Nutrients and Massachusetts Bay: A synthesis of eutrophication issues

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NUTRIENTS AND MASSACHUSETTS BAY:
A SYNTHESIS OF EUTROPHICATION ISSUES

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1.0 INTRODUCTION

The Massachusetts Water Resources Authority (MWRA) is planning the construction of a sewage outfall from the Deer Island Wastewater Treatment Plant. The new outfall will be located in Massachusetts Bay approximately 15 km from the Deer Island Plant at a water depth of 32 m. Improved sewage treatment, cessation of sludge discharge, and moving the wastewater discharge from within the confines of Boston Harbor is expected to result in a significant improvement in water and sediment quality within the Boston Harbor area. Operation of the new outfall is scheduled to begin in July 1995, initially with effluent from upgraded primary treatment; secondary treatment is scheduled to be phased in between 1996 and 1999.

The offshore outfall discharge will bring nutrients and organic matter directly into the Bay. Model projections suggest that these will cause little harm to the environment of Massachusetts Bay (EPA, 1988). Nevertheless, coastal eutrophication processes are complex and to a degree unpredictable, so eutrophication is a prime issue addressed by the MWRA outfall monitoring program, which involves baseline and postdischarge characterization. Surveys funded by MWRA over the past 2 years have gathered some of the most comprehensive data available to date on nutrients and water quality throughout Massachusetts Bay. These data can be used to update understanding of nutrient-related processes in the Bay, facilitate qualitative and quantitative predictions of ecological responses, and thereby guide development of monitoring activities that will adequately assess the actual influence of the outfall on the surrounding Massachusetts Bay environment.

1.1 PROBLEMS IN COASTAL EUTROPHICATION

In spite of uncertainties in a given bay's response to nutrient enrichment, it is axiomatic that nutrients influence algal biomass and productivity of water bodies. High nutrient loading to coastal waters can lead to excessive algal growth and to depletion of dissolved oxygen, particularly in bottom waters isolated from the atmosphere. Anoxia is perhaps the most dramatic endpoint of nutrient enrichment, but it is not the only concern. Changes in nutrient loads can also lead to changes in water clarity and can alter, directly or indirectly, the abundance, distribution, and mix of organisms, including the plankton and benthic communities and, thus, the food webs supporting fish and shellfish.

The response of a coastal water body to nutrient loading is complicated by physical factors, including flushing by water inflow. In principle, systems flushing faster can tolerate higher nutrient loads because nutrient concentrations and algal populations do not accumulate because they are exported. Also, systems that stratify, sealing lower waters from atmospheric gas exchange, inherently have higher potential for experiencing oxygen problems. Although a critical factor in determining responses to nutrient enrichment, water motion in coastal bays is highly complex. There is mixing of fresh- and sea-waters; strong tidal and wind forcing of currents; development of horizontal fronts as well as vertical stratification; variable hydrodynamical coupling of open deeper waters with associated subestuaries, embayments, marshes, and tidal flats; and usually an uncertain exchange of bay waters across the ocean/bay boundary. For such reasons, our present ability to predict coastal marine and estuarine responses to increasing nutrient loads is less advanced than it is for freshwaters.

Marine eutrophication also fundamentally differs from that in freshwater with respect to nutrient limitation. In marine waters, nitrogen rather than phosphorus usually seems to be the more limiting nutrient. The N/P ratio is often used as evidence of relative nutrient limitation; low N/P ratios reported for Massachusetts Bay (e.g., T. Loder in MWRA, 1990) are typical of marine waters and would suggest relative nitrogen limitation. Therefore, primary focus on nutrients in this report is on nitrogen.

1.2 PURPOSE OF THIS REPORT

This report examines and synthesizes some information from recent studies of Massachusetts Bay. The overall objective of the report is to develop guidance for monitoring the offshore sewage outfall relative to eutrophication issues.

Distributions of nitrogen, plant biomass, and oxygen in Bay waters and in Boston Harbor are examined, for this is basic information that is required to make an assessment of the consequences of changes in anthropogenic nutrient input. The primary data source examined was Townsend *et al.* (1990), a study that made measurements at stations throughout Massachusetts Bay through an annual cycle during 1989 and 1990. Other data sets are available. For example, with respect to nutrients and chlorophyll, there are data for nearshore and outfall regions only from summer 1987 and spring 1988 (MWRA, 1988, 1990) and a summary of historical data is given by Normandeau (1990). Some limited comparisons across data sets are made, but the scope and scale of the primary data set examined makes it particularly useful for a Baywide perspective.

The report structure is as follows.

1. Assessment of the present conditions and trophic status of Massachusetts Bay, with a primary focus on nitrogen, chlorophyll, and dissolved oxygen
2. Qualitative judgment on the ecological responses, with respect to eutrophication, of moving sewage discharge offshore from Boston Harbor
3. Discussion of appropriate indicators of eutrophication for Massachusetts Bay
4. Discussion of the implications of the findings for outfall monitoring and assessment of ecological changes in response to sewage discharge

2.0 NUTRIENTS AND THE MASSACHUSETTS BAY SETTING

2.1 NUTRIENT DYNAMICS

2.1.1 Nutrient Export from Boston Harbor into Massachusetts Bay

Nutrients flow out of Boston Harbor and must influence the nutrient budget of Massachusetts Bay. The quantity of nutrient export is difficult to establish, but it is argued below that the export is large, must strongly influence Massachusetts Bay, and has to be considered in assessing the present and future condition of the nearshore ecosystem, including the proposed outfall site. Thus, the initial focus of this section is on Boston Harbor.

Presently, MWRA sewage effluent is discharged directly into Boston Harbor. The total nitrogen loading to the Harbor (Menzie-Cura, 1991; Appendix A) is among the highest estimated for shallow estuarine and coastal waters (Figure 1), yet the Harbor does not have *in situ* levels of dissolved inorganic nitrogen (DIN = nitrate plus nitrite plus ammonium) that reflect the loading rate. Mean values for eight Harbor stations repeatedly visited over 2 years were 11.0 μM for surface waters and 9.4 μM for bottom waters [Robinson *et al.*, 1990 (their Table 8, p. 38)]; an equal mix of surface and bottom strata suggests a mean annual Harborwide average of 10.2 μM of nitrogen as DIN. Inclusion of phytoplankton particulate organic nitrogen, as estimated from chlorophyll (Appendix A), adds roughly 1.15 μM N. The annual average concentration of DIN + PON, or the N easily "available" to or already assimilated by phytoplankton, is estimated as 11.35 μM N. Given the annual volumetric loading of 1490 mmol/m^3 (= 1490 μmol/L), even assuming that only some fraction (as low as about 50%, Appendix A) of the total N load were readily available, the average Harbor concentrations are very low and a great deal of incoming N is unaccounted for within the water.

The budgetary discrepancy could in part result from removal and burial in Harbor sediments. Some long-term retention undoubtedly occurs; however, recent nitrogen budgets of some coastal bays suggest that bottom sediments store probably only a small fraction of nutrient input (e.g., Nixon, 1987). Most likely, the paradox of high loading and not so high *in situ* N concentrations in Boston Harbor [like Buttermilk Bay (Valiela and Costa, 1988)] arises because there is a high rate of flushing and concomitant export of nitrogen out of the Harbor. Since rapid flushing has implications for the present nutritional and trophic status of Massachusetts Bay, it is briefly examined.

The water residence time for the whole Harbor calculated by the freshwater fraction replacement method (Appendix A) is roughly 10.5 days. Using a tidal prism approximation method, the approximate half-time for volume replacement is a day or two, with about 97% of the Harbor volume exchanged in under 8.5 days (Appendix A); since some water just sloshes in and out of the Harbor and doesn't completely mix, flushing may be underestimated by this method. A hydrodynamic model of the Harbor developed by the [U.S.] Geological Survey (USGS) suggests a value of about 10 days as an upper bound for the whole Harbor; parts of the Harbor, of course, flush more rapidly and some more slowly (R. Signell, personal communication). The time scale for flushing appears to be on the order of days.

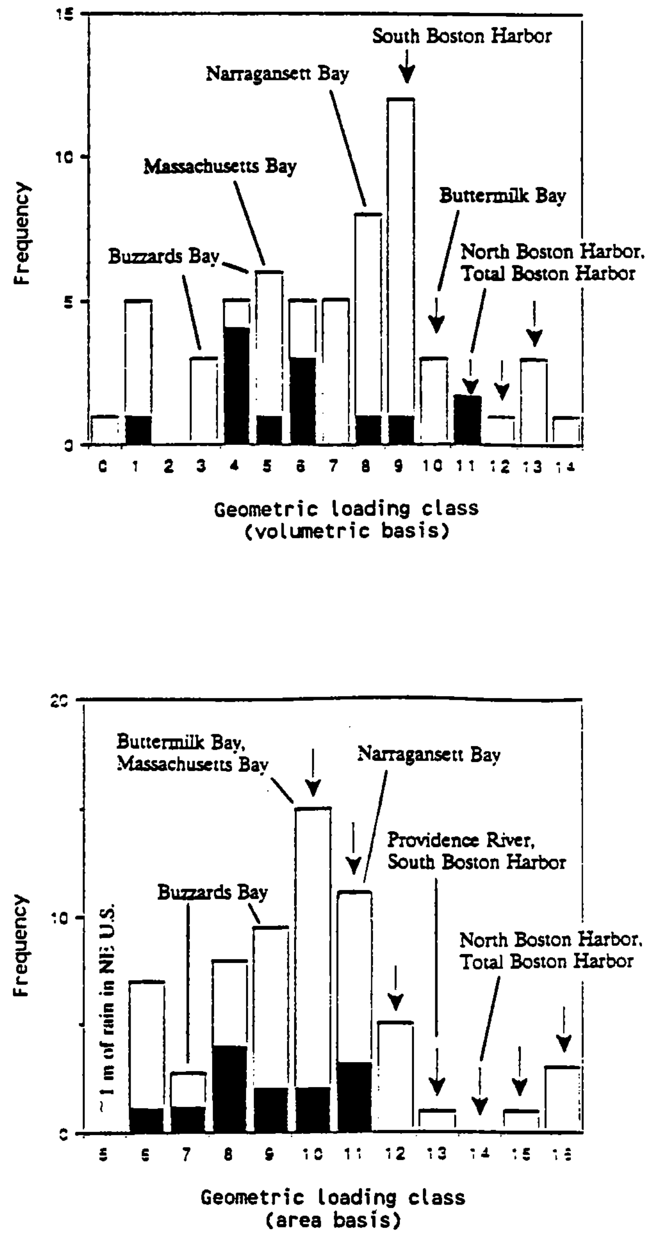


Figure 1. Geometric N Loading Class Frequency Distribution for Coastal Lagoons, Estuaries, Bays, Bights, and Seas.
[Modified from Kelly, 1990.]

The range for loading to Buzzards Bay is from Kelly *et al.* (1991). Buttermilk Bay is from Valiela and Costa (1988); Massachusetts Bay is from NOAA/EPA (1988); and the probable range is further discussed in Appendix A. Boston Harbor was calculated from Menzie-Cura (1990). A geometric class x is an interval range, defined as greater than 2^{x-1}, but less than or equal to 2^x. For example, class 7 is $>2^6$ and $\leq 2^7$, 65 to 128 mmol N m^{-2} (or m^{-3})y^{-1}. An uncertainty of about ± 1 loading class is expected at the midrange of the distribution. Arrows indicate classes represented by MERL microcosm gradient experiment (Nixon *et al.*, 1986). The dark bar histogram is the frequency distribution of loading classes for which annual benthic fluxes in the field (n = 13) also have been measured. For reference, the inorganic nitrogen content in about 1 m of rain input per year to a flat surface would provide a loading represented by about class 5.

One can also calculate water-column nitrogen residence times, using loading and *in situ* mass (Appendix A). The replacement time ("turnover time") for DIN + PON is about 3 days, assuming that 100% of N input is available, or 6 days if only 50% is available (Appendix A). The qualitative conclusion is that water-replacement rates and nitrogen-replacement rates are on comparable time scales; the implication is that the flushing action of water probably regulates nutrient turnover (cf. Kelly *et al.*, 1985), thereby causing export of a substantial portion of the nutrients presently added to the Harbor. The present discharge of sewage carrying the bulk of the Harbor's nutrient loading (Menzie-Cura, 1991) is to outer-Harbor areas, which are very dynamic regions with flushing higher than average. This fact lends additional support to the notion of rapid nutrient export.

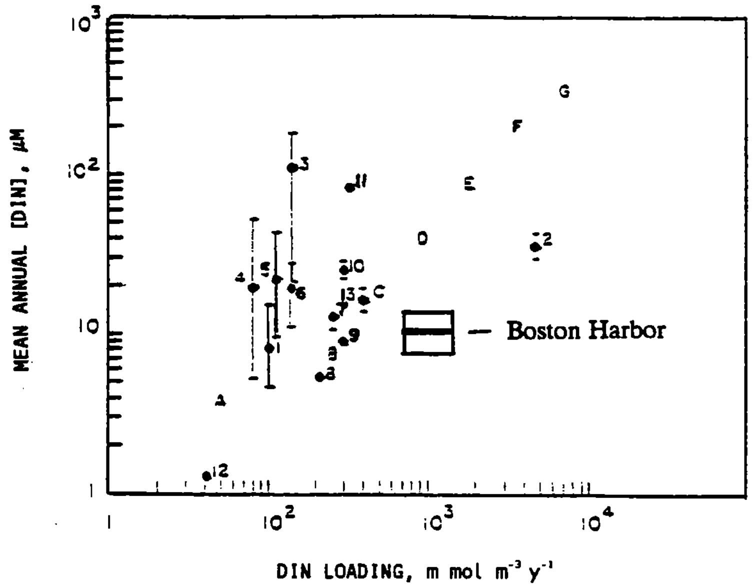
Firm quantification of nitrogen export may await budgeting of nitrogen buried in Harbor sediments, dredged and removed, or lost to the atmosphere through denitrification. Nevertheless, qualitatively we can compare the Harbor to some other coastal areas, with respect to nitrogen loading and mean annual DIN (Figure 2). The *in situ* Harbor DIN concentration, if uncorrected for flushing (top panel), is low relative to less loaded coastal ecosystems. If corrected for flushing (a range of 2 to 10 days is used for illustration, bottom panel), the Harbor mean DIN values tend to fall right in line with the pattern of a number of other coastal areas. This chemical evidence strongly supports the notion that flushing is significant on the time scales of days and export of much of the N delivered to the Harbor must follow.

2.1.2 An Enrichment Gradient into Massachusetts Bay

There is other, compelling evidence of the Harbor export of nitrogen to the waters of Massachusetts Bay. Townsend *et al.* (1990) reported vertical profiles of concentrations of various water-quality parameters at stations in Massachusetts Bay along three transects (Figure 3), each running from the nearshore (starting just east of Deer Island, off Marblehead, and northeast of Cohasset Harbor) to the eastern edge (or northeastward) of Stellwagen Bank. Using those data, mean values over the year for surface waters at 14 stations have been calculated (Appendix B). Depth-integrated annual means have been especially useful in characterizing eutrophication gradients in shallow coastal ecosystems (e.g., Oviatt *et al.*, 1984; Nixon *et al.*, 1986). It is desirable to have more frequent sampling intervals than were conducted during the Townsend surveys to arrive at an annual mean; values calculated and presented here must be considered approximate, but they will suffice to provide a synthesis revealing some broad patterns across the Bay.

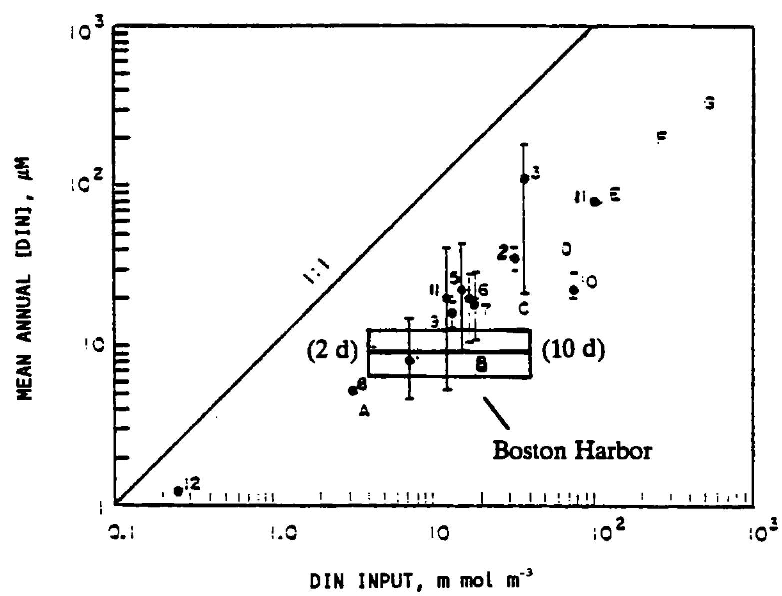
When displayed as contour maps of the Massachusetts Bay area, the Boston Harbor area is graphically evident as a strong point source of nitrogen radiating out into surface waters of Massachusetts Bay (Figure 4). Annual mean surface concentrations of DIN in micromoles per liter (μM), particulate organic nitrogen (PON, reported in micrograms per liter), and the sum of DIN plus PON [all calculated as micromoles per liter (μM)] each suggest a sharp exponential concentration gradient with distance away from the Harbor.

During 1989-1990, the station (6) closest to Deer Island (a main entrance into Boston Harbor and a point of sewage discharge) had a DIN concentration remarkably similar to the average for the Harbor (1987-1988), especially the Northern Harbor (Robinson *et al.*, 1990). Moreover, the DIN values given by Townsend *et al.* (1990) for Station 6 are similar to the DIN



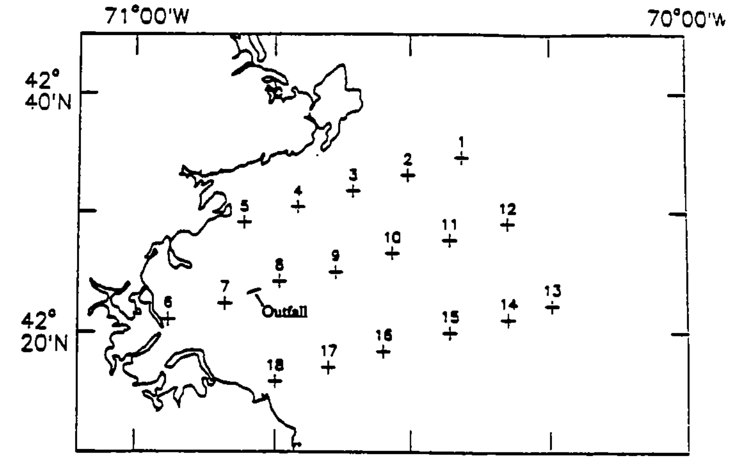
(a) Concentration Data Vertically and Horizontally Averaged over Most of the Stations Sampled in Each System.

Where possible, the range between upper and lower reaches is shown by a bar. 1: Narragansett Bay. 2: New York Bay. 3: Delaware Bay. 4: Chesapeake Bay. 5: Patuxent Estuary. 6: Potomac Estuary. 7: Pamlico Estuary (mid and lower portions only). 8: Apalachicola Bay. 9: Barataria Bay. 10: Northern San Francisco Bay. 11: South San Francisco Bay. 12: Kaneohe Bay. 13: Mobile Bay. A-G: MERL microcosms, June 1981 - June 1982.

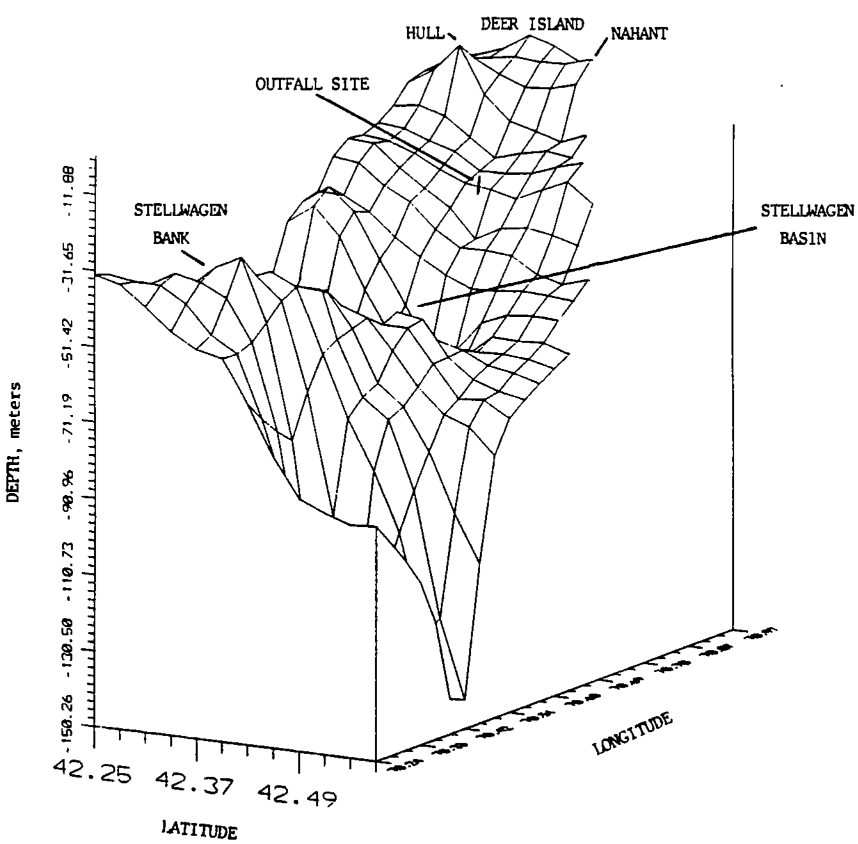


(b) Here, Inputs Have Been Multiplied by the Hydraulic Residence Time To Calculate Annual Loading.

Figure 2. Mean Annual Concentrations of Dissolved Inorganic Nitrogen As a Function of the Estimated Annual Input of Inorganic Nitrogen. [From Nixon, 1983.]

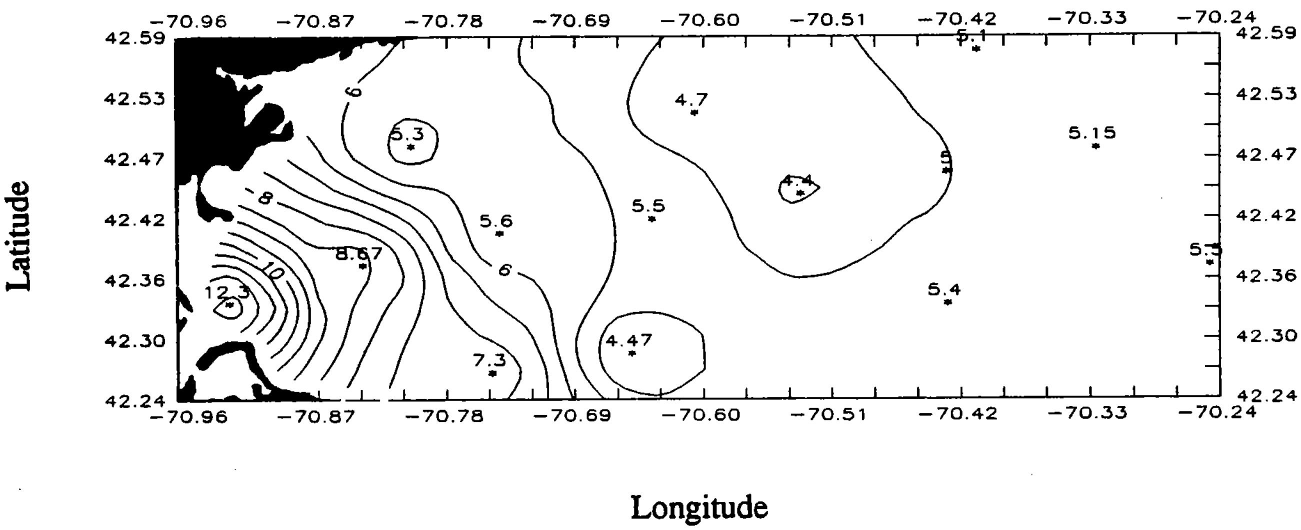


(a) Station Locations.

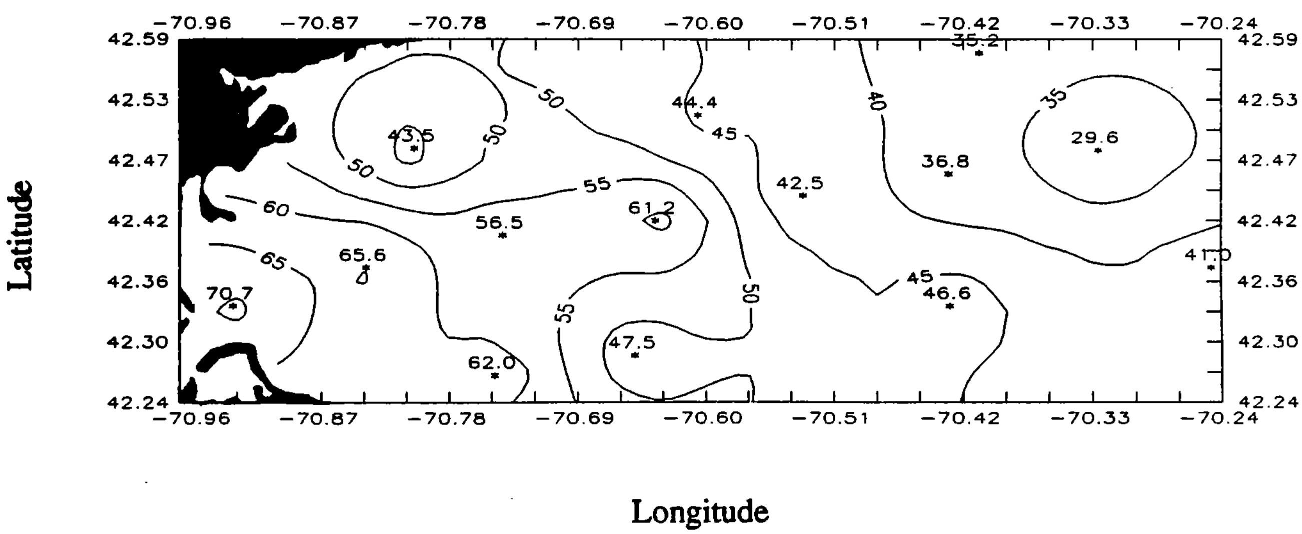


(b) Bathymetry Viewed from NE Corner of Bay (Station 1) Looking into Boston Harbor.

Figure 3. Station Locations of Townsend *et al.* (1990) in Massachusetts Bay.

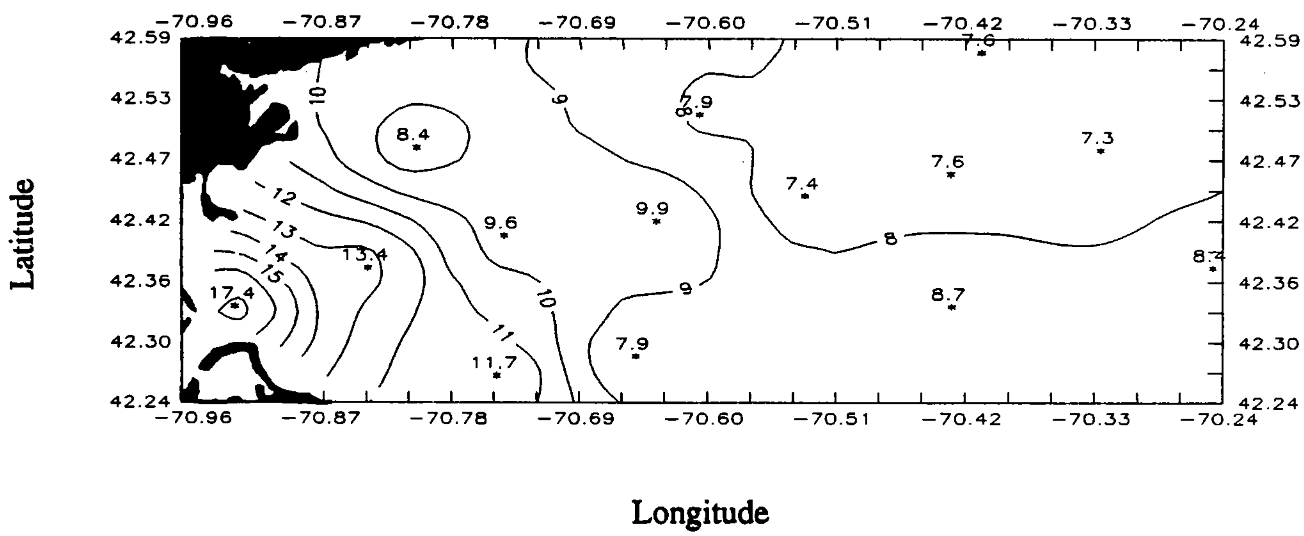


(a) DIN (μM)



(b) PON (μg)

Figure 4. Annual Surface Water Integrated Averages of Nitrogen at the Stations Sampled by Townsend *et al.* (1990) in Massachusetts Bay.



(c) DIN and PON (μM)

Figure 4. Annual Surface Water Integrated Averages of Nitrogen at the Stations Sampled by Townsend *et al.* (1990) in Massachusetts Bay. (continued)

concentration values reported for Broad Sound by Robinson *et al.* (1990; their Station 11). There was a progressive decrease of annual mean DIN outward from the Harbor along the middle transect of Townsend *et al.* (1990); nitrogen-enriched surface water extended to Station 8, which is eastward of the site of the proposed outfall (Figure 3). The area of nearshore enrichment extended to the south, encompassing Station 18 off Cohasset, but not to the north as far as Marblehead.

Interestingly, offshore surface waters in the euphotic zone (about the top 15 to 25 m) had about one-third to one-half the DIN concentration of the Harbor and adjacent waters ($\approx$6 m water depth) so the average standing mass (although not necessarily the flux) of N for primary producers may be about comparable. Judging this comparability as evidence of simple volumetric dilution throughout the Bay would neglect other possible points of nutrient input to the Bay; nonetheless, for budgeting purposes, mass and not concentration is an appropriate measure.

The PON contours suggest enrichment beyond Station 8, and the gradient with distance offshore is less sharp than that shown for DIN (Figure 4). Annual mean PON values were lowest at Station 12 at the eastern edge of the middle transect, slightly northeast of Stellwagen Bank in deep water. Inherently, one would expect PON exported from the Harbor to be settling from the water and therefore not be transported as far as dissolved constituents; if so, a gradient from a source would be sharper for PON than for DIN. In Massachusetts Bay, the more diffuse gradient for PON may indicate that PON is being produced in offshore surface waters *in situ* from rapid assimilation of DIN into phytoplankton N biomass. During summer stratification when surface outflow from the Harbor could be ejected the greatest distance out into the Bay, high chlorophyll levels were found in the region of the proposed outfall and farther offshore (Stations 7-9; Figure 5).

A steep gradient from the Harbor was also apparent using combined DIN + PON (Figure 4) because, on average, the combined N is dominated by DIN. PON made a greater contribution to the combined value at mid-Bay stations (8-10), about 41%-44% of the combined amount on average. At the shoreward stations, it was about 29%, 35%, and 38%, respectively, of the combined N at Stations 6, 7, and 18. Highly nutrient-loaded shallow coastal systems may have an even lower fraction of the combined N packaged in particles (Kelly *et al.*, 1985; Kelly and Levin 1986).

A nitrogen-"enriched area" extended outward roughly to Station 8 and southward from there to Station 18 (Figures 3 and 4). Within this region, water-column N was above offshore "background" concentrations ($\approx$ 5.5 to 6 μM DIN and $\approx$ 9 to 10 μM DIN + PON). This region is almost twice the area and well more than twice the volume of the Harbor. Using the mass of N above background in this region, a simple calculation suggests that the elevated N could be sustained easily by high nutrient outflow from the Harbor (Appendix A). The calculation suggests that the mass of N could be replaced in several days if all of the present load to the Harbor were being exported. To maintain steady state under such conditions requires that the average residence time of water in this area be a comparable length of time, i.e., days. Some recent physical investigations suggest water residence times for this area, which includes the outfall site, that may be similar to this time scale (R. Signell, personal communication). The inference is that nutrients presently ejected from the Harbor are subject to continued dispersal farther out into the Bay.

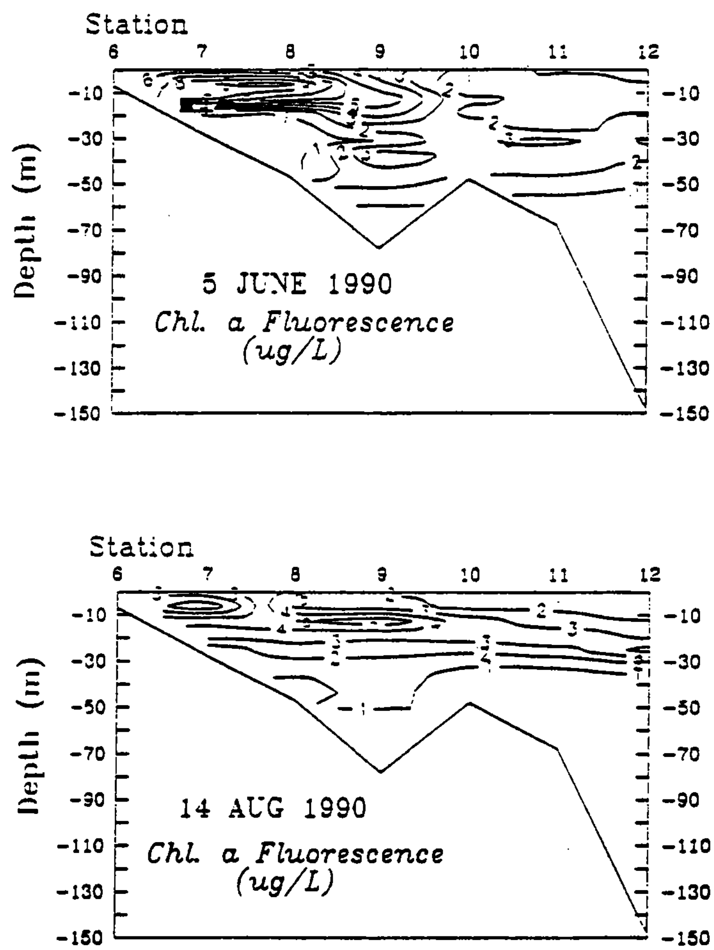


Figure 5. Chlorophyll Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay.
[From Townsend *et al.*, 1990.]

2.1.3 Seasonal and Vertical Nutrient Dynamics in Massachusetts Bay

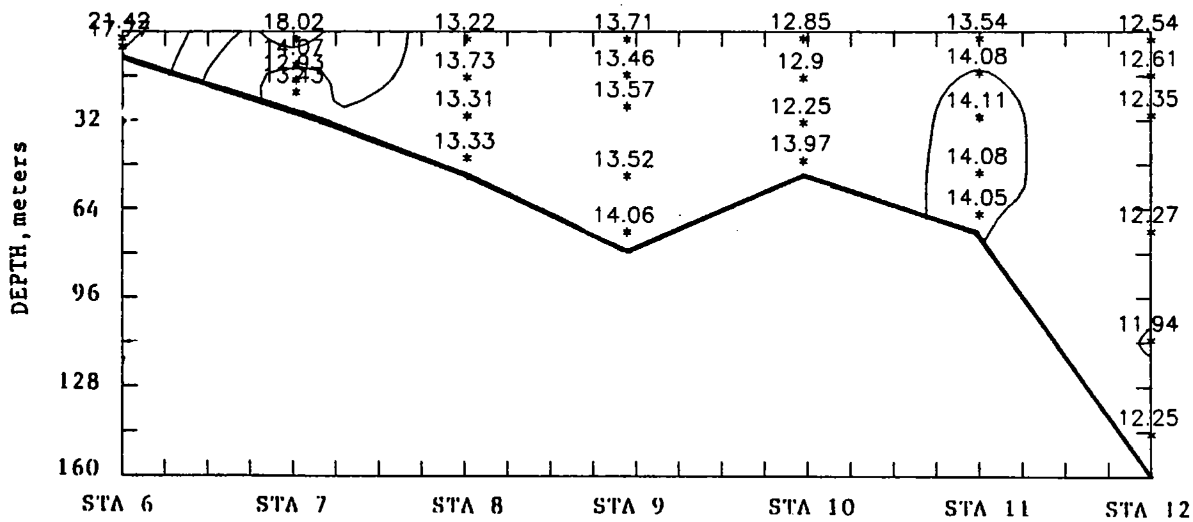
Besides annual values, seasonal views of water-column nutrient concentrations tell something of the dynamics of Massachusetts Bay relative to the Harbor. The seasonal progression of N concentrations across Massachusetts Bay starts with generally highest DIN + PON concentrations (only about 2% to 5% PON) during winter when the waters are vertically mixed and before the spring bloom is initiated (Figure 6). In February, combined DIN + PON was generally 13 to 14.5 μM throughout the Bay, except for higher values generally at Station 6 and the surface sample at Station 7. By April, there was conversion of some DIN into PON, and, assuming only small conversion to dissolved organic nitrogen forms, an apparent loss of N (as much as 5 to 7 μM) from the surface water. By June, the gradient of DIN + PON away from the Harbor extended farthest offshore of any of the months sampled. Except for the surface of Station 6, PON made up over 95% of the DIN + PON in surface samples throughout the area. By August, surface DIN values were still depleted (except at Stations 6 and 7). At that time, several inshore stations still had relatively high surface PON values (Figure 7), but most above-pycnocline DIN + PON concentrations had fallen to about 3 to 5 μM. With some mixing from the surface and deepening of the pyncocline by October ($\approx$ 40 to 50 m), the surface waters outward of Station 7 had begun to be renewed with dissolved inorganic nitrogen from below, and PON made up generally less than 50% of the DIN + PON value.

Ignoring horizontal features of water mixing and advection that may bring nutrients in and out of the Bay, a seasonal cycle thus could be depicted as a process of depletion of high wintertime levels of surface nitrogen via particle fallout from euphotic surface layers during spring through late summer. Renewal of the surface-originated nutrients occurs from vertical mixing in the fall through winter. Renewal from bottom waters may be a "recycled" source of nutrients in the sense that nutrients removed from the surface via particle settling get distributed back to the surface, perhaps at a different time and in a different place within the Bay. A number of processes — vertical mixing, diffusion of nutrients across a pycnocline or thermocline, "pumping" of nutrients by internal waves at a pycnocline, or active upwelling — do not necessarily infuse "new" nitrogen input to the broad-scale ecosystem whose boundaries are defined as all of Massachusetts Bay.

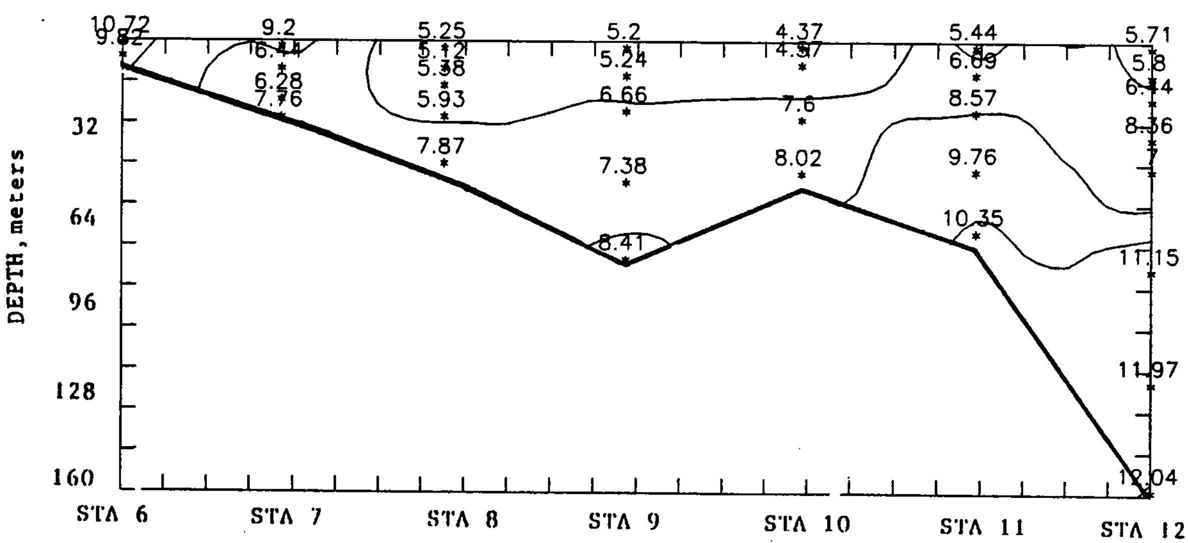
From a high of about 14 μM DIN + PON throughout the Bay in February to a low of about 4 μM within the upper 25 m of the water column in August, the difference implies net removal of about 10 μM during the productive season. How does this amount compare to other processes in the nutrient budget? Converted, using Redfield stoichiometry, to organic carbon lost to bottom layers, this represents 19.9 g C/m^2 or about 5.7% of an annual net primary production of 350 g C/m^2 (cf. Cura, 1991; Michelson, 1991). Actual removal from surface waters must be higher because there is input to the surface waters that is not accounted for by the above calculation.

If we assume that an amount of N equivalent to the Harbor input also comes into the Bay's surface water, when spread across the surface area of Massachusetts Bay it amounts to about 422 mmol N/m^2/year. If also removed to bottom waters and sediments over the year, this is equivalent to 33.6 g C/m^2/year.

By summing removal as calculated from standing stock depletion and input, the total estimated organic carbon removal to bottom waters below 25 m would be 53.5 g C/m^2 (i.e., 19.9 + 33.6), or about 15% of the *in situ* production of 350 g C/m^2 for the year. Generally, there is only a

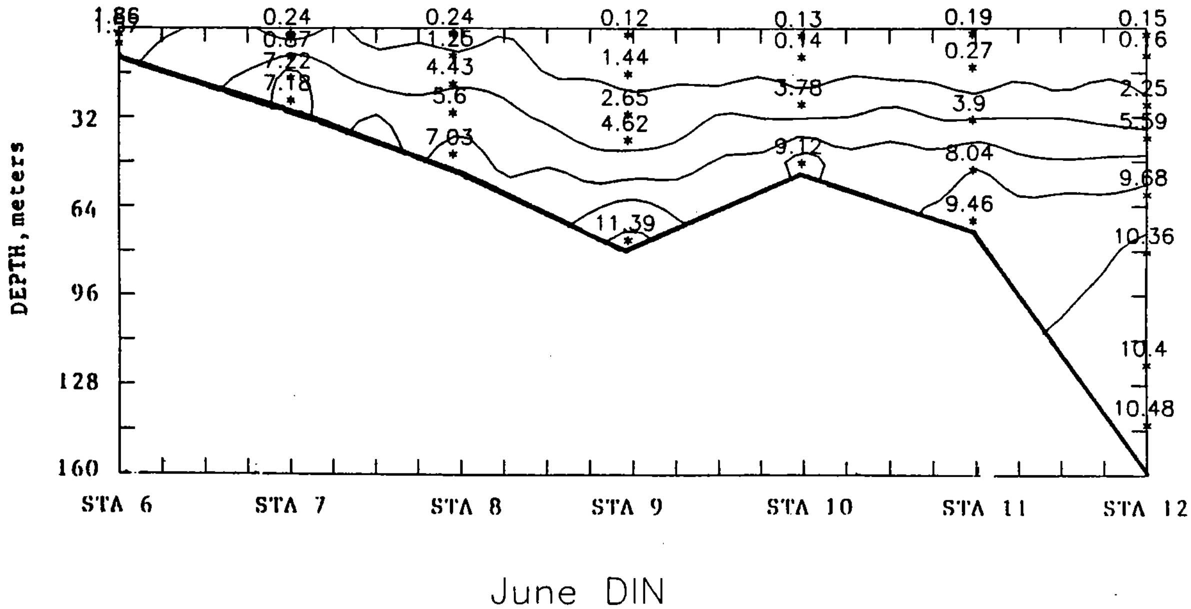


February DIN



April DIN

Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM].
(a) DIN by Months (February, April)



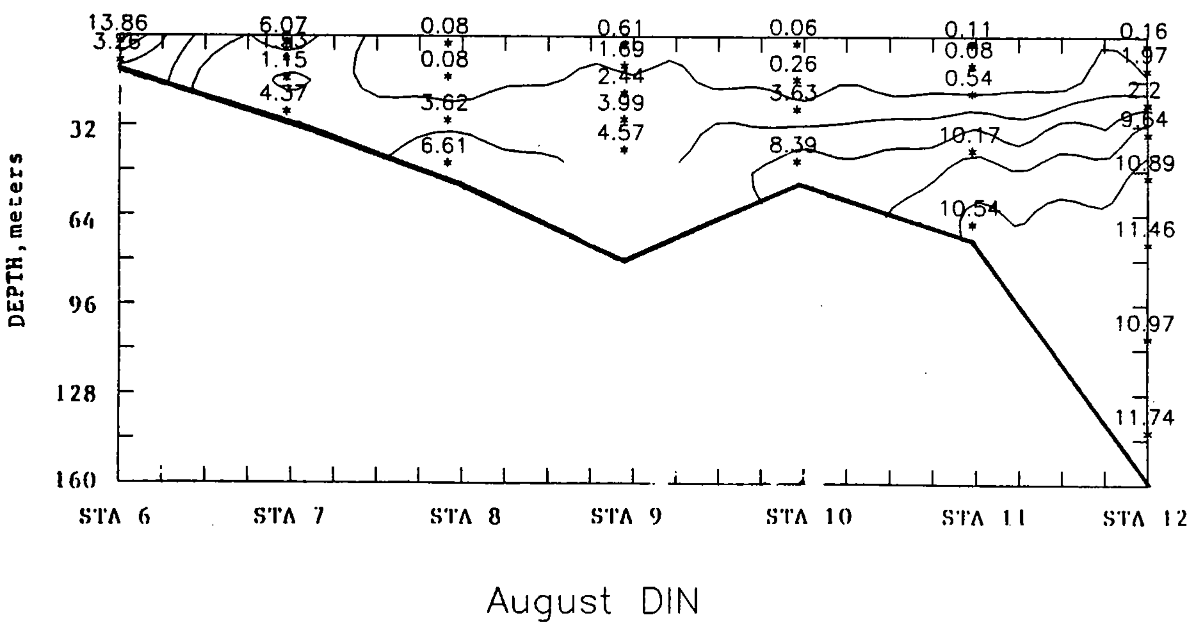


Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM]. (continued) (a) DIN by Months (June, August).

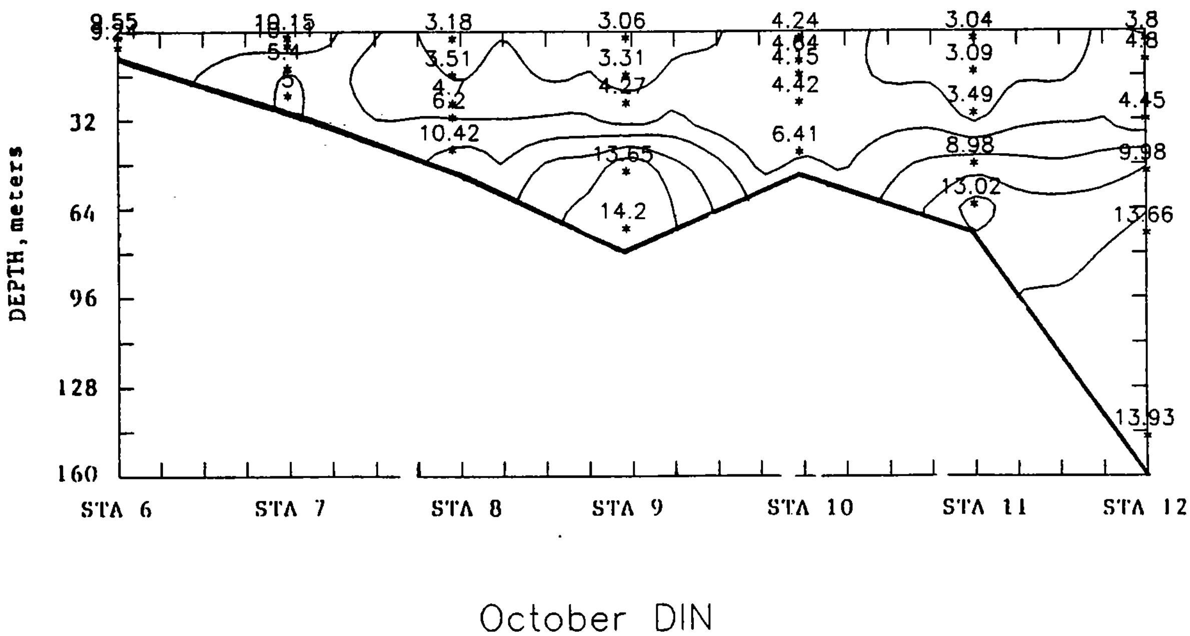


Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM]. (continued)
(a) DIN by Months (October).

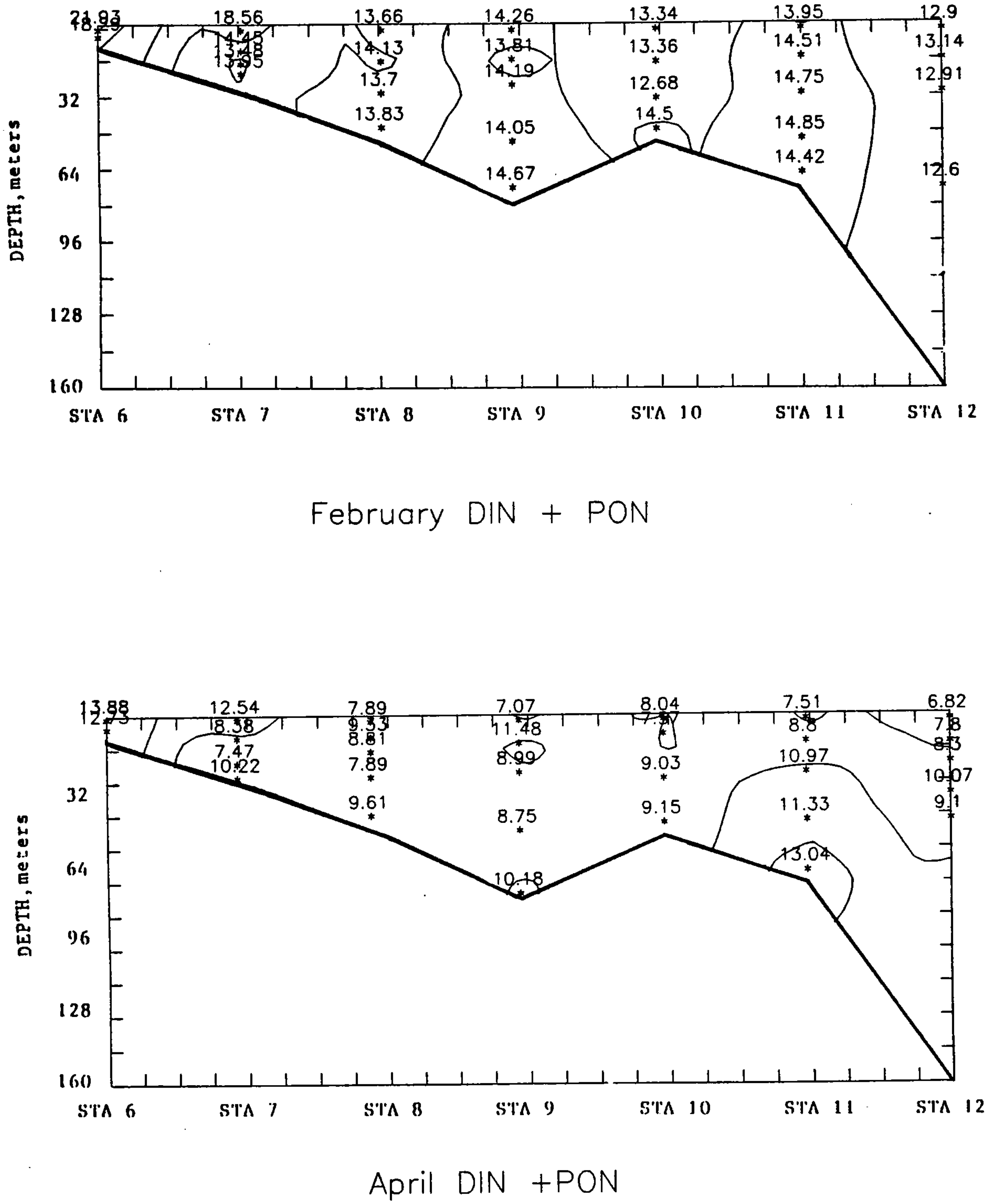


Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM]. (continued) (b) DIN + PIN by Months (February, April).

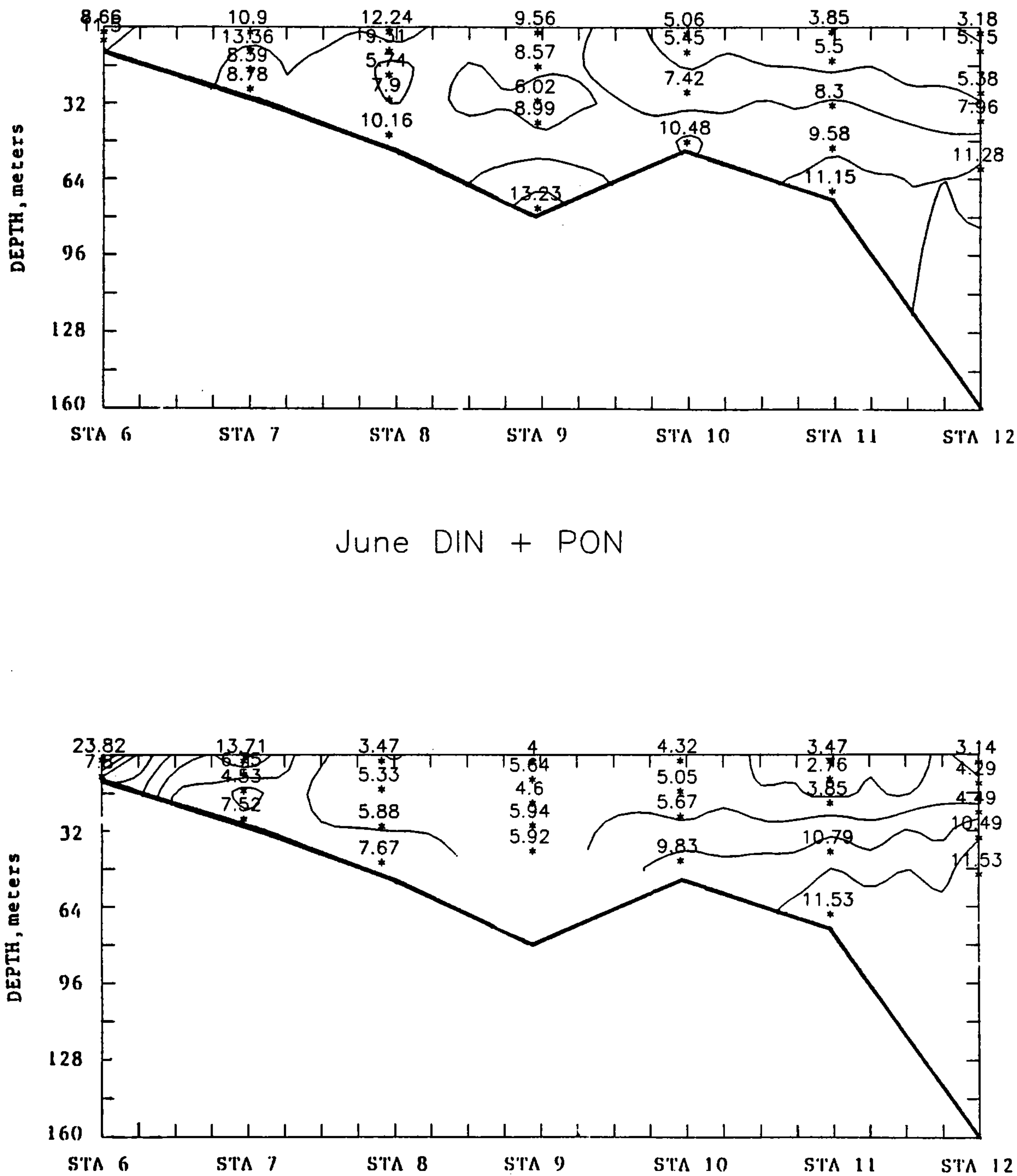


Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM]. (continued)
(b) DIN + PIN by Months (June, August).

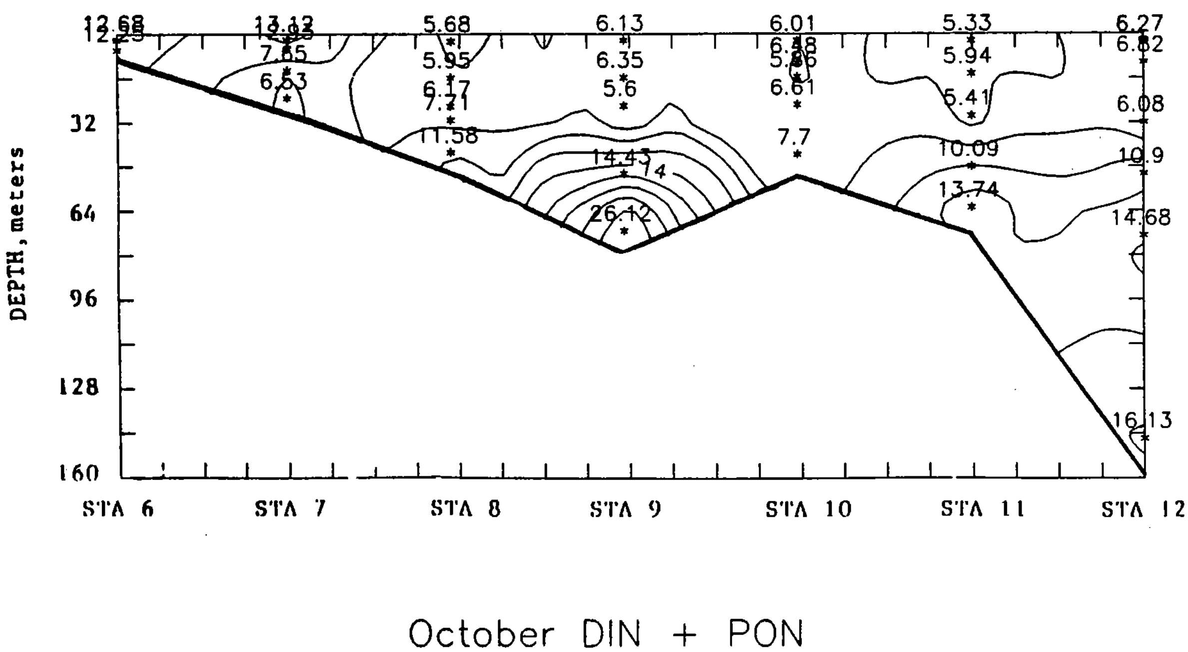


Figure 6. Nitrogen Profiles across the Middle Transect of Townsend *et al.* (1990) in Massachusetts Bay. [Contour interval: 2 μM]. (continued)
(b) DIN + PIN by Months (October).

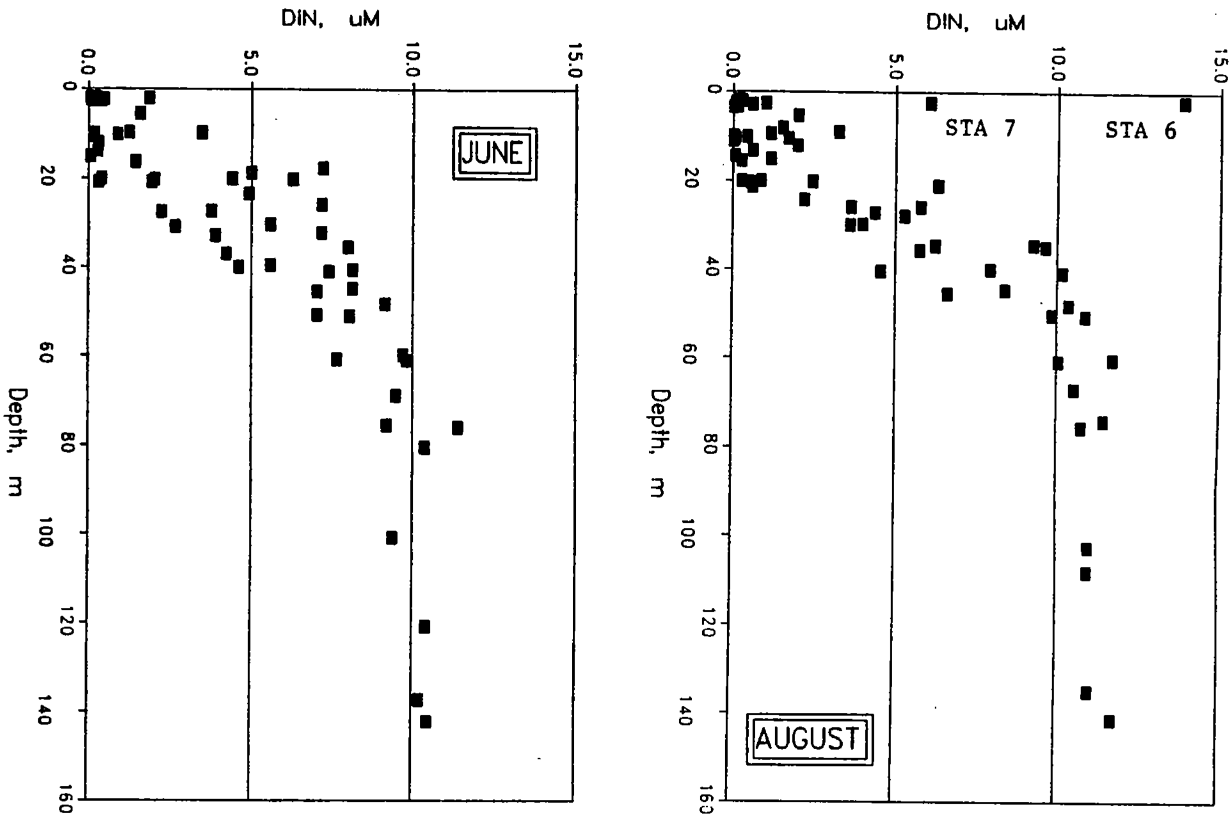


Figure 7. Nitrogen Concentrations vs Depth in Massachusetts Bay.
[From data of Townsend *et al.*, 1990.]
(a) DIN (μM) in June and August 1990.

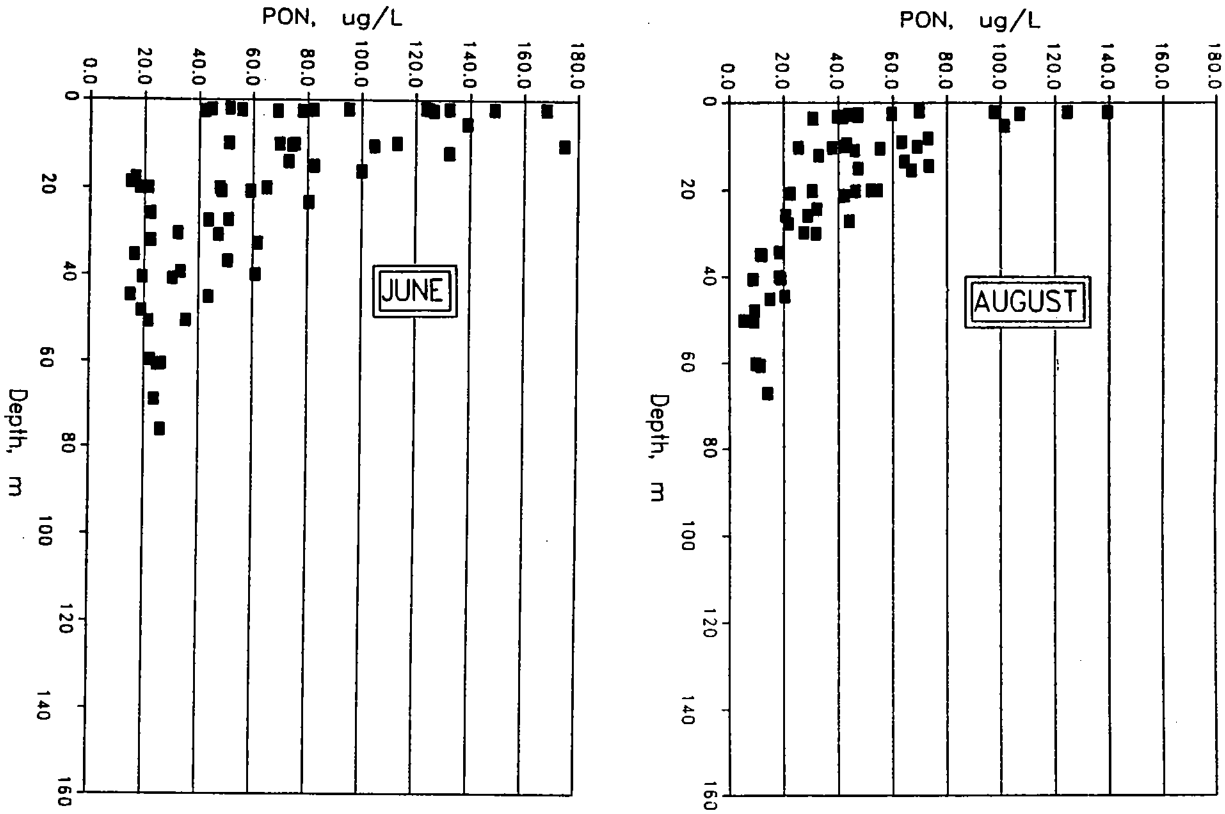


Figure 7. Nitrogen Concentrations vs Depth in Massachusetts Bay. (continued)
[From data of Townsend *et al.*, 1990.]
(b) PON (μg/L) in June and August 1990.

small net long-term storage in sediments (e.g., Nixon, 1987). Therefore, most of this organic matter must be consumed in bottom waters or sediments, but not buried. By these calculations, nearly 15% of the oxidation and remineralization of autochthonous organic-matter production would appear to take place in subpycnocline layers. The rest, or 85%, would therefore take place in the top 25 m of water, which is about the average depth of the Bay (Appendix A). If there are substantial other inputs on the Baywide scale, the implied removal would rise. After these very rough calculations, it is appropriate to discuss next what is known of the role of bottom-sediment communities in nutrient and carbon dynamics.

2.1.4 Present Role of Benthic Communities in Nutrient and Carbon Dynamics

Only limited information is available on benthic fluxes of oxygen and nutrients. Giblin *et al.* (1991) measured fluxes from soft-bottom, mostly muddy sediments of outer Boston Harbor, the Broad Sound area, and Massachusetts Bay just inshore from the proposed outfall site during September/October 1990. Bottom temperatures were near seasonal maxima, especially offshore, and fluxes can often be highest under these conditions (e.g., Nixon *et al.*, 1980).

Oxygen uptake and DIN release rates, although variable between cores, were generally highest in the Harbor and lowest at the deeper offshore stations. This pattern could result from the enrichment gradient, but also must be partially created by the general increase in depth and decrease in temperature to the offshore (cf. Hargrave, 1973; Kelly *et al.*, 1985).

An initial sense of the significance of the measured flux rates can be gained through simple calculations. Using the average DIN flux given by Giblin *et al.* for Stations 7 and 8 near the proposed outfall site of about 900 μmol/m^2/day, the combined N mass of an overlying 32 m of water in October [on the order of 256 mmol (= ≈ 8 μM N times 32,000 L for a 32-m water column); see Figure 6] could be replaced by the benthic flux in about three-quarters of a year. The replacement, in contrast, by export from the nearshore and Harbor area is more on the order of days (Section 2.1.2). In short, the nutrient flow into the water of the area far exceeds the flux from bottom sediments.

Annual benthic fluxes, not yet available, are required to place the role of the benthos in proper perspective. Based on comparison of different areas (Hargrave, 1973; Nixon, 1981; Kelly, 1990), it is known that annual benthic fluxes relate to the production and import of organic carbon, scaled by the depth of the mixed layer (or total depth in many coastal areas). For example, for an average 32-m mixed water column or a 15-m surface mixed layer (numbers appropriate for the outfall site), one could expect something on the order of 45 to 80 g C/m^2/year respired on the bottom for a net primary production level of 300 to 350 g C/m^2/year, or about 13% to 27% (Figure 8). There are some coastal sites, particularly nondepositional areas, where benthic fluxes fall below the general relation in Figure 8, and the associated values extrapolated from production probably should be viewed as maxima.

DIN fluxes are less predictable, but normally constitute a slightly smaller fraction of net production as organic nitrogen. This situation is evidenced by the fact that the O/N ratio for benthic fluxes often is high relative to a Redfield ratio (Nixon, 1981; Kelly, 1990). Giblin *et al.* (1991) report high O/N flux ratios in October near the proposed outfall site that are typical of coastal marine bottom communities. High O/N ratios can in part reflect a denitrification removal from the DIN flux owing to conversion to gaseous N_2.

We do not know the temperature/flux relationship for Massachusetts Bay sediments. For other areas, such as Narragansett Bay, it is exponential (Nixon *et al.*, 1976). Assuming that an exponential relation applies here, the values from Giblin *et al.* (1991) for October could be extrapolated to an annual value (Appendix A). The extrapolated value for sediments near the outfall site comes to about 15 to 18 g C/m^2/year, or only about 4% to 6% of the assumed primary production, and about 95 to 205 mmol N/m^2/year, or only about 2% to 5% of the inorganic nitrogen needed by the primary producers. The extrapolated range for benthic respiration is lower than expected from a regression line describing coastal ecosystems, but is still within the range measured for other areas (Figure 8).

There are obvious limitations to the above extrapolation, and it is only a very crude and tenuous guide for illustration. The extrapolation would be higher if the flux response to temperature is less sharp; on the other hand, it must overestimate fluxes for the whole outfall site region because the measurements are biased to soft-bottom depositional sediments in an area that has a considerable fraction of gravel and hard-bottom area. Hard-bottom areas may themselves contribute to benthic fluxes, but no estimates of this are available. There apparently is a seasonal "dusting" of fresh organic matter, presumably also being oxidized and creating a benthic flux, at these hard-bottom areas during the stratified period (Battelle and SAIC, 1991).

These crude benthic flux extrapolation are approximately in line with the very rough calculations on the role of bottom waters and sediments below about 25 m, which were made in Section 2.1.3. The salient point is that even maximal extrapolations of benthic fluxes do not suggest a quantitatively large involvement of the present benthos in consumption of pelagic organic matter and in recycling nutrients to the primary producers at the depth of the proposed outfall site.

With respect to possible nutrient-enrichment effects and benthic/pelagic coupling, there are two comments. The empirical regression (Figure 8) predicts only a slightly lower percentage of primary production to be consumed on the bottom even if production were to double; i.e., the coupling between benthic and pelagic processes is fairly consistent at mid to high levels of primary production, unless anoxia occurs (Kelly *et al.*, 1985; Oviatt *et al.*, 1986). In relation to this, probably the most important process with potential to alter the usual benthic/pelagic coupling relationship is the input of allochthonous organic matter in outfall effluent (cf. Oviatt *et al.*, 1987), an issue discussed in Section 3.0.

2.1.5 Some Uncertainties in Massachusetts Bay Nutrient Dynamics

It is postulated that surface and deepwater flows of water into Massachusetts Bay may occur from the northeast (Townsend *et al.*, 1990; and others). Whether this brings additional nutrients that are substantially involved in the dynamics of the Bay has yet to be determined and is difficult to calculate. Conceivably such inputs are significant, but the importance may depend on definition of the boundaries. For example, where a pycnocline shoals inshore at bottom sediments and tidal currents mix nutrients brought in from outside the Bay to surface waters, or where deep waters from offsite are actually upwelled, an input could have significance. In contrast, if vertical mixing occurs without intrusion of nutrients from outside a system boundary, nutrients are simply

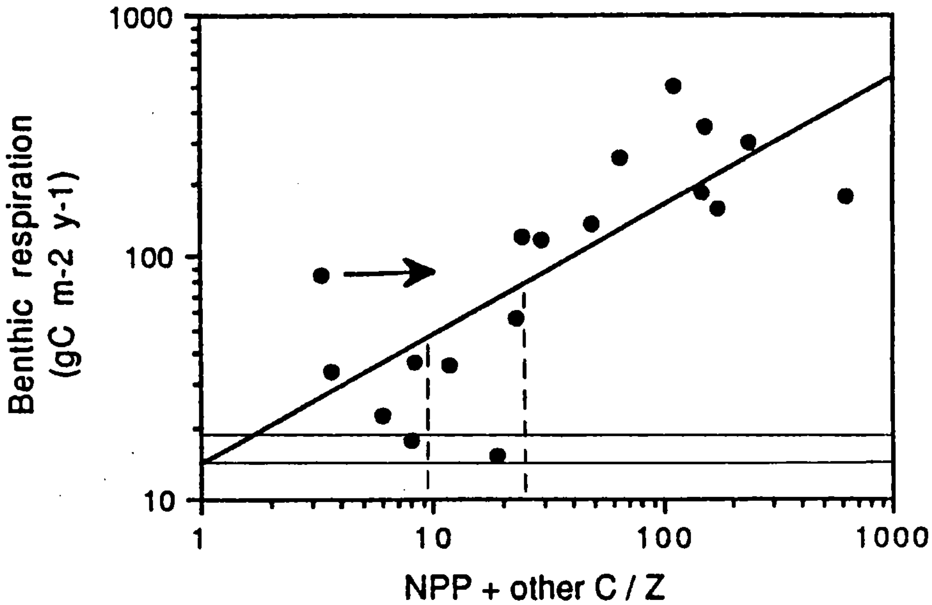


Figure 8. Relationship between Annual Carbon Input and Benthic Respiration in 18 Shallow Coastal Marine Ecosystems.
[Modified from Kelly, 1990.]

The sum of net primary production and allochthonous carbon input is scaled by the depth of the water, since most systems are well mixed. The data point with an arrow shows the shift that occurs if one instead uses the mixed-layer depth of one seasonally stratified system (following Hargrave, 1973). Dotted lines from the x axis that intercept the functional regression line indicate a range of benthic fluxes predicted assuming 300-350 g C/m^2/year and with a 32-m water depth or a 15-m mixed-layer depth. An extrapolated range of annual benthic respiration (solid lines from y axis) is described in text.

being recycled, and do not represent an input. This discussion introduces the question of whether the Bay is a net source or sink with respect to Gulf of Maine and oceanic waters outside its confines, which is unanswered. In part, this question can be resolved only with a dedicated effort to assess transport at certain critical boundaries, such as across the transect line from Cape Ann to Stellwagen Bank.

Having a complete nutrient budget for Massachusetts Bay may be important, but critical to outfall concerns is another question. One could argue that, on the broad scale, nutrients now going into Boston Harbor largely get dispersed into the Bay, the fundamental budgetary elements of nutrient input to the Bay will remain fairly unchanged with movement of the outfall offshore. Accordingly, the major related uncertainties may be ones focused not on broad Baywide scales, but more localized events. Unfortunately, over any time scale, small spatial-scale effects such as those created by heterogeneous, concentrated patches of nutrients and organic matter removed from, but traceable to, the outfall are among the most difficult to predict with reasonable certainty. The outfall monitoring program must be designed to address such events, to guard against errors of attribution.

2.2 NUTRIENTS AND PHYTOPLANKTON

2.2.1 Chlorophyll Gradient

The annual average concentration of chlorophyll (measured through *in situ* fluorometry, Townsend *et al.*, 1990) throughout Massachusetts Bay during 1989-1990 in part suggests the influence of nutrients in the inshore area. The inshore nitrogen-enriched area has slightly elevated levels of chlorophyll, an indicator of phytoplankton biomass (Figure 9). Highest inshore mean levels were seen at Stations 7 and 18, but inshore of about the 50-m depth contour, mean values are above 2.5 mg/m^3 (= μg/L).

The region of Massachusetts Bay far offshore has mean values primarily less than 2.5 mg/m^3 and many below 2.0 mg/m^3. The offshore exceptions are Stations 13 and 15, along the southern transect of Townsend *et al.* (1990). The vertically integrated chlorophyll in the surface 25 m or so at these stations, which bracket either side of Stellwagen Bank, is almost as high as found at many nearshore stations. The vertical distribution of chlorophyll is qualitatively different as compared to more inshore areas that have comparable mean values. For example, a maximum chlorophyll layer is usually found near the base of the pycnocline rather than higher in the water (Townsend *et al.*, 1990). Although chlorophyll at depth can sometimes simply indicate settling, its frequent occurrence suggests another hypothesis: that phytoplankton grow at this depth under lower-light conditions because they are successful at intercepting a nutrient flux across the pycnocline. At these offshore stations, the water is very clear, light penetrates deeply, the principal nutrient input to the surface waters may be vertical. Townsend *et al.* (1990) nicely illustrated the seasonal and vertical pattern of chlorophyll throughout the Bay. They give a conceptual model of the interaction of light and nutrients along a distance gradient inshore to offshore, expanding on the nutrient/light/depth ideas of Loder and Smayda (MWRA, 1990).

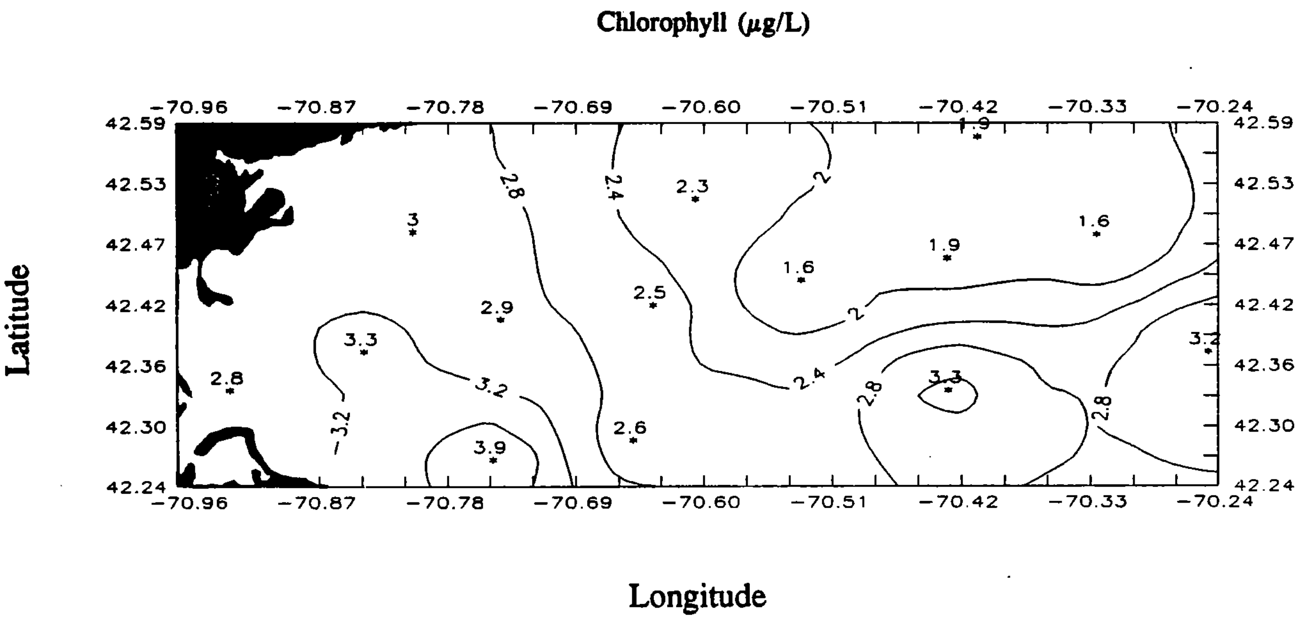


Figure 9. Annual Surface Water Integrated Averages of Chlorophyll at the Stations Sampled by Townsend *et al.* (1990) in Massachusetts Bay.

The annual mean values of chlorophyll of Figure 9 showed much of the spatial pattern described by a synoptic, remote-sensing survey by Michelson (1991) (cf. her Figure 12, p. 30). For example, her yearly annual mean values for 1978-1979 images were mostly in the range of 2.5 to 5.0 mg/m^3 within and just offshore of the Harbor. The nearshore "patch" of elevated chlorophyll above 2.5 mg/m^3 appears to be roughly the same size, shape, and location as suggested by data summarized by Figure 9 and similarly bends slightly to the South outside the Harbor and off Cohasset (Station 18). Conspicuously absent in Michelson's averaged images are the high chlorophyll values found on either side of Stellwagen Bank by Townsend *et al.* (1990); this finding is not surprising since the Coastal Zone Color Scanner samples surface waters only. Remotely sensed color data, if available, would be of great value with respect to monitoring surface waters of Massachusetts Bay, but cannot fully replace field surveys that sample throughout depth.

2.2.2 Relation between Annual Surface-Water Nutrients and Chlorophyll

Comparing annual mean nitrogen and chlorophyll concentrations (Figure 10) yields an illustration from which several observations may be made. (1) Stations 6 and 7 are not markedly different from the Harbor with respect to DIN, but Station 6 appears to have slightly depressed chlorophyll. (2) An apparent sharp rise in chlorophyll with increasing DIN from about 4 to 6 μM does not continue to the highest DIN concentrations in and immediately outside the Harbor. The apparent rise in chlorophyll associated with a small increase in DIN is slightly less dramatic if only shallower (<50 m) stations are considered. (3) Station 18 has moderate DIN levels but the highest average chlorophyll level. (4) Deepwater Stations 13 and 15, on either side of Stellwagen Bank, support high chlorophyll levels for their standing concentrations of DIN.

One interpretation of the pattern of the first three observations is that turbidity inshore, especially that spilling out from the Harbor (Townsend *et al.*, 1990) may limit light penetration and, thus, inshore plankton biomass production. In contrast, nutrient limitation must be a factor offshore. Lower chlorophyll concentrations at the deeper stations could occur because phytoplankton are lower in the water column, in essence trading off higher irradiance levels to be closer to the main source of nutrients (from below). Note that the effect of using an integrated photic zone mass, rather than concentration units, would be to shift offshore points upwards and to the right, each by a factor of about 2 to 3 relative to most inshore stations. By a mass measure, the chlorophyll response would be seen as less sharp than suggested by Figure 10. The response issue is further examined via comparisons to other areas (below).

There are additional factors that could help to create the observed pattern. A second interpretation of the response pattern is that inshore grazing activity by pelagic or benthic organisms in these shallower waters acts to crop chlorophyll. A third interpretation is that some other water-quality feature (e.g., toxicity) inhibits higher chlorophyll biomass at higher DIN concentrations. It is difficult to assess whether the second two mechanisms, although known to occur in other coastal areas, help to produce the pattern here.

There were two deepwater situations with high chlorophyll for their measured standing stock of nutrients. As suggested in Section 2.2.1, it is possible that nutrient flux from depth is intercepted at the base of the euphotic zone; because such DIN may be rapidly assimilated to PON, it might never appear as DIN within the surface layers during the productive season. If this is an accurate

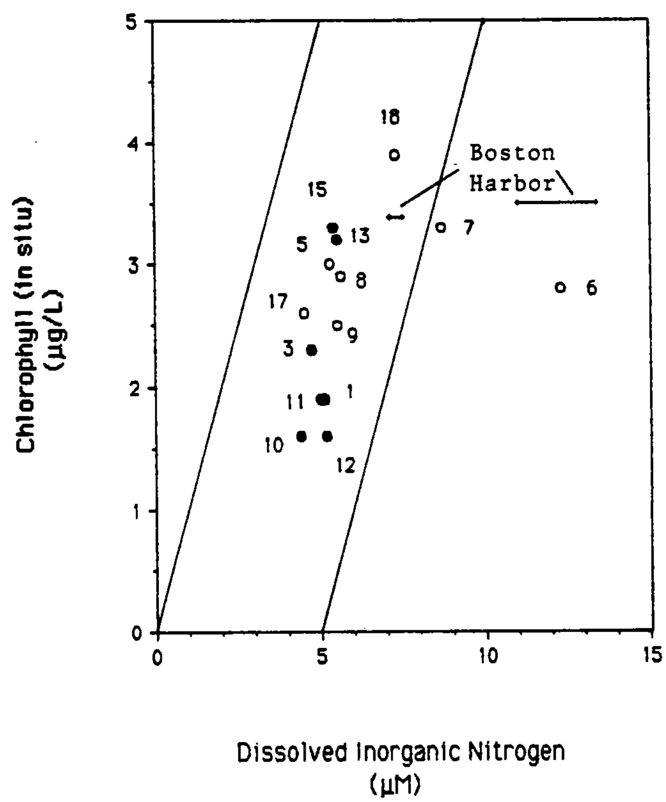


Figure 10. Annual Average Chlorophyll vs DIN for Stations in Massachusetts Bay and Boston Harbor.

Massachusetts Bay data were derived from Townsend *et al.* (1990) for surface water, as given in Appendix B. Open circles show those stations that are inshore of the 50-m depth contour. Boston Harbor data are from Robinson *et al.* (1990); the range is for two areas of the Harbor for DIN and two stations in those areas for chlorophyll. Isopleths depict slope of 1:1 relationship between chlorophyll increase (in micrograms per liter) per DIN increase (in micromoles per liter).

interpretation, these surface waters experience a rather high nutrient load that is not reflected in nutrient concentrations but is revealed as higher chlorophyll.

There are two speculations related to moving the principal point source of nutrient discharge to the offshore.

1. If the nutrients now flowing into waters where chlorophyll is light-limited were instead to be introduced as higher concentrations into clearer offshore surface waters, chlorophyll levels might reach levels higher than recorded presently at the highest nutrient levels. For example, projecting the initial curve linearly would suggest about a doubling of chlorophyll presently at Station 8 if the average DIN concentration doubled to Harbor values of about 11 μM (i.e., the rise follows roughly a 1:1 isopleth; Figure 10).
2. Additionally, surface chlorophyll levels might rise without a corresponding noticeable increase in surface DIN concentrations if subpycnocline waters enriched with nitrogen from a submarine outfall discharge create a flux of nitrogen (diffusive or advective) to euphotic waters. These two possibilities can be monitored and assessed by simultaneous profiles of nitrogen and chlorophyll throughout the year.

A perspective on chlorophyll/nutrient gradient and relationship in Massachusetts Bay can be gained by comparison to other areas (Figure 11). The range of mean values for both DIN and chlorophyll summarized for Massachusetts Bay is very small as compared to other New England coastal areas, specifically Buzzards Bay and Narragansett Bay. Interestingly, the apparent "yield" of chlorophyll at a given DIN concentration may vary across the Bays. However, each shows a chlorophyll response to enhanced nutrients, and each also has an apparent chlorophyll depression nearest a major sewage effluent source. Differences across the Bays could relate to a number of factors, including the involvement of benthic nutrient fluxes that may not appear in annual DIN averages, the intensity of water-column mixing, light limitation, effects of grazing organisms, etc. One might argue that comparing only surface waters (i.e., $\approx$ 10 to 20 m) throughout Massachusetts Bay (with the exception of the whole water column in the Harbor and Station 6) to whole water columns in other more shallow bays is not entirely appropriate. The main point of the comparison though is to suggest that Massachusetts Bay, from the standpoint of surface nutrients and chlorophyll concentrations, is not highly eutrophic.

The perspective on trophic status can be enlarged further (Figure 12) to include a coastal bay with much lower nutrient levels, as well as a Marine Ecosystems Research Laboratory (MERL) experimental enrichment gradient with much higher nutrient levels (Nixon *et al.*, 1986). The range depicted covers the range of nutrients observed in coastal waters in nature. Massachusetts Bay data do not indicate a highly enriched condition. The Massachusetts Bay/Boston Harbor data set appears most comparable to the MERL control tanks, which simulate a 5-m-deep water column in unenriched lower Narragansett Bay (cf. Nixon *et al.*, 1986).

Quite possibly, shifting the major point source of nutrients from inshore to offshore will not materially change the N input to Massachusetts Bay, as discussed above. If so, enhancement of phytoplankton biomass/nutrient levels on a broad Baywide scale is not expected.

Even so, some localized enrichments are possible, and the peak concentrations of chlorophyll, PON, and DIN could be shifted to the offshore as compared to the present. Using the general trend for DIN and chlorophyll (Figure 12), an increase of no more than about 1 mg/m^3 of

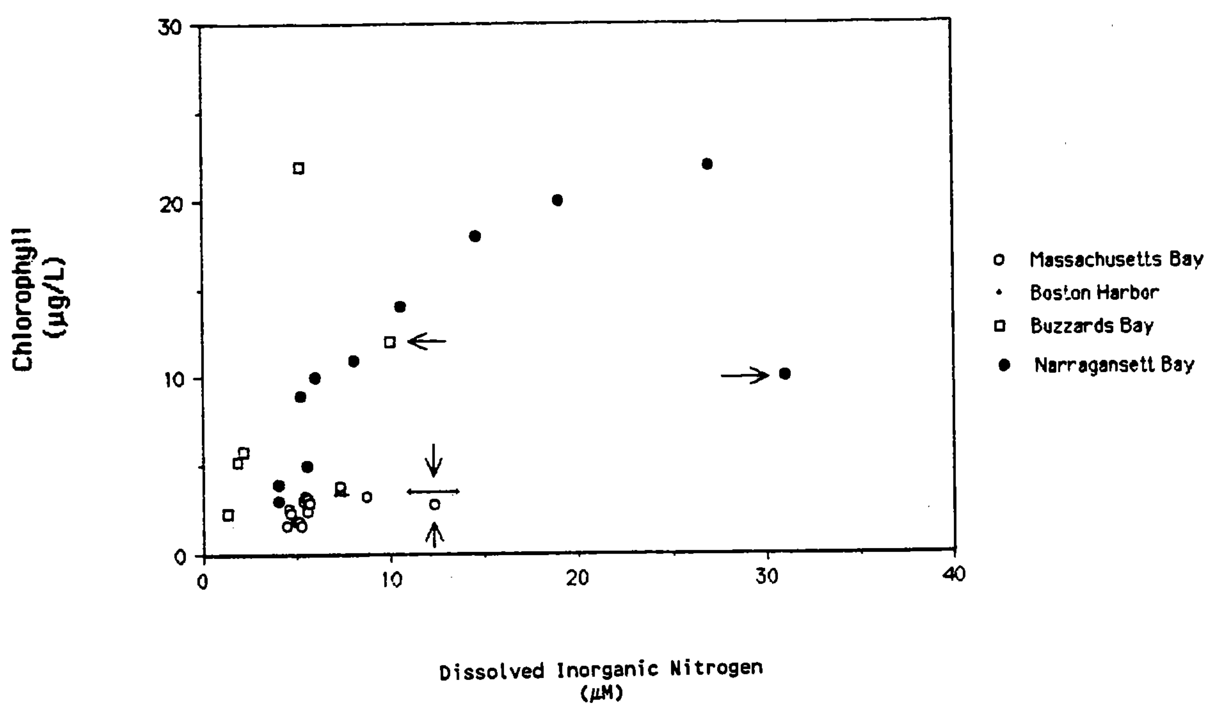


Figure 11. Annual Average Chlorophyll/DIN Relationship for Some New England Bays

The arrows indicate stations in each Bay near a major metropolitan sewage outfall. Data sources for Massachusetts Bay/Boston Harbor are given in Figure 10. The Narragansett Bay data are time-weighted annual means (n = 26) for the whole water column (7 to 28 m) [from Oviatt *et al.*, 1984]. The Buzzards Bay data are time-weighted annual means (n = 17) derived from Turner *et al.* (1989), for the whole water column (5 to 15 m) [from Kelly *et al.*, 1991].

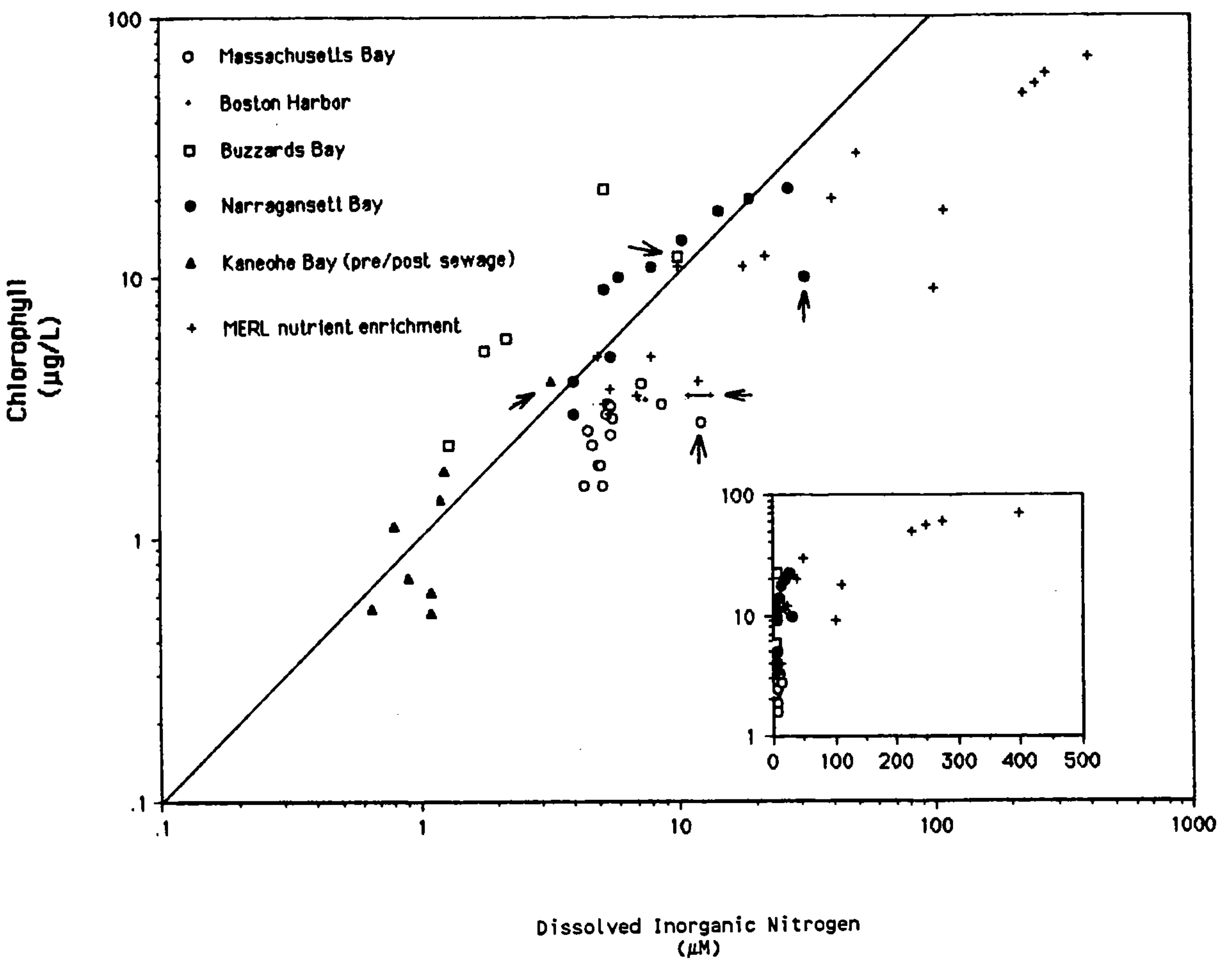


Figure 12. Annual Average Chlorophyll/DIN Relationship for the Approximate Range of Nutrient Levels Observed in Coastal Marine Waters.
Kaneohe Bay is from Smith *et al.* (1981), but is also summarized in Nixon *et al.* (1986), which gives values for the 2.3-year MERL mesocosm nutrient-enrichment study. Arrows as in Figure 11. Insert shows relationship plotted on a semilog scale. Isopleth on main plot depicts 1:1 slope between chlorophyll (in micrograms per liter) and DIN (in micromoles per liter).

chlorophyll appears realized for each 1-μM DIN increase. This rough relationship also applies to annual data from several tributaries of Chesapeake Bay (J. Garber, personal communication).Thus, even with the removal of a hypothesized inshore constraint of light on phytoplankton biomass, doubling of average DIN values of any stations offshore of Station 6 would not result in particularly high chlorophyll levels. Physical scenarios engendering different possibilities for local enrichments are discussed further later, and the monitoring plan being developed is to address these specifically.

Annual average chlorophyll concentrations offer an indication of the overall trophic status, but variability is also important. The brief occurrence of very high chlorophyll biomass is important to consider, for rapid decay of organic matter from an intense bloom under the right conditions is an agent to promote hypoxia or anoxia. Results of recent mesocosm experiments and comparative ecology of coastal ecosystems (Nixon and Pilson, 1983) seem to tell us that, as nitrogen loads increase, the range between maximum and minimum chlorophyll values may widen, and also that rapid, large-magnitude oscillations in chlorophyll biomass may accompany very high nutrient loading. In general, the observed maximum chlorophyll concentrations through the course of a year for many shallow coastal waters may be about 2 to 10 times the annual mean (Nixon, 1983).

Massachusetts Bay seems no exception. From Townsend *et al.* (1990), the surface-water integrated values for a given sampling date ranged as high as about 8.5 (Stations 7 and 8 in June) to 9.2 mg/m^3 (Station 18 in August). Some individual bottlecast measurements at a given depth were higher than this; some from either *in situ* fluorometry or extracted samples reached 13 to 14 mg/m^3 (Townsend *et al.*, 1990; their Figure 29, p. 50). From a historical summary, Cura (1991) estimates euphotic-zone chlorophyll in Massachusetts Bay to range seasonally from <1 to about 7.5 mg/m^3 and in Boston Harbor from <1 to about 6.8 mg/m^3, ranges similar to those calculated for each sampling date from the stations of Townsend *et al.* (1990). The annual maximum for coastal bays and estuaries typically can be from about 5 to well above 30 mg/m^3 (Boynton *et al.*, 1982; Nixon and Pilson, 1983). The most current as well as the historical mean and maximum chlorophyll values (by a variety of methods and at a variety of stations) for Massachusetts Bay are in a mid to low range, and do not in general indicate a very eutrophic situation.

2.3 NUTRIENTS, PRIMARY PRODUCERS, AND THE FOOD WEB IN MASSACHUSETTS BAY

Several estimates of primary production have been made for Massachusetts Bay, and these are summarized by Cura (1991) and Michelson (1991). For the region, estimates range from as low as 250 g C/m^2/year for Massachusetts Bay (possibly underestimating subsurface production) and 325 g C/m^2/year for Boston Harbor (both from Michelson, 1991), to 350 g C/m^2/year for Massachusetts Bay [from Cura (1991) and based primarily on studies by Parker (1980) and the MWRA (1988, 1990) in a limited area within the vicinity of the outfall region]. New estimates for stations throughout the Bay are forthcoming from the Townsend *et al.* (1990) data set. If one plots available values compared to nitrogen loading estimated for the area (Figure 13), they fall in line with those of other coastal areas. Importantly, the rates are not at a level where dramatic responses to further nutrient enrichment is expected (Kelly and Levin, 1986; Oviatt *et al.*, 1986; Nixon *et al.*, 1986); only a maximum of a doubling or so of present rates characteristically would be expected even with an order of magnitude increase in loading.

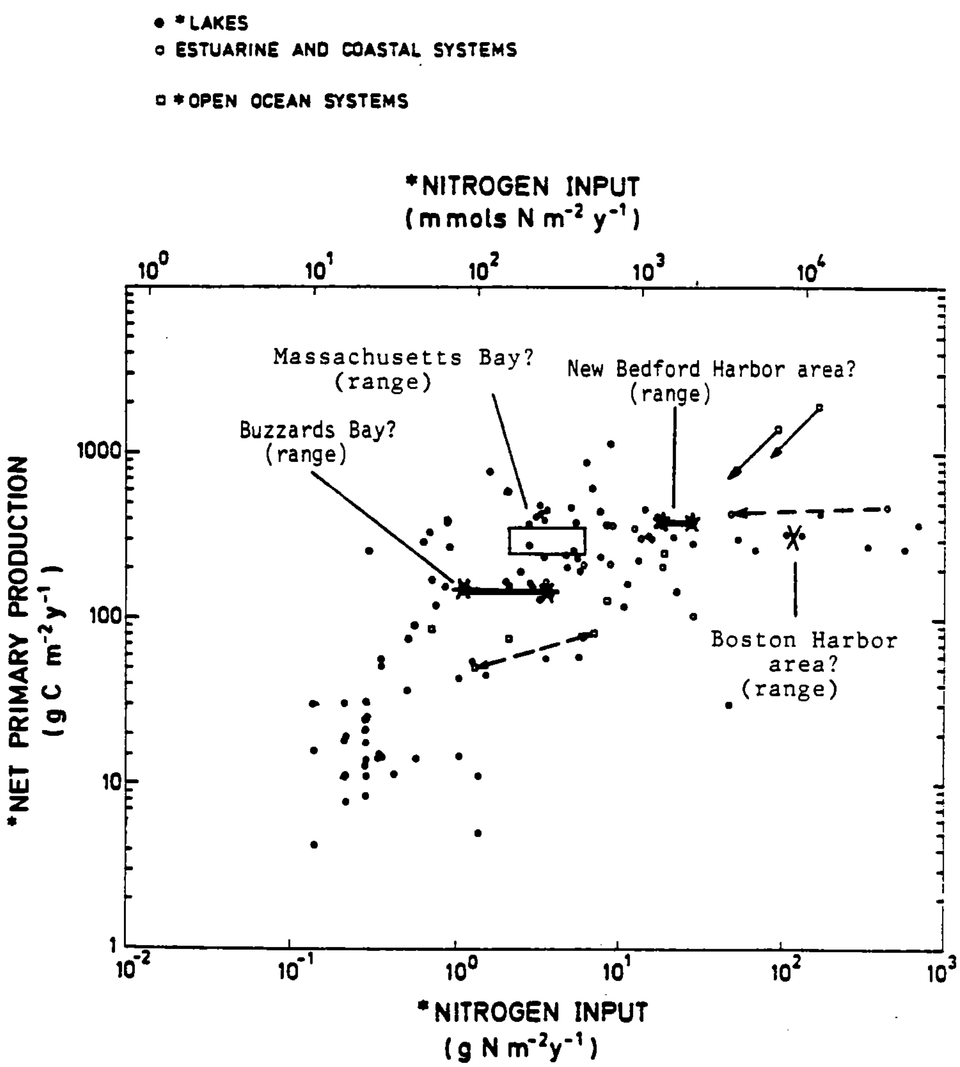


Figure 13. Annual Aquatic Net Primary Production vs Nutrient Loading [Modified from Kelly and Levin, 1986.]

Based on N loading for estuarine and marine systems and P loading to lakes, the *x* axis is scaled by a Redfield N/P ratio. Data sources for Buzzards Bay/New Bedford Harbor are in Kelly *et al.* (1991). Data for Massachusetts Bay/Boston Harbor are calculated from the literature. Loading ranges are suggested in Appendix A. For example, NOAA (1988) gives Massachusetts Bay N loading, calculated as 550 mmol/m²/year; without input from Boston Harbor, the Bay loading could be about 110-135 mmol/m²/year. Cura (1991) gives a range for NPP of about 250 to 350 g C/m²/ year in Massachusetts Bay. Menzie-Cura (1991) gives loading to Boston Harbor, calculated as 8643 mmol/m²/year, and Michelson (1991) suggests that NPP in the Boston Harbor area is about 325 g C/m²/year.

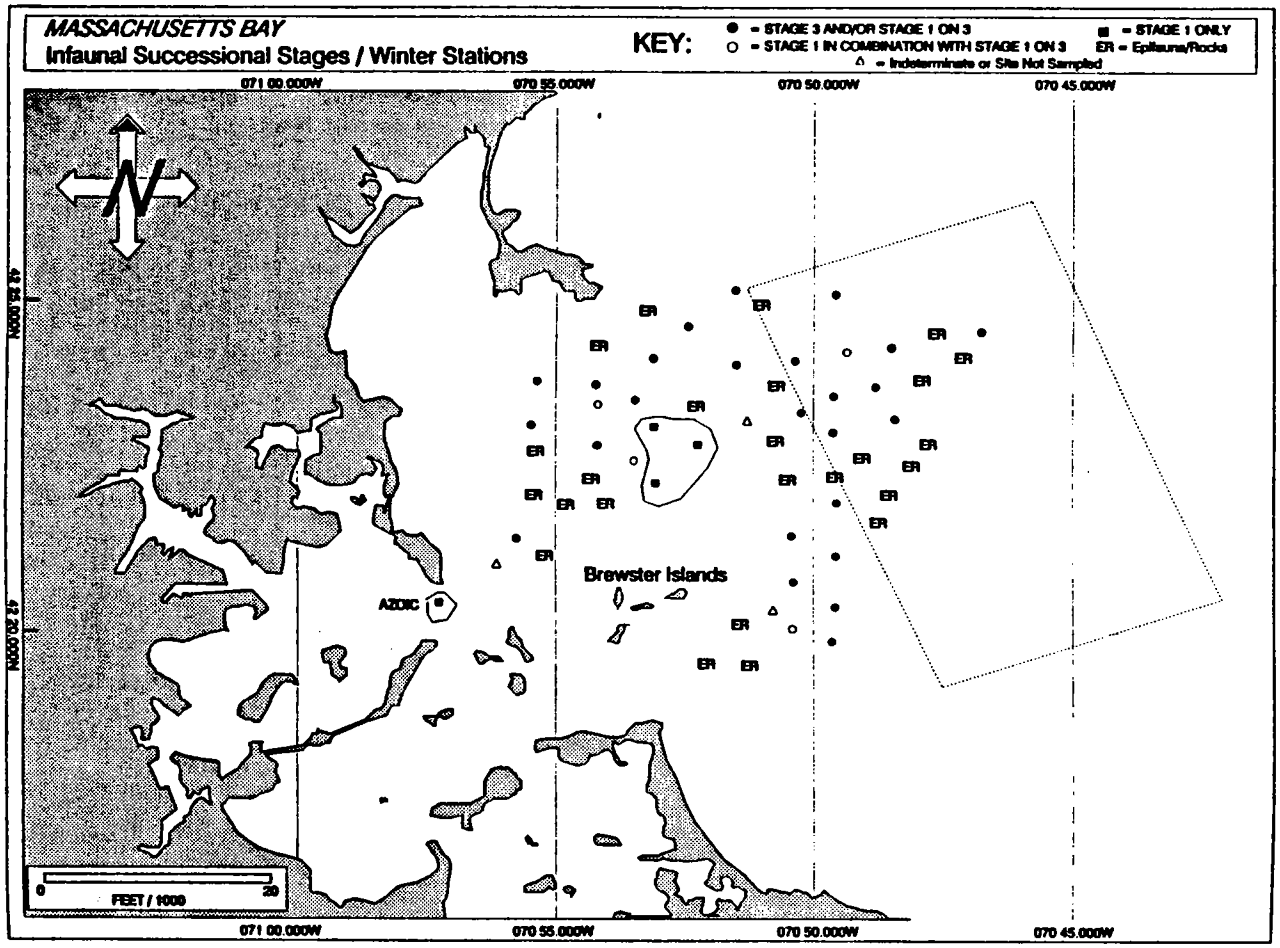
At the loading rates estimated for the Bay, external N loading can supply only a fraction of the nitrogen needs of the pelagic primary producers, on the order of 10%. The implication is that nutrient recycling is crucial for sustaining production, a conclusion also reached for all but the most heavily enriched coastal waters (Kelly and Levin, 1986). The present evidence and calculations suggest that a great deal of this recycling may occur within the surface mixed layer. With respect to pelagic activity, presently, there is limited information on pelagic microbial activity or zooplankton. The few station data on relative zooplankton biomass from Townsend *et al.* (1990) do not suggest much of an inshore enrichment gradient response (Appendix B).

Besides the quantity of algal biomass produced, the quality (e.g., the species composition) is significant to potential effects of nutrient enrichment. One factor that is suspected of influencing species composition includes the ratio of nitrogen to silicate nutrients available to phytoplankton. Diatoms in general have high silicate requirements, whereas flagellates (which can be among the nuisance or noxious species categories — cf. Cura, 1991) do not. Recent experimental mesocosm evidence suggests that manipulating the N/Si ratio, although having little overall metabolic or chlorophyll biomass effect at a given N loading, can indeed affect the species composition of the plankton and perhaps alter the flow of energy partitioned between the pelagic and benthic/demersal consumers (including fish) in near-coastal waters (Doering *et al.*, 1989).

Since sewage is relatively low in silicate as compared to marine waters, effects on species composition are conceivable (Officer and Ryther, 1980; Ryther and Officer, 1981). On the other hand, since much of the nitrogen presently put in with effluent to the Harbor may well be ejected into the Bay, it is unclear if the present status quo in Massachusetts Bay would be substantially altered with direct dissolved-nutrient input offshore.

Both gradual long-term changes and short-term population explosions of individual problem species, among the most difficult events to grapple with in a predictive sense, are nonetheless reasonable issues of concern and need consideration in the monitoring program. Projections relative to undesirable phytoplankton species changes could not be addressed within the timeframe for this synthesis, but could be cast in the light of phytoplankton species compositional changes across stations and through an annual cycle (data of Townsend *et al.*, 1990). As true for some other measures of eutrophication, the physical setting is a primary element that must be included in interpreting species fluctuations. Turbulent mixing and the light field strongly influence the dominant forms within the phytoplankton community (e.g., Margalef, 1978; Jones and Gowen, 1990).

Regarding other components of the food web, it is well established that pelagic activity drives benthic processes, and benthic measures are often used as a primary biological monitoring tool intended to be integrative of fluctuations in the overlying water. Data on the macrobenthic community throughout Massachusetts Bay have been summarized by Rhoads and Blake (Battelle and SAIC, 1991). Dominant species of soft-bottom areas are well known and the general quality of the benthic environment has been assessed by sediment camera imaging. Notably, there are several small clusters of benthic stations within the Broad Sound area and slightly farther offshore that presently show evidence of organic enrichment (Figure 14). These sites are within the area defined in earlier sections by elevated DIN, PON, and chlorophyll levels extending outside Boston Harbor. Recognition of certain particle depositional areas as possible foci for organic matter, particularly along the 25-m isobath, provides a framework for connecting pelagic

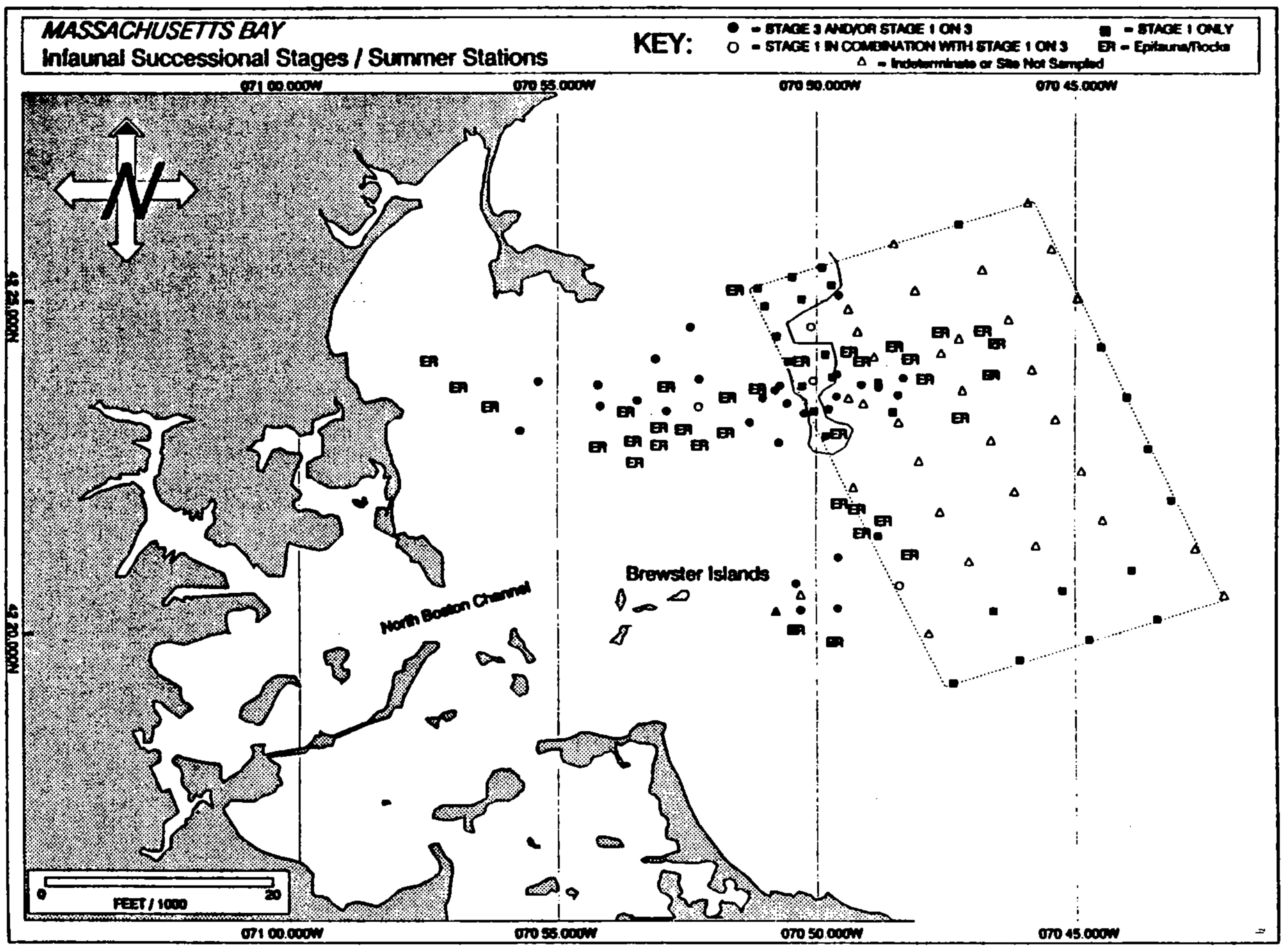


(a) Winter Surveys.

Figure 14. Infaunal Successional Stages in Massachusetts Bay As Determined from Recent Surveys.

[From Shea *et al.*, 1991.]

Stage I assemblages are surface-dwelling forms usually indicative of organic enrichment or recent physical disturbance. Stations where only Stage I assemblages were detected by camera imaging are delimited by a line in the Broad Sound area.



(b) Summer Surveys.

Figure 14. Infaunal Successional Stages in Massachusetts Bay As Determined from Recent Surveys. (continued)
[From Shea *et al.*, 1991.]

activity (and allochthonous input) with benthic responses. The present sedimentary biological andgeological evidence emphasizes the heterogeneous nature of benthic/pelagic coupling in the region in the Bay just inshore of the outfall site, but again suggests that an influence of present input to the Harbor may be felt well outside its confines.

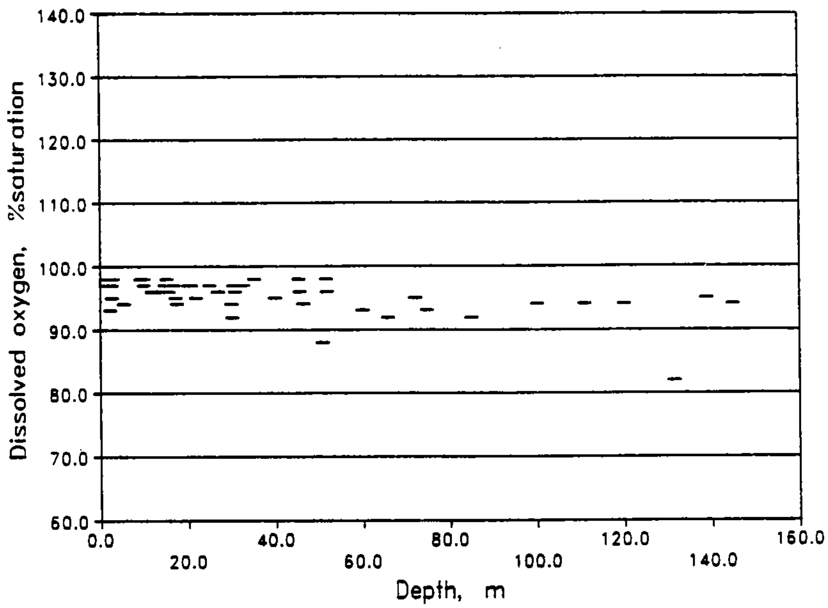
As a final note, the connection of fisheries to eutrophication processes is discouragingly difficult to quantify (e.g., Nixon *et al.*, 1986). In part, this must be true because fisheries can be highly influenced from the "top down" (by human predation) as well as from the "bottom up" through nutrient stimulation of food resources. A general relationship between primary production and fisheries yield in various marine and coastal systems has been established (e.g., Nixon, 1988) that describes how increases in fish yield might accompany increases in primary production; the relationship is not satisfyingly predictive. Considering that fish are some steps removed from nutrients in a food-web sense, the expected signal relative to noise is small. Therefore, while continued monitoring via trawls by State fisheries biologists will provide some background and is critical to toxicant burden monitoring, it is not foreseen that trawl data can be of extremely high utility relative to predicting or assessing outfall enrichment consequences.

2.4 NUTRIENTS AND DISSOLVED OXYGEN DYNAMICS IN MASSACHUSETTS BAY

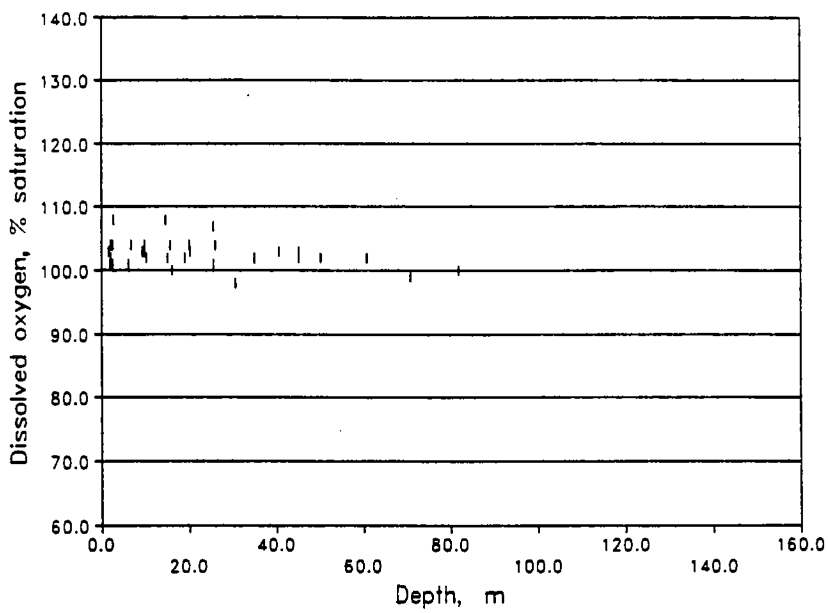
From the Townsend *et al.* survey, we can begin to develop a synoptic description of variability in dissolved oxygen throughout Massachusetts Bay. Figure 15 (series) displays the vertical distribution of dissolved oxygen (DO), expressed as percent of saturation to remove effects of salinity and temperature, including all stations and progressing from winter to autumn. The physical conditions of the water column are highly relevant to oxygen distributions and thus are discussed briefly also in this Section. Stations with samples below about 50 m are from Stations 1, 11, and 12 (in the northeast corner of the Bay on the seaward side of Stellwagen Bank), Station 3 (on the shoulder of the northern entrance to Stellwagen Basin), and Stations 9 and 15 (in Stellwagen Basin) (Figure 3).

Coincident with well-mixed conditions and similar nutrient concentrations (see Figure 6) throughout the Bay in February, the percent saturation of oxygen varied little across stations or with depth and was slightly below 100% saturation. The deepest stations were not occupied in March, but surface waters showed initiation of the spring bloom, with values between 100% to 110% saturation and little difference over the depths measured.

By April, the water column was beginning to stabilize (Townsend *et al.*, 1990). There was a thermocline at about 10 to 20 m, shoaling shoreward to a thermal front between Stations 7 and 8 (the outfall site) that was also evident in March. The top of the pycnocline was relatively close to the surface (about 5 to 10 m), both shallower and sharper offshore, except over Stellwagen Bank (Station 14), where density contours bent to the surface, perhaps suggesting upwelling. For the most part, a spring bloom was still evident at the surface and DO to 30 m throughout the Bay was at or above saturation, with the deeper waters only slightly below saturation. Judging from changes in total nitrogen values (e.g., Figure 6), a substantial fraction of surface nitrogen may already have been removed to deeper waters by this time. Colder temperatures in deeper waters at this time would inhibit decomposition, oxygen consumption, and nutrient remineralization.

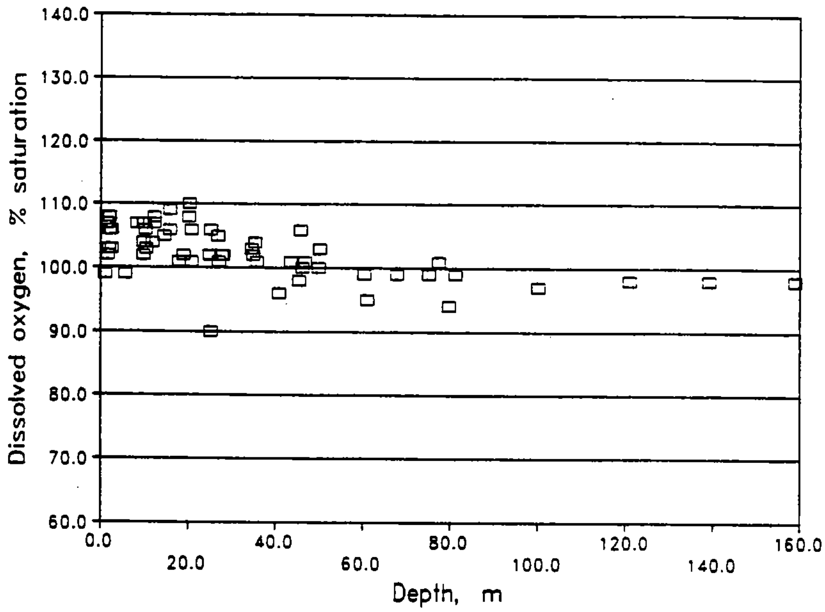


(a) February 1990

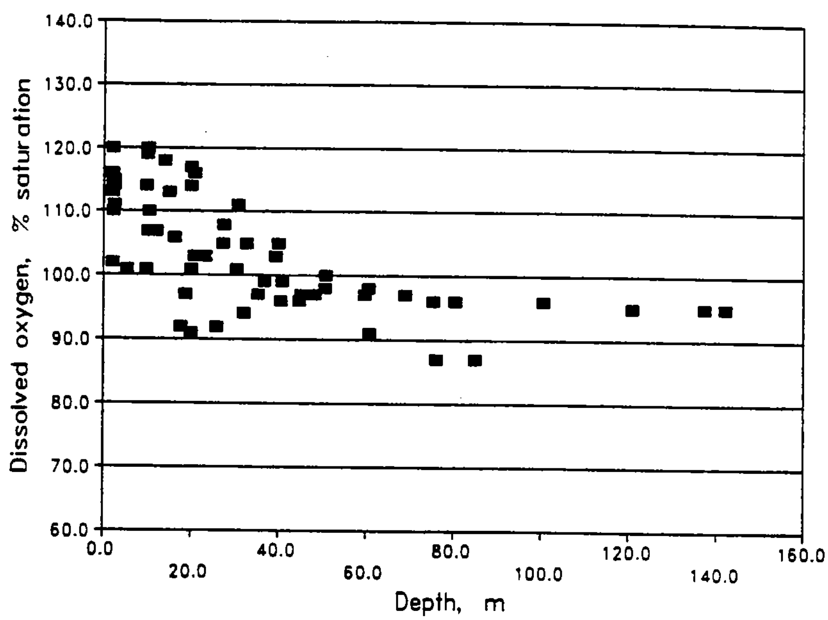


(b) March 1990

Figure 15. Dissolved Oxygen vs Depth throughout Massachusetts Bay at Different Months.
[Data from Townsend *et al.*, 1990.]

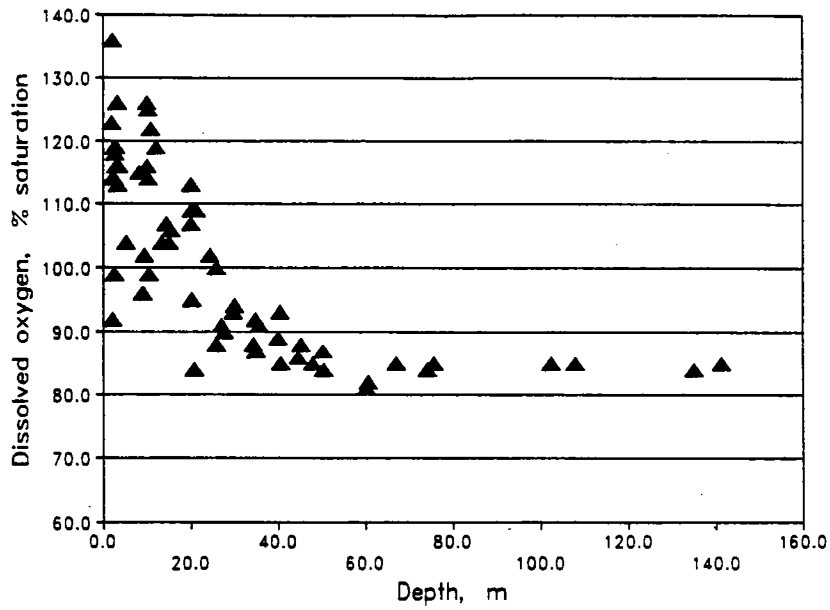


(c) April 1990



(d) June 1990

Figure 15. Dissolved Oxygen vs Depth throughout Massachusetts Bay at Different Months. (continued)
[Data from Townsend *et al.*, 1990.]



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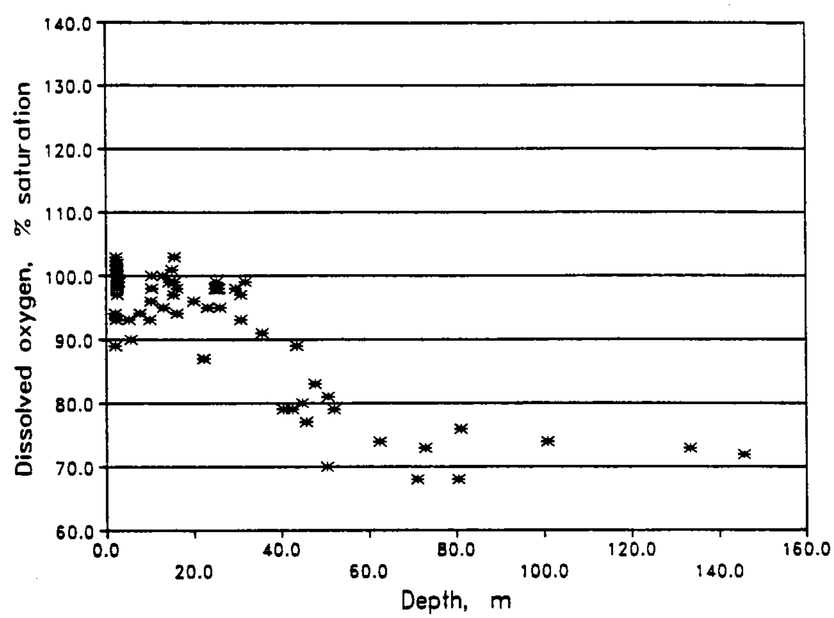


(f) October 1989

Figure 15. Dissolved Oxygen vs Depth throughout Massachusetts Bay at Different Months. (continued) [Data from Townsend *et al.*, 1990.]

By June, there was strong vertical thermal and salinity stratification, with a sharp pycnocline from about 5 to 15 m throughout most of the Bay. Some deep chlorophyll was apparent below this level at the offshore stations. At this time, there was more variability in oxygen saturation throughout surface waters: values ranged from about 90% to 120% saturation over the top 40 m, and many points to 30 m were above 110% (higher than in April). There was very high chlorophyll at about 10 m across Stations 7-9 (samples indicated $\approx$ 9 to 10 mg/m^3) and DO values there were among the highest, but other stations with chlorophyll maximum of about 2 to 3 mg/m^3 had equally high percent saturation. Below 40 m, values were uniformly less than saturation and in general slightly lower than in April, but still mostly above 90%.

Stratification was maximal by August (Townsend *et al.*, 1990). The pycnocline graded slightly from about 5 m inshore to perhaps 10 to 15 m offshore. Notably, at this time waters from about 20 to 30 m were fairly uniformly between 8 and 10 °C, and waters to 60 m in Stellwagen Basin and to about 100 m in the northeast corner outside the Bank were about 6 °C (Townsend *et al.*, 1990). The oxygen profiles show the most extreme variability of the year in the surface waters; lowest surface values were at Stations 6 and 7 coming out of Boston Harbor, and very high values above 110% were recorded at every other station. A strong exponential decrease in percent saturation was evident with depth, and most values below 25 m fell below 90% saturation, suggesting continued depletion from June.

Throughout both June and August, the many high surface DO values suggest continued net primary production at many points in the Bay; most likely, the high supersaturations are in evidence because of fairly stagnant surface conditions, with poor mixing and gaseous exchange with either the atmosphere or subpycnocline waters. In such conditions, when dissolved nutrient concentrations are very low, rapid nutrient recycling within the surface waters usually is a major mechanism to sustain production. Both horizontal, advected nutrient flux from enriched inshore waters and diffusive nutrient flux across the pycnocline may provide additional sources of nutrients at this season.

Surface mixing was initiated by October (actually the previous year, 1989), and surface DO values no longer were highly supersaturated. The pycnocline was deeper, starting at about 30 m or more through much of the Bay. Only waters below about 50 m were still trapped below the pycnocline, a depth below Stellwagen Bank and about equal to the northern shoulder entrance to Stellwagen Basin (Station 3) and at a depth that essentially caps off the northern end of Stellwagen Basin. The Basin and other deepwater temperatures were about 8 to 10 °C. Because the deep waters were still not ventilated from the surface and yet had continued to warm, it is not surprising that the lowest absolute DO concentrations of the year were recorded (4.63 and 4.65 mL/L at the bottom samples at Stations 9 and 15 in Stellwagen Basin — both about 68% saturation).

Viewed as a three-dimensional contour against the bathymetry across the stations sampled in Massachusetts Bay, the deepwater depletion of DO by early fall is very striking indeed (Figure 16). By this season, waters of the depth of the proposed outfall site (32 m) all were ventilated. However, below about 40 to 50 m, all DO values were further depressed from August. The annual progression of lowered DO in deep basin water may continue until October of some years, or until surface cooling and winds enable mixing throughout depth.

Grouping all station profiles together, such as in Figure 15, or contouring, as in Figure 16, emphasizes the similarity throughout the Bay of communication between surface and deep

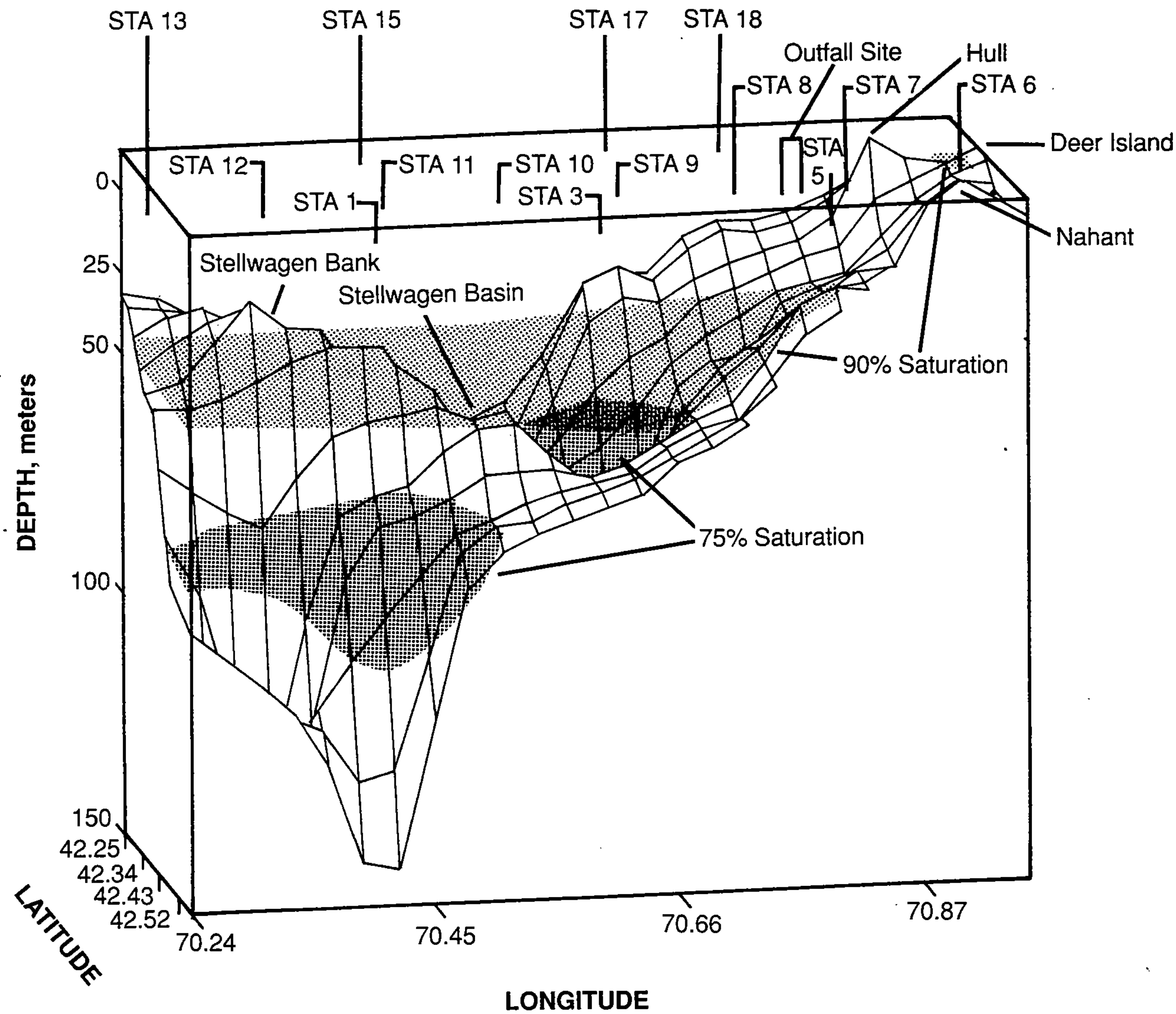


Figure 16. Three-Dimensional Graphic of Massachusetts Bay Bathymetry with Overlay of the Approximate Depths of 90% and 75% Oxygen-Saturation Isopleths.
[Based on October 1989 data of Townsend *et al.*, 1990.]
The view is from the northeast corner of the Bay, looking west/southwest into the entrance to Boston Harbor.

waters with respect to production, decomposition, and mineral recycling. There can be some horizontal differences in water masses and their chemistry and biology (e.g., Figures 3 and 8) as well as important horizontal exchanges of water into or within the Bay. The question of horizontal exchange bringing nutrients into the Bay (Townsend *et al.*, 1990) could involve the communication of nutrient-rich bottom waters, and is briefly addressed next.

Some deep waters (below 50 m) were constantly below pycnocline through October. Data from three groups of stations offer a horizontal contrast: Stations 1, 15, and 12 beyond Stellwagen Bank; Station 3 just north of Stellwagen Basin (a broad plateau entrance, about 50 to 60 m deep, for deeper water from the north to come into deeper waters of Stellwagen Basin to the south); and Stations 9 and 15 within Stellwagen Basin itself. DO decreased with increasing length of stratification in each of these groups, and each group showed the trend of concomitant increase in DIN as well as evidence of nitrification (conversion of ammonium to nitrate, which uses oxygen) (Figure 17). Yet the groups remained somewhat distinct at each sampling period. For example, samples from Stellwagen Basin characteristically had the lowest DO.

Station 3 bore a chemical signature similar to outer Bank stations of similar depth (50 to 60 m) only in June, but strongly deviated from surrounding deeper waters to the northeast and to the south as summer progressed. Based on their distinct chemistry (Figure 17), it is not clear that deeper waters of the three groups communicate with each other during this period. Subsequently, for bottom waters of Stellwagen Basin, a mostly vertical emphasis to the dynamics of nutrients and oxygen may be appropriate. Perhaps the vertical connection includes communication with shallower inshore areas. However, whether some of the organic matter originally produced in surface waters draws on nutrients brought into the Bay from outside Stellwagen Bank is still an open question.

In summary, the limited vertical and horizontal observations on DO in the Bay's water highlight a number of notions relevant to eutrophication and outfall monitoring.

1. Oxygen is a critical and sensitive measure of both production and consumption processes and should be a major focus for monitoring. High supersaturations suggest continuing sustained net production in summer, as bottom water DO begins to decline.

2. The pace of bottom-water oxidation processes appears, not surprisingly, influenced by temperature increases in deeper waters during summer, as well as by a continuing supply of organic matter from overlying surface waters. Deep-water remineralization of dissolved nutrients, also not surprisingly, occurs concomitantly with oxygen decrease.

3. Lowest DO concentrations (4.63 mL/L) were recorded in bottom waters of stations in Stellwagen Basin in October 1989. These approach a state standard of 6 mg/L ($\approx$4.2 mL/L) used as a "site determinative measure" in the outfall siting (EPA, 1988, p. 7-4). Stellwagen Basin deep waters may be somewhat isolated from more offshore oceanic deep waters because of bathymetry. The Basin may function as at least a temporary "internal" sink for organic matter processed in the Bay. However, the extent to which Stellwagen Basin bottom-water chemistry reflects communication with inshore or Cape Cod waters (versus just with its overlying surface waters) via an imported transport of organic matter is un-known. Such communication, of some significance to the influence of an outfall on bottom

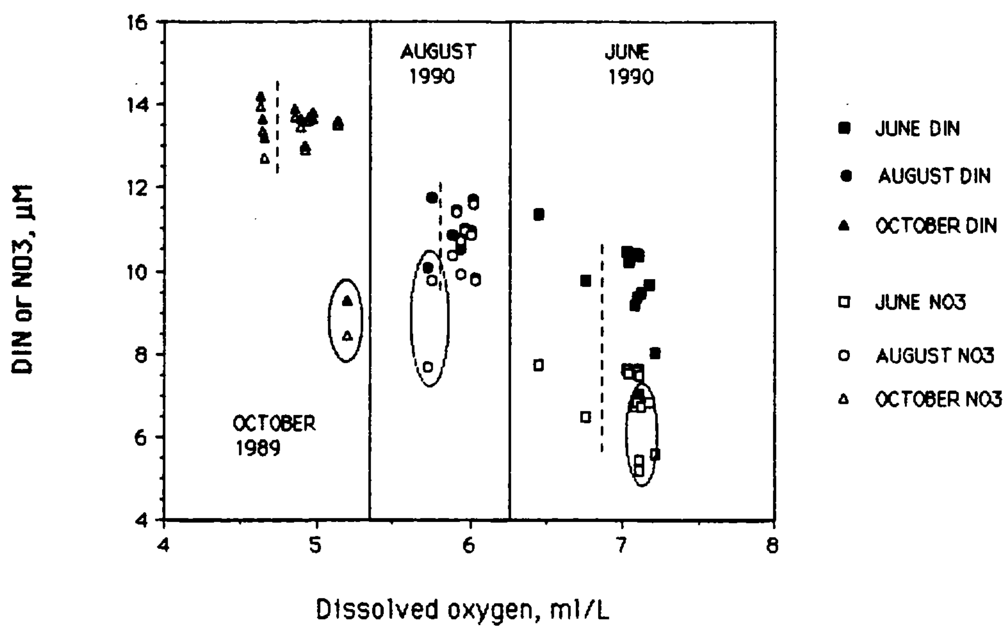


Figure 17. Dissolved Oxygen and Nitrogen Concentrations in Waters Greater than 50 m Deep.
[Data from Townsend *et al.*, 1990.]

Three groups of stations are identified: Station 3 is contained within the oval in each panel; Stations 1, 11, and 12 outside Stellwagen Bank are to the right of the dashed line; Stations 9 and 15 in Stellwagen Basin are to the left of the dashed line. Points are from discrete depths sampled at each station; the numbers taken at depths exceeding 50 m vary by station.

waters and sediments, could be clarified by transect studies extending from the outfall site downslope into Stellwagen Basin and then along its axis.

4. The influence of vertical stratification on water chemistry, and thus biological activity, is acute. Vertical chemoclines for oxygen and nutrients parallel quite accurately the seasonal fluctuations in the depth of the surface mixed layer. Indeed, vertical and horizontal components to the water-column monitoring are advised to facilitate a three-dimensional, volumetric picture of the proposed outfall effluent discharge fate and effects.

5. Judging from chemical as well as physical oceanographic data, the location of the proposed outfall (Figure 3) appears to be below pycnocline from about April to almost October. This depth may vary year to year and there may also be brief mixing events during the late summer period. Near-bottom water DO near the proposed outfall, from limited data examined here, may decrease to about 91% to 88% saturation by August (Stations 7 and 8, respectively). Station 7 (about 28 to 30 m deep) became ventilated and was at 95% saturation because of breakup of the pycnocline in that area sometime before October, thereby avoiding even lower DO depression experienced in deeper basins. Station 8, in slightly deeper water (about 47 to 50 m deep) did show continued DO decrease to 79% saturation, measured at 42.7 m in October. The difference between these two stations rather clearly defines the level of pycnocline barrier between surface and deep-water mixing at this date and location and moreover suggests how slightly prolonged time under stratified conditions during this critical period may affect water quality. Lowest bottom-water DO values for Stations 7 and 8 were recorded in October as 6.0 and 5.17 mL/L ($\approx$ 8.6 and 7.4 mg/L), respectively. By comparison, the EIS (based on earlier data of the MWRA) noted that the ambient DO subpycnocline could be as low as 6.5 mg/L at this time. EIS modeling used 8.0 mg/L as the ambient initial levels for model projections since it was recognized that outfall site location was normally above pycnocline by October. The initial ambient values used for modeling were appropriate given these new data.

A rough calculation (Appendix A) of how much carbon would be necessary to depress the DO in the lower 50 to 80 m of the northern end of Stellwagen Basin by about 0.5 mg/L suggests it unlikely, but not implausible, that this could be realized by a surface nutrient stimulation and subsequent sedimentation of organic matter. In the light of such calculations, it is intriguing to consider if the chemistry of the deeper waters in Stellwagen Basin is a good large-scale indicator of surface-water activities throughout Cape Cod and Massachusetts Bays; but it is not necessarily true that any bottom-water changes in Stellwagen Basin will reflect new discharges at the proposed offshore site. If rapid allochthonous POM transport toward these deeper waters occurred, a DO decrease on the order of 0.5 mg/L is possible (Appendix A). This, rather than broad-scale nutrient enrichment, is perhaps the more relevant mechanism for concern, and can be monitored.

Fairly efficient sedimentation of the discharged particulate C around the outfall, not rapid transport, has been predicted by models (EPA, 1988). Even so, processes such as particle focusing and near-bottom transport have not been sufficiently included in those predictions; nor have long-term transport and sinks for material been assessed. Therefore, over the long term, a lower layer transport that would affect the bottom waters of Stellwagen Basin seems a fair and addressable question for the monitoring program.

2.5 A CONCLUDING PERSPECTIVE ON MASSACHUSETTS BAY DYNAMICS RELATED TO EUTROPHICATION

The present effluent discharge strongly influences the water quality of the Harbor, but the confines of the Harbor are not the ultimate resting place of many of the effluent's constituents, particularly mobile forms like nitrogen, which cycle rapidly between dissolved and particulate forms. Because of vigorous, regular flushing by tides and by freshwater inflow, the concentrations of nutrients in the Harbor are fairly low, considering the level of loading. Much of the present nutrient input appears to be advected out of the Harbor into Massachusetts Bay. There is compelling evidence that Harbor nitrogen is dispersed well into Massachusetts Bay, and that there may be associated responses by pelagic and benthic communities. The benthic data support the notion of a high degree of patchiness, perhaps driven by particle focusing processes that concentrate matter to small-scale depositional sediments.

The significance of these conclusions is that a major change anticipated for nutrients is a simple shift in the major point of introduction, but not so much the magnitude of nutrients dispersed to Massachusetts Bay. Because of greater water depth at the proposed outfall site and dispersion through diffusers over its length (2 km), an increased initial volume dilution is fully expected, and the dissolved concentration gradient from the new point source can be projected as less sharp than the presently observed gradient away from the Harbor. The realized gradient will, of course, be determined by physical processes mixing and advecting waters vertically and horizontally from the site. A principal question of impact is connected with increased or concentrated transport to vulnerable bottom areas.

The waters of Massachusetts Bay stratify seasonally, with removal of nutrients and organic matter to bottom waters via *in situ* processes. Bottom waters and sediments have a role in the recycling of nutrients, particularly via seasonal mixing and renewal of nutrients, but the intensity of the benthic/pelagic fluxes is limited as compared to shallower coastal areas. Much of the present primary production throughout the Bay seems sustained by recycling, especially within the surface layers themselves. Nevertheless, there presently appears to be transport of organic matter produced in surface layers to deeper waters that is sufficient to lower oxygen concentrations substantially during a prolonged period of summer stratification. Thus, the addition of more organic matter and Biological Oxygen Demand (BOD) through subpycnocline discharges seems qualitatively a principal point worthy of more examination and a focus for monitoring.

3.0 EUTROPHICATION ISSUES FOR MASSACHUSETTS BAY

Perhaps the most significant difference in conditions relative to promoting eutrophication effects at the present effluent discharge site versus at the proposed outfall site will not relate to the magnitude of nutrients released but rather to physical stratification. In contrast with the Harbor, the offshore site (based on profiles of Townsend *et al.*, 1990; as well as others: MWRA 1988, 1990) stratifies for a portion of the year (Figure 18); the water column mixes sometime in late summer/early fall. This seasonal mixing in part may alleviate the potential stress of low DO as waters there reach seasonal maxima. With an outfall sited in deeper waters, although nutrients might be trapped below a critical depth and have less phytoplankton stimulation effects (e.g., Loder, Smayda in MWRA 1990), the concern would be greater for local DO depression because of more prolonged stratification.

Both vertical stratification and horizontal fronts between the inshore and offshore water masses occur surrounding the offshore site region, and these complicate the process of prediction relative to short- and long-term transport, and thus some potential consequences of enrichment and eutrophication (Figure 19). Prior to development and calibration of a three-dimensional hydrodynamical model for the outfall site, qualitative schematics suffice to illustrate concerns relative to designing outfall monitoring.

From the perspective of inshore/offshore dynamics, presently the summer stratification appears to aid surface transport of nutrients from inshore to waters over the proposed site, while such dispersion may be occasionally impeded by fronts. With movement of the source offshore, dispersion to the inshore could be occasionally impeded by some of those frontal conditions. At other times, subpycnocline nutrients and organic matter could conceivably advect to the inshore (to the west or laterally) and get mixed in shallows where turbulence destabilizes the water column all the way to the bottom sediments. In either case, the inshore surface source of nutrients and particles from the Harbor will have decreased dramatically.

The present level of nutrient flux to the surface waters around the outfall site (post discharge) could be exceeded during winter mixing, but primarily will be inhibited by the pycnocline for a good part of the year unless the discharge plume unexpectedly rises above this level. In a stratified condition, nutrient flux to the surface may well be less than at present, occur primarily across the pycnocline through diffusive and other processes, and potentially promote higher chlorophyll concentrations below the surface (like those present in stations bracketing Stellwagen Bank).

Three main issues of eutrophication are illustrated in Figure 20: DO, algal species compositional change, and overall food web change also involving the benthos and fisheries. A quantitative assessment relative to such concerns is far beyond the scope of this report and would be particularly difficult to accomplish, given the physical and ecological uncertainties involved. The best present strategy, especially considering the projections above with respect to nutrient enrichments, is to focus on adequate monitoring to address these concerns, even as better quantitative predictions are attempted. In that spirit, the schematics are to give some perspective on the critical points for monitoring.

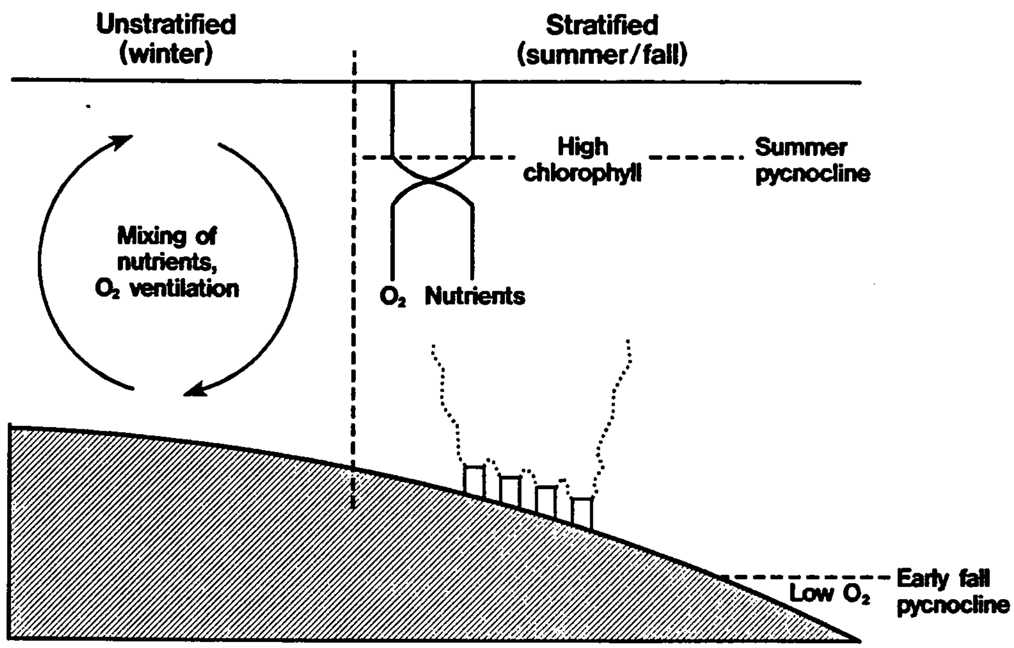


Figure 18. Schematic of Seasonal Changes in the Physical Regime Surrounding the Outfall.

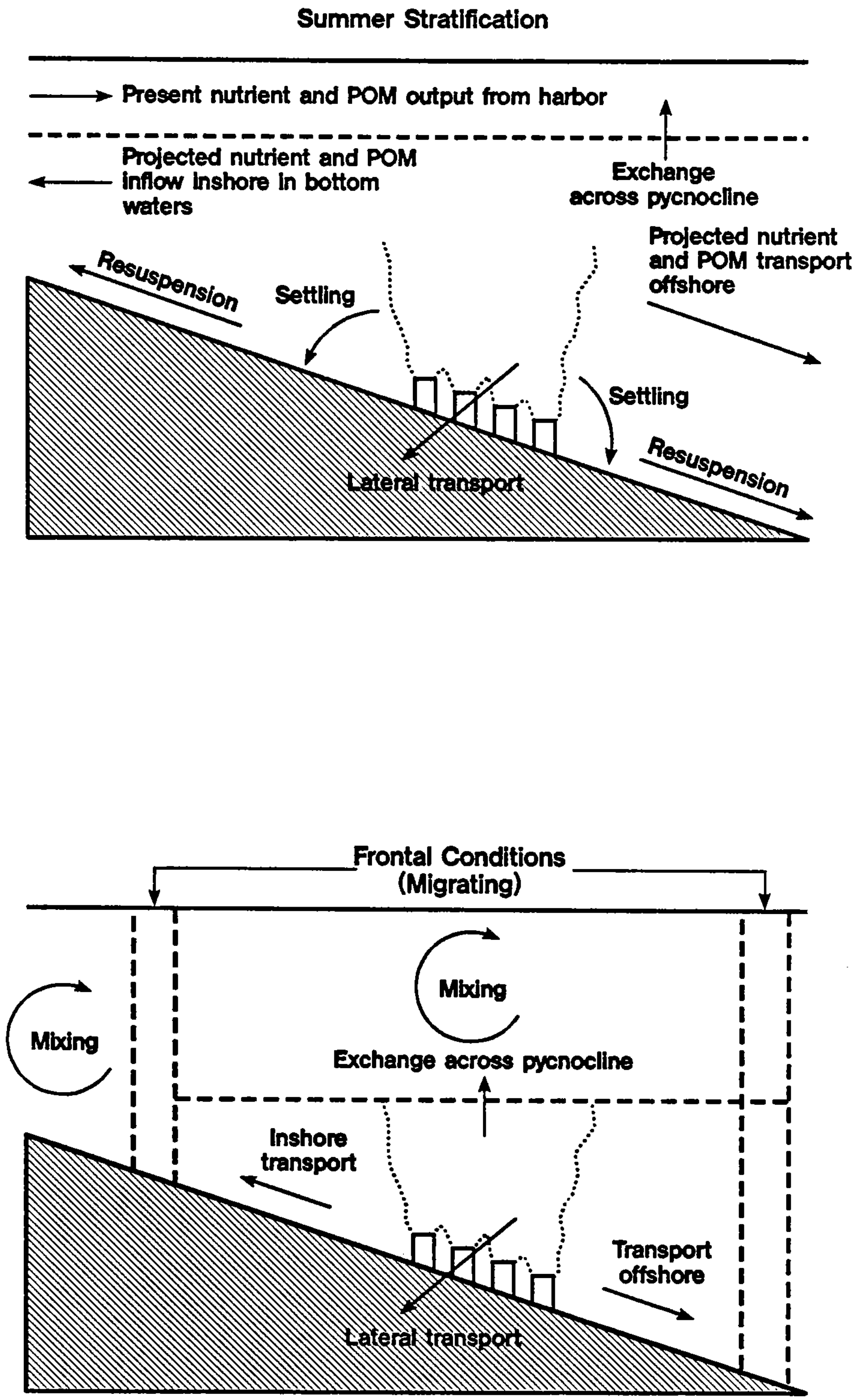
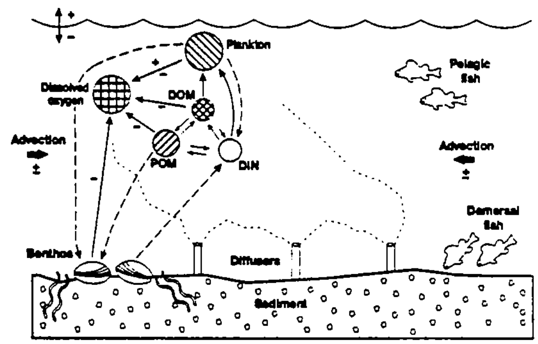


Figure 19. Schematic of the Influence of Vertical and Horizontal Mixing and Stratification on Mixing and Transport from the Outfall Site.

(a) Dissolved Oxygen Depletion

Contributing Factors

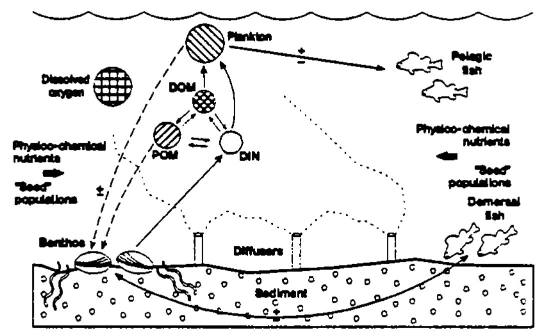
- Light
- Suspended matter
- Temperature
- Salinity
- Physics (mixing, stratification, advection, air-sea exchange)
- Nutrients
- Organic matter



(b) Stimulation of Problem Phytoplankton

Contributing Factors

- Nutrient quality and "quality" (N/P/Si)
- Light
- Suspended matter
- Temperature
- Salinity
- Physics (mixing, stratification, advection)



(c) Changes in the Food Web

Contributing Factors

- Light
- Suspended matter
- Temperature
- Salinity
- Physics (mixing, stratification, advection)
- Nutrient "quality"
- Plankton species

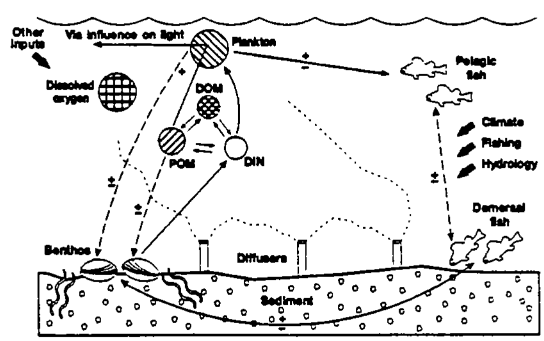


Figure 20. Schematic of Processes Contributing to Three Main Eutrophication Issues.

The influence of discharged material can be by a triad of substances: dissolved inorganic nutrients (DIN), particulate organic matter (POM), and dissolved organic matter (DOM). Usually, the focus is on effects caused by DIN and POM.

First, consider nutrient stimulation and effects on DO. During well-mixed conditions chlorophyll and production may rise, perhaps roughly as suggested by Figure 11. Some of the organic matter stimulated during the spring bloom will be transported to bottom sediments and be metabolized poststratification as the bottom waters warm (e.g., Nixon *et al.*, 1980). Given the loading rates and the limited increases in surface nutrients expected because of volumetric dilution, vigorous mixing, and advection during this period, the nutrient levels may not differ greatly from present conditions and the expected surface stimulation or prolonging of a spring bloom would not necessarily translate to major changes from slightly enhanced energy flow through the ecosystem, nor to general bottom-water DO problems.

Given the environmental setting described in the Section 2 and the effects of stratification on transport as given in Figure 19, it is arguable that the larger outfall issue is organic matter input and direct BOD loading below the pycnocline (Figure 20). Here, the issues revolve around the introduction of degradable dissolved organics and particulate organic matter (POM). POM in particular could be advected and transported to small areas of high deposition, perhaps both inshore and offshore. The introduction of allochthonous POM, more so than nutrient input, can lead to a net heterotrophic system, with generally lower DO concentrations (cf. Nixon *et al.*, 1986; Oviatt *et al.*, 1987). Via oxidation of the introduced organic matter, the balance between production and consumption (and the P/R ratio) can tip to the respiratory side even under some well-mixed conditions. The MWRA schedule for waste treatment and discharging at the outfall presently includes the cessation of sludge discharge (December 1991) and the reduction of total suspended solids and BOD to 31% and 25% of 1985 levels by October 1996, with additional improvement by December 1999 (MWRA, M. Connor, personal communication). Thus, compared to present conditions, the POM/BOD loads will be much lower at the new outfall. Nevertheless, given the suspected importance of this particular mechanism of impact on DO surrounding the outfall or transported to deeper waters/depositional areas, particle transport and near-bottom DO will be a principal emphasis for monitoring.

Possible consequences such as those of stimulation of problem or nuisance phytoplankton (Figure 20) can relate to nutrient quantities, but may often relate to nutrient quality. The influence of nutrient ratios is one issue. Both direct effects on plankton species composition and indirect effects from that upon the food web are somewhat separate but important issues not yet explicitly assessed in this examination of recent data from Massachusetts Bay. Specifically, consideration of fluxes of N, relative to silica, is warranted, since sewage effluent is very low in silica and may affect the ability of diatoms to compete successfully with flagellates. These concerns frame some additional focus for the monitoring program, as described below.

Like phytoplankton species changes, food-web changes are a eutrophication issue not specifically examined. Changes in the food web (Figure 20) can be created either through nutrient/ phytoplankton linkages or through impact of organic input and direct deposition to the benthos. At a broad scale and at the higher trophic levels, such impact can be very difficult to detect. Within the local vicinity of an outfall, they can be examined, for basic changes associated with enrichment are documented in the literature, and therefore signals that normal linkages within the food web are approaching critical points can be monitored.

4.0 A SUITE OF EUTROPHICATION/ENRICHMENT INDICATORS FOR MONITORING MASSACHUSETTS BAY

There is a fairly standard set of pelagic and benthic measures characteristically used to detect or describe eutrophication in coastal and estuarine ecosystems. From a cursory review of a very extensive literature on nutrient enrichment and sewage enrichment effects in the marine environment, mostly biased to shallower situations than Massachusetts Bay, one is lead to the following suggestion. The evidence is that simple nutrient and biomass-type measures — i.e., chlorophyll concentrations, faunal biomass (both zooplankton and benthic macroinvertebrates), benthic nutrient fluxes, and even fish yields — are all responsive to nutrient loading, if they are appropriately and carefully measured (cf. Kelly *et al.*, 1985; Nixon *et al.*, 1986; Nixon, 1988). Slight differences in relative sensitivity across these measures surely must occur, but will be difficult to forecast in specific cases. That there would be some fairly general correspondence is not surprising since these different parts of the food web are energetically related. The responses of many components should roughly parallel each other, unless the fundamental ecological structure is altered or the connections between parts are not strong due to physical constraints (see below).

Other indicators, particularly involving species change, are also responsive to nutrient and organic enrichment. The broad features of benthic responses are well known (Pearson and Rosenberg, 1978; Rhoads and Germano, 1986). Less well known and predictable is how phytoplankton species composition in nature changes as a generic response to increased nutrients.

In any given physical situation, parallel responses may not be fully realized across the broad spectrum of indicators. For example, in the case at hand, subpycnocline enrichment of POM could directly affect bottom-water DO, benthic fluxes, and benthic community structure to some extent. Yet this would not necessarily have a strong effect, nor necessarily a very direct effect, on the overlying plankton community. Note that the converse may be less true; in *most* cases plankton effects probably will influence the underlying benthos.

Another consideration is that the degree of response of any one indicator may, for a given nutrient or POM loading, vary with the environmental setting. Hypoxia or anoxia is a good example, as physics helps to determine whether low oxygen is realized. For example, the Chesapeake Bay, which receives far, far less nutrients per unit area or volume (and presumably BOD) than does Boston Harbor, but seasonally stratifies and flushes much more slowly, does indeed have a history of bottom-water hypoxia/anoxia.

The simple point being made is that, by monitoring with a suite of indicators, all known as responsive to enrichments, any scenario (e.g., Figures 19, 20) being played out surrounding the new outfall can be assessed. This is the most effective monitoring strategy. The understanding provided through this strategy facilitates more informed decisions on the effectiveness of any modifications to discharge, should they be deemed appropriate.

Much of the rationale and sampling design strategy for the suite of proposed monitoring measurements relative to eutrophication was detailed in the draft monitoring plan (Shea and Kelly, 1991) and need not be repeated here. The measures include many for which recent data have been examined for this report. They include chlorophyll, nutrients, DO, plankton species composition, several standard measures of soft-bottom community change (sediment-camera imaging indices, species-level change, and biomass of the top dominant species), measures of biological change in the hard bottom surrounding the outfall, and benthic flux monitoring.

Appropriate scales and sampling designs, both as baselines prior to discharge and after discharge commences, are being evaluated by an appointed committee for the monitoring program. Through the synthesis of information described in this report, there have been some insights gained on several aspects of the monitoring strategy, including issues that are more clearly defined as well as some technical approaches that may aid in characterizing scales of influence of the outfall discharge of Massachusetts Bay. These are highlighted in the next section.

5.0 SOME IMPLICATIONS AND RECOMMENDATIONS FOR MONITORING OF EUTROPHICATION

Two broad categories relative to eutrophication issues are here recognized. The first includes processes pertinent to expression of effects, nearfield and farfield, from the movement of the outfall. The issues associated with these may be addressed by baseline monitoring to clarify the processes. In this category are the two major issues of horizontal and vertical processes, including

1. Communication, via chemical exchange and transport of dissolved and particulate matter, between subpycnocline waters around the outfall and their neighboring bottom waters. This includes communication to shallower areas toward the Broad Sound area as well as southward, for some new information suggests drift in this direction. It encompasses the issue of particle-focusing processes and their influence on benthic communities. But, most significantly, it should also address the possibility of communication to deeper areas that remain stratified for longer periods. This latter issue might be effectively addressed by physical and chemical transect monitoring running offshore into Stellwagen Basin and along its axis.

2. Crosspycnocline exchange and transport processes, especially focusing on heterogeneity in an area about 10 × 12 km around the outfall that will provide a major frame of reference for monitoring (Shea and Kelly, 1991). During baseline characterization, fine-scale vertical profiling will help to describe the nature of the three-dimensional environment, and aid in characterizing the relationship of surface and subsurface interactions of relevance to the expression of enrichment-related effects.

The second category of implications derived from this synthesis involves a few technical concerns for the strategy of monitoring, particularly with respect to sampling frequency and scale.

A number of these are highlighted, with respect to certain measures and scales.

1. **Regarding spatial scales for water-column monitoring**
   There must be quantitative understanding of the environment around the outfall that is three-dimensional (such as in Figure 16) so that changes in the volumes and mass of a compound or element can be calculated. Changes, postdischarge, must be examined in comparison to a three-dimensional baseline characterization. The scale and overall strategy recommended in the draft monitoring plan should be slightly amended. That is, in addition to transect line (profiling in the vertical along the horizontal transect) as an inner track of about 1.5 km^2 around the diffuser and an outer track (presently about 10 × 12 km), adding an additional cruise track between will provide enough data density to permit three-dimensional contours. Presently, the outer track is linked inshore to approximately the 25-m depth contour and offshore to the 50-m depth contour. The purpose of this part of the monitoring remains to (1) characterize the pelagic effects in this field and (2) describe any directional movement of matter away from this region. If directional movement were indicated to certain "vulnerable" areas, this could be used to initiate higher-level monitoring studies at those locations. Moreover, this follows the overall monitoring strategy of establishing as much as possible the connection between fate and effects as the prime vehicle for establishing cause and effect. Notably, the

three-dimensional characterization will be used, along with data from sediments, to budget the fate of materials discharged from the outfall.

2. **Regarding sampling frequency for several measures**
   It appears that annual integrated water-column values are useful in depicting conditions. Water-column sampling, as proposed in the draft monitoring plan, should be sufficient to provide good estimates of these values. Secondly, a better description of the annual cycle of benthic fluxes predischarge, as proposed in the plan, seems advisable to confirm some calculations made here and to provide a baseline description adequate to suggest changes as a function of POM enrichment or phytoderived material. This characterization effort could include several stations in the vicinity of the outfall along a depth gradient, and would be most useful if coincident with characterization of the sediment environment, in terms of species, biomass, and camera-imaging.

3. **Regarding temporal scales**
   The sampling scheme for both water-column and benthic measures should indeed be heavily biased to summer stratification and extend to early fall for some measures.

4. **Regarding remotely sensed data for chlorophyll**
   If space platforms are available, their use should be continued to provide broad-scale synoptic information across the Bay, but this will not supplant field studies that can depict conditions throughout depth.

5. **Regarding phytoplankton species fluctuations**
   A main issue is by what indicator should this be measured? It seems clear that there is no surrogate measure for doing taxonomy, but the level to which this is taken still needs to be determined. Is the focus, at one extreme, on screening for some select number of problem species in a rather broad fashion throughout the Bay in waters that might traverse an outfall "plume?" Or is the focus, at the other extreme, on full taxonomic analysis on samples confined to a nearfield area around the outfall? In either extreme, depiction of species changes must be mapped onto the physical environment, including some measure of turbulence, light, nutrients and other physicochemical measures of the environment, if cause and effect are to be related.

6. Finally, it is again emphasized that a suite of indicators, some seemingly and potentially providing redundant information, provides a strategy to help to establish mechanisms of any observed effects, and this dictates strong coordination in the timing of the variety of measurements. The clear benefit of this strategy, given the complexity of the physical situation and the complexity of the discharge, is that it may provide some mechanistic understanding and, therefore, a highly informed basis for making decisions on any necessary modifications to discharge practices.

6.0 REFERENCES

Boynton, W.R., W.M. Kemp, and C.W. Keefe. 1982. "A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production." Pp. 69-89 in Kennedy, V.S. (Ed)., *Estuarine Comparisons*. Academic Press, New York, NY.

Cura, J.J. 1991. Review of phytoplankton data for Massachusetts Bay. Draft report to MWRA. February. 26+ pp.

Doering, P.H., C.A. Oviatt, L.L. Beatty, V.F. Banzon, R. Rice, S.P. Kelly, B.K. Sullivan, and J.B. Frithsen. 1989. "Structure and function in a model coastal ecosystem: Silicon, the benthos and eutrophication." Mar. Ecol. Prog. Ser. 52:287-299.

EPA. 1988. Boston Harbor wastewater conveyance system, supplemental environmental impact statement. Vol. 1. Environmental Protection Agency, Region I, Boston, MA.

Giblin, A.E., J. Tucker, and C. Hopkinson. 1991. Sediment oxygen demand and nitrogen flux in Massachusetts Bay. SAIC draft report to Massachusetts Water Resources Authority. December. 27 pp.

Hargrave, B.T. 1973. "Coupling carbon flow through some pelagic and benthic communities." J. Fish. Res. Board Can. 30:1317-1326.

Jones, K.J., and R.J. Gowen. 1990. "Influence of stratification and irradiance regime on summer phytoplankton composition in coastal and shelf seas of the British Isles." Estuarine Coast. Shelf Sci. 30:557-567.

Kelly, J.R. 1990. Paradigms of benthic-pelagic coupling of carbon and nutrients in coastal ecosystems as derived from annual field studies. ERC-124b, Ecosystems Research Center, Cornell University, Ithaca, NY 14853. Unpub. rep. 26+ pp.

Kelly, J.R., and S.A. Levin. 1986. "A comparison of aquatic and terrestrial nutrient cycling and production processes in natural ecosystems, with reference to ecological concepts of relevance to waste disposal issues." Pp. 165-203 in Kullenberg, G. (Ed.), *The Role of the Oceans as a Waste Disposal Option*. D. Reidel Publishing Co., Dordrecht, The Netherlands.

Kelly, J.R., V.B Berounsky, C.A. Oviatt, and S.W. Nixon. 1985. "Benthic-pelagic coupling and nutrient cycling across and experimental eutrophication gradient." Mar. Ecol. Prog. Ser. 26:207-219.

Kelly, J.R., I Valiela, and D. Hersh. 1991. Nutrients and the trophic status of Buzzards Bay, Massachusetts. Report to Buzzards Bay project. 47+ pp. (to be published)

Margalef, R. 1978. "Life forms of phytoplankton as survival alternatives in an unstable environment." Oceanol. Acta 1:493-509.

MWRA. 1990. Marine resources extended monitoring program. Vol. 2. Massachusetts Water Resources Authority, Boston, MA.

MWRA. 1988. Secondary treatment facilities plan. Vol. 5, Appendix Y: Nutrient analysis. Massachusetts Water Resources Authority, Boston, MA.

Menzie, C.A., J.J. Cura, J.S. Freshman, and B. Potocki. 1991. Boston Harbor: Estimates of loadings. MWRA Environmental Quality Dept. Tech. Rep. Ser. No. 91-4, February 1991. Massachusetts Water Resources Authority, Boston, MA. 108 pp.

Menzie-Cura & Associates. 1991. "Sources and Loadings of Pollutants to Massachusetts Bay." Draft report prepared for the Massachusetts Bay Program. 240 pp.

Michelson, A.R. 1991. Analysis of the spatial and temporal variability of primary production in Boston Harbor, Massachusetts and Cape Cod Bays. SM thesis, Boston University. 64 pp.

Nixon, S.W. 1981. "Remineralization and nutrient cycling in coastal marine ecosystems." Pp. 111-138 in Neilson, B.J., and L.E. Cronin (Eds.), *Estuaries and Nutrients.* Humana Press, Clifton, NJ.

Nixon, S.W. 1983. Estuarine ecology — A comparative and experimental analysis using 14 estuaries and the MERL microcosms. Chesapeake Bay Program, Annapolis, MD.

Nixon, S.W. 1987. "Chesapeake Bay nutrient budgets — A reassessment." Biogeochemistry 4:77-90.

Nixon, S.W. 1988. "Physical energy inputs and the comparative ecology of lake and marine ecosystems." Limnol. Oceanogr. 33:1005-1025.

Nixon, S.W., and M.E.Q. Pilson. 1983. "Nitrogen in estuarine and coastal marine ecosystems." Pp. 565-648 in Carpenter, E.J., and D.C. Capone (Eds.), *Nitrogen in the Marine Environment.* Academic Press, New York, NY.

Nixon, S.W., C.A. Oviatt, and S.S. Hale. 1976. "Nitrogen regeneration and the metabolism of coastal marine bottom communities." Pp. 269-283 in Anderson, J.M., and A. Macfadyen (Eds.), *The Role of Terrestrial and Aquatic Organisms in Decomposition Processes.* Blackwell Scientific Publications, Oxford, England, United Kingdom.

Nixon, S.W., J.R. Kelly, B.N. Furnas, C.A. Oviatt, and S.S. Hale. 1980. "Phosphorous regeneration and the metabolism of coastal marine bottom communities." Pp. 219-242 in Tenore, K.R., and B.C. Coull (Eds.), *Marine Benthic Dynamics.* University of South Carolina Press, Columbia, SC.

Nixon, S.W., C.A. Oviatt, J. Frithsen, and B. Sullivan. 1986. "Nutrients and the productivity of estuarine and coastal marine ecosystems." J. Limnol. Soc. South. Afr. 12(1/2):43-71.

NOAA. 1985. *National Estuarine Inventory Data Atlas.* Vol. 1: Physical and hydrologic characteristics. National Oceanic and Atmospheric Administration. U.S. Government Printing Office, Washington, DC.

NOAA/EPA. 1988. Strategic assessments of near coastal waters, northeast case study. Chap. 3: Susceptibility and concentration status of northeast estuaries to nutrient discharges. Strategic Assessment Branch, National Oceanic and Atmospheric Administration; Office of Marine and Estuarine Protection and Environmental Results Branch, Environmental Protection Agency; and College of Marine Studies, University of Delaware. NOAA, Rockville, MD. 49 pp.

Normandeau Associates. 1990. An evaluation of model structures, water quality monitoring programs, and historical data for the study of the Deer Island Ocean Outfall. Report to Massachusetts Water Resources Authority. June.

Officer, C.B., and J.H. Ryther. 1980. "The possible significance of silicon in marine eutrophication." Mar. Ecol. Prog. Ser. 3:83-91.

Oviatt, C.A., M.E.Q. Pilson, S.W. Nixon, J.B. Frithsen, D.T. Rudnick, J.R. Kelly, J.F. Grassle, and J.P. Grassle. 1984. "Recovery of a polluted estuarine system: A mesocosm experiment." Mar. Ecol. Prog. Ser. 16:203-216.

Oviatt, C.A., A.A. Keller, P.A. Sampou, and L.L. Beatty. 1986. "Patterns of productivity during eutrophication." Mar. Ecol. Prog. Ser. 28:69-80.

Oviatt, C.A., J.G. Quinn, J.T. Maughan, J.T. Ellis, B.K. Sullivan, J.N. Gearing, P.J. Gearing, C.D. Hunt, P.A. Sampon, and J.S. Latimer. 1987. "Fate and effects of sewage sludge in the coastal marine environment: A mesocosm experiment." Mar. Ecol. Prog. Ser. 41:187-203.

Parker, J.I. 1980. Phytoplackton primary productivity in Massachusetts Bay. Ph.D. thesis. University of New Hampshire, Durham, NH.

Pearson, T.H., and R. Rosenberg. 1978. "Macrobenthic succession in relation to organic enrichment and pollution of the marine environment." Oceanogr. Mar. Biol. Annu. Rev. 16:229-311.

Pilson, M.E.Q. 1985. "On the residence time of water in Narragansett Bay." Estuaries 8:2-14.

Rhoads, D.C., and J.D. Germano. 1986. "Interpreting long-term changes in benthic community structure: A new protocol. Hydrobiologia 142:291-308.

Robinson, W.E., T.J. Coffey, and P.A. Sullivan. 1990. New England Aquarium's ten year Boston Harbor monitoring program. First report (March 1987-July 1989). 108 pp. + app.

Ryther, J.H., and C.B. Officer. 1981. "Impact of nutrient enrichment on water uses." Pp. 247-261 in Neilson, B.J., and L.E. Cronin (Eds.), *Estuaries and Nutrients*. Humana Press, Clifton, NJ.

Shea, D., and J.R. Kelly. 1991. Massachusetts Bay sewage-outfall monitoring plan. Battelle Ocean Sciences draft report to the Massachusetts Water Resources Authority. April 12, 1991. 32 pp. + app.

Shea, D., D. Lewis, B. Buxton, D. Rhoads, and J. Blake. 1991. The sedimentary environment of Massachusetts Bay: Physical, chemical, and biological characteristics. MWRA Environ. Quality Dept. Tech. Rep. Ser. No. 91-6, June 1991. Massachusetts Water Resources Authority, Boston, MA. 139 pp.

Smith, S.V., W.J. Kimmerer, E.A. Laws, R.E. Brock, and T.W. Walsh. 1981. "Kaneohe Bay sewage diversion experiment: Perspectives on ecosystem responses to nutritional perturbation." Pac. Sci. 35:279-395.

Townsend, D..W. , L.M. Cammen. J.P Christensen, S.G. Ackleson, M.D. Keller, E.M. Haugen, S. Corwin, W.J. Bellows, and J.F. Brown. 1990. Seasonality of oceanographic conditions in Massachusetts Bay. Bigelow Laboratory for Ocean Sciences final report to Massachusetts Water Resources Authority. December. 98 pp.

Turner, J.T., D.G. Borlman, W. Lima, and R.W. Pierce. 1989. A seasonal study of plankton, larval fish, and water quality in Buzzards Bay, Massachusetts. Interim data report. Massachusetts Division of Water Pollution Control Research and Demonstration Project 87-15. May. 248 pp.

Valiela, I., and J.E. Costa. 1988. "Eutrophication of Buttermilk Bay, a Cape Cod coastal embayment: Concentrations of nutrients and watershed nutrient budgets." Environ. Manage. NY 12:539-553.

Appendix A

Notations on

CALCULATIONS,
STATISTICS ON MASSACHUSETTS BAY/BOSTON HARBOR,
and
DATA SOURCES

SECTION 2.1.1

(a) Nutrient Loading to Boston Harbor

From Menzie-Cura (1991), the annual total nitrogen input to the whole harbor is about 13,086 metric tons, mostly to the northern Harbor area (see their map 1) and the majority is from effluent discharge. Using the areas provided for northern Harbor and southern Harbor of 5.1×10^7 and 5.7×10^7 m^2, respectively, the total Harbor area is 1.08×10^8 m^2. From these, the area-based annual load to the whole Harbor then is 8643 mmol N/m^2. Using an average depth of the Harbor of 5.8 m (NOAA, 1985, as used in Robinson *et al.*, 1990), the volumetric loading to the whole Harbor is about 1490 mmol/m^3/year.

(b) Estimate of PON from Chlorophyll for Boston Harbor

From Table 9 of Robinson *et al.* (1990), the three-station average for chlorophyll is 3.06 μg/L. Assuming a ratio of C/Chl = 30 (by weight) and a C/N ratio of 6.625 (by atoms), this represents 1.15 μM as PON. This value compares to 10.2 μM N as DIN (see text). A low PON/(DIN + PON) ratio of 0.10 may be characteristic of a highly eutrophic condition (Kelly *et al.*, 1985).

(c) Forms of N in Loading

The forms of N included in total N loading are not described, but presumably this number represents "total Kjeldahl nitrogen" (TKN) as used in SEIS (EPA, 1988). For MWRA sewage effluent (e.g., from August 1990; M. Mickelson, personal communication) ammonium and nitrate are slightly less than 50% of the TKN. Ammonia also represents over 40% of the TKN in sludge, which contributes part of the N load to the northern Harbor (about 14% of that contributed by effluent, which itself is about 84% of the total estimated Harbor load). Some fraction of the organic nitrogen in effluent and sludge may be rapidly digested and converted to inorganic forms, so a reasonable lower bound for the fraction of total loading (Menzie-Cura, 1991) represented by DIN would be 50%. This fraction is used in the text.

(d) Water Residence and Flushing of Boston Harbor

A freshwater fraction replacement calculation follows from Ketchum (1950) [see Pilson, 1985], using the equation

$$V_f = (1 - S_b/S_o)\, V_b \quad ,$$

where V_f/Inflow (freshwater) = replacement time.

The area of the Harbor was assumed to be 1.08×10^8 m^2 (Menzie-Cura, 1991) and the average depth 5.8 m (NOAA 1985) to calculate V_b = Harbor volume. S_b, the volume-weighted mean salinity of the Harbor, and S_o, the salinity of ocean endmember, were assumed to be 30 and 31.5 ppt, respectively. These values are rough estimates from Robinson *et al.* (1990). The calculation is sensitive to these parameters — for example, a lower S_b increases V_f and lengthens the estimate of replacement time.

Menzie-Cura (1991) give a Harbor-wide freshwater input of 32 m^3/s [compared to about 21 m^3/s for MWRA discharge alone (EPA, 1988)]. Using the Harbor-wide value yields a replacement time of 10.5 days.

A tidal prism approximation method was calculated with the following assumptions: a tidal height of 1.2 m per 12-h cycle and an average depth of 5.8 m for the Harbor, as above. Calculations would be improved with knowledge of actual tidal volume incoming and precise Harbor volume. At 1.2 m influx per 5.8 m, this is 20.7% input per cycle. Assuming complete mixing of influx and Harbor water, then a half-time for full replacement can be calculated as

$$V_t/V_i = e^{-0.207\,(\mathrm{cycles})} \quad ,$$

where the fraction of the original volume (V_i) at time t is V_t/V_i. At $V_t/V_i = 0.5$, the half-time is 3.34 cycles or 1.67 days. Similarly, using $V_t/V_i = 0.03$ (i.e., 3% remaining or 97% flushed) the result is 8.46 days.

(e) Replacement Time for Nitrogen in Boston Harbor Water.

Assuming that the average DIN + PON in the Harbor is 11.35 μM (see text, and note b, above), the Harbor has 6.26×10^8 m^3 of water (note a, above), and the total N input is 1.21×10^{10} g N/year (note a, above), then the

$$\begin{aligned} \text{Replacement time} &= \text{Mass in Harbor/Input} \\ &= 0.00826 \text{ years} \\ &= 3 \text{ days} \end{aligned}$$

If 50% of the total N input is as DIN or readily available forms (note c, above), then the calculated time is 6 days. As suggested below, this calculation could be done using the mass of N enriched over background, but it is complicated to estimate this value for the Harbor. The calculation is intended only as a simple guide and should not be taken as a precise estimate.

SECTION 2.1.2

(a) Enriched Area outside Boston Harbor.

An enriched nitrogen area (where DIN was above background of 5.5 μM and PON + DIN was above background of 9 μM) represented by a semicircular quadrant extending eastward from Deer Island (Station 6) to the proposed outfall site (inside Station 8), and from south of Station 5 to just south of Station 18 was conservatively estimated as 177 km^2. Using appropriate depths through this area, the total volume is 2.4×10^{12} L. Using appropriate values of DIN and PON (Figure 4) for this area, the mass enrichment (above background) was calculated as about 6×10^9 mmol (DIN) and 7.5×10^9 mmol (DIN + PON). Comparing the DIN + PON mass enrichment to the total load to the Harbor (9.334×10^{11} mmol/m^2/year, from data in note a), one can calculate a rough replacement time as

$$\text{Replacement time} = \text{enriched mass/mass input to Harbor.}$$

For the above data, the replacement is then calculated as less than 3 days. Again, this gets longer if only DIN input to the Harbor is used, and will increase proportionally as the fraction of N exported from the Harbor decreases. As suggested in the text, a timescale of days is appropriate. Note also that if the total mass of N in the water, rather than that amount above the background level, is used in the calculation, the replacement time is estimated as less than 12 days.

SECTION 2.1.3

(a) Average depth of Massachusetts Bay.

The average depth of Massachusetts Bay was calculated from volumes and areas given by NOAA (1988) to be about 24 m. As noted in summary statistics at the end of this appendix, depending on the definition of the Bay, the average depth used could range from 23.6 to 50 m. The text suggests what value was used in a given calculation if the range was not used.

SECTION 2.1.4

(a) Rough Extrapolation of Benthic Fluxes to an Annual Value.

An exponential relationship between benthic fluxes and temperature (Nixon *et al.*, 1976) suggests an equation of the form

$$F_t = F_0 \, e^{0.16T} \quad ,$$

where F_t = the flux at temperature t (°C), F_0 = the flux at 0 °C, and T = temperature in °C. Knowing the temperature and flux from October from Giblin *et al.* (1990), one can then calculate a flux at any temperature by using the above equation. Then, assuming a sinusoidal temperature curve (based on data of Townsend *et al.*, 1990) for the bottom-water temperature near the proposed outfall site, one can extrapolate fluxes throughout the year. I assumed a maximum of about 12 °C in August and about 2 °C in January. Integration through the year then gives an annual value.

SECTION 2.3

(a) Nitrogen Loading to Massachusetts/Cape Cod Bay.

The estimated loading rate varies depending on the definition of the geographic area of the system (see end of this appendix), the sources included as input functions, and as a fraction of the loading estimate reference. Because of these, a loading rate should not be regarded as a precise number and it is most appropriate to regard estimates in an order of magnitude sense, as has been done in Figure 1 and Figure 13.

The two main references for loading are NOAA (1988) and Menzie-Cura (1991). NOAA (1988) gives the N load to Massachusetts Bay as 7994 tons/year and to Cape Cod Bay as ~380 tons/year. Menzie-Cura (1991), using different definitions for these areas, and including some additional sources like input from the Merrimack and Atmospheric Deposition, suggests N inputs ranging from about 2.4×10^7 to 3.2×10^7 kg/year for Massachusetts Bay (with Merrimack

River input) to about 2.7 to 3.4 $\times$ 10^7 kg/year to Massachusetts and Cape Cod Bays combined (with Merrimack River input but without dredging, which redistributes and is not a new input).

Using a variety of the loads and areas available, one can calculate a range of loading whereby N input could be as low as 100-200 mmol/m^2/year, especially if loading to Boston Harbor is not included and the area considered is both Massachusetts bays. Values range to as high as 962 mmol/m^2/year if the load to Massachusetts Bay only (Menzie-Cura, 1991) data with all appropriate sources are included. Many combinations of the area and load yield estimates in the range of 500-700 mmol/m^2/year on an area based at about 10-20 mmol/m^3/year on a volumetric basis. As suggested in Figure 1, an uncertainty of plus or minus about one loading class is appropriate as a guide, and the range given in Figure 13 is to suggest similar uncertainty.

SECTION 2.4

(a) DO in Stellwagen Basin.

For a 0.5-mg/L decrease in O_2 in lower waters of northern Stellwagen Basin, what mass of carbon must be oxidized? Assume that the area of northern Stellwagen Basin immediately "downslope" from the proposed outfall site is about 10 $\times$ 10 km^2 and the depth goes from about 50 to 80 m, or 30 m; then the volume represented is 3 $\times$ 10^{12} L of water. Assuming an respiratory quotient (RQ) of 1.0, then one can calculate that a drop of O_2 by 0.5 mg/L would take 5.6 $\times$ 10^8 g carbon. If expressed per unit area, this is 5.6 g C/m^2.

Assuming 10% C transport of surface production to deep water, then this requires 56 g C/m^2 enhanced production in the overlying water. Calculated as nitrogen using a Redfield ratio, this requires 704 mmol N/m^2. For a surface layer 25 m deep, this would amount to an increase of 28 μM DIN at the start of a winter/spring bloom for example, which is an enrichment level not currently seen even in the Harbor.

The load of carbon from the outfall (assuming solids are about 40% organic C) may be on the order of about 0.5 to 1.5 $\times$ 10^{10} g C/year (EPA, 1988, Table 5.1.1.a). This is approximately 9 to 27 times the amount calculated above needed for the given decrease in DO. Fairly rapid deposition of this matter is expected rather than efficient horizontal transport to deeper neighboring waters. Even so, the rough calculation suggests that rapid transport of only some of this allochthonous carbon potentially could affect basin DO. Moreover, stimulation of "new" POC as phytoplankton also transportable to neighboring areas could add additional DO demands and has not been included in this illustration calculation. The likelihood is not assessed by this crude calculation, but POM transport of either allochthonous (solids input) or autochthonous (phytoplankton production) POM toward deeper waters should be considered in the monitoring program.

SUMMARY STATISTICS FOR MASSACHUSETTS BAY/BOSTON HARBOR

The area of different sections of Massachusetts bays can be difficult to establish as definitions vary. Various definitions are given here and the text indicates what was used in a given calculation.

	Area (m^2)	Average Depth (m)	Reference
South Boston Harbor	5.7×10^7	—	Menzie *et al.* (1991
North Boston Harbor	5.1×10^7	—	Menzie *et al.* (1991)
Total Boston Harbor	1.08×10^8	5.8	Menzie *et al.* (1991), Robinson *et al.* (1990)
Massachusetts Bay	9.4×10^8	23.6	NOAA (1988)a
Massachusetts Bay	2.2×10^9	50	Menzie-Cura (1991)[b]
Massachusetts Bay	2.37×10^9	46.9	Menzie (1991)[c]
Cape Cod Bay	1.4×10^9	23.6	NOAA (1988)[d]
Cape Cod Bay	1.3×10^9	25.4	Menzie-Cura (1991)[e]
Total Massachusetts and Cape Cod Bays	3.67×10^9	39.3	Menzie-Cura(1991)[f]

[a]This reference does not define the area encompassed, nor is it described in NOAA (1985). Presumably it does not include deeper areas, however, especially Stellwagen Basin, and may only include an area west of a line running from about Cape Ann to Scituate.

[b]Area does not include Broad Sound or Boston Harbor and was derived by Menzie-Cura (1991) from "MIT Collegium, 1989." It apparently defines Massachusetts Bay, to include Stellwagen Basin, as an area west and north of lines running from Cape Ann to Provincetown and Provincetown to just south of Plymouth Bay.

[c]Same as note b, but includes Broad Sound and Boston Harbor.

[d]This area is defined in NOAA (1985) to include Plymouth/Duxbury Bays and the area south of a line running from there to Provincetown.

[e]Area defined does not include Plymouth/Duxbury Bays, but covers area south of a line running from just south of Plymouth Bay directly to Provincetown. This area was derived as described for note b.

[f]Includes Broad Sound and boston Harbor, as for note c.

Appendix B

TABLE OF ANNUAL AVERAGE VALUES CALCULATED FOR SURFACE WATERS OF MASSACHUSETTS BAY

Table B-1. Annual Average Concentrations for Surface Waters in Massachusetts Bay

Station[a]	Latitude	Longitude	Number of Cruises	Surface Depth Interval Considered[b] (m)	Chlorophyll *in situ*[c] (μg/L)	DIN[d] (μM)	PON[e] (μg/L)	DIN & PON[f] (μM)	Zooplankton Biomass[g] (cm^3/m^3)
1	42.577	70.409	5	20-30	1.9	5.1	35.23	7.62	
3	42.517	70.609	5	13-20	2.3	4.7	44.41	7.87	
5	42.485	70.805	5	9-10	3	5.3	43.49	8.41	
6	42.333	70.933	6	1-3	2.8	12.3	70.71	17.35	
7	42.37	70.839	6	2-12	3.3	8.67	65.6	13.36	6.0
8	42.401	70.743	6	10-17	2.9	5.6	56.49	9.64	6.95
9	42.415	70.636	6	10-16	2.5	5.5	61.21	9.87	
10	42.439	70.532	5	8-17	1.6	4.4	42.5	7.44	6.5
11	42.46	70.43	5	14-25	1.9	5	36.77	7.63	
12	42.483	70.326	5	10-27	1.6	5.15	29.64	7.27	6.5
13	42.37	70.247	6	10-25	3.2	5.5	41	8.43	
15	42.333	70.429	6	10-20	3.3	5.4	46.64	8.73	
17	42.285	70.65	5	2-21	2.6	4.47	47.5	7.86	
18	42.266	70.748	6	5-12	3.9	7.3	62	11.73	

[a]Station numbers are from Townsend *et al.* (1990). Table values here were derived from their Tables 1-6, the number of cruises (fall 1989 - summer 1990) available for each station is indicated as a separate column in the table.

[b]An appropriate range (5-6 cruises) of the deepest bottle samples included in calculating the average depth-integrated surface-water concentration is indicated. The depth distributions of sampling points varied across stations, and time and points were not equidistant depth intervals, so a weighting by depth was appropriate. At a station and time, the depth-integrated average (usually 2-3 points within the top 25 m) was made by assuming that a measured parameter value at a depth was representative of an interval one-half the distance to the next sampling depth, or to the surface or bottom if that were the case. Multiplying measured point concentrations times the volume (per meters squared) of that interval yields the mass in a given depth interval. The sum of the mass divided by the summed volume over several intervals yields the average concentration over the depth range. The *annual* average concentration was then calculated as the mean of 5-6 cruises, which were spaced over time through the annual cycle. The depth interval was chosen to include the maximum near-surface chlorophyll value at a given date.

[c]*In situ* chlorophyll from Townsend *et al.* (1990) tables, based on fluorometry.

[d]DIN: dissolved inorganic nitrogen, or ammonium + nitrite + nitrite.

[e]PON: particulate organic nitrogen.

[f]DIN + PON, as μM, assuming PON in μg/L divided by 14 gives μM N.

[g]Zooplankton biomass, as settled volume from tow 0-20-0 m (Station 7) or 0-35-0 m (Stations 8, 10, 12). Data are from 5 cruises at each station; inclusion of data from March, available for only Stations 10 and 12, would yield higher annual average, 7.5 and 8.2, respectively.

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