2005 Nutrient Issues Review

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EXECUTIVE SUMMARY

This report is one of a series of reviews on nutrient issues in the Boston Harbor and Massachusetts Bay region for the Massachusetts Water Resources Authority (MWRA). Past nutrient issue reviews have focused on eutrophication concerns (Kelly 1991, 1993), water column and sediment metabolism (Kelly and Doering 1995 and Howes 1998, respectively), and food web models for the bays (Hunt *et al.* 1999). The transfer of the MWRA sewage discharge from the harbor to the bay was predicted to cause only minor changes to Massachusetts Bay, while allowing substantial improvements in the harbor. The MWRA Ambient Monitoring Plan (MWRA 1991) recognized that some small changes might occur in the bay, but raised basic questions about system variability, processes, and function due to limited available data. After 15 years of monitoring, the prediction of minor impacts has proven to be reasonably accurate and most of the basic questions have been answered (MWRA 2003, Libby *et al.* 2006a).

This report presents a narrative summary of our understanding of the Massachusetts and Cape Cod Bay system gained from the monitoring and research conducted by MWRA and others over the past fifteen years. The report also considers questions that linger with respect to potential effects of MWRA's nutrient discharges:

(1) How sensitive is the bay's ecosystem to changes in nutrient loads within the current mode of delivery?

(2) What long term, cumulative effects should be expected in Massachusetts Bay as a result of diversion to the bay outfall?

(3) What has led to changes in phytoplankton assemblages that have been observed coincident with, but not necessarily in response to, the offshore diversion of discharge?

After nine years of baseline monitoring and five years of post-diversion monitoring, changes in the nutrient regimes in Boston Harbor and in Massachusetts Bay are unambiguous – dissolved inorganic nutrient (primarily ammonium – NH_4) levels have dramatically decreased in the harbor and harbor-influenced coastal waters while detectably increasing within a few kilometers of the offshore diffuser system (Libby *et al.* 2006a). In Boston Harbor, the dramatic decrease in dissolved inorganic nitrogen has been associated with a significant decrease in chlorophyll and particulate organic carbon (Taylor 2006), lower primary production, and an apparent change in seasonal primary productivity from a eutrophic pattern peaking in the summer to a more typical temperate coastal pattern similar to the nearfield dominated by winter/spring peaks in productivity (Libby *et al.* 2006b).

In contrast, there is no indication that the higher nutrient concentrations in the vicinity of the bay outfall (nearfield) have translated into significant changes in primary productivity or phytoplankton biomass, whether measured as chlorophyll, particulate organic carbon, or phytoplankton abundance. Moreover, dissolved oxygen levels in nearfield and farfield regions have not changed. Primary productivity in the region of increase nutrients has remained within the variability observed in the baseline period. Five years of post-diversion monitoring has documented apparent increases in the winter/spring phytoplankton biomass (chlorophyll) in the vicinity of the diffuser, but this increase is not statistically significant. Changes to the plankton community include an apparent regional increase in the frequency and duration of *Phaeocystis* blooms in the bays. During baseline, *Phaeocystis* blooms were observed roughly every third year, in winter/spring 1992, 1994, 1997 and 2000. More recently, these blooms have become annual occurrences since 2000. These changes in biomass and phytoplankton appear to be related to variations in regional patterns rather than a direct outfall effect.

Nutrient Changes and Long-term Outfall Effects

Observed changes in Massachusetts Bay in response to the September 2000 outfall diversion have been small. However, the potential for long-term outfall effects has not been fully evaluated. Key to understanding of any long-term effects is gaining insight into what magnitude of change is even possible. In this report, various mass balance approaches were used to understand the potential impact of the nutrient input from the MWRA outfall on primary production and phytoplankton biomass. These approaches included simple box models (e.g., nitrogen budgets), regression models, and a cross-system comparison of nutrient loading and primary production estimates. Application of the mass balance and regression approaches, while providing valuable information for the harbor, are not as conclusive when applied to open water areas due to significant variability and uncertainty associated with determining boundary conditions (salinity gradients; seasonal changes in horizontal and vertical structure). Therefore, the simple box model approach was applied to three regions to estimate the upper theoretical limits of effluent derived nitrogen on system metabolism: (1) Boston Harbor, (2) an expanded nearfield area of Massachusetts Bay, and (3) the farfield area of Massachusetts Bay.

The simple box model approach estimated increases in primary production in an expanded nearfield area to be quite small and consistent with monitoring program observations. Moreover, the estimated impact of the offshore transfer to productivity in the expanded nearfield area is well within the interannual variability that has been observed at the nearfield stations. These analyses indicate the net change in dissolved inorganic nitrogen delivery to the nearfield and the offshore waters, and the resulting change in productivity, is likely insignificant relative to baseline concentrations, consistent with previous findings. The box model and cross-system comparisons and review of monitoring data corroborate the conclusions reported by Hunt *et al.* (1999): that there would be no significant changes in nutrient concentrations or primary productivity in Massachusetts Bay under post-transfer conditions. Other than localized elevations of NH_4 in and around the outfall location, there has been no clear indication that the bay ecosystem has responded measurably to the relocation of effluent discharge. Although the box and regression models are quite general and span a large spatial scale, they illustrate the relative insensitivity of the ecosystem to post-transfer effluent delivery. Conceptually, and intuitively, the relatively low system response in Massachusetts Bay can be primarily explained by dilution and rapid mixing and transport due to the system’s vast volumes relative to the discharge volume and physical dynamics. Most of the nitrogen coming from wastewater sources has always reached Massachusetts Bay, but in past and present scenarios, has only contributed a small part of the available pool of nitrogen. Furthermore, current subpyncnocline discharge of effluent further prevents assimilation by primary producers and thus helps mitigate any potential eutrophic response. The depths in the bay are such that benthic-pelagic coupling is not likely to be significant unless there were large, measurable increases in productivity (Kemp *et al.* 1992).

This analysis support previous conclusions that the, transfer of effluent discharge to Massachusetts Bay was unlikely to significantly change system metabolism from the baseline conditions in the bay. However, continued data collection on system productivity and other information will support future seasonal, annual, and interannual nutrient and production mass-balances and may identify any future nearfield anomalies in productivity. The use of more refined calculations, such as those available in BEM, will provide additional contributions to identifying the true outfall impact and resultant environmental changes. In addition, potential short- and long-term net exchanges associated with the mass-balance estimates of nutrients (and carbon) would be furthered by the application of the BEM. The physics of the system ultimately determines the degree that nutrients are available for assimilation into organic matter over specific temporal and spatial scales. It has been demonstrated that the physics in the Massachusetts Bay system are highly variable on short time scales and even at longer time scales. Quantifying variability in the physics would require expanded sampling efforts that may not be value added given the ability of the model to mimic the system’s physics. Thus, further coupling of modeling with monitoring data will help elucidate the questions addressed in this report.

Phaeocystis Blooms

It is widely accepted that phytoplankton populations are influenced by a variety of factors. In coastal seas, long-term temperature effects are often intermixed with changes and trends in nutrient concentrations. A challenge faced by the monitoring program is to understand the interplay among nutrients and other factors that could be causing the observed changes in the *Phaeocystis* bloom dynamics. This is especially important given the geographically widespread occurrence of the blooms that suggest *Phaeocystis* is responding to regional scale processes rather than the outfall. Examination of the scientific literature and fifteen years of MWRA data suggests an apparent link between *Phaeocystis* bloom duration and interannual variation in spring seawater temperature in the bay. The MWRA data indicates *Phaeocystis* persists longer in years in which the attainment of a temperature limiting growth (14°C) is delayed. There is a strong correlation between the bloom duration and date the bay reaches 14°C; recently this has been in May or even June (cool years). Thus, it appears that the 2002-2005 summer *Phaeocystis* threshold exceedances are likely related to interannual temperature variability rather than any impact due to the bay outfall. Continued monitoring and incorporation of additional monitoring data into the analysis will allow more rigorous evaluation of the link between spring water temperature and *Phaeocystis* bloom duration in Massachusetts Bay.

In the nearfield area, some post-diversion changes in winter/spring nutrient ratios have been detected in the vicinity of the diffuser system – specifically a doubling of the DIN:Si ratio and a halving of the N_{ox}:N_{red} ratio both of which are largely due to the discharge of ammonium. The literature suggests such changes would favor *Phaeocystis* due to the difference in physiological nutrient kinetics of *Phaeocystis* vs. those of diatom populations that typically bloom in the winter-spring. An examination of *Phaeocystis* bloom magnitudes suggests that the post-diversion changes in nutrient availability and ratios have not led to larger blooms. Peak nearfield *Phaeocystis* abundance during the 2004 bloom was the highest that has been observed; however, the next three highest peak abundances were observed during the baseline period (1992, 1997 and 2000). The literature suggests that the increased frequency of *Phaeocystis* blooms may be related to lower N_{ox}:N_{red} ratios. However, the observed *Phaeocystis* blooms tend to be regional in extent and the change in N_{ox}:N_{red} ratio is restricted to within 10-20 km of the outfall (at most), thus the change in frequency is likely coincidental rather than causal. The nearfield monitoring data suggests an increase in bloom duration (noted as summer caution threshold exceedances in 2002-2005) which is correlated with decreases in the N_{ox}:N_{red} ratio. Thus, the decrease in the N_{ox}:N_{red} ratio in the nearfield may also have an impact on the duration of *Phaeocystis* blooms in this region. However, the temporal resolution of the sampling and regional nature of the bloom makes it difficult to confirm this observation. Testing for outfall effects on *Phaeocystis* bloom dynamics remains difficult and would likely require intensive, focused sampling across a gradient from outfall influenced to unaffected waters during a *Phaeocystis* bloom.

Summary

Many of the questions pertaining to the water column in the MWRA Ambient Monitoring Plan have been answered. The diversion of MWRA effluent from Boston Harbor to the ocean outfall in western Massachusetts Bay on September 6, 2000 has resulted in a change in the location, but not the quantity of nutrients to Massachusetts Bay (MWRA 2003, Libby *et al.* 2006a). The outfall clearly has a localized signature that is highlighted by the effluent plume's NH_4 concentrations, which are consistently confined to within 10-20 km of the outfall. Moreover, the monitoring data and modeling results suggest that substantial changes in the Massachusetts Bay ecosystem have not resulted from the transfer of the MWRA effluent from Boston Harbor to Massachusetts Bay. The potential impact of the loading to the bay continues to be minimal even under relatively conservative modeling assumptions. The predicted impact on Massachusetts Bay annual primary productivity is well within the range of interannual variability that has been observed in this parameter and no change has been seen in nearfield production data pre- to post-transfer. The change in *Phaeocystis* bloom occurrence appears to be a regional cycle unrelated to the MWRA outfall. The duration and magnitude of these blooms has been shown to be less a function of nutrient loading than it is of water temperature.

The Ambient Monitoring Plan questions that have not been fully answered require additional assessment and may necessitate changes in the monitoring approach. To understand the more subtle, long-term impacts that the bay outfall may have on the Massachusetts and Cape Cod Bay system will likely require monitoring focused on key ecological and biological processes and the factors that influence them. It is anticipated that MWRA will revisit the monitoring program in 2007 and seek to modify the program based on continued gains in the understanding of the ecosystem and to meet current regulatory requirements. During this process, decisions will need to be made as to whether it is necessary to augment or revise the current monitoring approach to examine and understand the observed subtle impacts of the outfall or if the program should focus specifically on regulatory and permit criteria.

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1. INTRODUCTION

This report is one of a series of reviews on nutrient issues in the Boston Harbor and Massachusetts Bay region for the Massachusetts Water Resources Authority (MWRA). Past nutrient issue reviews have focused on eutrophication concerns (Kelly 1991, 1993), water column and sediment metabolism (Kelly and Doering 1995 and Howes 1998, respectively), and food web models for the bays (Hunt *et al.* 1999). The transfer of the MWRA sewage discharge from the harbor to the bay was predicted to cause only minor changes in the bay, while allowing substantial improvements in the harbor. The MWRA Ambient Monitoring Plan (MWRA 1991) recognized that some small changes in the bay might occur but raised basic questions about system variability, processes, and function due to limited available data. After 15 years of monitoring, prediction of minor impacts have proven to be reasonably accurate and most of the basic questions have been answered (MWRA 2003, Libby *et al*. 2006a).

This report presents a narrative summary of our understanding of the Massachusetts and Cape Cod Bay system gained from the monitoring and research conducted by MWRA and others over the past fifteen years. The report also considers concerns that linger with respect to potential effects of MWRA's nutrient discharges: What is the potential long-term impact of MWRA nutrient inputs on primary production and phytoplankton biomass, what magnitude of change in primary productivity is possible, and explaining changes in plankton that have been observed coincident with, but not necessarily in response to, the offshore diversion of discharge.

1.1 Background

For more than a century, inadequately treated sewage and sewage sludge from the greater metropolitan Boston area was discharged into Boston Harbor. This began to change in 1986, when the MWRA initiated an ambitious program, the "Boston Harbor Project", of extensive upgrades to its regional treatment facilities to stop pollutants from entering the harbor. MWRA achieved a major milestone in 1991 when the discharge of sewage sludge into the harbor ended. The sludge is now converted to fertilizer. This step resulted in visible and measurable improvements to the quality of water and sediments in the harbor (Hunt *et al.* 2006a, 2006b). Other major components of the cleanup have provided continual improvements to the environmental quality of the harbor since 1991 (**Table** 1-1). The last major steps in the facilities improvements were the transfer of the effluent discharge from the mouth of Boston Harbor via a 15 km-long outfall offshore in Massachusetts Bay on September 6, 2000 (**Figure** 1-1) and the completion of the third battery of secondary treatment in March 2001.

Table 1-1. Major Upgrades to the MWRA Treatment System.

Date	**Upgrade**
December 1991	Sludge discharges ended
January 1995	New primary plant on-line
December 1995	Disinfection facilities completed
August, 1997 to March, 2001	Secondary treatment begins to be phased in
July 9, 1998	Nut Island discharges to the southern part of Boston Harbor ceased: south system flows transferred to Deer Island in the northern part of Boston Harbor – almost all flows receive secondary treatment
September 6, 2000	New Massachusetts Bay outfall diffuser system on-line
March 2001	Upgrade to secondary treatment complete

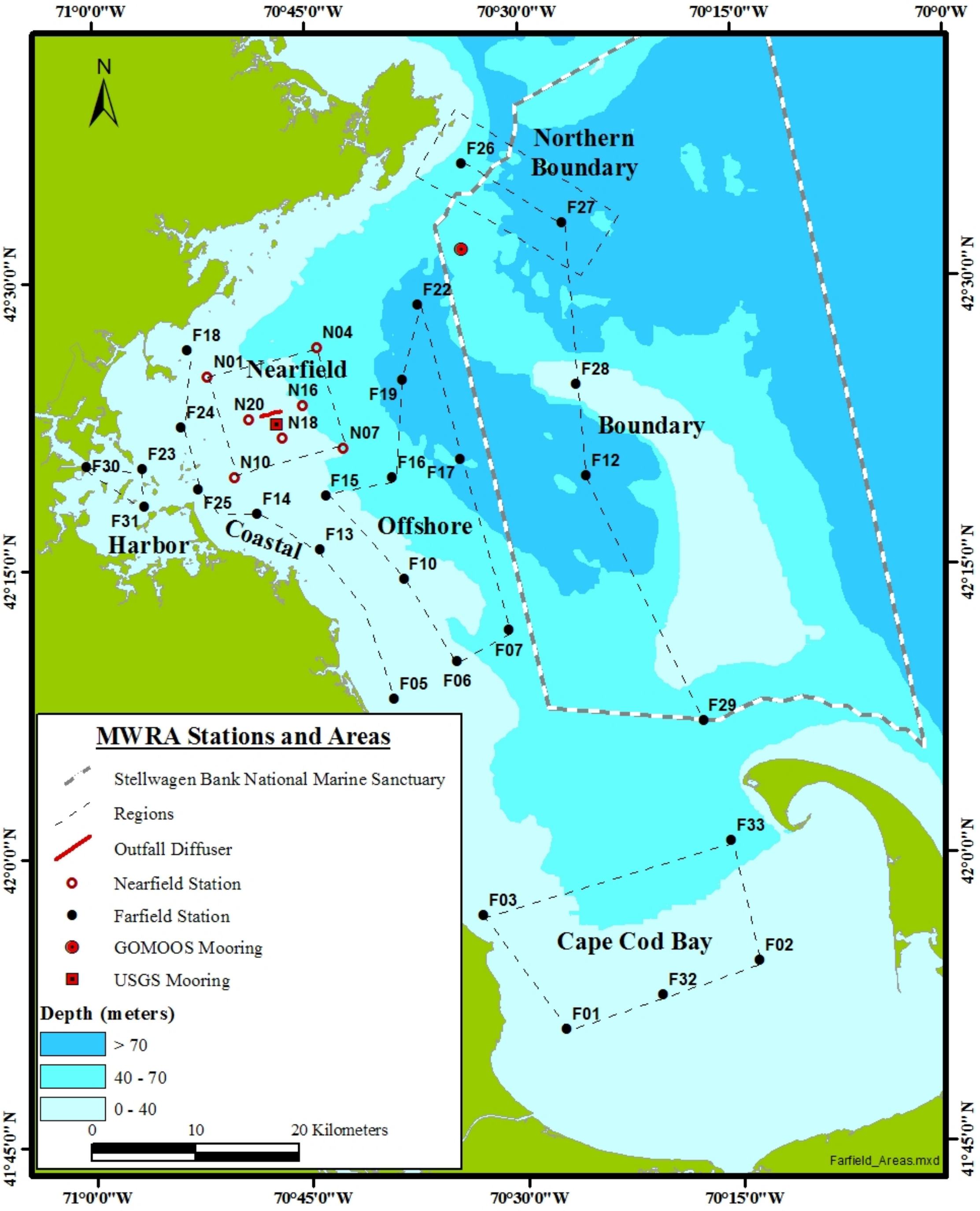


Figure 1-1. Map of Massachusetts and Cape Cod Bays showing the locations of MWRA stations, regional station groupings, and MWRA outfall.

The design, construction, and implementation of MWRA's Boston Harbor Project have received intense review and oversight from state, federal, and public sectors. During the planning phase, concerns were raised regarding environmental impact of the new outfall. To ensure that critical issues were addressed, the federal court overseeing the Project ordered that MWRA monitor Massachusetts Bay for potential impacts. The state convened an Outfall Monitoring Task Force to design a monitoring plan. MWRA initiated a data collection program in 1992 (MWRA 1991, 1997, 2004).

The overall purpose and goals of the Ambient Monitoring Plan are: (1) to verify compliance with the discharge permit, (2) to improve the ability to predict the environmental impact of the relocated outfall in Massachusetts Bay, (3) to measure the actual impact on the bay, and (4) to measure the recovery of the harbor as facility improvements decrease loadings. From 1991 through September 5, 2000, baseline data were collected to compare with the data collected after the outfall came online. Due to an extended construction period, this baseline provides insight into the bay's responses as various facility improvements came on line. Throughout the baseline period, MWRA's monitoring program increased scientific understanding of the physical and ecological processes within the harbor and bays, including the transport and fates of contaminants and nutrients in the system; and spatial, annual, and seasonal variability within Massachusetts Bay. The baseline monitoring data allow an understanding of the harbor and bays against which variability and changes can be evaluated and tested to determine if they result from the effluent diversion.

The Ambient Monitoring Plan was changed in response to new information since 1991 (MWRA 1997, 2004) and it is anticipated that additional modifications will occur in the future. The 1991 Monitoring Plan posed a set of monitoring questions that describe a series of possible environmental responses to the transfer of the discharge from the harbor to the bay outfall. These questions were originally conceived as a basis for evaluating changes and possible responses.

The diversion of MWRA effluent from Boston Harbor to the ocean outfall in western Massachusetts Bay on September 6, 2000 resulted in a change in the location, but not the quantity of nutrients to Massachusetts Bay (MWRA 2003, Libby *et al.* 2006a). Thus far, the only definitive water column changes from baseline are the significant decreases in nutrient (primarily ammonium – NH_4) concentrations in Boston Harbor and nearby coastal waters and an increase in the nearfield in the vicinity of the outfall (Libby *et al.* 2006a). This dramatic decrease in NH_4 in the harbor has been concomitant with significant decreases in chlorophyll and POC (Taylor 2006) and primary production. Moreover, the seasonal pattern in productivity appears to be changing from a summer peak to a winter/spring peak, which is a more typical temperate coastal pattern. In contrast, the higher NH_4 concentrations measured within 10-20 km of the outfall have not translated into significant changes in biomass, whether measured as chlorophyll, particulate organic carbon (POC), or phytoplankton abundance.

1.2 Report Content

This nutrient issues review addresses two of the remaining issues – the potential long-term impact of MWRA nutrient inputs to the bays and changes in *Phaeocystis* blooms – a species identified in the MWRA Contingency plan (MWRA 2001) as a nuisance species – that have been observed coincident with, but not necessarily in response to, the offshore diversion of discharge. Individual sections of the report focus on the following topics:

Section 2 – An overview of the current understanding of water quality conditions in the bays and what has changed since the transfer of effluent discharge to the bay outfall on September 6, 2000 addressing the questions in the Ambient Monitoring Plan (MWRA 1991).

Section 3 – Application of a mass balance approach to bound the potential impacts of MWRA nutrient input on primary production and phytoplankton biomass.

Section 4 – Examination of the apparent change and variability in the frequency, magnitude and duration of spring *Phaeocystis* blooms in Massachusetts Bay.

2. OVERVIEW OF WATER QUALITY IN BOSTON HARBOR AND MASSACHUSETTS BAY

Much has been learned about the Massachusetts Bay system over the course of MWRA's monitoring program. Our understanding of the circulation and importance of the Gulf of Maine to both water properties and biology of the system has led to changes in the way we envision the bay outfall may affect the bay. No longer is the system viewed as a simple upstream to downstream conveyor belt, but rather one that has a weak and seasonal counterclockwise circulation pattern that is often obscured by tidal and local/regional wind forcing. The influence of the Gulf of Maine has been observed on circulation, nutrient loading, DO, and nuisance species in the bays (Beardsley *et al.* 1997; Anderson 1997). Understanding these linkages remains critical for assessing the relative impact of the bay outfall on water quality in Massachusetts and Cape Cod Bays.

After nine years of baseline monitoring and five years of post-diversion monitoring, MWRA has been able to answer many of the questions that were posed when the program began (**Table** 2-1). As expected, monitoring has detected only minimal environmental response within the immediate vicinity of the outfall and overall conditions within the bays have not changed substantially from the baseline. However, there has been a dramatic and expected change in nutrient concentrations in Boston Harbor and the nearfield. Significant changes in harbor productivity and biomass concentrations have been observed (Taylor 2005, 2006; Libby *et al.* 2006a, 2006b), but no such changes have been clearly evident in the nearfield.

This section provides an overview of the current understanding of water quality conditions in Massachusetts and Cape Cod Bays and what has changed since the transfer to the bay outfall with specific attention to the questions outlined in the Ambient Monitoring Plan (MWRA 1991) and summarized in **Table** 2-1. This overview also serves as a basis for discussions presented in Sections 3 and 4, which focus on potential long-term effects on productivity due to the transfer and the documented change in the frequency of *Phaeocystis* blooms, respectively.

2.1 Water Circulation

Physical oceanographic data collected as part of this program and in conjunction with researchers at U.S. Geological Survey (USGS) and Woods Hole Oceanographic Institution (WHOI) show that circulation in Massachusetts and Cape Cod Bays is quite variable, seasonally dependent, and subject to both local and regional forcing. On a regional scale, circulation in the bays is often affected by the larger pattern of water flow in the Gulf of Maine (Beardsley *et al.* 1997). The western Maine coastal current usually flows southwestward along the coast of Maine and New Hampshire. Depending on prevailing oceanographic and metrological conditions, the current can enter Massachusetts Bay south of Cape Ann (**Figure** 2-1). Optimal conditions for input into Massachusetts Bay usually occur during the spring when winds out of the northeast bring significant freshwater inflow from the gulf into the bays. Water transport under these conditions generally follows a counterclockwise path along the coast to Cape Cod Bay. The Merrimack River and rivers further north in the Gulf of Maine provide most of the freshwater inflow to Massachusetts Bay (Manohar-Maharaj and Beardsley 1973, Geyer *et al.* 2004). Although they do not empty directly into the bay, their flow is much greater than the Charles River and other Massachusetts Bay tributary rivers. The spring freshet results in salinity stratification in early April. In late spring and summer, Cape Cod Bay becomes isolated from this circulation.

Table 2-1. Summary of nutrient-related monitoring questions and status as of summer 2006.

Monitoring Question	Status
Are the model estimates of short-term (less than 1 day) effluent dilution and transport accurate?	Model estimates accurate, confirmed by plume studies conducted in 2001 (Hunt *et al*. 2002)
What are the nearfield and farfield water circulation patterns?	Flow is controlled by general circulation in the Gulf of Maine, affected by tides and local wind (Beardsley *et al.* 1997). Bottom currents around the outfall can flow in any direction with no mean flow (Butman *et al*. 2002).
What is the farfield fate of dissolved, conservative, or long-lived effluent constituents?	Changes in farfield concentrations of salinity not detected within tens of meters of outfall and not observed in farfield water or sediments (Hunt *et al.* 2002).
Have nutrient concentrations changed in the water near the outfall; have they changed at farfield stations in Massachusetts Bay or Cape Cod Bay, and, if so, are they correlated with changes in the nearfield?	Changes consistent with model predictions. The effluent signature is clearly observed in the vicinity or the outfall but is diluted over a few days and tens of kilometers. (MWRA 2003, Libby *et al.* 2006a and others)
Do the concentrations (or percent saturation) of dissolved oxygen in the water column meet the state water quality standards?	Conditions unchanged from background in the five years of post-diversion data (MWRA 2003, Libby *et al.* 2006a and others)
Have the concentrations (or percent saturation) of dissolved oxygen in the vicinity of the outfall or at selected farfield stations in Massachusetts Bay or Cape Cod Bay changed relative to baseline or a reference area? If so, can changes be correlated with effluent or ambient water nutrient concentrations, or can farfield changes be correlated with nearfield changes?	Conditions not changed from background in the five years of post-diversion data (MWRA 2003, Libby *et al.* 2006a and others). Bottom water DO levels in the nearfield appear to be under control of regional physical oceanographic processes (Geyer *et al.* 2002)
Has the phytoplankton biomass changed in the vicinity of the outfall or at selected farfield stations in Massachusetts Bay or Cape Cod Bay, and, if so, can these changes be correlated with effluent or ambient water nutrient concentrations, or can farfield changes be correlated with nearfield changes?	No significant change detected in the five years of post-diversion data (Libby *et al*. 2006b).
Have the phytoplankton production rates changed in the vicinity of the outfall or at selected farfield stations, and, if so, can these changes be correlated with effluent or ambient water nutrient concentrations, or can farfield changes be correlated with nearfield changes?	There has been no observed change in the magnitude of seasonal or annual production rates in the nearfield. Frequency, timing and magnitude of fall blooms in the nearfield appear to be different, but this change appears to be regional and not associated with the discharge. Productivity rates have decreased in Boston Harbor and the seasonal patterns appear to be changing (from summer peak to winter/spring peak in annual maximum productivity), as the harbor transitions from eutrophic conditions to lower nutrient levels (Libby *et al*. 2003, 2006a, 2006b).

Monitoring Question	Status
Has the abundance of nuisance or noxious phytoplankton changed in the vicinity of the outfall?	No outfall-related change detected. Frequency of *Phaeocystis* blooms has increased, but the phenomenon is regional in nature (Libby *et al*. 2006a). Red tide blooms of *Alexandrium fundyense* have occurred in both 2005 and 2006, but again these are regional blooms with no apparent impact associated with the outfall (Anderson *et al*. 2005; Libby *et al.* 2006b).
Has the species composition of phytoplankton or zooplankton changed in the vicinity of the outfall or at selected farfield stations in Massachusetts Bay or Cape Cod Bay? If so, can these changes be correlated with effluent of ambient water nutrient concentrations, or can farfield changes be correlated with nearfield changes?	A clear change has not been detected with five years of post-diversion data (Libby *et al*. 2006b). Other than the frequency of *Phaeocystis* blooms and the 2005 and 2006 *Alexandrium* blooms, no changes have been observed in the phytoplankton community composition. Subtle changes in zooplankton have been noted with lower summer and fall abundances that may be related to presence of *Phaeocystis*, ctenophores or the lack of strong fall blooms, but none of the purported influences have been consistently linked (Libby *et al.* 2006b)

As the surface waters warm up in May and June, temperature stratification dominates over that due to the freshwater input. There is a strong and persistent pycnocline throughout most of Massachusetts and Cape Cod Bays in the summer that is occasionally punctuated by upwelling and storm mixing events (Bothner and Butman 2005). During the summer, winds are generally from the south which impedes the flow of surface water into the bay, but are conducive to upwelling along the coast and entry of deep gulf water into the bay. The water column generally remains stratified until late October, when surface cooling and wind stress cause the water column to become vertically mixed.

Wind-induced upwelling and downwelling cause large variations in the water properties at the outfall site by advecting the waters on- and offshore with response times of hours to days. Persistent, strong southerly or southwesterly winds in summer lead to upwelling. Upwelling causes a decrease in both surface and bottom water temperature, but most notably the surface

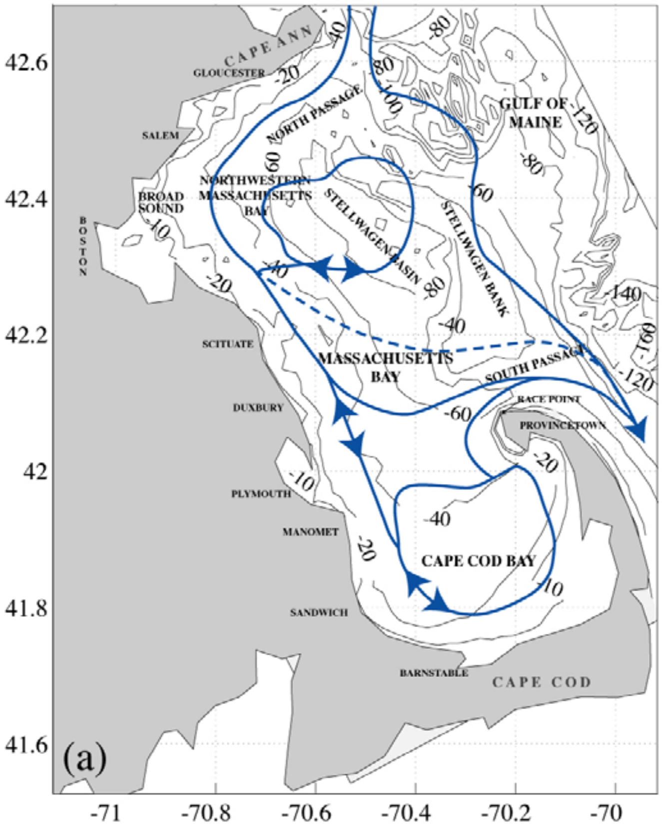


Figure 2-1. Summary of circulation within Massachusetts Bay (Lermusiaux 2001).

water. Downwelling causes an increase in bottom water temperature. Upwelling and downwelling have some influence on vertical exchange, but their main influence is the horizontal advection of gradients. Wind effects also include temporary destratification of the water column by large summer storms (for example, Hurricane Bob in 1991). A stormy early autumn can also lead to early fall turnover.

The importance of the input of Gulf of Maine water to Massachusetts and Cape Cod Bays cannot be overemphasized as research has shown it to be a major influence on circulation, water properties, and biology in the bays (Beardsley *et al.* 1997, HydroQual 2000, Anderson 1997). Massachusetts and Cape Cod Bays are clearly part of and influenced by the greater Gulf of Maine. Understanding this connection and taking it into account is critical in assessing the relative impact that the MWRA outfall may (or may not) have on water quality in Massachusetts and Cape Cod Bays.

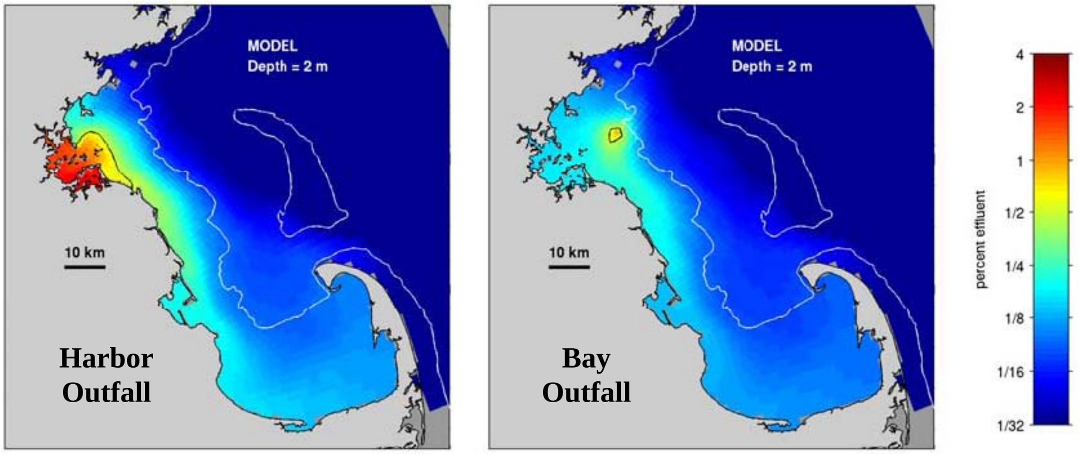
2.2 Effluent and Nutrient Plume

The general circulation within Massachusetts Bay together with local conditions and mixing determine the fate and transport of effluent discharged from the outfall. Vertical transport of the effluent plume is controlled by density gradients and horizontal transport is determined by tides and wind-driven flow. In the winter, the water column is well mixed and the effluent plume reaches the surface; from about April through October, the water column is stratified and the effluent plume is trapped below the pycnocline. The extent of horizontal exchange has been examined by USGS researchers and indicates that there is essentially no mean flow at the outfall location; bottom currents of approximately 6 cm s^{-1} are variable in direction (Butman *et al.* 2002.) The primary temporal and spatial scales of variability near the outfall are those of the tides and of local weather patterns. The key point is that although the long-term average net velocity is small at the outfall site, there is considerable “random” motion, which causes water parcels to be exchanged freely between the outfall site and other parts of the bay (Butman *et al.* 2002.) The largest displacements are observed in summer surface waters when the density gradient allows surface waters to slip relative to bottom waters, and thus surface waters move more readily in response to wind and tide. However, the effluent plume remains confined below the pycnocline and is only detectable within 10-20 km of the outfall (Libby *et al.* 2006a).

The impact of the effluent is minimized by dilution. A 2-km long diffuser with 271 open ports on 53 active risers disperses the effluent into the 30-m deep waters in the bay, where the effluent mixes rapidly with large volumes of seawater. This was documented by a study conducted in July 2001 that used rhodamine dye to track the distribution and estimate the dilution of the effluent plume (Hunt *et al*. 2002). During the study, there was moderate stratification of the water column, as is typical of the early summer. The field results confirmed model predictions that the initial dilution of the effluent is about 70:1 within the zone where the effluent’s momentum causes mixing. The effluent is rapidly diluted by oceanographic processes beyond this zone (Hunt *et al.* 2002) and is dispersed more gradually throughout western Massachusetts Bay. Drifter and model studies predicted that effluent constituents may move toward the shore or offshore where they are incorporated into the general circulation of the bays (Geyer *et al.* 1992). Recent modeling and monitoring results have confirmed that this is the case (Bothner and Butman 2005, Libby *et al*. 2006a).

Ammonium in the water column has proven to be a useful tracer of the effluent plume in the nearfield since the outfall came online in September 2000 (Libby *et al*. 2001). The effluent plume, as defined by the distribution of elevated NH_4 concentrations, surfaces when the water column is well mixed and remains trapped beneath the pycnocline during seasonal stratified conditions. In addition to illustrating the vertical extent of the plume, the NH_4 distribution also highlights the variability in its horizontal distribution (both direction and extent). As discussed above, the predominant circulation pattern in Massachusetts Bay is counterclockwise, but currents are quite variable and highly dependent upon winds. Although the effluent plume has occasionally been observed to extend beyond the nearfield, the plume as characterized by NH_4 concentrations is usually confined to or in close proximity to the nearfield (within 10 to 20 km of the outfall). Recent results reported by USGS (Bothner and Butman 2005) compare modeling output versus monitoring data for baseline and post-diversion years under both winter and summer conditions (**Figures** 2-2 and 2-3). It is clear that the model results that initially supported diversion of the discharge to the offshore outfall have been corroborated by the monitoring results. More importantly, the monitoring data indicate that the diversion has improved conditions in the harbor and coastal waters along the south shore while affecting only a limited area restricted to within 10-20 km of the bay outfall as represented in **Figures** 2-2 and 2-3 by the elevated "effluent" concentration (percent effluent was estimated based on effluent and ambient NH_4 concentrations; Bothner and Butman 2005).

(a) Winter Model Results



(b) Winter Monitoring Data

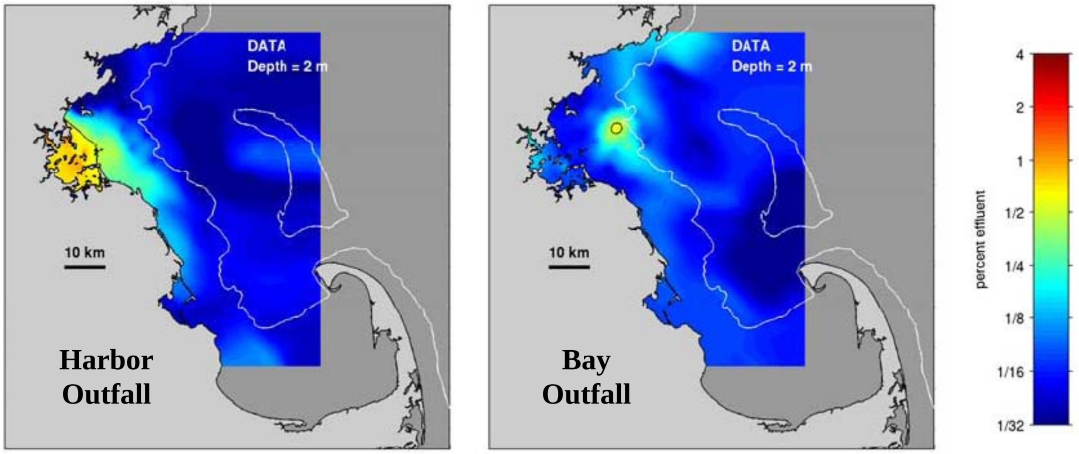
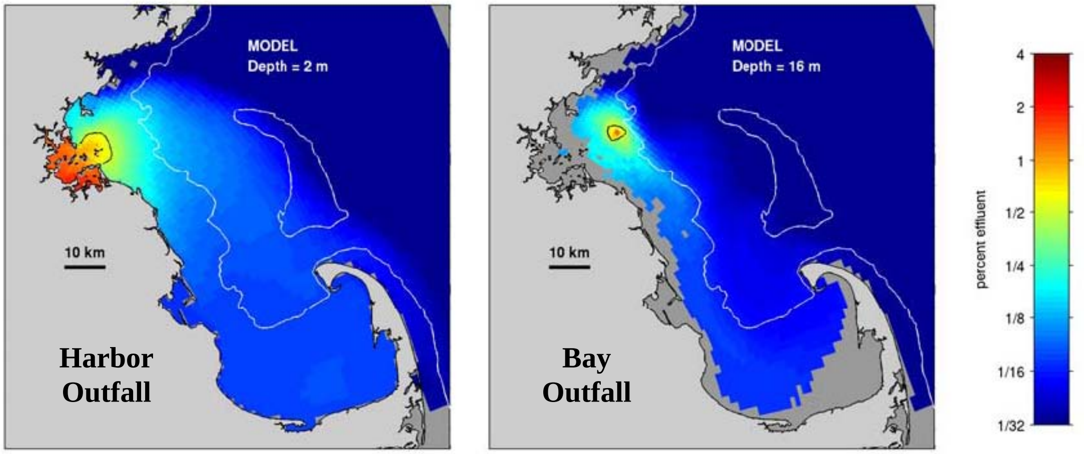


Figure 2-2. Comparisons of winter surface effluent concentrations for both harbor and offshore outfalls based on (a) model results and (b) ammonium monitoring data. (source Bothner and Butman 2005). The seasonal designations for both model results and monitoring data are winter=November, December, January, and February and summer=June, July, and August. The model uses results from an 18 month period (January 1990-August 1991) as in Signell *et al.* (1996). The monitoring results used pre-transfer NH4 data from February 23, 1992 to September 1, 2000 and post-transfer data from September 12, 2000 to December 29, 2004

(a) Summer Model Results



(b) Summer Monitoring Data

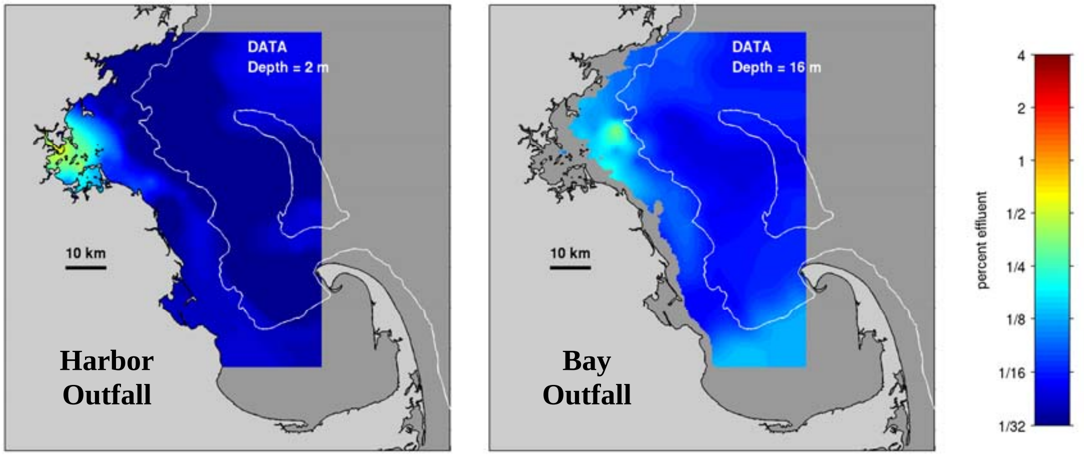


Figure 2-3. Comparisons of summer surface and mid-depth effluent concentrations for both harbor and offshore outfalls based on (a) model results and (b) ammonium monitoring data. (source Bothner and Butman 2005)

There are differences in the detail between the model and monitoring results presented in **Figures** 2-2 and 2-3. This is to be expected given the differences in model and monitoring spatial and temporal resolution, but some of the differences are due to the efficacy of using NH_4 as a tracer. Two high "effluent" areas are depicted in plots of the post-transfer monitoring data just south of Cape Ann in the winter **Figure** 2-2 and in Cape Cod Bay in **Figure** 2-3. These are due to NH_4 pools unrelated to the bay outfall. The winter data in **Figure** 2-2 likely depict the intrusion of nutrient rich coastal waters from the western Gulf of Maine into northern Massachusetts Bay. As for the high Cape Cod Bay levels in **Figure** 2-3, several studies have reported the high nutrient (including NH_4) phenomenon in Cape Code Bay during the summer that results from locally high rates of remineralization (Jiang *et al.* 2006; Becker 1992). These pools of elevated "effluent"/NH_4 are clearly unrelated to the bay outfall and illustrate one of the drawbacks of using NH_4 as a tracer of the effluent plume.

The transfer of the effluent discharge from Boston Harbor into Massachusetts Bay did not create a new source of nutrients to the system, but rather changed the location and water depth of effluent discharge. As expected, NH_4 concentrations have increased significantly at a number of stations in the nearfield closest to the outfall (N16, N18, and N20) since September 2000 (Libby *et al*. 2006b). This increase is evident throughout the year with the largest change observed in subpycnocline waters during the stratified summer months, as predicted. The discharge and presence of elevated NH_4 concentrations in bottom waters in the summer is an improvement compared to the tidal flushing of elevated surface water NH_4 into coastal and nearfield waters during the baseline period (**Figure** 2-3). In contrast, post-transfer NH_4 concentrations in Boston Harbor have been well below baseline levels.

The change in NH_4 concentrations in the nearfield and in Boston Harbor are consistent with model simulations which predicted that the transfer of effluent from Boston Harbor to Massachusetts Bay would greatly reduce nutrients in the harbor and increase them locally in the nearfield (Signell *et al*. 1996). This change was predicted to have little impact on concentrations in the rest of Massachusetts and Cape Cod Bays. The spatial patterns in NH_4 concentrations in the harbor, nearfield, and bays since the 2000 diversion have consistently confirmed this (Libby *et al.* 2006a). The overall shift in NH_4 between baseline and post-transfer years is illustrated in contour plots depicting changes in seasonal mean concentrations across the entire survey area (**Figure** 2-4). The seasonal means are based on MWRA threshold-defined seasons of winter/spring (February-April), summer (May-August), and fall (September-December). The reduction in Boston Harbor and near-harbor coastal station NH_4 concentrations is consistent across each of the seasons as is the increase in NH_4 concentrations in the nearfield area.

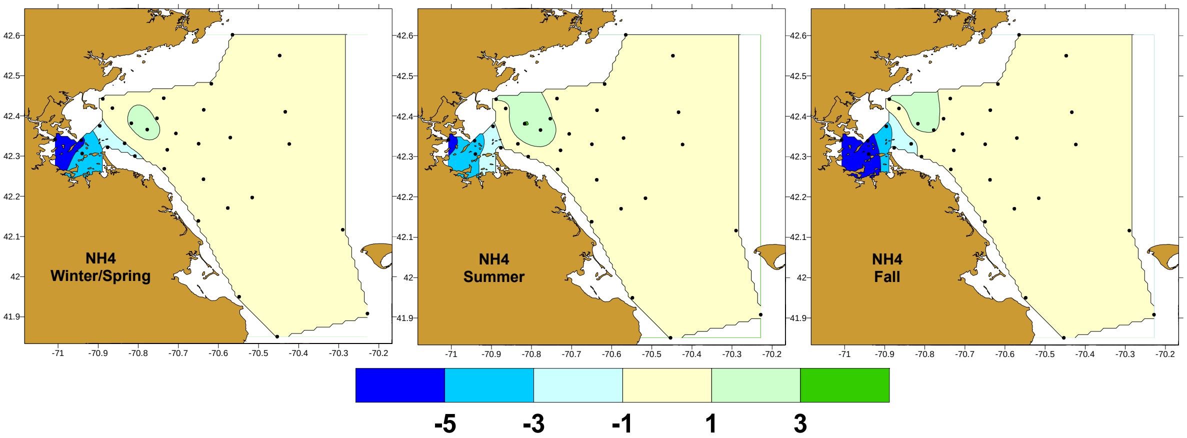


Figure 2-4. Change in seasonal NH_4 concentrations (µM) between baseline (2/92-8/00) when the discharge was in Boston Harbor and post-transfer (9/00-11/05) when the discharge was in Massachusetts Bay. Based on the difference of means calculated over all depths from each station, survey, season, and period. (See Figure 1-1 for station IDs)

In the 2005 annual water column report (Libby *et al.* 2006b), the baseline to post-transfer differences in NH_4 concentrations were examined statistically on a station-by-station basis and significant changes were found at many of the stations in Boston Harbor and the nearfield. The results of these t-test comparisons were based on alpha = 0.05 with Bonferroni correction using the actual number of comparison tests and are shown in **Table** 2-2. When examined on a seasonal basis, there were significant ($p \leq 0.0012$) or testwise-significant ($p \leq 0.01$) decreases in NH_4 concentrations at all but three of the possible 36 station/season combinations of Boston Harbor stations. The magnitude of these decreases ranged from 1 to 9 µM for the 36 season/station grouping. Significant/testwise-significant decreases in NH_4 were also found at coastal stations F14, F24, and F25 during the winter/spring and at nearfield station N10 in the fall (**Table** 2-2). NH_4 concentrations decreased at station N10 during both the winter/spring and fall by more than 1 µM (**Figure** 2-4). This station has historically exhibited patterns in water quality parameters consistent with harbor and coastal areas and is influenced by tidal flow from the inshore waters.

Table 2-2. Summary table of Studentized t-test results comparing baseline versus post-transfer seasonal NH_4 means by station. * indicates significant changes at alpha = 0.05 level with Bonferroni correction (i.e. alpha per family of tests is p=0.05, but since n=41 that corresponds to an alpha per test of p≤ 0.0012 (Abdi 2007)). The colored cells denote direction of change from baseline to post-transfer (red=increase, blue=decrease) for tests having an alpha per test of p≤0.01. The latter are referred to as "testwise-significant" in the text.

Region	Station ID	Winter/Spring	Summer	Fall
Boston Harbor	24	**0.0000***	**0.0046**	**0.0001***
	77	**0.0001***	**0.0004***	**0.0000***
	106	**0.0001***	**0.0012***	**0.0001***
	124	**0.0000***	**0.0019**	**0.0000***
	138	**0.0000***	**0.0076**	**0.0001***
	139	**0.0004***	**0.0017**	**0.0000***
	140	**0.0000***	**0.0065**	**0.0000***
	141	**0.0000***	**0.0002***	**0.0000***
	142	**0.0028**	**0.0002***	**0.0000***
	F23	**0.0016**	0.0839	**0.0010***
	F30	**0.0004***	0.0445	**0.0018**
	F31	**0.0003***	0.0145	**0.0041**
Coastal	F24	**0.0041**	0.2750	0.0280
	F25	**0.0009***	0.2531	0.1660
	F14	**0.0030**	0.7066	0.6731
	F13	0.1462	0.6403	0.8720
	F18	0.9416	**0.0008***	**0.0031**
	F05	0.0710	0.8673	0.1057
Nearfield	N01	0.7629	0.0924	0.0529
	N04	0.4396	0.3759	0.0346
	N07	0.5258	0.2700	0.1664
	N10	0.0277	0.4165	**0.0007***
	N16	**0.0075**	0.0117	0.0312
	N18	**0.0000***	**0.0003***	**0.0073**
	N20	**0.0011***	**0.0016**	**0.0000***
Offshore	F06	0.2377	0.0447	0.1529
	F07	0.1800	0.8083	0.3093
	F10	0.2754	0.2043	0.0804
	F15	0.3413	0.3681	0.1617
	F16	0.1429	0.5313	0.3179
	F17	0.6171	0.4098	0.2123
	F19	0.6333	0.7413	0.7967
	F22	0.4435	0.7850	0.1606
Boundary	F26	0.6811	0.8225	0.1728
	F27	0.9904	0.2908	0.6515
	F28	0.4686	0.6031	0.4394
	F12	0.8212	0.4936	0.3019
	F29	0.6160	0.3969	0.3310
Cape Cod Bay	F01	0.5022	0.8235	0.9395
	F02	0.8514	0.5479	0.3571
	F03	0.1849	0.9032	0.1150

Significant/testwise-significant increases in NH_4 were found at nearfield stations N18 and N20 during each season and at station N16 in winter/spring. Not surprisingly, these three stations are the closest (1-2 km) to the outfall. In the summer and fall, there were also a significant/testwise-significant increases in NH_4 concentrations at coastal station F18.

There are several generalizations that can be made based on the results presented in **Figure** 2-4. First, it is clear that there has been a decrease in NH_4 concentrations in Boston Harbor. Nearly all of the comparisons show a decreasing trend in values and many of them are significant. Some coastal waters and even station N10 in the nearfield have also seen a significant decrease in NH_4 concentrations – likely due to continued influence of harbor water quality at these stations. Second, while there has been an increase in NH_4 concentrations at most of the nearfield, the only significant/testwise-significant increases have been at stations closest to the outfall (N16, N18 and N20) and in the summer at Broad Sound station F18. A significant ($p=0.0008$) increase in NO_3 was also observed at station F18 in the summer. This was the only significant change in NO_3 concentrations even though relatively large (> 1µM) changes were observed at other stations and seasons (**Figure** 2-5). The largest increase in NO_3 concentrations was observed at station F18 (+3.34 µM) in the fall and this change was testwise-significant ($p=0.0042$). Station F18 is located in an area susceptible to upwelling and these significant/testwise-significant summertime and fall increases in NH_4 and NO_3 are likely due to a combination of higher bottom water concentrations (outfall or ambient) and upwelling favorable conditions. Station F18 may also be impacted by the effluent plumes from other regional waste water treatment plants (South Essex Sewer District and City of Lynn plants) that discharge >100 MGD of secondary treated effluent into the area. Nitrate concentrations showed an increase at most nearfield stations during the fall. These increases in NO_3, however, were mirrored by increases throughout the bays (**Figure** 2-5) for example, fall NO_3 concentration at the Northern Boundary stations F26 and F27.

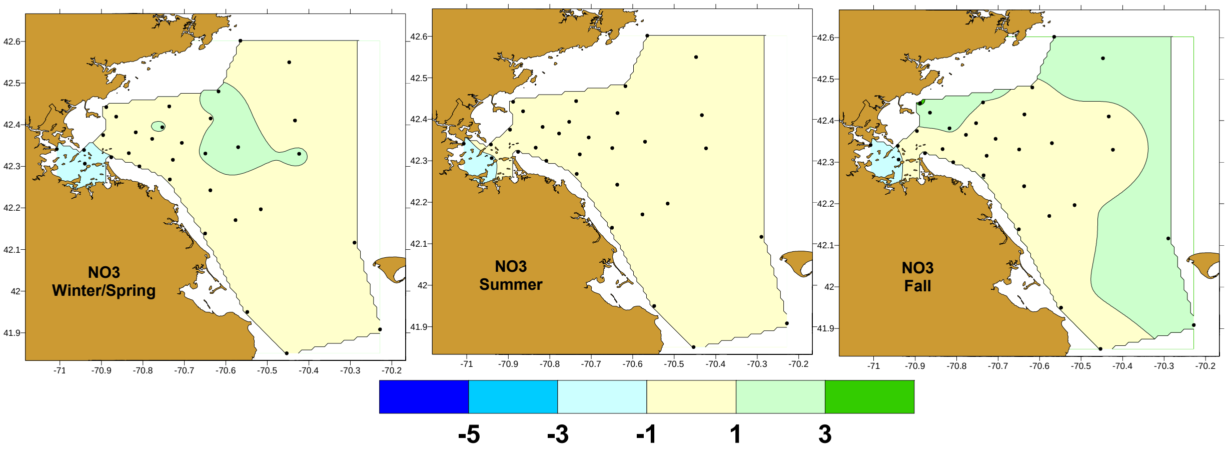
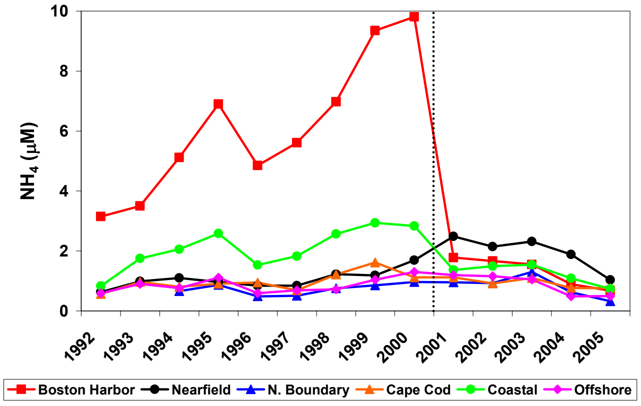


Figure 2-5. Change in seasonal NO_3 concentrations (µM) between baseline and post-transfer periods. Based on the difference of means calculated over all depths from each station, survey, season, and period.

The change in NH_4 and NO_3 concentrations is also manifest in annual mean concentrations for these areas. For example, the annual mean NH_4 concentration in Boston Harbor dropped sharply from 2000 to 2001 (**Figure** 2-6a). A similar sharp decrease was also seen at the coastal stations which are strongly influenced by water quality conditions in Boston Harbor. In contrast, the increase in annual mean NH_4 in the nearfield was not as dramatic as the decrease in harbor and coastal waters. Compared to 1999, however, the last full year before the bay outfall came online; annual mean NH_4 levels in the nearfield had almost doubled. This gradual transition may reflect the period when the discharge, high in NH_4 because of increased levels of secondary treatment, was located at the harbor mouth, and the signature reached into the bay near the present outfall.

Levels in the nearfield in 2005, however, had decreased from previous years to levels comparable to baseline values. This may have been due to the stormy winter/spring and fall of 2005 that likely increased dilution of the effluent NH_4 plume signal in the nearfield. However, this trend in declining NH_4 concentrations from 2003 to 2005 in the nearfield was also observed in each of the other areas of the bays. Overall, except for the apparent decrease in NH_4 concentrations since 2003, levels in offshore, boundary, and Cape Cod Bay waters have remained steady since 1992 suggesting no change at these farfield areas since the transfer to the bay outfall.

(a) Annual NH_4



(b) Annual NO_3

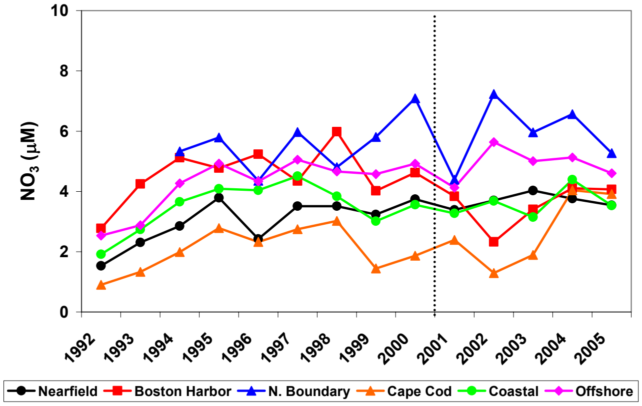


Figure 2-6. Time-series of annual mean (a) NH_4 and (b) NO_3 concentrations (µM) by area. Data collected from all depths and all stations.

The harbor and nearfield trends in annual mean concentrations of other inorganic nutrients are more variable with less change associated with the effluent transfer as seen in the example of NO_3 (**Figure** 2-6b). Year to year variability in NO_3, SiO_4, and PO_4 has more to do with timing of sampling and occurrence of blooms than any clear trends in overall nutrient levels. However, there does appear to be a trend of increasing NO_3 concentrations since the early 1990's in all of the areas except Boston Harbor. The largest change has been seen in the nearfield and offshore Massachusetts Bay and Cape Cod Bay waters. Although the changes in NH_4 concentrations in the nearfield can be ascribed to the relocation of the outfall, the data suggest that this increase occurred in addition to regional changes in nutrient concentrations. Changes in regional nutrient concentrations may be due to different loadings to the bays (riverine, offshore Gulf of Maine surface or bottom waters, etc.), changes in seasonal biological patterns (i.e. fewer and less intense fall blooms), or circulation shifts related to larger scale processes (e.g. North Atlantic Oscillation).

Overall, clear changes between pre- and post-transfer in nearfield, coastal and Boston Harbor nutrient regimes have been documented and are consistent with model predictions. Moreover, while the effluent nutrient signature is clearly observed in the vicinity of the outfall, it is diluted to background levels over a few days of plume travel time and tens of kilometers.

2.3 Impacts on Phytoplankton Biomass and Production

One potential impact of the altered nutrient regimes is a change in phytoplankton biomass trends as measured by chlorophyll and POC. It is known that trends in these parameters are tied to physical conditions, nutrient availability, and ecosystem dynamics. The seasonal phytoplankton biomass in Massachusetts and Cape Cod Bays is dominated by winter/spring and fall blooms, which are typically regional in nature (i.e. southwestern Gulf of Maine). Winter/spring phytoplankton blooms occur when increased light availability, nutrient replete conditions, and seasonal stratification of the physical environment trigger elevated growth rates that are not controlled by temperature-related increases in zooplankton grazing. Typically the timing of the fall bloom has been tied to decreased stratification and increased inputs of nutrients into the surface waters under good light availability. The MWRA monitoring program was designed to address concerns regarding outfall impact by answering the question – “Have changes in nearfield and farfield nutrient levels due to diversion changed the seasonal trends and concentrations of phytoplankton biomass?”

Over fifteen years of monitoring have shown that temporal variability of chlorophyll within the bay system tend to be bay specific. In the winter/spring, Cape Cod Bay often has higher chlorophyll as diatom blooms develop in the bay’s shallow waters earlier than in the deeper waters of Massachusetts Bay. During March/April, the input of fresher, buoyant surface waters from the Gulf of Maine is often conducive to phytoplankton blooms expressed in elevated levels of chlorophyll in Massachusetts Bay. It is difficult to determine whether the increase in chlorophyll results from the transport of phytoplankton into the bays or from the existence of physical and nutrient conditions conducive to increased production. In either case, the influence of the Gulf of Maine on chlorophyll biomass in waters entering Massachusetts Bay near Cape Ann is often apparent in satellite imagery. The 2004 winer/spring phytoplankton blooms for instance are highlighted in a series of satellite images shown in **Figure** 2-7. These images are from February to May and highlight the progression from a February Cape Cod Bay diatom bloom, to the March-April regional *Phaeocystis* bloom, and the sharp decline of this bloom in late April. The use of satellite images allows examination of the distribution of surface chlorophyll both within and outside of the bays and highlights the regional nature of blooms in these waters. The major blooms observed in Massachusetts Bay since SeaWiFS images became available (October 1997) have been regional in nature: that is, there has been a coincident regional expression of elevated chlorophyll values over much of the southwestern Gulf of Maine during each of the blooms.

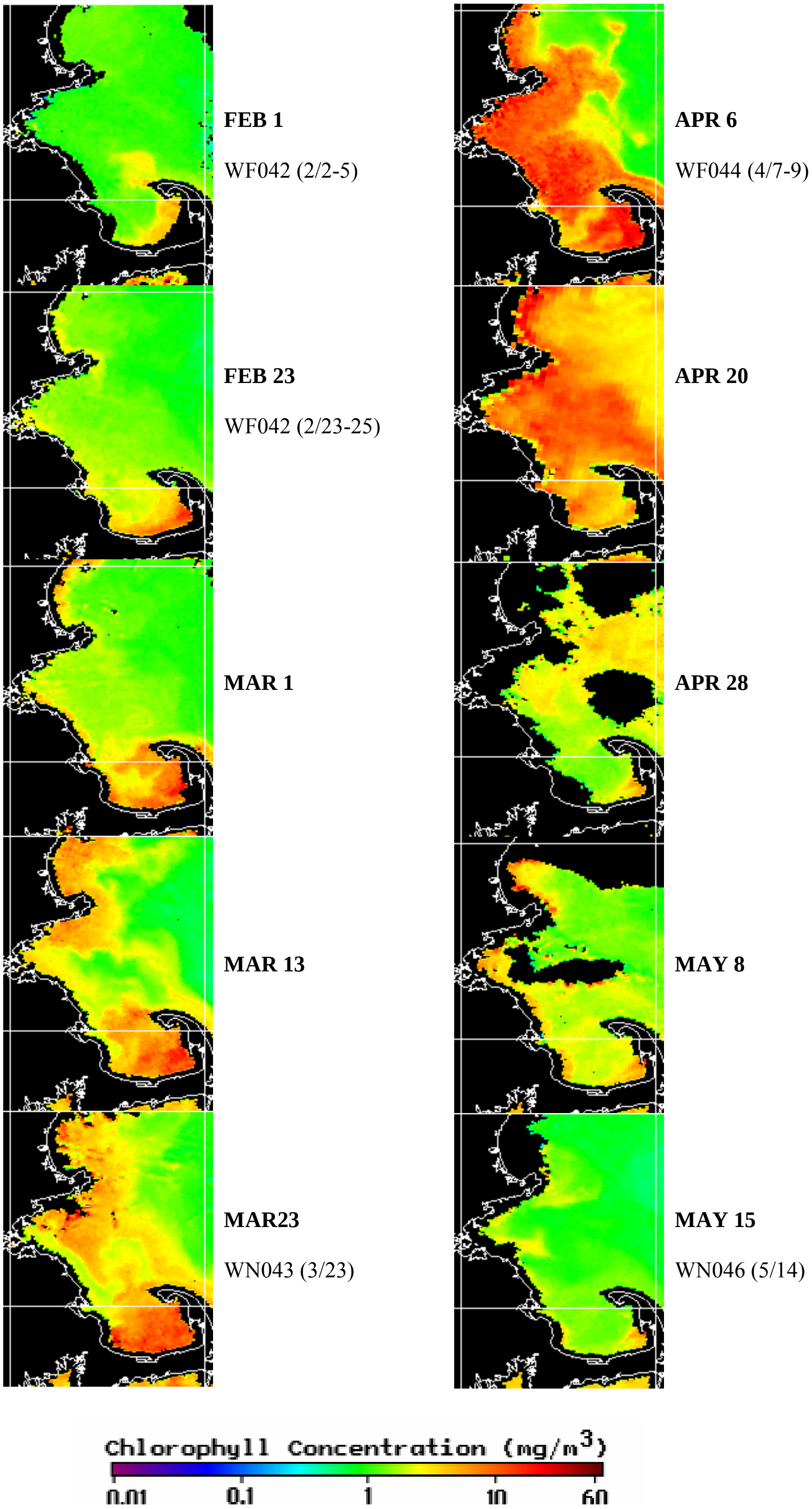


Figure 2-7. SeaWiFS images for southwestern Gulf of Maine for February to May 2004.

Post-transfer nearfield areal chlorophyll and POC concentrations were generally consistent with the baseline mean and seasonal patterns. The main departure from the baseline was in winter/spring with higher post-transfer chlorophyll values in early February (early winter/spring blooms in 2001 and 2002) and the consistent occurrence of March-April peaks in *Phaeocystis* (Libby *et al.* 2006a) The chlorophyll and POC data from Boston Harbor suggest that the harbor may be changing from a baseline pattern of biomass levels peaking in summer to a more typical temperate coastal water trend dominated by the winter/spring bloom (Libby *et al* 2006b). Although a comparison of seasonal and annual mean areal chlorophyll in the nearfield shows that there has been an apparent increase in winter/spring and annual mean levels since the bay outfall began discharging (**Figure** 2-8), none of the changes in seasonal or annual mean nearfield chlorophyll levels is significant.

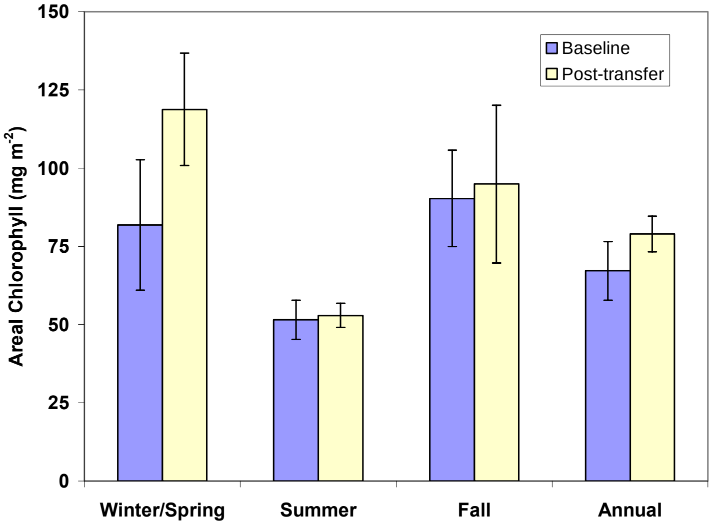
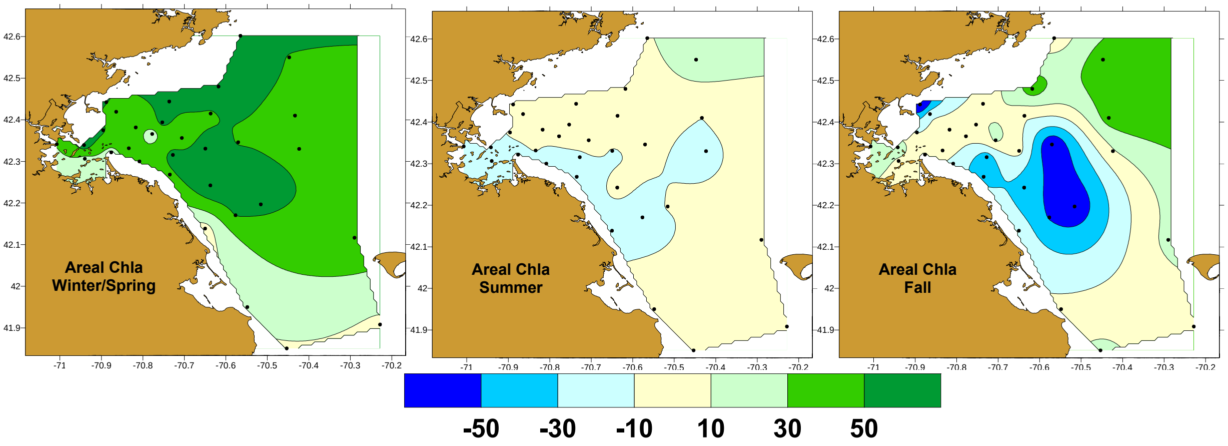


Figure 2-8. Comparison of baseline and post-transfer seasonal and annual mean areal chlorophyll (mg m^{-2}) in the nearfield through 2005. Error bars represent ±1 standard error. Winter/spring and summer means for 2000 included in baseline, 2000 fall mean in post-transfer, and 2000 annual mean not included.

In general, the winter/spring post-transfer period has been characterized by winter diatom (February) and early spring *Phaeocystis* (March-April) blooms of varying intensities. These blooms have been regional in extent and thus the winter/spring increase shown in **Figure** 2-8 may be due to a natural cycle in blooms rather than any localized change. On a per station basis, baseline to post-transfer differences in areal fluorescence were significant (based on t-tests with results of $p \leq 0.05$; Libby *et al.* 2006b) for only a limited set of stations in Boston Harbor (F23), coastal (F13 and F24), offshore (F06, F10 and F16), and boundary (F26 and F28) areas that had increased during the winter/spring and at harbor station F30 that had a significant decrease in summer concentrations (**Figure** 2-9a). Taken as a whole and corrected for the multiple comparisons, as discussed previously, none of these changes were significant. However, the fact that the lower p values were calculated for the changes at stations from different areas within the bay (harbor, coastal, offshore and northern boundary) suggests that the winter/spring increase is a regional phenomenon. The spatial trend in summer decreases in areal fluorescence is interesting in that the decreases are confined to Boston Harbor and stations along the south shore and southern Massachusetts Bay. This pattern, taken in context with the modeling and monitoring results (see **Figure** 2-3), suggests that the transfer to the bay outfall and subpycnocline capping of the effluent under stratified conditions effectively removes this source of nutrients from the euphotic zone in these waters.

(a) Areal Fluorescence Change



(b) POC Change

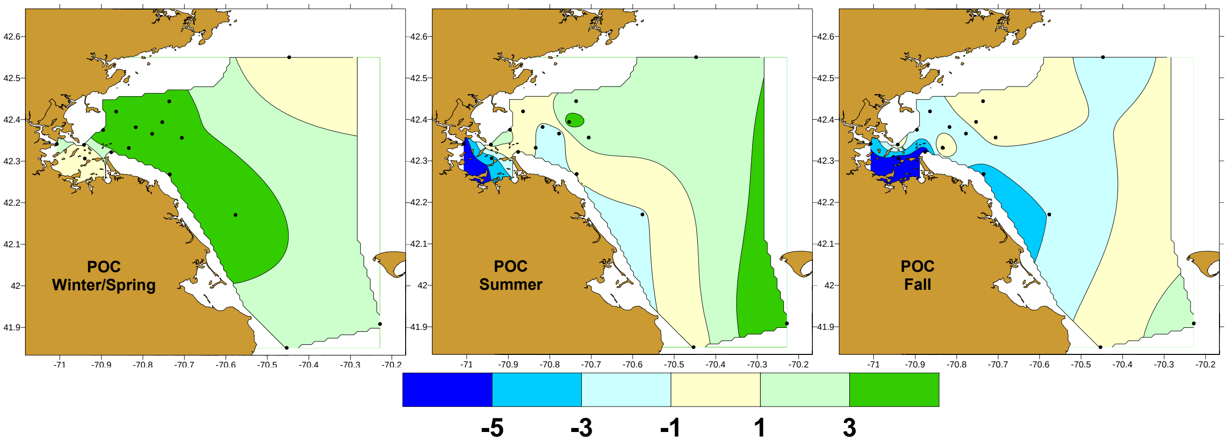


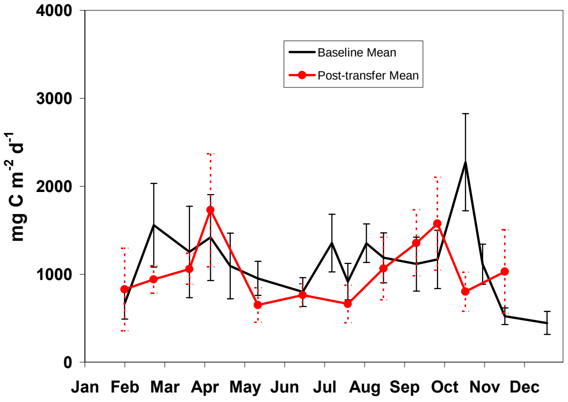
Figure 2-9. Change in seasonal (a) areal fluorescence (mg m^{-2}) and (b) POC concentrations (µM) between baseline and post-transfer periods. Based on the difference of means calculated over all depths from each station, survey, season, and period.

The winter/spring increase in areal chlorophyll fluorescence compared to baseline was coincident with increases in POC concentrations throughout most of Massachusetts and Cape Cod Bays (**Figure** 2-9b). The only Massachusetts Bay station that had a significant change (p=0.045) was station N20, which increased from a baseline mean of 20.6 µM to 29.2 µM post-transfer in the winter/spring. Nearly all of the Boston Harbor water quality monitoring stations had significant decreases in the summer and fall, although only two station/season instances (harbor station 140 fall and harbor station 141 summer) were significant after applying the multiple comparison correction. Summertime areal fluorescence and POC levels tended to decrease throughout western Massachusetts Bay and especially in Boston Harbor, while increasing further offshore (**Figure** 2-9). In the fall, the areal fluorescence change pattern was more complicated with slight increases in the harbor, nearfield, offshore, and Cape Cod Bay and decreases in coastal and southern Massachusetts Bay waters. POC concentrations, however, consistently show a decrease throughout much of Boston Harbor, coastal and boundary areas. There was a slight increase at stations N04, N10 and N16 in the nearfield and at stations F01 and F02 in Cape Cod Bay.

In the nearfield, graphical comparisons of survey, seasonal, and annual mean chlorophyll and POC values suggest that there has not been a substantial change since the diversion of effluent. Seasonal and annual mean chlorophyll concentrations in the nearfield have increased, but not significantly. However, winter/spring chlorophyll levels have increased significantly at individual stations throughout the region. The location of some of these stations in southern Massachusetts Bay is notable given the proclivity for transport of the effluent plume nutrients south of the nearfield during the winter/spring period, but increases in chlorophyll were also observed at the northern boundary stations due to the influence of the Gulf of Maine on the bays suggesting that these winter/spring changes are regional in nature. In Boston Harbor, there has been both a change in the seasonal chlorophyll and POC patterns and in the magnitude of the values. Since the transfer, the harbor has exhibited patterns in these parameters (and productivity) that are comparable to that observed in the nearfield and other temperate coastal waters. A clear relationship between changes in nutrients and chlorophyll levels in the nearfield, however, has not been observed over the first five years of post-transfer monitoring.

Productivity data from stations N04, N18 and F23 provides additional insight into the potential impact of additional nutrients in the nearfield and removal of the effluent nutrients discharged into Boston Harbor (**Figure** 2-10). Post-transfer areal production at the nearfield stations has continued to follow the pattern observed during the baseline, with the occurrence of spring and fall blooms and variable summer productivity (Libby *et al*. 2006b). Timing of the spring and fall blooms, however, is somewhat different from baseline years with the annual survey maximum occurring in April rather than October. The spring increase is due to the March/April *Phaeocystis* blooms, which have occurred every year since 2000. There have been no substantial fall blooms in the bays since fall 2000.

(a) Nearfield Areal Production



(b) Boston Areal Production

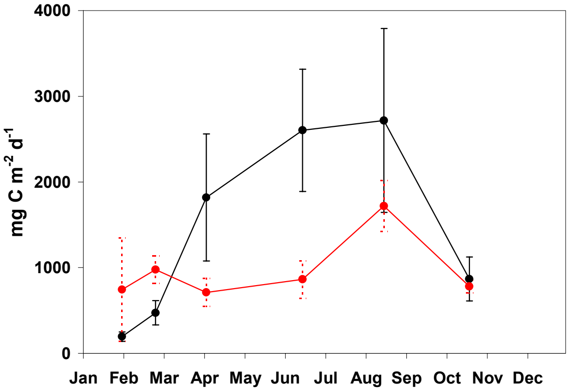


Figure 2-10. Time-series of baseline and post-transfer survey mean Areal Production (mg C m^{-2} d^{-1}) in the (a) nearfield and (b) Boston Harbor. Error bars represent ±1 SE. Data from all depths and stations.

The post-transfer productivity data at station F23 suggest that Boston Harbor may be transitioning from a eutrophic pattern to one more typical of Massachusetts Bay waters with a winter/spring peak, lower summer rates, and an intermittent late summer/fall peak (**Figure** 2-10). This is a change from the productivity pattern characteristic of the harbor while the sewage discharge was in the harbor which showed a steady increase in productivity from February to a summer peak before declining in the fall. The presence of a winter/spring bloom in 2001-2003 strongly suggested that this transition was occurring, with the cause presumably being the reduction in nutrients following the diversion of the outfall (Libby *et al*. 2004). In 2004, however, no spring or fall bloom was evident at the harbor station and the annual pattern in production was similar, though much lower, to the baseline pattern. In 2005, again there was no winter/spring bloom, but a late summer bloom was observed. The lack of winter/spring productivity blooms in 2004 and 2005 suggests that that the harbor station is not rapidly shifting to the nearfield pattern, but the overall decline in productivity seen at the harbor station indicates a shift to a less-enriched environment (**Figure** 2-10 and **Table** 2-3; Libby *et al*. 2006b).

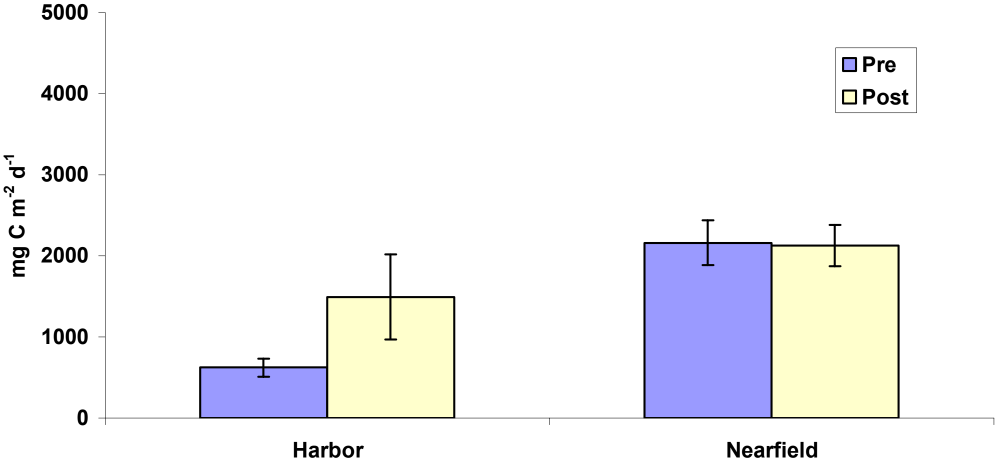
The greatest effect of the outfall discharge relocation on productivity is in seasonal productivity levels in Boston Harbor (**Figure** 2-11). In the winter/spring, the magnitude of the spring bloom peak productivity has nearly tripled from a mean of 623 mg C m^{-2} d^{-1} pre-relocation to 1491 mg C m^{-2} d^{-1} post-relocation (**Figure** 2-11; p=0.079). Summer harbor production decreased sharply, with a post-diversion peak of 1094 mg C m^{-2} d^{-1} compared to a pre-diversion peak of 3754 mg C m^{-2} d^{-1}. The post-diversion harbor summer peak in production is 30% of the baseline summer peak and represents an ecologically meaningful and statistically significant change (p=0.003). Fall productivity values for the harbor may be following a similar pattern with high values pre-diversion (3221 mg C m^{-2} d^{-1}) and low values post-diversion (1797 mg C m^{-2} d^{-1}; p>0.05, NS). No changes in the observed seasonal peak rates of primary production were observed in the nearfield (**Figure** 2-11).

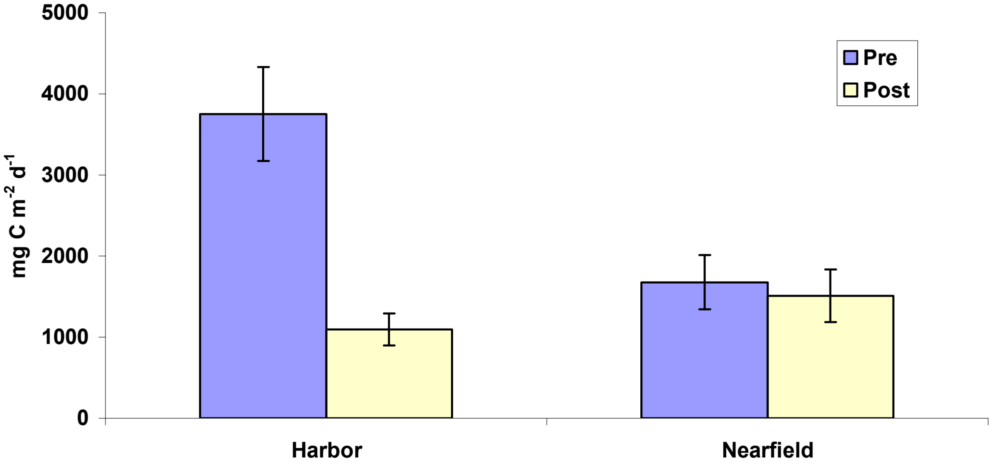
Interannual variability in annual mean production can be quite substantial (**Table** 2-3), but the Boston Harbor rates were consistently about 30 to 130% higher than nearfield rates over the baseline period (except for 1998 when all rates were very low). Since diversion to the bay outfall, the harbor and nearfield station rates have become comparable. The changes in nearfield station annual production (-0.2% and -19% at N04 and N16/N18, respectively) are not large nor are they significant.

Table 2-3. Annual mean production (gC m^{-2} y^{-1}).

Year	F23	N16-18	N04
1995	763	544	390
1996	1087	482	533
1997	862	612	480
1998	224	213	191
1999	658	503	395
2000	494	664	511
2001	404	559	569
2002	587	607	532
2003	311	293	295
2004	332	207	247
2005	251	244	343
Baseline Mean (1992-1999)*	719	471	398
Post-transfer Mean (2001-2005)*	377	382	397
Percent Change	-48%	-19%	-0.2%

*Bay Outfall began discharging September 6, 2000 – 2000 data not included for annual mean calculations.


(a) Spring Peak Areal Productivity


(b) Summer Peak Areal Productivity

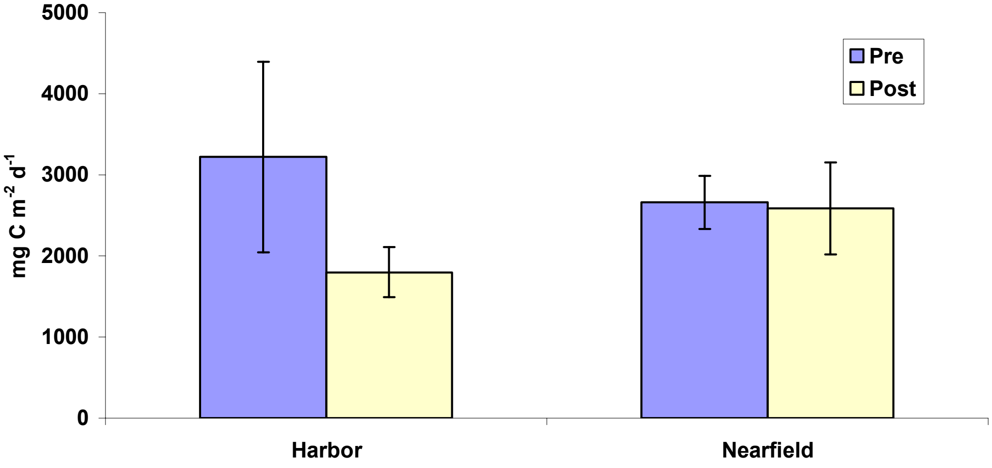

(c) Fall Peak Areal Productivity

Figure 2-11. Spring, summer, and fall bloom peak/maximum production (mgCm^{-2}d^{-1}) at nearfield (N04 and N16/N18) and Boston Harbor (F23) stations. Pre- vs. post- outfall diversion – spring and summer 1995-2000 vs. 2001-2005 and fall 1995-1999 vs. 2000-2005.

In Boston Harbor, however, there has been a nearly significant ($p=0.057$) reduction of 48% in annual production from baseline to post-diversion. In Boston Harbor, monitoring by MWRA shows decreases in annual mean chlorophyll (-26%) and POC (-28%); both are significant at $p \leq 0.05$ levels in the first five years after diversion to the bay outfall (Taylor 2006). All of these changes in production and biomass are coincident with significant decreases in NH_4 concentrations in the harbor (Taylor 2006). As discussed previously, there were significant increases in seasonal mean NH_4 concentrations at the inner nearfield stations (N16, N18 and N20). However, this increase has not had any apparent effect on primary productivity or phytoplankton biomass concentrations in the nearfield area.

The most notable changes since the outfall relocation are the large decreases in annual productivity within the harbor and the similarity in annual productivity at all three sites during the post-transfer period. The apparent changes in pre- and post-transfer production in Boston Harbor suggest that the removal of the effluent as a source of nutrients from the harbor is resulting in lower primary production rates and phytoplankton biomass concentrations (as chlorophyll and POC). In contrast, there has not been any definitive change in nearfield primary production as a result of the transfer. The expected level of change is addressed in Section 3 to better understand these observations.

Observed changes in the nutrient regimes following effluent diversion are unambiguous – DIN levels have dramatically decreased in Boston Harbor and nearby coastal waters (stations F14, F24 and F25) while increasing in the nearfield. In Boston Harbor, the dramatic decrease in DIN has been concomitant with significant decreases in chlorophyll and POC, lower production, and an apparent change in the seasonal productivity from a eutrophic pattern peaking in the summer to more normal temperate coastal pattern similar to the nearfield dominated by winter/spring peaks in productivity. In the nearfield, there are no clear indications that the higher DIN concentrations have translated into changes in productivity or phytoplankton biomass, whether measured as chlorophyll, POC, or phytoplankton abundance, although increases in winter/spring bloom biomass in the nearfield and subtle plankton community changes have been observed. Our current interpretation is that these changes are related to variations in regional patterns rather than a direct outfall effect.

3. A PRELIMINARY ASSESSMENT OF NITROGEN AND CARBON MASS BALANCES FOR THE BOSTON HARBOR AND MASSACHUSETTS BAY SYSTEM

On September 6, 2000, MWRA transferred effluent discharge from Boston Harbor to an outfall in Massachusetts Bay (see **Figure** 1-1). Many of the SEIS (Supplemental Environmental Impact Statement; EPA 1988) predictions and subsequent Bay Eutrophication Model projections (see Hunt *et al*. 1999) of this change have been validated by the long-term MWRA monitoring program (1992-2005; Section 2 and Libby *et al*. 2006a). The most obvious change was in nutrient concentrations, which decreased in the harbor and increased in the immediate vicinity of the offshore outfall. The pre-diversion assessments predicted little change in farfield nutrient regimes (except for Boston Harbor; Hunt *et al*. 1999), which have held true. The impact of the nutrients on primary production rates, phytoplankton and zooplankton biomass, abundance, and community assemblage was the subject of much speculation prior to the transfer. Concerns over potential eutrophication in Cape Cod and Massachusetts Bay drove much of the monitoring program and MWRA Contingency Plan.

The objective of this section is to address the following questions:

(1) How sensitive is the bay's ecosystem to changes in nutrient loads within the current mode of delivery?
(2) What long term, cumulative effects should be expected in Massachusetts Bay as a result of diversion to the bay outfall?

To address these questions this section briefly revisits pre-diversion reports (*e.g.*, previous Nutrient Issues reports, Kelly 1997, 1998; Hunt *et al.* 1999) and water quality model predictions, reviews relevant literature, and draws on mass balance analyses and evidence from the monitoring data for examples on how to determine bounds to system response to post-transfer conditions. For the latter, mass balances are evaluated and compared between pre- and post-diversion conditions and empirical models based on comparative systems approaches are considered in determining theoretical bounds to system responses in primary productivity. Additional work exploring the mass balances of N and C are undertaken to help elucidate their pre- and post-transfer nutrient budgets and to provide bounds on the magnitude of phytoplankton production and biomass change that might be expected due to the change in nutrient delivery to the system. This serves to provide an understanding of appropriate scales of measurement, and modeling, associated with elucidating the potential magnitude of the as-of-yet undetermined and likely subtle, changes that may occur in primary production and phytoplankton community structure.

The overview of system conditions in Section 2 provides a basis for understanding the impacts of MWRA nutrient loads and for assessing the various modeling approaches. Section 3.1 is a brief summary of previous loading and harbor export/mass balance estimates. This provides the conceptual background for examining post-transfer nutrient conditions and for estimating the potential impact of these inputs on primary production and biomass in the bays.

3.1 Summary of Historical Nutrient Loads to Boston Harbor and Massachusetts Bay

This section provides a brief review of efforts to assess and quantify nutrient budgets for Boston Harbor and Massachusetts Bay in pre- and post-transfer conditions. Three periods between 1991 and the present are associated with changes in the discharge of MWRA effluent to Boston Harbor and Massachusetts Bay (Alber and Chan 1994; Taylor 2005). The first period (Period A) extends from 1991 through 1998 (4/26/98) and represents a period of primary treated effluent discharges from two major wastewater treatment plants (WWTPs) to Boston Harbor: Deer Island (DI) to the North West Harbor and Nut Island (NI) to the mid-Central Harbor. Throughout this period, WWTP effluent accounted for almost 90% of the total nitrogen load to Boston Harbor. Period B (4/27/98-9/6/00) began in 1998 when an inter-island transfer was implemented that effectively combined the NI flows with the DI WWTP and resulted in one major effluent discharge location, the North West Harbor, and includes the phase-in of secondary treated effluent. Period B existed until September 2000 when secondary treated effluent was transferred from DI to an offshore diffuser located 15 km east of Boston Harbor. Period C (9/7/00-present) is characterized by the ending of direct discharges of WWTP effluent to Boston Harbor and is the current condition of effluent discharge.

Pre-Transfer Conditions

A series of reports and papers examined nutrient related issues associated with MWRA effluent prior to the transfer of effluent from Boston Harbor to its current offshore location in Massachusetts Bay. These included assessments on hydrology, nutrient loads, nutrient cycling and export, and models that supported predictions of future conditions.

Kelly (1997) conducted a study to estimate net annual exchanges of total nitrogen (TN) from Boston Harbor to Massachusetts Bay. Three fundamental questions were addressed in his paper: (1) Where does the input [of N] go? (2) Are there nitrogen sinks within the harbor? and (3) What is the relationship between harbor nitrogen cycling and the bay, which receives harbor outflow? To answer these questions, Kelly analyzed loading rates (Alber and Chan 1994) and water quality data collected within the study area between 1991 and 1994. Benthic nutrient cycling and denitrification rates were also incorporated in this analysis to yield an annual nitrogen budget for the harbor. Total nitrogen and dissolved inorganic nitrogen (DIN) input to Boston Harbor for this period were estimated at 8,470 mmol m^{-2} y^{-1} and 5,500 mmol m^{-2} y^{-1}, respectively (Alber and Chan 1994). More recent analyses conducted by Taylor (2005) report lower loading rates of TN and DIN to Boston Harbor for the same time period: 6,219 and 4,018 mmol m^{-2} y^{-1}, respectively[1].

Kelly used measurements of water column productivity for Boston Harbor in 1994 to estimate the amount of DIN assimilated by phytoplankton. Mean productivity (266 g C m^{-2} y^{-1}) accounted for approximately 60% of the total annual DIN load and 40% of the annual TN load. By contrasting water column TN concentrations with estimated water column exchange rates, Kelly suggested *in situ* production was not likely a consistent, significant nitrogen removal mechanism. This is because observed DIN concentrations were often too low to be attributed to removal by productivity alone. Rather, Kelly suggested, intuitively, that water residence times were the primary driver behind the observed DIN concentrations in Boston Harbor. He based this conclusion on comparison of volumetric loading of DIN and TN to Boston Harbor to observed water column concentrations, and to calculate the percent removal that would be necessary to attain these concentrations. First-order calculations resulted in required turnover times of 4.9 and 3.4 days for TN and DIN, respectively. Kelly then asked: What processes can remove, or assimilate, water column nitrogen within a 3 to 5 day average turnover time? Production and deposition were analyzed but the water residence time required for these processes to account for the observed water column nitrogen concentrations was 12 days; too long to be a consistent removal mechanism. Therefore, Kelly proposed that, based on existing estimates of tidal exchange between Boston Harbor and Massachusetts Bay, flushing accounts for a significant rate of removal of Boston Harbor nitrogen. In an effort to illustrate the effect of tidal exchange with Massachusetts Bay waters, Kelly included Boston Harbor mean annual nitrogen concentrations in a group of estuarine systems versus a gradient of loading rates, normalized by water residence time (per Nixon 1983). Thus, a “tentative conclusion” reached by Kelly is that tidal flushing is a major influence on Boston Harbor’s nutrient status. Kelly then established a simple annual nitrogen budget for Boston Harbor based on existing load

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[1] These values are based on reported loading rates of 1,834 and 1,185 kmol d^{-1} of TN and DIN, respectively. The conversion to areal loading rates are based on the mean area of Boston Harbor, 107,637,500 m^2, reported by Menzie *et al.* 1991.

estimates, burial and denitrification rates reported by Nowicki *et al*. (1997), and estimated tidal exchange rates (**Figure** 3-1). In this budget, about 2%, 8%, and 88% of total nitrogen loaded to the harbor is buried, denitrified, and exported via tidal exchange with Massachusetts Bay, respectively. An additional important source of nitrogen (both TN and DIN) missing from this annual budget is that from Massachusetts Bay (and offshore areas). Kelly identified this as a potentially important component of his budget analysis, but data on this source term was not readily available.

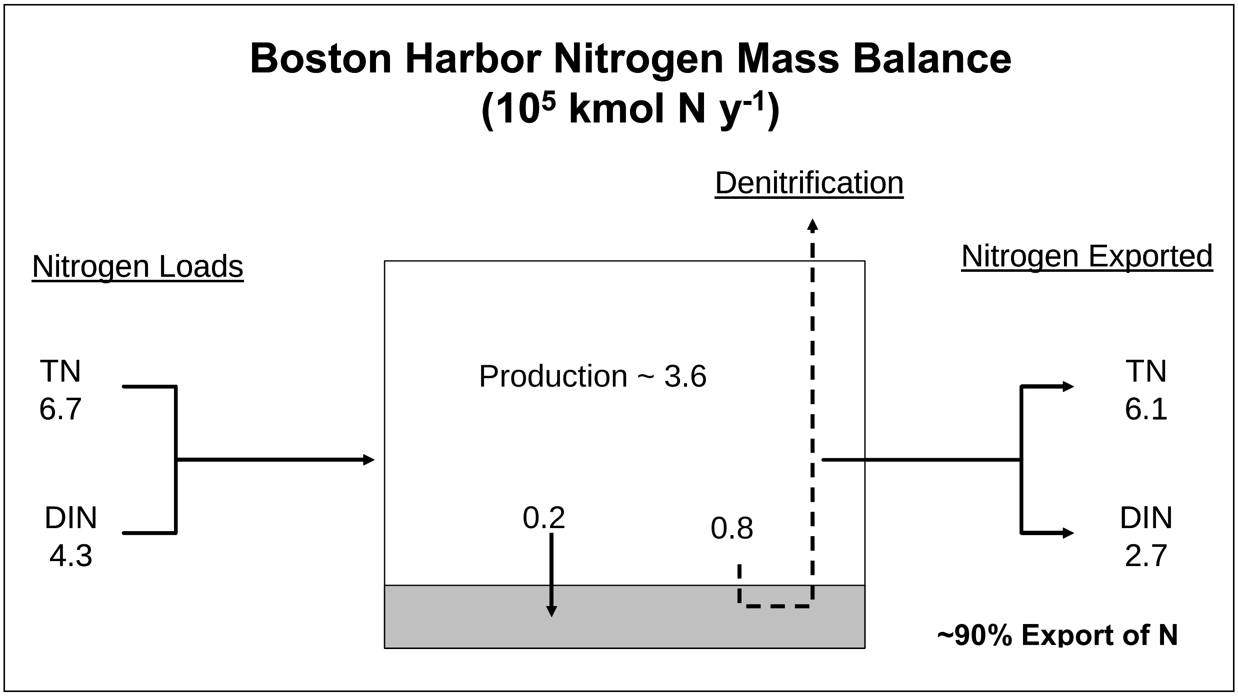

Figure 3-1. Annual nitrogen budget for Boston Harbor based on Kelly (1997) with revisions to loads from Taylor (2005). Converted to kmol N y^{-1} using harbor area from Menzie *et al.* (1991).

Kelly (1998) later proposed a revised Boston Harbor nitrogen (plus phosphorus and silicate) budget based, as previously, on terrestrial (including MWRA effluent) sources, but also including nutrient exchange between the harbor and offshore waters through tidal forcing. This "reanalysis" is focused on elucidating the influence of offshore water nitrogen imports due to the relatively large net volumetric exchange of Boston Harbor water with Massachusetts Bay water over each tidal cycle. This previously missing offshore term is very important in establishing a nitrogen mass balance for the Massachusetts Bays region because it clearly provides a significant mass input and export term.

To illustrate the offshore component in the mass balance, Kelly (1998) utilized water quality data from field surveys collected in 1994 to calculate net volumetric exchanges between Boston Harbor and Massachusetts Bay. A box-model framework was applied and net annual flux rates from two areas of Boston Harbor (north and south) to Massachusetts Bay were calculated from survey data that covered representative horizontal profiles (i.e., east-west through north and south harbor entrances). This box-model, mass-balance method was determined to be appropriate due to the measurable differences in salinity signatures between the Boston Harbor and Massachusetts Bay water columns. It should be noted that this approach would not have been successful, or appropriate, had a statistically significant salinity gradient not existed between the two water bodies.

Kelly used the volumetric exchange estimates to calculate net annual fluxes of nitrogen and other constituents between Boston Harbor and Massachusetts Bay (Kelly 1998). **Figure** 3-2 shows this annual budget which represents Period A. This budget is significantly different from his 1997 estimate (**Figure** 3-1). The primary difference is that the net export of nitrogen is greater than the sum of all inputs, even after removing the Massachusetts Bay input. This is somewhat problematic and Kelly (1998) admits that this imbalance exists and suggests that this could perhaps be due to rapidly exported benthic fluxes of DIN. But since his focus was on net fluxes at the harbor boundary (using real data), we feel that uncertainties associated with nitrogen inputs, burial, and denitrification might be the cause of his discrepancy. Thus, on an annual basis, there is approximately a 10% net import of TN to Boston Harbor that is not accounted for in Kelly 1998. Approximately 4% of total annual TN input is lost to burial and denitrification. If annual offshore inputs of TN are removed from the calculation then this value increases to about 13%. These small percentages strongly suggest that the vast majority of nitrogen loads to Boston Harbor are passed through the harbor/bay boundary to offshore waters and this is consistent with the preliminary calculations of net export of TN by Kelly (1997) based solely on physical exchange rates over spring and neap tidal periods.

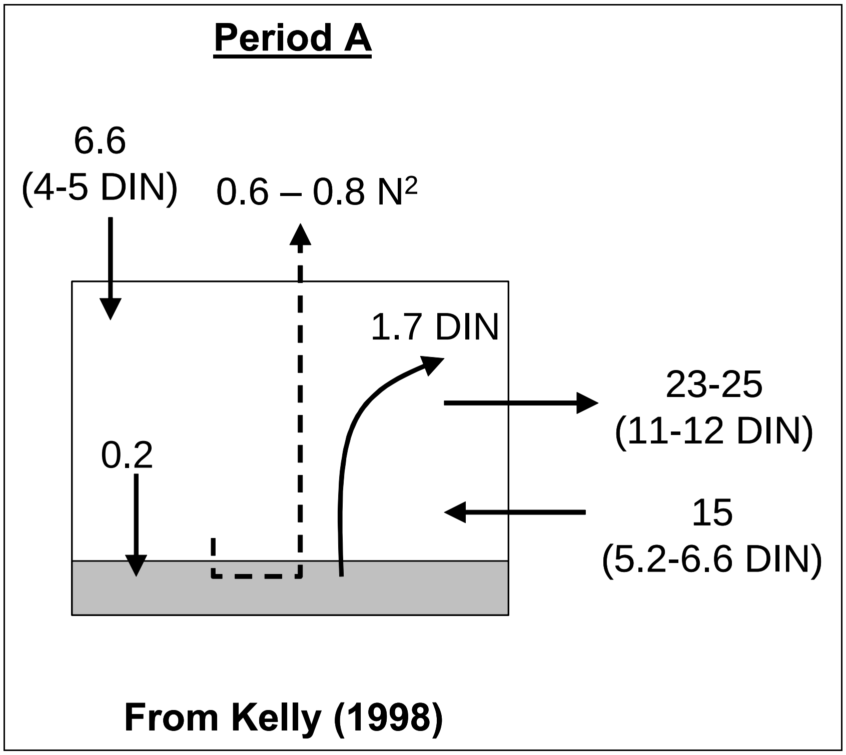

Figure 3-2. Annual total nitrogen budget for Boston Harbor calculated by Kelly (1998). Units are 10^5 kmols N y^{-1}.

Both Kelly papers report analyses of Boston Harbor exchanges of water and water column constituents that provide nitrogen (and other nutrient) budgets for Boston Harbor for pre-transfer conditions. Net annual loading values, physics, burial and denitrification rates, and field survey data were collectively analyzed and applied to the budget calculations. We agree with the conclusions reached by Kelly (1997, 1998) including the confirmation that pre-transfer nitrogen is predominantly exported from the harbor to the receiving waters of Massachusetts Bay; that ocean loading accounts for the highest loading term (mass) to Boston Harbor in both pre- and post-transfer conditions; and that the box-model, mass-balance method for determining net fluxes across system boundaries was appropriate given the availability of salinity and nutrient data.

Post-Transfer Conditions

In September 2000, the Deer Island Treatment Plant effluent discharge was transferred from the harbor to Massachusetts Bay at an offshore location about 15 km east of the harbor. This transfer rerouted about 82% of the total nitrogen load that previously existed in Boston Harbor (Taylor 2005). Kelly (1998) predicted partial annual budgets for nitrogen and other water column constituents in the post-transfer period based on nitrogen loading rates that were estimated at that time (subsequently revised by Taylor 2005). A comparison of the pre- and post-transfer budgets is shown in **Figure** 3-3. In essence, the post-transfer loading rate to the harbor is significantly reduced (88%) and although a slight reduction in the offshore loading term is predicted (Signell *et al.* 1996), it remains the primary source of TN to the harbor. Rates of burial, denitrification, and remineralization in the sediments are predicted to be less than those estimated for the pre-transfer period because they are a function of water column productivity which is governed, in part, by nutrient availability.

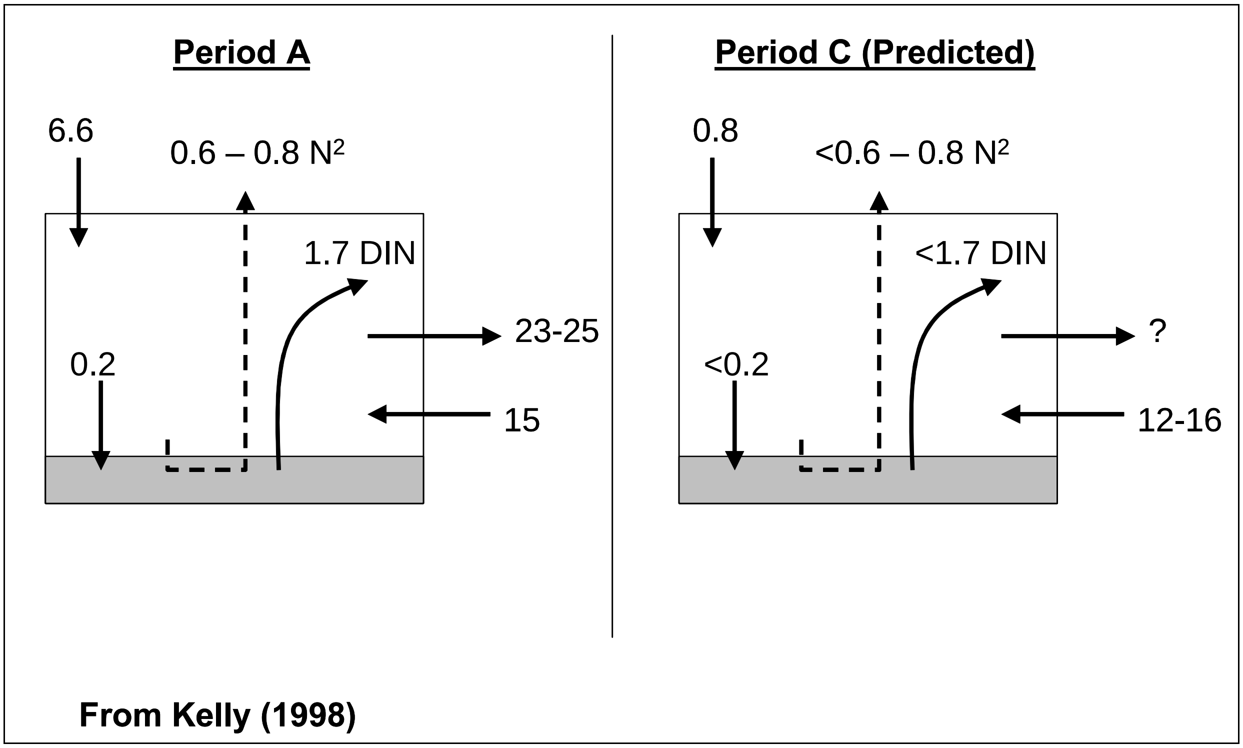

Figure 3-3. Annual total nitrogen budget for Boston Harbor for pre- and post-transfer periods (Kelly 1998). Units are 10^5 kmols N y^{-1}.

Kelly's predicted post-transfer budget does not indicate an export term from the harbor presumably due to the uncertainties present when this analysis was undertaken. However, the main point expressed by Kelly is that overall magnitude of ocean loading to the harbor is higher than that due to land derived sources under Period A conditions and this will obviously continue to be the case with the offshore outfall.

An analysis published by Hunt *et al.* (1999) focused on predicting whether post-transfer nitrogen inputs to Massachusetts Bay would result in measurable differences in water chemistry (i.e., nutrient concentrations) and ecosystem attributes (e.g., phytoplankton productivity and standing stock). They analyzed several modeling scenarios using the Bays Eutrophication Model (BEM), including sensitivity analyses, and concluded that measurable changes in water constituents would be limited to nearfield locations:

- Nitrogen entering at the boundaries of Massachusetts Bay exerts more influence on the total nitrogen concentrations in the farfield areas than the effluent discharge does;
- BEM and 3-D hydrodynamic model results demonstrate that nutrient concentrations above the background variability would be confined to a small area near the outfall;
- Elevated nutrient levels in the coastal region (from Boston Harbor southward towards Plymouth) would be unchanged or slightly lower with transfer of the effluent discharge location to Massachusetts Bay;
- BEM model results predict little change in spatial or temporal patterns of nutrient concentrations in Cape Cod Bay relative to the current and future effluent discharge locations;
- 3-D hydrodynamic model computations estimate the area in Massachusetts and Cape Cod Bays that would be under measurable influence from the discharge is small (only 7 km^2 which is <0.2 percent of the combined area of Massachusetts and Cape Cod Bays);
- 3-D hydrodynamic model computations predict that the effluent nutrient concentrations would be diluted to 200:1 within a few kilometers of the outfall diffuser, and thus indistinguishable from background;
- Change in the nutrient fields in Massachusetts Bay would be highly localized and have little to no impact on the phytoplankton and zooplankton species distributions and communities in the bay;
- Nutrient levels in Massachusetts and Cape Cod Bays would not be enriched to levels that promote the growth of nuisance species such as the "red tide" organism *Alexandrium*; and,
- BEM computations project small increases in the DO in bottom waters of the nearfield in the summer.

Based on these determinations, Hunt *et al.* (1999) concluded that adverse changes to the ecology and functioning of the Massachusetts Bay system would not occur as a result of the outfall relocation. Changes in phytoplankton community assemblages, rates of productivity, and standing stock would be significantly more influenced by regional, Gulf of Maine processes than the introduction of treated effluent from the Deer Island facility.

Therefore, based on the studies highlighted above, we suggest that the anticipated net effect of post-transfer nutrient delivery to Massachusetts Bay is minimal, if even measurable, except periodically in localized areas in the vicinity of the outfall, whereas measurable improvements would be found in Boston Harbor.

3.2 Discussion

This discussion focused on the two questions posed at the beginning of this section, and those predictions in Hunt *et al.* (1999). There is considerable interest in knowing the extent of impact the discharge of MWRA effluent has had on Massachusetts and Cape Cod Bays since the outfall came on-line. Each of these questions is addressed below to provide insight on what the likely effects are on system productivity and metabolism in Massachusetts Bay due to transfer of the nutrient discharge to the bay outfall.

(1) How sensitive is the bay's ecosystem to changes in nutrient loads within the current mode of delivery?

This question addresses the concern associated with the cascading effects of nutrient-driven increases in organic carbon production, which conceivably could affect system metabolism, benthic habitat, and primary producer species assemblages. The degree to which coastal ecosystems are sensitive to changes in nutrient delivery is dependent upon scale and variability. That is, do the nutrients added to the Massachusetts Bays system by the outfall represent an ecologically significant perturbation?

Hunt *et al.* (1999) focused on the sensitivity of Massachusetts Bay to post-transfer delivery of effluent. They did this in a variety of ways, including simulating the change in nitrogen concentrations by assuming nitrogen as a conservative constituent (biological processes turned off) and analyzing water quality changes by doubling the projected nitrogen loads in the fully developed BEM (biology turned on). The results were as follows:

- Approximately 93 percent of the nitrogen entering the Massachusetts Bays system is associated with influx from the Gulf of Maine, only 3 percent attributable to the MWRA outfall;
- When nitrogen was treated as a conservative tracer (i.e. biological processes turned of in the model), TN budgets in the vicinity of the nearfield (station N21) were composed of only 30 percent of the MWRA source and consistently <10 percent in fluxes to Cape Cod Bay;
- When simulating transfer to the new outfall location (assuming baseline loading rates), there is a predicted decrease (compared to the old harbor location) in chlorophyll-a in Boston Harbor and little or no impact on the levels of chlorophyll-a in Massachusetts Bay or Cape Cod Bay; also TN concentration profiles would not change significantly in Massachusetts Bay or Cape Cod Bay, but a slight increase might be measured in the vicinity of the outfall;
- When simulating transfer to the new outfall location and doubling the anticipated loading rate of nutrients and organic carbon to the new outfall location would not result in significant changes in TN concentrations in Cape Cod Bay, but would result in slight increases in TN in Boston Harbor and along the shore region of NW Massachusetts Bay as far south as Humarock (+0.5 mg/L);
- Doubling MWRA nutrient loads would result in an increase in DIN concentrations along the same shore region of NW Massachusetts Bay;
- DO levels are not significantly different in Massachusetts Bay bottom waters during post-transfer operations, nor when the loading rate of nutrients and organic carbon is doubled, except in the immediate vicinity of the outfall location where a slight decrease would be expected; and,
- The vast majority of N influx to the system is across the boundary with the Gulf of Maine, even compared to a hypothetical doubling of effluent nutrients

The sensitivity of the Massachusetts Bay system to nitrogen input was estimated by translating nutrient flux estimates into equivalent carbon production. This enabled examination of the relative magnitude of influence of nutrient additions to productivity in Massachusetts Bay by assuming nitrogen assimilation was 100% efficient; also physical influences were not considered (e.g., water residence times, stratification). It is thus possible in a box model scenario to determine the highest theoretical limit of productivity that the MWRA effluent load can support by assuming that 100% of the TN is available as DIN as it is transported from one region to the next (**Figures** 3-4 and 3-5). This exercise provides an estimate of the upper limits of influence that effluent-derived nitrogen may have on system metabolism within the areas depicted by the three model boxes. This is done by estimating net inputs and exports of TN across the boundaries of three boxes consistent with post-transfer conditions. Estimates of TN flux across various areas of Massachusetts Bay were made by applying the approach published by Dettmann (2001).

Dettmann (2001) developed a simple model that predicts net TN export from estuarine systems based primarily on freshwater water residence time (τ_{fw}) and a first-order rate constant (α) which was derived by optimizing the rate constant within a set of eleven estuaries. Two forms of the model are applicable to estuarine systems: (1) assumes that the load from offshore (ocean) sources (N_s) is significantly less than the load from terrestrial and atmospheric sources (N_l); and (2) the incorporation of the ratio, N_l:N_s within the equation in cases when N_s>N_l.

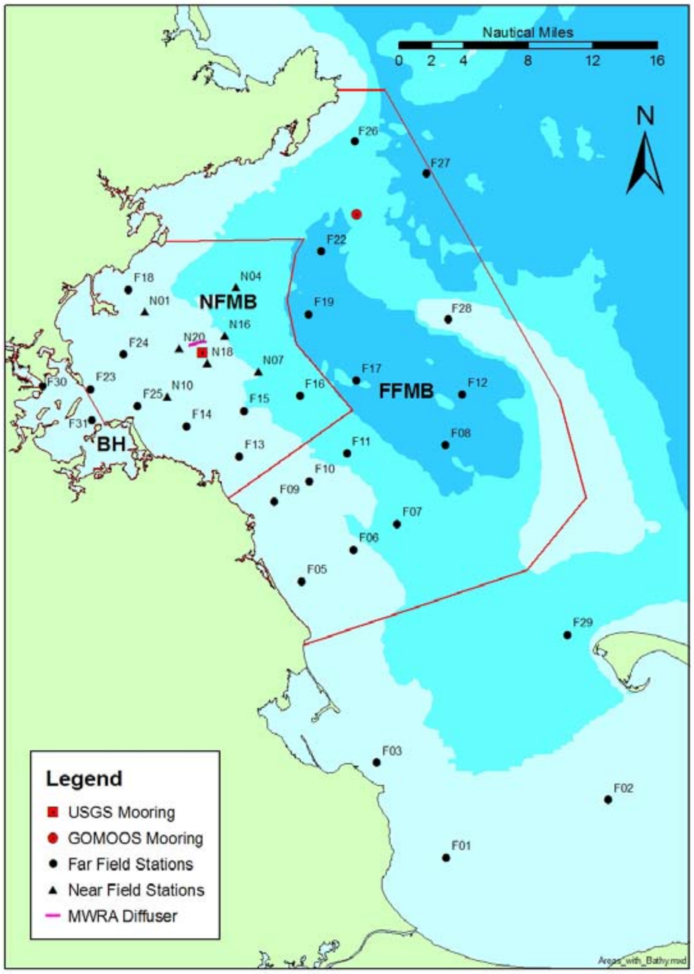

Figure 3-4. Simple box-model cell configuration showing the arrangement of Boston Harbor (BH), nearfield Massachusetts Bay (NFMB) and farfield Massachusetts Bay (FFMB) boxes.

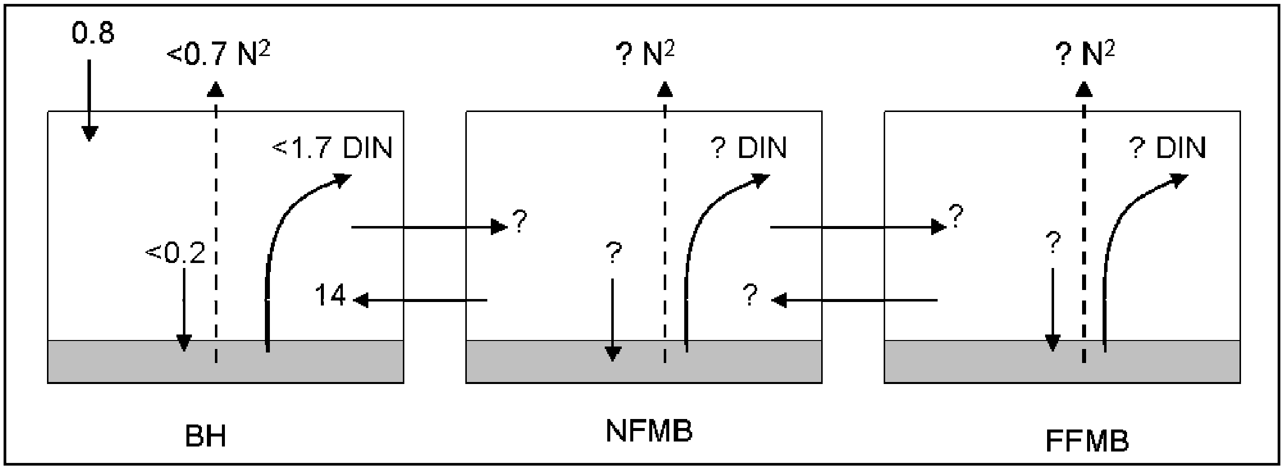

Figure 3-5. Conceptual model of post-transfer extended simple box-model associated with nutrient cycling and export through Boston Harbor (BH), nearfield Massachusetts Bay (NFMB) and farfield Massachusetts Bay (FFMB) boxes. Values (10^5 kmol) are total nitrogen unless noted.

The Dettman model provides an independent method to assess net TN export from systems like Boston Harbor. In fact, Dettmann (2001) includes Boston Harbor (pre-transfer conditions) in a series of model calibration exercises and provides an annual nitrogen budget for Boston Harbor and ten other estuarine systems. He reports that of his estimated total nitrogen inputs to the Boston Harbor system (7,891 mmol m^{-2} y^{-1}) approximately 87% (6,871 mmol m^{-2} y^{-1}) was exported to Massachusetts Bay on an annual basis during pre-transfer conditions. The set of eleven estuarine system annual TN budgets was then used to calibrate and corroborate his net export model which is reported as Equation (9) in Dettmann (2001).

$$F_{E(l)} = 1/(1+\alpha\tau_{fw})$$

Where $F_{E(l)}$ is the fraction of net annual export of nitrogen (land derived). This equation assumes that nitrogen from upland sources is much greater than that which fluxes across the seaward boundary (i.e., $N_l >> N_s$; where N_l is upland and N_s is seaward sources). Dettmann (2001) optimized the performance of this model by adjusting the value of α and concluded that the value of 0.3 ($month^{-1}$) results in the best fit for data associated with the 11 estuarine systems. Based on this model, where $N_l >> N_s$, the predicted net export of nitrogen from Boston Harbor is 91% of total inputs (N_l). This assumes a freshwater residence time of 0.33 months. This export value is close to the 88% that was calculated by Kelly (1997). But Kelly (1998) reported that oceanic nitrogen input to Boston Harbor is significant and that this must be considered in terms of annual budgets. If this is the case, then the assumption that $N_l >> N_s$ cannot be applied to this analysis. Rather, a separate derivation of the model must be considered which is reported as Equation (8) in Dettmann (2001).

$$F_{E(l)} = 1/([1+\alpha\tau_{fw}][1+N_s/N_l])$$

If this equation is applied to the Boston Harbor data, the predicted net annual export of nitrogen is 76% assuming that the ratio of $N_s{:}N_l$ is approximately 15 (based on Kelly 1998). Dettmann (2001) reported a site-specific value of 0.44 for α in Boston Harbor. If this value is applied in both $N_l >> N_s$ and $N_s \geq N_l$ versions of the model then values for the net annual export of nitrogen are 87% and 68%, respectively.

Therefore, based on Kelly (1997, 1998) and Dettmann (2001) the annual net export of land derived TN from Boston Harbor to Massachusetts Bay for pre-transfer conditions ranges from 68 to 91 percent. Kelly (1998) provided a predicted annual budget for post-transfer conditions and this also can be analyzed by the Dettman (2001) model.

Several estimates or assumptions are required to apply the Dettmann (2001) model to post-transfer conditions among the three simple box-model cells (**Figure** 3-5). First, TN load to each box must be estimated. Based on information reviewed above, post-transfer loads of TN are estimated for Boston Harbor and for MWRA effluent which is discharged from a diffuser located on the seafloor in NFMB. Although Kelly (1998) provided estimates of terrestrial and wastewater loads to Boston Harbor in post-transfer periods (i.e., Period C), we used more recent estimates of these loads reported by Taylor (2005). Taylor (2005) estimated that $1.23 \pm 0.74 \times 10^5$ kmols TN y^{-1} is received by Boston Harbor in post-transfer conditions (note: based on 36 months post-transfer data). The post-transfer load of MWRA effluent to the NFMB cell is estimated to be approximately 8.0×10^5 kmols TN y^{-1}, based on 2004 effluent monitoring data (MWRA pers. comm.). Second, water residence times are required for this analysis. The water residence time for BH is assumed to have increased by about 10% (Kelly 1998) to 0.36 months. Bulk, integrated water residence time estimates for Massachusetts Bay were reported by Geyer *et al.* (1992) as 20 to 40 days (0.7 to 1.3 months). For this exercise we assume that the water residence times are 0.8 and 1.3 months for NFMB and FFMB, respectively. We also assume a fully mixed water column and that there is a net transport from BH to NFMB to FFMB, consistent with the estimated water residence times such that the Dettmann (2001) model estimates sequential transport across cells.

One problem for this analysis is that net volumetric exchange between FFMB and NFMB is required to estimate a value for Ns. It is most certainly the case that $N_s \geq N_l$ due to the relatively large volumes, despite the added nitrogen input from MWRA diffusers in NFMB. This was demonstrated by Kelly (1998) and Hunt *et al.* (1999). Background concentrations of TN could be used, but an estimate of net annual import of TN across the boundary from FFMB to NFMB is necessary, but not available at this time. However, modeling analyses reported by Hunt *et al.* (1999) do provide estimates of boundary fluxes of TN and DIN from the GOM to Massachusetts Bay, and *vice versa*, but not for the specific spatial boundary scales of this analysis. To overcome this problem, we assume 100% export of TN from NFMB to FFMB to compare the highest theoretical influence of this introduced TN to the FFMB region, although the proportion of export across this boundary is likely to be substantially lower.

Under these assumptions, the Dettmann model suggests that approximately 43% of TN is exported from BH to NFMB. This is consistent with his previous results and accounts for about 0.5 x 10^5 kmols y^{-1}. This mass is combined with the post-transfer diffuser contribution of 8.0 x 10^5 kmols TN y^{-1} to make the total TN load to NFMB about 8.5 x 10^5 kmols y^{-1}. This equates to an areal load of about 1.3 mol TN m^{-2} y^{-1} and a volumetric load of 0.04 mol TN m^{-3} y^{-1}. If 100% of this load is assumed to transfer to the FFMB cell, this 8.5 x 10^5 kmol translates to an areal load of 0.5 mol TN m^{-2} y^{-1} and a volumetric load of 0.01 mol TN m^{-3} y^{-1} in FFMB.

Estimates of areal productivity corresponding to 100% assimilation of available land-derived nitrogen and a C:N molar Redfield ratio of phytoplankton biomass of 6.625 cell are shown in **Table** 3-1 for each simple box-model. Based on the uncertainties associated with the modeling and conservative approaches used, these values of both pre- and post-transfer productivity only represent upper limits. During period B, ~90% of MWRA's TN load was DIN. If 90% of the 6.69 x 10^5 kmols TN loaded to BH every year during pre-transfer was assimilated, then productivity would reach about 445 g C m^{-2} y^{-1}, not counting the vast net import of nitrogen entering along the seaward boundary as calculated by Kelly (1998).

Table 3-1. Comparison between pre- and post-transfer loads and productivity associated with these loads only (offshore inputs not included).

Cell Name	Pre-Transfer Annual TN Load (x 10^5 kmols y^{-1})	Pre-Transfer Productivity (g C m^{-2} y^{-1})	Post-Transfer Annual Load (x 10^5 kmols y^{-1})	Post-Transfer Productivity (g C m^{-2} y^{-1})	Pre vs. Post Change in Productivity (g C m^{-2} y^{-1})	Pre vs. Post Percent Change (%)
BH	6.69	494.12	1.23	90.85	-403.27	-82
NFMB	4.95	61.71	8.53	106.34	+44.63	+72
FFMB	4.00	19.56	6.88	33.64	+14.08	+72

The measured post-transfer production decreased dramatically at harbor station F23 from mean of 719±143 (SE) to 377±58 g C m^{-2} y^{-1} (-48%) for 2001-2005 (see **Table** 2-3). This decrease is of similar magnitude to that predicted due to the offshore transfer of the effluent discharge (~400 g C m^{-2} y^{-1}; **Table** 3-1) and further illustrates the importance of offshore nutrient inputs to production in Boston Harbor. Productivity measurements at stations N04 and N18 in the nearfield area suggest that there has been little change in annual production from pre to post-transfer. At station N04, mean annual production was constant with pre- and post-transfer values of 398±58 and 397±65 g C m^{-2} y^{-1} (-0.2%) and at station N18 (which is ~2 km south of the outfall) production decreased from a baseline mean of 471±68 g C m^{-2} y^{-1} to a post-transfer mean of 382±84 g C m^{-2} y^{-1} (-19%; see **Table** 2-3). This contrasts with the calculated change in **Table** 3-1 which estimates a 72% increase in productivity for NFMB. Note, however, that the magnitude of the change in Table 3-1 for NFMB is 44 g C m^{-2} y^{-1}, which is only is ~10% of the pre- and post-transfer nearfield production. The percent change in nearfield productivity associated with the transfer of the effluent to the offshore outfall, whether based on station observations (-0.2 and -19%) or this model (+10%), is low and well within the interannual variability that has been observed at the nearfield stations (Libby *et al.* 2006b).

These observed values can be further illustrated by placing them in context with a cross-system comparative regression developed by Nixon (1992) where a strong relationship between annual areal DIN input and primary production was discovered:

$$\text{Log PP} = 0.442 \log \text{DIN} + 2.332$$
$$r^2 = 0.93$$

where PP is primary production (g C m^{-2} y^{-1}) and DIN is annual input (mol m^{-2} y^{-1}).

Based on this regression, the average measured annual production vs. total DIN input to Boston Harbor for the pre- and post-transfer periods is plotted on the regression line derived by Nixon (1992) in **Figure** 3-6. This empirical, comparative approach is useful for demonstrating system response (in this case, primary productivity) to changes in total nutrient loads. The total DIN input values include direct atmospheric deposition (0.074 mol m^{-2} y^{-1}; Nixon 1997), watershed inputs as reported by Taylor (2005), and ocean boundary fluxes as reported by Kelly (1998). The resulting plot indicates that the relationship between the measured productivity and estimated loads to Boston Harbor fit remarkably well with Nixon's regression. This regression equation can also be reversed to estimate the approximate total annual flux of DIN to Massachusetts Bay (e.g., NFMB) necessary to result in the observed rates of production. For this calculation the mean of nearfield productivity summarized above (for stations N04 and N18) was used (389 g C m^{-2} y^{-1}). The resulting required DIN input to support the observed rate of production is 3.85 mol DIN m^{-2} y^{-1}.

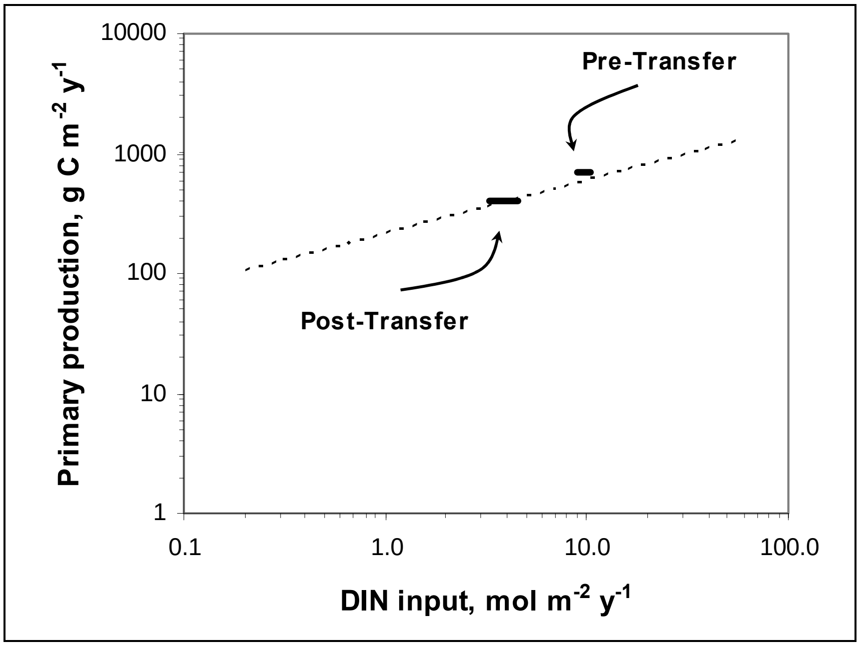

Figure 3-6. Annual high and low estimates of *total* DIN loads vs. measured primary production for Boston Harbor for pre-transfer and post-transfer periods. The dashed line indicates the functional regression line derived by Nixon (1992) in a multiple-system comparison (log PP = 0.442 log DIN + 2.332; r^2 = 0.93). Horizontal bars represent range of values associated with DIN loads.

The box model and cross-system comparisons discussed above, and the review of monitoring data corroborate the conclusions reported by Hunt *et al.* (1999): that there would be no significant changes in nutrient concentrations or primary productivity in Massachusetts Bay in post-transfer conditions. Other than localized elevations of NH_4 in and around the outfall location, there has not been a clear indication that the ecosystem has responded measurably to the relocation of effluent discharge. This is certainly due to the rates of dilution and vast volumetric exchange that occurs in the bay, which was a primary driver for proposing this relocation. Although the box and regression response models are quite general and span a large spatial scale, they help illustrate the relative insensitivity of the Massachusetts Bay ecosystem to post-transfer effluent delivery.

(2) What long term, cumulative effects should be expected in Massachusetts Bay as a result of diversion to the bay outfall ?

This question is addressed by considering the general factors that control metabolic rates within the water column and the benthos. The relatively insignificant anticipated and observed influence of the post-transfer nutrient delivery to Massachusetts Bay can be further put in to context with post-assimilatory processes. The conceptual model is that any additional organic carbon production will be subject to respiratory demands within the water column and, if labile carbon is eventually delivered to the benthos, additional oxygen demand can occur there.

There have been several comparative assessments of the relationship between organic matter supply and sediment (or benthic) oxygen consumption (Nixon 1981, 1982; Kemp *et al*. 1992, Boynton *et al.* 1996, Heip 1995, Borsuk *et al*. 2001a). Nixon (1981, 1982) illustrated this relationship quite well by comparing the sum of primary production and other organic inputs (g C m^{-2} y^{-1}) with benthic remineralization (g C m^{-2} y^{-1}) among a variety of coastal systems around the world. A highly significant relationship was discovered:

$$\text{Benthic Remineralization} = 15 + 0.238(\text{Primary Production} + \text{Organic Input})$$
$$r^2 = 0.94$$

Benthic remineralization can be equated to sediment oxygen demand (SOD) and therefore, an approximate relationship between organic carbon loading (primary production + organic input) and SOD emerges. Further study of this cross-system approach has led to the refinement of this relationship. Kemp *et al*. (1992) analyzed several factors that affect DO production and consumption in 18 estuarine data sets and found very strong evidence for depth-dependence:

$$\text{SOC} = 3.29 - 1.74\ \text{Log}(z)$$
$$r^2 = 0.99$$

Where SOC is sediment oxygen consumption (g O_2 m^{-1} d^{-1}) and z is depth (meters). Further analysis yielded a good relationship between the proportion of benthic areal respiration in the same data set.

$$\text{SOC as \% Total Respiration} = 55.5 - 27.8\ \text{Log}(z)$$
$$r^2 = 0.78$$

Kemp *et al*. (1992) hypothesize that SOC, equivalent to SOD, dominates community respiration in water columns (or bottom layers) of 5 meters or less. Both of these relationships are depicted graphically in **Figure** 3-7.

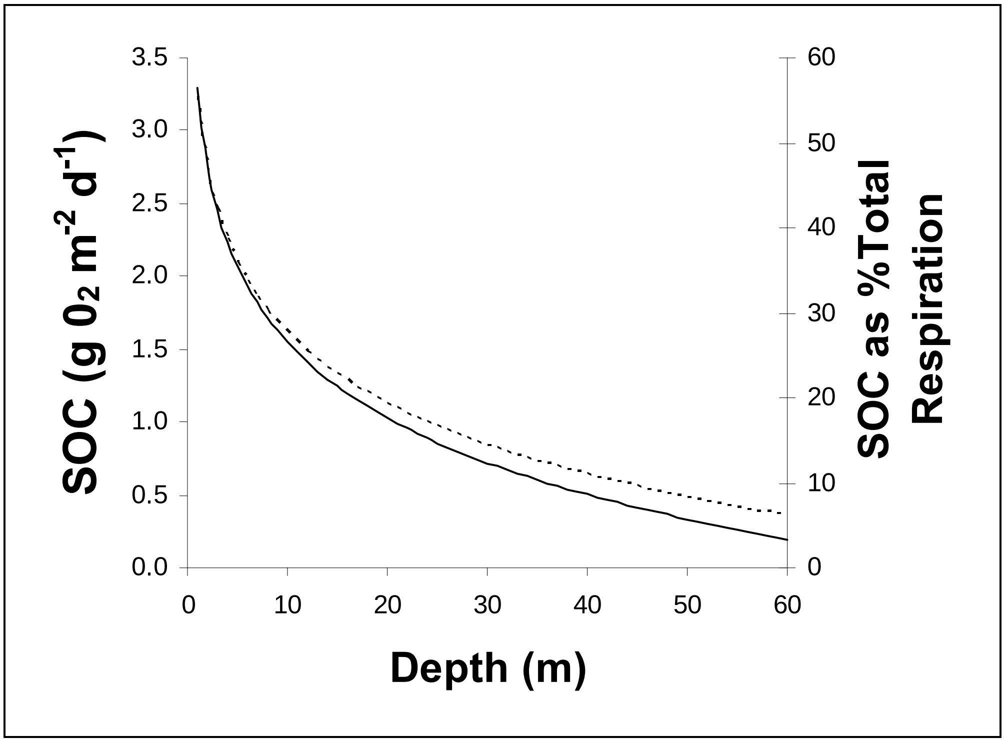

Figure 3-7. Relationship between depth (m) and sediment oxygen consumption (SOC, solid line) and SOC as a percent of total m^{-2} respiration (dashed line).

Borsuk *et al.* (2001a) recently published a revised comparative relationship, built upon the earlier work of Nixon (1981, 1982), Kemp *et al.* (1992) and others. This adaptation considers organic carbon load (L_c), depth (h) and three system-specific parameters (a, b, and k):

$$SOD = \alpha(L_c/[1+kL_ch])^b \quad (4)$$

Where, SOD is mol O_2 m^{-2} y^{-1}, L_c is mol C m^{-2} y^{-1} and h is water depth in meters. L_c is the sum of net primary production and external inputs. The parameter distributions of a, b and k are derived using a Markov Chain Monte Carlo method for analyzing the 34 system-specific parameter sets (including those analyzed by Nixon 1982) within this specific Bayesian hierarchical approach. Borsuk *et al.* (2001b) further describes methods to estimate bottom water DO concentrations and probabilities associated with exceeding established management standards.

These comparative studies indicate not only strong correlations between depth and benthic respiration rates, but that in relatively shallow systems, like areas of Boston Harbor, these rates are expected to be proportionally significant. In other words, benthic respiration has a greater influence in shallow coastal systems than in deep systems. For instance, when Kemp *et al.*'s relationships are applied to Boston Harbor (mean depth of about 5 m), the predicted SOC is 2.1 g O_2 m^{-1} d^{-1} and comprises 36.1% of total areal respiration. The model suggests that the remaining respiration (about 64%) occurs within the water column. However, in the deeper waters of Massachusetts Bay the proportion of water column respiration of surface water production is significantly greater. For instance, of the roughly 400 g C m^{-2} y^{-1} produced in the nearfield area of Massachusetts Bay (depth of about 36 m), the Kemp *et al.* model would predict that approximately 78% of the primary production is respired within the water column prior to it being available to the benthos. When the offshore regions are included in the estimate, the mean depths increase further and the supply of labile organic matter delivery to the benthos continues to diminish. In addition the volume increases, which also further attenuates the significance of surface water productivity to depth-averaged metabolism. A systematic, progressive increase in annual primary production Massachusetts Bay is also not expected to occur unless major changes in the nitrogen sources or other factors emerge.

Cumulative impacts in Cape Cod Bay and Massachusetts Bay as a result of the post-transfer discharge scheme are not supported. This is because the relative magnitudes of changes in nutrient loads and productivity, as discussed in Question (1) do not signify any likely changes beyond what has already been measured. The observed changes in surface water productivity are well within the envelope of system variability and that these slight differences would only be realized within a small subregion of Massachusetts Bay (perhaps ~10 km radius). Furthermore, any potential cumulative effect would need to be identified and measured beyond interannual and seasonal variability which has been shown to be periodically significant (Libby *et al.* 2006b). Moreover, marginal, localized increases in productivity are quickly lost within the noise of system variability and in a practical manner; the volume of Massachusetts Bay and environs is appreciably large in contrast to minor changes in carbon fixation.

3.3 Conclusions and Recommendations

Previous work on estimating fluxes of nutrients throughout the Massachusetts Bay system in pre- and post-transfer periods strongly suggests that the net effect of the transfer of effluent to the offshore outfall has resulted in significantly lower nutrient loads and rates of primary production within the water column of Boston Harbor and an insignificant contribution to change in Massachusetts Bay. This is primarily due to the differences in water column volumes and residence times between these two systems. In addition, the monitoring within Boston Harbor, the nearfield, and farfield areas of the bay has shown that dramatic changes (decreases) in system productivity and metabolism have occurred in Boston Harbor but very little change in system productivity has been observed in the immediate vicinity of the new discharge location despite an elevated NH_4 signal in the same area. Moreover, dissolved oxygen levels in nearfield and farfield regions have not changed, nor has the standing stock of phytoplankton. Conceptually, and intuitively, the relatively low system response in the NFMB and FFMB regions can be primarily explained by dilution and rapid mixing and transport due to the system's vast volumes relative to the discharge volume and physical dynamics. Most of the nitrogen coming from wastewater sources has always reached Massachusetts Bay, but in past and present scenarios, has only contributed a small part of the available pool of nitrogen. Furthermore, current subpycnocline discharge of effluent further prevents assimilation by primary producers and thus helps mitigate any potential eutrophic response. The depths are such that benthic-pelagic coupling is not likely to be significant unless there were large, measurable increases in productivity (Kemp *et al.* 1992).

Therefore, it is unlikely that post-transfer effluent discharge to Massachusetts Bay has or will significantly change system metabolism from baseline conditions in the bay. However, continued data collection on system productivity and other information will support future seasonal, annual, and interannual nutrient and production mass-balances and may identify any future nearfield anomalies in productivity. Additional contributions to resolving the impact and change can be realized with more refined calculations such as those available in BEM. Short- and long-term net exchanges associated with the mass-balance estimates of nutrients (and carbon) would be furthered by the application of the BEM which is currently being fostered and developed by UMB. The physics of the system ultimately determine the degree that nutrients are available for assimilation into organic matter over specific temporal and spatial scales. It has been demonstrated that the physics in the Massachusetts Bay system are highly variable on short time scales and even at longer time scales. The further coupling of modeling with monitoring data will help elucidate the questions raised in this section.

4. VARIABILITY IN THE FREQUENCY, MAGNITUDE AND DURATION OF *PHAEOCYSTIS* BLOOMS

Blooms of the marine prymnesiophyte phytoplankton genus *Phaeocystis* are a frequent component of the winter-spring phytoplankton of north-temperate coastal seas (Cadee and Hegeman 2002; Schoemann *et al*. 2005), including the Gulf of Maine (Bigelow, 1924). Within the Gulf of Maine, Massachusetts and Cape Cod Bays in particular have long been known for their 'rich *Phaeocystis* – diatom blooms' during April (**Figure** 4-1). The blooms are usually regional and have been observed at locations throughout the Gulf of Maine. The timing, duration and magnitude of these blooms are likely controlled by many factors. The ability of *Phaeocystis* to form high biomass blooms (>10 mg C L^{-1}; up to 200 x 10^{6} cells L^{-1}; Schoemann *et al*. 2005) that sequester a large portion of ecosystem resources and alter trophic pathways (Tang *et al*. 2001) make *Phaeocystis* one of the few 'keystone' phytoplankton taxa whose blooms significantly alter ecosystem function (Lancelot *et al.* 1994; Verity and Smetacek 1996) via modification of bio-geo-chemical cycling (Smith *et al*. 1991; Stefels *et al*. 1995) and alteration of food web structure (Rousseau *et al*. 2000). In addition to altered ecosystem function, mucus from gelatinous *Phaeocystis* colonies may have inimical effects such as fouling fishing gear (net clogging; Savage 1930) and fish mortality (Huang *et al*. 1999). In some regions, *Phaeocystis* bloom duration and bloom magnitude are sensitive to both long-term climate warming trends (Cadee and Hegeman 1986; 2002) and anthropogenic nutrient influences (Riegman *et al*. 1992; Cadee and Hegeman 2002; Tungaraza *et al*. 2003).

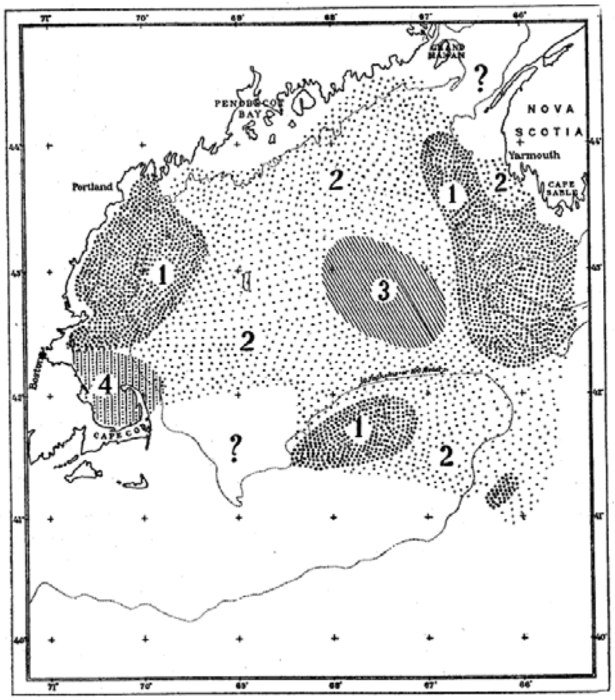

Figure 4-1. Characterization and distribution of general phytoplankton types across the Gulf of Maine in April 1920 taken from Bigelow 1924. Regions denote (1) rich diatom, (2) sparse diatom, (3) sparse *Ceratium* and diatom, and (4) rich *Phaeocystis* and diatom.

Phaeocystis blooms have been observed in Massachusetts Bay in the winter-spring period in nine of 14 years of MWRA monitoring (1992, 1994, 1997, 2000, 2001, 2002, 2003, 2004, and 2005). During this time, the blooms have only been observed during the months of March-May with lower abundances sporadically seen in months preceding and following the March-May bloom period. The Massachusetts Bay blooms are dominated by *Phaeocystis pouchetii*, which blooms during the winter/spring period in temperate seas. Summer blooms (July-August) of *Phaeocystis globosa* have been recorded in the Arabian Sea (Madhupratap *et al*. 2000). *P. globosa* blooms have not been observed during the MWRA monitoring program and all references to *Phaeocystis* in the bays in this report are to *P. pouchetii*.

Blooms of greater than one million cells per liter were confirmed in the nearfield during six of the 14 years (1992, 1997, 2000, 2003, 2004, and 2005). Note that three of the four largest bloom years (1992, 1997, 2000) occurred prior to relocation to the offshore outfall. Moreover, the pattern of occurrence and duration of the *Phaeocystis* blooms in Massachusetts Bay appears to be changing from two- to three-year intervals to blooms occurring annually. Speculatively, the apparent change in the pattern of occurrence of *Phaeocystis* blooms might be related to the outfall, however, the evidence is that it is not. Why

Phaeocystis occurs in relatively high abundance in some years and not in others is not well understood and continues to be the focus of research around the world. Factors contributing to the initiation of *Phaeocystis* blooms have been an area of active scientific research for decades with a focus on availability of light, nutrients, temperature, water movement, competition from other algal species for nutrients, and grazing. The silicate-*Phaeocystis* hypothesis postulates that diatoms outgrow *Phaeocystis* until silicate becomes limiting (Lancelot *et al*. 1987). Other work suggests that silicate depletion is not a necessary condition for a bloom because *Phaeocystis* can out-compete diatoms at higher irradiance or under eutrophic/highly nutrient-rich conditions (Peperzak *et al*. 1998). It has also been noted that colony formation by *Phaeocystis* tends to decrease vulnerability to grazing, at least temporarily, and may lead to changes in the food web structure.

This section focuses on the spatial extent and timing of the *Phaeocystis* blooms observed from 1992 to 2005 and provides context regarding factors that contribute to the occurrence, magnitude, and duration of the blooms. The evaluation uses MWRA monitoring data along with ancillary data sources and the literature to assess patterns of *Phaeocystis* blooms in the MWRA monitoring area of Massachusetts Bay to evaluate any changes in bloom timing or magnitude, and to identify factors that may be contributing to the observed bloom patterns.

4.1 Data Results

4.1.1 Spatial Patterns

Phaeocystis blooms in Massachusetts Bay are usually part of a large-scale regional feature, spanning large areas of the Gulf of Maine (Bigelow 1924) and sometimes south of Cape Cod including Buzzards Bay (Borkman 1994) and Narragansett Bay (Verity *et al*. 1988). However, in some years within the Gulf of Maine there may be local variability in the distribution and timing of *Phaeocystis* blooms, such as the inshore versus offshore bloom pattern differences observed by Bigelow (1924). Bigelow reported that for Massachusetts Bay the *Phaeocystis* blooms tended to occur after the winter/spring diatom bloom, while in offshore Gulf of Maine waters these blooms often preceded the annual spring diatom bloom. *Phaeocystis* often dominates the phytoplankton community, with *Phaeocystis* biomass often comprising >85% of total biomass during blooms (Davies *et al*. 1992). This feature allows the use of a combination of methods (microscopic identification of cells and remote sensing) to track the spatial and temporal extent of *Phaeocystis* blooms. For example, SeaWiFS images during the 2004 *Phaeocystis* bloom show nearly synchronous arrival of elevated phytoplankton chlorophyll across the entire region from Cape Ann south to Cape Cod Bay in late March 2004, followed by sustained elevated phytoplankton chlorophyll through late April (see **Figure** 2-7). These observations are consistent with the time course of *in situ* observations of *Phaeocystis* abundance at MWRA sampling stations. Also, anecdotal comparison of *Phaeocystis* observations from Massachusetts Bay stations often coincide with the appearance of *Phaeocystis* at other phytoplankton monitoring stations in the region such as in Buzzard's Bay (Turner pers. comm.) and other observations in Massachusetts Bay (off Provincetown by the Center for Coastal Studies[1]). This was also the case during 1992 and 1997 when monitoring off of Seabrook NH showed strong *Phaeocystis* blooms consistent with MWRA data (Normandeau 1997). *Phaeocystis* blooms were also observed in the Seabrook area in 1981 and 1983. Unfortunately, 1997 was the last year of phytoplankton monitoring for the Seabrook program. *Phaeocystis* blooms were observed in Narragansett Bay in 1996, 2000, and 2002-2005 (University of Rhode Island[2]) showing that there is not always agreement in the timing of *Phaeocystis'* appearance in Massachusetts Bay and in these more distant locations. Bigelow (1924) noted that even though these blooms tend to dominate the phytoplankton community when they occur *Phaeocystis* blooms are often short lived, which may account for poor agreement regionally some years.

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[1] Field observation data available at: www.coastalstudies.org/what-we-do/right-whales/ccb-field2006-March.htm

[2] Data available at: www.gso.uri.edu/phytoplankton/

Within the MWRA monitoring area, the pattern of regional abundance varies from year to year. Elevated *Phaeocystis* counts may occur in the nearfield or further offshore in any given bloom year. Fairly evenly distributed *Phaeocystis* abundance was observed in all regions of Massachusetts Bay in some years (1992), elevated abundance in the nearfield was observed in some years (2004), and elevated abundance to both the north (2001, 2003) and south of the nearfield (2002) has been observed (**Figure** 4-2). Overall, no consistent spatial pattern associated with the MWRA bay outfall is apparent in the monitoring data, with *Phaeocystis* abundance being neither consistently higher nor lower to the north ('upstream') or south ('downstream') of the outfall. A one-way ANOVA of regional *Phaeocystis* abundance (5 regions: nearfield, boundary, Cape Cod Bay, coastal, harbor) confirms this and shows *Phaeocystis* abundance was not statistically different in the five regions (p=0.75, n= 5 groups, F = 0.48).

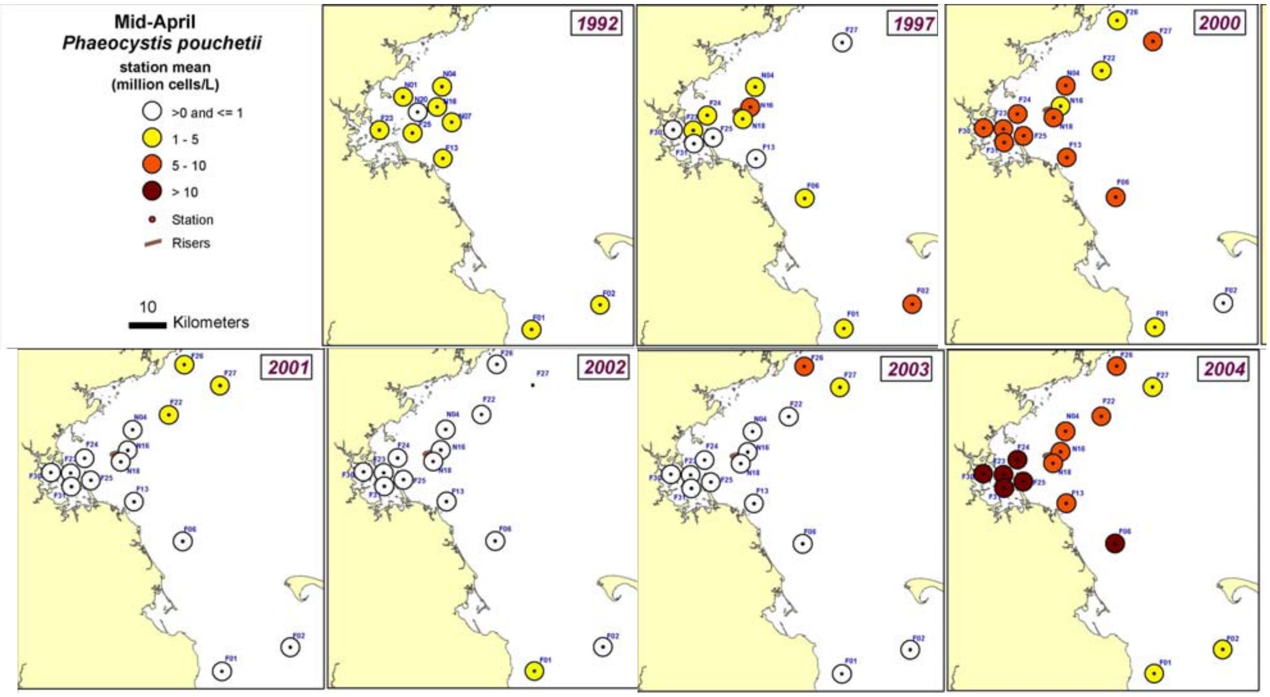


Figure 4-2. Spatial extent of the seven April blooms of *Phaeocystis*. Each point represents an MWRA plankton sampling station. The number and location of stations sampled for plankton has changed over the course of the program.

4.1.2 Temporal Trends

After recording spring *Phaeocystis* blooms in 1992, 1994 (farfield), and 1997, there have been blooms during consecutive years from 2000 to 2005 (**Figure** 4-3). Thus, the pattern has changed from spring *Phaeocystis* blooms occurring at ~3-year intervals to blooms occurring annually. Although it is clear that the periodicity of spring *Phaeocystis* blooms has changed, the reason(s) for this change remain elusive. Similarly, it is not clear why, unlike previous blooms which occurred primarily in late March and April, the blooms from 2002 through 2005 began earlier, and lasted until early May, thereby causing exceedances of the MWRA Contingency Plan "summer" *Phaeocystis* threshold. The *Phaeocystis* colonies observed in May are typically remnants of a senescent bloom (chlorophyll to phaeophytin ratio of 2:1 to 1:1 and colonies appeared to be senescent with 'empty' *Phaeocystis* cells, lower density of cells, and many fragmented/broken colonies), but the presence of even low abundances of this alga in May led to the 2002-2005 summer caution level threshold exceedances, since *Phaeocystis* was almost completely absent from samples collected after May 1st during baseline years.

Phaeocystis bloom initiation has been relatively consistent, with *Phaeocystis* typically first observed in February to early March followed by a bloom maximum in late March to early April. However, within the limits of detection imposed by the monitoring schedule, bloom duration appears to have increased significantly (linear regression, $r^2 = 0.49$, $n = 10$, $p = 0.0237$) at a linear rate of ca. seven days per year during 1992-2005 (**Figure** 4-4). This increased bloom duration was due to a delay in bloom termination which extended *Phaeocystis* presence into May or early June in 1992, 1997, 2002, 2003, 2004 and 2005. Most of the largest blooms have occurred in the latter years of monitoring, with the 2004 *Phaeocystis* bloom being the largest observed during the MWRA monitoring program with a nearfield survey maximum of 8 million cells l^{-1} in April 2004. The increased bloom duration was not accompanied by any statistically significant trend in bloom magnitude, with nearfield *Phaeocystis* maximum abundance showing no linear trend (linear regression $r^2 = 0.08$, $n = 10$, $p = 0.4272$) during 1992-2005 bloom years. Further, there was no correlation between *Phaeocystis* bloom duration and bloom magnitude (Pearson correlation $r = +0.0365$, $n = 10$, $p = 0.9201$).

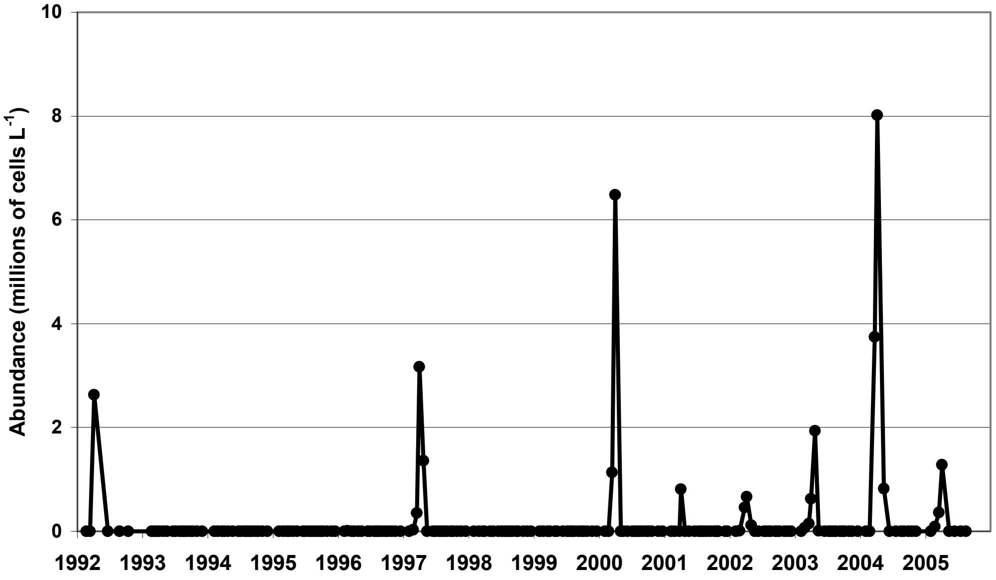


Figure 4-3. Time series of nearfield survey mean *Phaeocystis* abundance, 1992-2005.

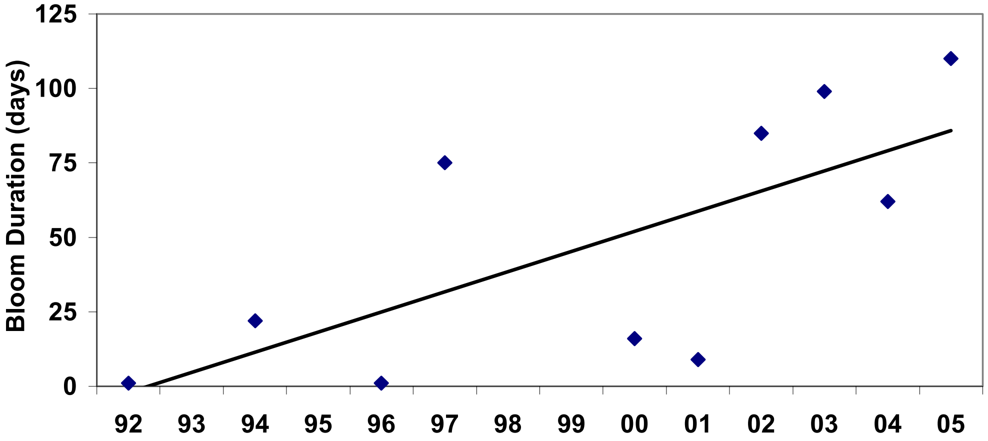


Figure 4-4. Increased *Phaeocystis* bloom duration in the nearfield 1992-2005. Linear trend (y = 6.76 X – 8.81) explains ca. 50% of the variance in the long-term trend of increasing bloom duration (r^2 =0.4926, n = 10, p = 0.0237).

4.2 Environmental Factors

4.2.1 Temperature Effect

Phaeocystis pouchetii has a thermal tolerance range of –2 to 14 °C (Jahnke and Baumann 1987) and a Massachusetts Bay *P. pouchetii* isolate has been shown not to grow in nutrient and light replete laboratory conditions at temperatures >14 °C (Hegarty and Villareal 1998). Thus, 14 °C appears to be the physiological temperature limit for *P. pouchetii* growth, and is the maximum temperature at which one might expect to observe *P. pouchetii* blooms in Massachusetts Bay. Examination of the temperature record from the Boston Buoy (NOAA buoy 44013) suggests that the extended duration of the *Phaeocystis* blooms in 2002, 2003, 2004 and 2005 is related to the presence of cooler waters (<14 °C) into June when compared against temperatures in 2000 and 2001 when surface waters reached 14 °C in mid May and the duration of the *Phaeocystis* bloom was abbreviated (**Figure** 4-5). The date of first attainment of 14 °C at the Boston Buoy is surprisingly variable, ranging from as early as late April (day 120) in warm years (2001) to early June (day 160) in cold years (1992,1993,1994, 2003, 2004, 2005). Although the correlation between surface temperatures and bloom duration was significant (p=0.027), the data set was limited to only six points that fall into two groupings. Data from the 1992 bloom are not applicable as phytoplankton was only sampled every other month during the winter/spring. The 1997 bloom was not included since surface water temperature data from NOAA buoy 44013 only extend through May 11 for that year. Although daily temperature data are not available for late spring 1997, surface water temperatures were relatively cold in mid-May with measurements of 7.2 °C on May 11 at the NOAA buoy and ~7.5 °C during the May 13 nearfield survey. Moreover, surface temperatures had only warmed to 14.5-16 °C by the June 18 nearfield survey. These observations suggest that the extended 1997 *Phaeocystis* bloom may have been associated with the presence of cool waters into June. These observations strongly suggest temperature is a factor in the extended duration of *Phaeocystis* blooms in 1997 and 2002-2005.

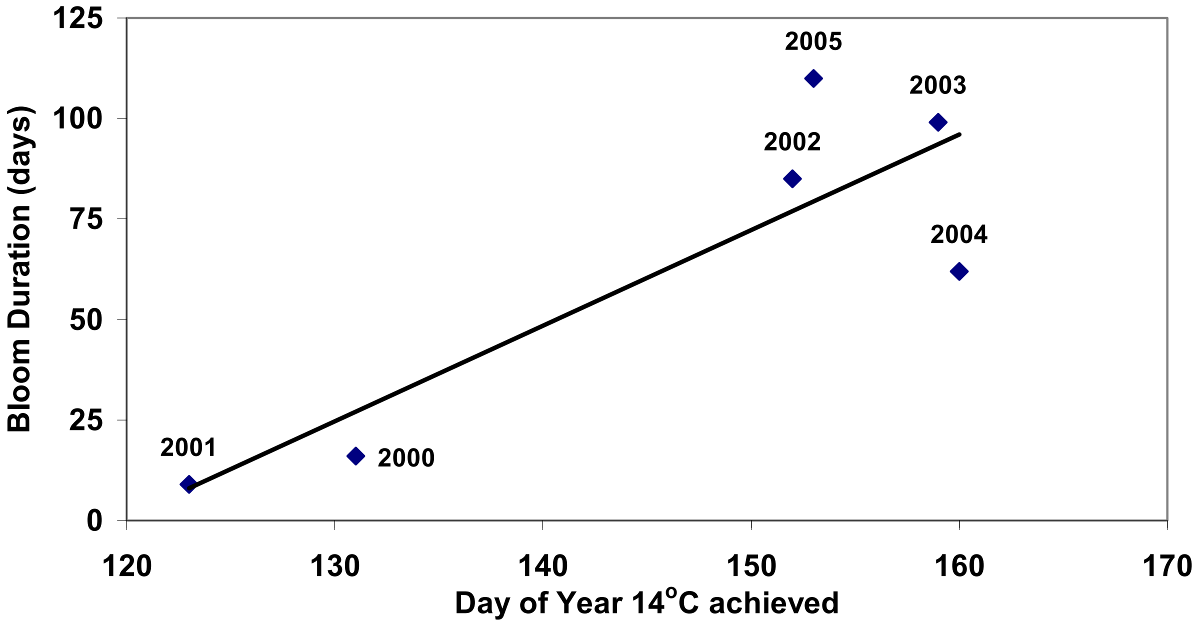


Figure 4-5. ***Phaeocystis*** **bloom duration related to first date of 14°C at the NOAA 44013 Buoy 2000-2005. Linear regression r^2 =0.75, n = 6, p = 0.0265.**

4.2.2 Nutrient Effects

Several hypotheses have been advanced suggesting nutrients as contributing factors for *Phaeocystis* bloom initiation and maintenance. The leading hypotheses include the silica- *Phaeocystis* limitation hypothesis (Lancelot *et al*. 1987; Reid *et al*. 1990), the eutrophication hypothesis which alters N:P ratios (Cadee and Hegeman 1991; Philippart *et al.* 2000) and the oxidized – reduced N (NO_3/NH_4) hypothesis (Tungaraza *et al.* 2003). The silicate-*Phaeocystis* hypothesis postulates that diatoms outgrow *Phaeocystis* until silicate becomes limiting then *Phaeocystis* can out-compete the diatoms (Lancelot *et al*. 1987; Reid *et al*. 1990). However, other work suggests that silicate depletion is not a necessary condition for a *Phaeocystis* bloom because *Phaeocystis* can out-compete diatoms at higher irradiance (Peperzak *et al*. 1998; Peperzak 2002).

For each of the *Phaeocystis* blooms observed during the monitoring program, a SiO_4 vs. DIN temporal pattern consistent with the silicate hypothesis was observed in nearfield surface waters as shown in **Figure** 4-6 for the larger blooms in 1997, 2000, and 2004. Silicate and DIN concentrations decreased at close to a 1:1 ratio until SiO_4 concentrations were ≤3 µM and DIN levels were still relatively high (5-6 µM). While *Phaeocystis* has been observed to bloom when SiO_4 is still high in nearfield surface waters, contrary to the silicate hypothesis, these 'blooms' have had relatively low abundance (e.g. 1994 and 2001, data not shown). Two different patterns are seen during non-*Phaeocystis* bloom years. In 1996, SiO_4 and DIN concentrations declined at about a 1:1 ratio from early February to mid March, but when low SiO_4 levels were reached in March, the DIN concentrations were also depleted (**Figure** 4-6). During 1998, the year without a winter/spring bloom (Keller *et al.* 2001), both SiO_4 and DIN concentrations remained relatively high (5-6 µM) through at least the end of March. A combination of these two patterns was observed in 1993 when nutrient concentrations remained elevated through March and then deceased to moderate SiO_4 and low DIN concentrations in April. The data suggest that the silicate-*Phaeocystis* hypothesis applies to the occurrence of these blooms in Massachusetts Bay.

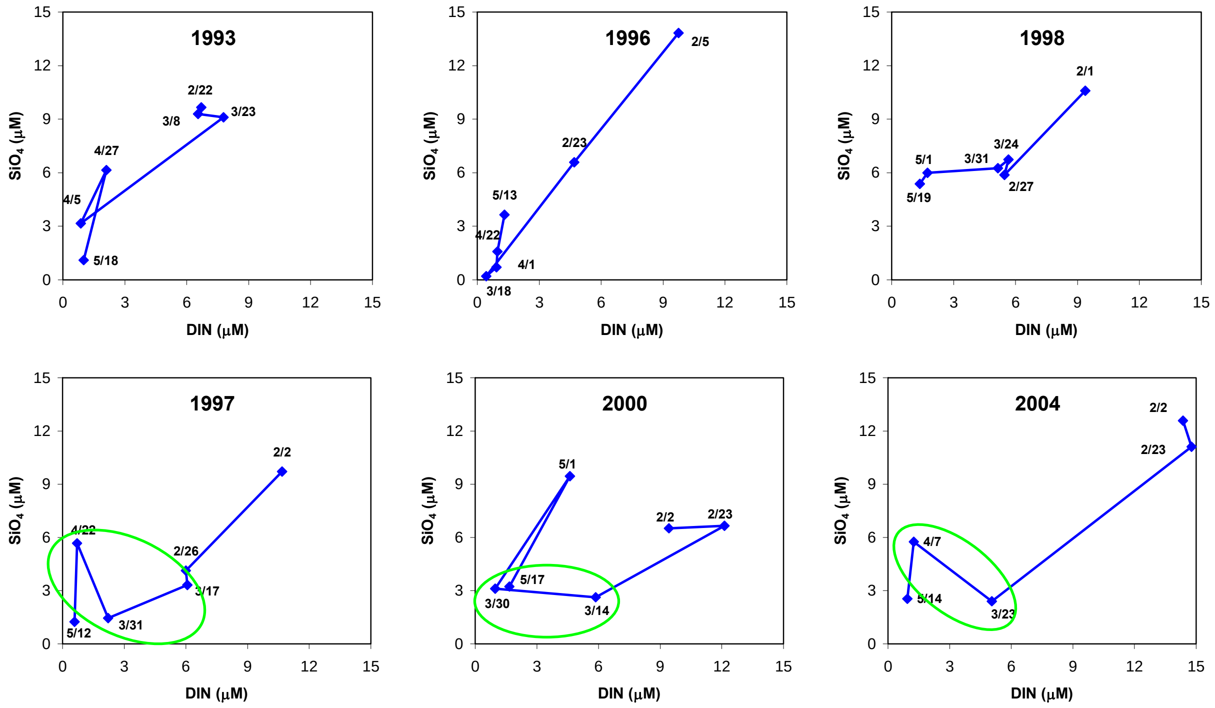


Figure 4-6. SiO_4 and DIN concentrations in surface waters for February through May during non-*Phaeocystis* (1993, 1996, 1998) and *Phaeocystis* (1997, 2000, 2004) bloom years. The ovals indicate the time period of the blooms.

A combination of winter DIN:Si ratio of greater than ~2:1 and an elevated DIN concentration (greater than ~6 um DIN) were observed in years with the largest *Phaeocystis* blooms (**Figure** 4-6). This observation is consistent with model simulations that indicate that the balance of diatom:flagellates is most strongly influenced by perturbation of winter DIN:SI of ca. 3:1 or greater (Tett and Lee, 2005). In the North Sea (Belgian Coast), variation in winter weather patterns and subsequent modification of the delivery of riverine N-delivery and modification of DIN:Si ratio and total DIN explained much of the long-term variation in *Phaeocystis* bloom magnitude (Breton *et al*. 2006). Similar annual hydroclimatic variation may partially explain the differences between *Phaeocystis* bloom years and non-bloom years in Massachusetts Bay.

The eutrophication (altered N:P or N:Si ratios) hypothesis suggests nutrient concentrations may be less important than their relative proportions. During the transition from a winter/spring diatom bloom to a *Phaeocystis* bloom, SiO_4 concentrations decrease to the point where silica either becomes limiting to diatoms or the N:Si ratio becomes high enough that *Phaeocystis* can out-compete the diatoms for nitrogen. In the nearfield, DIN concentrations have increased (increasing N:Si ratios) commensurate with the transfer of the outfall offshore (**Figure** 4-7a). A comparison of pre- (1992-2000) and post- (2001-2004) transfer winter/spring (February-May) surface DIN and NH_4 nearfield mean concentrations shows the mean DIN concentration increased from 4.2 to 5.9 μM (40%) and the mean NH_4 concentration doubled from 1 to 2 μM (both are significant with $p<0.05$ and $p=0.001$, respectively). In contrast, there was a slight decrease in SiO_4 concentrations. The increase in DIN relative to SiO_4 has resulted in an increase in the N:Si ratio (**Figure** 4-7b) which suggests a higher proclivity for *Phaeocystis* dominated blooms under the ratio hypothesis. However, given the regional nature of *Phaeocystis* blooms this nearfield change is not likely to impact the occurrence of these blooms which was region wide.

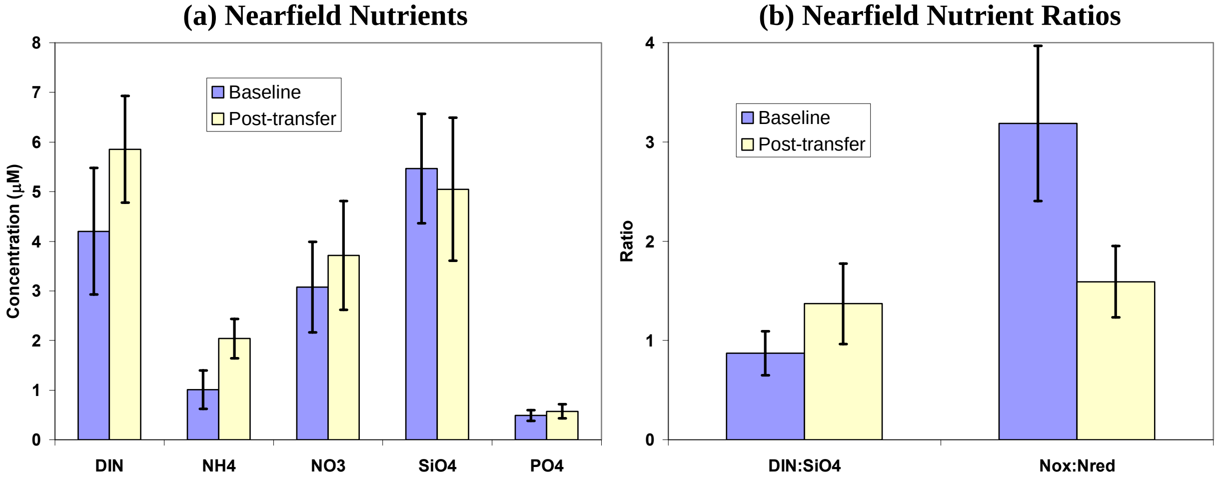


Figure 4-7. Winter/Spring Nearfield mean surface nutrient (a) concentrations (μM) and (b) ratios for baseline (1992-2000) and post-transfer (2001-2004) time periods.

The different silicon demand of *Phaeocystis* (very low) and winter/spring diatoms (high) is not the only nutrient related mechanism influencing *Phaeocystis* blooms. A difference in the ability to assimilate oxidized (NO_3, NO_2) and reduced (NH_4) forms of DIN has been reported to influence the competition between diatoms and *Phaeocystis* (Peperzak *et al*. 1998). In diatoms, production of the enzyme responsible for NO_3 uptake (nitrate reductase) is blocked by NH_4 concentrations of >1-2 μM (Dugdale and Wilkerson 1991). Thus, NH_4 becomes the main DIN source available to diatoms at concentrations greater than this threshold level. Relative to a mixed winter/spring diatom community, *Phaeocystis* has a greater maximum NH_4 uptake rate (V_{max}), but requires higher NH_4 levels (K_s) to achieve higher uptake (Tungaraza *et al.* 2003). Tungaraza *et al.* report *Phaeocystis* blooms in the North Sea may partly be in response to low ratios of NO_3+NO_2 (oxidized N) to NH_4 (reduced N; or $N_{ox}:N_{red}$ ratios) due to anthropogenic NH_4 additions to coastal waters. In addition to drawing down SiO_4, the initial diatom bloom also draws down NO_3 and NH_4 altering existing $N_{ox}:N_{red}$ ratios. In situations where NH_4 supply rate is high (near eutrophic coasts, due to remineralization by zooplankton), those phytoplankton species that can most rapidly utilize this NH_4 resource often bloom. *Phaeocystis* may fill this niche in the spring in Massachusetts Bay, utilizing NH_4 more rapidly than does the winter/spring diatom assemblage.

Even though *Phaeocystis* blooms are a natural feature of the Gulf of Maine, the nearfield bloom magnitude and duration could conceivably be modified by the outfall's modification of the local DIN environment. The nearfield $N_{ox}:N_{red}$ ratio has halved from pre- to post-transfer, falling from 3.2 to 1.6 (significant difference with $p=0.028$; **Figure** 4-7b) as nearfield NH_4 concentrations doubled due to the outfall discharge. During the baseline period, the nearfield $N_{ox}:N_{red}$ ratio typically fell from six in February to three in late March and to 0.3 in May. Since 2001, the typical nearfield $N_{ox}:N_{red}$ has started lower in February (3) and fallen to 1.5 to 0.3 by March and May, respectively. Thus, the increased NH_4 and concomitant change in the $N_{ox}:N_{red}$ ratio could potentially alter competition for available DIN, favoring *Phaeocystis* over diatoms. Although it is not possible to examine the species level competition with the monitoring data, an examination of bloom magnitude suggests that the post-diversion decrease in $N_{ox}:N_{red}$ ratio has not necessarily led to larger blooms (**Figure** 4-8a). Peak nearfield *Phaeocystis* abundance during the 2004 bloom was the highest that has been observed, but the next three highest peak abundances were observed during the baseline period (1992, 1997 and 2000). The frequency of *Phaeocystis* bloom appears to be related to lower $N_{ox}:N_{red}$ ratios, but since the blooms tend to be regional in extent and the change in $N_{ox}:N_{red}$ ratio restricted to within 10-20 km of the outfall (at most) this is likely coincidental rather than causative. Based on the bloom durations calculated using the monitoring data (limited due to intervals between surveys), the increase in bloom duration noted in **Figure** 4-4 shows a strong correlation with decreases in the $N_{ox}:N_{red}$ ratio (**Figure** 4-8b). The only year that did not conform to this trend was 1997, which had a relatively extended bloom and a high $N_{ox}:N_{red}$ ratio. The decrease in the $N_{ox}:N_{red}$ ratio in the nearfield may have an impact on the duration of *Phaeocystis* blooms, but the limited temporal resolution of the sampling makes it difficult to confirm this observation.

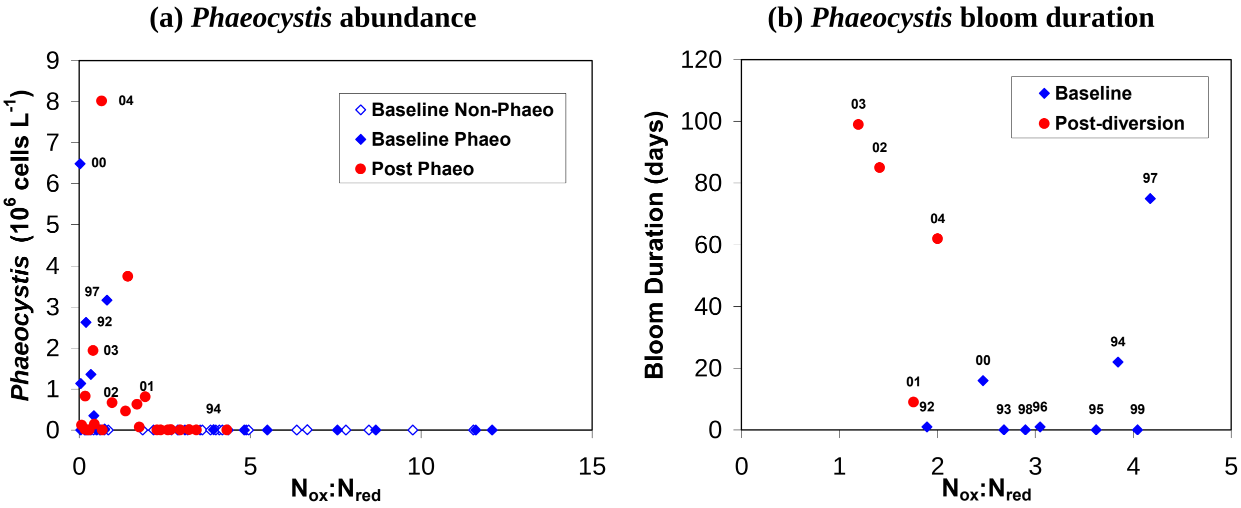

(a) *Phaeocystis* abundance **(b) *Phaeocystis* bloom duration**

Figure 4-8. Comparisons of (a) nearfield survey mean *Phaeocystis* abundance vs. survey mean $N_{ox}:N_{red}$ and (b) *Phaeocystis* bloom duration vs. seasonal mean $N_{ox}:N_{red}$ for nearfield surface waters. Data binned into baseline (1992-2000) and post-transfer (2001-2004) periods.

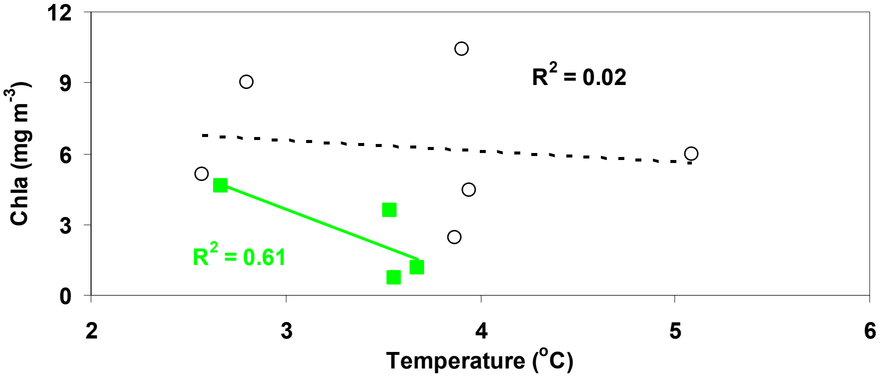
Demonstration that the changes resulting from the outfall transfer contribute to the observed change in *Phaeocystis* bloom patterns and duration would require intensive time series sampling before and during a *Phaeocystis* bloom. Alternatively, accumulation of additional years of monitoring data would be necessary to enable comparison of nutrient and phytoplankton conditions within the nearfield to those beyond the influence of the outfall. Such a comparison may yield indications of the magnitude of the outfall's influence on *Phaeocystis* blooms, as well as the regionally varying dynamics of *Phaeocystis* blooms in the southern Gulf of Maine.

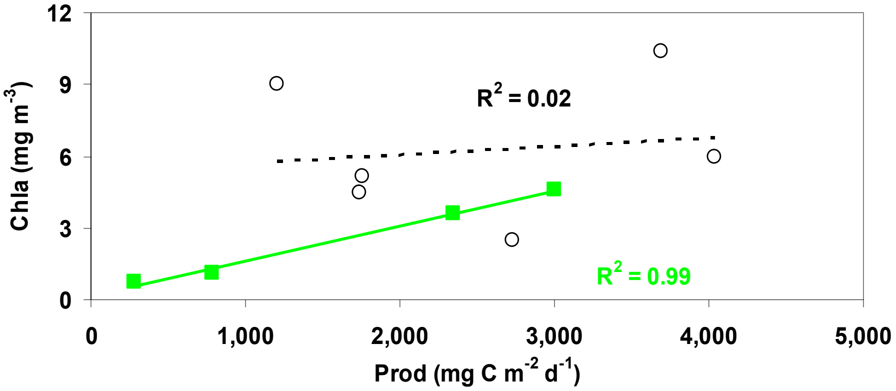
4.2.3 Ecological Effects

An evaluation into the timing and magnitude of the winter/spring bloom in 1998 suggested that the magnitude of the bloom (using chlorophyll as a proxy for phytoplankton biomass) was inversely correlated with temperature and zooplankton abundance (Keller *et al.* 2001). As subsequent data were collected, this relationship lessened as *Phaeocystis* blooms began to be an annual event rather than the ~3 year cycle observed during the baseline. The tendency for *Phaeocystis* blooms to dominate phytoplankton biomass (Davies *et al.* 1992) allows one to extend the interannual comparisons of Keller *et al.* 2001 to a comparison of *Phaeocystis* vs. non-*Phaeocystis* bloom years (diatoms dominant). When the data are stratified according to whether or not a *Phaeocystis* bloom occurred, a very strong inverse relationship between temperature and winter/spring bloom biomass is apparent for the non-*Phaeocystis* years, but only a weak one for the *Phaeocystis* years (**Figure** 4-9a). Also, ecological dynamics appear to differ during years with a *Phaeocystis* bloom including an apparent disconnection between *Phaeocystis* bloom period production rates and phytoplankton biomass and a decrease in zooplankton abundance with increasing phytoplankton biomass (**Figures** 4-9b and c).

For example, data from the consecutive *Phaeocystis* blooms from 2000-2004 suggest that there is a negative relationship between the occurrence of these blooms and the abundance of zooplankton (**Figure** 4-9c). The data show the late spring and early summer nearfield zooplankton means for 2001-2004 were well below the baseline means in June and July (Libby *et al.* 2006a). The literature on *Phaeocystis* grazing offers mixed conclusions about both the degree of grazing pressure and the trophic value of *Phaeocystis* (Schoemann *et al*. 2005, Long and Hay 2006). *Phaeocystis* colonies are surrounded by a tough outer 'skin' (Hamm *et al*. 1999) and colony formation may temporarily decrease vulnerability to grazing (Hansen and Van Boekel 1991; Gasparini *et al*. 2000). In some systems, such as the polar seas and northern coastal areas, *Phaeocystis* blooms coincide with periods of maximal zooplankton production and abundance (Hansen and Van Boekl 1991, Meyer-Harms *et al.* 1999). In other areas including Narragansett Bay (Verity and Smayda 1989), the Irish Sea (Claustre *et al*. 1990) and the southern North Sea (Gasparini *et al*. 2000) poor nutritional value is reported, often followed by reduced fecundity (Verity and Smayda 1989, Turner *et al*. 2002) and reduced copepod abundance (Bautista *et al*. 1992) during *Phaeocystis* blooms.

Nearfield copepod abundance in April and May 2001 to 2004 (months of and after peak *Phaeocystis* abundance) are within the baseline range, but by June and July (two and three months after the bloom) post-transfer copepod abundance is less than 50% of the baseline mean (**Figure** 4-10a). The small zooplankton *Oithona similis* is typically the numerically dominant copepod species in the bay (Libby *et al*. 2006a), but its abundance was well below the baseline mean from late February through August 2001 to 2004 (**Figure** 4-10b). The presence of relatively low copepod abundances during these surveys is offset by the fact that some copepod species, such as *Calanus finmarchicus*, were generally at the baseline mean in April and reached abundances well above the baseline in May (**Figure** 4-10c). Since *Phaeocystis* blooms have occurred consistently since 2000, these pre- and post-transfer trends in zooplankton abundance suggest a link between these blooms and zooplankton dynamics. The MWRA Massachusetts Bay nearfield *Phaeocystis* and zooplankton abundance data was further examined to identify and quantify any relationships between *Phaeocystis* and zooplankton abundance. All data were checked for normality prior to statistical analysis using the Kolmogorov-Smirnov (KS) test. The data


(a) Peak Bloom Chla vs Temperature


(b) Peak bloom Chla vs Production

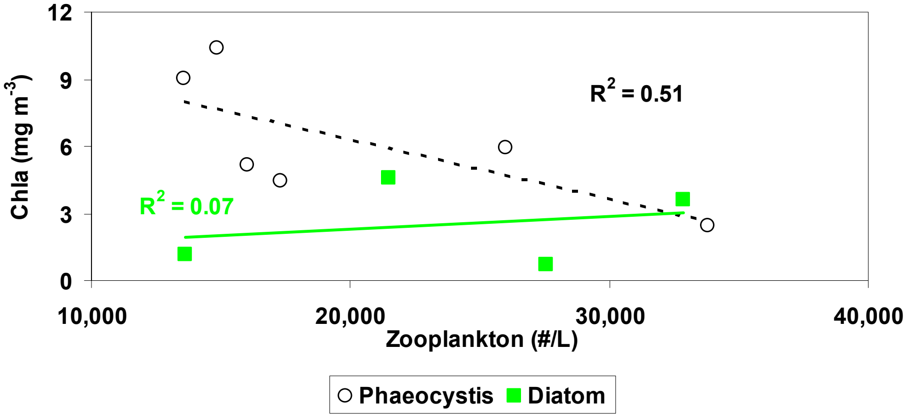
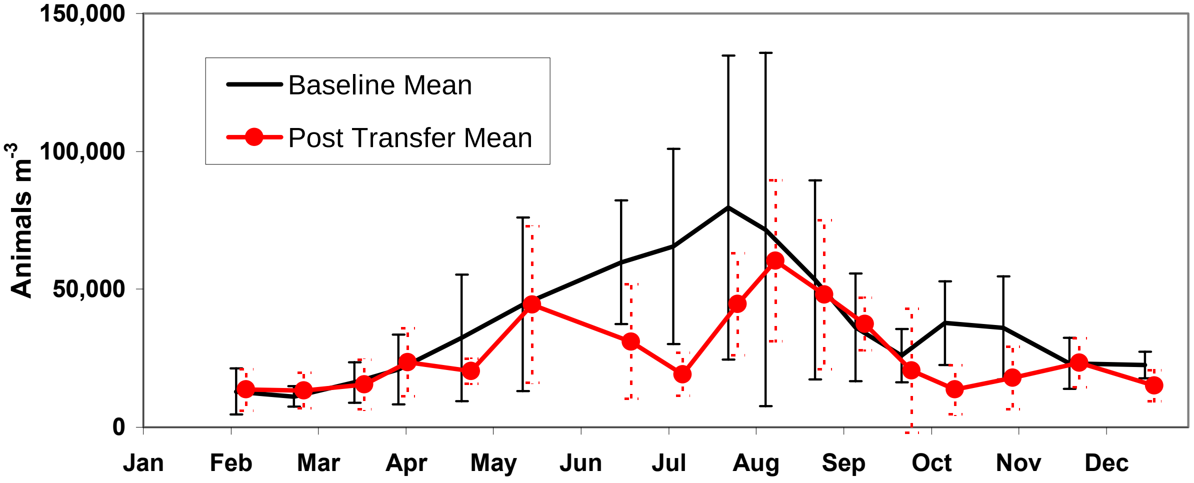
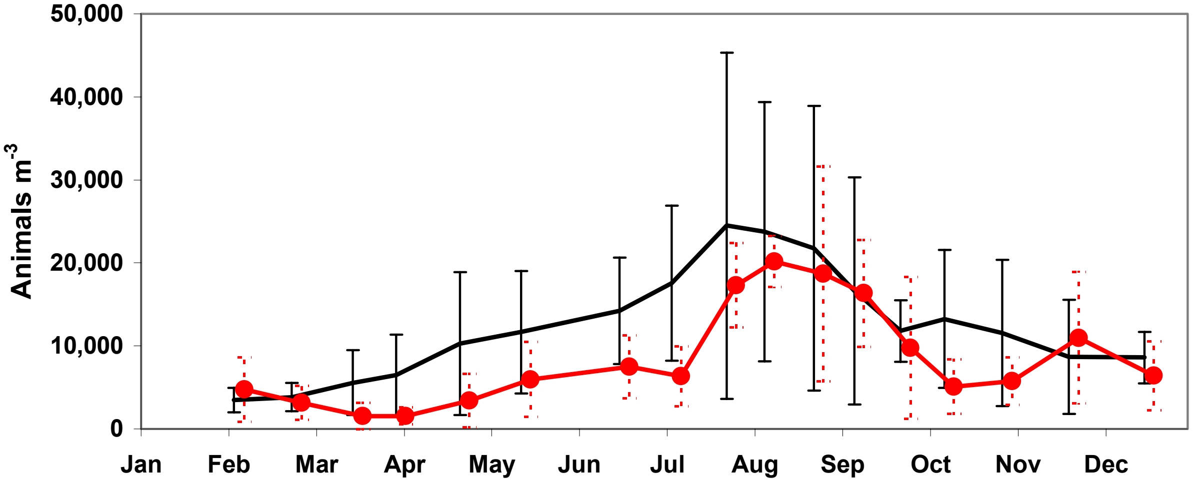

(c) Peak Bloom Chla vs Zooplankton Abundance

Figure 4-9. Spring bloom period (February to April) comparisons of peak bloom chlorophyll concentration vs. (a) temperature, (b) bloom production and (c) zooplankton in the nearfield. Non-*Phaeocystis* years (95, 96, 98 and 99) and *Phaeocystis* years (97, 00, 01, 02, 03 and 04).


(a) Total Copepods


(b) Total *Oithona similis*

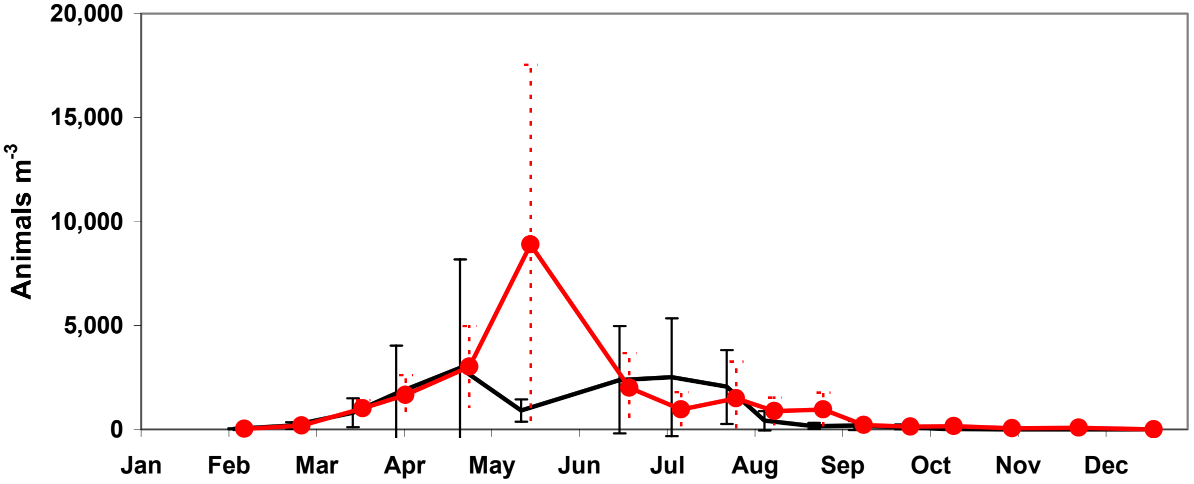

(c) Total *Calanus finmarchicus*

Figure 4-10. Time-series of baseline (2/1992-8/2000) and post-transfer (9/2000-11/2004) survey mean nearfield abundance (animals m^{-3}) of (a) total copepods, (b) total *Oithona* and (c) total *Calanus*. Error bars represent ±1 SD. Data collected from all nearfield stations sampled.

approximated a normal distribution so the Pearson correlation coefficient was used throughout. The Pearson correlation analyses were undertaken to identify co-varying *Phaeocystis* and zooplankton patterns and 2-sample tests were performed to identify differences in zooplankton abundance when *Phaeocystis* is present (+) or absent (-). Data were examined in monthly, two-month, and four month (February-May) time bins. Data were also examined for time-lagged responses with time-lag windows of one to three months. The correlation analyses revealed seasonally varying taxon-specific patterns of *Phaeocystis* – zooplankton variation.

The monthly interval data revealed a positive correlation between *Phaeocystis* and *Calanus* abundance ($p<0.05$) in February. *Calanus* was the only taxa with positive correlation with *Phaeocystis* abundance, and it was highly dependent on elevated *Phaeocystis* and *Calanus* levels in late February of 2003. The correlation is not significant if the late February 2003 value is removed. All other taxa had weak negative correlations with *Phaeocystis* abundance, with the strongest negative correlation observed later in the *Phaeocystis* season (i.e., April and May). For example when all data (*Phaeocystis* present and absent) were examined, *Oithona* abundance and total copepod abundance in April and May surveys were negatively correlated with *Phaeocystis* abundance, but not significantly ($p=0.07$ to 0.11). Looking at only April and May surveys when *Phaeocystis* was present, total zooplankton abundance was negatively correlated with *Phaeocystis* abundance ($p\leq0.05$; see **Figure** 4-11). This regression is strongly dependent on the very high *Phaeocystis* abundances observed in April 2004 (p value falls to 0.2253 upon removal of the elevated 8 million cells L^{-1} *Phaeocystis* value) . The zooplankton response to *Phaeocystis* may not be accurately captured by a linear model. Peperzak (2002) has suggested that inimical *Phaeocystis* effects may not be seen until levels of 1-2 x 10^6 cells L^{-1} are attained. The *Phaeocystis* – total zooplankton data (**Figure** 4-11) are consistent with this view as reduced (levels below ~4,000 animal m^{-3}) April and May zooplankton were only observed in the two years having *Phaeocystis* abundance of > 2 x 10^6 cells L^{-1}.

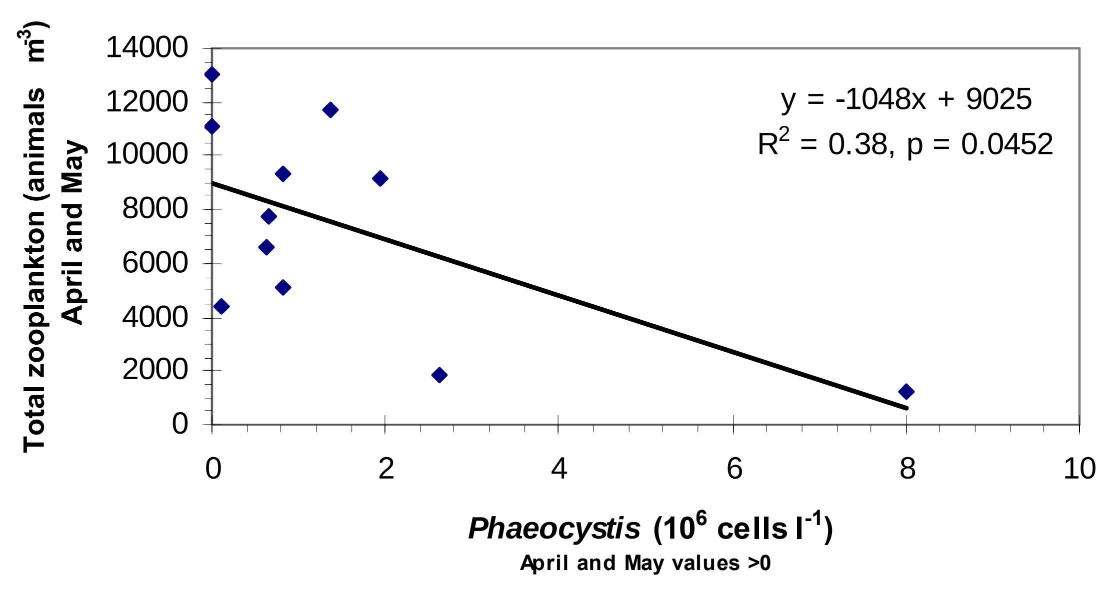


Figure 4-11. Nearfield survey mean total zooplankton vs. *Phaeocystis* abundance for surveys that occurred in April or May (1992-2004) with *Phaeocystis* >0 cells liter^{-1}.

Time lags of up to two months (i.e., February *Phaeocystis* vs. April zooplankton, March *Phaeocystis* vs. May zooplankton, etc.) were also examined by binning *Phaeocystis* and zooplankton cruise data into monthly averages. Only *Calanus* showed a time-lagged response to *Phaeocystis*, with April *Calanus* abundance positively correlated with March *Phaeocystis* abundance (Pearson $r = +0.89$, $n = 11$, $p = 0.003$) and May *Calanus* abundance correlated with April *Phaeocystis* abundance (Pearson $r = +0.92$, $n = 11$, $p < 0.0001$). At a 2-month time lag, May *Calanus* was positively correlated with March *Phaeocystis* (Pearson $r = +0.62$, $n=12$, $p = 0.0306$). However, in all time lagged cases above, the correlation was dependent on 2004 values and removal of 2004 data shifted the correlation below alpha=0.05.

The strong dependence of the correlative relationships described above on one or two observations during elevated *Phaeocystis* levels (i.e., > 2 million cells L^{-1}) suggested the possibility of a square-wave or threshold response rather than a linear response of the zooplankton community to *Phaeocystis* abundance. To examine this, the nearfield data were divided into *Phaeocystis* present (+; *Phaeocystis* >0 cells l^{-1}) or absent (-) and the corresponding abundance of various zooplankton taxa during these two conditions was compared using unpaired t-tests. Data were examined for normality (KS test) and equal variance prior to t-test. All data sets approximated normality so no transformation was applied. Welch's correction was applied to t-tests if the variances of the two data sets were not equal.

As with regression analysis, the strongest *Phaeocystis* effect was seen in April and May (**Table** 4-1). Mean April and May (combined) total zooplankton abundance when *Phaeocystis* was absent (16,030 animals m^{-3}) was about twice the abundance observed when *Phaeocystis* was present (7,406 animals m^{-3}). Most of the difference in total zooplankton abundance when *Phaeocystis* was present or absent appears to be due to the response of *Oithona*. *Oithona* abundance in March and April when *Phaeocystis* was absent was 6,156 animals m^{-3}; *Oithona* abundance decreased by more than 50% when *Phaeocystis* was present to 2,846 animals m^{-3}. While these differences in total zooplankton and *Oithona* are highly unlikely to have occurred by chance alone ($p < 0.05$), the difference was sensitive to the method of data averaging and the resultant decrease in the number of observations.

Table 4-1. Total zooplankton abundance (nearfield mean values) during various time periods with *Phaeocystis* absent or present. Based on data collected from 1992-2004.

Time period	*Phaeocystis* Absent (animals m^{-3})	*Phaeocystis* Present (animals m^{-3})	P-value of unpaired T-test
February	4,356	4,544	0.8685
March	6,789	5,245	0.4834
April	12,700	6,228	0.1256
May	18,980	9,468	0.1504
Feb_Mar	5.065	4,965	0.9267
Mar+Apr	9,941	5,675	0.0691
Apr+May	**16,030**	**7,406**	**0.0202**
Feb-May	9,610	5,998	0.0619
June-July	28,780	23,960	0.6584

The analyses summarized in **Table** 4-1 used all available observations in each time period. If the observations were averaged by time period, and the time periods compared when *Phaeocystis* was present or absent, the April+May zooplankton difference was no longer statistically significant. Zooplankton observations were available in both April and May in only eight years. Of these three (1995, 1996, 1997) lacked *Phaeocystis* blooms and five (1999, 2001-2004) had *Phaeocystis* blooms. Comparing total zooplankton in these two groups of averaged data by t-test yielded no significant difference ($p = 0.1765$)

between the mean of averaged April+May zooplankton when *Phaeocystis* was absent (mean = 9,820 animal m^{-3}; n = 3 years) versus the total zooplankton level (mean = 1,889 animals m^{-3}; n=5 years) observed when *Phaeocystis* was present Other groups examined (*Acartia*, *Calanus*, and nauplii) had no significant difference in abundance when *Phaeocystis* was present versus absent.

These analyses suggest a mixed seasonally varying and taxon-specific response to *Phaeocystis* in Massachusetts Bay. Observed patterns of elevated *Calanus* early in the season (Feb-Mar) and reduced *Oithona* and total zooplankton abundance late in the season (April-May), followed by increased *Oithona* later in the summer in elevated *Phaeocystis* winter-spring years may reflect the influence of *in situ* processes such as differential growth and reproductive success that may be influenced by *Phaeocystis*. Alternatively, different oceanographic regimes (i.e., variable influence of nearshore vs. offshore water masses; Turner *et al*. 2006) having different fauna (*Calanus*-dominated vs. *Oithona* dominated) may be operative in and co-varying with *Phaeocystis* vs. non-*Phaeocystis* bloom years. Further, annual variability in winter-spring temperature may alter the rates of *Phaeocystis* bloom development and/or rates of zooplankton development. This variation may in turn modify grazing interaction between zooplankton and *Phaeocystis* by, for example, altering the timing of spring zooplankton population increase. *Phaeocystis* may have a growth rate dependent palatability and nutritional value for some zooplankton. For example, *A. tonsa* consumed stationary-phase (~ senescent) *Phaeocystis globosa* at a rate 30-times that observed for exponential-phase (~ growing) *Phaeocystis* (Long and Hay 2006). If this growth phase related *Phaeocystis* palatability is operative for Massachusetts Bay zooplankton and *Phaeocystis*, then one might expect temperature-dependent sensitivity to the timing of zooplankton population peaks and *Phaeocystis* blooms, similar to the match–mismatch idea presented by Cushing (1969, 1990).

4.3 Discussion and Conclusions

Phaeocystis is eurythermal and euryhaline allowing for a worldwide distribution of the six recognized species (Verity and Medlin 2003, Schoemann *et al*. 2005). *Phaeocystis* has a heteromorphic life that is unusual for marine phytoplankton, with an alternation between small (3-8 um diameter) solitary free-living flagellated haploid cells and large (millimeter to centimeter diameter) gelatinous colonies of non-flagellated diploid cells. Of the six species known, three species *P. pouchetii, P. globosa* and *P. antarctica*, form blooms of the colonial form (Shoemann *et al*. 2005). The three bloom-forming species are typically found only in cold waters (*P. pouchetii and P. antarctica*) or in the higher latitudes of the temperate zone (*P. globosa*), with *P. globosa* having growth (in culture) to 27°C (Guillard and Hellebust 1971). Although *Phaeocystis* is typically a cool water bloom former (<14 °C), it may occasionally form blooms at ambient temperatures of >20°C (as in the South China Sea; Huang *et al*. 1999 and the Arabian Sea; Garrison *et al*. 1998). The temperature and salinity ranges of both *P. globosa* and *P. pouchetii* (Medlin *et al*. 1994) coincide with the ranges of these variables observed in Massachusetts Bay. The lobed colony formations (Hegarty and Villareal 1998; Borkman pers. obs.) and the observed temperature thresholds of *Phaeocystis* in Massachusetts Bay (Hegarty and Villareal 1998) indicate that *P. pouchetii* is the dominant bloom forming *Phaeocystis* species in Massachusetts Bay. The ability of *Phaeocystis* to form high biomass blooms (up to 10 mg C L^{-1}; Schoemann *et al*. 2005) that sequester a large portion of ecosystem resources and alter trophic pathways (Tang *et al*. 2001) make *Phaeocystis* one of the few 'keystone' phytoplankton taxa whose blooms significantly alter ecosystem function and are important in global biogeochemical cycles (Lancelot *et al*. 1994; Verity and Smetacek 1996). In addition to altered ecosystem function, mucus from gelatinous *Phaeocystis* colonies may have inimical effects such as fouling fishing gear (net clogging; Savage 1930) and fish mortality (Huang *et al*. 1999). Additionally, *Phaeocystis* toxic effects have been observed on larval fish (Aanesen *et al*. 1998; Stabell *et al*. 1999) and toxins have been isolated from *Phaeocystis* (Eilertsen and Raa 1995; Stabell *et al*. 1999; Hansen *et al*. 2003).

Massachusetts Bay *Phaeocystis* bloom magnitude is comparable to that observed in other areas of the Gulf of Maine. The maximum observation of Massachusetts Bay *Phaeocystis* abundance during MWRA monitoring (1992-2005) was 15.5 million cells L^{-1} observed at station F24 in April 2004. Comparable levels (15 million cells L^{-1}) were observed in Cape Cod Bay at station F02 in April 1997. Although these samples are among the five highest phytoplankton abundances recorded during the 1992-2005 MWRA monitoring program, they are on the lower end of the range reported for *Phaeocystis* in coastal waters globally. *Phaeocystis* spp. blooms generally reach a maximum of tens to low hundreds of millions of cells L^{-1} in coastal waters globally (Schoemann *et al*. 2005). In comparison to *Phaeocystis* levels observed in other systems (Schoemann *et al*. 2005), the relatively modest magnitude of Massachusetts Bay *Phaeocystis* blooms may limit or even prevent any long-lasting, ecosystem altering effects from occurring. *Phaeocystis* blooms may have a threshold effect, with inimical effects only being observed when levels exceed 1-2 million cells L^{-1} (Peperzak, 2002). Rousseau *et al.* (2000) saw a decline in trophic efficiency related to a shift from a linear diatom-mesozooplankton food web to a bacteria dominated microbial food web following a massive *Phaeocystis* bloom in the shallow Belgian coastal zone. This transition occurred after a 2-month long *Phaeocystis* bloom that reached a maximum of 1,100 mg C m^{-3}. Using typical *P. globosa* carbon equivalents (Schoemann *et al*. 2005), this bloom maxima was ca. 80 x 10^6 cells L^{-1}, approximately five-times the maximum value observed in MWRA monitoring.

The MWRA monitoring program has revealed an apparent link between *Phaeocystis* bloom duration and interannual variation in spring seawater temperature, with *Phaeocystis* persisting longer in cool years in which the attainment of a 14°C threshold is delayed. In 14 years of Massachusetts Bay monitoring, the most significant change in *Phaeocystis* bloom patterns was an increase in bloom duration from ~20 days or less prior to 2001 to 60 days or greater post-2001 (**Figure** 4-4). Within the confines of the periodic monitoring data available, this bloom duration increase appears to have been due to a delay in the date of bloom termination, which in turn was related to variability in the exceedance of the physiological temperature threshold of 14°C (**Figure** 4-5). *Phaeocystis* blooms are of increased duration in cool years in which the water temperature stays below 14°C into May or even June. In the Dutch Wadden Sea (in the North Sea off the coast of the Netherlands) a 30 year time series (1970 - 2000) has shown an increase in *Phaeocystis* bloom duration related to warming temperature patterns (Cadee and Hegeman 2002). However, in the Wadden Sea the trend has been for *Phaeocystis* blooms to increase their duration by starting earlier in warm winters characteristic of the 1990s relative to the cold winters of the 1970s. Blooms (>1,000,000 cells L^{-1}) lasted ca. 20 days in the 1970s, increasing to ca. 100 days in the 1990s (Cadee and Hegeman 2002). Incorporation of each additional year's monitoring data into this analysis will allow more rigorous evaluation of the link between spring water temperature and *Phaeocystis* bloom duration in Massachusetts Bay.

Phytoplankton populations are under multi-factorial control (Smayda 1998) and in coastal seas, long-term temperature effects are often accompanied by trends in nutrient concentration due to eutrophication (Phillipart *et al*. 2000; Cadee and Hegeman 2002). In the nearfield area of western Massachusetts Bay some post-diversion changes in winter-spring nutrient ratios have been detected – namely an increase in DIN to Si ratio (from ca. 0.8 to 1.5) and a decrease in the N_{ox}:N_{red} ratio (from ca. 3.2 to 1.6; **Figure** 4-7) mainly due to the discharge of ammonium. Due to the different physiological nutrient kinetics of *Phaeocystis* and winter-spring diatoms, these changes would tend to favor *Phaeocystis*, which, unlike diatoms, does not require silica beyond trace amounts (Peperzak *et al.* 1998) and has a greater affinity and capacity for ammonium uptake relative to diatoms (Tungaraza *et al*. 2003). Nearfield data suggest that there could be an association between bloom duration and N_{ox}:N_{red} ratio (**Figure** 4-8b), but this cannot be further substantiated given the temporal resolution of the monitoring data. Comparison of *Phaeocystis* abundance at existing MWRA monitoring stations failed to detect any outfall nutrient effect, as manifested by increased *Phaeocystis* bloom magnitude. Further, three of the four largest *Phaeocystis* blooms recorded by the monitoring program occurred prior to offshore diversion of the MWRA submarine outfall.

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