) aid a search for new patterns, potentially to develop empirical models useful in making predictions, and (3) allow new or improved calculations on ecological and nutrient dynamics relative to the eutrophication issues that are central to the monitoring program.

The principal themes examined in this report focus on three issues described below. For each, the specific data and topics explored are listed, then the main conclusions highlighted. Many notions raised in the previous synthesis report are reinforced by the recent data collected by the monitoring program.

Issue (1) *How would diversion of the MWRA effluent affect nitrogen distribution and nitrogen flux in Massachusetts Bay? Kelly (1991) proposed the notion that most of the nutrients in MWRA effluent discharged to the Harbor is not retained in the Harbor and probably goes to the Bay quite rapidly. A variety of data is summarized and calculations are presented to provide perspective on the present situation, the scales of change anticipated, and the proper monitoring design to detect future change in Massachusetts Bay.*

This first issue centers on the present and future influence of the MWRA outfall effluent on nitrogen distribution in Massachusetts Bay. Evaluation of this influence includes examination of

- the budget for nitrogen input to Boston Harbor as related to the present nitrogen export,
- the present distribution of nutrients in the Bay, particularly as it reflects the role of Boston Harbor as the strongest source in the region near the proposed outfall site, and
- the present and possible future role of various sources in structuring nutrient distributions in the Bays and in the region near the proposed outfall site.

The main conclusions drawn from the examination of the data are:

- The available evidence strongly and consistently supports the concept that most of the nitrogen in the MWRA effluent presently discharged to Boston Harbor is exported to the Bay.
- There are sharp differences in the form of nitrogen exported seasonally from the Harbor to the Bay. The implication from a variety of data is that organic N dominates the N exported from the Harbor during the summer, whereas DIN export is high and dominant in winter.
- Presently, the nutrients exported into the area surrounding the future diffuser track are delivered directly to the surface layers, especially during the stratified period and, thus, are completely available to the phytoplankton. In the future, the initial dilution will be faster than that presently occurring at the edge of the Harbor. Also, in the nearfield, vertical flux of nutrients to the surface productive layers is constrained during stratification by vertical density gradients. With the new outfall the vertical nutrient flux will increase; however, calculations suggest that the increased flux from bottom waters will still be much less than the present horizontal transport from inshore to surface layers surrounding the outfall diffuser. Thus, the overall nutrient supply to surface productive layers may decrease.

From these conclusions one can argue that the principal scale of monitoring for change is more local than regional. At the regional scale, the nutrients being discharged today are circulated throughout the Bays, just as they will be in the future. While monitoring efforts principally need to focus on local ecological changes that are likely and detectable, research and monitoring should also continue to identify factors that could influence broad-scale variability in nutrients in Massachusetts and Cape Cod Bays.

Issue (2) *What is the response of plankton to nutrients? This issue is multifaceted and involves the question of how changes in nutrient distributions will affect accumulation of chlorophyll, an indicator of plankton biomass, an excessive amount of which can alter water quality in an undesirable manner. A second, related, question is whether changes in nutrient distributions will promote growth of certain plankton species that directly alter water quality and/or produce toxins that may accumulate in food chains, including those affecting commercial fish or shellfish.*

This issue was addressed with recent monitoring data on chlorophyll and plankton taxonomy. Two major topics were approached with the following emphases:

- An examination of annual, seasonal, and fine-scale trends in data on chlorophyll concentrations was conducted to describe patterns in chlorophyll concentrations relative to nitrogen concentrations.
- The complexity of predicting phytoplankton species change was reviewed, incorporating historical perspectives on Massachusetts Bay and Cape Cod Bay and presenting broad spatial trends in the plankton species and communities documented in 1992 monitoring data. The possible influence of nutrient concentrations and ratios (nitrogen/silicate) are briefly explored.

More data are available to assess chlorophyll responses to nutrients than to determine possible nutrient effects on species. Therefore, more conclusions from the data relate to the question of stimulation of chlorophyll by nutrients:

- For 1992, the range in annual average surface chlorophyll fluorescence was about 1.4 to 4.3 μg L^{-1} at 46 stations in Massachusetts and Cape Cod Bays. The higher chlorophyll concentrations were found at the edge of Boston Harbor and, as with nutrients, there was a gradient of decreasing concentration of chlorophyll concentration with distance away from the Harbor into Massachusetts Bay.
- Significant linear regressions were obtained using the annual average chlorophyll concentrations versus the annual average concentration of different forms of nitrogen in the water column. Similarly strong trends were also evident at a seasonal scale, using data summarized for the surface layer of 21 nearfield stations sampled intensively during the stratified period of 1992. The regressions provide some predictive capability of the influence of nutrients on chlorophyll, but the capability is restricted because factors other than nutrients also influence chlorophyll along the Harbor-Bay gradient.
- Stations near the Harbor have, on average, chlorophyll concentrations that are less than might be predicted for their high nutrient concentrations. High

turbidity may influence light availability near the Harbor and thus limit chlorophyll relative to clearer offshore waters.

- Some thought experiments were conducted to consider the nutrient-chlorophyll response around the new outfall. These employ regressions developed from the 1992 data, including one that, in principle, provides for possible interactive effects of nutrients and light upon chlorophyll. Results suggest that upon transfer of the present nutrient concentration within the Harbor to the middle of the nearfield, the chlorophyll concentration, on average, could increase by about a factor of two, but this does not consider the additional dilution imparted at the offshore discharge site. Moreover, since we now can project that a decrease in surface nutrient flux from Harbor export will accompany the effluent diversion, the projections of surface chlorophyll change include the possibility of a decrease, on average, throughout the nearfield. Therefore, the principal change due to relocation of the outfall may not be the average chlorophyll concentration, but a shift in the vertical distribution of chlorophyll to a lower depth. Such a shift would accompany the projected sharp decrease in the present supply from inshore to surface waters offshore, an increase in the future nutrient supply from nearfield bottom-waters, and presence of a relatively deep photic zone in the nearfield.

- Patterns between chlorophyll and salinity were revealed from analysis of the summer season and higher-resolution sampling in the nearfield/Harbor area. Several lines of evidence suggest there is stimulation of chlorophyll concentration (perhaps 1-2 $\mu g\ L^{-1}$) in immediate receiving waters several kilometers outside the Harbor, a phenomenon which may relate to export of nutrients into an area with increased water clarity.

- Review of the influence of nutrients on plankton species indicates that the present capacity to formulate predictions is limited. The results from the few experimental studies that have examined the response of plankton communities to long-term nutrient enrichment have been complex, but blooms of noxious species and shifts in relative proportions of diatoms to flagellates have been infrequent and often have not followed scenarios inferred from short-term, simple experiments. 1992 monitoring data provide information relevant to the potential alteration of nitrogen/silicate ratios in the future nearfield area. Initial comparisons of stations with differing nutrient concentrations or ratios suggest that the relatively small changes in nearfield nutrient levels that are projected are within the bounds already experienced within the Bay. Thus, it would be difficult to conclude from the data that projected nutrient changes will yield major species composition shifts to nuisance/noxious forms.

Issue (3) *What are present rates of metabolism in the Bay under existing nutrient conditions? What are the current trends in bottom-water dissolved oxygen (DO) concentrations and how do these relate to metabolism and the fate of pelagic production in Massachusetts Bay?*

The 1992 monitoring data provide information on seasonal patterns in primary production, benthic respiration, and bottom-water dissolved oxygen (DO) fluctuations. Data analyses were performed to describe rates of change for bottom-water DO concentrations in Massachusetts Bay during 1992 and to identify factors affecting DO concentrations.

The main conclusions reached relative to this third issue are:

- Variability in primary production is high and small changes in time and space will be difficult to detect. Rates of benthic metabolism do not indicate a major role for the benthos in the consumption of primary production within the nearfield region of the Bay.

- An analysis of the nutrient sources supporting primary production in the nearfield shows that nitrogen presently exported from the Harbor quantitatively dominates nitrogen input to the surface layers and could support virtually all the primary production. In contrast, benthic regeneration and diffusive fluxes across the pycnocline into euphotic surface layers supply only on the order of 10% of the needs of primary producers. In the future, the flux from inshore will be curtailed and the flux from bottom-waters to the surface layer will increase. Calculations suggest that the overall nitrogen to the surface layer will be substantially diminished and primary production may therefore decline in much of the nearfield.

- Bottom-water DO declines during the stratified season in the nearfield and in Stellwagen Basin. Sustained rates of consumption appear to be about 0.01-0.06 mg O_2 L^{-1} d^{-1}. These rates were estimated from gradual changes monitored over weeks to months, and may imperfectly estimate respiration rates. Unfortunately, attempts to measure water column respiration directly in bottle incubations were unsatisfactory. In 1992, rates were often below limits of detection using a short-term (hours) incubation method. Nonetheless, the data suggest that a relatively small fraction of primary production is presently consumed within the bottom waters of Massachusetts Bay. Preliminary calculations suggest that much of the organic matter produced in the surface layers may be consumed within the upper water column.

- Because changes occur slowly, DO concentration trends across surveys should suffice as a principal tool to unambiguously record DO and monitor its rate of change.

Even though bottom-water DO concentrations change slowly enough to be monitored by the present frequency of summer and early fall surveys, it is argued that improved estimates of water column rates of oxygen consumption would be beneficial to understand the ecosystem. Precise respiration rate estimates would enable us to confirm or refute the notion that consumption of organic matter presently occurs rapidly within the upper water column and would also provide data useful to validate or improve water-quality modeling and predictions.

Preface

This report draws largely upon water column studies that were conducted in 1992 as part of the Massachusetts Water Resources Authority's (MWRA) Baseline Monitoring Program for the offshore effluent outfall. Such an effort requires the talent and dedication of many people. The 1992 water quality studies were performed by researchers from Battelle Ocean Sciences (BOS), the University of Rhode Island (URI), and the University of Massachusetts at Dartmouth (UMD). From BOS, Carl Albro, Chip Ryther, Jack Bechtold, Kevin King, and Paul Dragos carried out the surveys and maintained the instruments. Carl Albro, John Hennessy, Ellie Baptiste, and Rosanna Buhl were responsible for processing, management, and QA review of an enormous volume of data. From URI, Peter Doering coordinated all aspects of sampling for the nutrient and metabolism studies, the data processing and management, and contributed to data interpretation. Peter, Laura Reed, and Edwin Requintina were the mainstays of field sampling; they processed and analyzed literally thousands of laboratory samples. From UMD, Jeff Turner and Dave Borkman sampled, identified, and counted the phytoplankton and zooplankton; provided succinct interpretive guidance; and expended extra effort to provide quantitative results for samples that were "slimed" by colony-forming species. Numerous other individuals from each institution also assisted at various critical times.

Individual survey results have been reported to the MWRA in survey reports and key monitoring parameters have been summarized in an annual report. The data from the MWRA monitoring program were intended to serve a dual purpose: (1) to establish a baseline against which meaningful changes could be measured in Massachusetts and Cape Cod Bays and (2) to enhance the understanding of the ecosystem and improve predictability of change due to offshore effluent discharge. A small measure of the success of the 1992 data collection efforts is, hopefully, reflected in this report.

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1.0 Introduction

This report updates a previous report (Kelly, 1991) and uses it, in part, as a template to relate some of the progress that has been made on several eutrophication issues in the past two years. Over the last five years there have been numerous surveys, data summaries, and modeling efforts that have helped to describe the environment and ecological features of the Massachusetts and Cape Cod Bays ecosystems; some studies have also attempted to predict the consequences of nutrient enrichment and diversion of the MWRA effluent (e.g., EPA, 1988; MWRA, 1990; Townsend *et al.,* 1991; Cura, 1991; Smayda, 1992; EPA, 1993).

The most recent data from the 1992 MWRA water quality monitoring studies (Turner, 1993; Kelly *et al.,* 1993), as well other recent studies funded by the Massachusetts Bays Program (Geyer *et al.,* 1992; Becker, 1992), add to the growing body of information that enhances understanding of the Bays' ecology. In this report, the monitoring data are used to (1) confirm previously described nutrient patterns, (2) aid a search for new patterns, potentially to develop empirical models useful in making predictions, and (3) allow new or improved calculations on ecological and nutrient dynamics relative to the eutrophication issues that are central to the monitoring program.

Several key issues related to eutrophication were recognized in the development of the MWRA monitoring program. The key parameters chosen as indicators for water column enrichment were nutrients, chlorophyll, dissolved oxygen and metabolism, and plankton species (MWRA, 1991). Interactions among these parameters are examined in this report.

The principal issues examined in this report include the following:

(1) How would diversion of the MWRA effluent affect nitrogen distribution and nitrogen flux in Massachusetts Bay? This issue is discussed in Section 2. The examination focuses on three types of information relevant to the issue:

- the budget for nitrogen input to Boston Harbor as related to the present potential for export,
- the present distribution of nutrients in the Bay, particularly as reflective of Boston Harbor as the strong source in the region near the proposed outfall site, and
- the present role of northern river sources in structuring nutrient distributions in the Bays and in the region near the proposed outfall site.

A variety of data is summarized and calculations are presented to provide perspective on the present situation, the scales of change anticipated, and the proper monitoring design to detect future change in Massachusetts Bay.

(2) What is the response of plankton to nutrients? This issue is multifaceted and is addressed in Sections 3 and 4. Section 3 discusses present spatial and temporal trends in chlorophyll, a measure of plankton biomass, relative to nitrogen concentrations. From these patterns, simple projections of change in chlorophyll are provide in light of expected changes in nutrients discussed in Section 2. Further, scales of measurement for chlorophyll and suitable indicators for monitoring relative to those scales are briefly discussed. Section 4 reviews the complex issue of plankton species composition as related to nutrients, and centers on the issue of "nuisance" or "noxious" species that might be anticipated in the future. Discussion examines present trends relative to nutrient distributions, present and expected. Both the possible influence of nutrient concentrations and ratios (nitrogen/silicate) are explored briefly.

(3) What are present rates of metabolism in the Bay under existing nutrient conditions? How do trends in bottom-water dissolved oxygen (DO) concentrations in Massachusetts Bay relate to metabolism? Section 5 uses the 1992 monitoring data to examine what can be described in terms of seasonal patterns in production, respiration, and in bottom-water DO changes. A final focus is to review the ability of the monitoring program to assess changes in DO concentrations and certain process rates related to this important endpoint.

A final section summarizes major findings and relates scientific progress on these major eutrophication issues.

2.0 Nitrogen Loading and Distribution in Massachusetts Bay

A prime issue related to diversion of the MWRA effluent to an offshore outfall centers on what role present discharges to the Harbor play in nutrient budgets and ecological dynamics of Massachusetts Bay. Boston Harbor has received extremely high nutrient loads for decades and approximately 88% of the total nitrogen input comes from MWRA effluent discharge (Alber and Chan, 1994). The topics examined in this section start with a synthesis of the qualitative and quantitative understanding of what happens to the nitrogen presently put into the Harbor, particularly with respect to nitrogen export. To corroborate a main conclusion of the synthesis — that a high percentage of nitrogen input to the Harbor is rapidly exported — the distribution of nitrogen in Bay receiving waters is examined. Chemical gradients are described using water-column monitoring data collected in the area extending from the Harbor and including a 100 km^2 (10 x 10 km) region surrounding the future outfall in western Massachusetts Bay, referred to as the "nearfield" in the context of MWRA monitoring. Brief consideration is then given to another possible source of nutrients — flow from rivers to the north into Massachusetts Bay. Such a source, if strong, could also play a role in the ecological dynamics in the nearfield. Finally, some features of nutrient dynamics in the nearfield area are examined and a perspective offered on expected scales of change with effluent diversion.

2.1 Export of Nitrogen to the Bay

When the predecessor to this report was prepared, a logical focus for nutrients in Massachusetts Bay was nitrogen (N) and the logical starting point for discussion was Boston Harbor (Kelly, 1991). Historical data on water quality indicated that, despite the high nutrient loads, nitrogen concentrations of the Harbor water were not particularly high. Flushing characteristics, gradients into Massachusetts Bay, and other observations supported the notion that much of the input nitrogen was quickly transported to the Bay. However, as stated then ... "Firm quantification of nitrogen export may await budgeting of nitrogen buried in Harbor sediments, dredged and removed, or lost to the atmosphere through denitrification" (Kelly, 1991). Although there was some information to make rough estimates of organic matter burial in Boston Harbor sediments, the lack of information on denitrification processes was recognized. MWRA responded by supporting direct measurements of sediment denitrification and benthic flux of nutrients, which were initiated in September 1991.

Initial benthic flux data reports were available later in the year when Christensen (1991) speculated that denitrification might remove at least 25%, and perhaps as high as 70%, of the nitrogen entering the Harbor. Benthic flux studies and denitrification measurements conducted in 1991 (Giblin *et al.,* 1992; Kelly and Nowicki, 1992; Nowicki, 1994) led to a more comprehensive set of measurements throughout 1992 (Giblin *et al.,* 1993; Kelly and Nowicki, 1993). Measurements of denitrification at several Harbor stations, as well as at several Massachusetts Bay stations, continued in 1993 and 1994.

The benthic flux data now collected for Boston Harbor are extensive. In terms of spatial and temporal coverage of sediment denitrification in estuarine ecosystems, the present data set has few parallels (cf. Seitzinger, 1988; Kemp *et al.,* 1990). The subtidal sediment communities examined include those near MWRA discharges, at more distant depositional areas, and at a less depositional reference site. Direct measurements of denitrification using the method of Nowicki (1994) and indirect stoichiometrically-modeled estimates (Giblin *et al.,* 1993) agree surprisingly well. Absolute denitrification rates at the depositional sites are very high for subtidal estuarine sediments, but not compared to N loading to the Harbor. The most recent annual input-output budget for N in Boston Harbor (circa 1992, after cessation of MWRA sludge discharges) suggests that sediment denitrification may remove less than 10% of the N input (Kelly and Nowicki, 1993).

While the MWRA effluent N load is fairly constant through the year (M. Hall, MWRA, personal communication), sediment denitrification rates are generally much higher during warm temperatures. Thus, a greater fraction of N input appears to be denitrified during the summer months (~12% in 1992 — Kelly and Nowicki, 1993). In part, this seasonal variation in the relative importance of denitrification could be a function of time lags — some N_2 gas loss may come from degradation of organic nitrogen that was deposited during late winter-spring but was not metabolized until seasonal warming occurred.

Other studies since 1991 have added to the evidence that suggests tidal flushing exports a substantial portion of the N input. A rough calculation of sediment burial (Kelly and Nowicki, 1992) has not yet been improved by direct measurements, but the modeling exercises of Adams *et al.* (1992) estimate that N burial in the Harbor is probably similar to the original estimate (i.e., only several percent of the input). Signell and Butman (1992) have solidified understanding of the flushing dynamics of the whole Harbor; their work confirms flushing times on the order of days, particularly in the immediate areas of MWRA discharges. Finally, Adams *et al.* (1992) used a simple box model with flushing rates similar to Signell and Butman; their model predicted that denitrification could account for only about 5% of the N input to the Harbor. Model sensitivity analyses forecast about 22% as an upper bound.

In summary, considerable evidence supports a conclusion that denitrification does not remove a majority of the N input to Boston Harbor. The amount of N degassed or buried in the Harbor appears to be approximately 10% of the inputs which, by difference, leaves roughly 90% exported to western Massachusetts Bay. The export flux estimated in this way incorporates the uncertainties of all other flux estimates in the budget; thus the evidence is indirect evidence, but it is balanced by evidence of enrichment in the nearshore receiving system (next Section).

Coda. During 1992, the Harbor appeared to be in transition. The benthos resembled a macrofaunal community recovering from excessive organic enrichment (cf. Kelly and Kropp, 1992; Blake *et al.,* 1993). Some changes may be related to sludge abatement, although such a conclusion is premature. Certain macrofauna, including ampeliscid amphipods which often are among initial recolonizers of perturbed sediments, can affect sediment denitrification rates (Kelly and Nowicki, 1993; Nowicki *et al.*, 1994; Pelegri *et al.*, 1994). In 1992 amphipods became abundant in the northern Harbor and have been observed in abundance in cores from the benthic flux station nearest the old MWRA sludge discharge pipe (ceased December 1991).

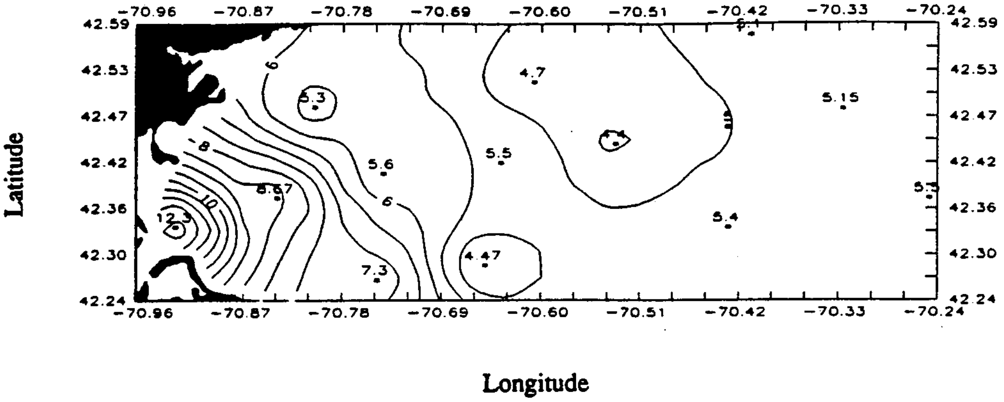
A consequence of benthic organismal recolonization and community change may be increased denitrification in highly organic sediments where the flow of oxygen, necessary for coupled nitrification—denitrification, is increased by increased bioturbation as the community progresses to include more deep-dwelling organisms. As ecological changes progress in the Harbor, denitrification may temporarily take on greater importance; in part, such a phenomenon would be related to the metabolism of organic nitrogen temporarily "buried" in sediments and be uncoupled from current inputs. Therefore, as nitrogen loads to Boston Harbor are reduced, benthic fluxes in general and denitrification in specific may become more important in the Harbor nitrogen budget for some period as the water and sediments adjust to new "steady state“ conditions. An increased sediment denitrification conceivably would act to accelerate the pace of recovery from nutrient-enrichment effects in the Harbor that are not quickly modulated by rapid tidal flushing. Thus, while the studies now show denitrification plays a small role in modifying the present export of nitrogen from the Harbor, denitrification may be ecologically significant to the overall recovery rate of the Harbor. Continued monitoring of benthic fluxes and denitrification in the Harbor is therefore warranted.

2.2 The Distribution of Nutrients in the Bay, with a Focus on Western Massachusetts Bay

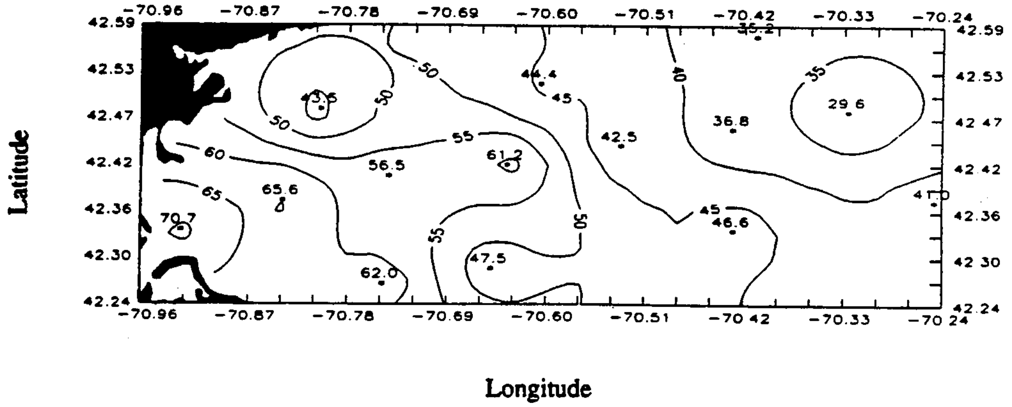
There is a strong and consistently-noted environmental gradient stretching out from the Harbor into the Bay, which provides additional evidence of nitrogen export. For example, from extensive sampling during the early 1970's, Pandan (19~~9~~77) noted that water-column dissolved nutrient (NO_3, PO_4) and suspended solid concentrations usually decreased sharply from the Boston Harbor to locations in Massachusetts Bay that are presently sampled as the nearfield for the MWRA program. More recently, data from 1989-1990 (from Townsend *et al.*, 1991 as summarized by Kelly, 1991) displayed sharp decreases in dissolved and particulate nitrogen concentrations from the mouth of Boston Harbor into western Massachusetts Bay (Figure 2-1). Estuarine-shelf nutrient concentration gradients may be more the rule than the exception for nutrient-enriched estuaries and physically-stratified inner shelf receiving waters. For example, a pronounced peak in water column nitrogen concentration has been measured at the inner shelf seaward of New York Harbor, as well as near the mouth of Delaware Bay and Chesapeake Bay (e.g., Matte and Waldhauer, 1988; Battelle, 1992).

Annual patterns for 1992. Sampling for the 1992 MWRA monitoring was conducted at 46 stations throughout Massachusetts and Cape Cod Bays (Figure 2-2). A distinct gradient from Boston Harbor was evident in annual mean dissolved inorganic nitrogen ($DIN = NH_4 + NO_3 + NO_2$) concentration for surface water (Figure 2-3). Annual mean calculations are described and values tabulated in the Appendix (Table 1).

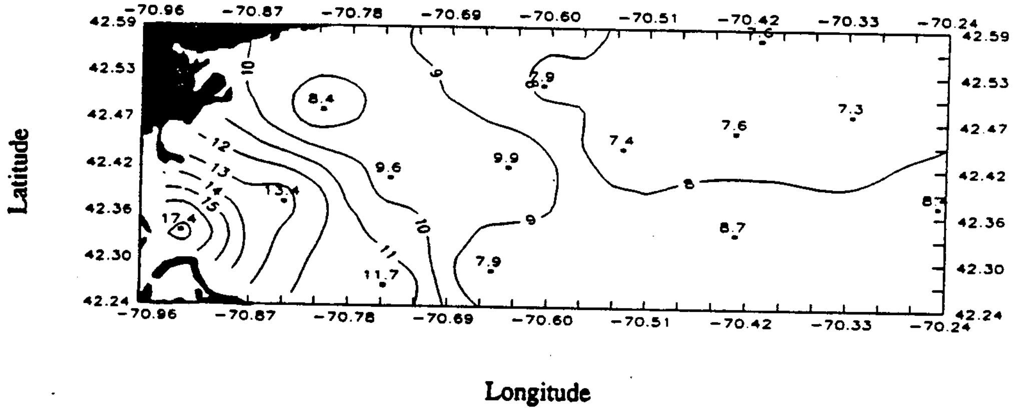
Comparability of 1992 and 1989/1990 data summaries notwithstanding (see Appendix), the 1989/1990 DIN means may have been generally higher because, unlike 1992, the first 1990 survey sampled winter baseline conditions prior to initiation of the winter-spring bloom. Townsend *et al.*'s (1991) stations did not extend into Cape Cod Bay, but Figure 2-3 suggests that in 1992 the surface waters in Massachusetts Bay had more DIN than surface waters in Cape Cod Bay. For 1992, the lower annual means for DIN in Cape Cod Bay, in part, are due to the initiation of the winter-spring bloom earlier in that region than in Massachusetts Bay. Consequently, DIN and other dissolved nutrients were already reduced throughout the water column by the first survey in late February 1992 (Kelly *et al.*, 1992); note that a number of seasonal differences in nutrients between the two Bays have been emphasized recently (Geyer *et al.*, 1992; Kelly *et al.*, 1992). These broad comparisons illustrate that



(a) DIN (µM)



(b) PON (µg)



(c) DIN and PON (µM)

Figure 2-1. **Annual surface water integrated averages of nitrogen at the stations sampled by Townsend *et al.* (1991) in Massachusetts Bay. [Source: Kelly, 1991]**

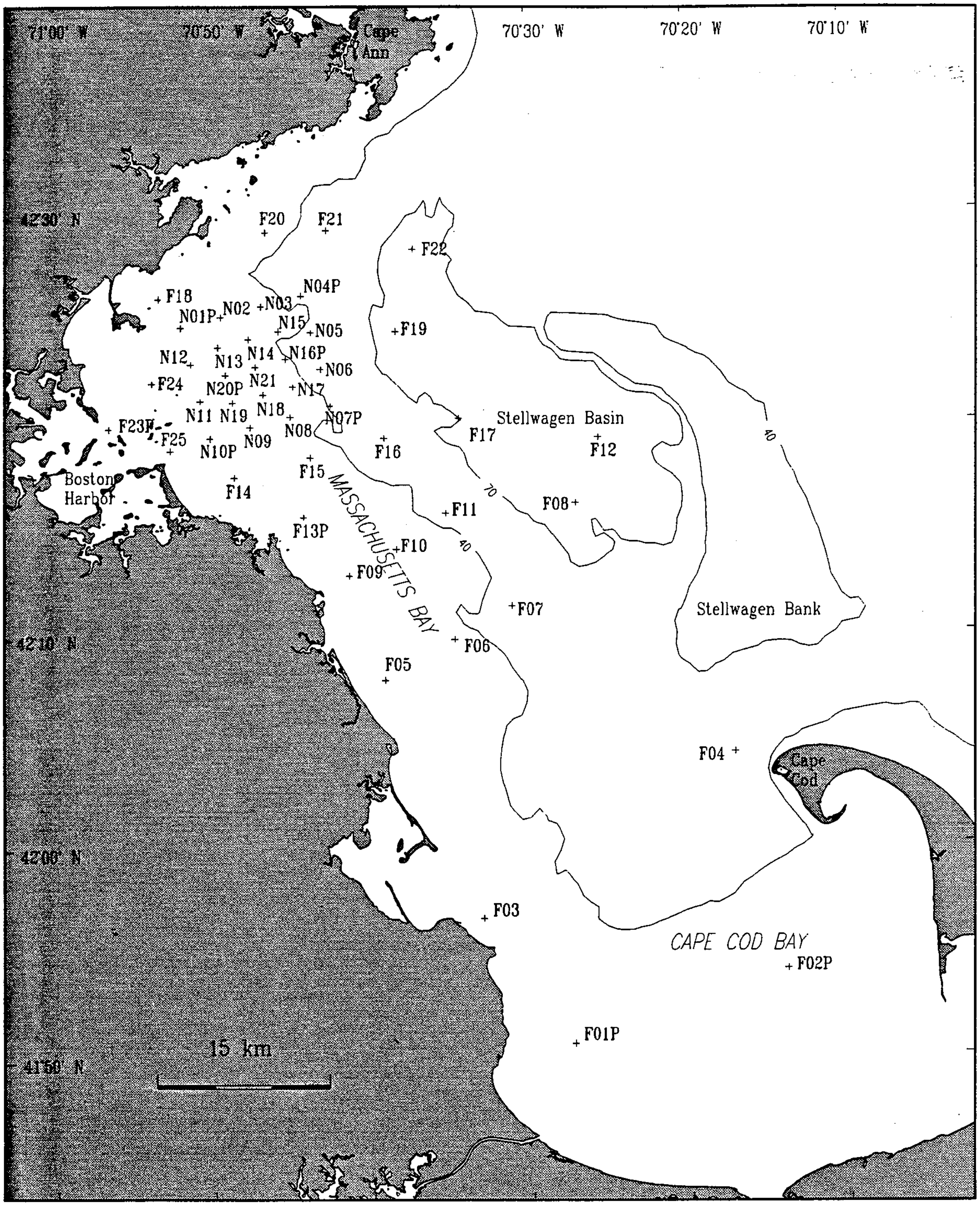


Figure 2-2. **Water quality sampling stations in Massachusetts and Cape Cod Bays. Station Codes - F: Farfield, N: Nearfield, P: Biology/Productivity. Depth contours are in meters.**

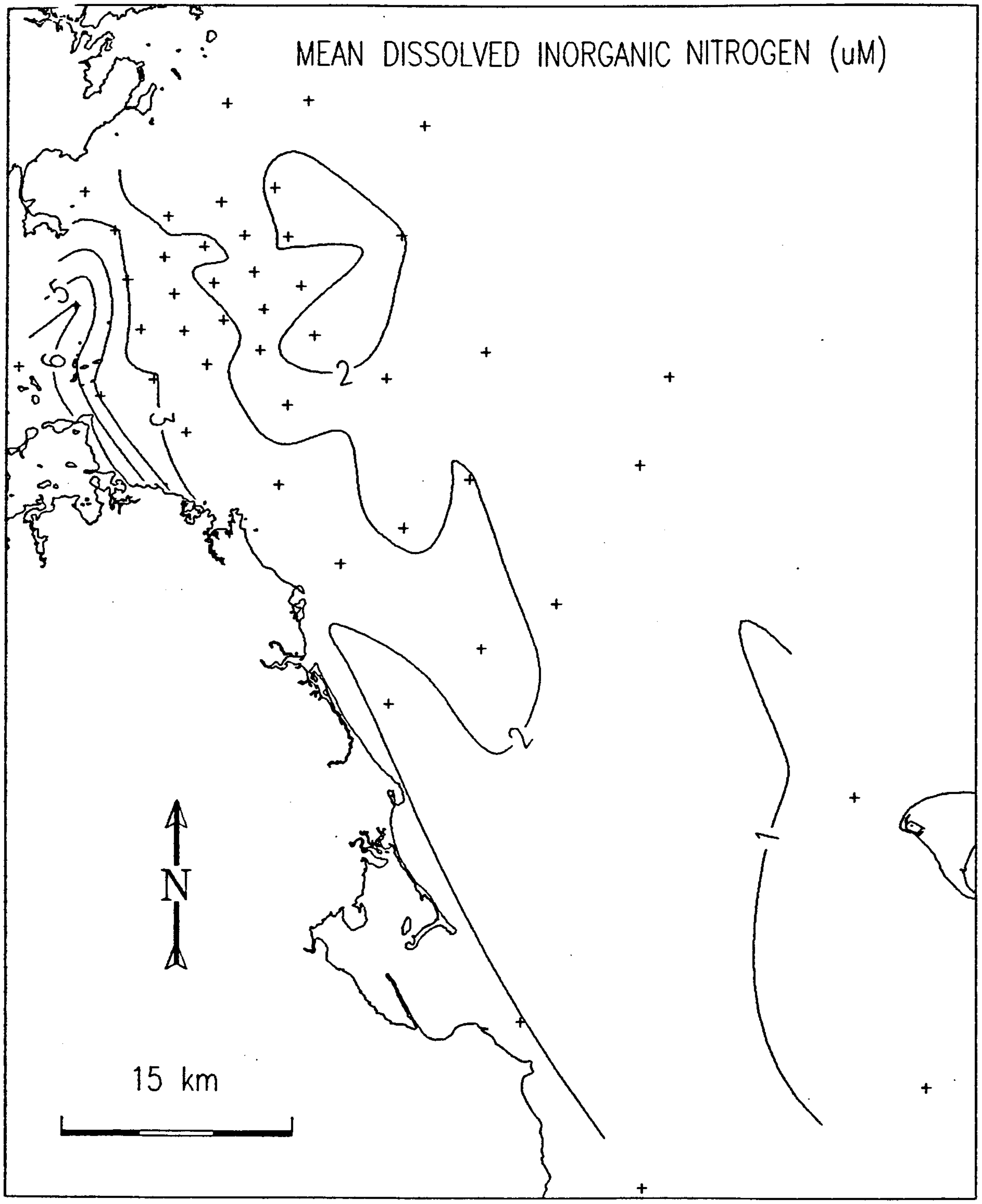


Figure 2-3. Annual surface average DIN in the Bays in 1992.

variability in major seasonal events, both in space (Cape Cod Bay vs. Massachusetts Bay) and in time (1989/1990 vs. 1992), influence annual mean concentrations that are based on six seasonal surveys. Regardless, the two annual data summaries compiled to date for western Massachusetts Bay depict a very similar gradient from Boston Harbor. Mean annual DIN concentrations decrease roughly 5-7 μM from the edge of the Harbor to the middle of the nearfield (Figures 2-1 and 2-3), with the gradient appearing slightly sharper in 1989/1990.

The 1992 MWRA monitoring design included select stations in western Massachusetts Bay where the strong DIN gradient exists. At these, the most comprehensive set of analyses of nitrogen forms to date was performed. Particulate nitrogen (essentially particulate organic nitrogen and hereafter referred to PON) and dissolved organic nitrogen (DON) were measured at eight "P" stations in the region and at station F25 seaward of Nantasket Roads, the southern exit from Boston Harbor to Massachusetts Bay (see Figure 2-2). The sum of DIN + PON + DON, termed total N (TN), thus could be calculated (Appendix). The gradient in mean annual N forms from Boston Harbor to western Massachusetts Bay region is further described in a series of graphs (Figures 2-4 to 2-7).

Mean annual surface PON concentrations show a decrease from Harbor to the nearfield and southward along the coast (Figure 2-4). A protruding lobe is suggested outside the southern Harbor (station N10P). There was a protruding lobe of DIN at station F24 outside the northern Harbor (Figure 2-3). Unfortunately, the full suite of N measurements was not made at station F24; but the smooth gradation depicted from the northern Harbor to the nearfield in Figure 2-4 (and subsequent plots) is probably more lobe-like in actuality. The general PON decrease from Harbor-edge to nearfield was about 3 μM in 1992; this compares with a slightly smaller decrease, <2 μM [i.e. about 20 μg/L, Figure 2-1] in 1989/1990. The relative decrease in PON concentration with distance offshore seems less abrupt than for DIN, a phenomenon that also was striking in the previous summary (Kelly, 1991). However, the pattern for PON + DIN for 1992 (Figure 2-5) is essentially the same as the

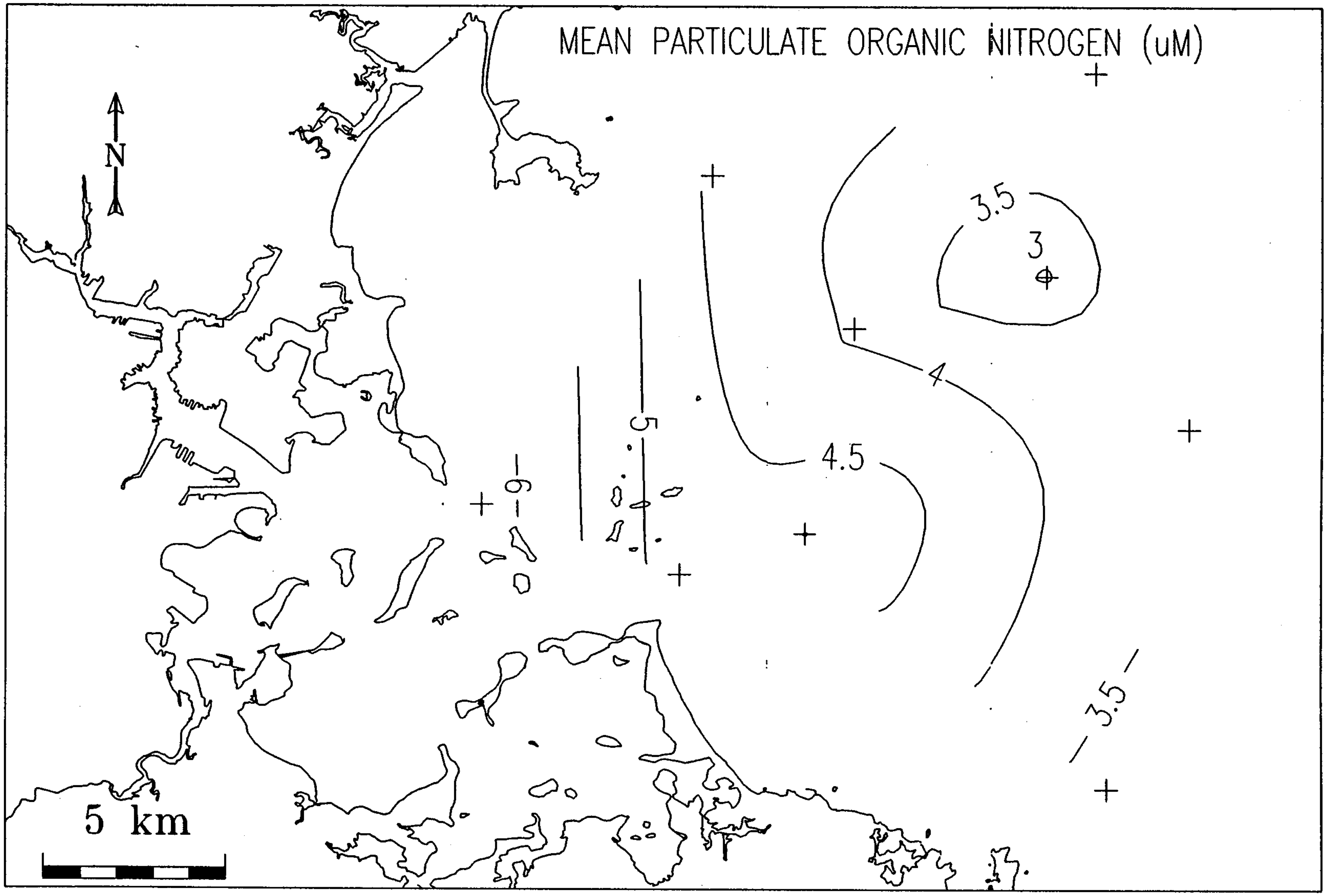


Figure 2-4. Annual surface average PON in western Massachusetts Bay in 1992.

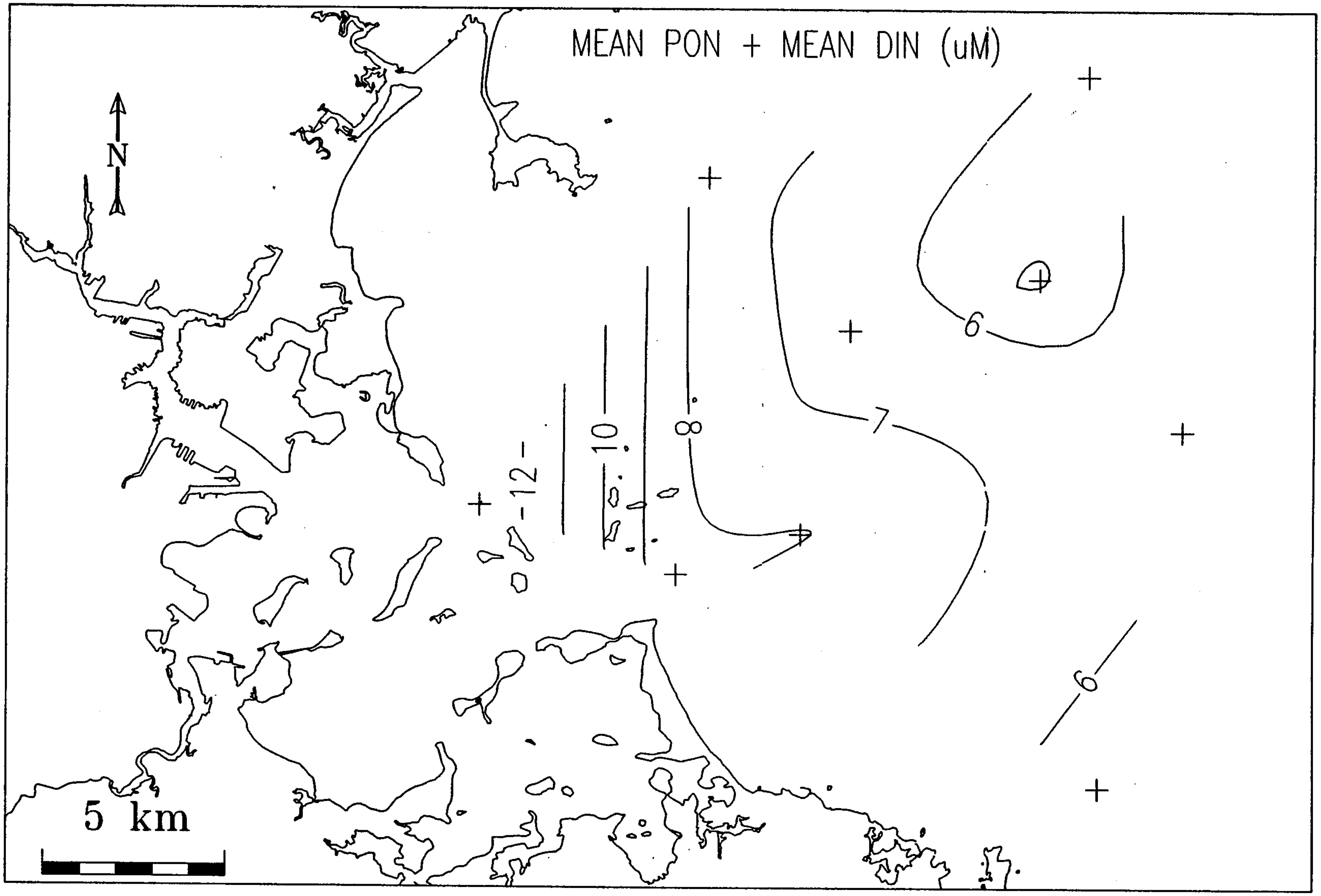


Figure 2-5. Annual surface average PON + DIN in western Massachusetts Bay in 1992.

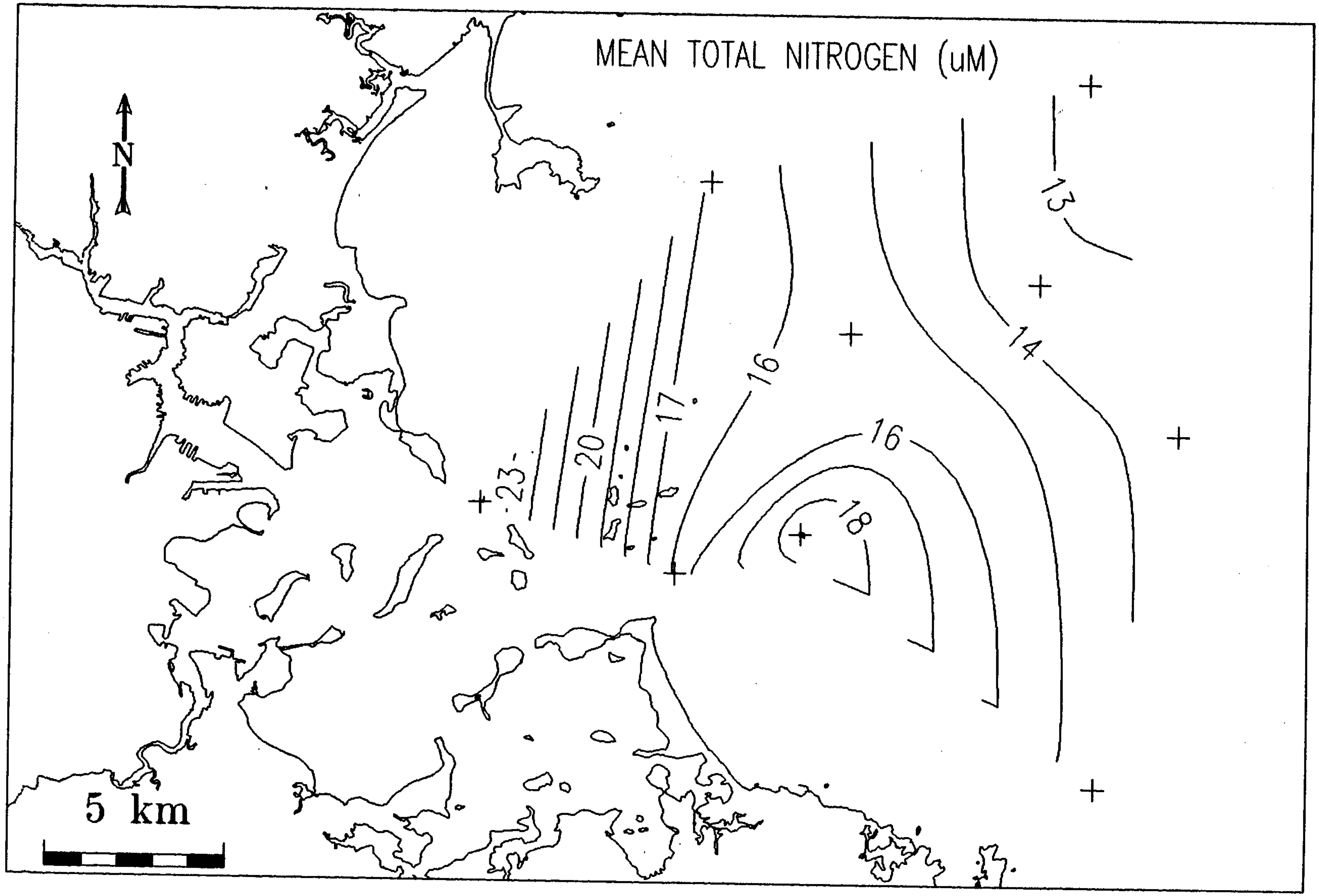


Figure 2-6. Annual surface average TN in western Massachusetts Bay in 1992.

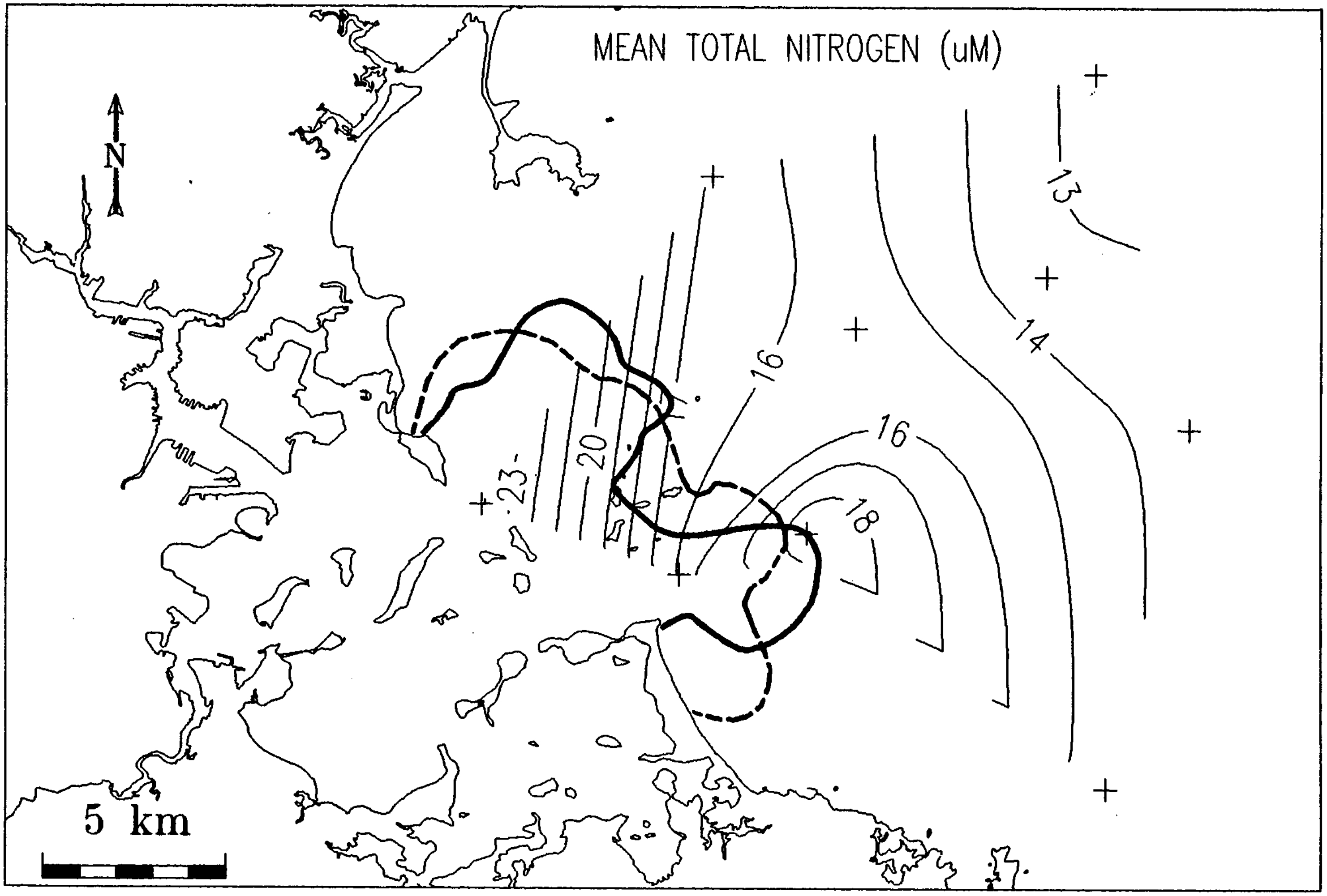


Figure 2-7. Annual surface average TN in western Massachusetts Bay in 1992. As Figure 2-6, except with suggested limits of tidal processes (see text). The solid line radiating from the Harbor suggests where tidal outflow carries water and the dotted line suggests where tidal inflow water to the Harbor comes from.

1989/1990 gradient (Figure 2-1). The decrease in PON + DIN from Harbor to nearfield in both cases is on the order of 7-8 μM.

The pattern for total nitrogen is similar to the individual forms (Figure 2-6). Rather than a lobe at station N10P, the contoured data depict a small local maxima. A decrease of about 10-11 μM total N occurred from the Harbor at President Roads (station F23P) to the east side of the nearfield almost 20 km away. Interestingly, the present U.S. Geological Survey (USGS) modeling of the Harbor and Bays suggests that physical dilution processes decrease nitrogen by about 10 μM from the edge of the Harbor to the nearfield (Signell, personal communication).

In Figure 2-7, the contours for TN are repeated, but overlain on the map are bold lines that radiate from the Harbor — these represent some geographical limits of tidal exchange suggested from modeling of Signell and Butman (1992). A bold, bi-lobed dotted line radiates from the two Harbor exit channels, which are separated by the string of outer Harbor islands. This line depicts, from the model, where seawater comes from as the Harbor fills during an average flood tide. The bold, bi-lobed solid line depicts where the ebb-tide water goes back out to the Bay. Signell and Butman (1992) point out how the ebb flow is more channelized and thus more jet-like; this results in a difference between flood and ebb flow that will produce net exchange (export) rather than just sloshing of the same water back and forth with the tide. Interestingly, station N10P appears to be at the ebb-jet limit, but outside the floodwater return limit. The same comments seem to apply to station F24, noted above. On average, concentration peaks occur near these stations and high-resolution studies have noted tidal fronts close to these locations (Kelly and Albro, 1994). The observations in the receiving system offer indirect evidence that the main features of tidal flushing are well-captured by Signell's model. The implication of such a conclusion strongly reinforces that rapid flushing occurs (Signell and Butman, 1992) with a consequence being high nutrient export.

The water quality monitoring has shown that advection of Harbor-type water off Nantasket Roads at station N10P (east of the southern exit of the Harbor to the Bay) occurs as a function of the stage of the tide. A combination of wind, density-driven circulation, *and* tides promote the actual advection, but a shallow surface lens of less dense seawater regularly has been observed east of station N10P into the nearfield (cf. Kelly *et al.* 1993a). Besides short-term variability, there are distinct seasonal trends in the distribution of different forms of N in the water of western Massachusetts Bay.

Seasonal aspects — winter versus summer 1992. Water quality gradients away from the Harbor varied by season. During the colder season, inorganic and organic nitrogen forms appeared to be mixed "conservatively" from the Harbor edge into western Massachusetts Bay (Kelly *et al.* (1992) and Kelly *et al.* (1993c)). For example, in February DIN (Figure 2-8) was correlated with salinity ($R^2 = 0.63$, $n = 45$), as was TN ($R^2 = 0.90$, $n = 15$). The concentration gradient from the Harbor can be explained by dilution of fresher, high-nutrient Harbor water into Bay seawater. Thus, in winter before phytoplankton growth typical of a winter-spring bloom is active, the data show that physical dilution processes largely control chemical distributions of nitrogen.

In contrast, with warm water temperatures of summer, plankton can utilize dissolved nutrients and "package" them into organic forms faster than physical mixing processes can act to regulate distributions. In August (Figure 2-8) there was a low, although significant ($p < 0.05$), correlation between DIN and salinity ($R^2 = 0.18$, $n = 30$). A more highly significant ($p < 0.008$) linear relationship was found between salinity with TN ($R^2 = 0.40$, $n = 16$). Besides individual surveys, for the summer on average (see Appendix Table 3) the stations from the Harbor to the nearfield had strong trends in organic N forms (PON and DON) with salinity (Figure 2-9). Thus, there is a general decrease in TN with increasing salinity as Harbor-Bay mixing occurs, which, as in winter, results in a distance-concentration gradient from the Harbor; these data support a conclusion that export during summer occurs primarily in organic forms rather than DIN.

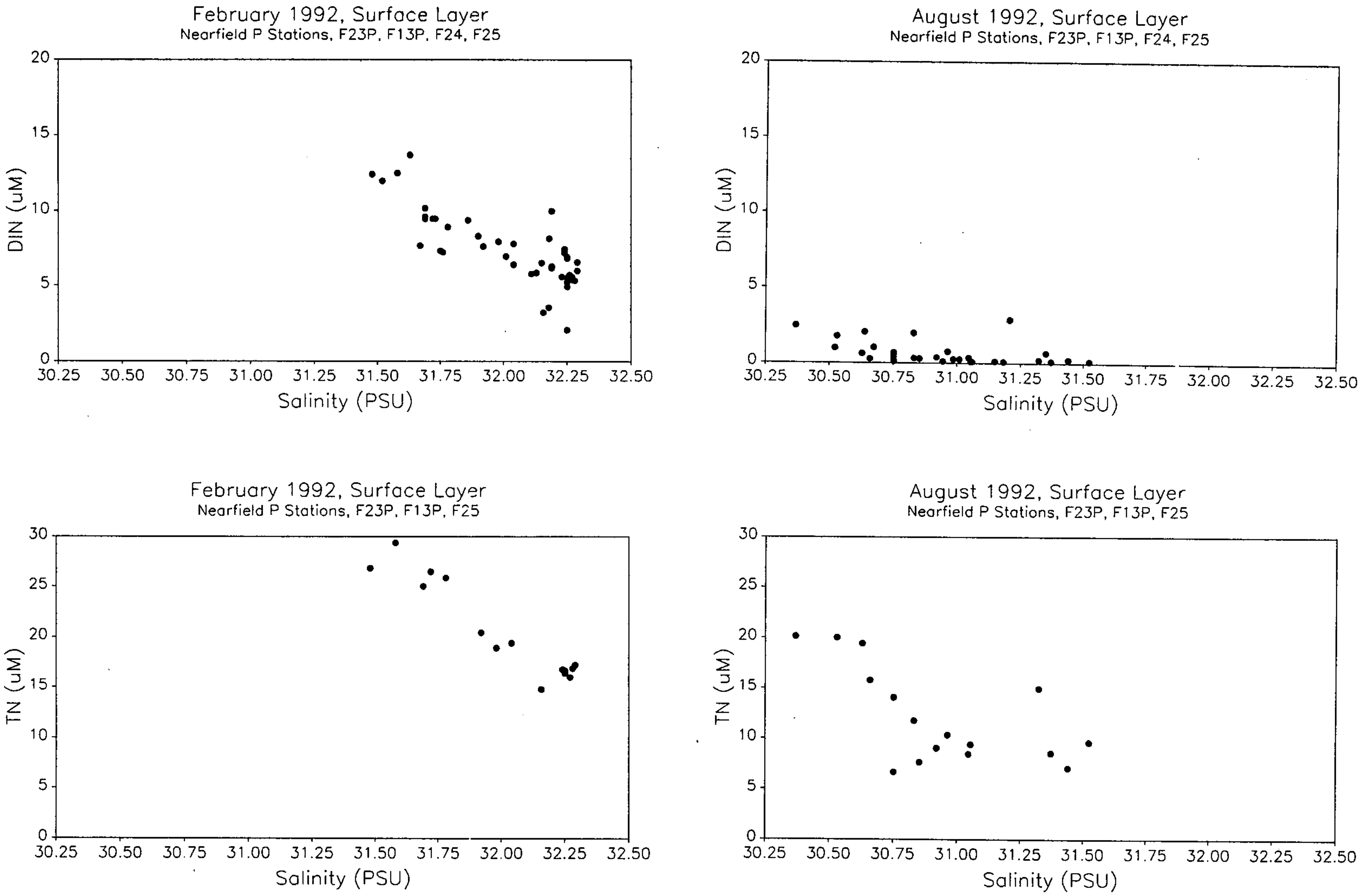


Figure 2-8. **DIN and TN in western Massachusetts Bay in February and August 1992. Data are all individual measurements at the indicated stations from the surface to the sub-surface chlorophyll maximum.**

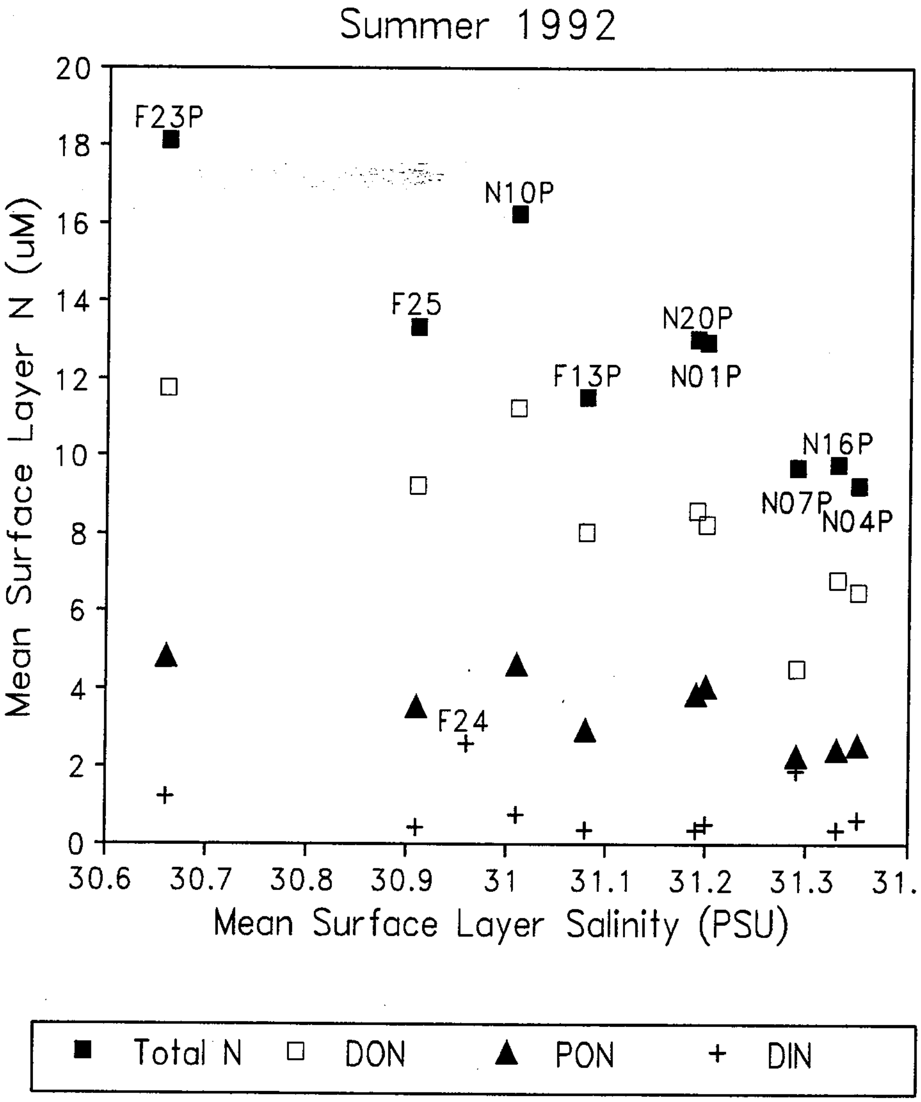


Figure 2-9. **Surface layer nitrogen concentrations in western Massachusetts Bay as a function of salinity. Means represent the average of June and August data for both surface and chlorophyll maximum samples at each station.**

The TN decrease from the Harbor to the nearfield area may be slightly sharper in winter (a TN decrease of about 13-15 μM) than in summer (a TN decrease of about 9-10 μM). This could imply more efficient nutrient export in winter, but there are too few data to assess this possibility. However, Figure 2-8 shows that there was a seasonal difference in TN concentrations at all stations (~10-15 μM). A lower TN in summer results from the seasonal decrease in DIN from winter to summer (Figure 2-8). The annual cycle of dissolved nutrient assimilation and net production in spring-summer, followed by net dissolved nutrient remineralization in late fall-winter creates this seasonal difference in the background DIN concentrations in seawater; this cycle, which temporarily carries N to sediments, happens independently of inputs to the Harbor and exchange with the Bay. In marked contrast to DIN, PON and DON concentration ranges, as well as their patterns with salinity (e.g., Figure 2-9), were similar between seasons, which leads us to an overall conclusion. Similar organic N export is suggested in winter and summer and the only the clear difference in receiving waters is a seasonal change in the gradient of DIN from the Harbor. The implication is that organic N dominates the N exported from the Harbor during the summer, whereas DIN export is high and dominant in winter.

The extent to which exported PON and DON, either in summer or winter, can be assimilated (or decomposed and then assimilated) by phytoplankton has not be assessed and their importance as an input to Bay nutrient cycles is not well characterized. However, direct assimilation of organic forms is slower than DIN, and the main pathway from organic N forms to uptake by phytoplankton may be through remineralization processes that convert organic to inorganic forms. By this means, organic export in summer could provide a time-release fertilization compared to inorganic enrichment.

Summary. In spite of much seasonal and short-term variability, the monitoring data reveal striking and consistent spatial patterns with respect to N in the water column in western Massachusetts Bay. Total nitrogen was correlated with salinity during both winter and summer, which suggests conservative dilution and dispersion into the Bay's surface waters and adds evidence to confirm rapid, efficient Harbor nitrogen export. From examination of different N forms, including relative concentrations and patterns with salinity, a seasonal difference in the dominance of inorganic (winter) vs. organic (summer) export of N is implied.

2.3 The Northern River Contribution to Massachusetts Bay and the Nearfield Region

Harbor budgets and gradients into the receiving waters identify Boston Harbor as a principal source of nutrients to western Massachusetts Bay. However, other inputs into Massachusetts Bay have been suggested as significant; in particular, input into the Bay across its northern boundary deserves mention in this review.

Menzie-Cura (1991) estimated sources of nutrients to Massachusetts and Cape Cod Bays. They calculated the *potential* input to the ocean by the Merrimack River was about 1.1×10^7 kg N/yr (equivalent to the 1.3×10^7 kg N/yr to the Harbor — Alber and Chan, 1994). If *all* the N carried by the Merrimack were transported south around Cape Ann and into the Massachusetts Bays, it would represent greater than 30% of the entire N input currently estimated.

"Loading" rates are difficult to estimate in any case, but a prime difficulty for this northern boundary case is that no estimate has been made of *actual* transport into the Bay. A second problem is conceptual: external loading, initially conceived and best applied to *relatively closed, well-mixed* water bodies with a defined hydraulic residence time (Vollenweider, 1975 and 1976), is not easily applied to a *large, open, and heterogeneous* system like Massachusetts Bay. For this reason, it is difficult to assess potential loading from northern boundary sources at scales that are meaningful. For example, the transport that occurs across the northern boundary may meander as a surface current over Stellwagen Bank and go out the Bay without ever interacting with the nearfield area in any way. Lastly, there is another is major issue relative to the possible influence of nutrient input: the significance, to ecological processes, of external loading compared to internal loading at the particular scale of interest. Making calculations for the Bay as a whole, even assuming the entire *potential* Merrimack contributed to the total N input to the Bay, all estimated external N sources would furnish only about 18% of the annual N needs for primary production[1]. Assuming the external loads are well estimated, the calculation implies that 72% of primary production needs are supplied by internal loading, i.e. recycling in water and sediments. Therefore, pelagic responses to nutrients in Bay waters, aside from areas near major point sources, will be less influenced by total Baywide external "loading" than they are by local and mesoscale processes (physical and biological) which regulate "internal" nutrient cycles (see also Kelly, 1991; Kelly and Levin, 1986). With these practical and conceptual problems highlighted, one must recognize that the influence of the Merrimack River on Massachusetts Bays' nutrient dynamics is very poorly known, no matter how well discharges from the Merrimack River into the ocean are quantified.

Knowledge of water quality at the most northerly water column stations in the 1992 MWRA monitoring program, relative to the nearfield, allows some insight on the potential influence of flow from the north into the nearfield. Throughout the year there were times when a shallow and slight fresher surface layer was observed in the northern area of Massachusetts Bay, and this could indicate transport of a body of river-influenced water. The average DIN within the surface 10 m for stations F20, F21, and F22 (the "Northern Transect", see Figure 2-2) was 1.55 μM for the year, with a concentration range of 0-5 μM (n=30). Sixty-three percent of the samples had less than 1 μM DIN. In contrast, the mean DIN concentration was less than 1.5 μM for only two of the twenty-one nearfield stations. Thus, the nearfield was generally more enriched in DIN than the northern stations; flow of water from the north into the nearfield would dilute higher dissolved nutrient water with lower nutrient water.

Unfortunately, there are no data on PON at the Northern Transect stations, and TN cannot be evaluated. However, the northeast corner of the nearfield, station N04P, was intended

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[1] The percentage is calculated using the input of 3.5 x 10^7 kg N/yr and the area of Massachusetts and Cape Cod Bays as 3,500 km^2 (Menzie-Cura, 1991), relative to an average net primary production of 300 gC m^{-2} yr^{-1}, assuming a C/N ratio of 6.625 (by atoms).

to serve as an indicator for waters moving from the north into the nearfield that is "upstream" of the outfall diffuser, and TN was measured here. Occasional transport of water from the north to this corner of the nearfield has been implied by examination of temperature, salinity, and other water characteristics (e.g., Kelly *et al.,* 1992). Figure 2-10 shows that TN at this station within the surface layer was consistently low compared to other "P" stations (refer to Figure 2-2 for locations). Of the 10 "P" stations in the Bays, Station N04P had the lowest average TN (12.2 μM) in 1992. As with DIN, the implication is that water from this corner flowing into the nearfield would generally dilute nearfield TN concentrations.

While nutrient concentration does not describe flux or loading (*sensu* Vollenweider 1975, 1976), these data certainly refute any notion that flow of surface water from the north would *enrich* nearfield N concentrations. Indeed, the gradient shown in Figure 2-9 implies overwhelming dominance of inshore sources, and Boston Harbor, on the present nearfield's nutrient concentrations. It may be demonstrated in the future that the northern rivers do play an important role in surface transport of water into Massachusetts Bay, but it is possible that they are more significant for transport of biological organisms than for nutrients (e.g. Franks and Anderson, 1992).

2.4 Scales of Change Expected with Proposed Offshore Outfall

The available evidence strongly and consistently supports the concept that most nitrogen discharged with MWRA effluent is exported from Boston Harbor (Kelly, 1991). From this, one may argue that a principal scale of monitoring is local rather than regional, because the nutrients now being discharged are circulated throughout the Bays, much as they will be in the future. With respect to the local scale, the monitoring data enable new understanding of nutrient dynamics in the nearfield area, as illustrated by the calculations below, focused on the stratified period.

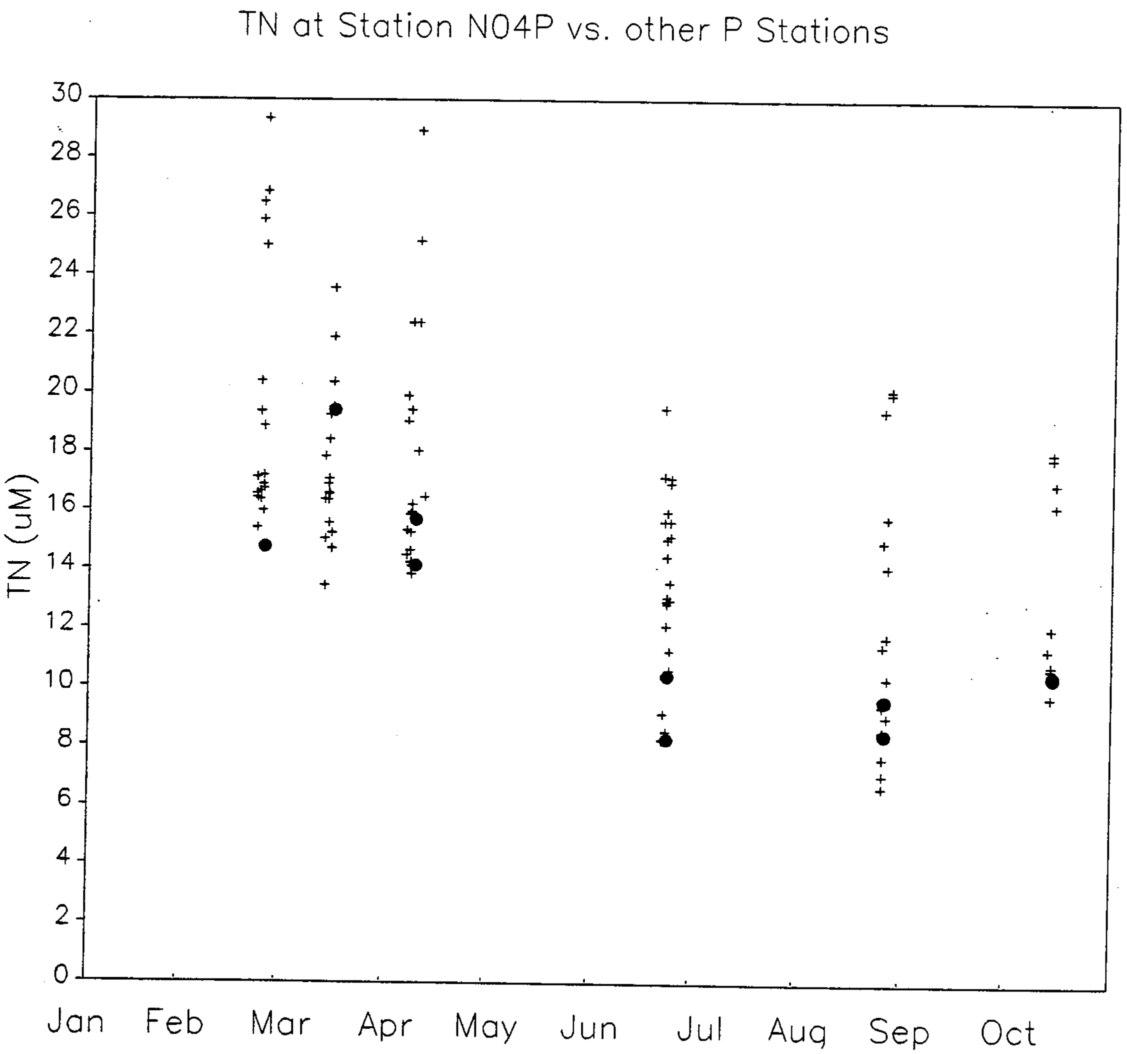

Figure 2-10. **Total nitrogen at "P" stations during 1992. Station N04P is indicated by the dark dots.**

The 1992 data suggest nearly conservative mixing of total nitrogen across nearfield surface waters during summer (e.g. Figure 2-9). While this implies a large horizontal nutrient flux it suggests little *net* vertical exchange as loss or gain of N to bottom waters during stratification. Note that to maintain the observed relationship with salinity, gross sedimentation of organic matter (PON) could occur if on average it were balanced by a continuous vertical diffusive flux of dissolved nitrogen across the pycnocline, or sporadic upwelling of bottom waters. Neither sedimentation nor upwelling is easily quantified, but the vertical diffusive flux can be estimated. Comparison of vertical and horizontal fluxes estimated for present and projected conditions next provides a strong sense of how nutrient dynamics in the nearfield may be altered with the effluent diverted directly to the nearfield.

A diffusive flux from bottom to surface waters is implied because there is a vertical gradient in DIN concentration within the nearfield during stratification. A simple calculation of the diffusive flux of nitrogen can be made, following a Fickian diffusion model and assuming a three-layer system (e.g., Okubo, 1971):

$$F = K\,(N_2 - N_1)/b \qquad \text{(Equation 2-1)}$$

where, F= vertical flux (μmol m^{-2} d^{-1}), N_2 and N_1 represent seasonal averages for DIN in surface and bottom waters, K is the coefficient of eddy diffusivity (cm^2 sec^{-1}), and b = thermocline thickness. Note that K is not molecular diffusion (which is much slower) and is difficult to measure accurately.

Using the samples bracketing the subsurface chlorophyll maximum, near the bottom and top of the pycnocline, $N_2 = 2.77$ and $N_1 = 0.55$ μM of DIN for the period of June to October 1992 at the nearfield stations. The mean sampling depths for N_2 and N_1 were 22 m and 7 m, respectively. Thus, b was generally less than 15 m and a range of 10 to 15 m will be assumed. Studies of shelf waters suggest K values of about 0.03 to 0.44 cm^2 sec^{-1}, using a variety of estimation methods (cf. Sharp and Church, 1981). Note that K is not molecular diffusion (which is much slower) and is quite difficult to measure. Geyer *et al.* (1992) estimated K of 0.11-0.14 cm^2 sec^{-1} for relatively stagnant bottom water in Stellwagen Basin. One might expect a higher K in the nearfield, which is more dynamic. Dye studies by Rocky Geyer (WHOI) were conducted to quantify this process during August 1993; results are not completed, but K seems lower than 0.1 cm^2 sec^{-1} for the few days of the study (Geyer, personal communication). To calculate a reasonable average upper bound for vertical flux during the stratified season, K was assumed to range from 0.1 to 0.2 cm^2 sec^{-1}. Using this range, the diffusive (i.e., not including advective upwelling) nutrient flux[2] from nearfield bottom waters to the surface layer could be 127-384 μmol N m^{-2} d^{-1}.

An equivalent, balancing sedimentation flux would account for only 3% of the approximate 1 g C m^{-2} d^{-1} (~ 12.5 mmol N m^{-2} d^{-1}) primary production occurring during the period (Cura, 1991; Kelly *et al.*, 1993). A value closer to 20% might be expected for seasonally stratified coastal systems of similar depth (e.g. Hargrave and Phillips, 1986). It is also likely that vertical advection (upwelling) infuses additional nutrients to the surface layers. The presence of fronts, surfacing thermoclines, and noticeable expansion/contraction of the thermocline thickness and position within the nearfield during summer 1992 (Kelly *et al.*, 1993a) suggests some bottom water venting. Moreover, Geyer *et al.* (1992) stressed that their surface temperature data suggested that upwelling and disruption of stratification occurs on a sporadic basis.

For comparison to vertical flux, assume that during the stratified period, as much as 88% of the 1992 input to Boston Harbor (cf. Kelly and Nowicki, 1993), or 2.16 x 10^9 mmol N d^{-1}, is exported[3]. Export occurs to surface waters extending into at least the middle of the

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[2] The estimated flux range is derived from Equation 2-1, using N_1 =0.55 μM and N_2 =2.77 μM. The lower estimate of 127 μmols N m^{-2} d^{-1} is derived from K=0.1 cm^2 sec^{-1} and b=15 m (i.e., slow diffusion across a wide boundary layer) and the higher estimate of 384 μmols N m^{-2} d^{-1} is derived from K=0.2 cm^2 sec^{-1} and b=10 m (i.e., faster diffusion across a thinner boundary layer).

[3] Input is 1.3 x 10^7 kg N yr^{-1} or slightly below 9 x 10^{11} mmol N yr^{-1}. Divide by 365 days and multiply by 88% to get 2.16 x 10^9 mmol N d^{-1} as a potential horizontal flux.

nearfield (about 175 km^2). Thus[4], the average horizontal surface flux for this region would be 12.3 mmol N m^{-2} d^{-1} — more than 30 times the calculated diffusive flux but equivalent to the primary production demand (above). If one assumed that only 50% of the Harbor N was exported to the nearfield, the associated average horizontal flux across the area would be 6.2 mmol N m^{-2} d^{-1}, which is still more than 16 times the calculated diffusive flux.

These simple calculations suggest that the present export from the higher-nutrient waters inshore is the major nutrient flow into nearfield surface layers. A fundamental change in the delivery of nutrients will take place with diversion of effluent to the Massachusetts Bay outfall since nutrients will be input to bottom waters. Under stratified conditions, the present horizontal flux will be strongly limited because the source to the Harbor will be curtailed, but the vertical flux in the nearfield will increase. If the difference in TN ($\leq$15 μM) between the Harbor edge and the nearfield is indicative of the maximum increase in nearfield bottom waters, the diffusive flux calculation (Equation 2-1) can be repeated. The result is a diffusive flux[5] on the order of 1-2 mmol N m^{-2} d^{-1}. When stratification is persistent, this flux will be a major input to the surface waters; but note that it represents a flux that is only about 10-16% of the horizontal flux to the surface waters that occurs now.

Summary. In the present situation, MWRA effluent-derived N flows into the nearfield area surrounding the future diffuser. It is delivered directly to the surface layers, especially during the stratified period, and therefore is available to the plankton. In the future, the initial dilution will be faster than at the edge of the Harbor and, thereafter, the delivery to the productive surface layers will be more highly constrained by vertical density gradients that restrict exchange. Therefore, monitoring and research efforts should focus on understanding local ecological changes, predicting response to the nutrient flux changes such

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[4] 2.16×10^9 mmol N d^{-1}, divided by 175×10^6 m^2 = 12.3 mmol N m^{-2} d^{-1}

[5] Using Equation 2-1, with N_2-N_1 = 15 μM, K=0.1 cm^{-2} sec^{-1}, and b=15 m, then F= 987 μmol m^{-2} d^{-1}.

as are estimated here, and on how well the monitoring program will be able to determine both local and regional change in the Harbor and the Bay.

3.0 Influence of Nutrients on Chlorophyll

Chlorophyll, a measure of phytoplankton biomass, is one of the most variable parameters in the MWRA monitoring program. In spite of its variability, chlorophyll is a prime indicator of the response of phytoplankton to nutrients. After a brief review of time-space variability of chlorophyll observed during 1992, this section explores patterns that may be derived between chlorophyll and nutrients on annual and seasonal timescales.

Kelly (1991) presented a summary of average conditions of chlorophyll and dissolved inorganic nitrogen (DIN) in Massachusetts Bay surface waters. Perhaps because of preoccupation with the issue of Harbor flushing and nutrient export, there has been little emphasis on his observation that the average chlorophyll concentrations in western Massachusetts Bay were not high and, moreover, that only modest changes in chlorophyll might result from the new outfall. A main objective of this section is to use monitoring results to focus a discussion on the chlorophyll response expected with effluent diversion to Massachusetts Bay.

3.1 Spatial and Temporal Scales of Chlorophyll Variability

At fine scales of measurement, chlorophyll distributions often have a high degree of patchiness. Patches exist at the scale of meters to kilometers over both horizontal and vertical space throughout the nearfield (e.g., Figure 3-1). Kelly and Albro (1994) used high-resolution profiling to describe how some small-scale chlorophyll variability at the western edge of the nearfield may be due to tidal action, resulting in water and material exchanges between the Harbor and the Bay. The 1992 monitoring program has also shown that fluctuations in concentrations of chlorophyll from near zero to bloom levels can be detected at fixed locations over periods of hours to days.

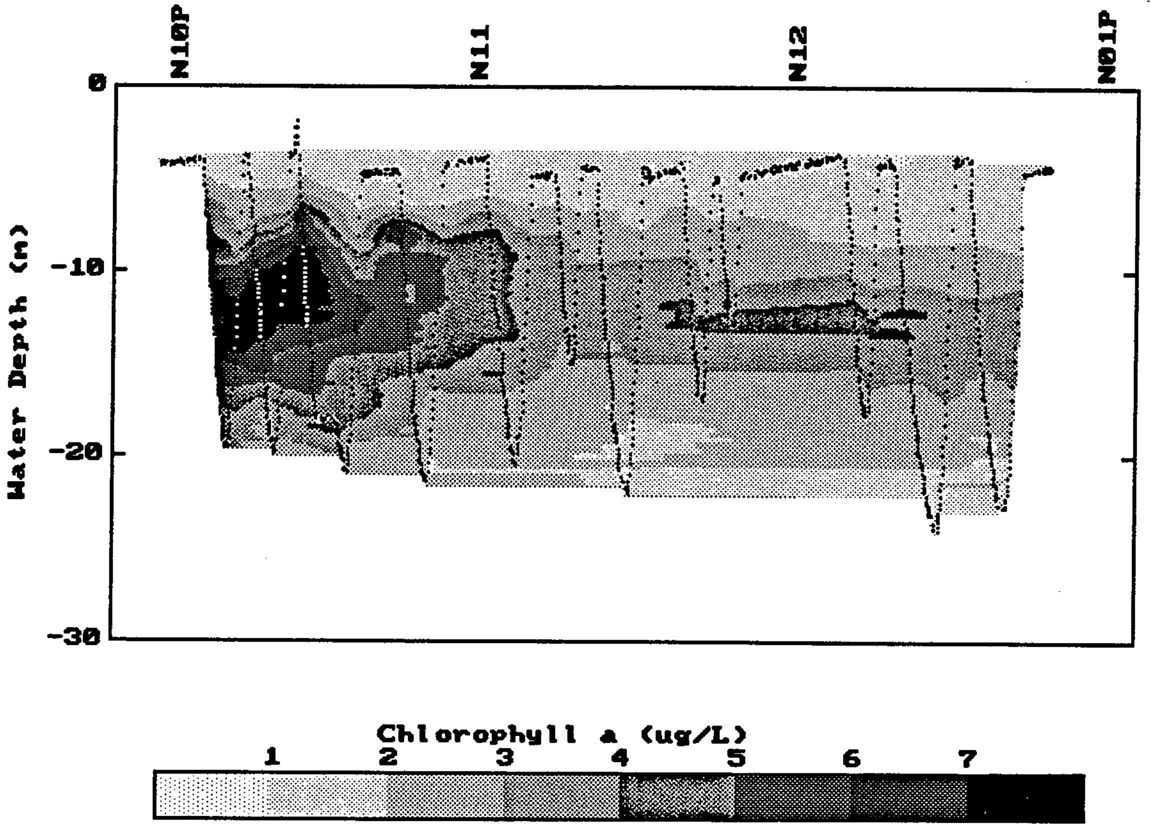

Figure 3-1. An example of chlorophyll variability from Station N01P to Station N10P in mid-July 1992. The transect is about 10 km in length.

At intermediate scales of observation, monitoring has demonstrated that chlorophyll concentration gradients from shore are persistent in some areas of the Bay. For example, shallow inshore stations (particularly near Boston Harbor) and deeper offshore stations characteristically have different annual cycles (e.g., Figure 3-2). Differences in part reflect differences in the general level and seasonal availability of nutrients as a function of proximity to nutrient sources in conjunction with mixing of the water column. In shallow coastal areas near an inshore nutrient source, the surface layer often has the highest summer chlorophyll concentrations, whereas in deeper, more stratified water, highest chlorophyll concentrations are frequently found within or near the pycnocline.

At a geographic scale, the broadest included in the monitoring program, the 1992 chlorophyll data show clear differences between Massachusetts and Cape Cod Bays, some of which parallel differences in nutrient concentrations (cf. Kelly *et al.,* 1993c). For example, the winter-spring peak in chlorophyll occurred earlier, was more intense, and more prolonged in surface waters of Cape Cod Bay stations than at stations in Massachusetts Bay. The prolonged spring bloom virtually depleted both nitrogen and silicate in Cape Cod Bay but not silicate in Massachusetts Bay. From April through October, surface chlorophyll concentrations were consistently higher in Massachusetts Bay, particularly western Massachusetts Bay, than in Cape Cod Bay. In spite of seasonal differences in near-surface trends, both Bays experienced sporadic *mid-water* chlorophyll blooms during the summer stratified period. Both Bays also had a sharp autumn peak in chlorophyll as water column stratification weakened, but before complete re-mixing occurred.

One way to observe gradients and patterns over the backdrop of high local variability, broad inshore-offshore trends, and regional-level distinctions is to examine averages over time and space. Indeed, the monitoring plan was designed, in its frequency and timing of sampling, to collect data suitable for calculating annual and seasonal averages. Using the 1992 data, annual means for chlorophyll (using chlorophyll *a*-calibrated fluorescence readings) were calculated from surface readings (about 2-5 m deep) at all 46 stations occupied on the sixfarfield/nearfield surveys (see Appendix Table 1).

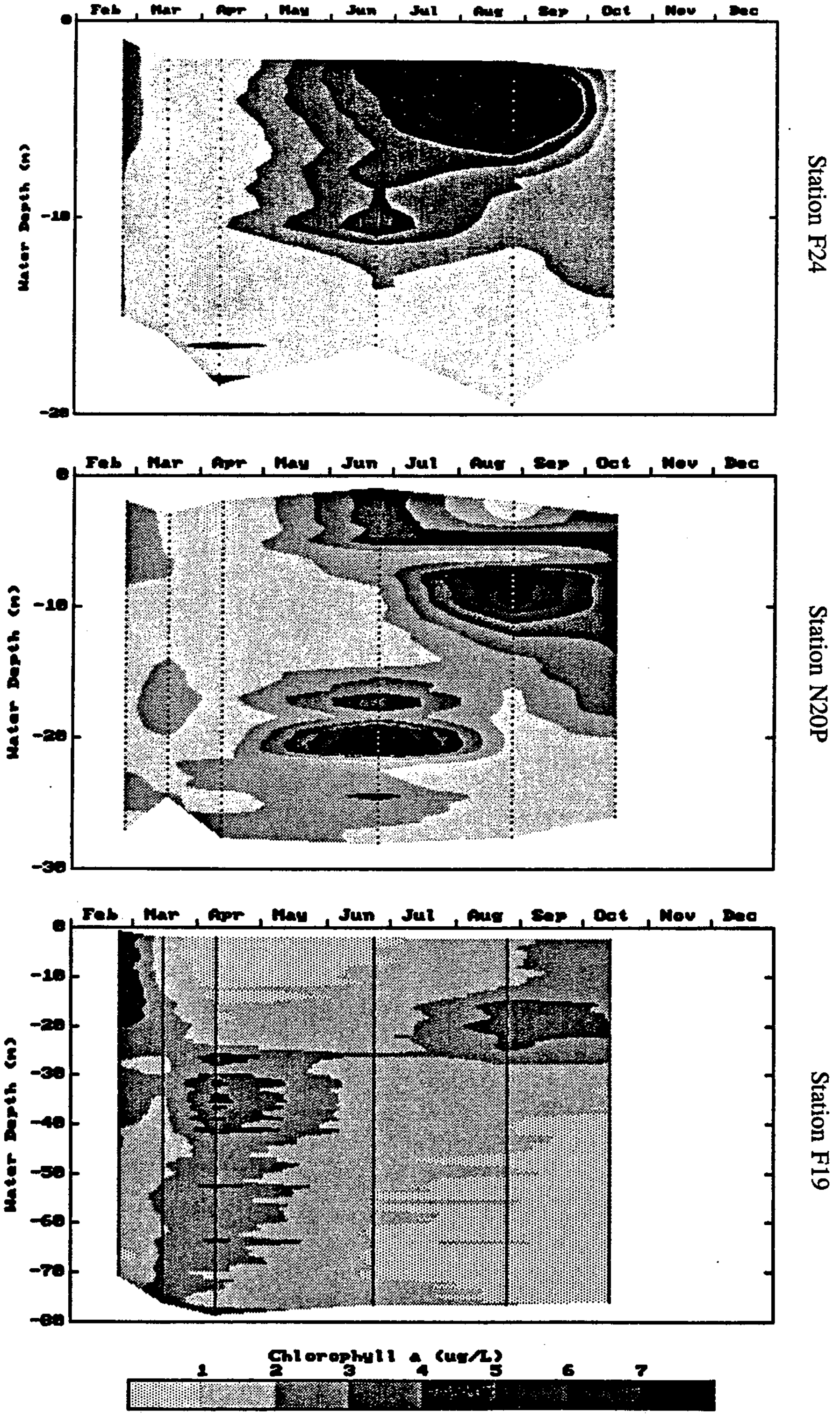

Figure 3-2. Seasonal cycle of chlorophyll along a transect from Broad Sound to Stellwagen Basin. Note different depth scales. [Source Kelly *et al.* 1993c]

The range for 1992 annual surface averages was about 1.4 to 4.3 μg/L. This range is similar that calculated for the surface layer in 1989/1990 from the data of Townsend et al. (1991) by Kelly (1991) [see Appendix]. Figure 3-3 shows the Bays-wide chlorophyll pattern for 1992, as contoured over all nearfield and farfield stations. The greater small-scale variability implied for the nearfield can be attributed to a higher density of stations in the nearfield and is not necessarily a reflection of variability in western Massachusetts Bay compared to other areas. Highest average chlorophyll concentrations occurred westward of the coastal arc from Nahant Bay to Cohasset. On average, surface chlorophyll concentrations at Cape Cod Bay stations were similar to those of the nearfield stations, in spite of the striking differences in timing of seasonal production events that was mentioned above.

Focusing on the western Massachusetts Bay region emphasizes a gradient in chlorophyll concentration with distance from the Harbor (Figure 3-4). Highest average concentrations are associated with locations that are strongly tidally regulated (stations F23P and F25) or lie within the zone of direct tidal influence (stations F24 and N10P). Outside the area of the direct tidal influence, on average there was an apparent plume from the western edge of the nearfield almost to the middle of the field (Figure 3-4). A variety of individual observations have identified patches, if not a coherent plume, of water that are enriched in chlorophyll, and found along the western edge of the nearfield; a body of data support the concept that patches represent release of water temporarily entrained in the Harbor-Bay tidal ebb and flow cycle (cf. Kelly *et al.,* 1993a; Kelly and Albro, 1994). The exact location and intensity of such chlorophyll-enriched water parcels varies with season, winds, and perhaps other factors. Thus, the average picture of a plume may be misleading; for example, chlorophyll plumes extending several kilometers southward along the coast have been suggested at times. Finally, both Nahant Bay (station F18, see Figure 2-3) and the northwest corner of the nearfield off Nahant (station N01P) had relatively high average chlorophyll values. The occurrence near Nahant Bay may relate to something other than tidal dynamics, such as upwelling or a coastal nutrient source from the north.

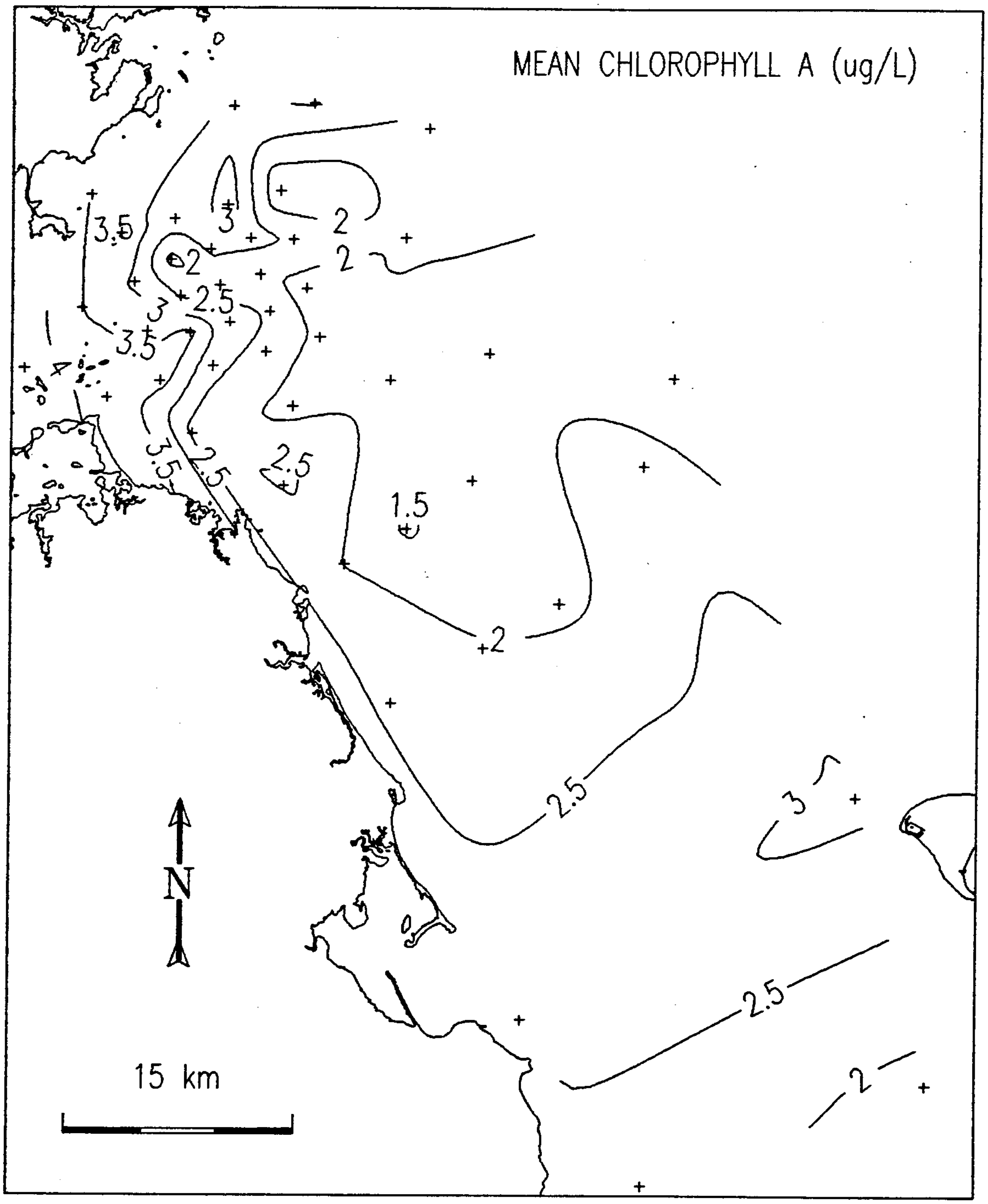

Figure 3-3. Annual surface average chlorophyll in the region in 1992.

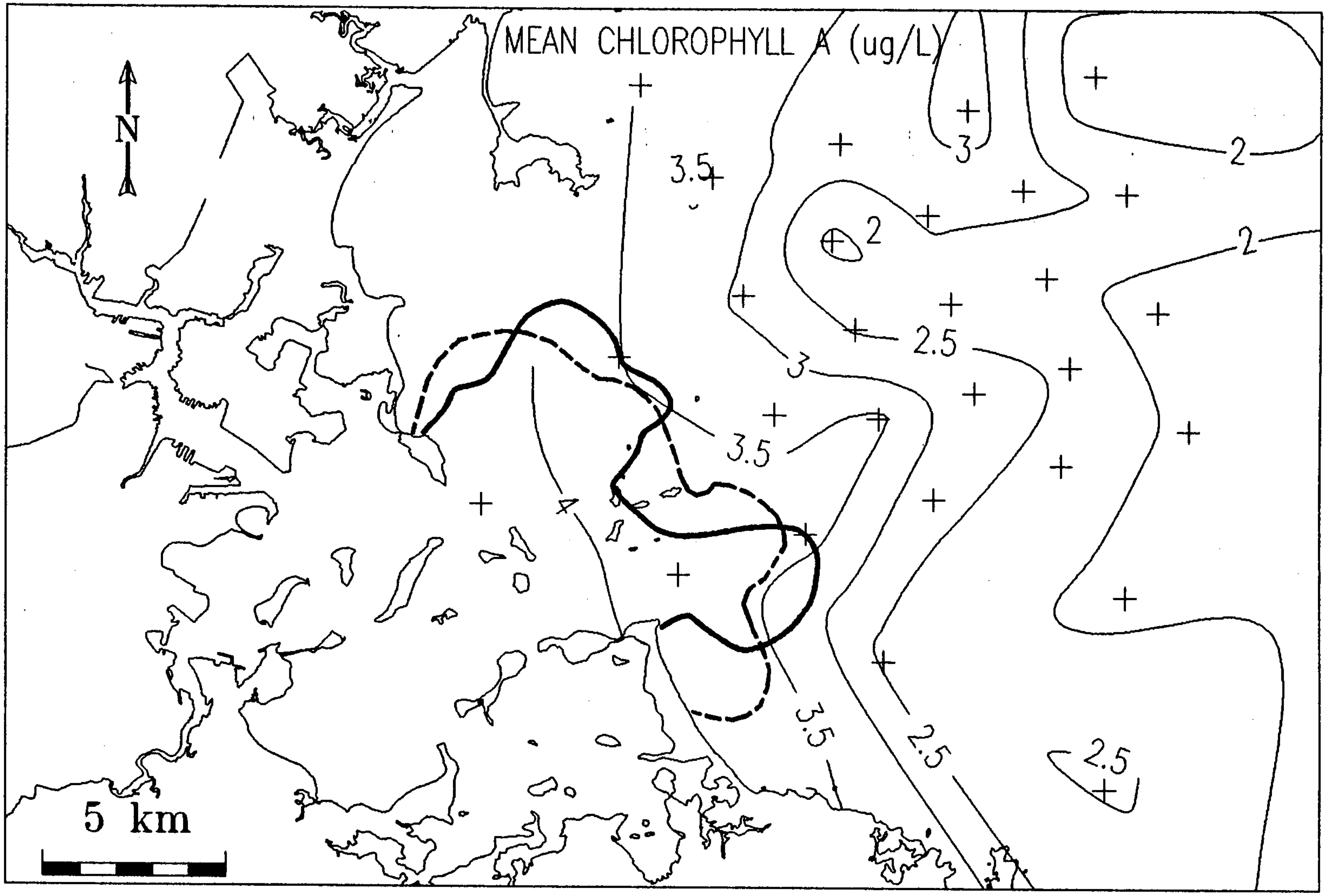

Figure 3-4. Annual surface average chlorophyll in western Massachusetts Bay in 1992. Solid and dotted lines are as in Figure 2-7.

3.2 Annual Surface Water Nutrient—Chlorophyll Relationships in the Bays

A prediction by Kelly (1991) that annual mean chlorophyll would rise *as much as* 1 μg L^{-1} for each 1 μM increase in annual mean DIN concentration is revisited here using 1992 survey data[1]. The prediction was not based solely on data from Massachusetts Bay, and relied more on field data from a variety of other natural systems, as well as experimental data on marine eutrophication studies in the MERL mesocosms. The range of concentrations encompassed by those studies was from very oligotrophic to quite eutrophic, spanning annual mean DIN concentrations from <1 μM to about 400 μM and annual mean chlorophyll concentrations from about 0.5 to nearly 80 μg L^{-1}.

For all stations sampled during combined nearfield/farfield surveys in 1992 (n=6), the mean annual surface chlorophyll and DIN concentrations were compared (Figure 3-5). The figure shows much scatter, but the results broadly parallel those derived with the Townsend *et al.* (1991) data (cf., Figure 10 of Kelly, 1991) and in particular repeat the observation that stations near the Harbor (in this case, stations F23P and F24) have chlorophyll well below that predicted by a 1:1 trend. The pattern for 1992 data shows slightly higher chlorophyll "yield" at a given DIN concentration than observed using the averages derived for 1989/1990. This makes the 1992 data more consistent with data previously compiled for other shallow marine ecosystems (cf. Figure 12 of Kelly, 1991). Interestingly, those stations in this 1992 summary which seem to have especially higher "yield" (e.g. chlorophyll above 2 μg L^{-1} at DIN <2 μM) include all four Cape Cod Bay stations, as well as several in Massachusetts Bay north of the nearfield — all of which had higher chlorophyll than the nearfield—Harbor area early in the year in February (Kelly *et al.*, 1992).

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[1] A 1:1 increase conforms to common observations on chemical composition of marine plankton and is consistent with monitoring data. Assuming a Redfield ratio for C/N (6.625, by atoms), then 1 μg Chl L^{-1} per 1 μM N would predict an average C/Chl ratio of 80:1 (by weight). POC/PON trends from monitoring data are consistent with the Redfield model. POC/Chl ratios are variable, but the range includes 80:1.

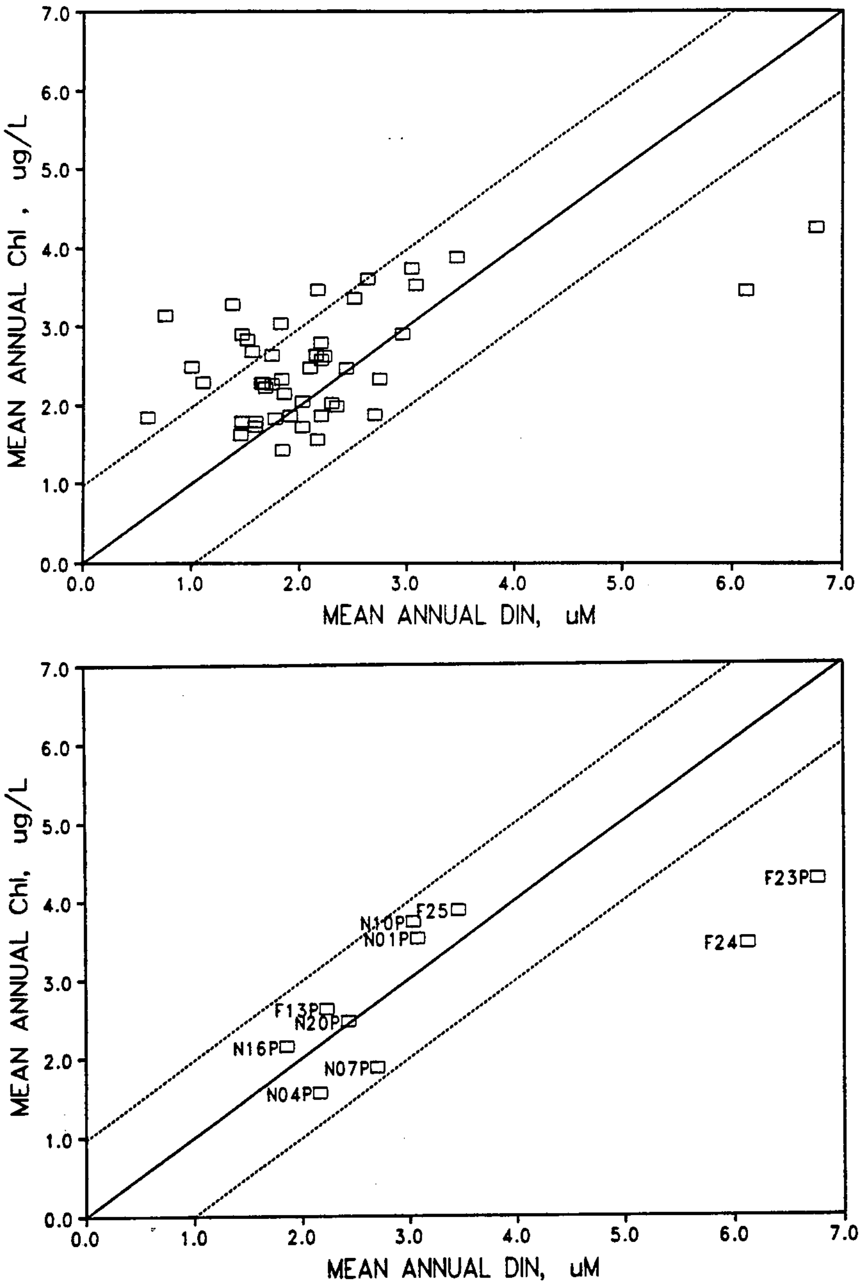

Figure 3-5 Annual surface average chlorophyll vs. DIN. Top: All Massachusetts and Cape Cod Bay Stations. Bottom: Select stations in the Harbor - nearfield region. Solid and dotted lines on both figures depict 1:1 isopleths where the rise in chlorophyll would be 1 $\mu g\ L^{-1}$ for a 1 μM rise in DIN.

A regression for chlorophyll on DIN for all station data in Figure 3-5 was significant at the 95% level (R^2= 0.30, n= 46). The slope of the regression was 0.34 ± 0.08 (± stnd. error). The slope was largely determined by the two near Harbor stations (F23P and F24) with higher mean DIN concentration. Inclusion of near-Harbor stations in 1992 (as well as in 1989/1990 — Kelly, 1991) suggests that a chlorophyll rise due to higher nutrients would be far less than 1:1. Excluding those two stations there was a small range in chlorophyll and DIN and a weak correlation between them (R^2= 0.15, n= 44).

A more striking pattern was observed between chlorophyll and DIN (Figure 3-5) when data was limited to the select stations of the Harbor-nearfield area where additional wet chemistry is conducted (see below). In this case, the regression (without stations F23P and F24) was significant at the 95% level (R^2= 0.66, n= 8), with a slope estimate not different from 1 (1.31 ± 0.38, ±stnd.error). Stations F23P and F24 fell well below this trend and their inclusion lead to a weaker correlation (R^2= 0.49, n= 10) and lower slope (0.39 ± 0.14).

At the select group of stations in Figure 3-5, chlorophyll concentrations were examined relative to other forms of nitrogen. Significant linear relationships with PON and Total N (TN) were found (Figure 3-6). In both cases, station F23P fell slightly below the trend line for the other stations; station F24 was omitted because organic N was not measured there. Without station F23P, the regression for chlorophyll and PON was significant at the 95% level (R^2= 0.66, n= 8), with a slope of 1.0 (±0.29, stnd. error). Including station F23P, the slope was not significantly lower (R^2= 0.72, n= 9, slope = 0.81 ± 0.19). The relationship between chlorophyll and PON thus suggested that roughly 1 μg L^{-1} chlorophyll is created for each 1 μM N assimilated into tissue (Figure 3-6).

The relationship between TN and chlorophyll differs from DIN, but it also may be a more robust predictor of chlorophyll than DIN. Chlorophyll and TN had the highest correlation for all the N forms at the select group of stations (R^2= 0.73, n=8 without station F23P; R^2= 0.71, n=9, with station F23P) (Figure 3-6). The slope estimated for chlorophyll and

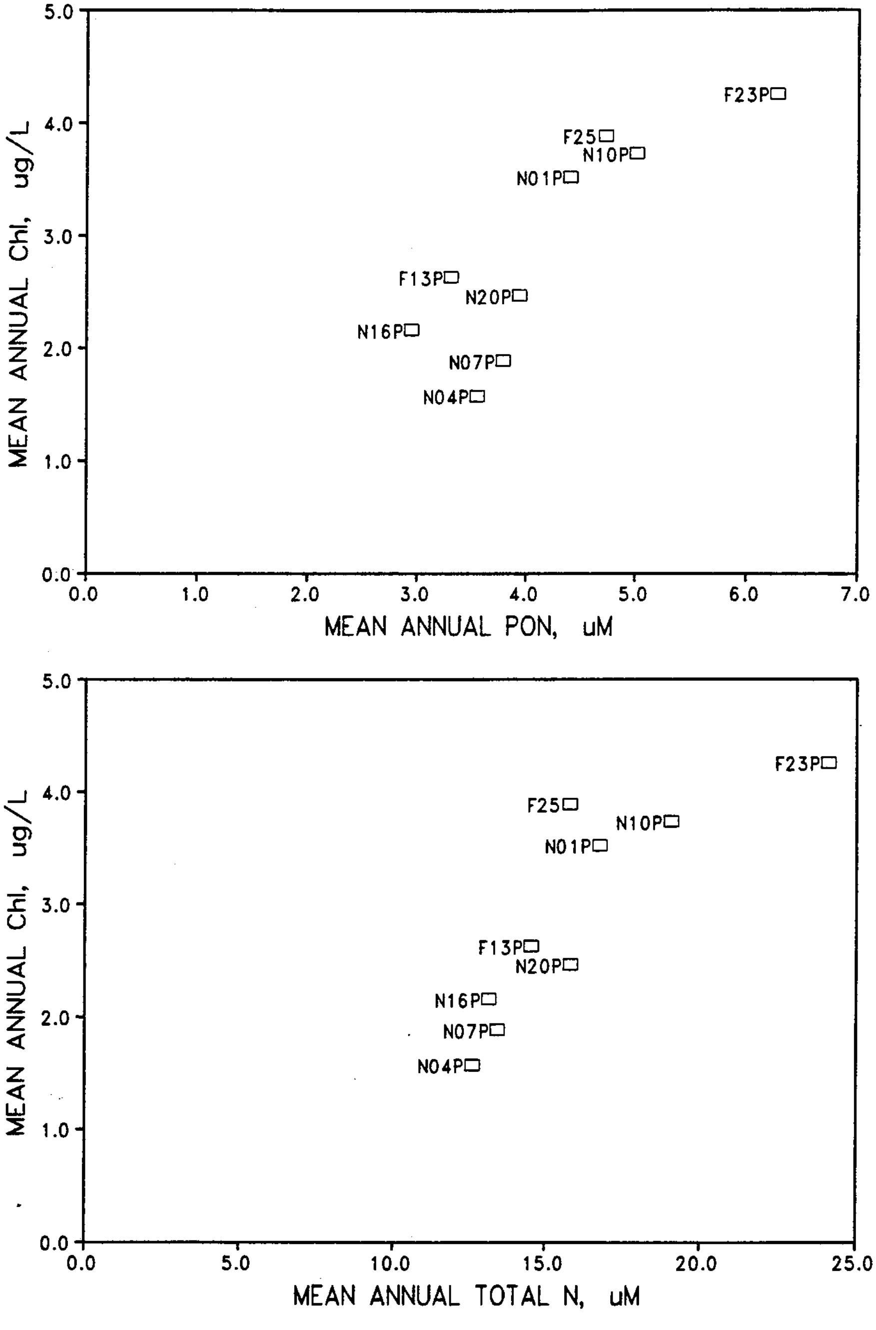

Figure 3-6 **Annual surface average chlorophyll vs. nitrogen forms for select stations in the Harbor - nearfield region. Top: Chlorophyll vs. PON. Bottom: Chlorophyll vs. total N.**

TN was low: 0.35±0.09 (stnd. error) without, and 0.23±0.05 (stnd. error) without station F23P. High frequency fluctuations (relative to sampling) between active chlorophyll growth and DIN concentrations could create some scatter in annual means due to timing of measurements; by using TN and incorporating that labile N within organic forms, such effects may be partially buffered. TN includes both PON and DON; only a portion of DON is likely to be used and recycled by plankton. DIN and DON, having sharper concentration gradients than PON across the stations, strongly influence (and lower) the overall slope of TN and chlorophyll. In any event, TN was strongly related to chlorophyll, the pattern may be useful as a predictor, and the premise that chlorophyll may rise no more than about 1 μg L^{-1} for a nitrogen increase (DIN, PON, or TN) of 1 μM is applicable.

Reviewing the above regression analyses, the inclusion of high-nutrient concentration Harbor-edge stations often lowered the slope estimate and decreased the R^2 value. This finding obviously suggests that the two higher-nutrient stations are not fully described by the empirical relationships derived for the remaining stations. Interestingly, their inclusion affected cases when dissolved nutrients were included in the N form, but not the case with PON alone; this additionally suggests that the deviation at stations F23P and F24 may be more from altered linkages between dissolved nutrients and chlorophyll than from a fundamental change in the chlorophyll/PON ratio. Thus, while significant regressions were obtained using data for all stations, their predictive capacity for enriched conditions is uncertain because the higher nutrient conditions presently occur at locations that may have different factors limiting chlorophyll.

Before describing some thought experiments to predict the magnitude of chlorophyll changes in the nearfield, it is helpful to have a short examination of environmental controls relevant to the difference between shallow near-Harbor locations and nearfield waters. The basic question is why station F23P generally does not reach chlorophyll levels that could be expected for its nitrogen level. There are a number of factors that can affect the relationship between chlorophyll and nutrients, e.g., grazing, light, and toxicity (cf. Kelly, 1991). From the monitoring program, we have no information on grazing or toxicity, but we do know that a light gradient exists along the nutrient-chlorophyll gradient from the Harbor. Townsend *et al.* (1991) advanced the concept that chlorophyll was limited by both nutrients and light along the gradient from the Harbor into the Bay — nutrients being more limiting offshore and light, because of increased turbidity, being more limiting inshore. The 1992 transmissometry readings (attenuation of a beam of transmitted light over a 25-cm pathlength), which are correlated to total suspended solids (and thus turbidity), indicated a gradient from the Harbor to the Bay (Figure 3-7). Coincident with this gradient is a general increase in vertical mixing as well as the depth of the euphotic zone (where light is sufficiently high to allow phytoplankton growth, usually taken from the surface to the depth having 0.5 to 1% of surface-incident photosynthetically active radiation). 1992 data confirm that the euphotic zone graded from about 10 m or less at the edge of the Harbor to 25-30 m at the eastern edge of the nearfield. Moreover, results of Kelly (1993) and Kelly et al. (1994) confirm other empirical formulations (e.g., Cole and Cloern, 1987) which provide strong indirect evidence that variations in integrated water column primary production and chlorophyll biomass in the Harbor-Bay region of western Massachusetts Bay are related to light availability. In sum, the notion that chlorophyll near the Harbor is, in part, restricted by light is a viable hypothesis.

To address the interaction of nutrients and light a sophisticated approach involves explicit modeling of light, nutrients, etc. as linked, complex mathematical functions that are based in physical and physiological theory and have a variety of assumptions. The approach generally decrements a theoretical "maximum" production or biomass for sub-maximal conditions of the various factors. Such an approach has been adopted in developing a water-quality model for the Bay (Hydroqual, 1993). Simulations being conducted will soon provide predictions for a number of effluent discharge scenarios (MWRA, personal communication); these will be interesting to compare against the patterns and simple linear regression model results presented here. A simple approach used here jointly considered the light (beam attenuation as proxy, Figure 3-7), chlorophyll, and nutrient gradients to see if one could describe data from all stations by one set of rules. For the 1992 station summary, a composite variable, the average chlorophyll multiplied by average beam attenuation

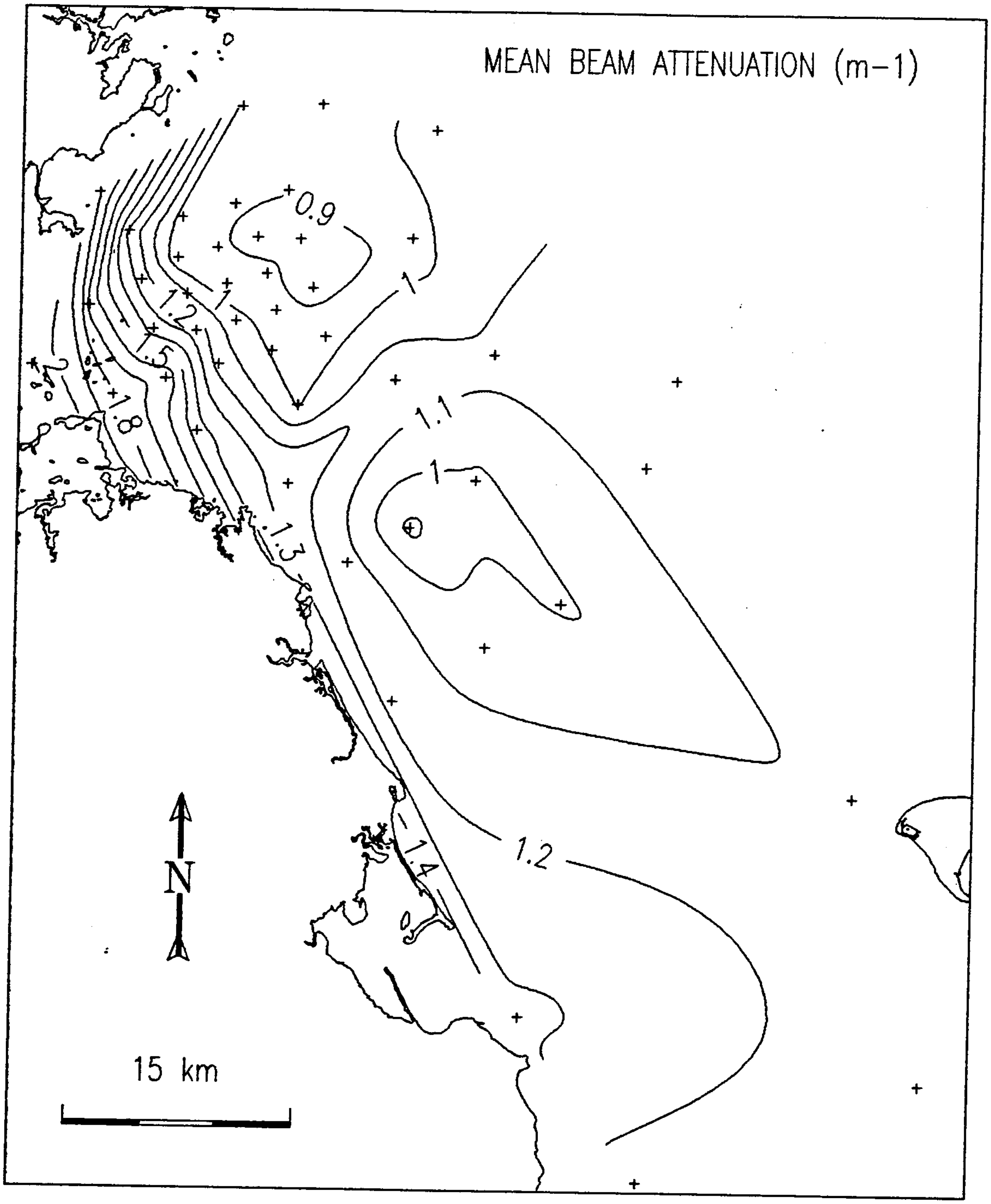

Figure 3-7. Annual surface average beam attenuation in the region in 1992.

(turbidity), was related strongly to average nitrogen concentrations. For DIN, using all stations (n=46, R^2=0.52) or TN at select stations (n=9, R^2=0.79), linear regressions were stronger using the composite variable than chlorophyll alone. In contrast to the analyses above, when stations F23P and F24 were included, each regression had a higher R^2 but no change in the estimated slope. This finding suggests that all stations may indeed follow some generalizable rules, but a principle attractiveness of the formulation lies in the inclusion of a relative turbidity term, which is hypothesized to be the essence of why F23P is lower in chlorophyll. The composite variable suggests that if nitrogen in the nearfield increased, but turbidity did not, a higher chlorophyll would be predicted than if turbidity increased also. This formulation absolutely needs testing against more data as they become available and also needs founding relative to theory; nevertheless it is used a series of thought experiments that next provide a range of predictions of chlorophyll concentration in the nearfield in the future.

There are two parallel series of thought experiments. The first assumes an unamended transfer, to the surface water in the middle of nearfield, of the concentration of DIN or TN measured in 1992 at the Harbor edge. If no light or other compensation actually exists for the present environment near the Harbor (relative to the Bay), the average chlorophyll concentration at F23P would directly provide a prediction (4.26 μg L^{-1}). If regression models for the select stations without F23P are used (see Figure 3-6), the predicted range[2] is 5.9 to 8.1 μg Chl L^{-1}. If regression models with the composite variable are used, the predicted range is 3.74 to 4.43 μg Chl L^{-1} if nearfield turbidity increased to the level of station F23P, or 8.3 to 9.83 μg Chl L^{-1} if turbidity remained at the level presently measured at the middle of the nearfield[3].

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[2] Based on least squares linear predictive regression models for DIN and TN from select stations: Chlorophyll = 1.31 (DIN) - 0.72 and Chlorophyll = 0.35 (TN) - 2.55. Chlorophyll was predicted using nutrient concentrations of station F23P, DIN = 6.77 μM and TN = 24.11 μM.

[3] Based on the least squares regression model for DIN (n=46) and the model for TN at select stations (n=9) as described in text. The models are: (Chlorophyll * Beam attenuation) = 1.06 (DIN) + 0.71 and (Chlorophyll * Beam attenuation) = 0.65 (TN) - 6.33. Calculated ranges are based on DIN= 6.77 μM, TN= 24.11 μM, and Beam attenuation= 2.11 m^{-1} (F23P) or 0.95 m^{-1} (nearfield).

The second thought experiment simulates the outfall diversion: move the nutrient source, but increase dilution and during summer, trap nutrients below the thermocline, while curtailing the present nutrient export to nearfield surface water from the Harbor (see Section 2). A variety of modeling and calculations suggest there will be a substantial increase in effluent dilution (conservatively, a two- to four-fold increase for the nearfield on average compared to the present immediate receiving area) and trapping below the thermocline will reduce surface summer concentrations further (see Sections 2 and 5). For the calculation, assume that average nearfield nutrient concentrations are 1/4 of station F23P; then DIN would be the same as the mid-nearfield at present, about 1.75 μM. Assume TN will be similar to mid-nearfield concentrations also, about 14.5 μM. Using the same chlorophyll-nutrient regressions as above, the predictions of the average chlorophyll are 1.57 to 2.53 μg Chl L^{-1}, or within the present range of the nearfield. Using the composite variable, the predicted range is 2.7 to 3.25 μg Chl L^{-1} if turbidity is unchanged or 0.83 to 1.5 μg Chl L^{-1} if turbidity increases to Harbor-edge levels.

Although simplistic, these thought experiments provide a conceptual guide and numerical bounds of prediction of average annual chlorophyll concentrations. Removal of apparent light limitation at the Harbor would by itself allow increase chlorophyll, but as embodied in the composite variable formulation, increased turbidity and self-shading from increased plankton could moderate chlorophyll increases. If there were sustained chlorophyll levels as high as 8-9 μg Chl L^{-1} — still not high compared to eutrophic systems (e.g. Kelly, 1991; Nixon *et al.*, 1986) — these would be easily detected. While such a 400% increase in surface chlorophyll concentration in the nearfield is possible, it seems less likely than a slight decrease on average. With future surface nutrient concentrations likely to be less than near the Harbor now, the nutrient-chlorophyll response, even with lessening of light limitation, could be difficult to detect.

In summary, even with the present strong nutrient source and a sharp gradient in nitrogen concentrations away from the Harbor, the accompanying chlorophyll gradient, although shown clearly in the data, encompasses a limited range in average chlorophyll concentrations and there is only a factor of 2-3 difference from minimum to maximum (Figure 3-4). A chlorophyll enrichment, even at the most nutrient-enriched monitoring location at the edge of the Harbor, is persistent and detectable, but it is not large relative to the overall time and space variability in chlorophyll. In other words, the apparent nutrient-induced signal-to-noise ratio in this parameter is low. Neglecting the nutrient-enriched stations F23P and F24, the remaining patternless cloud of points in Figure 3-5a is testament to a low signal-to-noise ratio. This observation is undoubtedly promulgated by the data summary, which used the farfield sampling frequency (six per year) for all stations. Indeed, high-resolution profiling and extensive hydrocast bottle sampling in space and time within the nearfield region were instituted with recognition of this signal-to-noise issue. Appropriate statistical analyses of the extensive monitoring data are planned for the purpose of defining the detectable scales of change for chlorophyll and will be reported elsewhere, but patterns derived from frequent sampling in the nearfield during 1992 next illustrate some resolvable trends.

3.3 The Nearfield Region in the Summer Stratified Period

The stratified season is of special interest because bottom waters become sealed from the atmosphere and dissolved oxygen may decrease, in part as a consequence of surface production of organic matter that settles to the bottom. Also, this season is when nitrogen presently exported from the Harbor appears to be dispersed directly into surface productive layers of the nearfield.

The nearfield itself is of special interest, not only for its expected response to the upcoming changes in effluent discharging (above), but also from the perspective of the monitoring's sampling design. Frequent nearfield sampling was intended to provide a higher power to detect change. Thus, it is of substantial interest to investigate whether the higher frequency and intensity of measurements in the nearfield provide a picture of variability that seemingly represents more "noise" than "signal" of response to the gradient of conditions from the Harbor.

Temporal variability in the nearfield was described in the annual report for 1992 (Kelly *et al.,* 1993c). Briefly, the water column was only weakly stratified in April, a thermal gradient had begun by May, and a fairly stable stratification was evident from June to mid-October. There were seven intensive nearfield surveys during this stratified period (June, mid-July, late July, mid-August, late August, mid-September, mid-October). For the period, mean concentrations for various parameters were calculated for the surface "layer" (surface to subsurface chlorophyll maximum) for each of the twenty-one stations (Appendix Table 2). A subsurface chlorophyll maximum was usually within the pycnocline and was found at about 12 m inshore and 17-18 m offshore.

Inspection of the nearfield data summarized for the stratified period reveal some striking spatial trends. The average surface-water chlorophyll concentration uniformly decreased by a factor of two from inshore to offshore (Figure 3-8). Values were slightly higher in the middle of the western transect of stations closest to shore (stations N11 and N12), suggesting influence from the President Roads area by way of the North and South Channels leading from there to Broad Sound.

Regardless of the mean chlorophyll concentration, minima <1 μg Chl L^{-1} were detected everywhere during the period (Figure 3-9). In contrast, maximum chlorophyll values increased as a function of increasing mean chlorophyll concentration. This feature was also reported in controlled nutrient enrichment experiments (e.g., Nixon *et al.,* 1986). While this point may seem obvious, quantifying the relationship between maxima and means is not trivial because ecosystems can be sensitive to extreme (i.e., maxima) conditions and undesirable events like anoxia may result from episodic events as well as from the overall trophic status implied by mean conditions. Both statistics (mean, maximum) are of interest; the frequency of summer sampling captured a strong sense of maximum events and provided meaningful average values.

The inshore-offshore nearfield data for the stratified period showed a linear relationship between chlorophyll and salinity (R^2= 0.73, n= 21) even though the range of mean salinity

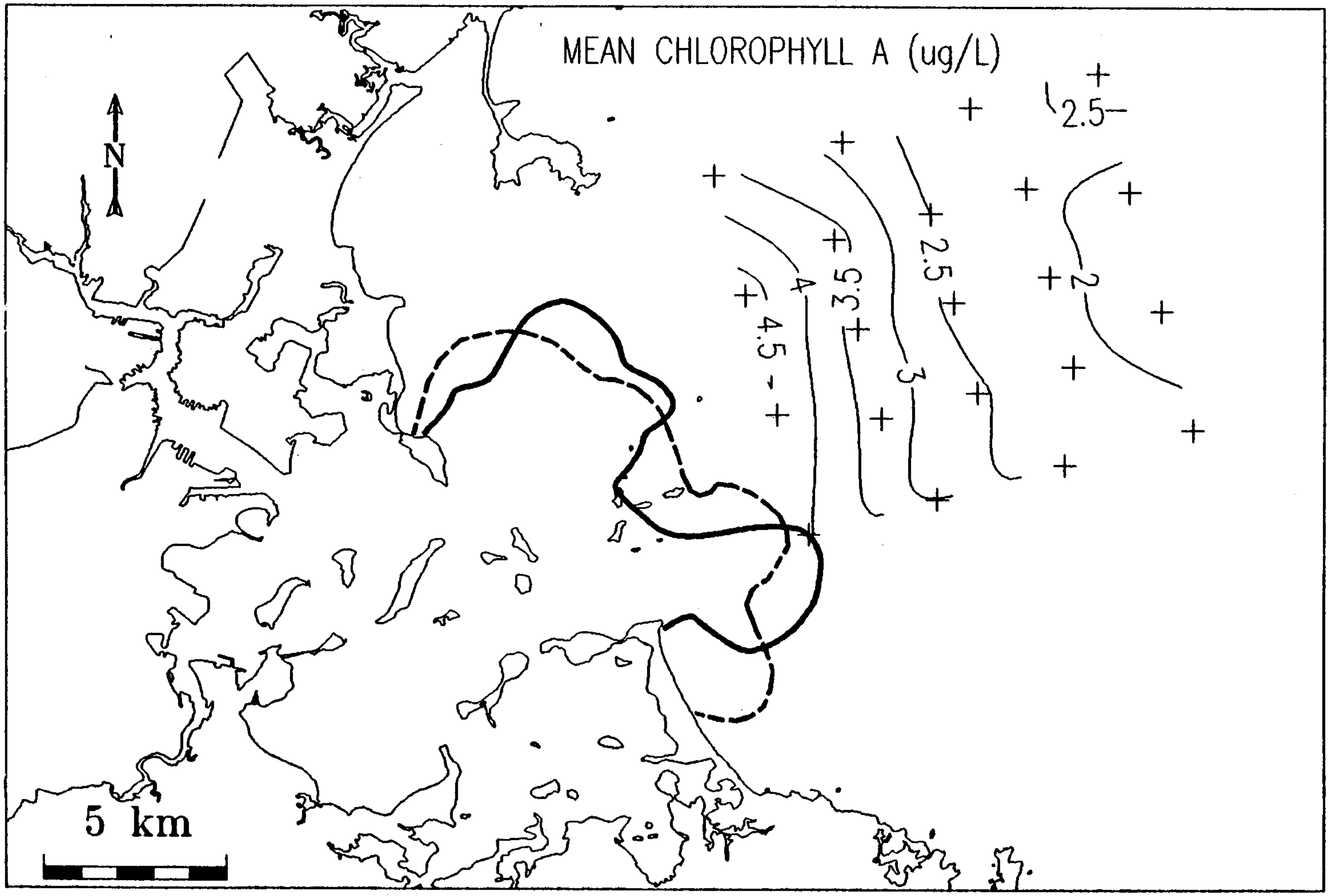

Figure 3-8. Summer surface layer chlorophyll in 1992 at nearfield stations.

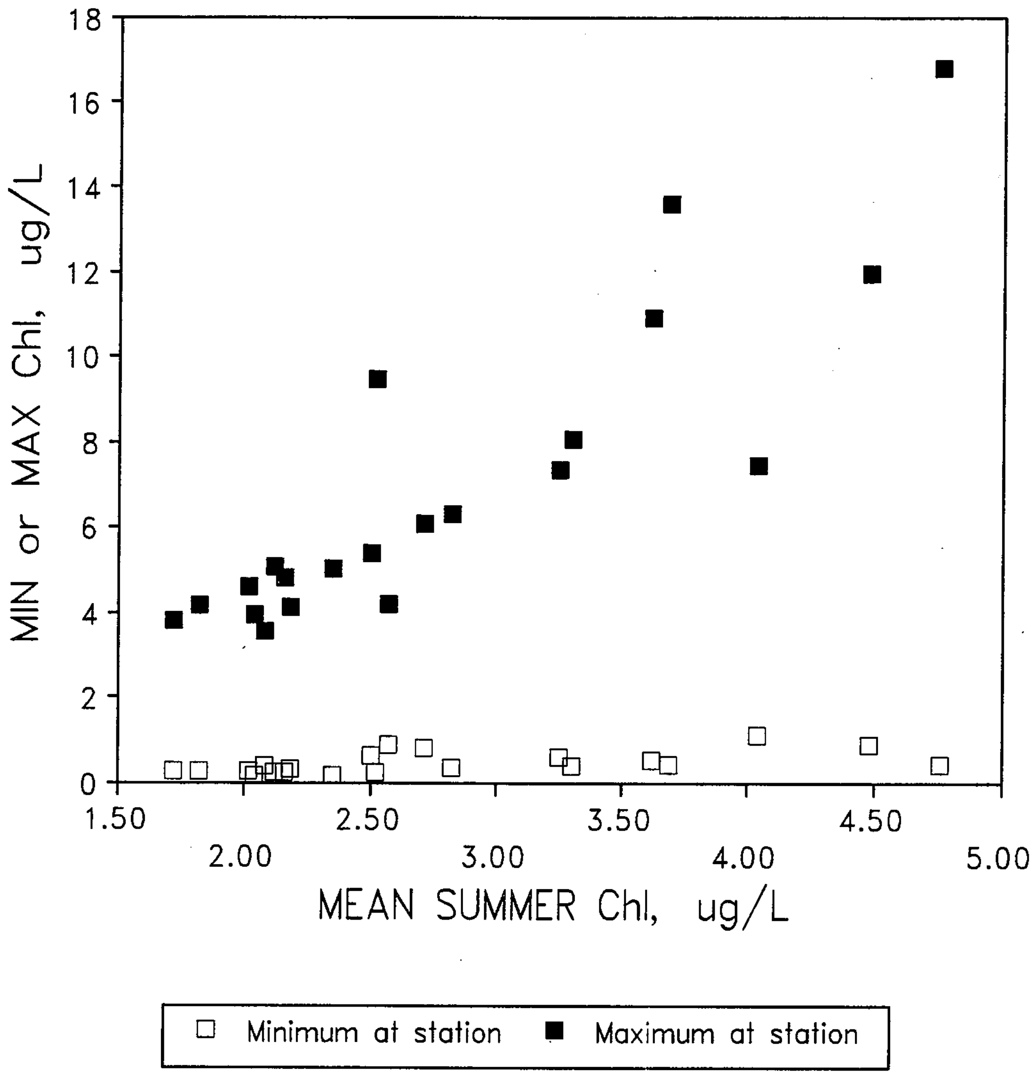

Figure 3-9. **Maximum or minimum chlorophyll during summer 1992 relative to mean values at nearfield stations.**

was only slightly more than 0.2 PSU (Figure 3-10a). Stations from the western side of the nearfield on average had higher chlorophyll and lower salinity. Beam attenuation (Figure 3-10b) was also a linear function of salinity ($R^2 = 0.51$, $n = 21$) and was correlated with chlorophyll ($R^2 = 0.58$, $n = 21$). Note that beam attenuation was in general much lower offshore than at the edge of the Harbor (cf. Figure 3-7). Again, the data suggest the present importance of nutrient export along the transect from Boston Harbor across the nearfield in western Massachusetts Bay.

The decrease in chlorophyll concentration with distance from the Harbor and the linear trend with salinity could indicate conservative export of chlorophyll, and thus an "outwelling" of organic matter from the Harbor. But to examine this possibility, one must include stations near the Harbor in the analysis. The Harbor-edge stations have been included in a summer period comparison in Figure 3-11 by calculating averages for the surface layers in June and August for the select group of stations that had strong N—salinity patterns (as summarized in Figure 2-9 and reported in Appendix Table 3). For this comparison, the station averages and patterns for chlorophyll, nitrogen, and salinity were generally similar to those shown previously (cf. Figure 3-11 vs. Figure 3-10). However, an extremely interesting feature is shown in Figure 3-11 which refutes the notion of conservative dispersion of chlorophyll produced inshore.

Assuming stations F23P and N04P represent inshore and offshore endmembers, conservative mixing between the Harbor and eastern edge of the nearfield could produce a straight line drawn between data points for F23P and N04P in Figure 3-11. However, stations between the Harbor and the nearfield that are at intermediate salinity (especially stations F24, F25, N01P, and N10P) had average chlorophyll concentrations higher than an implied conservative mixing line. The enhancement could arise from active growth of chlorophyll between the Harbor and the middle of the nearfield, but still at stations within the exported water plume that has high total N. In this interpretation, the area where chlorophyll stimulation occurs need not be extensive. For example, stimulation could occur in the few

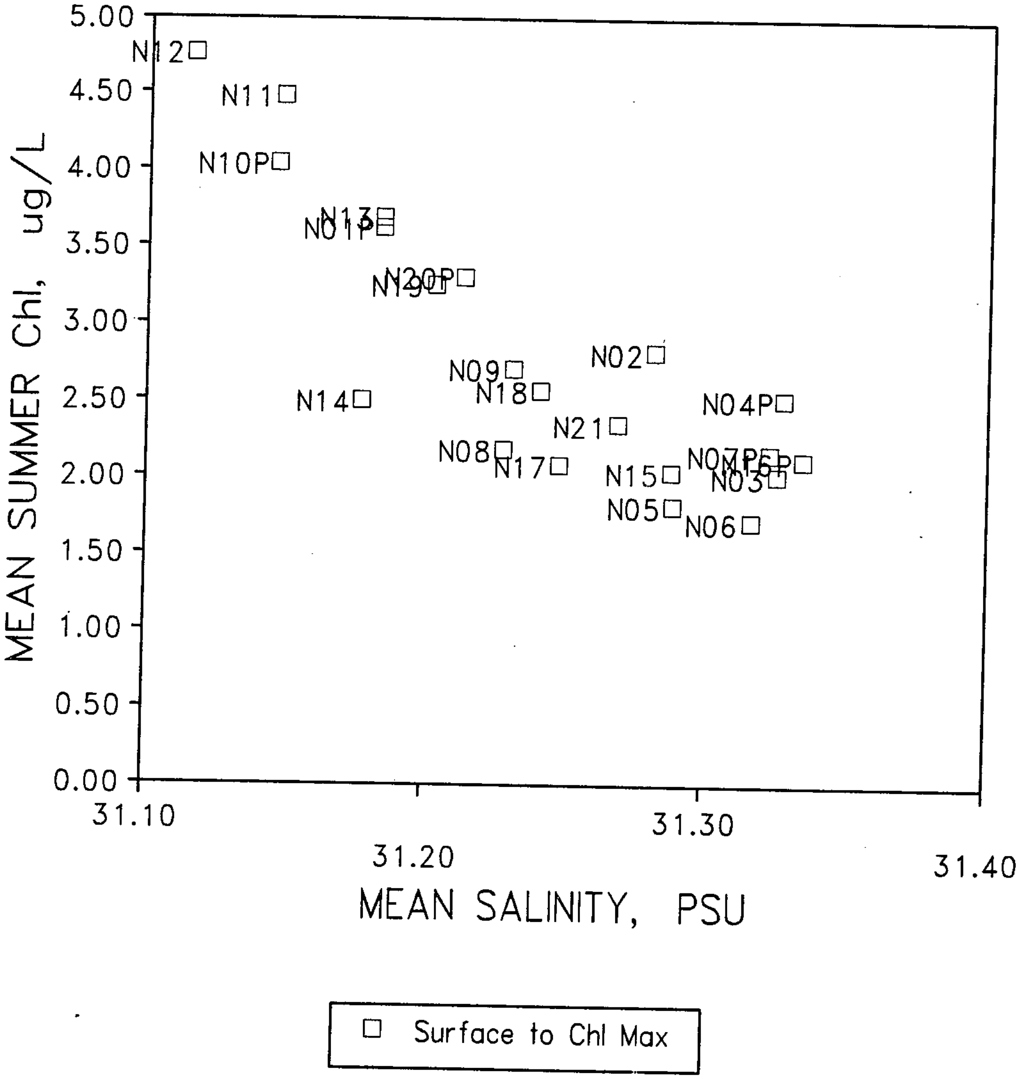

Figure 3-10a. **Summer surface layer chlorophyll and salinity in 1992 at nearfield stations.**

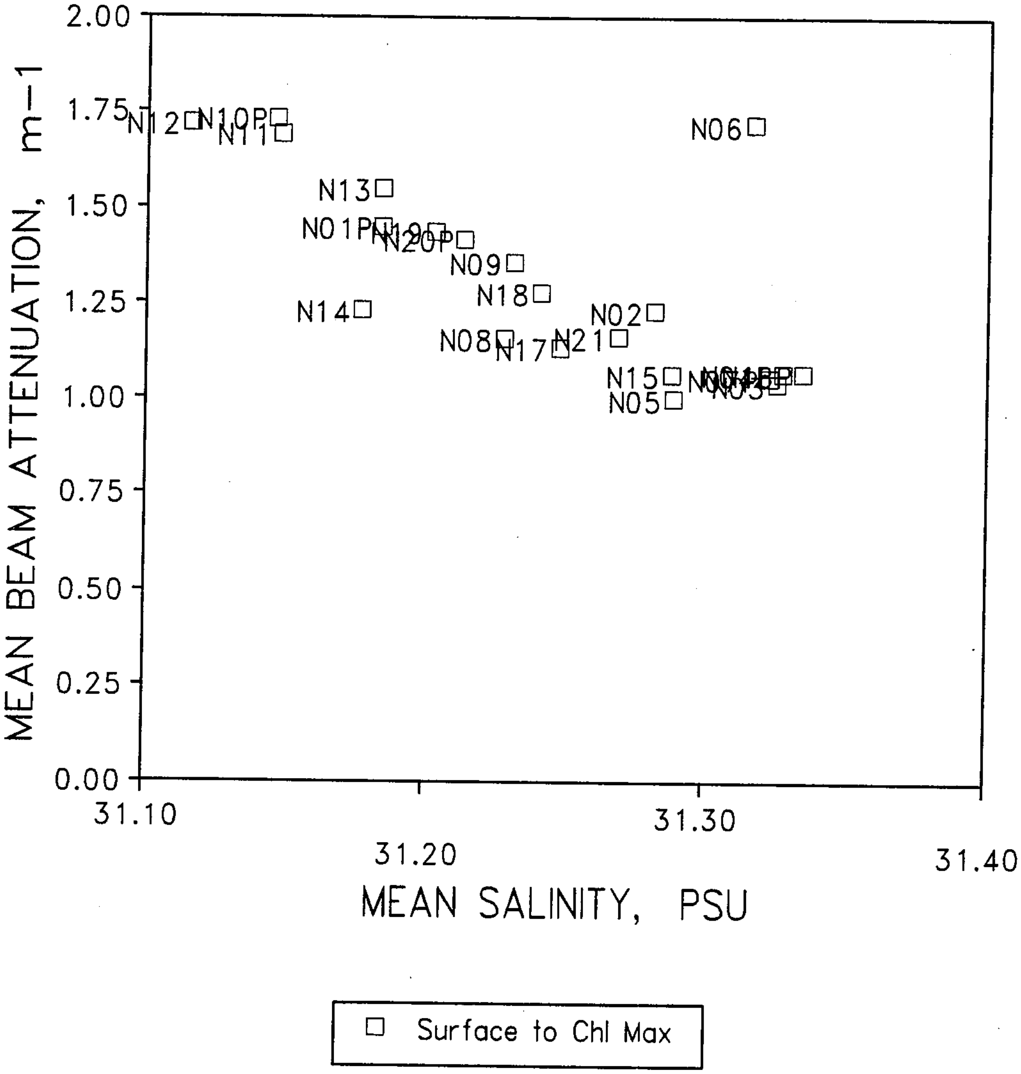

Figure 3-10b. Summer surface layer beam attenuation and salinity in 1992 at nearfield stations.

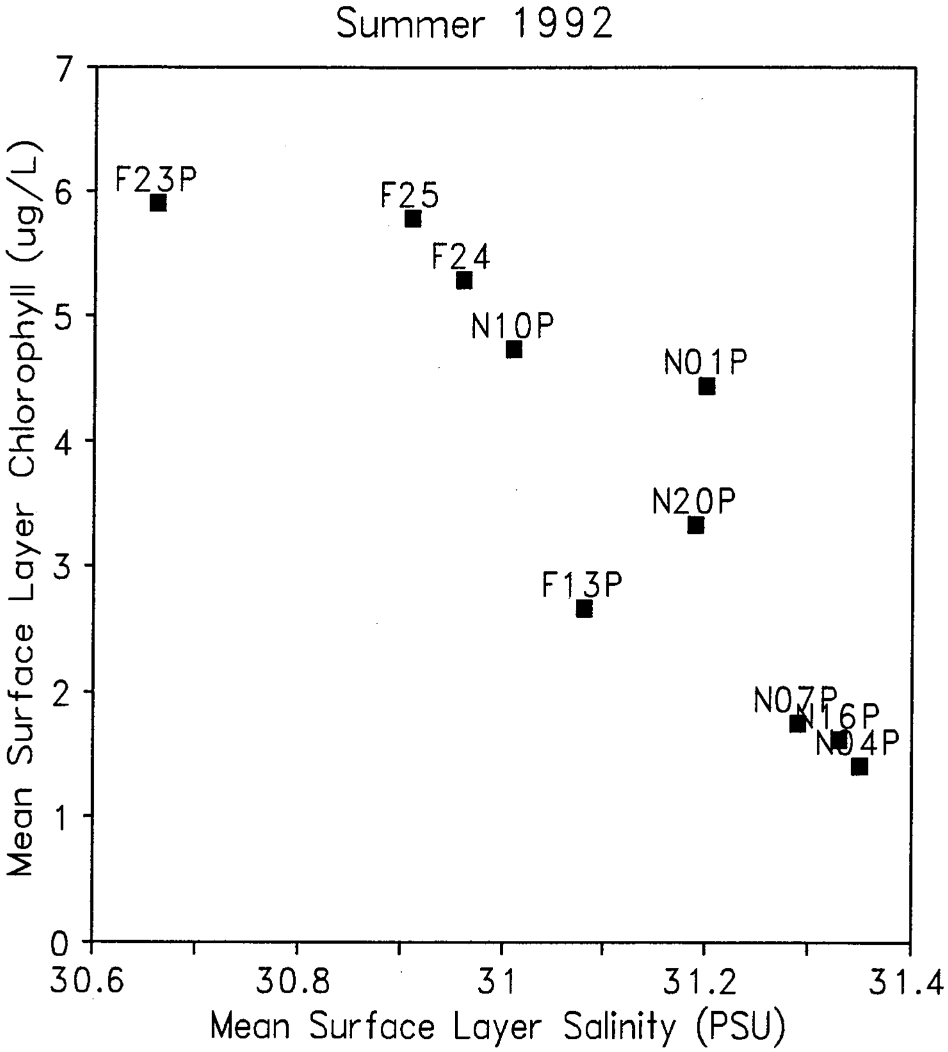

Figure 3-11. Summer (June - August) surface layer chlorophyll and salinity for select stations in the Harbor - nearfield region.

kilometers between the Harbor and stations F25, F24, and N10P, for one can draw a straight line from F25 to N04P and include most points.

There could be several mechanisms for chlorophyll stimulation. The western Massachusetts Bay area receiving that interacts with Boston Harbor by tidal exchange may function as a temporary trap to accumulate nutrients in surface waters. For example, station N10P appears to have a high TN concentration that is not fully commensurate with conservative mixing (see Figure 2-9). Such data supports the notion of a localized nutrient accumulation (via recycling or temporary retention of particles) possible because of unique physical mixing dynamics — this station is located essentially at the tidal front between inshore and offshore waters (Signell and Butman, 1992; Kelly and Albro, 1994). A second possible mechanism for chlorophyll stimulation outside the Harbor is an abrupt release from light limitation due to a turbidity drop, which is essentially the phenomenon addressed in the previous section. The apparent increase in chlorophyll over conservative mixing at the intermediate stations in questions, about 1-2 μg Chl L^{-1}, is of the same order as would be predicted by the composite (chlorophyll-turbidity) variable trend with TN described above, using summer turbidity data for these stations.

Kelly and Albro (1994), in analysis of data from an independent, high-resolution mapping study of Harbor-nearfield transects in June 1993, were able to show that the Broad Sound area (station F24) and that east of Nantasket Roads (F25, N10P) had enhanced chlorophyll that could not result from simple mixing of inshore and offshore waters. The enhanced chlorophyll occurred where turbidity decreased sharply. Thus, they speculated that the reduction in turbidity at the edge of the tidal front might allow for growth since nutrients in Harbor export must still be available at this location.

Regardless of mechanism, there now are two analyses which suggest non-conservative transport of chlorophyll from the Harbor to the nearfield. Notably, Kelly and Albro (1994) was based on high-resolution sampling on a single day, but the summer seasonal pattern in Figure 3-11 suggests sustained occurrence. Stimulation of chlorophyll in western

Massachusetts Bay occurs under present conditions, at least during summer. Herein lies a principal notion related to our recurrent theme of chlorophyll response to nutrients. If the higher chlorophyll concentrations found in waters just outside the Harbor are indeed produced because light is more available or some other Harbor-related limitation is removed, the interesting fact is that the increase is not very large, perhaps 1-2 μg Chl L^{-1}, and is similar to the more conservative annual average predictions made above. The summer chlorophyll "enhancement," presumably a consequence of the present nutrient export, is less than the present difference in chlorophyll concentrations between stations on the western and eastern sides of the nearfield area. Thus, the nearfield as a whole may exhibit little change in average chlorophyll concentration, but the cross-nearfield gradient should diminish with effluent diversion and the resulting chlorophyll gradient could become centered in the nearfield rather than along its western edge.

3.4 Summary Prospectus on Chlorophyll as an Indicator of Change: Scales of Response to Nutrients

Across stations in 1992, there were relatively small but well-ordered differences in annual average surface chlorophyll fluorescence. Moreover, the differences seemed to follow nitrogenous nutrients, but perhaps indicate a modest limitation on chlorophyll exists at higher nutrient levels near the Harbor. Annual averaging is clearly a useful scale for measuring change related to nutrient enrichment and it is a proper scale for addressing some of the major energy flows which, in part, follow from plankton biomass distributions. At this scale, meaningful change, in the context of the monitoring program, might be an average chlorophyll concentration above the present observed range observed or predicted for the Harbor-nearfield area or for other healthy shelf systems. Predictions that can be made at present generally suggest modest changes in chlorophyll concentration within most of the nearfield area around the proposed outfall. In fact, the average concentration could decrease, because much of the present export of MWRA-origin nutrients now sustains chlorophyll concentrations across much of the nearfield; at a minimum, the existing chlorophyll gradient from shore should be altered. More than the average chlorophyll in the nearfield surface layer, the most noticeable change in chlorophyll distribution in response to the new outfall location may be a change in vertical distribution: one can anticipate uniform deepening of the position of a chlorophyll concentration maxima in the water column towards the interface between the surface and bottom water layer, which will function as a prime nutrient source to euphotic layers during stratification.

The stratified period is, of course, a critical one for the Bay. It is encouraging to demonstrate that patterns of nutrients and chlorophyll, and gradients with distance and salinity, are readily discernible with the present intensive summer monitoring strategy in the nearfield.

At finer scales of resolution, the notion that extreme events, as well as mean conditions, are a function of higher nutrient supply merits further investigation, but in any case it provides an argument for continuation of frequent sampling in space and time. High-resolution *in situ* tow data (Figure 3-1, also Kelly and Albro, 1994), have revealed the scales of spatial and temporal patchiness in chlorophyll. Earlier efforts (e.g., Kelly *et al.,* 1992; 1993a,b) have illustrated that temporal variability in chlorophyll is often rapid; because of this, concentrations are unpredictable (even without including uncertainties of predicting nutrient-related responses) at any fixed station within the dynamic, three-dimensional fluid that is the Massachusetts Bay nearfield water column. A notion that chlorophyll patch size, persistence, and position in the water column are related to differences in nutrient supply (location and intensity) may be advanced from inspection of the patterns at different locations in the Bays (Figure 3-2 and Kelly *et al.,* 1993c); changes in patch characteristics are a potential indicator of response along the Harbor-Bay gradient which may add to the ability to detect and describe subtle changes in the ecosystem. Large and persistent chlorophyll patches, perhaps as much as any localized or acute event, are relevant to eutrophication and promotion of low oxygen events in western Massachusetts Bay.

Given limited scientific understanding, and no explicit regulatory guidance, on those chlorophyll levels that might be considered undesirable in an ecosystem context, it is helpful to maintain a hierarchy of sampling scales in the monitoring program to ensure that any relevant changes are detected and evaluated. Development of scale-dependent and independent indicators should continue as data gathering efforts continue, with the goal of offering predictive guidance (as to the types of changes that need to be detected by monitoring efforts) from finer scales to seasonal and annual time frames. It is realistic to presume that high-resolution sampling for chlorophyll (e.g., moorings, towed *in situ* profiling, or frequent standard surveys) may provide early warning of meaningful change in the ecosystem, but at the same time care also must be exercised in its use. We do not yet know the importance of localized and acute peaks in chlorophyll (in space or time) and one must recognize that this type of monitoring indicator (cf. Kelly and Harwell, 1989) can raise concern where no meaningful change ever becomes realized.

4.0 Influence of Nutrients on Plankton Species Composition

In addition to stimulation of overall phytoplankton biomass measured as chlorophyll, the influence of nutrients on plankton species in Massachusetts Bay is a concern, for some changes potentially could affect food webs. Moreover, a principal issue is stimulation of certain problem species. This includes species often classed as "nuisance" because they may cause a surface scum or cause discoloration of otherwise clear water and those classed as "noxious" for their toxins, either accumulated in the next link of grazing food chain (endotoxins) or spewed into the water itself (ectotoxins). The influence of nutrients on species composition is inherently difficult to predict. The 1992 monitoring data is examined here to update our knowledge on these topics. In considering the potential effects, the previous discussion on nutrients must be kept in mind, particularly that changes are expected to be localized near the outfall.

4.1 Nutrients and Plankton Species

It is generally recognized that there is a limited ability to predict species-level variation in the plankton as a function of nutrient concentrations (or loading) (Cura, 1991; Townsend *et al.*, 1991; Kelly, 1991; Smayda, 1992; ASA, 1993). There is a substantial ability in the monitoring design to relate the total phytoplankton biomass response to nutrients and other aspects of the environment (e.g., Section 2); however, both the strength and limitation of a biomass indicator is that it is essentially blind to individual species fluctuations. With biomass regulated by energetic and nutritional constraints and, therefore, bound by the laws of physics and thermodynamics, it is somewhat "robust" to many types of species replacements, where any of several species capable of playing a similar functional role and having fairly similar physiological requirements may flourish in a specific time or place.

Smayda (1992), in reviewing species successional patterns and available geographic distributions at that time, wrote (p. 8)

> "The general conclusion from this limited data set is that despite some regional variations in successional patterns, the phytoplankton composition and dominant species throughout the region are generally the same. The phytoplankton flora of neither Massachusetts Bay, nor Cape Cod Bay stand out as regionally distinctive communities. In fact, the winter-spring bloom component is similar to that south of Cape Cod Bay, including Narragansett Bay, Long Island Sound and New York Bight (Smayda, 1973). Features distinctive from the flora south of Cape Cod include: the prominence of the nuisance species *Phaeocystis pouchetii*; the occurrence, spreading and blooming of the toxic dinoflagellate *Alexandrium tamarense*, and *Emiliania huxleyii* events."

A Massachusetts and Cape Cod Bays-wide bloom of *Phaeocystis pouchetii* in April 1992 (cf. Kelly *et al.,* 1993a,c; Turner, 1993) was also observed by Turner (personal communication) in Buzzards Bay, to the south of the Cape. There are indeed scores of resident plankton species common to the Bays, as revealed by Borkman's and Turner's (MWRA Water Column Monitoring Reports) careful identification of species from nearly 200 samples collected in 1992.

The possibility of "red tide" or other noxious/nuisance species being encouraged by nutrients, as suggested by Don Anderson (e.g., Franks and Anderson, 1992), has been acknowledged. But this issue "for the moment has to be considered of a relatively low order given the observations already made about expected [small] changes in Massachusetts Bay" (ASA, 1993). To track possible noxious/nuisance species, the 1992 monitoring program included additional special sampling designed to quantify the less-numerous, but larger (>20 μm) cells of dinoflagellate species considered by phytoplankton specialists to be of greatest ecological concern .

Most scientists (e.g., Nixon *et al.,* 1986; Franks and Anderson, 1992; Smayda, 1992; ASA, 1993) argue effectively that the problem is complex, that factors other than nutrients can often be primary influences, and that the complexity of the marine ecological response to enrichment begs for controlled experimental work to address issues of the level and quality (i.e., ratios of elements) of nutrient enrichment. Suitable experiments are difficult to design and interpret, so the limitations of the results must always be recognized, as should the fact that results are also "rarely wholly unambiguous" (ASA, 1993). Historically, controlled enrichment experiments have taken two forms: e.g., short-term (days-weeks) bottle-size container studies (cf. Smayda, 1992, for MWRA, 1988, 1990) and larger-scale, longer-term (years) mesocosm studies conducted at the URI MERL facility (Nixon *et al.,* 1986, Oviatt *et al.,* 1986; Doering *et al.,* 1989).

Short-term, bottle-type enrichment studies conducted with water samples from western Massachusetts Bay in July and September 1987 resulted in several general conclusions (Smayda, 1992):

- "Nuisance species did not bloom in the 2-day enrichment experiments. The community structure remained more or less intact.
- The most abundant species were stimulated to higher growth rates, although the highest nutrient levels used appeared to have repressed growth of individual species."

Smayda (1992) also reviewed results of some longer experiments. Selected conclusions that speak to the issue of species changes, from among many of his conclusions (MWRA, 1990) on results of 7-day experiments using water samples collected approximately in the middle of the nearfield and enriched with "secondary effluent," follow:

1. "The phytoplankton community at station P2 (42° 22'42" N, 70° 47'10" W) sampled on 18 April 1988 (see MCA Rpt. No. 88-1 for companion field studies) was dominated by diatoms typical of late-spring bloom communities in New England coastal waters. This bloom studied between 2 March — 4 May was in decline.
2. Chronic exposure of this community to nine secondary sewage effluent treatments, including two controls; three treatments at 40:1 seawater:effluent

enrichment; two treatments at 90:1; and two treatments at 150:1 did not alter community structure.

3. The dominant species present initially persisted irrespective of effluent treatment. The following diatoms dominated: *Thalassiosira nordenskioeldii, Skeletonema costatum, Nitzschia delicatissima, Nitzschia seriata, Thalassionema nitzschiodes* and several *Chaetoceros* species. *Thalassiosira pseudonana* and *Nitzschia closterium* were of secondary importance.

4. Nannophytoplankton < 10 μm, which considerably exceeded diatom abundance numerically, but contributed only 25% of the chlorophyll biomass in the initial community, did not increase dramatically with effluent level, contrary to expectations.

...

6. Nuisance algae blooms of nannophytoplanktonic species would not be expected during the winter-spring period in response to the above sewage effluent treatments; the diatom component would be stimulated to greater abundance.

...

25. Rapid utilization of $Si(OH)_4$ by diatoms suggests both an export of surplus nitrogenous nutrient and increased N:Si ratio with downstream distance would occur along the dispersal gradient. The availability of $Si(OH)_4$ will determine the duration, magnitude, and spatial pattern of diatom blooms stimulated by nitrogenous nutrient in the discharged effluent.

26. The winter-spring diatom bloom in recipient waters can be expected to be more intense and more prolonged. Summer blooms of nannophytoplankton, predominantly non-diatoms, are likewise expected to be more prolonged and intense.

...

29. The potential for nuisance algal blooms during summer-autumn periods will be increased, and should be evaluated experimentally.

30. Whatever else is done with regard to regulation of nutrient delivery, the maintenance of high silica loading is strongly recommended to ameliorate potential nuisance bloom stimulation and hypoxia by nitrogen loading."

The latter few conclusions, whether or not fully warranted from the short April experiments, raise interesting issues. Some of these have been examined in the second type of controlled experiments mentioned above using the MERL mesocosms. The mesocosms have used water from Narragansett Bay, thereby including much of the same phytoplankton community as in Massachusetts and Cape Cod Bays (see above); many experiments have dealt with well-mixed conditions simulating a well-mixed 5-m water column and others have studied stratified systems. ASA scientists, among the many researchers involved in MERL studies, have written that

> "In the few studies which have addressed these questions (i.e. response of plankton communities to long term nutrient enrichment) the results have been complex, and often not in accord with the preconceived wisdom. In particular, there have not been blooms of noxious species, and often the relative proportions of diatoms to flagellates have not followed expected scenarios (Nixon *et al.,* 1984; Doering *et al.,* 1989; Oviatt *et al.,* 1989)." (ASA, 1993)

Recent experimental results suggest that manipulating the N/Si ratio had little overall metabolic or chlorophyll biomass effect at a given N loading (Doering *et al.,* 1989). Thus, some long-term, large scale results are counter to some conventional notions, such as the prediction in the last Smayda conclusion (No. 30) above. Doering *et al.* (1989) wrote:

> "With respect to the ideas expressed in the 2 papers by Officer and Ryther (1980) and Ryther and Officer (1981) our results are equivocal. On the one hand, alteration of the N: Si ratio of nutrients in sewage effluent did nothing dramatic by way of 'ecosystem improvement'. Although enhanced fish growth resulted, this was not as dramatic as that caused by addition of nutrients in a ratio resembling sewage effluent. By contrast, however, our results do support many of the ideas concerning diatoms, especially the importance of N:Si supply ratio and grazing by both zooplankton and benthos in regulating diatom abundance."

Results of Doering *et al.* (1989) did confirm that N/Si ratios can affect the phytoplankton species composition. Large manipulations of the ratio can alter the flow of energy partitioned between the pelagic and benthic/demersal consumers (including fish) and even perhaps modify trophic efficiency with respect to biomass realized by fish which prefer to graze on diatoms. However, we should always remain cautious in using experimental results conducted at scales and conditions that imperfectly mimic those relevant in nature and in this case one must be conscious of the experimental nutrient concentrations relative to environments of concern. The Doering *et al.* (1989) experiment used enrichments typical of very highly-enriched coastal systems. Their time-weighted average DIN concentrations in enriched tanks, independent of the Si loading, were in the range of 53-65 μM — a factor of 8 or more greater than the annual average for the Harbor-edge station F23P in 1992. Recognizing such differences and drawing from other MERL results, including those of Oviatt *et al.* (1989), it has been suggested that nutrient enrichment with silica-limited, nitrogen-rich wastewater, at concentrations likely to occur at the proposed MWRA outfall site, are not likely to affect dinoflagellate populations (ASA, 1993). This statement seems well-reasoned but, of course, one must still maintain a monitoring program that is attentive to less probable, but perhaps consequential events.

4.2 Results from 1992

Nutrient export from the Harbor into the surface of the Bay in a sense provides an experiment in nature that has continued for decades. The monitoring data at different stations in the Bay occur across a nutrient concentration gradient and therefore provide an appropriate database to review for evidence of broad differences in species composition in relation to nutrients.

Species composition and nitrogen concentration. Coastal stations (F23P and F13P), nearfield "P" stations, and Cape Cod Bay stations (F01P and F02P), receive different quantities and maintain slightly different levels of nutrients (e.g., see Section 2). On average, the coastal stations have the highest nutrient concentrations. The Cape Cod Bay stations have nutrient concentrations similar to the mid-range of the nearfield stations. Within the nearfield, stations N04P and N07P to the eastern side of the nearfield characteristically have lowest, and stations N10P and N01P highest, nutrient concentrations. Chlorophyll levels reflect these nutrient differences (Section 3). The taxonomic composition by major phytoplankton groups is given in Figure 4-1a,b for these stations. Further details are in the sources cited in the figures. There is seasonal variation, especially from higher dominance of diatoms (February and March) to strong dominance by *Phaeocystis* in April, to a mixed community in summer and fall. Station F23P, highest in nutrients, has an overall balance of major phytoplankton groups quite similar to less nutrient-enriched conditions, and, like chlorophyll, often did not have highest total cell counts. Figure 4-1 presents a coarse view; multivariate analyses have not yet been conducted and such techniques might be able to discriminate fine differences in communities across some groups of stations. But notably from the summary in Figure 4-1, the seasonal pattern is stronger than the individual station variability; i.e., overall the stations are relatively similar in spite of nutrient differences. A similar observation is apparent for zooplankton, as described in more detail by Turner (1993) and shown in Figure 4-2. For zooplankton, about the only major distinction was the presence of more estuarine species at F23P, which has slightly lower salinity most of the year.

Species composition and nitrogen/silicate ratios. Two aspects are of interest, a difference between Cape Cod Bay and Massachusetts Bay during 1992 and comparison of the present outfall area at the edge of the Harbor with the future outfall area in Massachusetts Bay.

First, a brief discussion is provided regarding Cape Cod Bay and Massachusetts Bay relative to biology and N/Si ratios in the water column; Kelly *et al.* (1993c) summarized differences fully. In 1992, two blooms of potentially-noxious phytoplankton (*Phaeocystis pouchetti* in April, and *Ceratium longipes* in June) appeared to relate to changing N/Si ratios in Cape Cod Bay. The *Phaeocystis* bloom occurred also in Massachusetts Bay but the *Ceratium* bloom was not detected there. In Cape Cod Bay (but not Massachusetts Bay), silicate, along with nitrogen, was virtually depleted by the time *Phaeocystis* bloomed in April; Cape Cod Bay was lower than Massachusetts Bay with respect to silicate concentrations at this time (Figure 4-3a). By June, the mid-water in Cape Cod Bay had adequate N and P supplies for the intense development of *Ceratium,* which does not require silicate; subsequently, when *Ceratium* was detected in very high numbers, silicate was higher in Cape Cod Bay than elsewhere (Figure 4-3b). Looking at the data for June, one would think that Cape Cod Bay

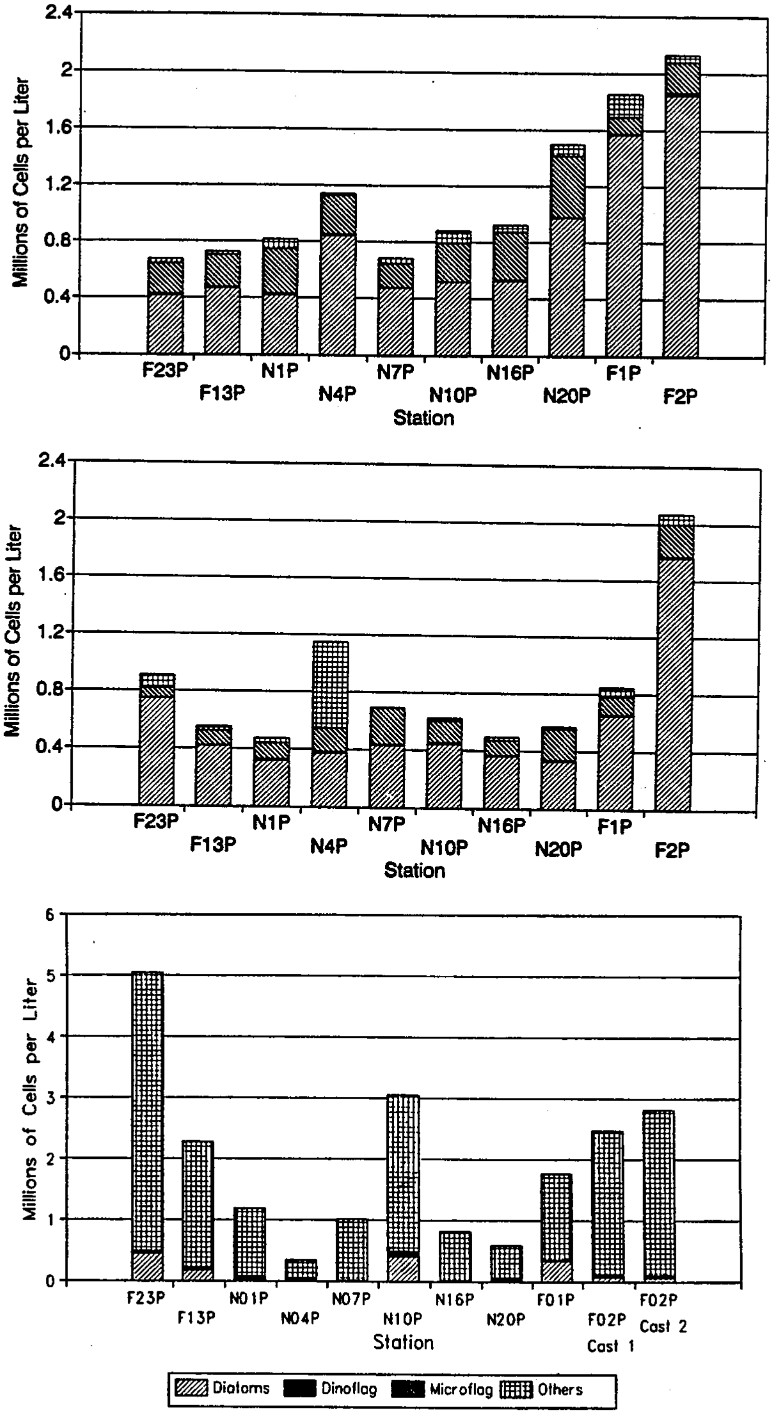

Figure 4-1a. Phytoplankton groups by stations in winter-spring 1992 for Station F02P, replicate casts were taken in some months. All samples are from the surface. [Sources: Kelly *et al.* 1992, 1993a]

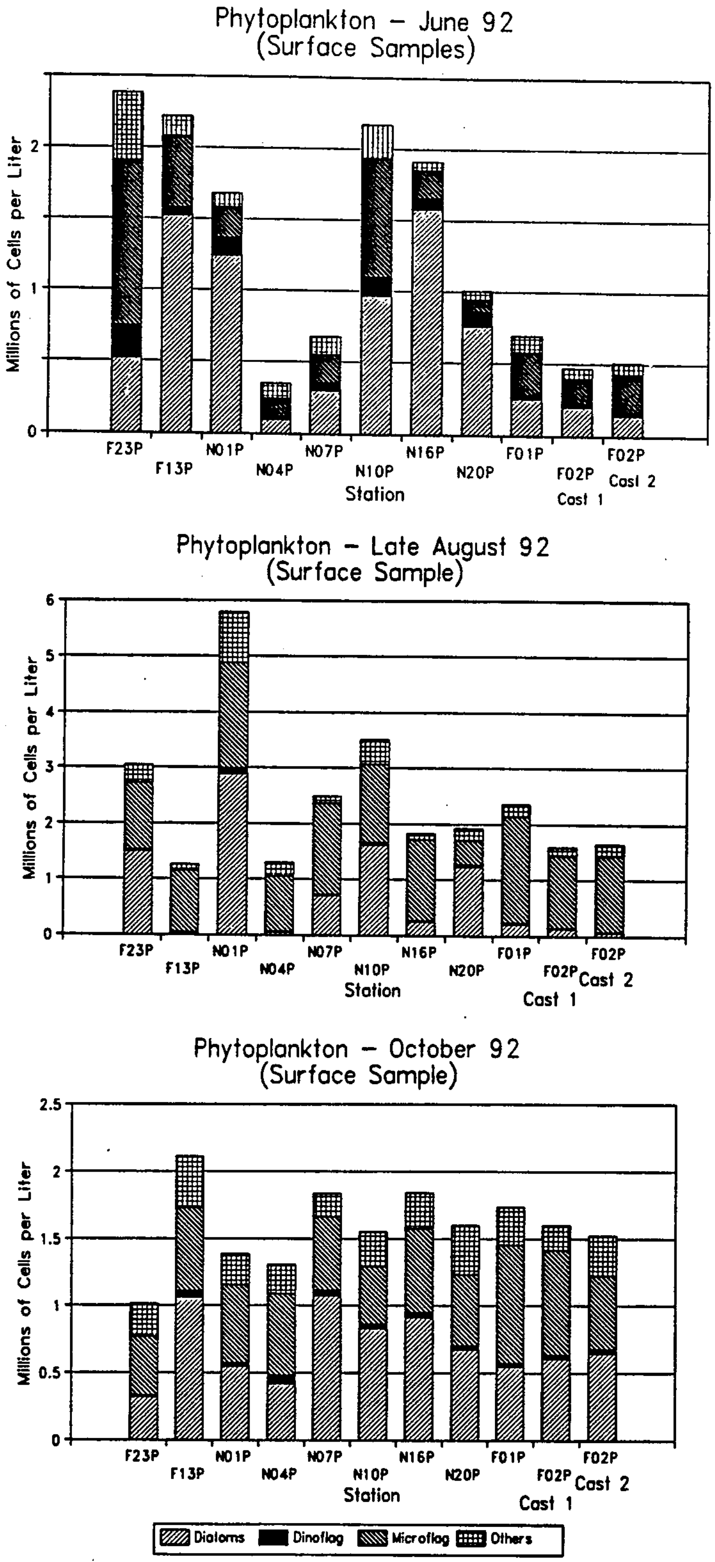

Figure 4-1b. **Phytoplankton groups by stations in summer-fall 1992. For Station F02P replicate casts were taken in some months. All samples are from the surface. [Sources: Kelly *et al.* 1993a,b]**

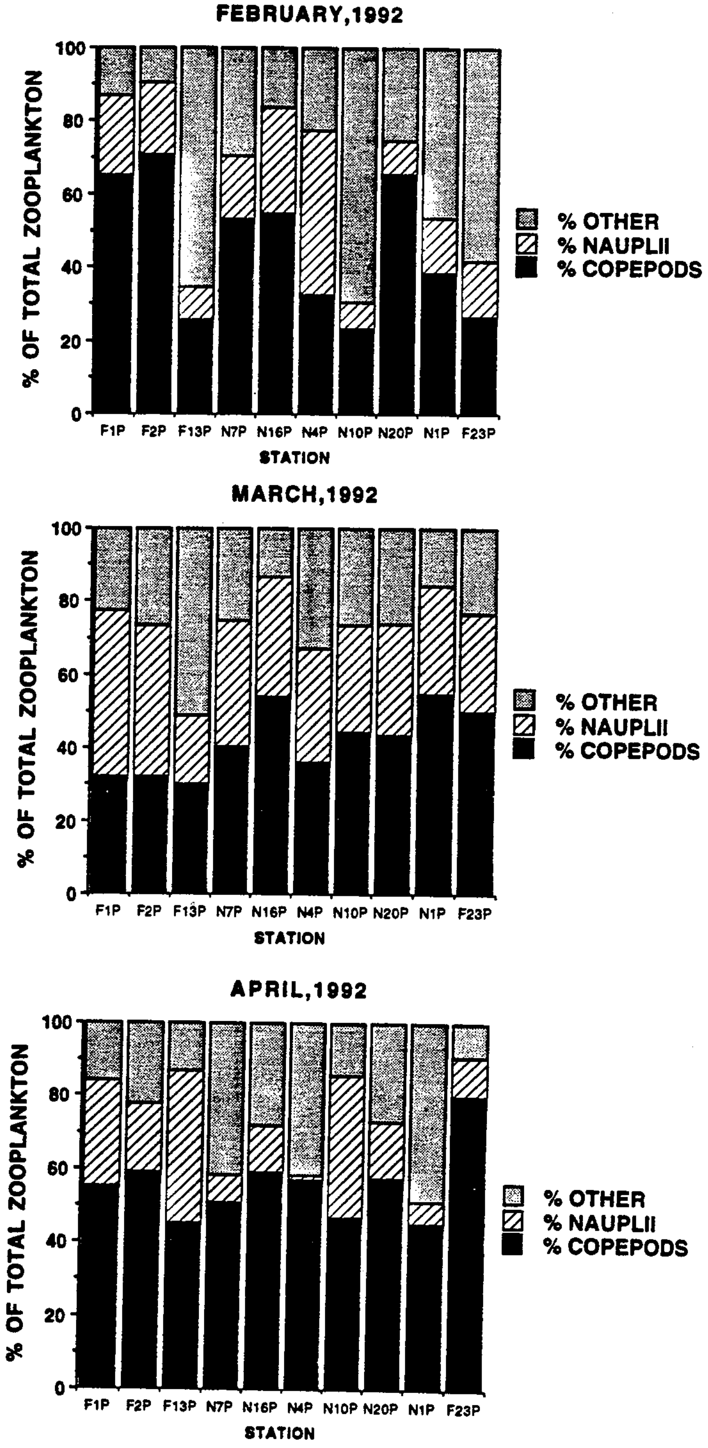

Figure 4-2a. Zooplankton groups by stations in winter-spring 1992. [Source: Turner, 1993]

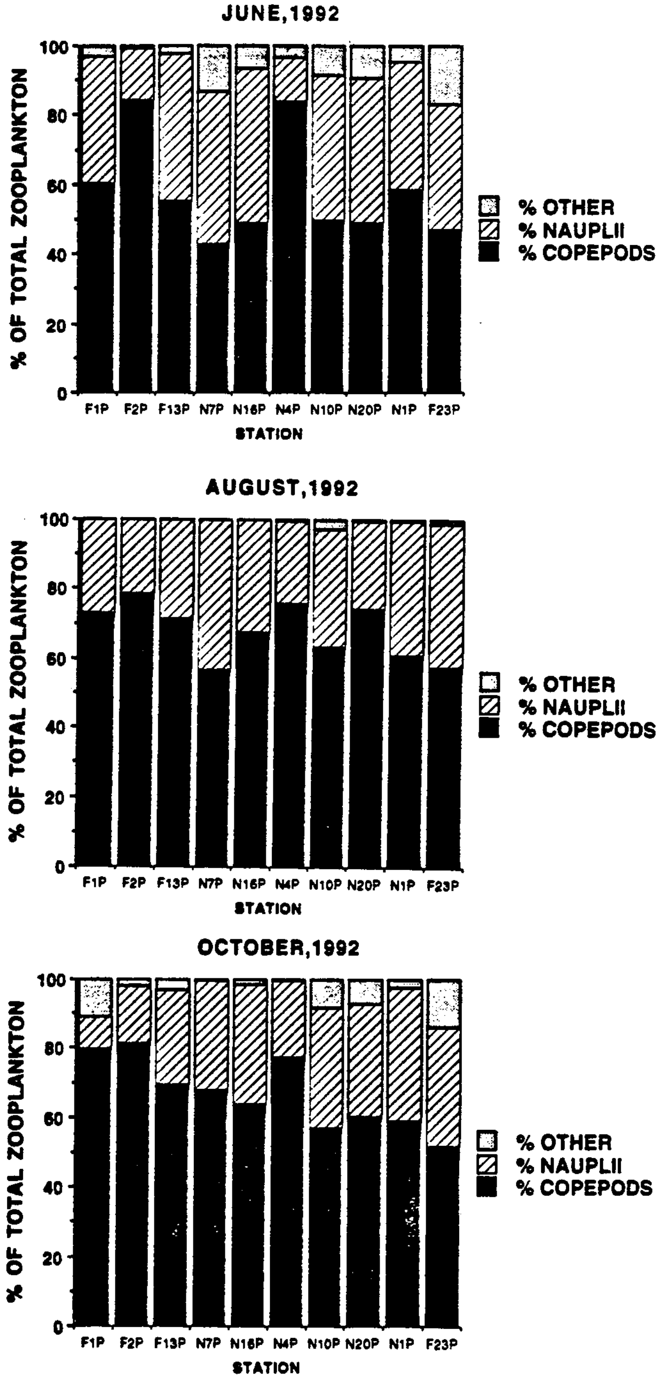

Figure 4-2b. Zooplankton groups by stations in summer-fall 1992. [Source: Turner, 1993]

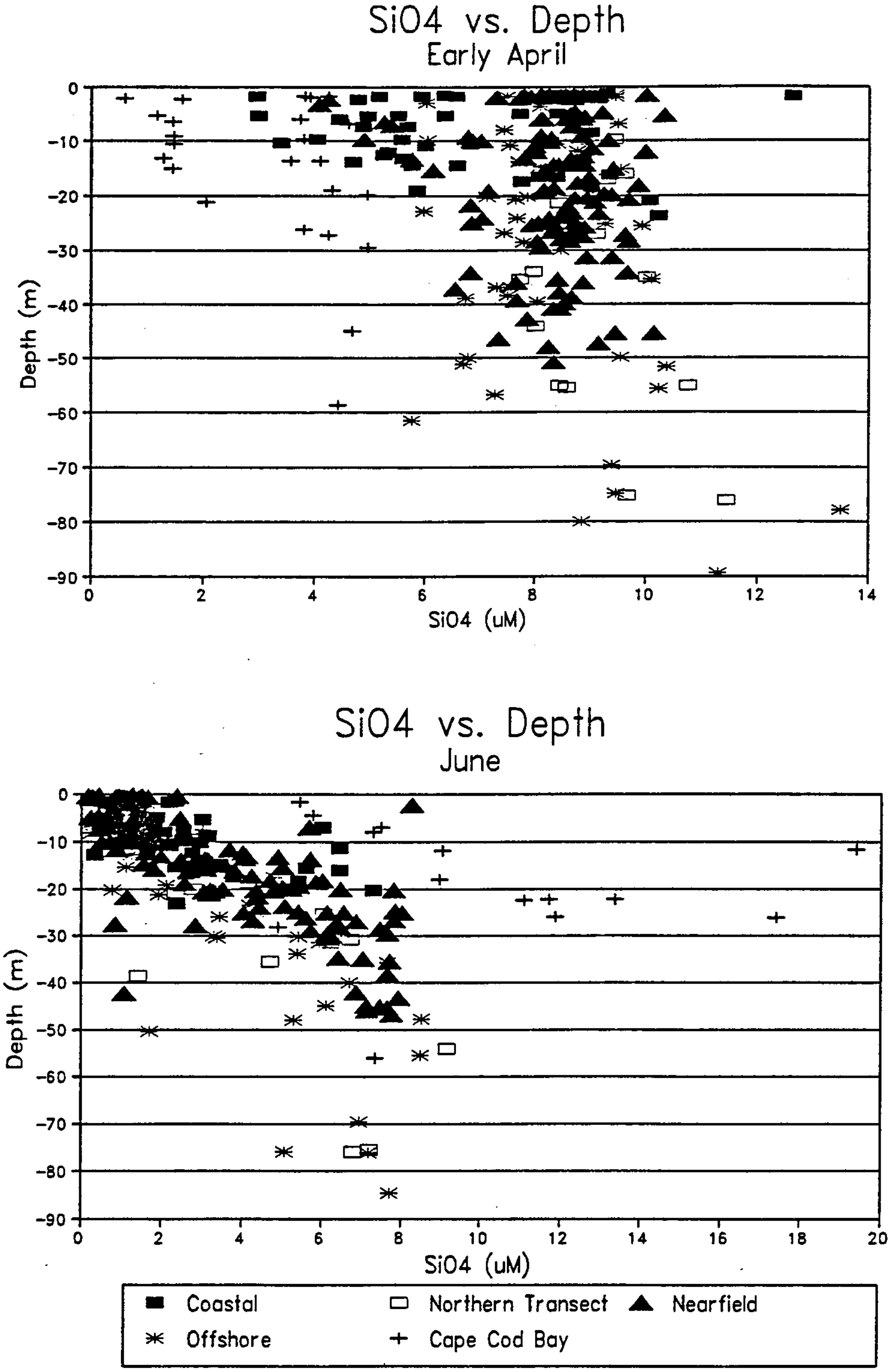

Figure 4-3. **Silicate vs. depth for all stations on combined survey cruises in early April and June 1992. [Source: Kelly *et al.* 1993a]**

water, because of higher Si (and lower N/Si ratios) would be prime for diatoms. Following this period it may have been, but at this time the conditions were probably a consequence of dinoflagellate bloom. One important suggestion from these data is that biology is not just passively influence by water chemistry; biology can actively influence chemistry as well. A second suggestion is that winter-spring diatoms do appear to have the potential to create a nutritional situation advantageous for nuisance dinoflagellates to exploit, a sequence that may be part of the normal cycle of events in coastal and shelf waters.

In the case just described, it appears that a successional scenario, as alluded to in Smayda's conclusions above (No. 25), may occur in the development of many nuisance blooms. It is not clear why the Cape Cod Bay winter-spring bloom appears to be initiated earlier and may characteristically be more prolonged (cf. Kelly *et al.*, 1992), but this appeared to be in part why silicate became more depleted. Higher Si depletion does not seem related to more available N; levels of TN in February were not high relative to other stations. Interestingly, Cape Cod Bay stations in 1992 had more biological attributes of a eutrophied system than Massachusetts Bay stations (Kelly *et al.* 1993c) but there is no evidence linking Cape Cod Bay attributes to higher nitrogen concentrations or external loading, which both appear to be much higher near Boston. Additional studies funded by the Massachusetts Bays Program in 1993/1994 will help understand the unusual character and dynamics in Cape Cod Bay.

N/Si ratios in the principal areas of concern. Figure 4-4 compares the near-Harbor stations and the nearfield stations (all depths) in winter 1992. A clear relation between DIN and silicate is evident for both, particularly the Harbor-edge. There, the N/Si ratio from a functional regression is about 2.8:1 (R^2=0.90, n= 17; Figure 4-4). The majority of data for the nearfield had slightly less N per atom of Si but most had ratios greater than 1:1.

In the summer (Figure 4-5), the Harbor-edge stations had a ratio of about 1.7 (functional regression slope; R^2=0.62, n= 39). There were some cases where DIN was very low and silicate was still easily detectable; these of course have a lower ratio. The bottom waters for the nearfield that are indicated in Figure 4-5 are where the future outfall will discharge,

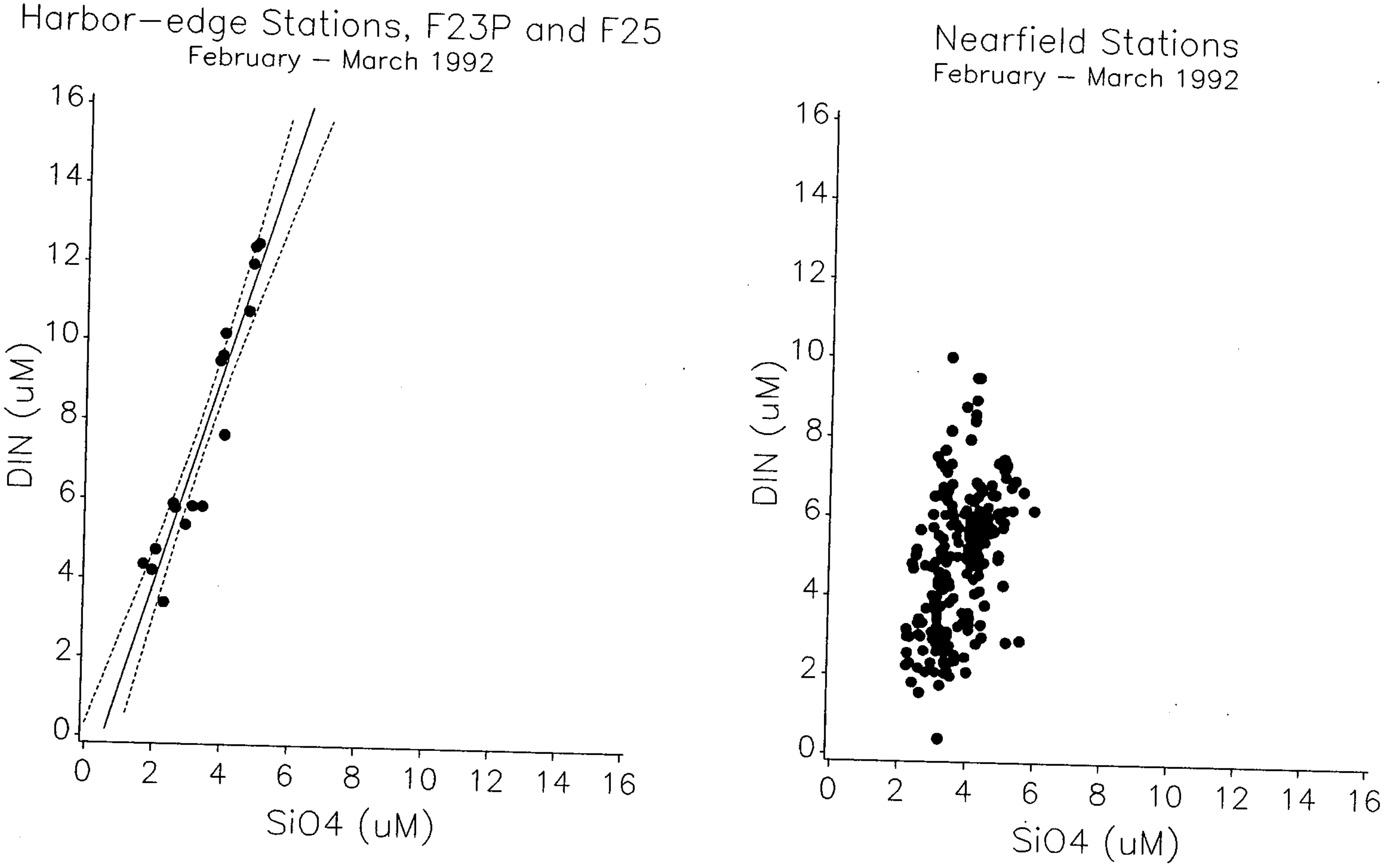

Figure 4-4. Dissolved inorganic nitrogen and silicate in western Massachusetts Bay in winter 1992. A Linear regression line, with 95% confidence Limits (dotted lines) is indicated for the Harbor-edge stations.

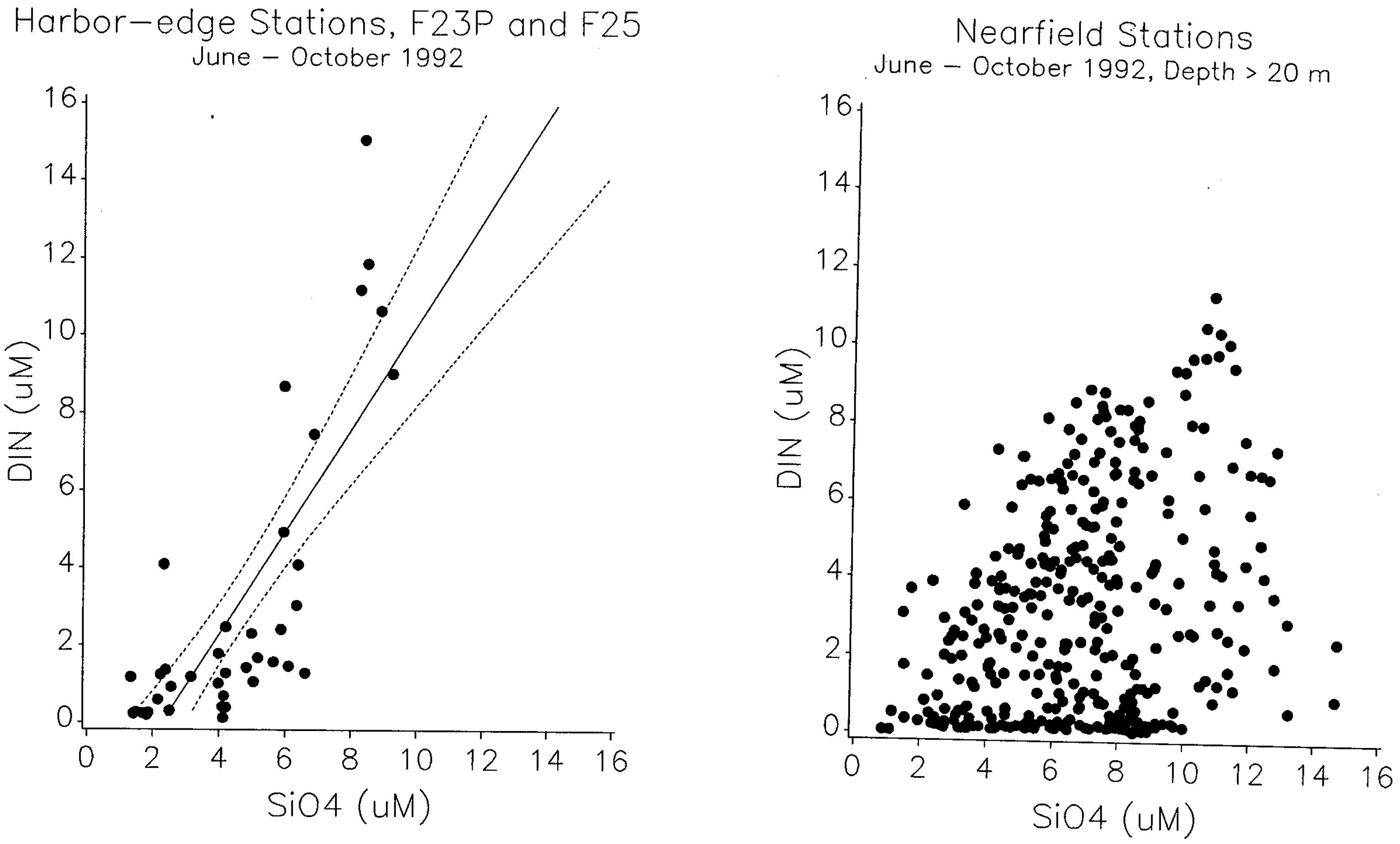

Figure 4-5. Dissolved inorganic nitrogen and silicate in western Massachusetts Bay during summer 1992. Right is for depths below the pycnocline only. A linear regression with 95% confidence limits is indicated for the Harbor-edge stations.

subsequently acting as a source to the surface layer. Bottom water N/Si ratios approached 1.7 but were generally lower than this. The scatter in the ratio in summer data reinforces the notion of the nearfield as a complex mixing zone for inshore and offshore waters (Kelly *et al.* 1992, 1993a,b). At times during 1992, silicate in bottom waters to the offshore was observed mixing into the nearfield area. The fact that ratios can be lower offshore also may relate to discharges to the Harbor that are relatively N-enriched. The recent inclusion of silicate as a part of effluent chemistry analysis will help characterize the MWRA effluent source. The Harbor's relative enrichment in N in summer may also be a partial consequence of a larger role of sediments in nutrient cycles at higher temperatures. With respect to this, Giblin *et al.* (1994) noted that N/Si ratios in Harbor sediment fluxes are generally higher than for depositional sediments in the nearfield.

It is not simple to predict how sediment fluxes will affect Harbor water quality in the future, after effluent diversion. Since sediment fluxes will also affect the Harbor's future nutrient export (concentrations and nutrient ratios) to the Bay, nutrient export is likewise difficult to predict, but it must be included to predict future surface water concentrations and ratios in the nearfield, especially during summer.

The principal point of the comparison, though, is to provide an empirical basis for understanding the present and expected variation in N/Si ratios. Some of the nearfield area now has N/Si ratios that are as high as the Harbor area where effluent is presently discharged, in part because those western stations receive exported Harbor water. The rest of the area, especially the deeper bottom waters will, in the future, likely become more enriched in N than silicate and the ratios may rise, but probably no more than presently observed in the Harbor. Overall, the present nutritional differences between Harbor and nearfield are most pronounced during stratification, yet even then are not large and do not appear to create major community shifts.

The Harbor and nearfield station phytoplankton species composition were similar in 1992. Diatoms, not dinoflagellates of nuisance/noxious nature, were a substantial component of the community even during summer. This indirect evidence supports the premise that the *expected* N/Si ratio changes by themselves may not induce major undesirable shifts in phytoplankton species. This notion will be controversial, but it is a basis for examining the facts of the problem and seems in accord with some controlled experimental mesocosm results summarized earlier.

In general, the 1992 monitoring results show similarity, with some exceptions, in species composition throughout the Bays, as earlier noted by Cura (1991) and Smayda (1992). The Bays therefore may be viewed as biologically well-mixed, much as the system is physically well-mixed and open. The present monitoring design captures major seasonal events and allows some limited insight into the influence of nutrients on plankton species. An improved study design, and the best "experimental analog" to when the future outfall is in operation, might be to follow the "blobs" (Signell and Buttman, 1992) of Harbor water with higher nutrients and chronicle the plankton response as these are advected through and mixed with the Bay water. Although perhaps difficult to do, the technology exists to perform such studies.

5.0 Nutrients, Metabolism, and Bottom-Water Dissolved Oxygen

A prime concern with nutrient enrichment is depletion of dissolved oxygen (DO) caused by excessive production coupled with rapid consumption of organic matter. Mid-water hypoxia/anoxia, which has been documented in the New York Bight (e.g., Falkowski *et al.*, 1980), is possible with rapid decay of particulate organic matter. Another concern is with progressive depletion of bottom water, as has occurred in recent years in other U.S. coastal waters. Kelly *et al.* (1993c) have previously reviewed the 1992 annual cycle of DO in Massachusetts Bay. Frequently, surface waters were supersaturated, not surprising for a productive system. In general, mid-depth waters of Massachusetts Bay were near saturation for most of the year; at some stations, water within the pycnocline was, at times, supersaturated during summer stratification. The lowest DO saturation levels measured were in the range of 70-75% and the lowest concentrations were near 7 mg L^{-1} or about 1.0 mg L^{-1} above the state standard of 6 mg L^{-1} used as a "site determinative measure" in outfall siting (EPA, 1988).

Depression of dissolved oxygen (DO) is a principal issue in Massachusetts Bay since the nearfield area and most of the Bay is seasonally stratified from about late April to mid-October. Some production may be continuously or rapidly transferred to bottom waters and therefore not really observable in any chlorophyll-averaging scheme. Metabolism (production and consumption of organic matter) is a measure complimentary to chlorophyll and may provide a useful indicator for monitoring; metabolism in the water column and sediments is reviewed in this section. A preliminary nitrogen budget is assembled to illustrate relative strengths of sources supporting nearfield production and 1992 bottom-water DO trends are discussed in the context of consumption of primary production.

5.1 Metabolism in Massachusetts Bay

Figure 5-1a shows estimates of integrated water column production, derived from oxygen changes across a light-dark gradient (Kelly *et al.,* 1993c) for six nearfield "P" stations sampled in 1992. The production rates for 1992 were converted to carbon from oxygen. They may be biased towards high estimates because the assumed Photosynthetic Quotient (PQ) of 1.25 may underestimate the actual PQ (Doering *et al.,* 1993). Also plotted in Figure 5-1a are the 1973/1974 Parker (1974) data for *in situ* measurements made a few kilometers south of the proposed outfall diffuser; these data were summarized by Cura (1991, Table 9) and are also shown in Smayda (1992, Figure 6B). A third data set displayed in Figure 5-1a was summarized by Smayda (1992, Appendix Table IV). Originally reported in MWRA (1988, 1990), these data were derived from incubator measurements of water from three stations in western Massachusetts Bay in 1987/1988. Those three stations were near stations N10P, N21 and F24 of the 1992 water quality monitoring program. Some of the pre-1992 measurements, notably in 1987/1988, were made at approximately weekly intervals but, in this comparison, are plotted all under the same month, thereby obscuring the rapid rise and fall (one to two weeks) in peak winter-spring production rates. The three annual-cycle studies are not strictly comparable because of differences in methods, but they represent virtually all of the available computations of daily primary production rates in Massachusetts Bay. Townsend *et al.* (1991) made suitable measurements but did not present integrated daily rates.

Figure 5-1a illustrates the considerable variability in production evident in each study both in space and over short time periods. Considerable time-space variability is expected for production, because integrated water column production is strongly influenced by chlorophyll concentrations (see Section 3). Two seasonal peaks in production nevertheless seem suggested, characteristically occurring in late spring (March) and late summer/early fall (late August or September). Both within and across studies, the production rate within any month often varies by an order of magnitude, so results will depend on the week and location

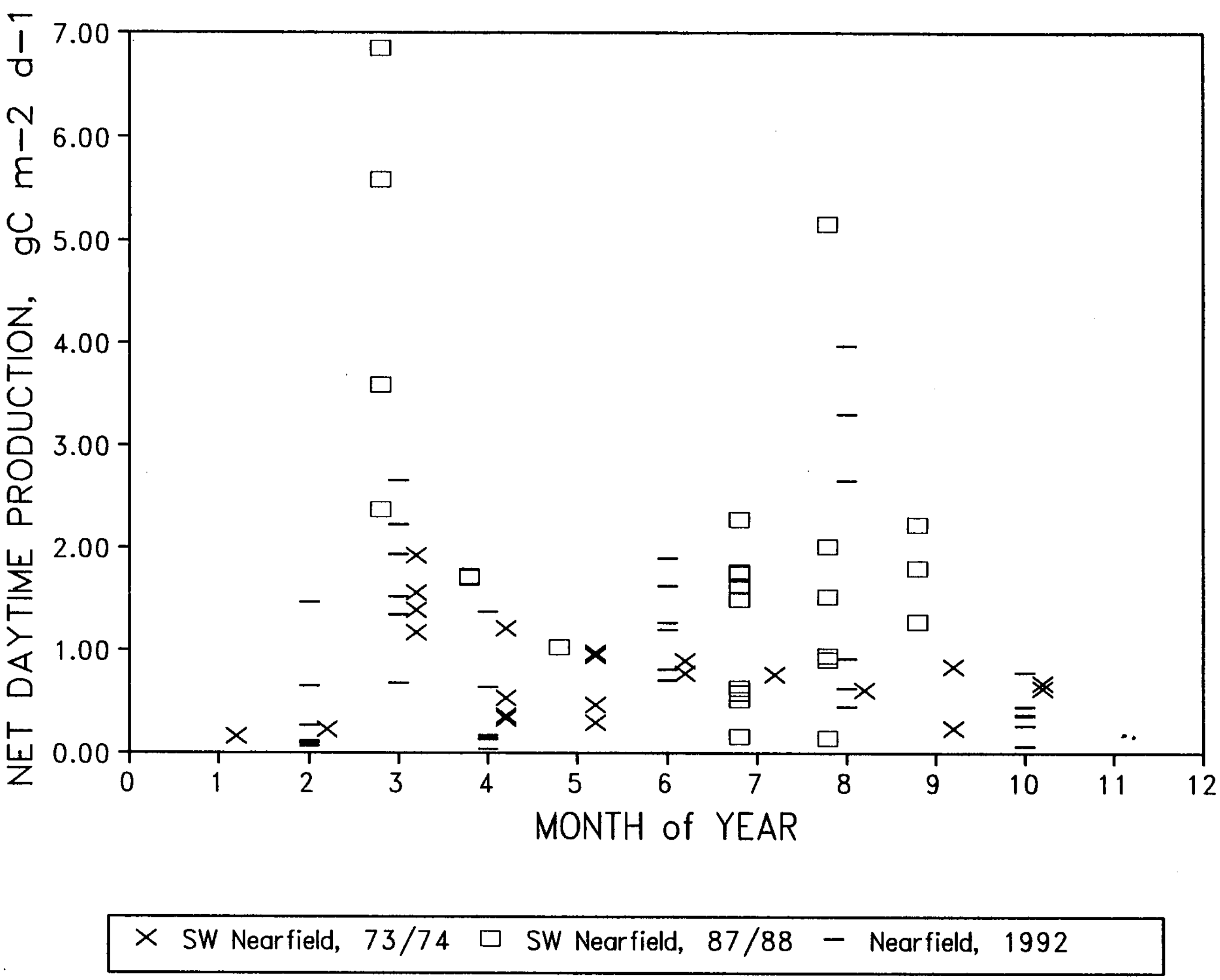

Figure 5-1a. Primary production in the nearfield area. [Sources: Parker (1974) as cited in Cura (1991) and Smayda (1992); MWRA (1988, 1990); and Kelly *et al.* (1993). 1992 data represent the average rate derived from incubation of surface and chlorophyll maximum samples at each of the six stations. Note that the three studies are offset slightly for each month. See text for additional details.]

chosen for each production measurement. This observation is most evident in March and in summer, when more measurements have been made.

The three studies in Figure 5-1a have fairly similar patterns and roughly equivalent rates. Comparability is difficult to assess, given high time-space variability, differences in methodology, and the sensitivity of calculations to assumptions (cf. Doering *et al.*, 1993; Kelly *et al.*, 1993c; Frenette *et al.*, 1993). With a relative insensitivity of production to nutrient loading at the high nutrient loads typical of most estuaries (e.g., Kelly, 1991; Nixon, 1992; ASA, 1993), and where light limitation may also complicate the picture, it may be difficult to determine changes in integrated water column production with the present monitoring design. Regardless, the measurements give some indication of the energy flow, which is required for understanding carbon and nutrient flow in the region of the proposed outfall.

Annual primary productivity has only been coarsely estimated. Reflecting on the data in Figure 5-1a, it seems reasonable to assume an annual productivity near 350 g C m^{-2} in the nearfield (cf. Cura, 1991). An appropriate range may be 350 to 500 g C m^{-2} y^{-1} (Smayda, 1992) but, for the sake of computations, 350 g C m^{-2} y^{-1} will be used; this amount[1] implies a nitrogen requirement of about 4400 mmols N m^{-2} y^{-1}.

Using an empirical model derived from many marine areas, Kelly (1991) suggested that about 15 to 27% of produced carbon (at about 350 g C m^{-2} y^{-1}) might be consumed by an underlying soft-sediment benthic community at the water depth (about 32 m) and mixed-layer depth (about 15 m) in the nearfield. The few empirical measurements of benthic flux available in 1991 suggested that a smaller percentage might be consumed. New data obtained in 1992/1993 by Anne Giblin, Charles Hopkinson, and Jane Tucker of the Marine Biological Laboratory have recorded somewhat higher rates than previously measured (Giblin

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[1] 350 g C m^{-2} y^{-1} divided by 12 g C/mole, divided by 6.625 mole C/mole N = 4.4 moles N m^{-2} y^{-1}, or 4400 mmols N m^{-2} y^{-1}. The 6.625 C/N conversion is a standard assumption based on the Redfield ratio.

et al., 1994). Using high and low temperature oxygen uptake rates (October and February), an exponential relationship with temperature was derived: Flux (mmol O_2 m^{-2} d^{-1}) = 9.33 $e^{0.066(Temp)}$ (R^2= 0.49, n= 11). This is a normal pattern that suggests a Q_{10} $\approx$2.0, or roughly a doubling of rates for each 10 °C increase in temperature — approximately the annual rise in nearfield bottom waters from winter to early fall. Using the equation, the monthly average temperature distribution in bottom water, and assuming a Respiratory Quotient (RQ) of 1.0 to convert from oxygen to carbon, roughly 62 g C m^{-2} y^{-1} would be consumed in softer, depositional (silt-clay and/or fine sand) sediments found in the nearfield[2]. This value represents about 18% of primary production.

The bottom in the nearfield area is not highly depositional, so the value and percentage estimate may be high. Knebel's (1992) assessment suggests that, at most, about 29% of the entire Massachusetts Bay area may be similar to the soft sediments sampled for benthic fluxes (including both Knebel's "depositional" and "sediment reworking" bottom types). The percentage of soft sediments typical of those surveyed for benthic fluxes likely includes only a small portion of the "sediment-reworking" type; this type might be as low as 5-10% of the bottom in the nearfield.

Metabolism in non-depositional, coarse sands and hard-bottom areas has not been measured. However, fluxes for most elements and compounds may be low, an expectation based on results of benthic fluxes from an initially bare hard bottom in mesocosm studies (5 m water column). In mesocosms at the Marine Ecosystems Research Center (MERL) at the University of Rhode Island, a sedimented flocculent layer accumulated in a short-term study (about a week following a large diatom bloom — Hunt, 1983; Kelly and Nixon, 1984) as well as a one-year experiment (Doering, 1989). In both experiments, opportunistic benthic macrofauna (including spionid polychaetes dominant in Massachusetts Bay) colonized and "bound" sediment, similar to what may happen during stratification in Massachusetts Bay

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[2] Approximate monthly temperatures were used to calculate a rate for each month from the model. The average rate of these months was 14.1 mmols O_2 m^{-2} d^{-1}. That rate, multiplied by (365 d/y x 1 mmol C/mmol O_2 x 12 mg C/mmol C) and divided by 1000 mg/g = 61.7 g C m^{-2} y^{-1}.

(Shea *et al.*, 1991). Concurrent with this, initially clean hard bottom in the mesocosms achieved oxygen and nitrogen flux rates that were about 10-20% compared to tanks with intact depositional sediments from Narragansett Bay. Note that the mesocosm studies were conducted in a shallower and warmer ecosystem than that of interest in Massachusetts Bay and experimental results likely produced relatively high fluxes; but in the absence of other information, the 10-20% range is useful as a guide to a calculation made next. Assuming that a maximum of 29% of the area consumes carbon at the annual rate given above and the remaining area (71%) consumes carbon at 20% of this rate, the consumption of pelagic carbon by the nearfield benthos would be less than 8% of the overlying production[3]. The calculation suggests a relatively small role for the benthos in pelagic carbon consumption processes.

Calculations, similar to those for oxygen flux above, for regeneration of dissolved inorganic nitrogen (DIN) and loss as N_2 (based on direct denitrification measurements of Barbara Nowicki in 1992/1993) yield annual values on the order of 550 mmols N m^{-2} y^{-1} for DIN and 175 mmols N m^{-2} y^{-1} for N_2. Without correction for the percent of bottom that is soft-sediment, the sum of these rates represents 725 mmols N m^{-2} y^{-1}, or about 16% of the surface layer production as N, which is similar to that calculated for summer by Giblin *et al.* (1994). Correction, as above, lowers the value to about 7%.

Some annual N production must be buried, not returned as remineralized flux. However, this is usually only a small fraction of the flux itself. A calculation, using 0.1 cm y^{-1} deposition (probably maximal — see Knebel, 1992) and about 0.15% N in surface sediment (Giblin *et al.*, 1993), yields an estimate of about 96 mmols N m^{-2} y^{-1} possibly buried or, after the 29% area correction, well less than 1% of primary production, which is comparable to estimates for other coastal and shelf systems.

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[3] (62 g C m^{-2} y^{-1} x 0.29) + (62 g C m^{-2} y^{-1} x 0.2 x 0.71) = 27 g C m^{-2} y^{-1}, or 7.7% of annual primary production.

These preliminary annual calculations will be improved as more data are collected for Bay sediments. For the critical summer period, Figure 5-1b draws together data and calculations on various flows of nitrogen (e.g., from Section 2) that might furnish nutrients to primary producers. Although preliminary, this figure conveys the significant progress that has been made in quantitative understanding of the nearfield area[4]. Since the indicated flows are calculated for the whole nearfield, the input flux at the western side of the nearfield must be higher than the eastern side; the gradient is therefore not emphasized by these average flows. Nonetheless, the figure illustrates that in the present situation of MWRA effluent discharge to the Harbor, the flow of nitrogen from the Harbor is the dominant supply to the surface photic layer and could nearly supply the nutrients required for observed rates of net primary production (NPP). Pelagic nutrient recycling has not been assessed directly and is not included in the diagram, but it is likely substantial. The DIN flux from bottom sediments supports a DIN flux across the pycnocline into the photic layer; each rate is small compared to primary production needs and to the present supply from inshore.

As effluent is diverted and becomes a direct input to subpycnocline bottom waters of the nearfield, flows will change dramatically. As suggested in Section 2, the input from inshore then will be restricted to roughly 10% of the present flow. Benthic fluxes in the nearfield are expected to increase (Hunt and Steinhauer, 1994) and crude calculations (Section 2) suggest that the diffusive flux across the pycnocline might be 1 mmol N m^{-2} d^{-1}. Therefore, the inshore and bottom-water fluxes of N to the nearfield photic zone would become more balanced than at present. Because of these expected shifts in relative strengths of the major nitrogen sources, it is likely that the vertical distribution of phytoplankton and NPP will shift and chlorophyll-NPP maxima should be observed near the depth where the dominant nitrogen flow enters the photic layer. Additionally, however, the projections indicate that the total nitrogen flux to the photic zone will be reduced compared to the present; because of this

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[4] The reader is invited to compare this Figure 5-1b to the similar, but purely qualitative, schematics of major processes in the nearfield presented as Figures 18-19 in Kelly (1991).

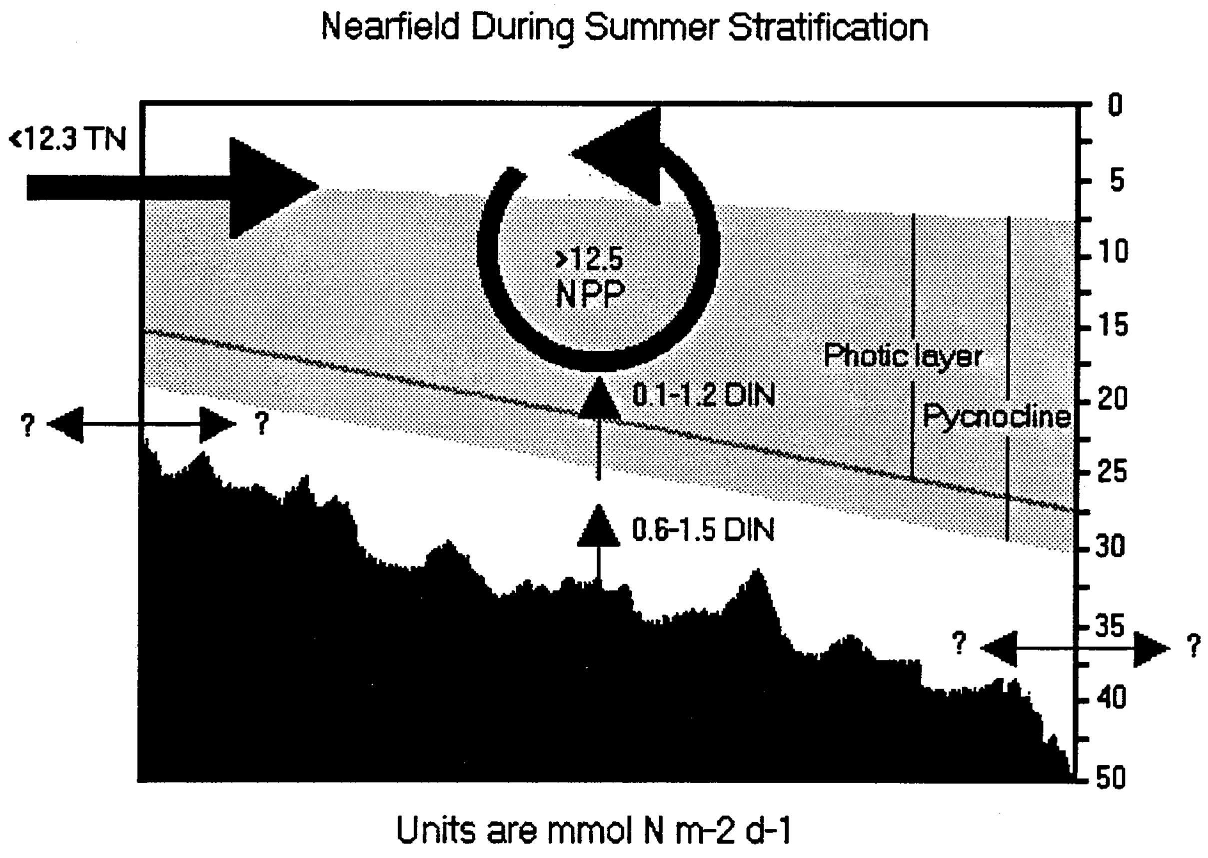


Figure 5-1b. Net primary production (NPP) and supporting nitrogen flows during summer stratification. Flow estimates are indicated for near-surface input from inshore (as Total N, TN), flux across the pycnocline into the photic layer (as Dissolved Inorganic Nitrogen, DIN) and flux into the bottom water from benthic recycling (DIN). The view is to the north, with the inshore to the left and offshore to the right. The approximate depth ranges are indicated for the photic layer and the pycnocline between surface and bottom layers; these ranges often grade slightly from inshore to offshore. The horizontal distance is greater than 10km and the depth scale is in meters, so the vertical exaggeration is >200:1.

NPP in the nearfield can be expected, on average, to decrease, rather than increase, with the diversion of effluent to the new offshore outfall.

Besides predictions, a basic conclusion is suggested at both annual or seasonal timescales and evident in Figure 5-1b — rates of benthic metabolism and nitrogen recycling are significant to the nearfield nitrogen budget, but do not indicate that the benthos is presently the major supplier for production or consumer of pelagic organic matter. If only a small fraction of pelagic production reaches the bottom in the nearfield, then it must be consumed in the water column or transported away as particles to depositional areas, some of which are outside of the nearfield region. Trends in water column DO are reviewed next to aid understanding of oxygen consumption in the water column. Changes in bottom-water DO over the stratified period in Massachusetts Bay during 1992 are examined for two regions: the nearfield stations and a group of five stations positioned in deep water along the north-to-south axis of Stellwagen Basin. Stellwagen Basin is of particular interest because of low DO values observed there by Townsend *et al.* (1991) and also because it may be a regional depositional area, collecting organic matter from a wide area, perhaps including transport from the nearfield.

5.2 Time Trends of Bottom-Water DO in the Nearfield

In 1992, the lowest DO concentrations in nearfield bottom waters (below 20 m) were measured in October. Lowest values were just below 7 mg L^{-1}, about 75% of the saturation value (Figure 5-2). The temporal trend showed a decrease in DO concentration (and % saturation) in May, a rebound through the next few surveys in June and July, followed by a period of continuous decrease starting in late July-early August and extending to October, after which DO concentrations began to rise again (Figure 5-2).

The downward trend in DO observed in May followed the cessation of the winter-spring bloom, sinking of chlorophyll, and initiation of strong stratification. As noted by Kelly *et al.* (1993c), the May 1992 DO data seem suspiciously low. In 1993, however, a DO

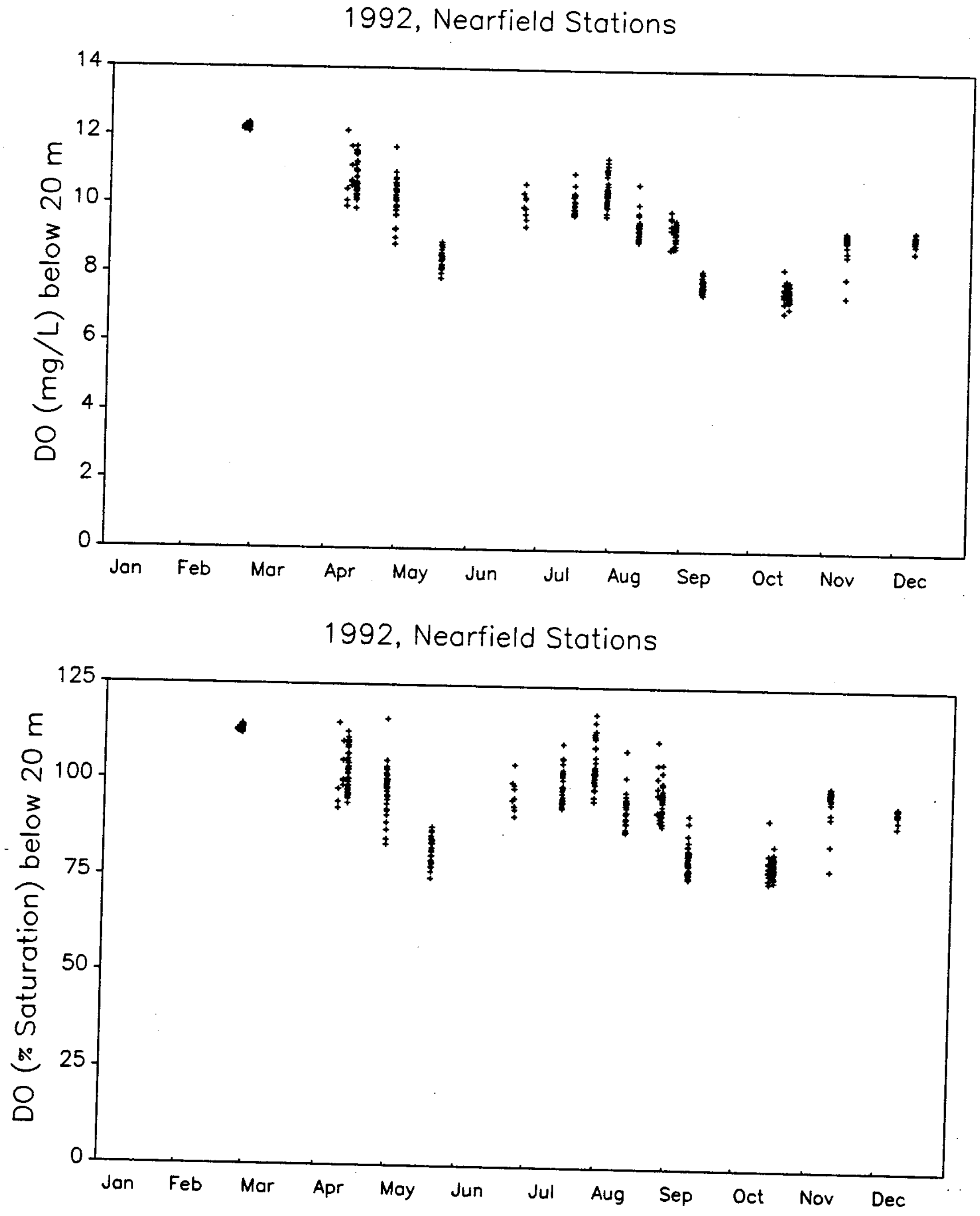
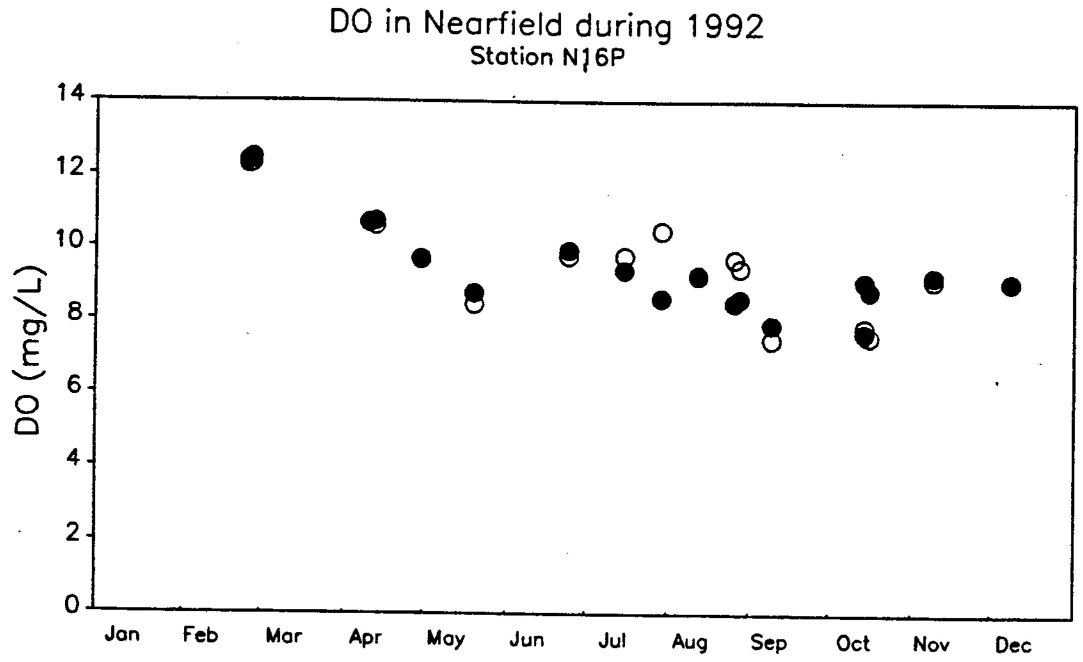


Figure 5-2. **DO concentration (top) and % saturation (bottom) in water greater than 20 m deep in the nearfield (from 21 stations, n = 349 data points). Note several anomalous, suspect low points in February.**

decrease also has been observed in late Spring and subsequently followed by a slight rise in DO concentration in early summer (Kelly *et al.* , 1994). The May data may indeed suggest an initial respiration response to the winter-spring events as bottom waters began to warm. After May, an upward trend continues for more than one survey in 1992; thus, there must also be a phenomenon which allows the bottom water to be partially reoxygenated and produce the early summer rebound. Two mechanisms are possible. First, production at low light within the pycnocline may contribute. This has been noted as higher DO concentrations at the subsurface chlorophyll maximum (Kelly *et al* 1993a,b). Second, the variability in the density structure of many nearfield stations suggests that physical stratification may, on occasion, be at least partially disrupted and this could contribute to increased oxygen in bottom waters following an initial depletion (Kelly *et al.* 1993a,c).

Related to this, Figure 5-3 shows an interesting trend, using a single nearfield station as an example. The DO concentration for the sample closest to the surface (about 1.5-2.5 m below) and the sample nearest the bottom (about 5 m above) were strikingly similar, differing only slightly in July and August, when the bottom waters were actually a bit higher in DO. Since diffusive flux is a function of a concentration gradient, little flux between surface and bottom waters can be implied from data shown in Figure 5-3; the changes in bottom waters during stratification therefore would not seem to be artificially low due to atmospheric-originating, diffusive-driven replenishment and would seem instead to be a function of pycnocline metabolism and/or advection. When examined as % of saturation (Figure 5-3), the bottom sample was characteristically below saturation during the stratified period. The degree of saturation was lower in spite of similar or higher concentrations because the bottom waters are much colder during this time, so the saturation value is higher. This observation may provide an argument to use the saturation deficit and a concept like Apparent Oxygen Utilization (AOU) (Redfield, 1958) in an effort to characterize changes in DO that are due solely to biological processes. However, in this case where, over the summer some advection may occur, a more sophisticated approach seems unwarranted.



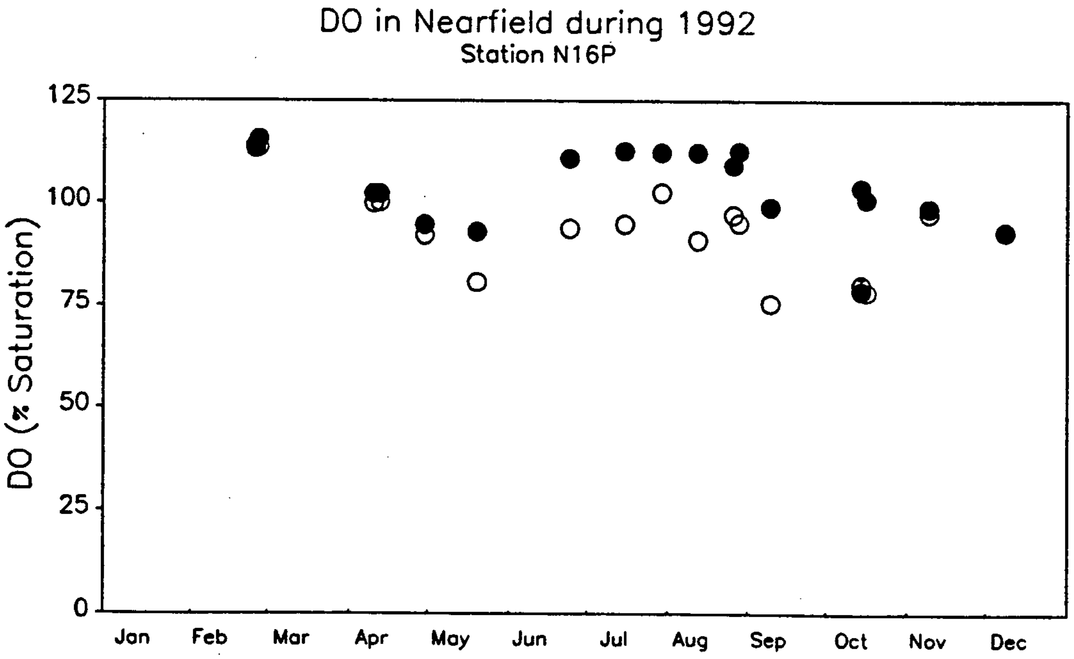


Figure 5-3. **DO, concentration (top) and % saturation (bottom) in surface (dots) and near-bottom (open circles) samples at Station N16P during 1992.**

If partial or full resaturation of DO does occur, by any mechanism, then neither production nor sedimentation of organic matter from the winter-spring bloom can be a reliable harbinger of the potential oxygen depletion later in the summer, contrary to expectations for a fully stratified system. The pace of dissolved oxygen change during seasonal stratification may be an important monitoring indicator, but the factors governing change need to better understood.

The fluctuation in DO in nearfield bottom water (Figure 5-2) during the stratified period (April to October) makes use of the time-trend data problematic for directly estimating *in situ* respiration rates. For example, using data from April to October, a linear regression of DO concentration on time is significant ($R^2 = 0.496$, $n = 349$), with the slope suggesting a decrease of 0.0136 (± 0.0007, stnd. error) mg O_2 L^{-1} d^{-1}. However, the rates for both the first DO decrease (April to May) and the second DO decrease (late July to October) are higher than this, roughly 0.037 to 0.060 mg O_2 L^{-1} d^{-1}. Assuming the range of these rates, 0.013 to 0.06 mg O_2 L^{-1} d^{-1} and applying them over a 15 m subpycnocline bottom water layer, one calculates oxygen consumption[5] as 0.2 to 0.9 g O_2 m^{-2} d^{-1}. Compared to an average primary production near 2.6 g O_2 m^{-2} d^{-1} (≈ 1 g C m^{-2} d^{-1}) during the stratified period, such rates, if indicative of respiration, would represent consumption of about 8 to 35% of overlying primary production. As for benthic fluxes, estimates may be improved as more monitoring data are accumulated and particularly with improvements in measurements of *in situ* respiration (below). Recognizing their limitation, these rough calculations, which include any benthic contribution to oxygen consumption, presently suggest a minor role for consumption of overlying production by the water and sediments below the pynocline.

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[5] E.g., 0.0136 mg O_2 L^{-1} d^{-1} x 15 m x 1000 L/m^3 = 0.2 g O_2 m^{-2} d^{-1}.

5.3 Time Trends of Bottom-Water DO in Stellwagen Basin

Using data for five Stellwagen Basin stations and focusing on waters deeper than 50 m, the trend for DO concentration in 1992 is shown in Figure 5-4. The pattern for these stations was similar over time, each event showing values within about 1 mg L^{-1} of each other and often much tighter than that. For these stations, there was less frequent sampling and none in May. The results suggest a more consistent, progressive decrease in DO for these bottom waters than was observed in the nearfield (cf. Figure 5-2). During stratification (April to October) the DO decrease was essentially linear ($R^2 = 0.88$, $n = 30$) and estimated as 0.015 (±0.001, stnd. error) mg O_2 L^{-1} d^{-1}, coinciding with an increase in bottom-water temperature from a low of 3.3 °C in February to a high of 7.2 °C in October. The rate implied for June to October was a bit faster, 0.023 (±0.001, stnd. error) mg O_2 L^{-1} d^{-1} ($R^2 = 0.98$, $n = 19$).

Interestingly, only in April and October was the bottom water lower in DO than the surface water (Figure 5-5). As for the nearfield, no large diffusive flux from the surface to the bottom seems implied, although flux from subsurface production within a deep chlorophyll maximum in the pycnocline may occur. However, an increasing saturation deficit is shown for the stratified period from April to October, compared to the surface waters which were always slightly supersaturated. In comparison to the nearfield, bottom waters may be somewhat sealed from the surface for the entire stratified period.

With the same cautionary notes as given for the nearfield, using the time trends (about 0.015 to 0.023 mg O_2 L^{-1} d^{-1}) as an approximation to respiration for a 30-m bottom layer (the total depth is about 80 m), an areal rate was calculated. The result is 0.45 to 0.69 g O_2 m^{-2} d^{-1}. Assuming a 50-m bottom layer (all subpycnocline water), resultant rates would be about 0.75 to 1.15 g O_2 m^{-2} d^{-1}. Calculated rates are thus comparable to results for the thinner, shallower nearfield bottom layer. Another perspective on the depletion is that, from the saturation deficit achieved between April and October, a net depletion of about 2.5 g O_2 L^{-1} is implied. This suggests consumption of about 0.9 g C L^{-1} assuming a Respiratory Quotient

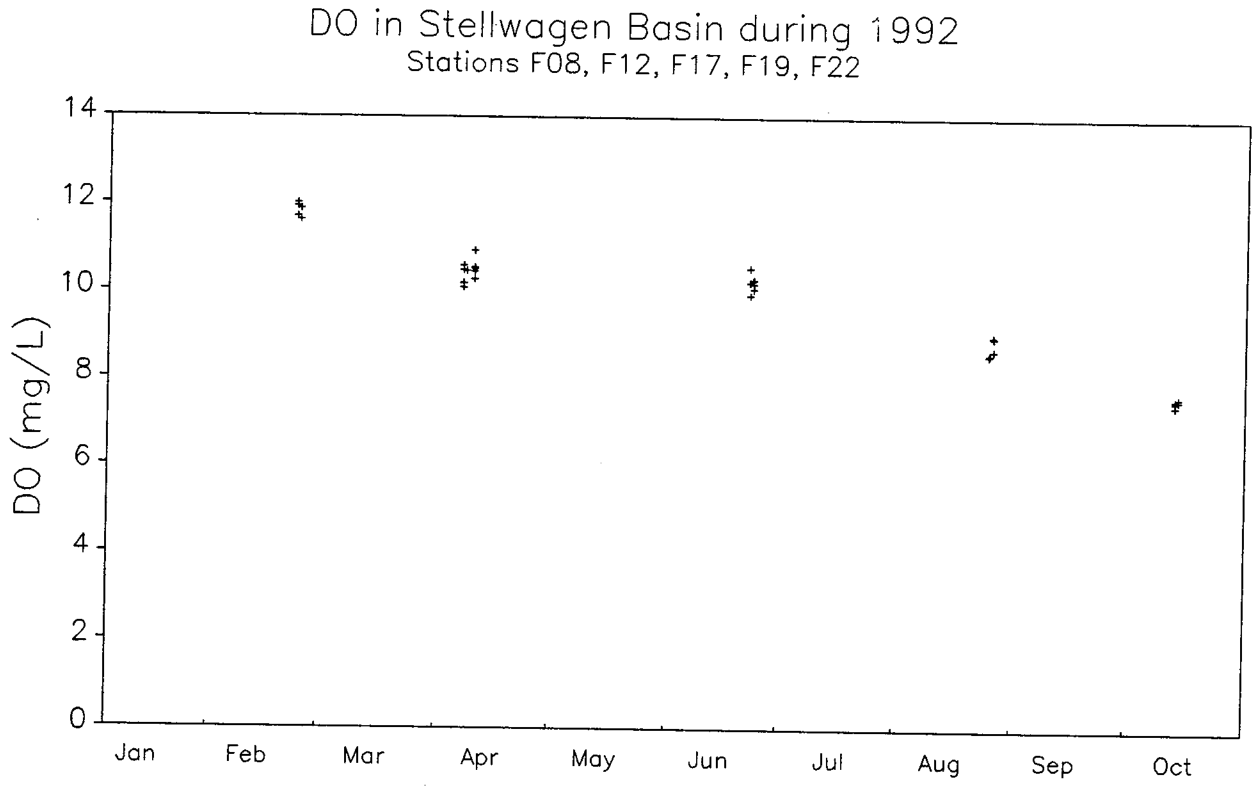

Figure 5-4. DO in bottom water (>50 m) of Stellwagen Basin during 1992.

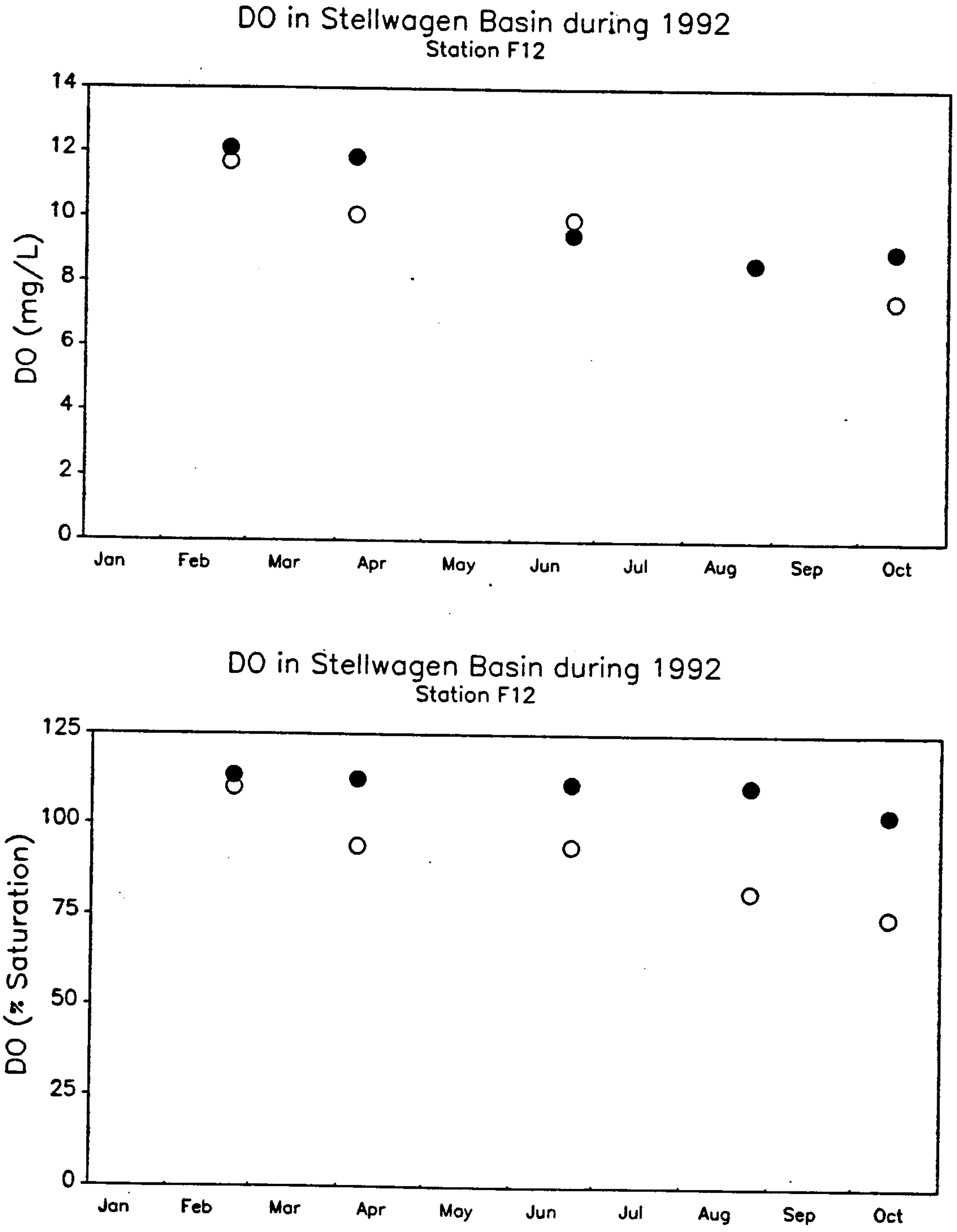

Figure 5-5. **DO concentration (top) and % saturation (bottom) in surface (dots) and near-bottom samples (open circles) at Station F12 in Stellwagen Basin during 1992.**

(RQ) of 1. For the 30-m bottom layer, roughly 27 g C m^{-2} would appear to be consumed by oxidative processes. The bottom layer and sediments thus might consume about 10% of the overlying water's annual primary production from about April to October, assuming that production is slightly lower than in the nearfield – in the range of – 250 to 300 g C m^{-2} y^{-1} over the Basin. These calculations, as for the nearfield, leave the impression that production, unless transported out of the area, is primarily consumed and recycled in the upper layers. Results of USGS efforts on sedimentation and particle transport may help clarify transport issues, but in the case of Stellwagen Basin, the observed subpycnocline oxygen decrease can be supported by production in the overlying water *without* significant import of organic matter that was produced elsewhere.

Calculations notwithstanding, there seem at least three interesting aspects of this review with respect to monitoring of Stellwagen Basin DO. One important observation is that the rate of DO decrease in Stellwagen Basin is quite low and therefore well-monitored by repeated surveys during the year at about the frequency conducted in 1992. A second important observation is that the five stations, arrayed on the axis of Stellwagen Basin running from well north to well south of the nearfield (see Section 2), had similar time trends for bottom waters in 1992. Tentatively, the rough synchrony across stations could be evidence that processes shaping DO changes were more a function of *in situ* respiration and vertical (diffusive and sedimentation) exchanges than a function of directional advection of water or import of organic matter. From the monitoring perspective, the Basin's seeming axial bottom-water similarity, if consistent across years, provides a stable baseline against which to detect change from directional transport of oxygen consuming organic matter if significant transport does occur from the nearfield in the future. Thirdly, it should also be recognized that we do not know the lowest DO concentrations attained in 1992 in these deep waters of Stellwagen Basin. The water column was still stratified in October 1992 and no further sampling was conducted in the farfield until February 1993.

5.4 Scales of Variability

Calculated peak bottom-water respiration rates in either the nearfield or in Stellwagen Basin are on the order of 0.01 to 0.06 mg O_2 L^{-1} d^{-1}. Those rates are established from gradual changes monitored over months and are not without problems of interpretation. The monitoring program also includes measurements of subpycnocline respiration using dark BOD bottles and incubation times of about 6 hours. Unfortunately, the rates do appear to be low enough that this short incubation time is inadequate for directly measuring water respiration changes. A minimum detectable respiration rate in short incubations is about 0.1 mg O_2 L^{-1} d^{-1}, or as much as an order of magnitude greater than what actually may occur in bottom water. It is therefore not surprising that the bottle incubations have not been successful at detecting respiration rates in general and that they have been more successful in waters with high temperatures and high chlorophyll concentrations. In short, the 6-hour incubations presently do not fulfill the intended role of detecting the potential for lowered oxygen in cold bottom waters. The bottle incubation times could be increased, and/or time series studies conducted, to improve the ability to monitor, and therefore detect changes in, respiration rates and to provide better estimates for comparison to water-quality modeling. Moreover, it would be useful to have good rate estimates for water column processes to confirm the strong, but indirectly reached, impression from calculations presented here that most of the organic consumption appears in the upper water column of the seasonally stratified Massachusetts Bay ecosystem. Even without such rates, however, the present monitoring design maintains frequent visits to the nearfield area and the record of changes over time should be adequate to detect gradual and significant trends in DO.

6.0 Summary

Three principal eutrophication issues are addressed in this report. For each, the main results from available evidence and theoretical calculations are provided in this summary.

The first issue centers on the present and future influence of the MWRA outfall effluent on nitrogen distribution in Massachusetts Bay.

- The available evidence strongly and consistently supports the concept that most of the nitrogen in the MWRA effluent presently discharged to Boston Harbor is exported to the Bay.
- There are sharp differences in the form of nitrogen exported seasonally from the Harbor to the Bay. The implication from a variety of data is that organic N dominates the N exported from the Harbor during the summer, whereas DIN export is high and dominant in winter.
- Presently, the nutrients exported into the area surrounding the future diffuser track are delivered directly to the surface layers, especially during the stratified period and, thus, are completely available to the phytoplankton. In the future, the initial dilution will be faster than that presently occurring at the edge of the Harbor. Also, in the nearfield, vertical flux of nutrients to the surface productive layers is constrained during stratification by vertical density gradients. With the new outfall the vertical nutrient flux will increase; however, calculations suggest that the increased flux from bottom waters will still be much less than the present horizontal transport from inshore to surface layers surrounding the outfall diffuser. Thus, the overall nutrient supply to surface productive layers may decrease.

From these conclusions one can argue that the principal scale of monitoring for change is more local than regional. At the regional scale, the nutrients being discharged today are circulated throughout the Bays, just as they will be in the future. While monitoring efforts principally need to focus on local ecological changes that are likely and detectable, research and monitoring should also continue to identify factors that could influence broad-scale variability in nutrients in Massachusetts and Cape Cod Bays.

The second issue involves several facets of potential response by plankton to nutrients with diversion of the outfall to the offshore site.

- For 1992, the range in annual average surface chlorophyll fluorescence was about 1.4 to 4.3 μg L^{-1} at 46 stations in Massachusetts and Cape Cod Bays. The higher chlorophyll concentrations were found at the edge of Boston Harbor and, as with nutrients, there was a gradient of decreasing concentration of chlorophyll concentration with distance away from the Harbor into Massachusetts Bay.

- Significant linear regressions were obtained using the annual average chlorophyll concentrations versus the annual average concentration of different forms of nitrogen in the water column. Similarly strong trends were also evident at a seasonal scale, using data summarized for the surface layer of 21 nearfield stations sampled intensively during the stratified period of 1992. The regressions provide some predictive capability of the influence of nutrients on chlorophyll, but the capability is restricted because factors other than nutrients also influence chlorophyll along the Harbor-Bay gradient.

- Stations near the Harbor have, on average, chlorophyll concentrations that are less than might be predicted for their high nutrient concentrations. High turbidity may influence light availability near the Harbor and thus limit chlorophyll relative to clearer offshore waters.

- Some thought experiments were conducted to consider the nutrient-chlorophyll response around the new outfall. These employ regressions developed from the 1992 data, including one that, in principle, provides for possible interactive effects of nutrients and light upon chlorophyll. Results suggest that upon transfer of the present nutrient concentration within the Harbor to the middle of the nearfield, the chlorophyll concentration, on average, could increase by about a factor of two, but this does not consider the additional dilution imparted at the offshore discharge site. Moreover, since we now can project that a decrease in surface nutrient flux from Harbor export will accompany the effluent diversion, the projections of surface chlorophyll change include the possibility of a decrease, on average, throughout the nearfield. Therefore, the principal change due to relocation of the outfall may not be the average chlorophyll concentration, but a shift in the vertical distribution of chlorophyll to a lower depth. Such a shift would accompany the projected sharp decrease in the present supply from inshore to surface waters offshore, an increase in the future nutrient supply from nearfield bottom-waters, and presence of a relatively deep photic zone in the nearfield.

- Patterns between chlorophyll and salinity were revealed from analysis of the summer season and higher-resolution sampling in the nearfield/Harbor area.

Several lines of evidence suggest there is stimulation of chlorophyll concentration (perhaps 1-2 $\mu g \ L^{-1}$) in immediate receiving waters several kilometers outside the Harbor, a phenomenon which may relate to export of nutrients into an area with increased water clarity.

- Review of the influence of nutrients on plankton species indicates that the present capacity to formulate predictions is limited. The results from the few experimental studies that have examined the response of plankton communities to long-term nutrient enrichment have been complex, but blooms of noxious species and shifts in relative proportions of diatoms to flagellates have been infrequent and often have not followed scenarios inferred from short-term, simple experiments. 1992 monitoring data provide information relevant to the potential alteration of nitrogen/silicate ratios in the future nearfield area. Initial comparisons of stations with differing nutrient concentrations or ratios suggest that the relatively small changes in nearfield nutrient levels that are projected are within the bounds already experienced within the Bay. Thus, it would be difficult to conclude from the data that projected nutrient changes will yield major species composition shifts to nuisance/noxious forms.

The third issue concerns a characterization of metabolism, the status of bottom-water dissolved oxygen (DO) concentrations, and the factors influencing metabolism and DO in Massachusetts Bay.

- Variability in primary production is high and small changes in time and space will be difficult to detect. Rates of benthic metabolism do not indicate a major role for the benthos in the consumption of primary production within the nearfield region of the Bay.

- An analysis of the nutrient sources supporting primary production in the nearfield shows that nitrogen presently exported from the Harbor quantitatively dominates nitrogen input to the surface layers and could support virtually all the primary production. In contrast, benthic regeneration and diffusive fluxes across the pycnocline into euphotic surface layers supply only on the order of 10% of the needs of primary producers. In the future, the flux from inshore will be curtailed and the flux from bottom-waters to the surface layer will increase. Calculations suggest that the overall nitrogen to the surface layer will be substantially diminished and primary production may therefore decline in much of the nearfield.

- Bottom-water DO declines during the stratified season in the nearfield and in Stellwagen Basin. Sustained rates of consumption appear to be about 0.01-0.06 mg $O_2 \ L^{-1} \ d^{-1}$. These rates were estimated from gradual changes monitored over weeks to months, and may imperfectly estimate respiration rates. Unfortunately, attempts to measure water column respiration directly in bottle incubations were unsatisfactory. In 1992, rates were often below limits of detection using a short-term (hours) incubation method. Nonetheless, the data suggest that a relatively small fraction of primary production is presently consumed within the bottom waters of Massachusetts Bay. Preliminary calculations suggest that much of the organic matter produced in the surface layers may be consumed within the upper water column.

- Because changes occur slowly, DO concentration trends across surveys should suffice as a principal tool to unambiguously record DO and monitor its rate of change.

Even though bottom-water DO concentrations change slowly enough to be monitored by the present frequency of summer and early fall surveys, it is argued that improved estimates of water column rates of oxygen consumption would be beneficial to understand the ecosystem. Precise respiration rate estimates would enable us to confirm or refute the notion that consumption of organic matter presently occurs rapidly within the upper water column and would also provide data useful to validate or improve water-quality modeling and predictions.

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8.0 APPENDIX
TO

Nutrients and Massachusetts Bay: An Update of Eutrophication Issues

DATA SUMMARIES FOR NITROGEN AND CHLOROPHYLL IN 1992

DATA SUMMARIES FOR NITROGEN AND CHLOROPHYLL IN 1992

This appendix contains three tables, each of which summarizes nitrogen forms and chlorophyll at MWRA water column monitoring stations over a certain period of time and over a specified depth range. Values from these tables are used extensively in Sections 2 and 3 of the accompanying text report.

Table 1 presents data for all stations surveyed during the six farfield/nearfield surveys in 1992. These include surveys in February, March, April, June, August, and October. N is usually six, but varies with station: it is less than six if the station was not occupied at each survey and it is more than six for the six nearfield "P" stations because each was occupied more than once during some combined farfield/nearfield surveys. Means for various parameters are presented; means represent the simple average of the values determined on samples for the surfacemost Niskin bottle at each occupation. A sample N is provided which applies to chlorophyll and DIN, fewer samples were analyzed for organic nitrogen forms. The maximum depth of a this sample for these surveys is shown for each station. CHL is chlorophyll, post-calibrated from *in situ* fluorescence readings, and is in μg L^{-1}. BEAM ATTEN is beam attenuation, taken from *in situ* transmissometer readings, and has units of m^{-1}. DIN is dissolved inorganic nitrogen, which was measured at all stations. PON (particulate organic nitrogen) and TDN (total dissolved nitrogen, includes dissolved organic nitrogen [DON] plus DIN) was measured at select stations and TOT N (total nitrogen) is the sum of PON + TDN. The concentration for all N forms is μM.

Note that this surface annual average differs from an integrated surface layer annual average calculated previously by Kelly (1991) from Townsend *et al.* (1991) data.

Table 2 presents similar parameters for nearfield stations only, for the stratified period of 1992, including all seven surveys and station occupations from June to October. However, for this "surface layer" summary, all data from the surface Niskin bottle to the depth of the subsurface chlorophyll maximum sample (usually n=3 on each cast) have been used in calculating mean values. The units are the same as in Table 1; also reported here are the mean salinity, as well as the minimum chlorophyll and maximum chlorophyll values (for any individual reading) for each station.

Table 3 presents data for select stations only and for only the summer period, including only the two farfield/nearfield surveys in June and August 1992. This surface layer summary, as in Table 2, includes all data from the surface Niskin bottle to the depth of the subsurface chlorophyll maximum sample (usually n=3 on each cast) to calculate mean values. Units are the same as Table 1. Organic nitrogen forms were measured at the surface and subsurface chlorophyll maximum at all station except F24. Note that this table presents DON rather than TDN. DON + DIN = TDN.

Table 1: SURFACE ANNUAL MEANS FOR ALL STATIONS IN 1992

STATION	N	MAX z	MEAN CHL	MEAN BEAM ATTEN	MEAN DIN	MEAN PON	MEAN TDN	MEAN PON +DIN	MEAN TOT N
F01P	6	2.42	2.24	1.155	1.69	3.28	10.07	4.97	13.35
F02P	6	5.48	1.84	1.163	0.6	3.27	9.56	3.87	12.83
F03	6	2.3	2.83	1.323	1.52				
F04	5	5.05	3.13	1.119	0.76				
F05	6	2.5	2.28	1.151	1.65				
F06	6	2.44	2.03	1.044	2.3				
F07	6	4.67	1.83	0.986	1.78				
F08	5	3.17	2.3	1.147	1.11				
F09	6	2.2	1.99	1.123	2.35				
F10	6	3.78	1.43	0.878	1.85				
F11	6	2.34	1.72	0.982	2.03				
F12	5	2.4	1.62	1.164	1.46				
F13P	6	2.32	2.62	1.258	2.23	3.31	11.21	5.54	14.52
F14	6	2.26	2.33	1.44	2.75				
F15	6	2.5	1.8	0.992	1.47				
F16	6	2.98	1.87	1.185	1.92				
F17	5	2.4	1.79	1.111	1.6				
F18	6	2.7	3.46	1.81	2.17				
F19	6	2.2	2.05	0.935	2.03				
F20	6	2.3	2.9	0.971	1.465				
F21	6	2.19	3.04	0.912	1.83				
F22	6	2.4	2.28	1.047	1.675				
F25	6	3.36	3.88	1.81	3.47	4.73	11.04	8.2	15.77
N01P	10	2.7	3.52	1.264	3.08	4.41	12.34	7.49	16.75
N02	5	4.76	2.64	0.951	1.75				
N03	4	2.27	3.28	0.913	1.38				
N04P	9	3.6	1.57	0.905	2.17	3.56	9.04	5.73	12.6
N05	4	1.9	2.48	0.851	2.1				
N06	4	2.4	1.73	0.834	1.59				
N07P	11	5.36	1.88	0.961	2.71	3.79	9.62	6.5	13.41
N08	5	2.47	2.33	0.989	1.84				
N09	5	2.1	2.63	1.161	2.15				
N10P	11	3.22	3.73	1.67	3.05	5.01	14.06	8.06	19.07
N11	5	2.2	3.36	1.354	2.51				
N12	5	2.31	2.9	1.237	2.96				
N13	5	2.6	1.87	0.927	2.21				
N14	4	2.7	2.58	0.903	2.21				
N15	4	2.5	2.68	0.85	1.57				
N16P	10	2.7	2.15	0.925	1.86	2.95	10.2	4.81	13.15
N17	4	2.6	2.495	0.944	1				
N18	5	2.4	2.8	1.034	2.2				
N19	5	2.7	3.6	1.261	2.64				
N20P	11	2.84	2.46	1.102	2.43	3.94	11.86	6.37	15.8
N21	5	2.5	2.27	0.914	1.75				
F23P	6	2.8	4.26	2.11	6.77	6.28	17.83	13.05	24.11
F24	6	2.5	3.45	1.661	6.13				

Table 2: SURFACE LAYER MEANS FOR NEARFIELD STATIONS IN THE STRATIFIED PERIOD OF 1992

STATION	N	MAX z	MEAN SALINITY	MIN CHL	MAX CHL	MEAN CHL	MEAN DIN	MEAN BEAM ATTEN	MEAN PON	MEAN TDN	MEAN DON	MEAN TOT N
N01P	30	18.3	31.184	0.54	10.92	3.62	1.56	1.45	4.035	10.78	9.22	14.815
N02	21	21.7	31.282	0.34	6.31	2.82	1.07	1.229				
N03	21	21.3	31.326	0.26	4.6	2.02	0.53	1.04				
N04P	30	27.1	31.328	0.24	9.48	2.52	0.49	1.067	2.89	6.71	6.22	9.6
N05	21	26.4	31.289	0.28	4.17	1.82	0.63	1				
N06	21	23	31.317	0.27	3.79	1.72	0.47	1.716				
N07P	30	21.6	31.323	0.25	4.81	2.16	1.04	1.056	2.86	7.02	5.98	9.88
N08	21	16.9	31.228	0.3	4.12	2.18	0.39	1.157				
N09	21	15.5	31.231	0.81	6.07	2.71	0.55	1.355				
N10P	30	15.5	31.146	1.11	7.45	4.04	1.06	1.728	4.24	12.55	11.49	16.79
N11	21	14.7	31.148	0.9	11.95	4.48	0.71	1.69				
N12	21	15.1	31.116	0.43	16.8	4.76	1.04	1.716				
N13	21	16.9	31.184	0.42	13.6	3.69	0.68	1.549				
N14	18	15.2	31.177	0.63	5.4	2.5	0.71	1.23				
N15	21	20.5	31.288	0.17	3.94	2.04	0.62	1.063				
N16P	31	35.1	31.335	0.24	5.06	2.12	0.62	1.069	2.43	7.33	6.71	9.76
N17	21	20.6	31.248	0.39	3.56	2.08	0.88	1.133				
N18	21	17.1	31.241	0.88	4.2	2.57	0.29	1.278				
N19	21	14.8	31.203	0.6	7.33	3.25	0.58	1.436				
N20P	30	18.3	31.213	0.37	8.05	3.3	0.79	1.417	3.95	10.25	9.46	14.2
N21	21	15.2	31.269	0.17	5.02	2.35	0.36	1.159				

Table 3: SURFACE LAYER MEANS FOR HARBOR-NEARFIELD "P" STATIONS AND SPECIAL STATIONS IN SUMMER 1992

STATION	N	MAX z	MEAN CHL	MEAN SALINITY	MEAN BEAM ATTEN	MEAN DIN	MEAN PON	MEAN TOT N	MEAN DON
F13P	6	10.2	2.66	31.08	1.42	0.32	2.95	11.49	8.03
F23P	6	10.56	5.9	30.66	2.55	1.21	4.83	18.11	11.76
F25	6	7.92	5.78	30.91	2.39	0.42	3.54	13.34	9.2
N01P	9	18.32	4.44	31.2	1.51	0.49	4.04	12.93	8.23
N04P	9	20.43	1.41	31.35	0.89	0.61	2.55	9.2	6.48
N07P	9	20.73	1.75	31.29	1	1.9	2.29	9.7	4.51
N10P	9	13.86	4.74	31.01	2.04	0.73	4.6	16.2	11.22
N16P	9	20.81	1.62	31.33	0.97	0.35	2.43	9.75	6.81
N20P	9	13.71	3.34	31.19	1.36	0.33	3.88	13	8.57
F24	6	11.24	5.29	30.96	2.08	2.61			

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