The 2005 New England red tide of *Alexandrium fundyense*: observations, causes, and potential outfall linkages

Massachusetts Water Resources Authority

Environmental Quality Department
Report ENQUAD 2007-10

Citation

Anderson DM, Libby PS, Mickelson MJ, Borkman DG, He R, McGillicuddy DJ. 2007**. The 2005 New England red tide of *Alexandrium fundyense*: observations, causes, and potential outfall linkages.** Boston: MWRA Report 2007-10. 85 pp. + appendices

The 2005 New England red tide of *Alexandrium fundyense*: observations, causes, and potential outfall linkages

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December 2007
Technical Report No. 2007-10

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EXECUTIVE SUMMARY

In 2005, after several years without paralytic shellfish poisoning (PSP) toxicity, Massachusetts Bay experienced a massive bloom of *Alexandrium fundyense,* the alga that causes PSP, which led to widespread shellfishing closures and extensive field studies. This report is an early effort to present and integrate the data from those numerous sources, especially those related to Massachusetts Bay and the MWRA outfall, with the goal of documenting and understanding the 2005 *Alexandrium* bloom.

The report presents data on *Alexandrium* cell counts, phytoplankton counts, nutrients, and hydrography collected during 30 surveys, conducted by 4 institutions. MWRA conducted 11 of those surveys, 8 of which were rapid-response surveys. It also draws on state shellfish PSP data, buoy and drifter data, and a variety of model results.

The onset of the 2005 *Alexandrium* bloom in the Gulf of Maine was similar to blooms seen in previous years in that the bloom began off the coast of western Maine. However, the 2005 Maine coastal bloom was unusually widespread and intense. Toxicity reached the New Hampshire coast in early May. A sequence of two northeasterly storms in May transported *Alexandrium* cells into Massachusetts Bay, affecting virtually the entire coastline, and ultimately the bloom reached the coasts of the outer cape and islands. The bloom persisted until late June.

Factors affecting the regional bloom
2005 had strong northeast winds, very high river flow, and abundant overwintering resting *Alexandrium* cysts. Numerical model sensitivity tests showed that cysts were the most important factor causing the gulfwide bloom. The high abundance of cysts in western Gulf of Maine sediments provided a large vegetative cell inoculum for the bloom, though the source of those cysts is not well documented. Given the abundant cysts, the model simulation showed a substantial gulfwide bloom whether or not 2005's unusual river flow and winds were used.

Factors affecting the bloom in Massachusetts Bay
There had been no harvesting closures in Massachusetts Bay from 1994 to 2004, presumably a reflection of moderate to low PSP toxicity along the western Maine coast in those years. Toxicity in Massachusetts Bay is linked by transport of cells in the western segment of the Maine coastal current, and wind-mediated flow into Massachusetts Bay at Cape Ann.

Model simulations showed that the 2005 winds caused an earlier bloom in Massachusetts Bay than would otherwise have occurred, but by June the bloom would have been as large with "typical" winds. That is not to say that wind is not important: looking at the detail within each simulation, episodic bursts of northeast winds caused onshore advection and concentration of cells. These model simulations suggest that high *Alexandrium* cyst abundance in the sediments, which led to high vegetative cell abundance in the water column off the coast of Maine and New Hampshire was the most important factor in the 2005 bloom both overall in the Gulf of Maine and within Massachusetts Bay.

Pattern of shellfish toxicity within Massachusetts Bay
Normally toxicity at the south shore of Massachusetts Bay gradually occurs after toxicity at Cape Ann, and at lower levels. But the observations for 2005 showed high PSP and cell abundance nearly simultaneously throughout the bay. A few observations even showed higher and earlier toxicity and abundance to the south, though the sampling frequency was not adequate to provide confidence in that pattern. The high levels of cells and toxicity at the south shore hypothetically could have resulted from stimulation of the bloom as it passed by the MWRA outfall, but it seems more likely that they resulted from anomalous hydrographic conditions due to two strong northeasterly storms (May 7-8 and

May 24-25). The wind events brought *Alexandrium*-laden water into the bay all the way to the coast, where downwelling aggregated the upward-swimming motile cells in the nearshore surface waters. The storms also deepened the mixed layer and entrained nutrients to the surface waters, fueling the bloom.

Pattern of *Alexandrium* abundance in Massachusetts Bay in relation to other phytoplankton

The 2005 *Alexandrium* bloom in Massachusetts Bay occurred in the context of a diverse phytoplankton community. Community dominants followed the following pattern of succession: (1) *Phaeocystis*, followed by the first northeaster storm; (2) centric diatoms with low levels of the pennate diatom *Thalassionema*, followed by the second northeaster storm; (3) high levels of *Thalassionema* and peak levels of dinoflagellates (including *Alexandrium*); (4) post-bloom microflagellates. *Alexandrium* roughly paralleled the pattern of abundance of *Thalassionema* (at about 1/20 of the cell abundance; *Alexandrium* was never a dominant).

Pattern of *Alexandrium* abundance in Massachusetts Bay in relation to nutrients

Ambient ammonium (NH_4) in the outfall nearfield was unusually low in May 2005, presumably due to slightly lower MWRA nitrogen loading than normal, much greater initial dilution of effluent due to stronger currents, and storm-related mixing and advection.

At the simplest level, examination of nutrient concentrations and the abundance of *Alexandrium* cells did not reveal patterns or trends that could be used to infer a cause-effect relationship.

By late June, cell abundance dropped dramatically, including near the outfall, indicating there was no outfall effect stimulating growth and prolonging the bloom.

Dispersion modeling showed that in worst-case quiescent conditions dinoflagellates like *Alexandrium* would only increase by 10% within the outfall plume. That increase is transient because oceanic dispersion more than keeps pace with any stimulation of growth by outfall NH_4. A sensitivity test was also conducted to evaluate effects of the MWRA discharge on the bloom. Ambient NH_4 levels were used as a surrogate for the MWRA discharge, as most of the nitrogen in the discharge is present as NH_4. This is a highly conservative assumption, as ambient waters also contribute NH_4. The model showed that ambient NH_4 could have caused, at most, an increase of 10-14% in *Alexandrium* levels in the "downstream" area. This is an upper bound of the potential outfall effect as previous modeling work indicated that the MWRA outfall provides only a very minor (~3%) portion of the total nitrogen load to the bays, while the Gulf of Maine provides more than 90%.

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A few abbreviations:

PSP	paralytic shellfish poisoning
ECOHAB	a research program on the ecology of harmful algal blooms
MWRA	Massachusetts Water Resources Authority
DIN	dissolved inorganic nitrogen (NH_4 + NO_3 + NO_2)
NH_4	ammonium
NO_2	nitrite
NO_3	nitrate
PO_4	phosphate
SiO_4	silicate
Petrie99	abbreviation for to the nitrate field provided by Petrie *et al.* (1999)
BoF	Bay of Fundy
EMCC	Eastern Maine coastal current
GOM	Gulf of Maine
MCC	Maine coastal current
WMCC	Western Maine coastal current
WHOI	Woods Hole Oceanographic Institution
HMZ	hydraulic mixing zone (sometimes called the zone of initial dilution, sometimes called the near field)
RSB	a spreadsheet-like empirical model of initial dilution described in Roberts (1999)
MDS	multidimensional scaling, a kind of statistical ordination

1 INTRODUCTION

1.1 *Alexandrium fundyense* Blooms in the Gulf of Maine Region

1.1.1 Patterns of PSP

Paralytic shellfish poisoning (PSP) is a recurrent and serious problem in marine coastal waters of the U.S. caused by blooms of several toxic dinoflagellate species in the genus *Alexandrium*. In the northeastern U.S., the causative organism is *Alexandrium fundyense*. Potent neurotoxins called saxitoxins produced by this species are accumulated by filter-feeding shellfish and other grazers and are passed on to humans and animals at higher trophic levels, leading to illness, incapacitation, and even death. PSP is a relatively recent phenomenon within the northeastern U.S., but is now recurrent and widespread, affecting vast expanses of the Gulf of Maine (GOM) coastline (Anderson 1997).

Toxicity was historically restricted to the far eastern sections of Maine near the Canadian border, with the first documented PSP in 1958 (Hurst 1975; Shumway *et al.* 1988), but in 1972, a massive, visible red tide of *Alexandrium fundyense*[1] stretched from southern Maine through New Hampshire and into Massachusetts, causing toxicity in southern areas for the first time (Mulligan 1975). Virtually every year since the 1972 outbreak, western Maine has experienced PSP outbreaks, and on a less-frequent basis, New Hampshire and Massachusetts have as well (Anderson 1997). This pattern was viewed as a direct result of *Alexandrium* resting cysts being retained in western Gulf of Maine waters once introduced there by the 1972 bloom (Anderson and Wall 1978).

1.1.2 *Alexandrium fundyense* Distributions

Within the GOM region, a number of “habitats” can be identified in which toxic *Alexandrium* blooms occur (Anderson 1997; **Figure** 1-1). The most important of these can be defined as: 1) eastern Maine - the eastern Maine coastal current (EMCC) region; 2) western Maine, New Hampshire, and northern Massachusetts – the western Maine coastal current (WMCC) region; 3) Massachusetts and Cape Cod Bays; 4) Georges Bank, the Great South Channel, and Nantucket Shoals; and 5) isolated salt ponds and embayments. Our knowledge of *Alexandrium* dynamics within each of these habitats varies greatly as a result of unequal research emphasis and different levels of shellfish monitoring. It is nevertheless clear that despite the geographic proximity of these areas, the mechanisms of bloom formation and toxicity development for each differ significantly.

Alexandrium fundyense blooms in the GOM have been the subject of an intense five-year investigation through the ECOHAB-GOM program. A series of large-scale field surveys provided data that were combined with mooring observations and numerical model simulations to document the complex dynamics of *Alexandrium* blooms within this region (Anderson *et al.* 2005a). A synthesis of the results of many of those studies was provided by Anderson *et al.* (2005b) and McGillicuddy *et al.* (2005) in the form of conceptual models of *Alexandrium* bloom dynamics. Townsend *et al.* (2001) also formulated a conceptual model for *Alexandrium* blooms that develop in eastern Maine waters. Derived from different approaches, these models have many features in common and some differences as well. These models are discussed in more detail below.

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[1] Both *A. tamarense* and *A. fundyense* occur in the Gulf of Maine. We consider these to be varieties of the same species (Anderson *et al*. 1994; Scholin *et al*. 1995). Neither antibody nor oligonucleotide probes can distinguish between them, and only detailed analysis of the thecal plates on individual cells can provide this resolution. This is not practical for large numbers of field samples. Accordingly, for the purpose of this study, the name *A. fundyense* or simply *Alexandrium* is used to refer to both forms.

1.1.3 Gulf of Maine Circulation

The patterns of PSP within the region are best viewed in the context of the large- and small-scale hydrographic characteristics of the GOM and adjacent waters. One key feature is the Maine Coastal Current system (MCC), described by Lynch *et al.* (1997) as a composite of multiple legs or segments with multiple branch points (**Figure** 1-1).

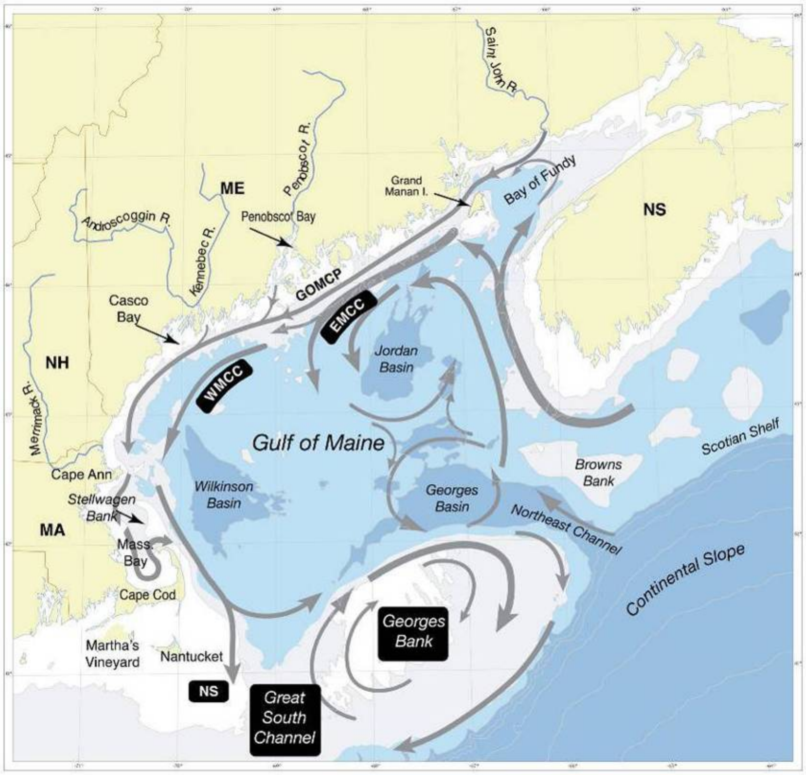

Figure 1-1. Map of the Gulf of Maine, showing dominant circulation patterns (arrows) and *Alexandrium* habitats (black boxes). (WMCC and EMCC are the western and eastern Maine coastal currents; GOMCP is the Gulf of Maine coastal plume). Although shown separately, Nantucket Shoals (NS), Great South Channel, and Georges Bank are viewed as one habitat due to the hydrographic connections. Circulation from Pettigrew *et al.* 2005; habitats - see Anderson (1997).

The upstream, eastern segment of the MCC extends from Grand Manan basin in the Bay of Fundy to Penobscot Bay. This current, hereafter termed the EMCC, derives from inflow from the Scotian Shelf and freshwater from the Saint John River (Bisagni *et al.* 1996). The EMCC often veers offshore south of Penobscot Bay, which defines a branch point. Some EMCC water continues offshore, and some passes around the Penobscot River plume and returns shoreward to form the western segment or WMCC, which is then augmented by freshwater outflow from the Penobscot, Kennebec/Androscoggin, Saco, and Merrimack Rivers. Near Cape Ann, Massachusetts, another branch point is found, with some WMCC water entering Massachusetts Bay, and some traveling along the eastern flank of Stellwagen Bank. Downstream, the Stellwagen segment undergoes another fork into a Nantucket segment, exiting the GOM at the Great South Channel, and a Georges Bank segment that travels to and around Georges Bank.

At the Cape Ann branch point, water may flow into Massachusetts Bay. Massachusetts Bay (which includes Cape Cod Bay), is a semi-enclosed basin bounded on the east by the relatively shallow waters of Stellwagen Bank (**Figure** 1-1). The dominant circulation regime in the bay is a counterclockwise flow that enters the bay just south of Cape Ann, travels south through most of Massachusetts Bay, and exits through a deep channel between the southern end of Stellwagen Bank and Provincetown at the tip of Cape Cod, heading offshore toward Georges Bank and southern waters via the Great South Channel (Geyer *et al.* 1992). Superimposed on this pattern are episodic intrusions of low-salinity water from the WMCC, which enter Massachusetts Bay around Cape Ann (Butman 1975; Franks and Anderson 1992a). Depending on the local wind stress, water from the WMCC can either enter the bay or bypass it entirely, traveling instead along the eastern flank of Stellwagen Bank toward the Great South Channel, Georges Bank, and Nantucket shoals.

1.1.4 Cyst Seedbeds

A key element of *Alexandrium* bloom dynamics is the existence of two large, benthic accumulations of dormant cysts, termed "seedbeds". These were revealed in survey cruises that mapped out the concentrations of cysts in bottom sediments from Massachusetts Bay to the Bay of Fundy (Anderson *et al.* 2005b), and have been confirmed in several subsequent surveys (D.M. Anderson, unpublished data). Two large areas of accumulation are evident – one at the mouth of the Bay of Fundy, and the other offshore of Penobscot and Casco Bays in central Maine. These are depicted as dashed, outlined areas in **Figure** 1-2. In between these deposition sites, cysts are found, but at much lower concentrations. Likewise, in shallow, nearshore waters along the coast, *Alexandrium* cysts are found, but at concentrations that are typically 10-100 times lower than in the offshore seedbeds (Anderson *et al.* 2005b).

Cells are presumed to germinate from these cyst deposits, swimming to surface waters where they divide and initiate the blooms. Germination is controlled by an internal, annual clock, as well as by temperature and light (Anderson *et al.* 2005b). The conceptual model (**Figure** 1-2; Anderson *et al.* 2005b; McGillicuddy *et al.* 2005) begins with cysts that germinate within the Bay of Fundy seedbed, causing localized, recurrent blooms in that area that are self-seeding with respect to future outbreaks as well as "propagatory" in nature - i.e. some cells escape the retention zone and enter the EMCC. Some EMCC cells are entrained into the WMCC, while others eventually deposit cysts offshore of Penobscot and Casco Bays, creating the large, offshore cyst seedbed in that area. In subsequent years, these latter cysts serve as a seed population for blooms that are transported to the south and west by the WMCC, causing toxicity along the coasts of western Maine, New Hampshire, and Massachusetts before the cells are either lost due to mortality or encystment, or are advected out of the region.

Several conceptual models (Anderson *et al.* 2005b and McGillicuddy *et al.* 2005; Townsend *et al.* 2001) argue that *Alexandrium* blooms occur earlier in the western Gulf of Maine than in eastern waters due to more favorable growth conditions at that time, such as warmer temperatures that support faster growth and more water-column stratification. As the season progresses, nutrients become limiting in the western region, and cell growth diminishes and cysts are formed (McGillicuddy *et al.* 2005). Townsend *et al.* (2001) also point out that early season growth conditions are not favorable for *Alexandrium* growth at the upstream (eastern) end of the EMCC due to turbulence and deep mixing and that cells begin to grow and accumulate at the western end of the EMCC as those waters stratify and light no longer limits uptake of the abundant nutrients.

Toxicity occurred every year in southwestern Maine during the 1970s, '80s, and early '90s (Franks and Anderson 1992b) and continued through 2005 albeit at lower levels of severity from 1994-2004 (Anderson *et al.* in prep.). The working hypothesis for Massachusetts Bay toxicity is that part of the *Alexandrium* bloom causing toxicity to the north may be carried south by currents, and some portion of those currents may enter the bay near Cape Ann. Generally the pattern in Massachusetts Bay loosely echoes that to the north, but at lower levels and with a variability that might be related to winds and currents. **Figure** 1-3 shows that there was little or no toxicity within Massachusetts Bay from 1994 to 2004, and no harvesting closures. Just north of Cape Ann is a station at Gloucester (Annisquam); it is more readily influenced by the Western Maine Coastal Current than stations within Massachusetts Bay. Toxin was detected at Gloucester albeit at low levels in 1994, 1995, and again in 2000, and 2003 (not shown).

In marked contrast to this interval of low or absent toxicity in Massachusetts and western GOM waters, for approximately three months in mid-2005, a massive bloom of *Alexandrium* occurred in southern New England coastal waters (Anderson *et al*. 2005c). This outbreak was the largest since the 1972 event, and ultimately resulted in shellfish harvesting closures that extended from eastern Maine through Massachusetts to its offshore islands of Martha's Vineyard and Nantucket. Additionally, a 40,000 km^2 area of offshore (federal) shellfish resources was closed (see Section 2.2). Press reports estimated the economic loss from this outbreak to exceed $50 million to the Massachusetts shellfish industry alone (M. Hickey, unpublished data). Maine and New Hampshire also experienced significant losses.

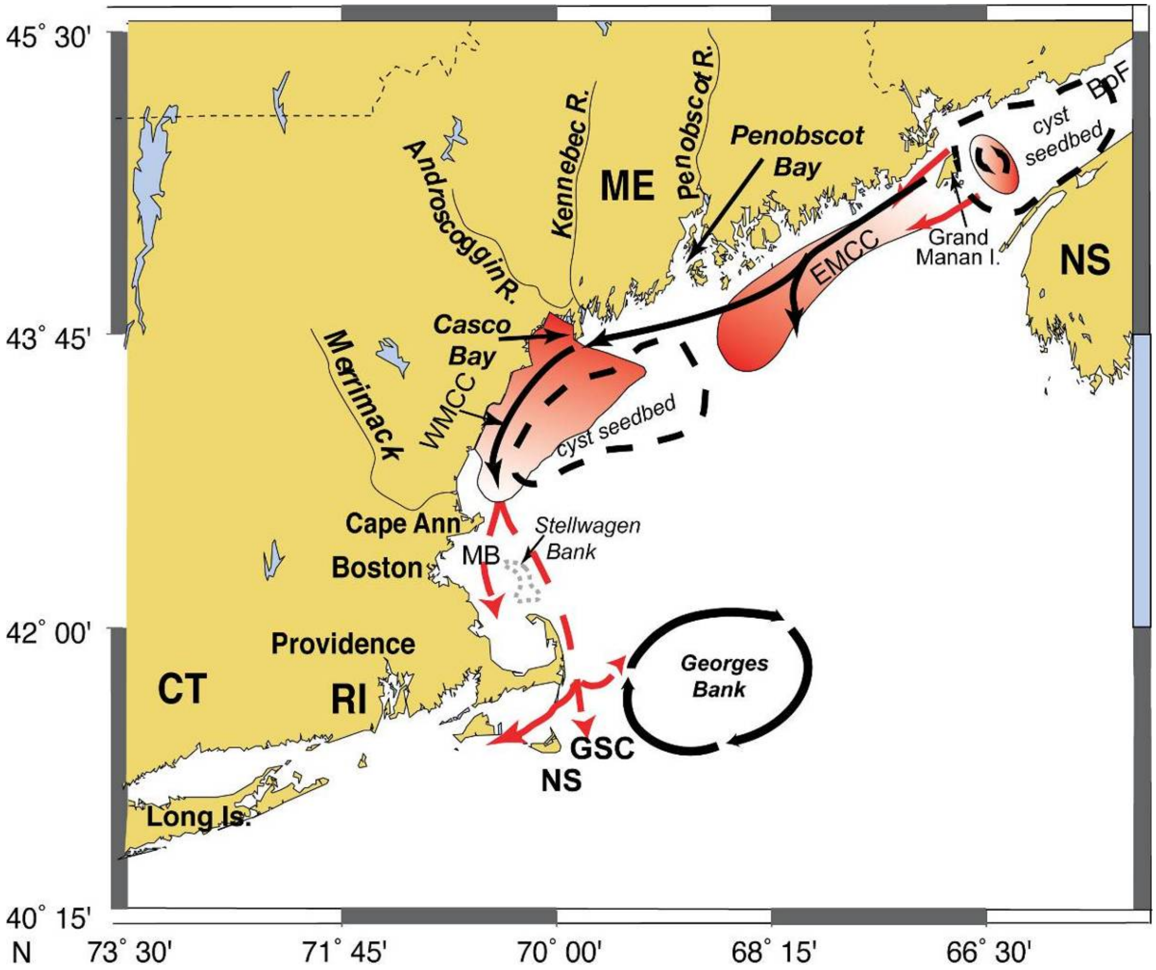

Figure 1-2. Conceptual model of *Alexandrium* bloom dynamics and PSP toxicity. Shown in solid black lines are the EMCC and WMCC coastal current systems, and in dashed black lines the cyst seedbeds in the Bay of Fundy (BoF) and mid-coast Maine. The red shaded zones show areas where *Alexandrium* vegetative cells accumulate at higher concentrations relative to adjacent waters. Red dashed lines how the delivery or transport pathways of these establish bloom populations to southern waters. GSC = Great South Channel; NS = Nantucket Shoals; MB = Massachusetts Bay; Modified from Anderson *et al.* 2005b.

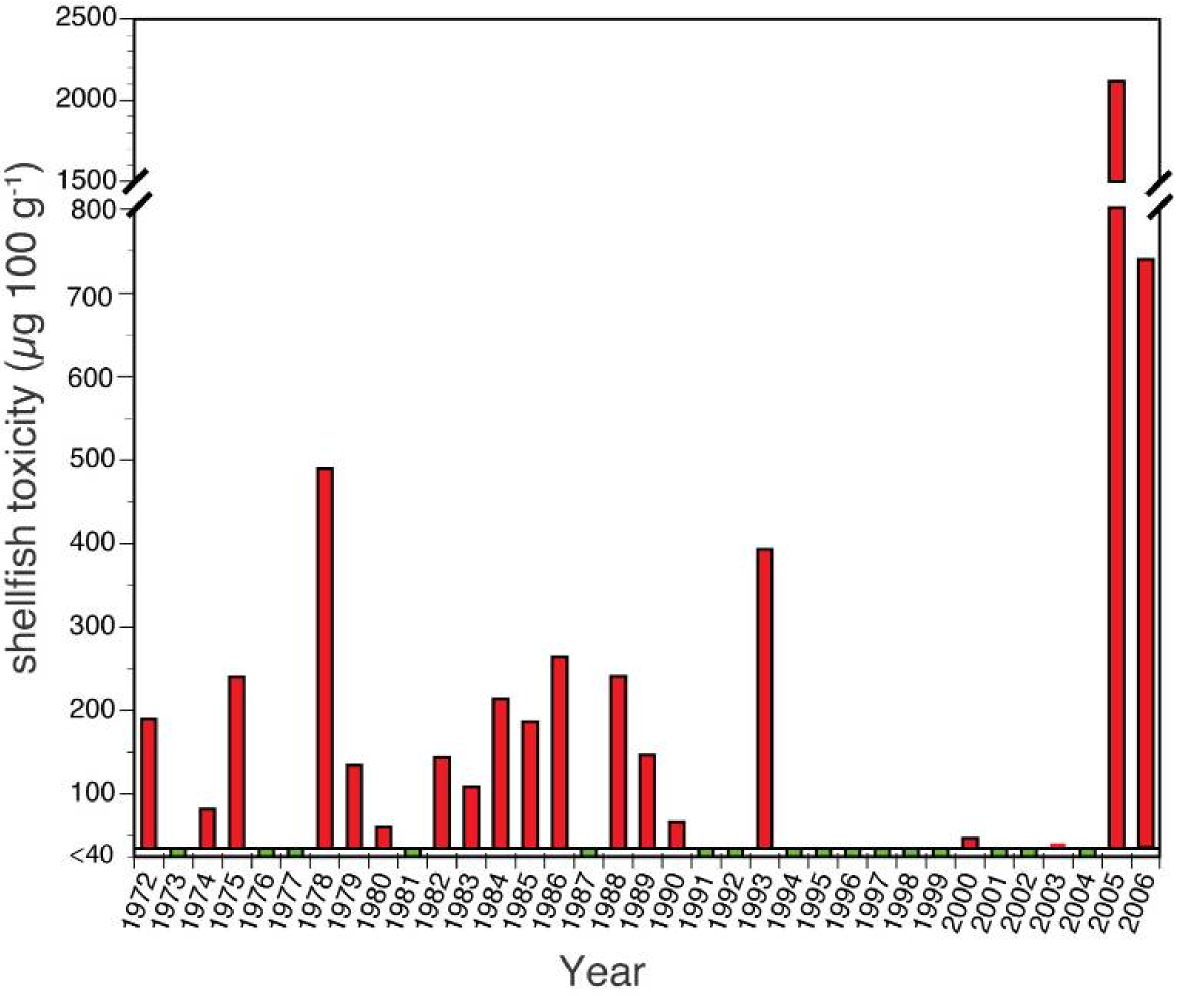

Figure 1-3. Maximum levels of PSP toxicity in Massachusetts Bay, 1972- 2006. Units are µg saxitoxin per 100 g shellfish meat. Source: Massachusetts Division of Marine Fisheries.

1.2 MWRA *Alexandrium* rapid-response surveys and report objectives

The MWRA transferred the discharge of secondary treated sewage effluent from Boston Harbor to an outfall in Massachusetts Bay in September 2000. Since 1992 MWRA has monitored Massachusetts and Cape Cod Bays to verify compliance with NPDES permit requirements and to determine whether change within the system exceeds ecological thresholds. The impacts documented include a dramatic decrease in dissolved inorganic nitrogen (DIN) levels in the harbor and increases near the bay outfall. The change in DIN in Boston Harbor is associated with significant decreases in chlorophyll, particulate organic carbon, primary production, and changes in seasonal productivity patterns. The higher nutrient load near the outfall, however, has not translated into significant changes in phytoplankton biomass or productivity. Coincidentally, regional changes have been observed such as increases in seasonal phytoplankton biomass and apparent changes in the frequency, duration and magnitude of *Phaeocystis* blooms throughout the bay. The major red tide bloom of *Alexandrium* in 2005 increased interest in the potential effect the outfall may have in exacerbating such blooms, although a direct association between the outfall and these regional blooms has not been observed.

The major red tide bloom of *Alexandrium* in 2005 provided an opportunity to test whether the outfall may exacerbate such blooms at a local scale. Before 2005, *Alexandrium* was too rare to test this concern. As mentioned above, the last toxicity event associated with *Alexandrium* was recorded in 1993. Nonetheless, MWRA developed a standing "rapid response plan" (Libby 2006) and when the first signs of the 2005 bloom were observed MWRA initiated the intensive sampling program. All told, from early May to early July, MWRA conducted 11 surveys focused on the *Alexandrium* bloom including 8 *Alexandrium* Rapid Response Surveys (ARRS; Appendix A). These were complemented by 14 surveys by Woods Hole Oceanographic Institution (WHOI), 4 surveys by Center for Coastal Studies, and 1 survey by University of Massachusetts at Dartmouth. The common goal of all groups studying the bloom was to determine the extent and magnitude of the bloom, and to communicate results quickly between researchers and resource managers.

1.2.1 *Alexandrium* Rapid Response Surveys

MWRA's ARRS plan (Libby 2006) provides details on surveys that are designed to respond to a large or unusual red tide bloom of *Alexandrium* in Massachusetts Bay. The response plan supplements regular MWRA outfall monitoring surveys, which are not designed to describe such a bloom or provide all the desired information relevant to possible outfall effects. The goal of ARRS is to characterize and understand a major *Alexandrium* bloom in Massachusetts Bay and determine whether the MWRA outfall influences the bloom. Data are collected during the surveys to obtain insight on the bloom dynamics and to evaluate the potential influence (impact) of outfall discharge on the bloom (e.g., localized and downstream, change in magnitude of bloom, etc.).

The current understanding, as described in detail in Section 1.1, is that *Alexandrium* blooms that affect Massachusetts Bay originate off the mid-coast of Maine in April to May and are transported southward with the western segment of the Maine coastal current (Franks and Anderson 1992a). The blooms may enter Massachusetts Bay under appropriate conditions - sustained winds from the northeast at the time an offshore bloom is passing the northern boundary of Massachusetts Bay (Anderson *et al.* 2005d). There is no evidence that *Alexandrium* blooms in Massachusetts Bay are initiated internally.

The north to south transport of blooms within Massachusetts Bay is also evident in data from the Massachusetts Department of Marine Fisheries (DMF) PSP monitoring program. An examination of DMF data from 1980-1999 indicates that there are some clear patterns that are relevant to this project (D.M. Anderson, A. Solow, unpublished). These researchers focused on data from stations in Gloucester, along the south shore of Massachusetts, and in Cape Cod Bay (**Figure** 1-4). During each of the years that PSP toxicity was observed, it was first seen at the Gloucester station and then subsequently at stations to the south. The toxicity always progressed from the north to the south (see **Figure** 1-5 for 1993 data). Additionally, the peak level of toxicity was generally higher to the north. Moreover, the data indicate the higher the toxicity in Gloucester, the higher the likelihood that toxicity will be observed to the south. The historic PSP toxicity data not only support the plume advection hypothesis (Franks and Anderson 1992a; Anderson *et al.* 2005d), but also allow the transport time through the system to be estimated. Based on the 1980-1999 data, there is a delay of about 2 weeks between when toxicity is first observed at the MA DMF Gloucester station and when it is measured at the south shore sites and another week or two before it is seen in Plymouth or Sandwich (**Figure** 1-6)

It was hypothesized that, in the absence of a major hydrographic forcing that might alter the typical advective pattern discussed above, if an *Alexandrium* bloom is more intense (higher abundance and higher toxicity) along the south shore than to the north, or if toxicity is observed earlier at stations along the south shore than at Gloucester, these patterns would be consistent with, and suggestive of, an outfall effect.

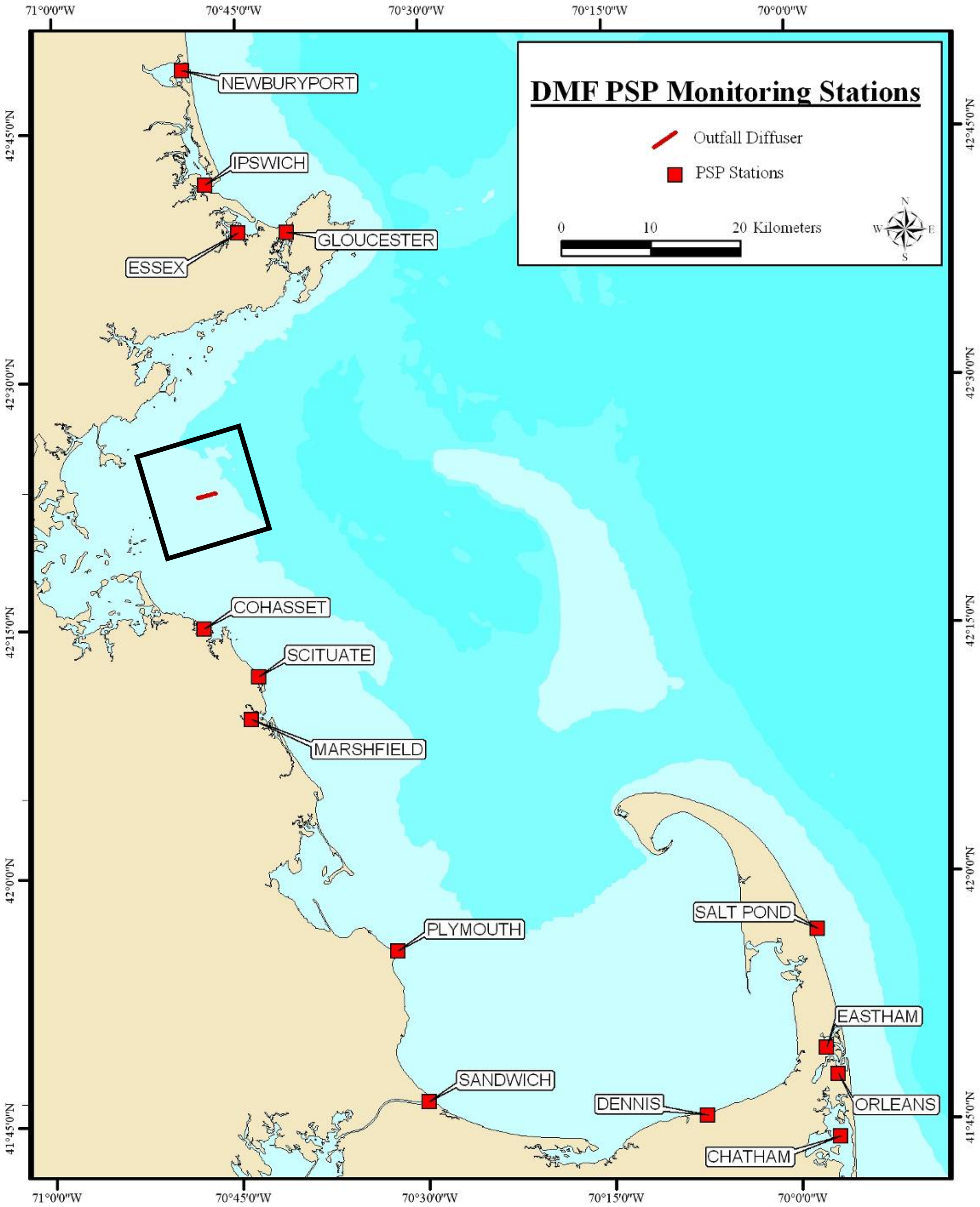

Figure 1-4. Massachusetts Department of Marine Fisheries primary shellfish toxicity PSP-monitoring stations. Nearfield area is represented by the rectangle around the outfall diffuser.

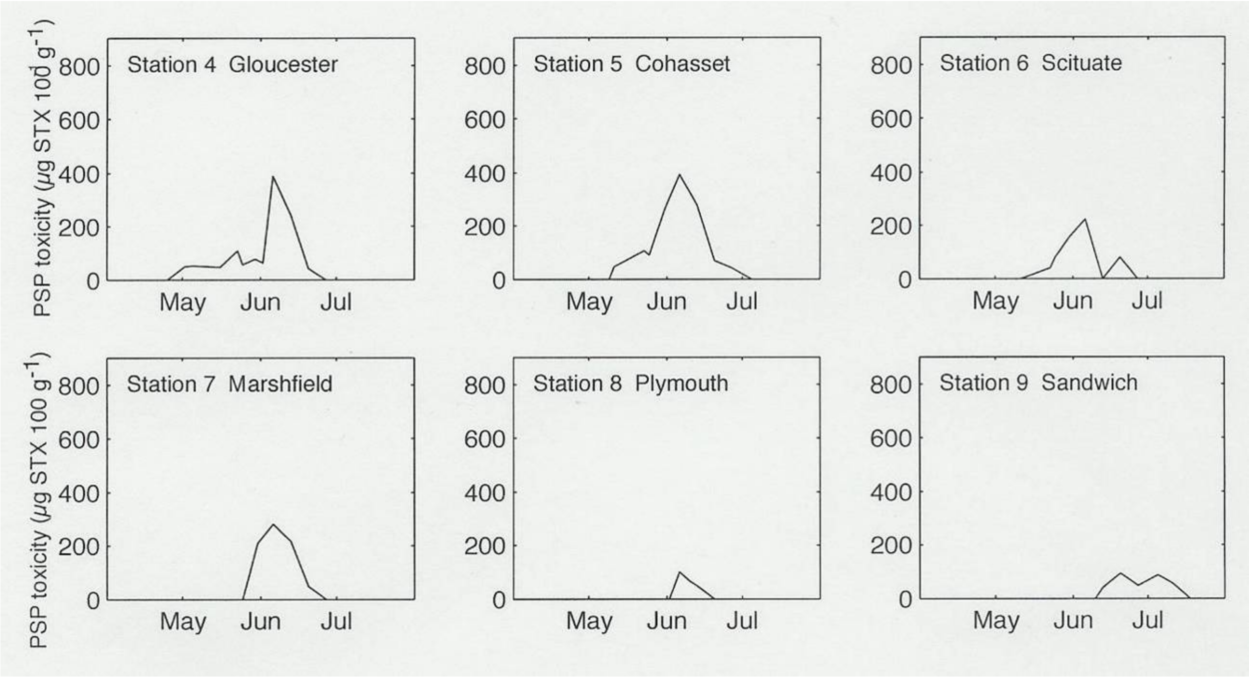

Figure 1-5. Time series of PSP toxicity in 1993 from Gloucester, south shore and Cape Cod Bay DMF stations. (D.M. Anderson *et al.*, unpublished data)

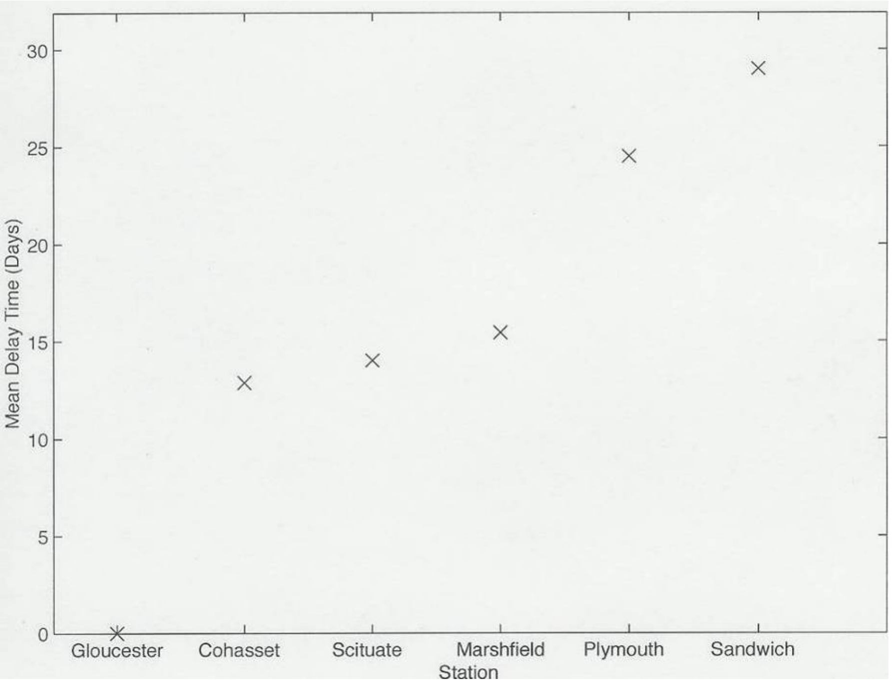

Figure 1-6. Average delay between PSP toxicity in Gloucester and stations along the south shore and in Cape Cod Bay based on 1980-1999 DMF data with 1996 omitted. (D.M. Anderson *et al.*, unpublished data)

1.2.2 Report Objectives

The main objective of the ARRS effort and this report is to evaluate what if any impact the bay outfall had on the 2005 *Alexandrium* bloom in Massachusetts Bay. The regional nature of *Alexandrium* blooms was understood prior to the 2005 event as discussed in Section 1.1, but the effect of the bay outfall on the local expression (exacerbation or prolongation) of the bloom was unknown. It was hypothesized that two patterns that might be suggestive of an outfall effect would be if South Shore shellfish became more toxic or *Alexandrium* abundances were higher than on the North Shore, or became toxic first. During the 2005 red tide event, this was in fact observed, as the highest cell counts, and early and persistent toxicity were observed along the South Shore near Plymouth and Sandwich. These findings and a reevaluation of the hypothesis are discussed in detail in this report.

This report also evaluates other evidence relevant to a possible outfall effect on *Alexandrium*, such as:

- Were cells abundant "downstream" of the outfall?
- Were cells more abundant at the depth of the outfall plume?
- Did cells linger near the outfall, even as the bloom declined regionally in July?
- Did *Alexandrium* become more abundant relative to other phytoplankton?
- Were nutrient conditions an important regulator of *Alexandrium* abundance?
- Is the outfall an important source of nutrients relative to other sources?
- Could cells grow fast enough to outpace oceanic dispersion?
- Does modeling fail to fit observed abundance without invoking faster growth and nutrient stimulation within the bay?

Finally, the report strives to both document and characterize the 2005 New England red tide. Given the unprecedented, coordinated monitoring effort undertaken by MWRA and WHOI, a comprehensive documentation of the surveys, data, and interpretations is warranted. Appendix A presents a compilation of the surveys conducted within or in the vicinity of the bays. This includes survey details and station location maps. Due to frequent modifications to the surveys as the bloom developed and continued, there was not a set of nominal station locations across all surveys. **Figure** 1-7 shows the general location of stations and instrumented buoys during each survey discussed in this report and appendices. Note, however, that many of the MWRA rapid response surveys (AF05#) reoccupied the same sets of stations and **Figure** 1-7 does not represent all stations sampled (multiple symbols on top of each other). **Table** A-1 and figures in Appendix A present details on the number of stations sampled and their locations during each survey.

The successful coordinated monitoring effort in 2005 has already served as an example and model for monitoring that was done during the subsequent 2006 *Alexandrium* bloom. One of the objectives of this report is to lay ground work for framing questions pertaining to future red tides and the responsibility and role of MWRA in monitoring them. This includes revisiting the current MWRA contingency plan (MWRA 1997) threshold for *Alexandrium* of only 100 cells L^{-1}. This threshold was established using the 1992-2000 baseline monitoring date, during which time there was only one minor *Alexandrium* related toxicity event in the bays in 1993. Given the high abundances observed during the 2005 (and 2006) bloom, it may need to be modified to be a useful, but not overly conservative, threshold. Finally, we discuss whether or not these events have changed the bloom dynamics for *Alexandrium* in Massachusetts Bay such that blooms might now originate *in situ*, rather than being advected into the Bay from established populations in northern waters

As noted above, MWRA conducted a series of rapid response surveys focused on the May/June 2006 *Alexandrium* bloom. The results of these surveys are cited herein, but the main focus of this report is the 2005 *Alexandrium* bloom.

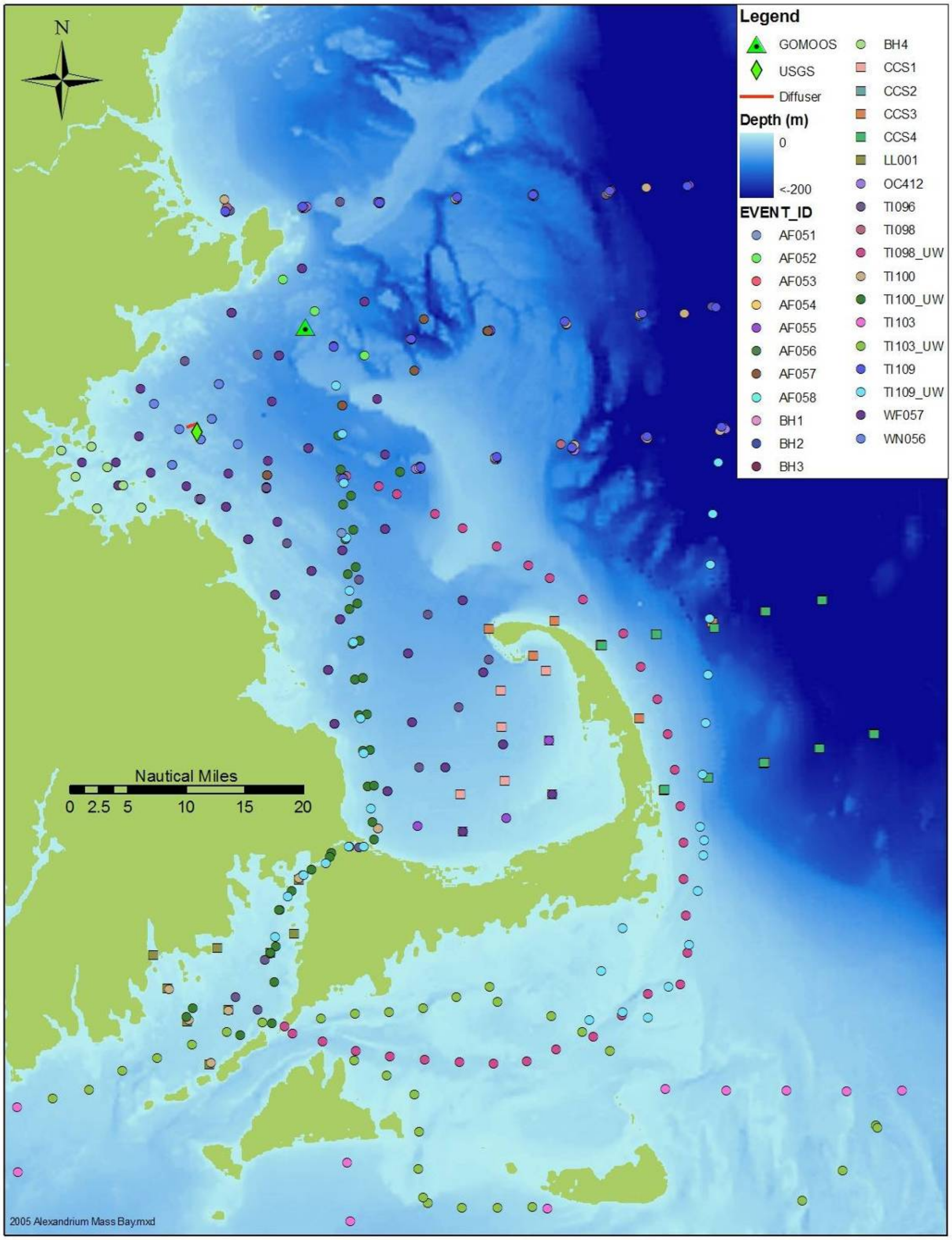

Figure 1-7. Locations of stations sampled during the 2005 New England red tide. Note that multiple visits to stations are not represented –see Appendix A for survey-by-survey details.

2 OBSERVATIONS

Examinations of algal blooms require a variety of biological, chemical, and physical observations. Our current understanding of *Alexandrium fundyense* (hereafter *Alexandrium*) red tide blooms, as detailed in Section 1, emphasizes the role that meteorological and physical oceanographic forcings play in bloom development and distribution of cells and toxicity associated with the bloom in the Gulf of Maine. This section begins by setting the stage on which the bloom occurred and then describes the distributions of shellfish toxicity. The initial conditions for the bloom as depicted by *Alexandrium* cyst distributions are also presented. The bulk of this section characterizes the *Alexandrium* bloom over the course of more than two months. The bloom is also put in context of the phytoplankton community structure. Lastly, we examine the bloom in light of the overall regional nutrient distributions and specifically related to localized nutrient loading at the MWRA bay outfall.

2.1 Meteorology and Physical Oceanography

The winter/spring of 2005 was marked by above-average precipitation throughout coastal Massachusetts. This included substantial storm events that produced a deep snowpack in January and heavy rainfalls in April and May. The combination of new and released precipitation led to extremely high river flows across New England (**Figure** 2-1). The flow of the Merrimack was the highest it has been during the MWRA monitoring program for the April-June period, with three substantial discharge events. This nutrient-laden, freshwater pulse is thought to have exacerbated the development of *Alexandrium* bloom in the Western Gulf of Maine (Anderson *et al.* 2005c).

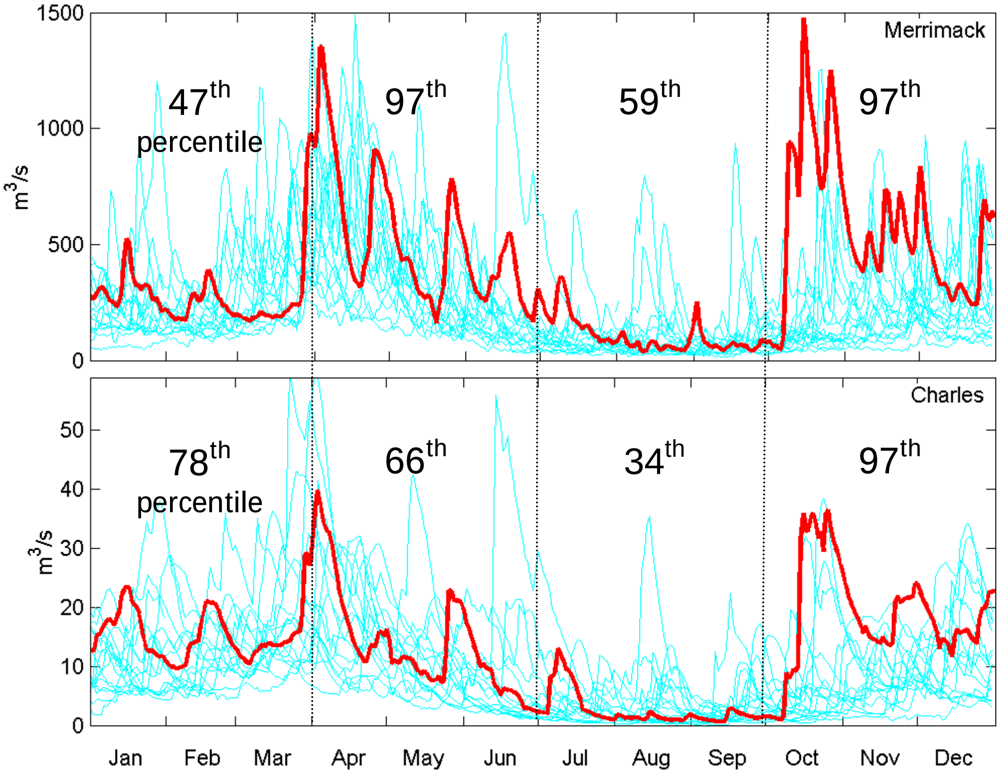

Figure 2-1. Comparison of the 2005 discharge of the Charles and Merrimack Rivers (thick red lines) with the observations of the past 15 years (1990-2004; thin blue lines). Percentile for 2005 discharge shown for each season.

In May 2005, two northeaster storms (a "northeaster" is an extratropical cyclone with strong winds out of the northeast) impacted the Massachusetts coast with sustained winds >10 m s^{-1} over multiple days. Timeseries of the velocity and salinity at the GoMOOS-A buoy are shown in **Figure** 2-2, along with timeseries of the Merrimack flow and the wind forcing. The near-surface currents (3rd panel) show four major pulses to the SE (into Massachusetts Bay), with speeds greater than 80 cm s^{-1}. Currents of this magnitude are rare at this mooring location, and their occurrence was affected by strong freshwater forcing combined with downwelling-favorable winds. The first of these pulses (in early April) corresponds to a moderate wind event during high river flow. The next two current velocity pulses, in early and late May, occurred during intense northeasterly storms (note wind pulses in **Figure** 2-2 panel 2). Another strong wind event occurred in mid-June, resulting in the fourth large velocity pulse.

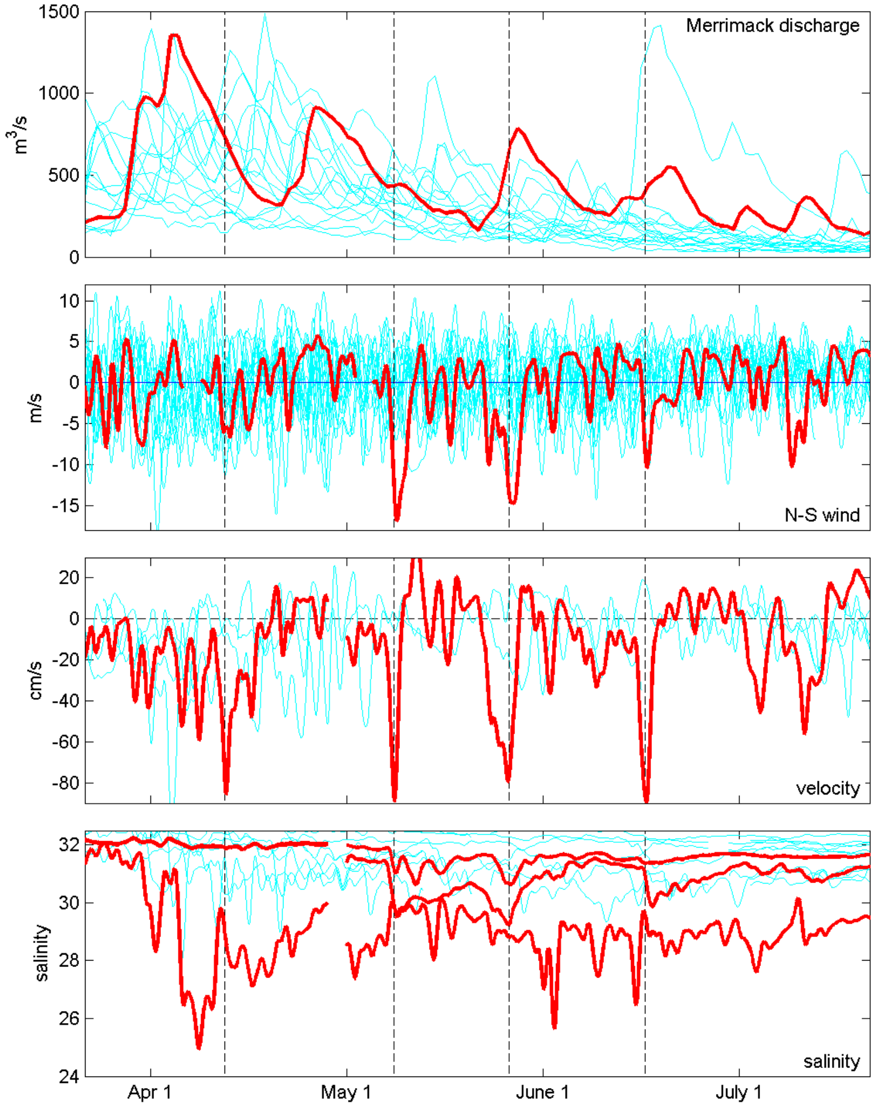

Figure 2-2. Merrimack discharge; N-S winds at the Boston Buoy; NE-SW current velocity at GoMOOS-A; salinity at GoMOOS-A at 5, 25 and 50-m depth. For salinity, the lower curves correspond to shallower depths. All of the data are low-pass filtered to remove the tides. Year 2005 is shown in red; previous years in light blue. The vertical dotted lines mark the four current velocity pulses indicating flow into Massachusetts Bay (April 11, May 7-8, May 24-25, June 15). Note that the GoMOOS data only extend back to 2001, whereas the other data extend to 1990.

The northeaster storms in May 2005 contributed to a monthly mean wind out of the north-northeast for the GOM and Massachusetts Bay. This is in stark contrast to the typical monthly mean winds for May, which is out of the southwest as represented by the 1948-2005 average in **Figure** 2-3. The comparison also hints at why *Alexandrium* blooms have been relatively infrequent events in Massachusetts Bay since they typically occur in the western GOM in May, but are often kept out of the bay by prevailing winds.

In May 2005, not only did the northeaster storms bring GOM waters and associated nutrients and *Alexandrium* cells into Massachusetts Bay, but the winds also pushed the waters inshore and to the south. Model projections of sea surface elevation show water piling up along the South Shore (**Figure** 2-4). The University of Massachusetts Boston (UMB) modeling results also suggest the formation of eddies in the bay once wind forcing relaxes after both of the May northeaster storms. **Figure** 2-5 highlights the modeled surface currents and compares the modeled and measured salinity from May 11, 2005. UMB modelers are currently working on the details of these findings, but they are presented here to highlight potential mechanisms for mixing and transporting water in the bays that contribute to the formation of some of the water quality patterns observed in the data.

On a related note, an important aspect of the wind forcing on Massachusetts Bay is the average north-south component of wind stress, which determines the preponderance of upwelling- or downwelling-favorable conditions in the western bay. The most notable feature of the 2005 wind forcing is the strong downwelling during the month of May (**Figure** 2-6). May is normally transitional between winter downwelling- and summer upwelling- favorable conditions, and so the net north-south wind stress is typically close to zero. The 2005 conditions in May were more characteristic of the winter period, with strong net downwelling. This was the result of two strong northeaster storms that occurred in May, as indicated by the time series data in **Figure** 2-2.

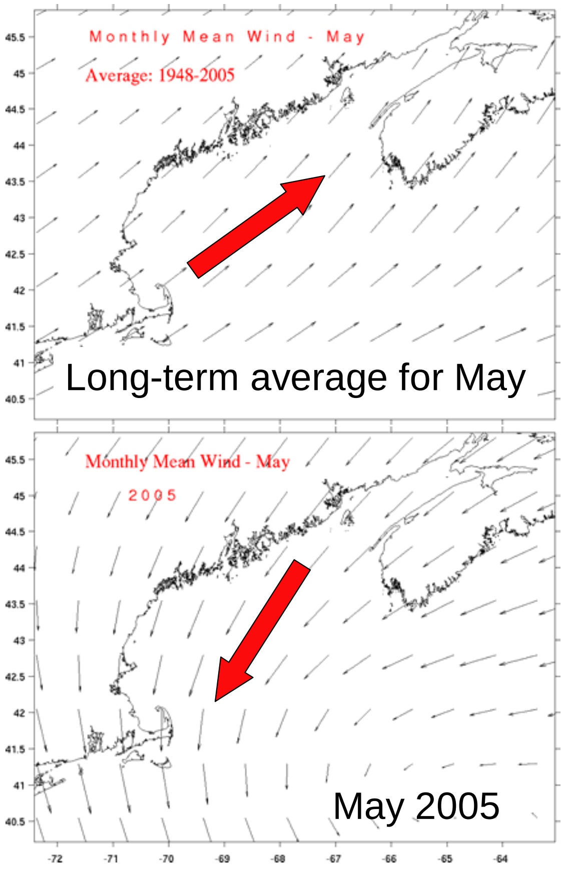

Figure 2-3. Monthly mean wind patterns for May 1948-2005 vs. May 2005. (Source: NOAA NCEP Reanalysis)

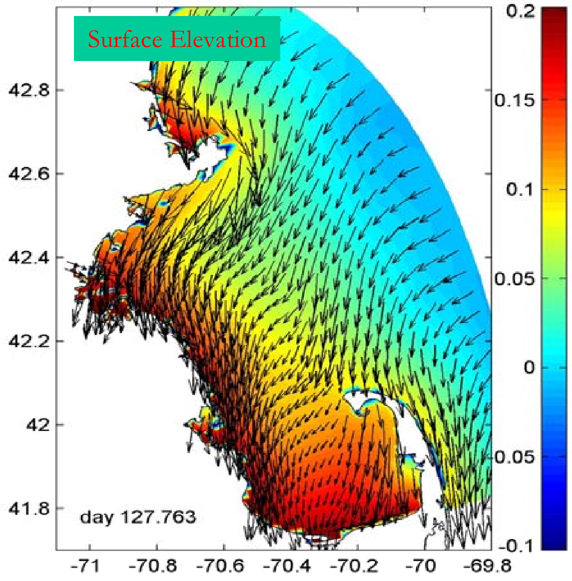

Figure 2-4. Sea surface elevation and surface currents on May 7, 2005. (Model results from M. Jiang and M. Zhou, UMB)

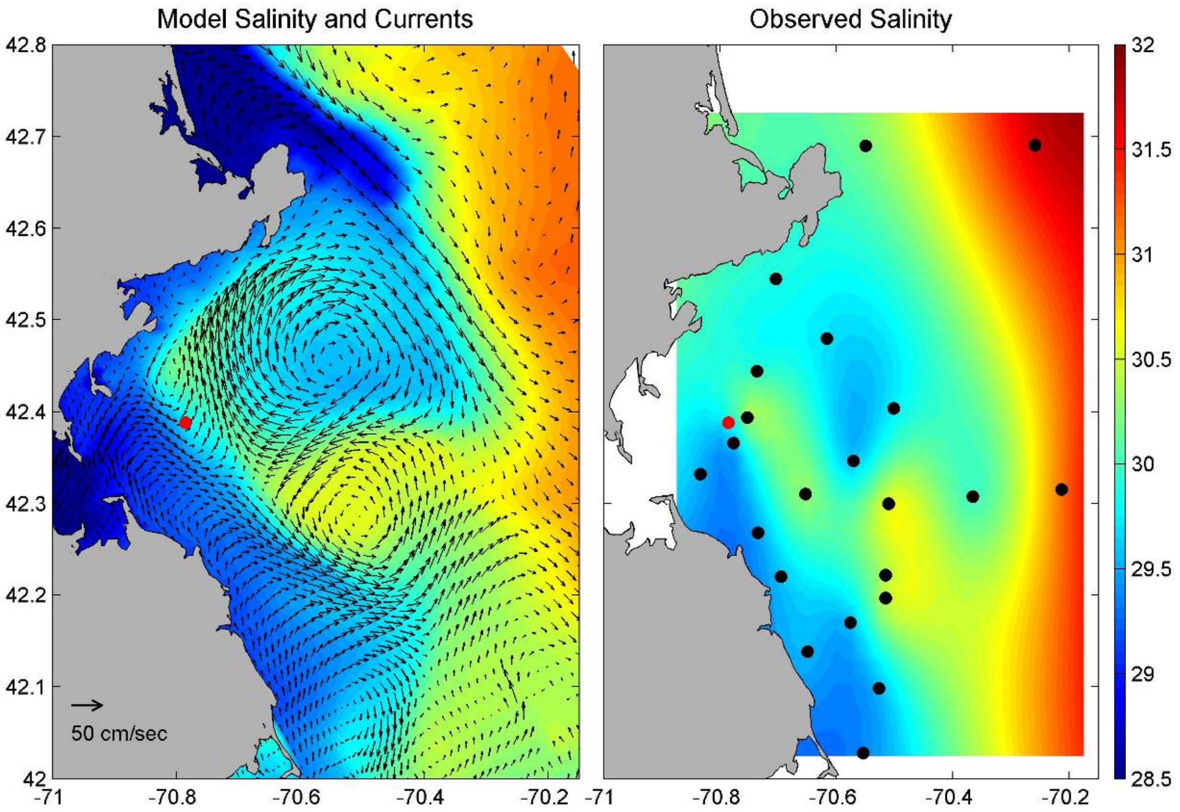

Figure 2-5. May 11, 2005 modeled salinity and currents compared to contoured observed salinity (PSU, scale to right). MWRA outfall shown as red dot and sampling locations as black dots. (Model results and plots from M. Jiang, University of Massachusetts at Boston).

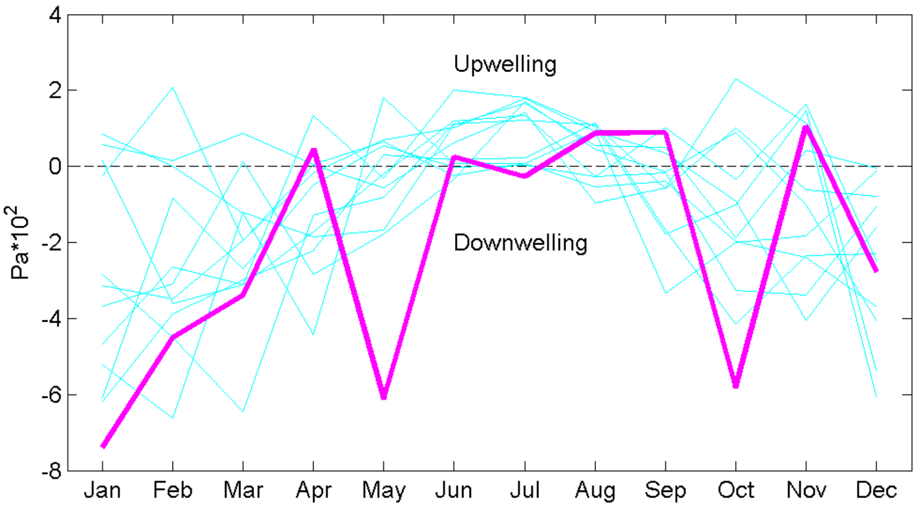

Figure 2-6. Monthly average N-S wind stress at Boston buoy for 2005 (thick red line) compared with the previous 11 years of observations (1994-2004; thin blue lines). Positive values indicate northward-directed, upwelling-favorable wind stress.

The strong winds during May also had a marked influence on the wave height during this period. Measurements at the Boston Buoy indicate wave heights greater than 5 m during the two storms in May. The month of May in 2005 was considerably rougher than in any other year during the Harbor Outfall Monitoring program, as indicated by a comparison of the mean wave height (**Figure** 2-7).

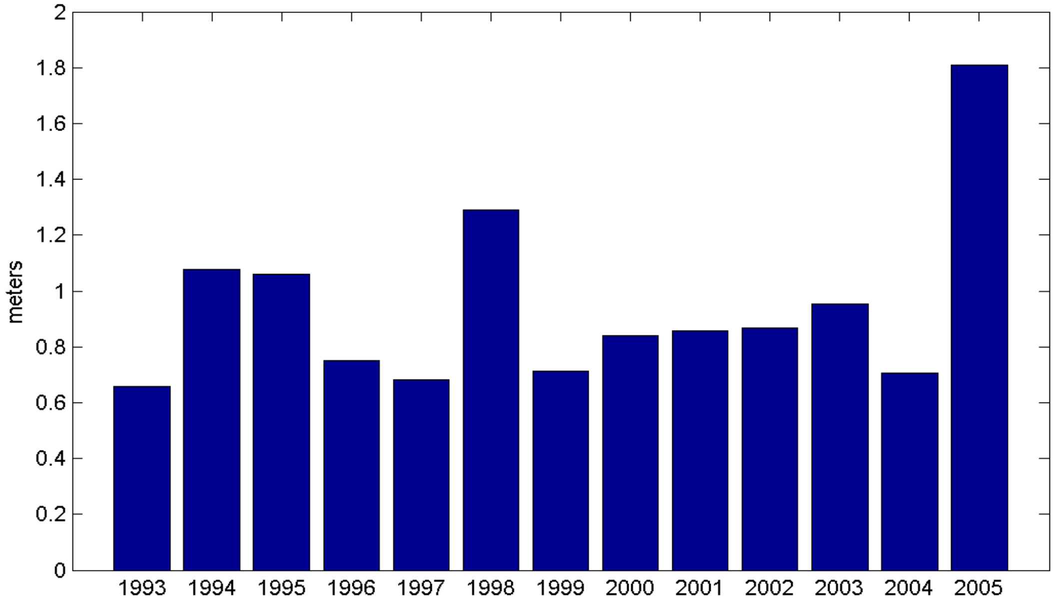

Figure 2-7. Average significant wave height in May at the Boston Buoy.

Preliminary results from an analysis ranking storms from 1990-2005 based on bottom wave stress and wind stress indicate that these May 2005 storms were substantial (B. Butman, submitted). Out of ~500 storms 'defined' by integrated wind and wave stress, the 2005 May 7-8 and 24-25 storms were #24 and #14 as ranked by wave stress, and #24 and #9 ranked by wind stress. Also, the October 2004 – May 2005 time period was the stormiest 'winter' period as defined by wave bottom stress. The second stormiest was the October 1992 – May 1993 period; it may be coincidental, but it is of note that prior to the 2005 bloom, the last major PSP outbreak in Massachusetts was in May and June, 1993 (**Figure** 1-3).

The strong wind events not only caused strong transport into Massachusetts Bay, but they also resulted in intense mixing of the source waters for the bay and in downwelling-favorable conditions. Thus, the scenario has the first northeaster storm driving Gulf of Maine waters (rich with *Alexandrium*) well into Massachusetts Bay. Runoff and strong water column mixing presumably supplied ample nutrients to fuel the bloom and abundances continued to increase as May progressed. A second powerful northeaster storm hit the area in late May again deepening the mixed layer and entraining nutrients to the surface waters throughout the area. These northeast winds pushed *Alexandrium* even further into the area, including well into Cape Cod Bay. Downwelling-favorable conditions may have further concentrated these upward-swimming motile cells in the nearshore surface waters, further exacerbating PSP shellfish toxicity.

2.2 Shellfish Toxicity

This section provides a narrative the PSP toxicity events during the 2005 *Alexandrium* bloom. The timing and geographic extent of shellfish harvesting closures is given in **Figure** 2-8, and the trends in PSP toxicity scores in **Figure** 2-9, categorized by region (excluding Maine).

The first detectable PSP toxicity of 2005 was recorded along the western Maine coast by the Maine Department of Marine Resources (DMR). Toxicity was detected unusually early in April, unusually far west along the Maine coast, and was unusually widespread among western Maine stations (D. Couture, pers. comm.). DMR closed the Maine coast south of Portland on May 4 as the levels rose near the 80 µg saxitoxin (STX) 100 g^{-1} regulatory limit.

The first high (>80) level of toxicity of 2005 (567 µg STX 100 g^{-1}) was measured one day later (May 5) 8 km offshore of New Hampshire (NH) on Star Island (part of Appledore Island) by the NH Department of Environmental Services (DES), prompting immediate closure of offshore NH islands on that day (C. Nash, pers. comm.). This was the first measurement of the season at Star Island because earlier sampling is operationally difficult. The toxicity measured at Star Island was a surprise to many observers because coastal NH had no toxicity, and coastal Maine had only low toxicity at that time.

The northeaster struck the next weekend (May 7-8). Massachusetts Department of Marine Fisheries (DMF) began sampling their northern stations (see **Figure** 1-4) on May 11 and detected toxicity north of Cape Ann. A week later, toxicity was detectable inside Massachusetts Bay at four of five stations on the western shore as far south as Sandwich (at the eastern entrance to the Cape Cod Canal). On the basis of these results plus additional information on cell abundance and winds, DMF closed the Massachusetts coast north of Cape Ann, and a day later the coast north of the Cape Cod Canal.

While toxicity remained high offshore at Star Island, inshore NH toxicity was undetectable until May 18. DES then extended the offshore closure to the remainder of the NH coast.

After the second northeaster storm on May 24-25, toxicity measured on May 26 in southern Massachusetts Bay was higher than levels measured at Massachusetts stations north of Cape Ann. Furthermore, for the first time ever, mussels at the western end of the Cape Cod Canal (which links Cape Cod Bay to Buzzards Bay) became nearly toxic (70 µg STX 100 g^{-1}), presumably from cells introduced from one bay to the other via tidal exchange through the canal. DMF subsequently extended the closure to portions of Buzzards Bay (Bourne and Wareham), eastern Cape Cod Bay, and the outer Cape nearly as far south as Chatham.

On June 1, toxicity was detected at Chatham. That result plus cell abundance data prompted the decision to close Monomoy Island (at the “elbow” of Cape Cod), a major shellfish resource and wildlife refuge. Toxin scores continued to increase throughout the region (i.e., Scituate Yacht Club with 2,116 µg STX 100 g^{-1} on June 8; New Hampshire’s Star Island reached 1,224 µg STX 100 g^{-1} on June 20), and reached closure levels on Nantucket and the eastern and southern shores of Martha’s Vineyard as well. Cell abundances were very high near these offshore islands, with toxin scores as high as 4,378 µg STX 100 g^{-1} in mussels on Martha’s Vineyard on June 17.

On June 16, the widespread cell distribution, as well as wind and drifter tracks showing the transport pathways of water and cells (see below) led federal officials to close 40,000 km^2 of offshore shellfish resources that lie outside state jurisdictions. This closure was in addition to a larger offshore closure that had been in place since 1989. In late June and early July, toxicity began to drop region-wide, and state agencies began re-opening the closed areas, with the first Massachusetts re-opening occurring on July 1. As of this writing (December 2007), portions of the federal offshore closure remain in effect.

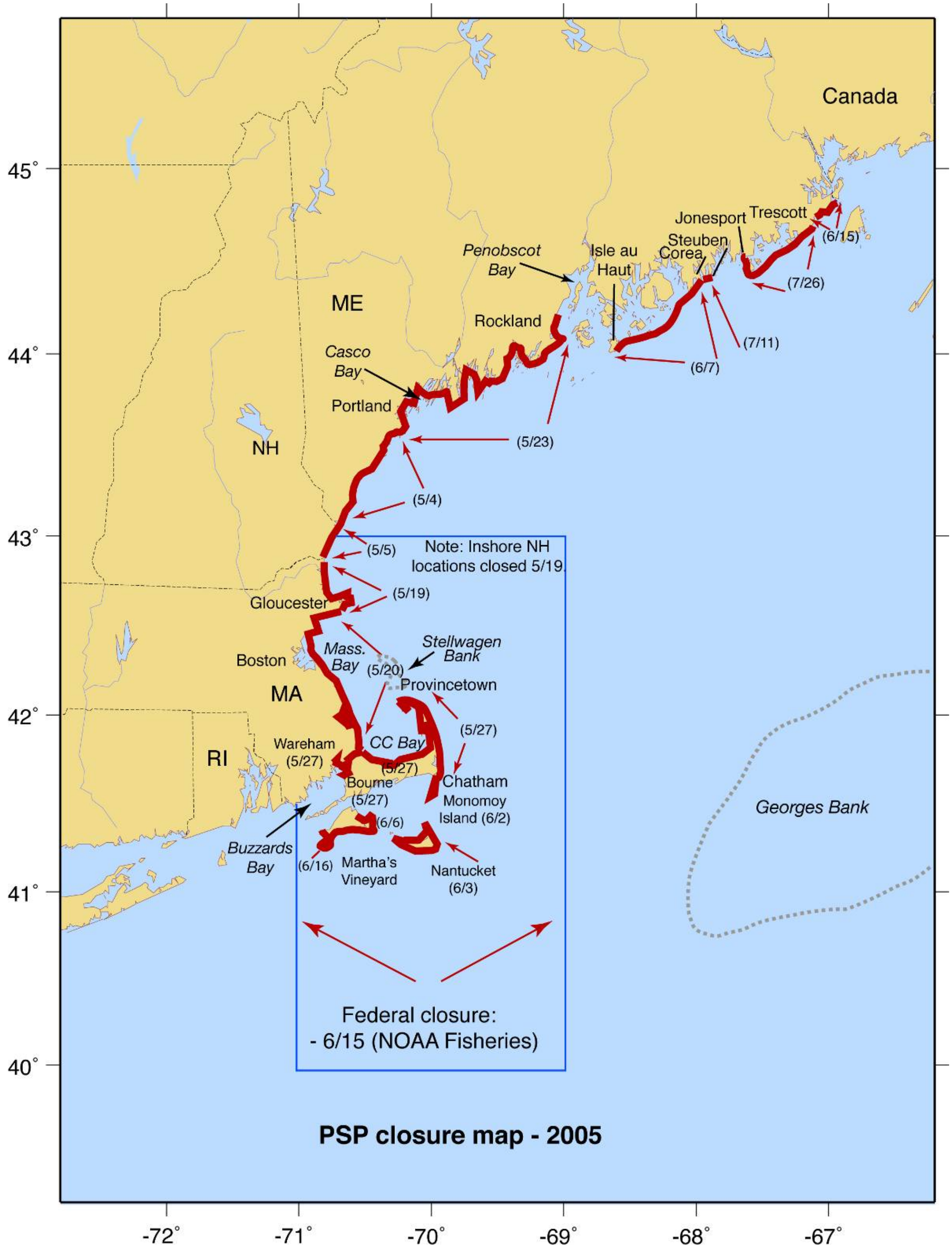

Figure 2-8. Shellfish closures along the coastlines of Maine, New Hampshire, Massachusetts, and the adjacent offshore waters due to detection of PSP toxins during the 2005 *Alexandrium* bloom. Issuance dates of these closures are indicated. (Source: Anderson *et al.* 2005c - compiled from information provided by the Massachusetts Division of Marine Fisheries, the New Hampshire Department of Environmental Services, and the Maine Department of Marine Resources.)

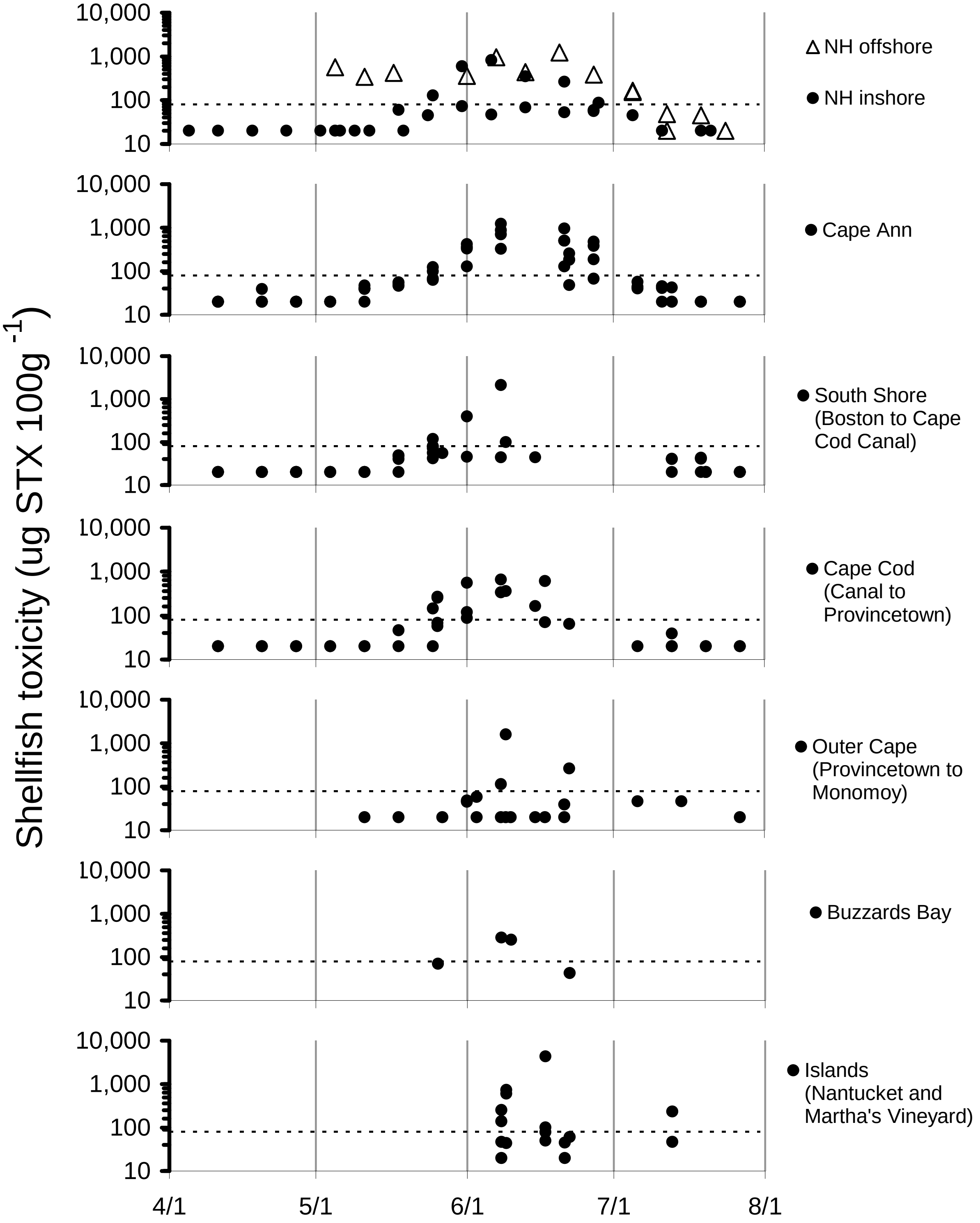

Figure 2-9. PSP toxicity in mussels from Massachusetts and New Hampshire, grouped into seven regions. Closure level of 80 µg 100g^{-1}is indicated by the dotted line. Non-detects <40 are plotted as 20.

2.3 *Alexandrium* cyst distributions

One factor contributing to the 2005 bloom may have been the presence of large numbers of fresh *Alexandrium* cysts in western Gulf of Maine sediments. Through an ECOHAB-funded project, these cysts were mapped in fall 2004, shortly after a large late-summer, early-fall bloom of *Alexandrium* that caused extensive late-season toxicity in the Gulf of Maine. As seen in **Figure** 2-10, cysts were particularly abundant in a large area offshore of Penobscot and Casco Bays – four to five-fold more abundant than in the Bay of Fundy, which has historically been viewed as the area with the largest *Alexandrium* cyst deposits in the region (Anderson *et al.* 2005b).

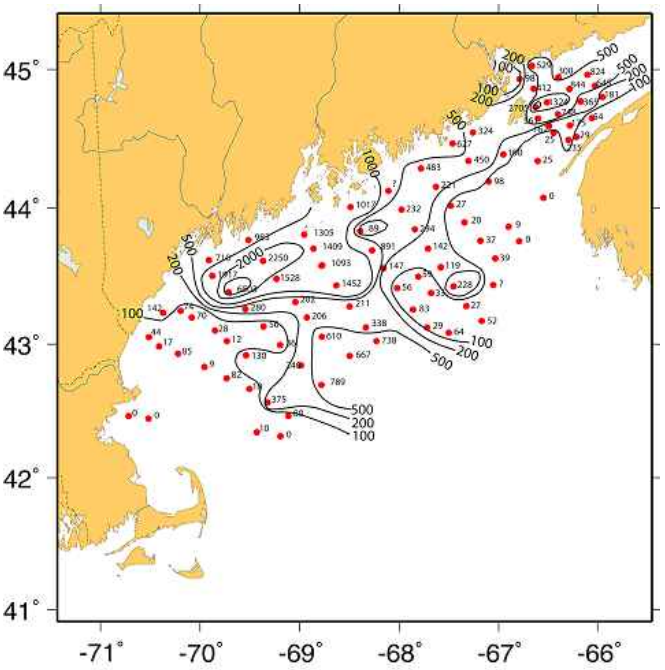

Figure 2-10. *Alexandrium* cyst abundance in the top cm of sediment (cysts cm^{-3}), mapped in October 2004. (Source: D.M. Anderson, unpublished data).

Table 2-1 shows *Alexandrium* cyst abundance in the top cm of sediment in two domains, calculated from a cyst survey conducted in 1997 (Anderson *et al.* 2005b), and another in 2004 (D. M. Anderson, unpublished data). The GOM domain represents the area from Cape Ann to Grand Manan; the Bay of Fundy is east of Grand Manan (**Figure** 1-1). Since the 1997 survey area was smaller than that for 2004 (see Figure 3-2), only the data within the 1997 survey area are used from 2004. In this way, the cyst abundances are comparable, since they are from the same area. **Table** 2-1 shows that in 2004, cyst abundance was nearly 9-fold higher in the Gulf of Maine (not including the Bay of Fundy) than it was in 1997. The increase in the Bay of Fundy was not as large (~2.3-fold), and thus the increase for the Gulf of Maine plus Bay of Fundy was 5.3-fold.

Table 2-1. Cyst abundance (total, x 10^6) in Gulf of Maine surface sediments in 1997 and 2004, and the percent increase from 1997 levels. (From Anderson *et al.* 2005c)

	GOM	Bay of Fundy (BOF)	Total cysts in GOM + BOF
1997	2.46	2.84	5.3
2004	21.6	6.68	28.3
% change from 1997	880%	240%	530%

2.4 Characterization of the 2005 *Alexandrium* Bloom

This section highlights the spatial and temporal patterns in *Alexandrium* abundance, physical parameters and nutrient concentrations during the 2005 bloom. A complete set of horizontal and vertical contour plots of all of key parameters is provided in Appendices B and C for each of the surveys conducted in May-July 2005.

A time series of the *Alexandrium* cell counts aggregated by region is presented in **Figure** 2-11. Although MWRA responded almost immediately to the potential of an *Alexandrium* bloom entering Massachusetts Bay, cell counts were already elevated (>1,000 cells L^{-1}) during the survey conducted on May 11, 2005 (AF051). Cell counts continued to increase from early to late May throughout the bays. The peak abundance in the bays was 38,565 cells L^{-1}, measured in surface waters just off Sandwich at the northern end of the Cape Cod Canal on May 28, 2005 during a WHOI survey (TI096). Surface water *Alexandrium* abundances of >30,000 cells L^{-1} were also observed in the nearfield at station N18 on May 28. Subsurface samples peaked at >30,000 cells L^{-1} three weeks later on June 18, 2005 at station 8M in central Cape Cod Bay. High *Alexandrium* abundances (>10,000 cells L^{-1}) were consistently measured within the bays and in the Gulf of Maine on nine different surveys over the three-week period from late May to mid June. *Alexandrium* abundance decreased following the mid June survey to values <3,000 cells L^{-1} during the June 22-23 survey (AF056) and <500 cells L^{-1} in late June (AF057). In July, *Alexandrium* abundances were <10 cells L^{-1} in Massachusetts and Cape Cod Bays. The general temporal patterns in *Alexandrium* abundance by region were similar to those seen for associated toxicity (see **Figure** 2-9) albeit surface water abundance peaked earlier than both subsurface abundance and toxicity.

The spatial distributions of cell counts from samples collected during the course of the bloom from early in the bloom, to its peak in late May and early June, and to its decline in late June are presented in **Figure** 2-12 to **Figure** 2-19. Concurrent salinity, temperature and nutrient distributions are also presented. A complete set of horizontal contour plots for all of the surveys, parameters and sampling depths is presented in Appendix B.

May 10-18 2005 survey data. A region-wide cruise was conducted on the R/V *Oceanus* during the early stages of the *Alexandrium* bloom from May 9-18, 2005 (Cruise OC412). Cells were already present at moderate concentrations in Massachusetts Bay (~ 1000 cells L^{-1}), and were distributed to the north as far as the Bay of Fundy in lower concentrations (**Figure** 2-12A). The cells were predominantly associated with low-salinity water close to the coast (**Figure** 2-12C), with the exception of the outer reaches of Massachusetts Bay. Temperatures were warmer in the immediate nearshore waters and within Massachusetts Bay than they were further out in the Gulf, or to the east in the EMCC and Bay of Fundy regions. The cold core of the EMCC is also evident in these data, extending alongshore and offshore from the eastern Maine coastline (**Figure** 2-12B). The salinity distribution **Figure** 2-12C) shows a continuous, alongshore band of low-salinity water (<31.4) extending from the Bay of Fundy to Massachusetts Bay. This feature resembles the Gulf of Maine Coastal Plume described by Keafer *et al.* (2005a). **Figure** 2-12D shows the concentration of nitrate plus nitrite ($NO_3 + NO_2$) over the cruise domain. The highest concentrations are in the eastern Maine region, associated with the EMCC and the outflow from the Bay of Fundy, with concentrations in excess of 8 µM. Some additional structure is seen in the western region, with a band of moderately high concentrations (2.5 – 6.6 µM) extending from Casco Bay in the vicinity of major freshwater sources (i.e. Penobscot and Androscoggin/Kennebec Rivers). Along the NH and MA coastlines, however, NO_3 was essentially depleted from inshore surface waters.

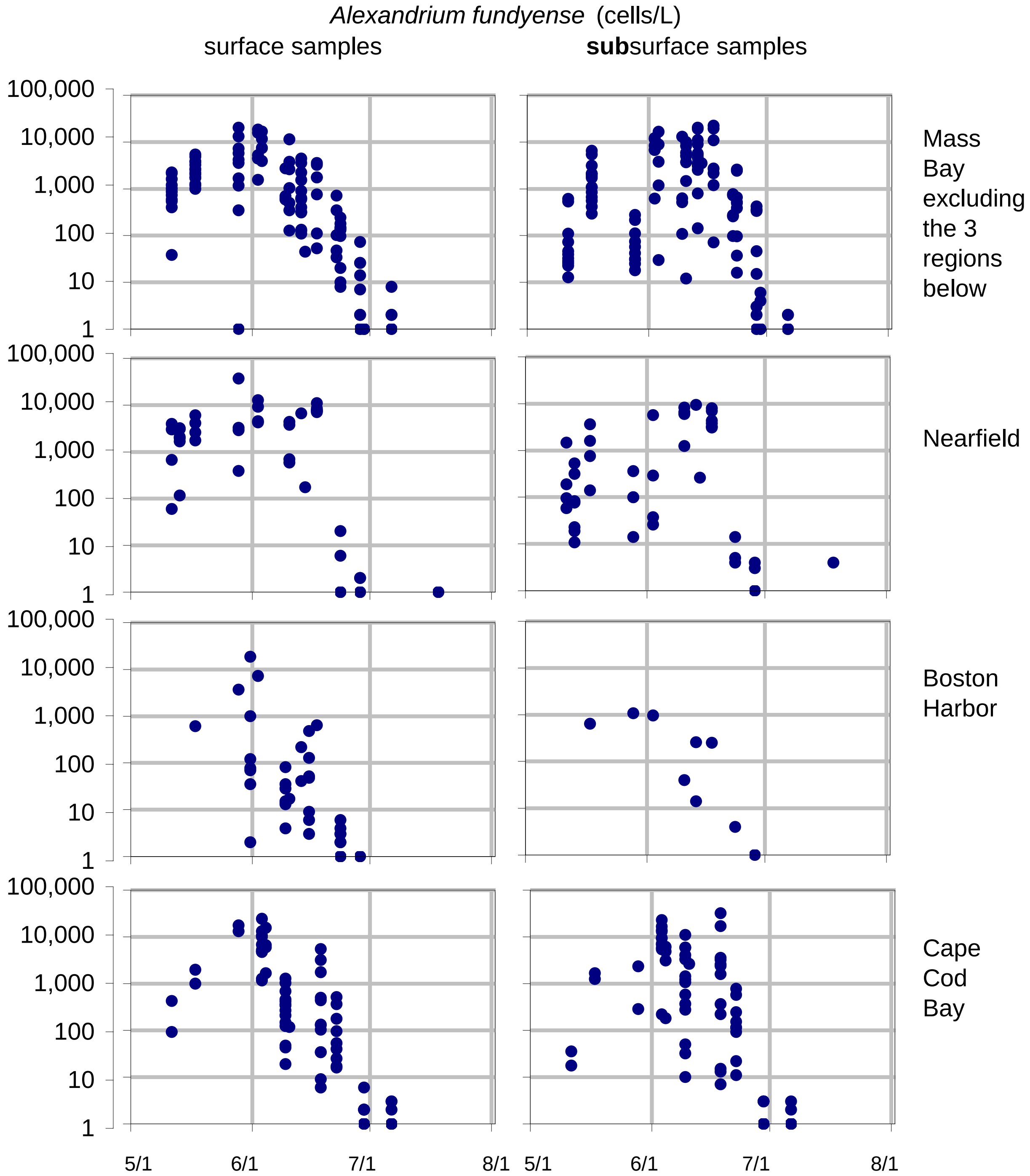

Figure 2-11. *Alexandrium* abundance+1 in surface and subsurface samples, grouped into four nonoverlapping regions ("Massachusetts Bay" here excludes the other 3 regions shown: the outfall nearfield region, Boston Harbor, and Cape Cod Bay.)

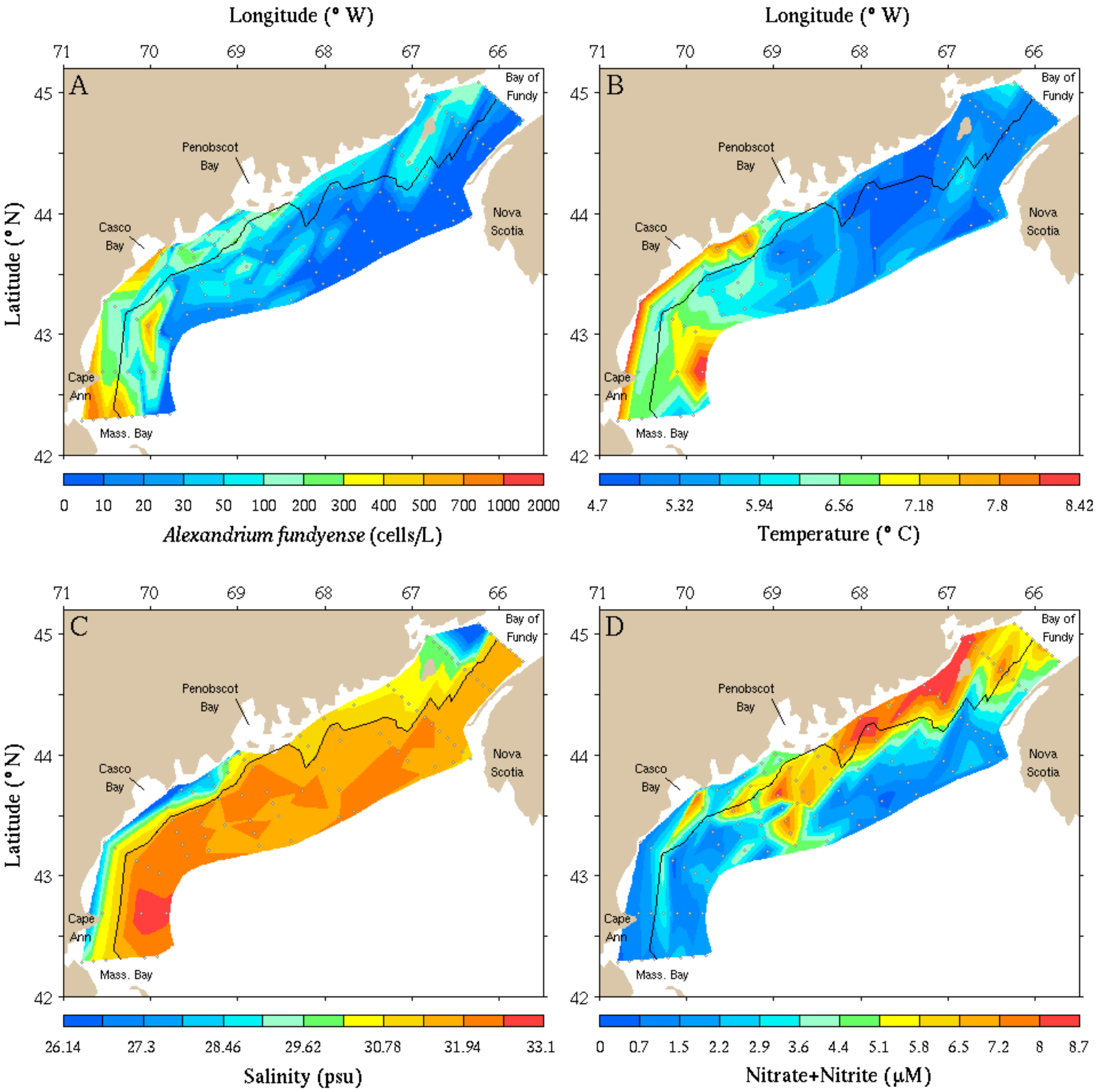

Figure 2-12. May 10–14 2005 near-surface distribution of *Alexandrium* abundance, hydrographic properties, and nutrients during initial bloom detection on R/V Oceanus cruise OC412 Leg I. (A) *Alexandrium* abundance; (B) temperature; (C) salinity; (D) nitrate and nitrite. The dark line in all panels depicts the 31.4 PSU salinity contour, indicative of the Gulf of Maine Coastal Plume. Stations are indicated by small open squares.

Focusing in on Massachusetts Bay, similarly high *Alexandrium* abundance and low NO_3 concentrations were observed within the bay during survey AF051. **Figure** 2-13 depicts the surface data from both of these coincident surveys. The distribution of *Alexandrium* cells in Massachusetts Bay at that time suggests that there was a gradient, with higher abundances in the nearfield and northern Massachusetts Bay and lower abundances to the south and further offshore, suggesting that the cells were being carried into the bay from the north by the prevailing currents. The surface waters entering the bay had low salinities and high silicate (SiO_4) concentrations (4-5 µM), consistent with an upstream, freshwater influence. The elevated SiO_4 concentrations supported a May diatom bloom of *Chaetoceros*, which was relatively abundant for this time of year (~0.5 million cells L^{-1}; see Section). Although NO_3 concentrations were low (0-2 µM) in early May, the availability of NO_3 (albeit at low levels) supported both the *Alexandrium* and diatom blooms. Surface water NH_4 concentrations were very low.

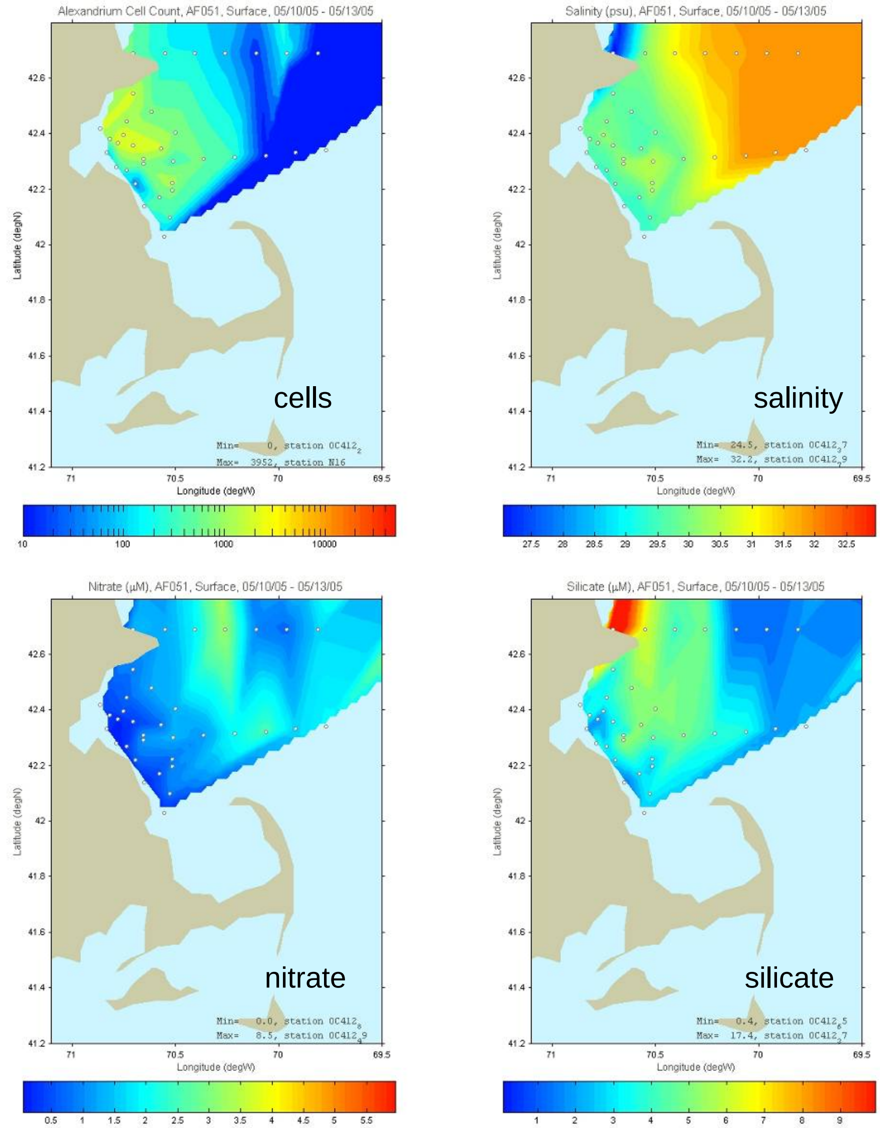

Figure 2-13. May 10-13 2005 *Alexandrium*, salinity, nitrate, and silicate in surface waters of Massachusetts Bay on surveys OC412 and AF051. Note color scale change from Figure 2-12, and log scale for *Alexandrium*.

Figure 2-14 shows the *Alexandrium* cell distributions at the surface, 10, and 20 m depths for Legs 1 and 2 of survey OC412. The second leg was conducted one week later and cell abundance had increased over this interval – approximately doubling. Cells were not only abundant in surface waters, but also at 10 and 20m, indicative of the deep mixing associated with strong storms prior to and during this interval. A survey on May 17 (AF052) showed an increase in *Alexandrium* abundances (>5,000 cells L^{-1}) in western Massachusetts Bay with surface water abundance peaking at 6,028 cells L^{-1} at station N04 in the nearfield and a subsurface maximum 6,475 cells L^{-1} at station F07 along the South Shore (Appendix B).

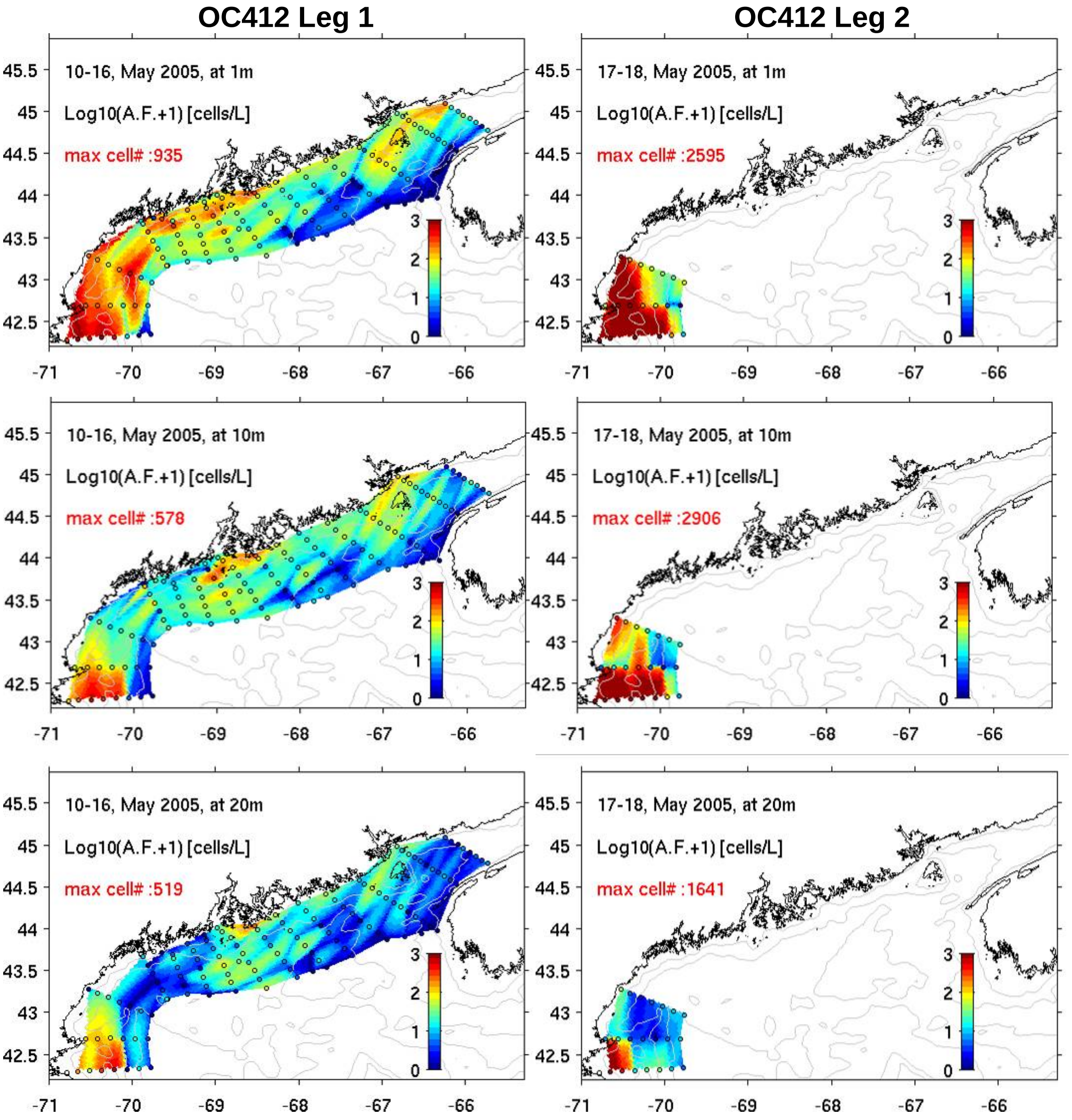

Figure 2-14. May 10-18 2005 *Alexandrium* log abundance at the surface, 10m, and 20m depths (top row, middle, bottom) on cruise OC412 legs 1 and 2.

May 27-28 2005 survey data. Another intense northeaster storm on May 24 and 25 was associated with an increase in the intensity and extent of the bloom. *Alexandrium* abundances increased dramatically with surface counts exceeding 10,000 cells L^{-1} over most of Massachusetts and Cape Cod Bays on surveys AF053 and TI096 (**Figure** 2-15). Exceptionally high cell concentrations for the southern New England region (~40,000 cells L^{-1}) were detected in samples collected in Cape Cod Bay just to the north of the canal off of Sandwich. Planozygotes, the precursors to *Alexandrium* resting cysts, were observed in these samples. Lower abundances were found further offshore in the Gulf of Maine and in Buzzards Bay. Low salinity water was evident to the northeast and along the shoreline and slightly elevated NH_4 and SiO_4 concentrations (2-4 µM) were associated with the lower salinity waters. Even though both nitrogen and silica were available, there was a marked decrease in the number of diatoms in Massachusetts Bay waters from mid to late May. Along with elevated *Alexandrium* abundances, the samples from May 28 also contained cells that are characteristic of freshwater or estuarine waters (*Dinobryon* spp., *Scenedesmus* spp., and *Asterionella formosa*).

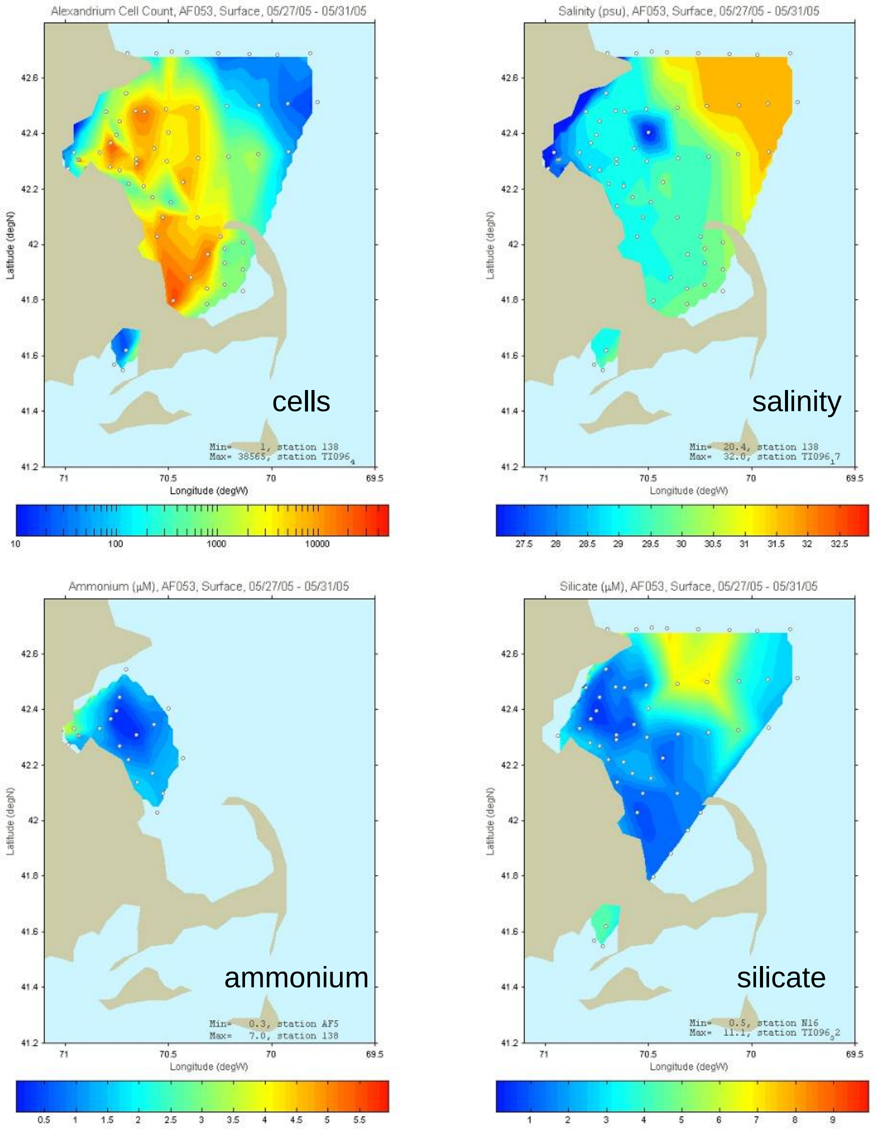

Figure 2-15. May 27-28 2005 *Alexandrium*, salinity, ammonium, and silicate in surface waters on surveys AF053 (all), TI096 (all except NH_4), and CCS1 (*Alexandrium* and salinity only).

June 2-4 2005 survey data. Subsequent expanded cruises in early June indicated that the bloom extended to the Atlantic-facing areas of outer Cape Cod and eastern portions of Nantucket Sound (**Figure** 2-16). *Alexandrium* abundances remained high (>10,000 cells L^{-1}) with peak counts >20,000 cells L^{-1} at stations in Cape Cod Bay and to the east of Cape Cod. Elevated *Alexandrium* abundances (10,000-20,000 cells L^{-1}) also continued to be observed in western Massachusetts Bay and were coincident with lower salinity surface waters (28-29 PSU). Elevated cell counts were also observed along the eastern shore of Cape Cod and to the south. Notably high cell concentrations were detected near Monomoy Island where tern mortalities reported at that time have subsequently been shown to be caused by saxitoxins (T. Leighfield, pers. comm.) Cells were observed in moderate concentrations (2,400 cells L^{-1}) in the Cape Cod Canal and the northeast end of Buzzards Bay, consistent with the harvesting closure there. Cell concentrations remained well below 1000 cells L^{-1} and shellfish areas remained open in other areas of that bay (see **Figure** 2-8). Elevated concentrations of SiO_4 were measured well offshore (5.4 µM), but were generally depleted throughout the rest of the bays.

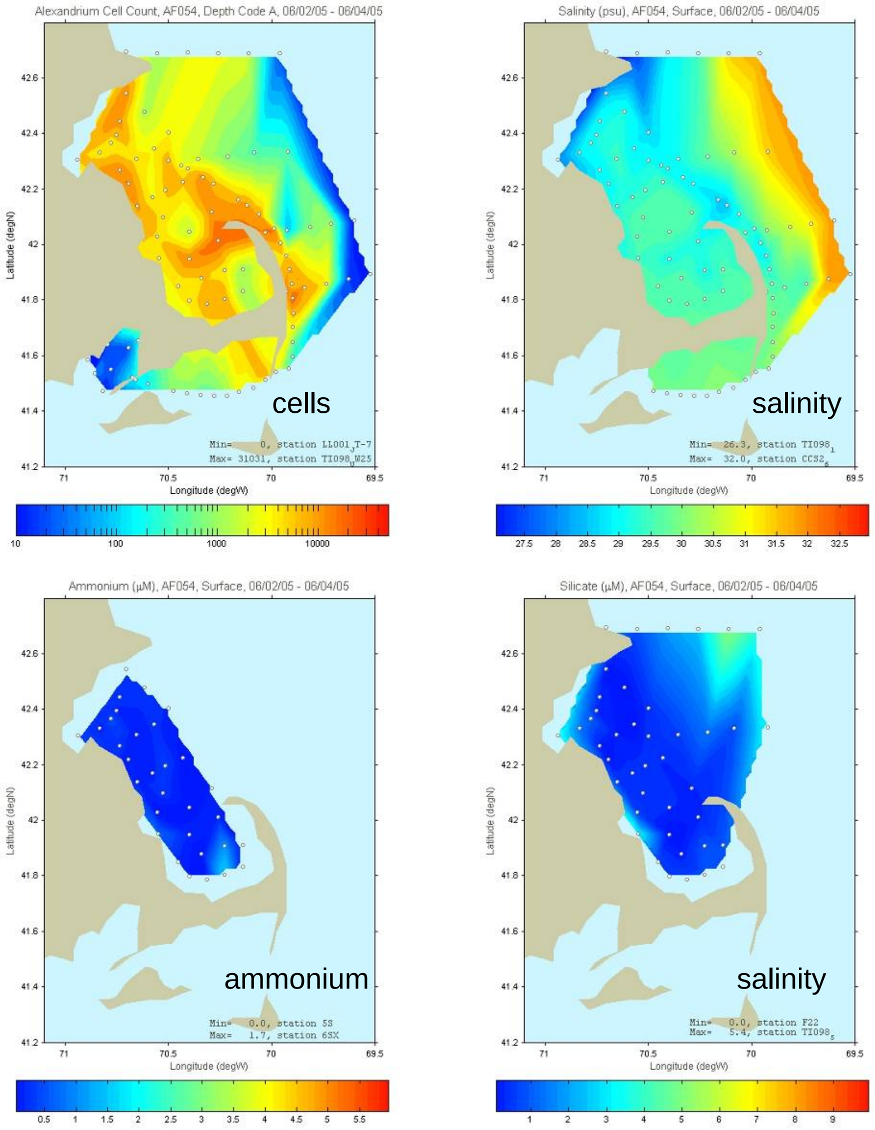

Figure 2-16. June 2-4 2005 *Alexandrium*, salinity, ammonium, and silicate in surface waters on surveys AF054 (all), TI098 (all except NH_4), and CCS2 (*Alexandrium* and salinity only)

June 9-10 2005 survey data. Figure 2-17 shows results of coincident (June 9-10) surveys conducted in Massachusetts and Cape Cod bays, offshore waters and Buzzards Bay. There was a decline in *Alexandrium* abundance in the bays with only one surface sample being >10,000 cells L^{-1} (at station F17 ESE of the nearfield) while most surface water abundances had dropped to <1,000 cells L^{-1}. The highest *Alexandrium* abundances (>60,000 cells L^{-1}) were once again documented in offshore waters east of Cape Cod on June 9. Elevated abundances (>3,000 cells L^{-1}) observed in the nearfield and to the east and northeast were associated with a water mass with lower salinity (28.5-29.5 PSU). Salinity values of 27-28 PSU were measured in the surface waters north of Cape Ann. This low salinity water to the north was coincident with *Alexandrium* abundances of >12,000 cells L^{-1}. These waters were also virtually depleted in nutrients as shown for NO_3 and SiO_4 in **Figure** 2-17. The June 9-10 data suggest movement of surface water into the bays from the north contributing to the low salinity and the elevated *Alexandrium* abundances that were measured. Note, however, that the higher *Alexandrium* abundances were observed at 10 m depth than at the surface in the bays during this survey perhaps due to the greater availability of nutrients at depth (Appendix B, **Figure** B-5).

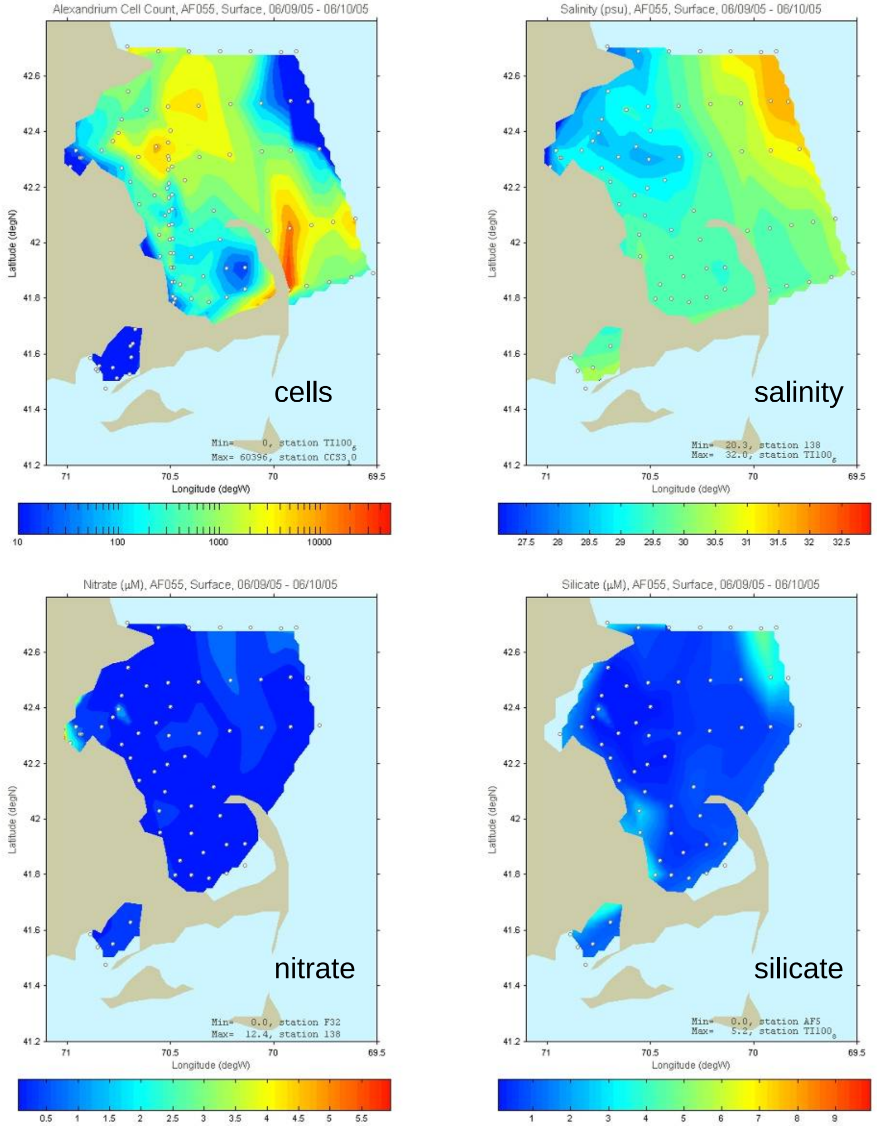

Figure 2-17. June 9-10 2005 *Alexandrium*, salinity, nitrate, and silicate in surface waters on surveys AF055 (all), TI098 (all), and CCS3 (*Alexandrium* and salinity only)

June 13-18 2005 survey data. Figure 2-18 shows that in mid-June, concentrations of *Alexandrium* continued to decrease in the surface waters of Massachusetts Bay with a maximum of 10,919 cells L^{-1} measured at station N10 in the western nearfield area. This decrease was coincident with an increase in counts in the 10-m depth samples, which peaked at 32,094 cells L^{-1} in central Cape Cod Bay. Elevated abundances of 10,000 to 20,000 cells L^{-1} were found in subsurface samples throughout much of the bay. High counts also were observed to the south of both Nantucket and Martha's Vineyard islands, and very low abundances to the west, south of Narragansett Bay (~10 - 100 cells L^{-1}). The *Alexandrium* bloom extended into waters far from the mainland coast and it was truly a widespread bloom. Silicate and nitrate were generally depleted at surface and 10-m sampling depths (Appendix B, **Figure** B-6) though present at moderate concentrations in the vicinity of Boston Harbor (2-4 µM). Ammonium concentrations were slightly higher (~2 µM) in the harbor and nearfield surface waters and also at depth in this same area and to the south.

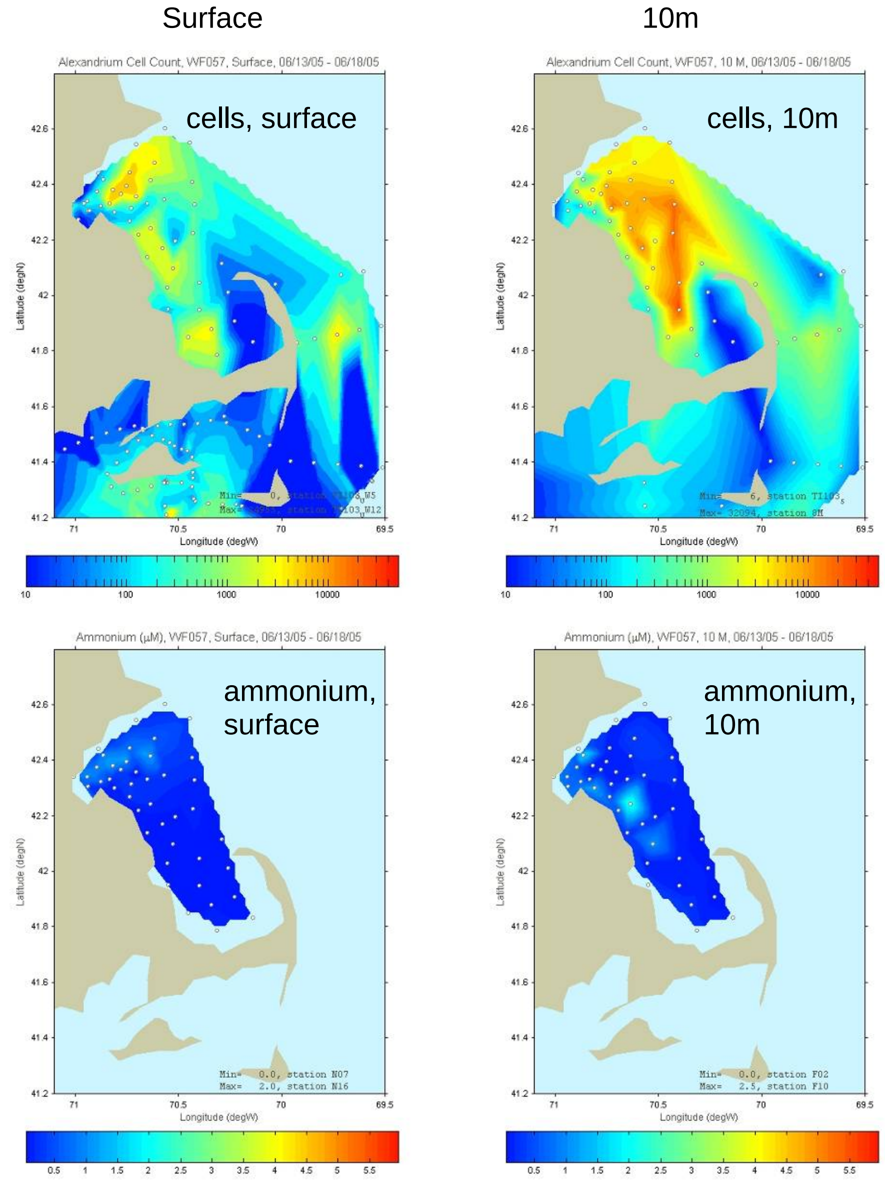

Figure 2-18. June 13-18 2005 *Alexandrium* and ammonium in surface and ~10 m waters on surveys WF057 (all) and BH3, CCS4, and TI103 (*Alexandrium* only).

From early to mid June, the bloom spread further south towards Nantucket and Martha's Vineyard, which were closed to shellfishing in early June. With those closures, more than 3/4 of the productive shellfish beds (~ 2400 km^2) in the state of Massachusetts were closed due to presumed high levels of PSP toxin. All of the New Hampshire coast, and more than half of the Maine coastline, was closed as well (**Figure** 2-8). Cell counts for a transect along the border between Massachusetts and Rhode Island (data not shown) revealed that *Alexandrium* cells were present at low concentrations nearshore (~100 cells L^{-1}), and slightly less abundant offshore (~10 cells L^{-1}) in mid June. One week later, the distribution changed, as offshore cell concentrations along the border were in the 300-400 cells L^{-1} range, while nearshore concentrations fell to 10 cells L^{-1} or below. This was likely due to the advection of the Martha's Vineyard offshore population (>10,000 cells L^{-1} in mid June) into the offshore waters of Rhode Island.

June 28-29 2005 survey data. Figure 2-19 shows that *Alexandrium* abundances declined rapidly over the course of the next several weeks and by the end of June were <10 cells L^{-1} in both surface and subsurface waters throughout most of Massachusetts and Cape Cod Bays. Surface water values of >10 cells L^{-1} were only found at the stations in northeastern Massachusetts Bay and were even higher (1,000 cells L^{-1}) further offshore east of Stellwagen Bank in the Gulf of Maine. High abundances (10-20,000 cells L^{-1}) were observed at several stations at 20 m depths at offshore stations east of Cape Ann and Stellwagen Bank. In late June surface salinity was low, typical of input from the Gulf of Maine south of Cape Ann. Nutrients were generally depleted from the surface to 10-m waters throughout the bays (Appendix B).

July 6 & 18 survey data. *Alexandrium* abundances during the July surveys were all <10 cells L^{-1}. These results combined with declining toxicity in Massachusetts and Cape Cod Bays as reported by MA DMF indicated that the bloom was effectively over in southern New England. As the bloom was ending, the MWRA ARRS monitoring continued, to examine if cell abundances might remain elevated near the outfall due to the continued availability of effluent-derived nutrients. The results, however, showed no apparent outfall effect causing a localized increase in growth or duration of the *Alexandrium* bloom.

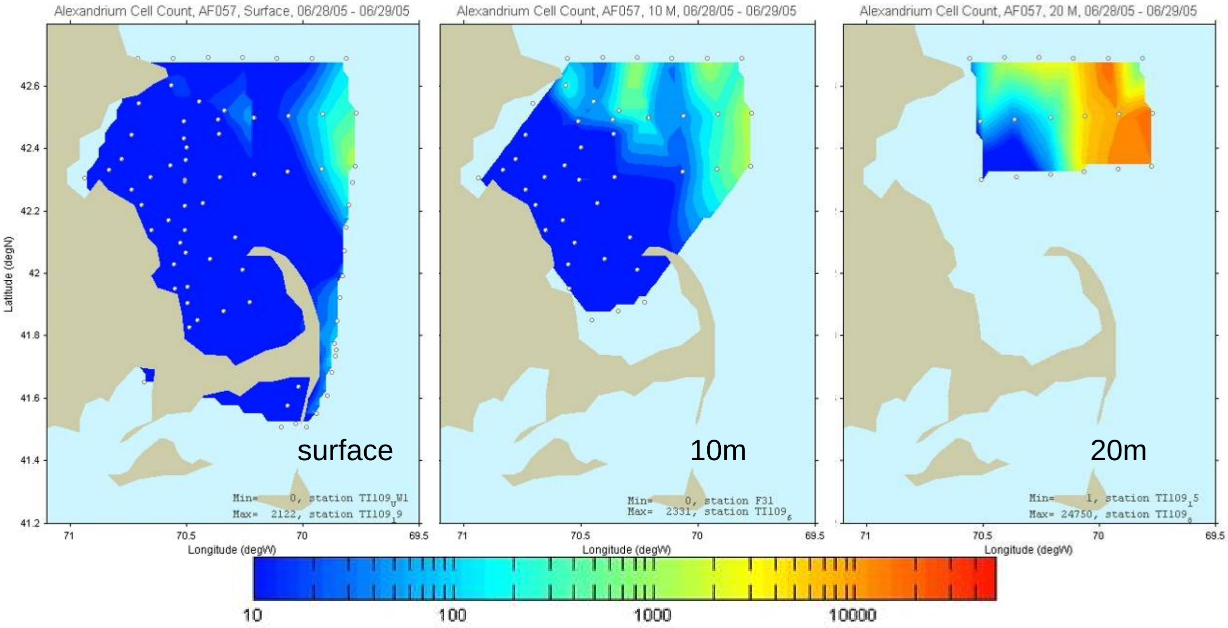

Figure 2-19. June 28-29 2005 *Alexandrium* in surface, 10-m, and 20-m waters on surveys AF057 and TI109.

2.5 Drifter Observations

The southerly transport pathway of the cells causing this bloom, as well as the chaotic nature of the flow field can be inferred from the track of three surface drifters released from cruise OC412 near Cape Ann on May 9 (**Figure** 2-20). This release was coincident with the initial detection of high concentrations of *Alexandrium* in Massachusetts Bay and just after the first northeaster storm. While the three units were deployed within a few hours of each other and were separated by less than ten kilometers, their pathways were significantly different. One traveled along the eastern flank of Stellwagen Bank, down the outer Cape and entered Nantucket Sound, carving a pathway between Nantucket and Martha's Vineyard. This matches the pattern and timing of nearshore toxicity extremely well. Other drifters followed an offshore trajectory, with both traveling east of Stellwagen Bank, after which one was transported down the Great South Channel, while the other headed east towards Georges Bank.

It is notable that none of these drifters entered Massachusetts Bay, where very high *Alexandrium* cell concentrations were found. However, this drifter release occurred subsequent to the first northeaster storm, after the period of transport conditions favorable for entry into the bay. By the time of the next storm with winds from the northeast, the drifters had already passed by the potential entry point to the bay. **Figure** 2-21 shows a surface drifter released south of Martha's Vineyard by WHOI on June 22, 2005 in waters with 3,600 cells L^{-1} of *Alexandrium*. The drifter moved to the northwest and was off of Narragansett Bay by early July (a concern to Rhode Island officials, but this did not result in any shellfish toxicity) before heading south of Long Island for the rest of the summer. Another set of drifters, one surface and one 3-m-deep subsurface drogue, was released in the outfall nearfield area on June 28. The surface drifter was east of Stellwagen Bank within 3 days and continued out to sea toward Georges Bank. The subsurface drogue also headed east initially before heading south into Cape Cod Bay from July 2 to July 10, 2005 (**Figure** 2-21).

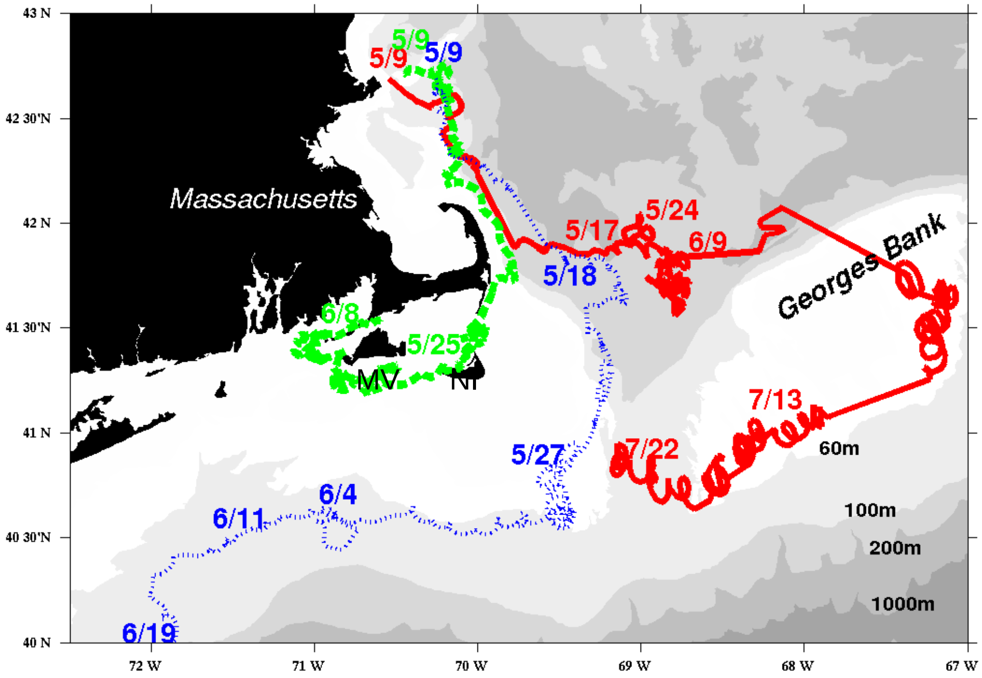

Figure 2-20. Trajectories of three surface drifters released during cruise OC412 near Cape Ann on May 9, 2005. For reference, several dates are labeled along each drifter track. Bathymetry is shown as different shades of gray (white is shallow and dark is deep). MV—Martha's Vineyard; NI—Nantucket Island. Source Anderson *et al.* 2005c - data courtesy of Jim Manning (NOAA, Northeast Marine Fisheries Science Center, Woods Hole).

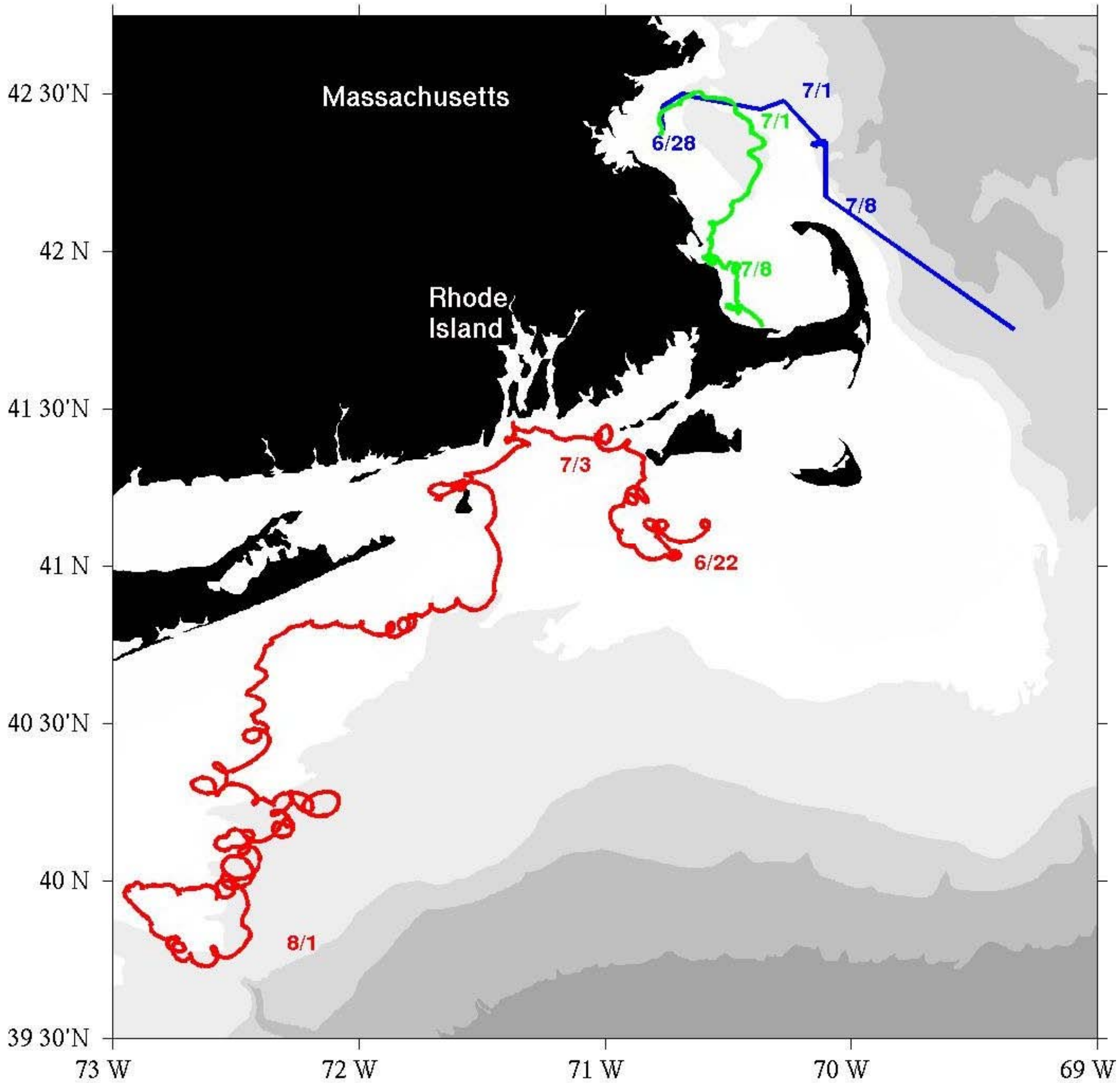

Figure 2-21. Trajectories of three drifters, one released on June 22 south of Martha's Vineyard (WHOI survey TI105; red - surface) and two released on June 28 near the bay outfall (Battelle survey AF057; blue – surface and green – 3-m drogue). For reference, several dates are labeled along each drifter track. Bathymetry is shown as different shades of gray (white is shallow and dark is deep). Data courtesy of Jim Manning (NOAA, Northeast Marine Fisheries Science Center, Woods Hole).

2.6 Phytoplankton community structure

Whole-water phytoplankton samples were also collected during the MWRA rapid response surveys to characterize the community structure and thus put the *Alexandrium* bloom into an ecological context. The *Alexandrium* bloom occurred within the seasonal succession of other, much more abundant (hundreds of thousands to millions of cells L^{-1}), species of phytoplankton. The nature of interactions between *Alexandrium* and other phytoplankton species during the initiation, maintenance and termination of blooms is not well understood, but a description of the phytoplankton communities observed during the 2005 bloom follows.

2.6.1 Temporal Succession of Phytoplankton

Phytoplankton sampling was conducted on 12 surveys in Massachusetts and Cape Cod Bay from April to July 2005, before, during, and after the red tide bloom. A total of 18 stations were sampled and a total of 154 phytoplankton samples were collected, analyzed and included in this analysis and 105 phytoplankton species or groups were identified.

Multivariate analyses were used to 1) identify patterns of phytoplankton community variation and 2) identify the variables (phytoplankton species/groups) most influencing the observed patterns of phytoplankton community variation preceding, during, and after the 2005 red tide event. The statistical analysis software Primer (Plymouth Routines In Multivariate Ecological Research; Clarke and Gorley 2001) was used for multivariate analysis. Specifically, the entire 105 species/groups by 154 sample data matrix was first analyzed by calculation of a similarity matrix (Bray-Curtis dissimilarity matrix), on log (x+1) transformed and standardized data and application of multidimensional scaling (MDS) analysis to identify patterns of all phytoplankton samples. The goal was to identify dominant patterns of variation (spatial vs. temporal) in all samples. Once the dominant modes of variation were identified, the samples were binned by cluster or MDS grouping and analyzed by a procedure (SimPer) that identified the percentage each phytoplankton species/group contributes to the discrimination of each identified group. The goal was to identify the number of variables (phytoplankton species/groups) responsible for the dominant pattern of Massachusetts Bay phytoplankton variation during the April to July 2005 period. Once identified, the abundance of dominant phytoplankton species/groups were plotted to describe patterns in their abundance.

Cluster and MDS analyses showed that temporal variation of community composition (**Figure** 2-22) was much greater than either horizontal (station to station) or vertical (surface vs. mid-depth) variation (data not shown). The statistical indicator of MDS goodness-of-fit is referred to as the stress value. The low value (0.15) indicates that the similarity matrix (in this case dissimilarity) was reasonably represented by the two-dimensional plot. The dissimilarity between samples in time (i.e., comparing April phytoplankton community to July community) was greater than dissimilarity based on either station or depth. This interpretation is consistent with the observed succession from a late winter-spring *Phaeocystis*-dominated community (April) to a diatom bloom (May) to dinoflagellate bloom (June).

The phytoplankton data were binned and averaged by survey date and data were reanalyzed as a 105-species by 12-survey matrix. Samples now clustered into four distinct groups by survey date (**Figure** 2-23): a single survey in April, a mid-May group, a late-May to early June group (coincident with the peak of *Alexandrium* abundance), and a late June to July group. The stress value of 0.04 indicates the matrix is well represented by this two-dimensional plot.

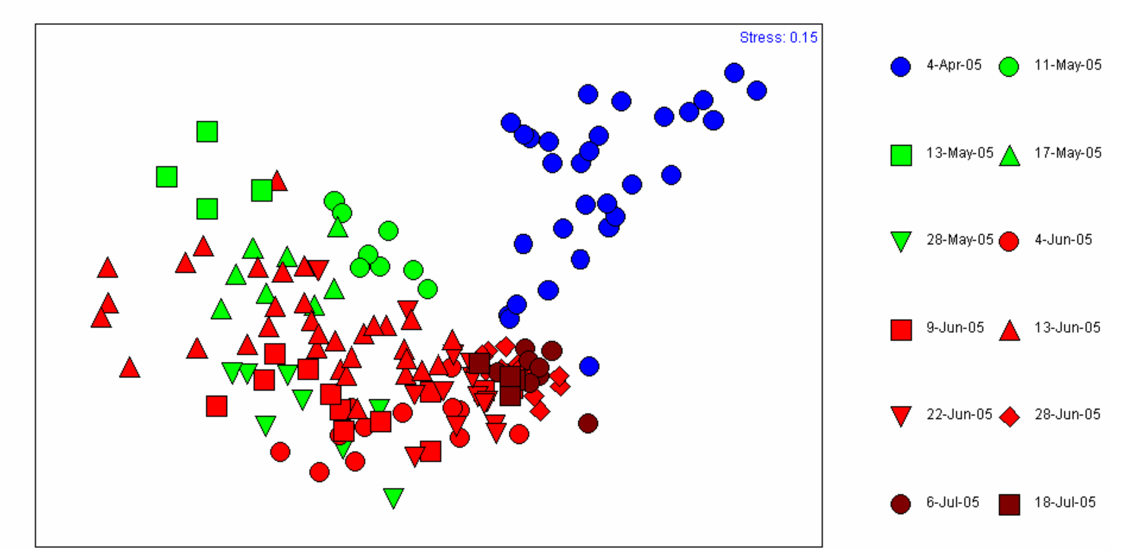

Figure 2-22. Multidimensional scaling (MDS) plot of all phytoplankton samples labeled by survey date. Note smooth transition in roughly counterclockwise pattern from April (blue; upper right) to May (green), June (red) and then July (brown). (Date of first day is given for multi-day surveys).

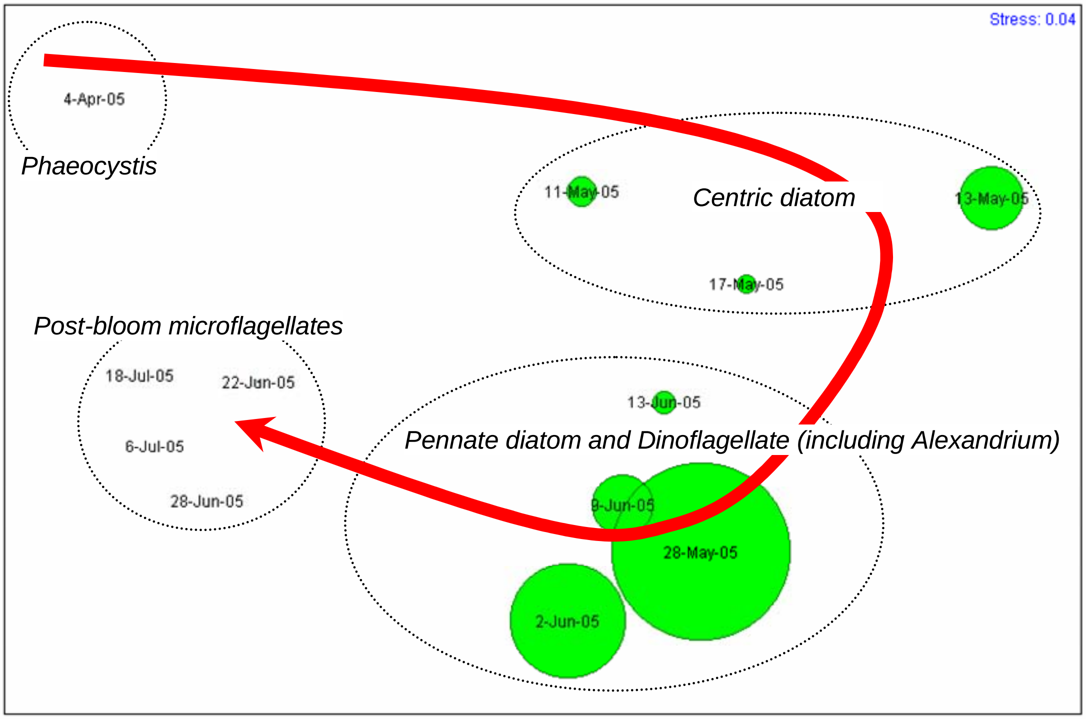

Figure 2-23. MDS plot of phytoplankton samples labeled by survey date showing sorting of samples into 4 distinct temporal groups : Upper left (April); Upper right (early May); lower right (late May – mid June); and lower-middle left (late June to July). Red arrow emphasizes that temporal sequence. *Alexandrium* abundance bubbles (green) overlaid to illustrate *Alexandrium* bloom peak in mid May to mid June. Maximum *Alexandrium* abundance (on 28 May 2005) corresponds to 9,900 cells L^{-1}.

These MDS four groupings qualitatively corresponded to observed shifts seen in **Figure** 2-24**,** a time series graph of phytoplankton community composition from April to July 2005. In April *Phaeocystis* was dominant (MDS group 1). After the first Northeaster storm on May 7-8, a centric diatom was dominant (MDS group 2); *Alexandrium* was present at moderate levels. Then after the May 24-25 Northeaster, pennate diatoms and dinoflagellates were dominant (MDS group 3); *Alexandrium* peaked in late May. By late June, these groups had all declined and the community was dominated by microflagellates (MDS group 4). This pattern is characteristic of temperate coastal phytoplankton community development in the winter-spring to early summer (Smayda 1980).

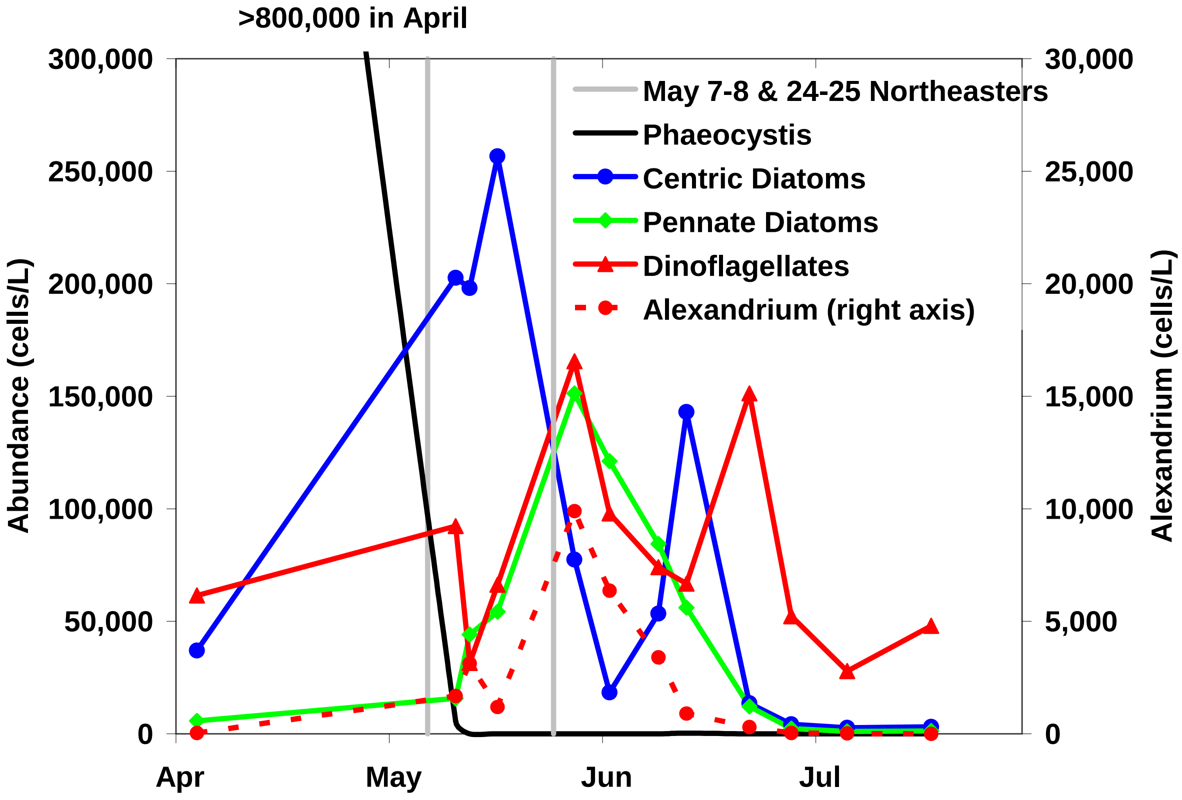

Figure 2-24. Succession of dominant functional phytoplankton groups during the April to July 2005 *Alexandrium* bloom. Survey-average abundance of all phytoplankton samples.

The MDS plots help to identify the transition dates (within the confines of periodic sampling) at which the phytoplankton community changes from one successional stage to the next. The SimPer procedure (Clarke and Gorley 2001) was used to further quantify which species/phytoplankton groups were most important in defining each of these four successional groups. **Table** 2-2 summarizes the results of the SimPer procedure and lists the dominant species or phytoplankton groups during each of the four successional stages. Note that throughout the April to July period, variation in microflagellate abundance was only 1.2-fold, thus relatively consistent. The entire April to July period was also relatively consistent in dinoflagellate abundance, with only a 1.6-fold variation in total dinoflagellate abundance between successional stages. Most of the variation in phytoplankton community composition during April-July 2005 was in *Phaeocystis* abundance (early April), centric diatom abundance which displayed ~40-fold variation in abundance between its peak (in early May) and it nadir in late June – July, and in pennate diatoms which displayed a 25-fold variation between the peak during a *Thalassionema nitzschioides* bloom in early June and the nadir in late June – July.

Table 2-2. Summary of dominant species and phytoplankton functional groups during four phases of phytoplankton succession related to the 2005 *Alexandrium* bloom as identified by SimPer, with corresponding abundance. Abundance of *Alexandrium* included for reference; it was never a community dominant. Bold highlights maximum abundance period for each species or group. Units are cells L^{-1}.

type	Spp. or Spp. group	April Pre-bloom (*Phaeocystis*)	11-17 May Early bloom (Centric Diatoms)	28 May-13 June Peak bloom (Pennates + Dinoflagellates)	22 June-18 July Post bloom (Microflagellates)
other	*Phaeocystis pouchetii*	**871,723**	2,246	157	0
centric	*Chaetoceros socialis*	**19,928**	9,145	2,223	0
centric	*Chaetoceros debilis*	4,120	**169,524**	32,223	2,536
centric	*Thalassiosira* sp. group 3 10-20 um	1,547	**10,465**	2,985	847
centric	*Thalassiosira rotula*	319	6,224	**22,684**	8
centric	*Chaetoceros* sp. group 2 10-30 um	494	8,877	**11,616**	1,710
pennate	*Thalassionema nitzschioides*	3,059	24,469	**60,401**	3,327
pennate	*Pseudo-nitzschia delicatissima* complex	1,134	9,243	**16,364**	688
dino	*Alexandrium fundyense*	35	1,149	**3,442**	127
dino	*Heterocapsa rotundata*	33,266	16,953	**16,994**	11,050
dino	*Heterocapsa triquetra*	75	875	**6,849**	3,760
dino	*Gymnodinium* sp. <20um	20,576	38,310	48,507	**61,983**
micro	*Cryptomonas* sp. <10 um	101,731	113,332	138,989	**219,752**
micro	unidentified micro-phytoflag <10 um	955,247	910,152	879,190	**1,022,091**
centric	Total Centric Diatoms	37,014	**219,137**	73,076	5,912
dino	Total Dinoflagellates	61,534	63,320	**101,069**	69,846
pennate	Total Pennate Diatoms	5,693	38,060	**103,192**	4,207
	Total Phytoplankton	**2,049,915**	1,344,070	1,335,495	1,339,196

At its peak mean abundance encountered in this analysis (9,900 cells L^{-1}; cruise average on 28 May 2005), *Alexandrium* comprised only 0.65% of total phytoplankton, 1.8% of non-microphytoflagellate phytoplankton and 6.0% of dinoflagellates numerically. [Note that much higher individual station *Alexandrium* abundance levels were observed in Cape Cod Bay (40,000 cells L^{-1} on 28-29 May 2005; Anderson *et al.* 2005c) and Massachusetts Bay (30,900 cells L^{-1} station N18, 2 meters, 28 May 2005) during the May-June 2005 bloom.] Even at the individual station maximum bloom level (30,900 cells L^{-1}), *Alexandrium* comprised ~3% of total phytoplankton, 5.5% of non-microphytoflagellate phytoplankton and 16% of dinoflagellates numerically. Clearly the 2005 *Alexandrium* bloom occurred in the context of a diverse Massachusetts Bay phytoplankton community. Below, 2005 Massachusetts Bay phytoplankton patterns are compared to observed long-term (1992-2005) patterns.

2.6.2 2005 Phytoplankton Compared to Long-term Patterns

Figure 2-25 compares the 2005 Massachusetts Bay phytoplankton pattern to the patterns seen during MWRA monitoring during 1992-2005 using MWRA nearfield data. Total phytoplankton was elevated compared to baseline during February 2005, with mean nearfield total phytoplankton abundance during late February (1.19 x 10^6 cells L^{-1}) that was twice the baseline mean level of 0.56 x 10^6 cells L^{-1}. This elevated value was largely due to elevated microphytoflagellate abundance and a bloom of the centric diatom *Thalassiosira nordenskioeldii*. Microphytoflagellate abundance during February – April 2005 was ca. 2-fold (early February, March and April) to 3-fold (late February) the baseline level of ca. 0.6 x 10^6 cells L^{-1}. By March, and through October 2005, total phytoplankton levels had declined to near or below baseline mean levels (**Figure** 2-25).

Massachusetts Bay diatom abundance was reduced to ca. 10% of baseline levels in early February 2005, with ca. 15,000 diatom cells L^{-1} versus a long-term mean value of 123,000 cells L^{-1} (**Figure** 2-26). This was the second-lowest early February value recorded during 1994-2005; early February 2004 nearfield diatom abundance (11,000 cells L^{-1}) was slightly lower. By late February, during the *Thalassiosira nordenskioeldii* bloom, diatom abundance (377,000 cells L^{-1}) had recovered to slightly above the long-term mean level (316,000 cells L^{-1}). Diatom abundance during March 2005 (162,000 cells L^{-1}) was only ca. 50% of the long-term mean level (360,000 cells L^{-1}), and by April 2005, during the presumed *initiation* of the *Alexandrium* bloom and one month *prior* to the peak of the *Alexandrium* bloom, nearfield diatom abundance (17,000 cells L^{-1}) was <10% of the long-term April value (276,000 cells L^{-1}). During May and June 2005 diatom abundance (250,000 to 300,000 cells L^{-1}) was ca. 50% of the long-term mean abundance level (400,000 to 600,000 cells L^{-1}). Massachusetts Bay diatom abundance remained relatively low for the remainder of 2005, with nearfield diatom abundance levels during July through October 2005 that were among the lowest observed during the 1992-2005 monitoring.

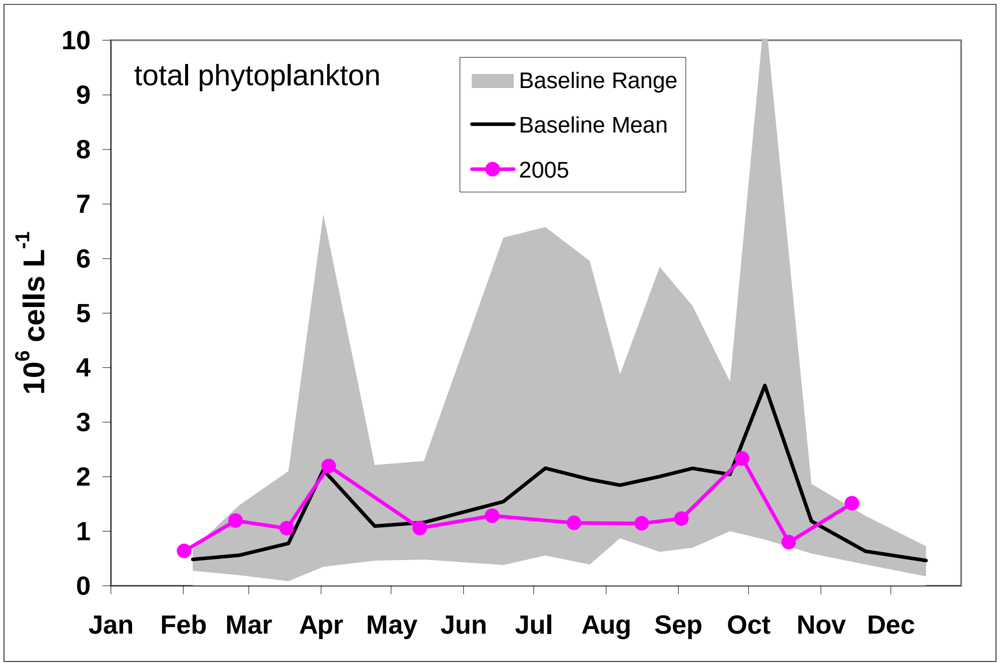

Figure 2-25. Total phytoplankton pattern during 2005 compared to the 1992-2000 baseline mean and range of values. Values are average of samples collected in the 'nearfield' area of Massachusetts Bay during MWRA monitoring.

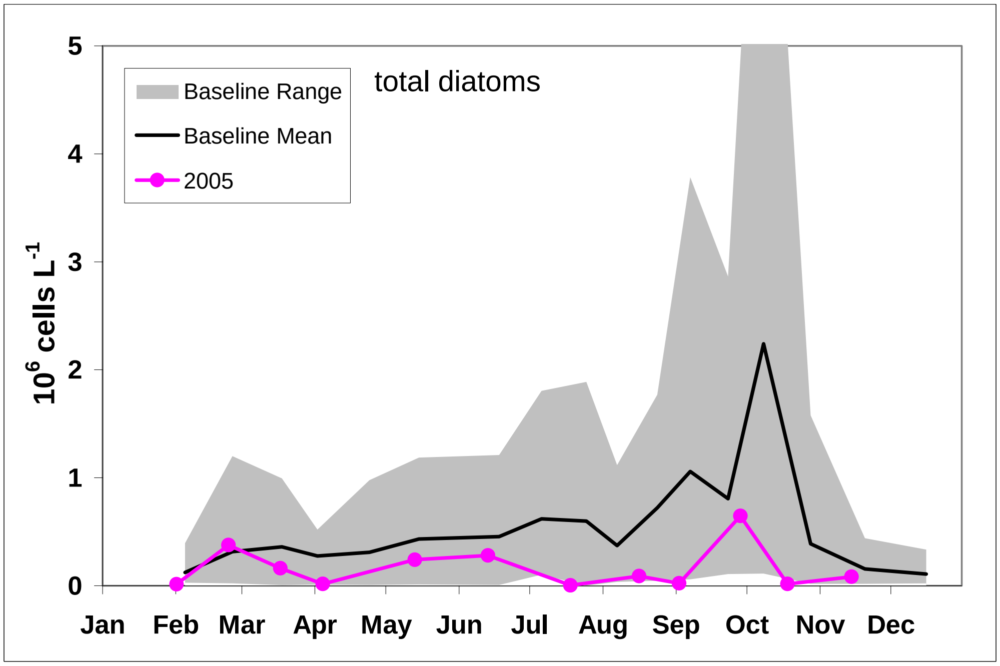

Figure 2-26. Total diatom pattern during 2005 compared to the 1992 2000 baseline mean and range of values. Diatom counts for most of the year were below average. Values are average of samples collected in the 'nearfield' area of Massachusetts Bay during MWRA monitoring.

Figure 2-27 shows dinoflagellate abundance, which was elevated relative to long-term mean levels during late February 2005. 2005 nearfield dinoflagellate abundance (2,000 cells L^{-1}) was ~4-fold the long-term mean level of 500 cells L^{-1}. *Heterocapsa rotundata*, *Heterocapsa triquetra* and an unidentified *Protoperidinium* spp. comprised the most abundant dinoflagellates during February 2005. Dinoflagellate abundance during March (389 cells L^{-1}) were ca. 50% above the long-term March level (275 cells L^{-1}) while during April 2005 dinoflagellate abundance (197 cells L^{-1}) was ~50% below the long-term April level (428 cells L^{-1}). During the *Alexandrium* bloom in May and June 2005, total dinoflagellate abundance was 1.5-fold to 2-fold above the corresponding long-term mean level. In May, mean dinoflagellate abundance was 2,300 cells L^{-1}, with *Alexandrium* present at 50 to 3,000 cells L^{-1} at individual stations. Surface *Alexandrium* abundance (2,000 – 3,000 cells L^{-1}) was much higher than that at 10-15 m (50 to 150 cells L^{-1}). Other common dinoflagellates included *Heterocapsa triquetra*, *Scrippsiella trochoidea* and unidentified *Protoperidinium* spp. Nearfield mean dinoflagellate abundance increased to 4,950 cells L^{-1} in June, near the high end of the range of observed values. From July and through the remainder of 2005, dinoflagellate abundance was markedly reduced compared to long-term mean levels with nearfield mean dinoflagellate abundance that was 6 % (July and early September) to 40% (late September) of the corresponding long-term mean abundance level.

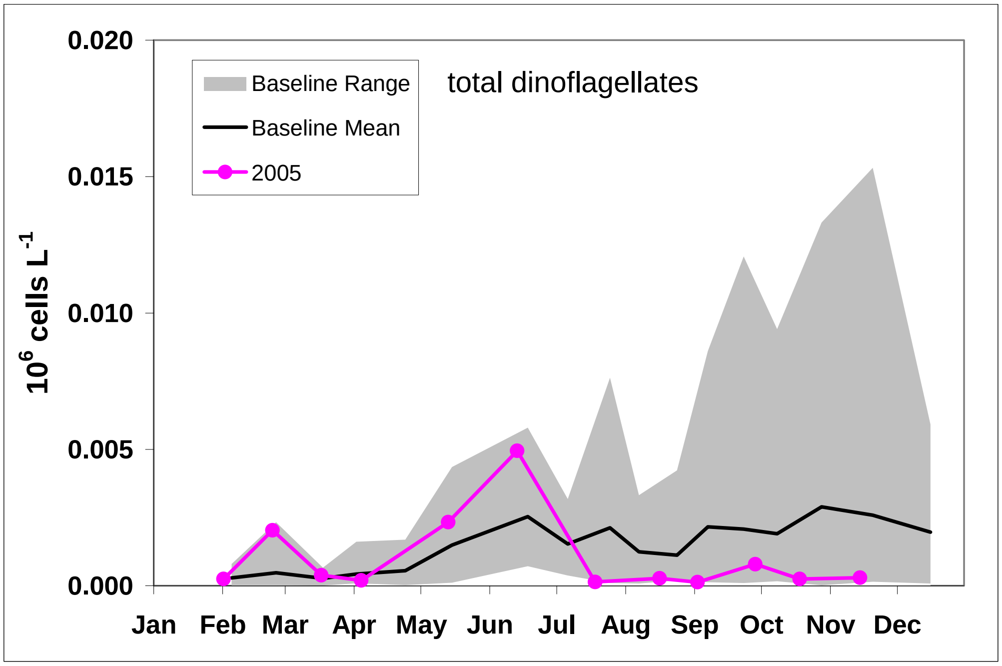

Figure 2-27. Total dinoflagellate (including *Alexandrium*) pattern during 2005 compared to the 1992-2000 baseline mean and range of values. Values are average of samples collected in the 'nearfield' area of Massachusetts Bay during MWRA monitoring.

These phytoplankton community structure analyses indicate that the 2005 *Alexandrium* red tide occurred against a backdrop of the usual seasonal post-*Phaeocystis* bloom diatom (late spring) to flagellate (summer) phytoplankton sequence, punctuated or interrupted by two northeaster storm events. This sequence, outlined in **Table** 2-2, was *Phaeocystis*=>Centric diatom (*Chaetoceros debilis*)=>Pennate diatom (*Thalassionema nitzschioides*) and dinoflagellates=>Microflagellates, with the *Alexandrium* bloom peaking during the pennate diatom- and dinoflagellate-dominated period during 28 May to 13 June 2005. The first storm (7-8 May) may have introduced *Alexandrium* into Massachusetts Bay (Anderson *et al.* 2005c) and also appears to have ended the ongoing *Phaeocystis* bloom, with a centric diatom community prevailing after the storm. The second storm (24-25 May 2005; see **Figure** 2-2) coincided with the transition from the centric diatom to pennate diatom (mainly *Thalassionema nitzschioides*) and dinoflagellate community. During this period dinoflagellates, including *Heterocapsa rotundata* (17,000 cells L^{-1}), *Heterocapsa triquetra* (7,000 cells L^{-1}) and *Alexandrium fundyense* (3,500 cells L^{-1}) (see **Table** 2-2), were abundant. The buoyant, relatively low salinity (freshwater influenced) coastal plume (Franks and Anderson 1992a; Keafer *et al.* 2005a and 2005b) that the storms 'pushed' into Massachusetts Bay (Anderson *et al.* 2005c) not only transported *Alexandrium* cells, but also may have favored their growth through physical and chemical modification of the environment. The stratification imparted by a freshwater layer is beneficial to, although not necessary for dinoflagellate growth (Tyler and Seliger 1978; Yentsch *et al.* 1986; Smayda 2002). Further, reduced salinity may increase toxicity in *Alexandrium* (Grzebyk *et al.* 2003; Etheridge and Roesler 2005). The dissolved organic substances ('humics') added by freshwater enhance the growth of many dinoflagellates including *Alexandrium* spp. (Carlsson *et al.* 1998; Gagnon *et al.* 2005). Elevated macro nutrient concentrations during this period also may have an effect of increased *Alexandrium* toxicity (Poulton *et al.* 2005). While this unique series of meteorological and oceanographic events led to the 2005 *Alexandrium* bloom, whether there were any biological factors within the phytoplankton community during 2005 that predisposed it to an *Alexandrium* bloom remains unclear.

The comparison of 2005 phytoplankton succession relative to the long-term patterns indicated that diatom levels were markedly reduced (to 10% of the corresponding long-term mean level) during the April 2005 period of *Alexandrium* bloom initiation in Massachusetts Bay, and diatom abundance was reduced to 50% of long-term mean levels during the peak of the *Alexandrium* bloom in May and June of 2005. Because of the differential silicon requirements of the dominant forms of phytoplankton (prymnesiophytes (*Phaeocystis*), diatoms and dinoflagellates) present in Massachusetts Bay during the winter-spring-summer succession, silicon concentration can be an important determinant of phytoplankton diatom to flagellate succession (Officer and Ryther 1980; Tilman 1977; Smayda 1990; Egge and Aksnes 1992). Further, Si-availability can be an important determinant of diatom species selection, with a 4-fold range in diatom Si:C content, and a tendency for selection of low Si-requiring species as ambient Si- concentration declines (Rousseau *et al.* 2002).

In temperate coastal seas *Alexandrium* blooms generally occur after the winter-spring diatom bloom, and coincident with or after a period of Si decline. For example, in Hiroshima Bay (Seto Inland Sea, Japan) *Alexandrium* blooms (to 100,000 cells L^{-1}) usually occur during periods of decreased Si concentration after *Skeletonema*-dominated diatom blooms (of $10x10^6$ cells L^{-1}) (Itakura *et al.* 2002, Yamamoto *et al.* 2004). Further, smaller *Skeletonema* blooms and larger *Alexandrium* blooms occurred in years having decreased Si concentration (Yamamoto *et al.* 2004). However, modulation of the physical environment via annual variation in freshwater discharge and concomitant and co-varying influences on stratification, residence time and Si concentration appear to be more important than Si concentration alone in influencing *Alexandrium* blooms in Hiroshima Bay (Yamamoto *et al.* 2004). In the Gulf of Maine, the role of Si in *Alexandrium* bloom initiation and maintenance is unclear. In the Western Gulf of Maine, *Alexandrium* blooms can occur while *in situ* Si concentrations are relatively high (5- 10 µM) and with a Si to DIN ratio of >5:1 (Love *et al.* 2005) implying that diatom-modulated Si concentration is not an important component of bloom formation there. However, other analyses have indicated that annual and longer-term variation in Si supply and subsequent variation in the Si:N ratio may be an important determinant of the relative magnitude of diatom versus *Alexandrium* blooms (Townsend *et al.* 2005).

Phytoplankton succession may be mediated by factors other than differential nutrient utilization. Allelopathy, or chemical interaction between plants that may either be positive (growth promoting) or negative (growth retarding or prohibiting), has been hypothesized as a determinant of phytoplankton succession (Keating 1977), including marine phytoplankton (Pratt 1966; Smayda 1980 and 1997; Maestrini and Bonin 1981; Gross 2003). *Alexandrium* spp. are among those marine phytoplankton having experimental evidence of allelopathy (Arzul *et al.* 1999; Fistarol *et al.* 2004; Tillmann *et al.* 2007), with allelopathic activity varying with toxicity and growth phase (Arzul *et al.* 1999). One hypothesized mode of allelopathy that may affect *Alexandrium* bloom initiation and maintenance is inhibition of competitive species growth. The growth of a diatom (*Chaetoceros gracile*) and one dinoflagellate (*Gymnodinium mikimotoi*) was inhibited, while the growth of another dinoflagellate (*Scrippsiella trochoidea*) was not inhibited when exposed to the filtrate (presumed to contain allelopathic agents) of *A. tamarense* culture (Arzul *et al.* 1999). The effects were seen over one week, and some species (*Chaetoceros gracile*) 'recovered' or resumed growth following presumed degradation of the added *Alexandrium* filtrate (Arzul *et al.* 1999). The *A. tamarense* concentration required to achieve growth inhibiting effects was ca. 2 x 10^6 cells L^{-1}, a concentration 50-times the peak *Alexandrium* concentration of 40,000 cells L^{-1} observed in a concentrated surface layer in Cape Cod Bay during the 2005 bloom. Interestingly, *Scrippsiella trochoidea*, which commonly co-occurs with *Alexandrium* in Massachusetts Bay, was not affected by *A. tamarense* filtrate in one study (Arzul *et al.* 1999) but was in another study (Fistarol *et al.* 2004). The effects of *A. tamarense* filtrate on a natural marine phytoplankton community (collected in Trondsheimfjord, Norway in September 2001 (14°C water)) was evaluated and found to decrease the community growth rate (Fistarol *et al.* 2004). After exposure to *Alexandrium* filtrate, *Skeletonema costatum*, *Scrippsiella trochoidea* and *Leptocylindrus* spp. declined over the short (3 day) incubation period, and growth rate of the community exposed to *Alexandrium* filtrate (0.08 d^{-1}) was half that of the control community (0.16 d^{-1}; Fistarol *et al.* 2004).

A second mode of allelopathic interaction that may influence *Alexandrium* blooms is inhibition of *Alexandrium* growth by other species. In this mode, similar to the hypothesis in which *Alexandrium* blooms form after a Si-depleted diatom population crashes (Townsend, unpublished data), *Alexandrium* bloom initiation is hypothesized to be inhibited by the presence of other species. In a series of studies, the filtrate of a diatom culture (*Thalassiosira* spp.) and several dinoflagellate cultures (*Prorocentrum dentatum*, *Prorocentrum compressum* and *Heterocapsa* spp.) reduced the growth rate of *A. tamarense* by 30-40% (Sekiguchi *et al.* 2000, 2004). Interestingly, similar species including *Thalassiosira* spp. (present at ~10,000 cells L^{-1}) and *Heterocapsa* spp. (both *Heterocapsa triquetra* (~7,000 cells L^{-1}) and *Heterocapsa rotundata* (~17,000 cells L^{-1}) were dominant members of the phytoplankton community during the 2005 Massachusetts Bay *Alexandrium* bloom (see **Table** 2-2). Despite some experimental *in vitro* evidence, proof of *Alexandrium*'s allelopathic interaction on *in situ* phytoplankton populations remains elusive, in part due to the difficulty in showing the ecological effects of allelopathy in aquatic systems (Gross 2003). The 2005 Massachusetts Bay *Alexandrium* bloom occurred within the context of a diverse phytoplankton community that was about 1,000-fold (mid-May during bloom initiation; see **Table** 2-2) to 400-fold (late-May to early-June during the *Alexandrium* bloom peak) more abundant than *Alexandrium*. The sequence of storm events and synchronous changes in the rest of the phytoplankton community indicate that the 2005 Massachusetts Bay *Alexandrium* red tide was likely the result of a phytoplankton sequence related to water mass changes rather than phytoplankton species succession within a single water mass.

2.7 Nutrients

A concern is whether the MWRA outfall nutrient discharge significantly stimulates *Alexandrium* in Massachusetts Bay. The outfall monitoring program showed that *Alexandrium* was rare during 1992-2004, even after the outfall was transferred to Massachusetts Bay in 2001. Clearly no obvious bloom developed exclusively around the outfall, but when cells are rare the counts lack the precision to reliably detect a minor stimulation. The 2005 bloom provided a background level of the high counts necessary to detect such a change, if it were to occur. Although the 2005 bloom presented the first good opportunity to test for outfall effects, it is ironic that the currents that brought that bloom into Massachusetts Bay also moved and mixed the cells too strongly and variably for us to detect a relatively small stimulus. The winds and freshwater accelerated the coastal current, brought the cells into the bay, and increased dilution of the outfall plume. Furthermore, patches of cells in high abundance were so dynamic that it was hard to tell precisely from where they had come. These factors add to the challenge of analyzing and interpreting the nutrient data in the context of the 2005 bloom. In this section, the overall distribution of nutrients and their dynamics in Massachusetts Bay is examined with specific focus on bay outfall nutrient loading, and potential effects on abundance.

2.7.1 MWRA Loading and Ambient Nutrient Concentrations

In May and June 2005, there was considerable variability in effluent flow from the MWRA discharge that was clearly associated with the storm/rain events (**Figure** 2-28). There was a small increase in flow associated with the first northeaster storm on May 7 (>500 MGD), but the highest flow was measured on May 25 during the second storm (800 MGD). Most of the daily flow observations in May and June 2005 were below 370 MGD, which is the long-term year-round daily-average flow for the MWRA system. The lowest flows of ~300 MGD were measured in late June. However, nutrient loading is not directly related to flow, as during high flow, nutrient concentrations tend to be lower due to dilution by the stormwater inputs to the system. For instance, there was an increase in NH_4 loading on May 25 coincident with the increased flow, but the loading was comparable to that observed during the second half of June. In general, most of the NH_4 loading measurements in May and June 2005 were below the annual mean daily NH_4 load of 24,000 kg/d as indicated by the thin black line in **Figure** 2-28. When the flow and NH_4 loading are compared to previous post-diversion data, May 2005 had a higher monthly mean flow than 2001-2004, but NH_4 loading during both May and June 2005 was below the minimum observed for those months since the outfall went online (**Figure** 2-29).

The overall distribution of ambient nutrients in May and June 2005 was consistent with the meteorological and oceanographic forcings discussed in Section 2.1. The freshwater signature of elevated SiO_4 was observed within the riverine plumes during many surveys, but the typical NH_4 signature of the MWRA discharge was not evident (e.g. **Figures** 2-13, 2-15, and 2-16 in Section 2.4 and Appendices B and C). Nearfield survey-mean nutrient concentrations are presented in **Figure** 2-30 for 2005 and compared to previous baseline and post-diversion levels. Panel A: For NO_3, the pattern observed for 2005 was similar to the mean for the baseline and 2001-2004 with a few minor differences. In May and June, NO_3 levels were very consistent across the three sets of survey mean data. Panel B: Silicate concentrations were also quite similar and generally within the baseline range for both 2005 and other post-diversion years. The most obvious difference was the low survey mean SiO_4 concentration (<1 µM) in June 2005, which was below the baseline minimum and 3 µM lower than the mean for the 2001-2004 surveys. This was likely driven by the May/early June diatom blooms (centric and pennate) that occurred in 2005, utilizing this nutrient. Panel C: Overall, the largest differences observed in survey mean nutrient concentrations in the nearfield in 2005 were the lower NH_4 levels compared to 2001-2004 levels. This was particularly evident for the May and June surveys (**Figure** 2-30) whose means were nearly 3 µM lower than those observed for 2001-2004 and the May 2005 value was even lower than the baseline minimum.

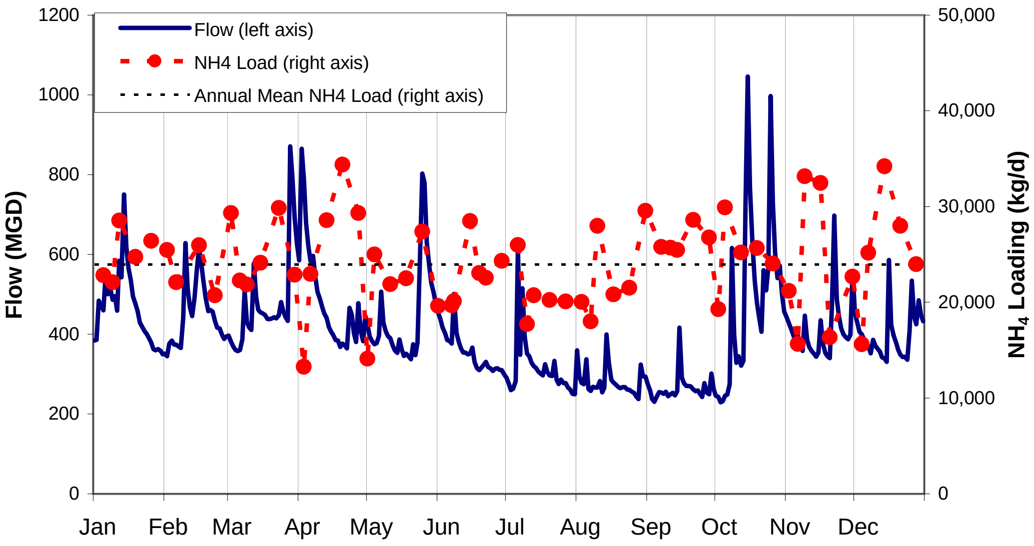

Figure 2-28. Daily flow and NH$_4$ loading in 2005 (based on daily and weekly measurements, respectively) at the Deer Island Treatment Plant.

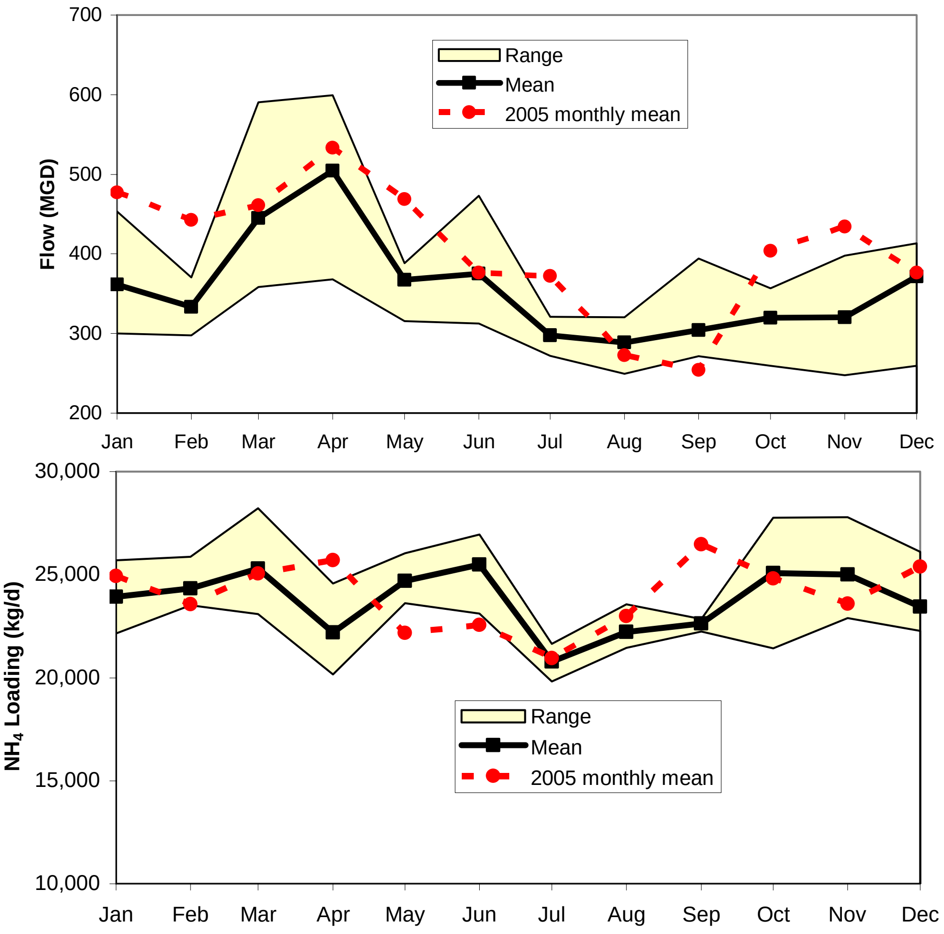

Figure 2-29. Monthly-mean flow (top panel) and NH$_4$ loading (bottom panel) in 2005 compared to mean and range of values measured in 2001-2004.

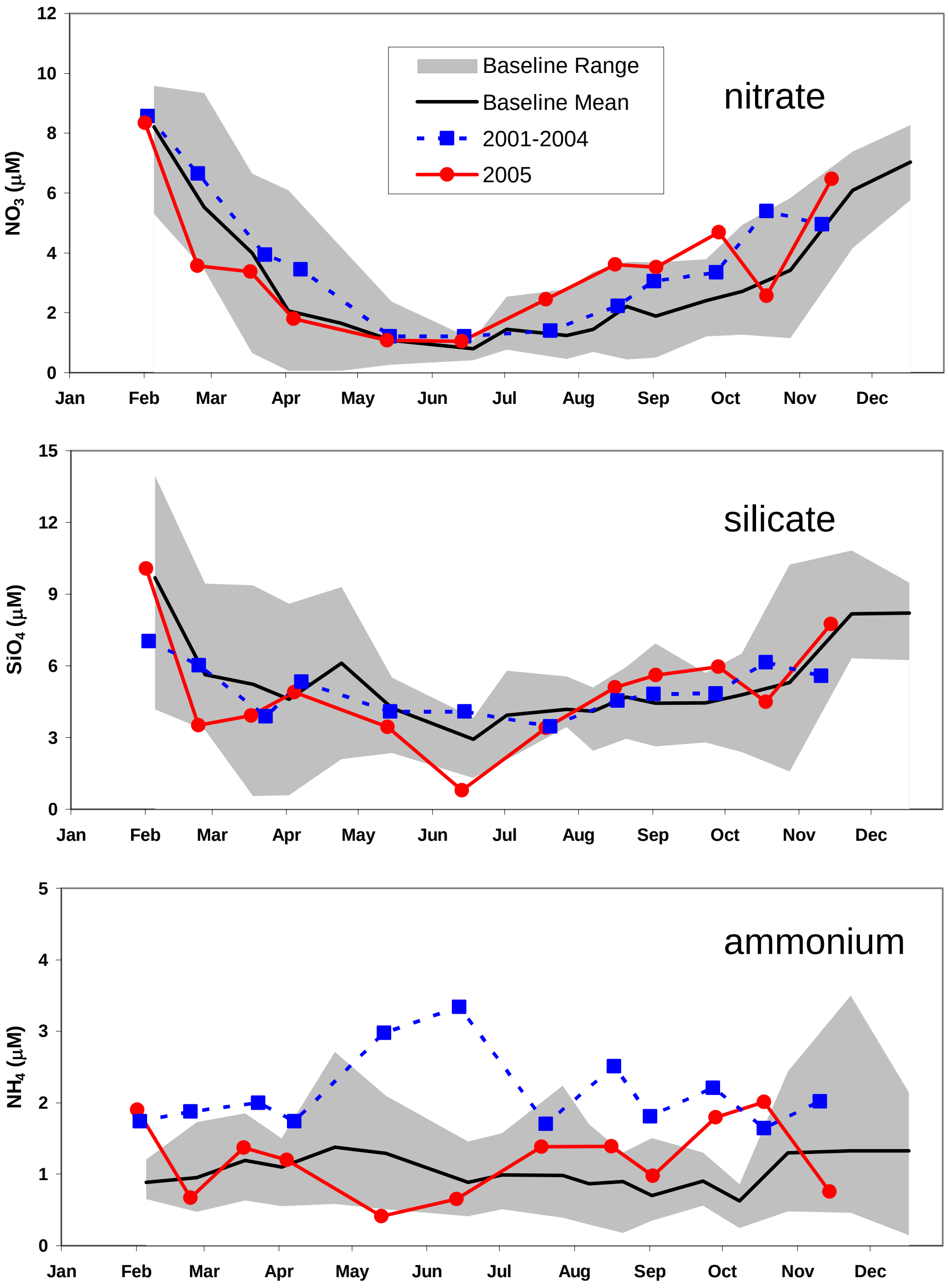

Figure 2-30. Survey-mean nearfield nutrient concentrations comparing baseline mean and range, 2001-2004 mean and 2005 values.

MWRA nitrogen loading in May and June 2005 was low compared to other years in the post-diversion period (**Figure** 2-29). The lower loading combined with the high winds, waves and currents noted earlier likely diluted the effluent plume so quickly that the NH_4 signal was lost. Another factor that controls both the distribution/dilution of NH_4 is water column stratification. When the water column is well mixed, there tends to be an area of elevated NH_4 concentrations in the surface waters of the nearfield within ~10 km of the outfall. When it is stratified, the NH_4 in the effluent plume remains below the pycnocline and is observed at even higher concentrations (than in the unstratified case) within 10-20 km of the outfall (low flow in summer and lack of utilization and increased regeneration at depth also contribute to higher NH_4

levels; Libby *et al.* 2004). These conditions were not present in May and June 2005 and the effluent plume's NH_4 signature was not evident in the monitoring data.

The degree of stratification in the water column can be approximated by comparing the near surface and near bottom water densities. As a general rule of thumb, a density difference of $<1\ \sigma_t$ indicates weak stratification and as $\Delta\ \sigma_t$ increases so does the strength of stratification. **Figure** 2-31 shows plots of $\Delta\ \sigma_t$ or degree of stratification for each of the nine survey periods in May thru early July 2005. During the first two survey periods, the water column was generally well mixed with only the areas to the north and south of Cape Ann exhibiting weakly stratified to stratified conditions. This stratification was driven by salinity as the fresher water in the coastal riverine plumes entered the bay from the north (see **Figure** 2-15 and Appendix B). A similar pattern in stratification was observed in late May with somewhat higher stratification south of Cape Ann due to the lower salinity waters resulting from high river flow following the second Northeaster storm (compare **Figure** 2-17 to **Figure** 2-31). By early June, most of Massachusetts Bay was stratified and although surface water temperatures had increased from late May through June, the stratification continued to be driven by salinity and input of fresher waters from the Gulf of Maine off of Cape Ann. Stratification typically develops in the bay over a period from April to June. The difference in 2005 is that it started later and salinity was the primary factor in its development into June (Libby *et al.* 2006). Typically, salinity and the freshets in April initiate seasonal stratification development, but it is the warming of the surface waters in May and June that leads to the establishment of stratified conditions. In 2005, the water column became stratified, but the storm conditions that led to it being driven by salinity also produced relatively high currents over most of the May and June time frame, which would tend to disperse the plume.

2.7.2 Dilution within the hydraulic mixing zone (HMZ)

The chemical signature of the effluent plume was not observed during the May and June 2005 surveys, but mathematical modeling approaches can provide an indication of how the atypical oceanographic conditions during this period may have affected plume dilution and dispersion. The plume's dimensional characteristics and effluent dilution can be estimated using the mathematical Roberts-Snyder-Baumgartner (RSB) model as described in Roberts (1999). RSB is based on extensive experiments on multiport diffusers in density-stratified currents of arbitrary direction reported in Roberts *et al.* (1989). It uses semi-empirical formulations based on the relative magnitudes of the dominant length scales of the problem. The model output consists of the plume characteristics (dilution, rise height, and wastefield thickness) at the end of the HMZ. In 2001, two dye studies were performed to document initial dilution at the MWRA's bay outfall to verify outfall performance (Hunt *et al.* 2002a and 2002b). These studies were conducted under weakly stratified (April) and strongly stratified (July) conditions. In addition to verifying that the outfall performance was as predicted and designed, the plume studies showed that there was strong agreement between field and modeled results under two distinctly different stratification conditions. This provides us with a high degree of confidence in the RSB model as a tool to examine the influence of varied oceanographic conditions and stratification regimes on the effluent plume dispersion.

The RSB model was run under variable current speeds and degrees of stratification to bound the conditions observed during May and June 2005. Effluent flow was held constant across all scenarios at 17.5 m^3/s or about 400 MGD. The currents were set at 0 and 40 cm/sec. While no-current is not likely, it does present a 'worse-case' scenario. The 40 cm/sec current is quite fast for the bay, but is only half of the maximum current observed at the GoMOOS buoy A during the three northeaster storms that occurred over this time period (see **Figure** 2-2). Three stratification levels were assumed – well-mixed ($\Delta\ \sigma_t = 0$), weakly stratified ($\Delta\ \sigma_t = 0.7$), and strongly stratified ($\Delta\ \sigma_t > 4$). The weakly stratified conditions were similar to levels observed mid-May 2005 and the strongly stratified conditions were similar to some of the higher stratification levels seen in mid to late June towards the end of the bloom (**Figure** 2-31).

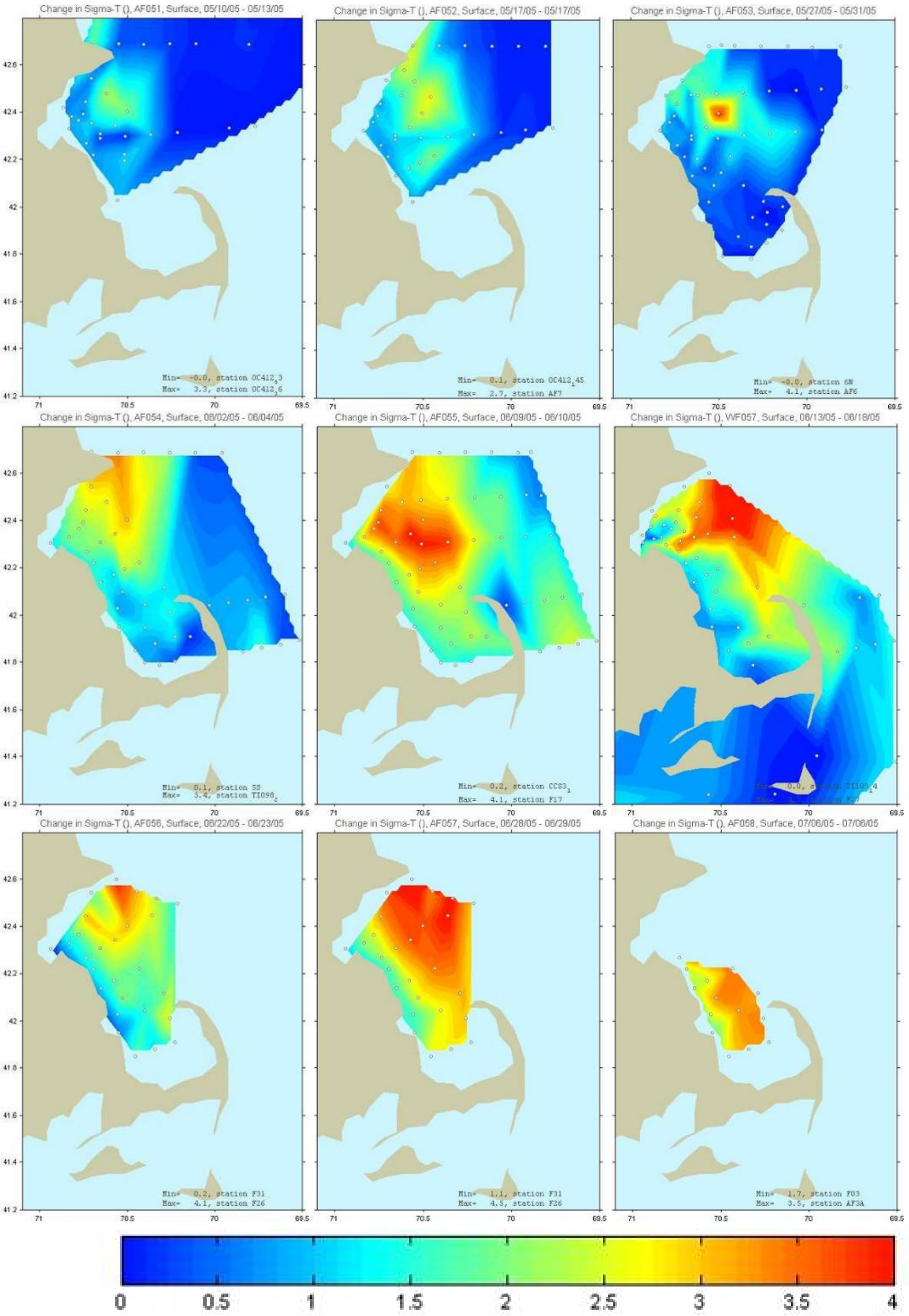

Figure 2-31. Degree of stratification based on difference between surface and bottom water density values for all nine survey time periods.

The RSB model results are presented in **Table** 2-3 and **Figure** 2-32. Under the no-current scenarios there is little difference (100±20) in the minimum level of dilution in the HMZ. The HMZ is the area in which mixing and dilution occur as a result of the turbulence and buoyancy generated by the outfall discharge itself. The primary difference under this scenario was controlled by stratification with the depth of the plume reaching the surface under well-mixed and weakly stratified conditions and remaining below the pycnocline at 13.1 m under strongly stratified conditions. Although the no-current scenarios are unlikely, the model results are comparable to observed differences in NH_4 distributions in the nearfield during well-mixed and stratified conditions (slightly more dilute and in the surface versus more concentrated and below the pycnocline, respectively). The more interesting results are for the strong-current scenarios. The minimum dilution in the HMZ is 4-6 times higher than for the no-current scenarios (**Table** 2-3). Even under strongly stratified conditions the dilution is much higher than the no-current, well-mixed scenario. Additionally, not only is the strong current increasing the level of dilution, but it is also pushing the effluent plume deeper into the water column. During the three northeaster storms, currents reached a maximum of 80 cm/s at GoMOOS buoy A and likely were higher than 40 cm/s in the nearfield, which suggests even higher levels of dilution and potentially a deepening of the plume based on the mid-May weakly stratified condition results in the RSB model.

Table 2-3. Summary of initial dilution for the six scenarios current speed/stratification scenarios.

	No Current	**Strong Current (40 cm/s)**
Minimum Dilution (X times)		
Unstratified	128	791
Weakly stratified (mid-May 2005)	127	520
Strongly stratified (mid-June 2005)	82	319
Depth of Plume (m)		
Unstratified	0	0
Weakly stratified (mid-May 2005)	0.2	6.2
Strongly stratified (mid-June 2005)	13.1	15.7

Similar results were found (M. Zhou, M. Jiang, pers. comm.) using passive tracers released from the outfall site in a hydrodynamic model (ECOM-si; Blumberg 1991; Signell *et al.* 2000) incorporating 2005 wind forcing. The particles were released from the outfall site from May 5 to May 15. Prior to the May 7 northeaster, the particles rose to the surface and moved slowly to the south. During the storm, the particles moved very fast to the south and were pushed down to >30m depths and then after the storm winds relaxed the particles tended to accumulate in the nearfield for a few days. These results combined with their model simulations showing the formation of eddies following relaxation of the NE winds (see **Figure** 2-5), highlight how dynamic the physical oceanographic conditions were during the *Alexandrium* bloom. The strong and varying currents would move cells about and also disperse the outfall's chemical signature.

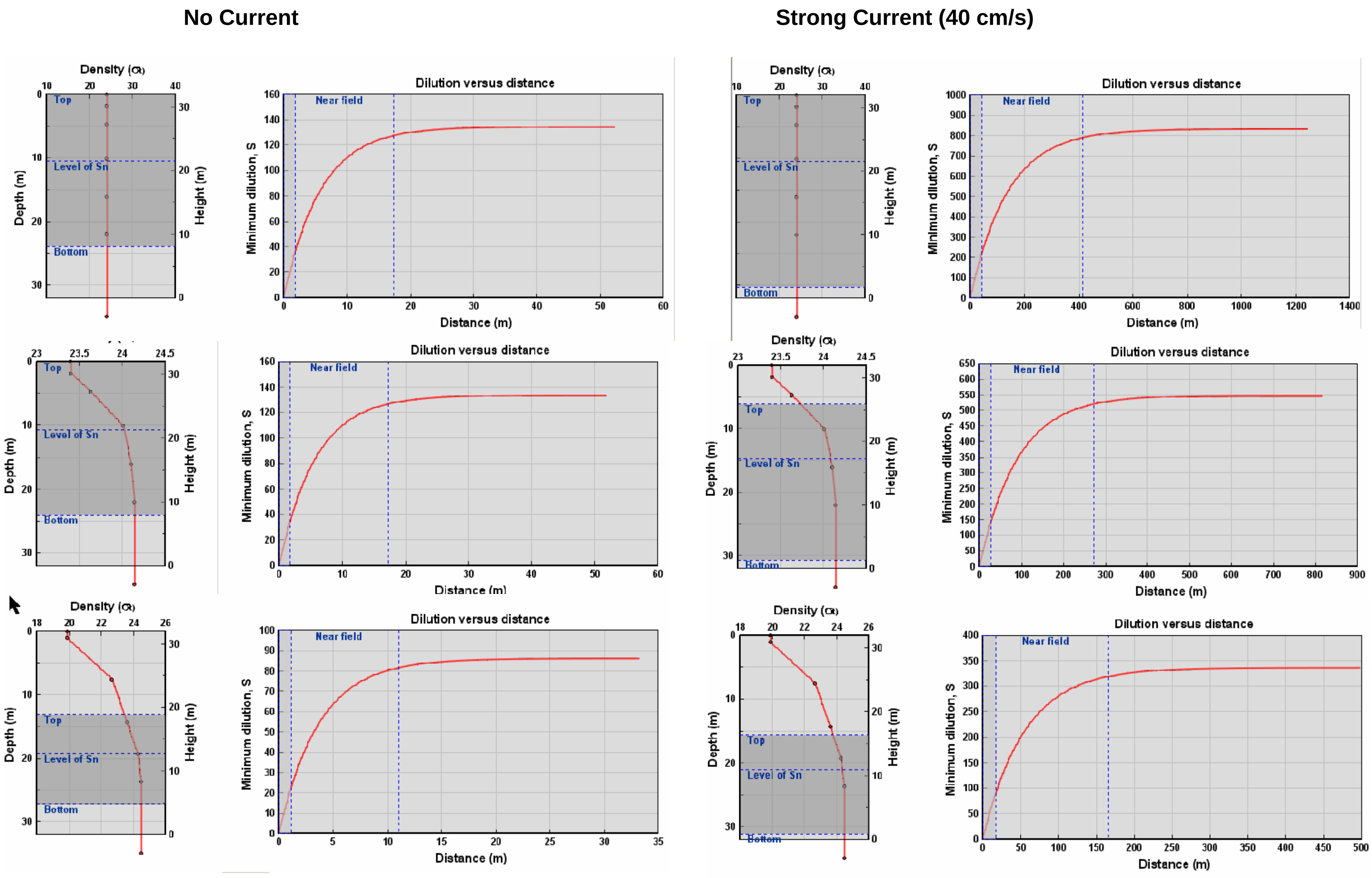

Figure 2-32. RSB model output for the six current speed/stratification scenarios.

“Level of Sn” (Sn is RSB’s symbol for nearfield dilution) is the depth of the plume at the nearfield boundary as indicated by the vertical dashed line in the distance plots.

2.7.3 Dilution beyond the hydraulic mixing zone

The RSB model (**Figure** 2-32) describes dilution of effluent within the HMZ, a process that takes a few minutes and is expedited by the effluent's momentum and buoyancy. After that, the effluent is further diluted on the time scale of hours to days by oceanic dispersion, a process analogous to diffusion, according to an equation derived in Roberts and Webster (2002):

$$S_f = \left[erf\left(\frac{3/2}{(1+8\alpha L^{-2/3} t)^3 - 1} \right)^{1/2} \right]^{-1}$$

where Sf is the plume centerline dilution beyond the HMZ (Sn is dilution within the HMZ)
L is the length of the outfall
t is time
α is a coefficient in the equation $\varepsilon = \alpha\ W^{4/3}$, where ε is the rate of dissipation of turbulent kinetic energy and W is the width of the plume)

For arguments sake, how much phytoplankton growth might we expect based on MWRA's NH_4 loading? To evaluate this, we modified the Roberts and Webster (2002) ocean dispersion model to include algal growth and nutrient uptake to estimate the impact of MWRA loading on phytoplankton. The following assumptions made for these calculations approximate typical dilution, loading and ambient conditions. Although, as discussed previously, conditions in May and June 2005 were not typical.

Assumptions based on typical conditions:

1.	Sn = 100	(initial dilution of effluent and its contaminants within the HMZ)
2.	L= 1200 m	(length of the MWRA outfall)
3.	$\alpha = 0.01\ cm^{2/3}\ s^{-1}$	(a typical value suggested by Roberts and Webster (2002))
4.	Co = 1200 µM NH_4	(effluent concentration of the NH_4)
5.	Cb = 0.5 µM NH_4	(background concentration of NH_4)

The heavy solid red line of **Figure** 2-33 shows the decreasing concentration of discharged NH_4 in a parcel of water in relation to the number of days that water has traveled downstream of the outfall. Time zero in the figure corresponds to the time the parcel leaves the HMZ. The concentration at time zero is 12.5 µM (resulting from the assumed 100-fold dilution of the 1200 µM effluent NH_4 into seawater with ambient concentration of 0.5 µM NH_4). As time progresses the concentration of NH_4 in the parcel decreases to 1 µM (twice the ambient concentration) at 4.6 days.

The foregoing assumed that NH_4 is conservative (that is, not taken up by algae). Now consider the nonconservative case by including algae which take up NH_4 and grow. We extended the equation above for S_f to include uptake of NH_4 by the algae and growth of those algae. We considered 3 cases for doubling time (the no-growth case is described above):

Doubling time	**Analogous to**	**Maximum growth rate**	**Line in figure**
infinite	No growth	0	thick solid
3 days	Dinoflagellates	$0.23\ d^{-1}$	thin solid
1 day	Diatoms	$0.69\ d^{-1}$	dotted

And for the calculation we made the following assumptions:

1. Xb = 2 µg chlorophyll L^{-1} (background concentration of algae)
2. Xo = 0 (effluent concentration of algae)
3. Algal nitrogen:chlorophyll composition = 1 so that a cell with y µg chlorophyll also contains y µmoles of nitrogen.
4. Growth and uptake is limited only by NH_4 concentration (other factors remain constant).
5. Growth rate is half of its maximum value at 1 µM ammonium (Ks=1).
6. Growth and uptake are tightly coupled (algal composition is constant).
7. Algal specific mortality matches the algal specific growth rate at the ambient NH_4 concentration.
8. Algal mortality is regenerated as NH_4. Algal abundance is expressed as µM NH_4

The thin solid green line in Figure 2-33 shows the response of dinoflagellates; they increase in abundance to a peak equivalent to 2.2 µM NH_4 at 3.6 days but are gradually diluted to the background concentration of 2 µM. The dotted green line shows the response of the faster-growing diatoms; they increase in abundance to a peak of 2.6 µM at 2.4 days but again the increase is ephemeral. This exercise illustrates why it is challenging to detect a local increase in algal abundance downstream from, and caused by, the outfall. Field studies have to sample at the right location (about 2-4 days travel time "downstream") and even there may only find a 10-30% increase in abundance of dinoflagellates and diatoms, respectively. Furthermore this is an upper bound to the algal response as the doubling times we assumed require adequate light so the response would be less in a plume trapped below the pycnocline.

The overall distribution of ambient nutrients in May and June 2005 was not unexpected or out of the normal range of variability (see Section 2.4 and Appendices B and C). The freshwater signature of elevated SiO_4 was observed within the riverine plumes during many surveys, but the typical NH_4 signature of the MWRA discharge was not evident. In June as stratification became stronger, there were higher concentrations of NH_4 in the bottom water layer, but they were not clearly associated with the outfall. MWRA nitrogen loading in May and June 2005 was low compared to other years in the post-diversion period. The lower loading combined with the high winds, waves and currents noted earlier presumably diluted the effluent plume so quickly that the NH_4 signal was lost. The diatom and dinoflagellate blooms that were coincident with the *Alexandrium* bloom likely contributed to the lack of a NH_4 plume signature due to rapid biological utilization, but it is likely that the storms and associated strong currents and mixing were the primary factors leading to reduced NH_4 levels in the nearfield. Lastly, the sampling design for the surveys provided broad spatial coverage of the bay and may have been too broad (often <5 stations in the nearfield area) to detect a localized plume signature.

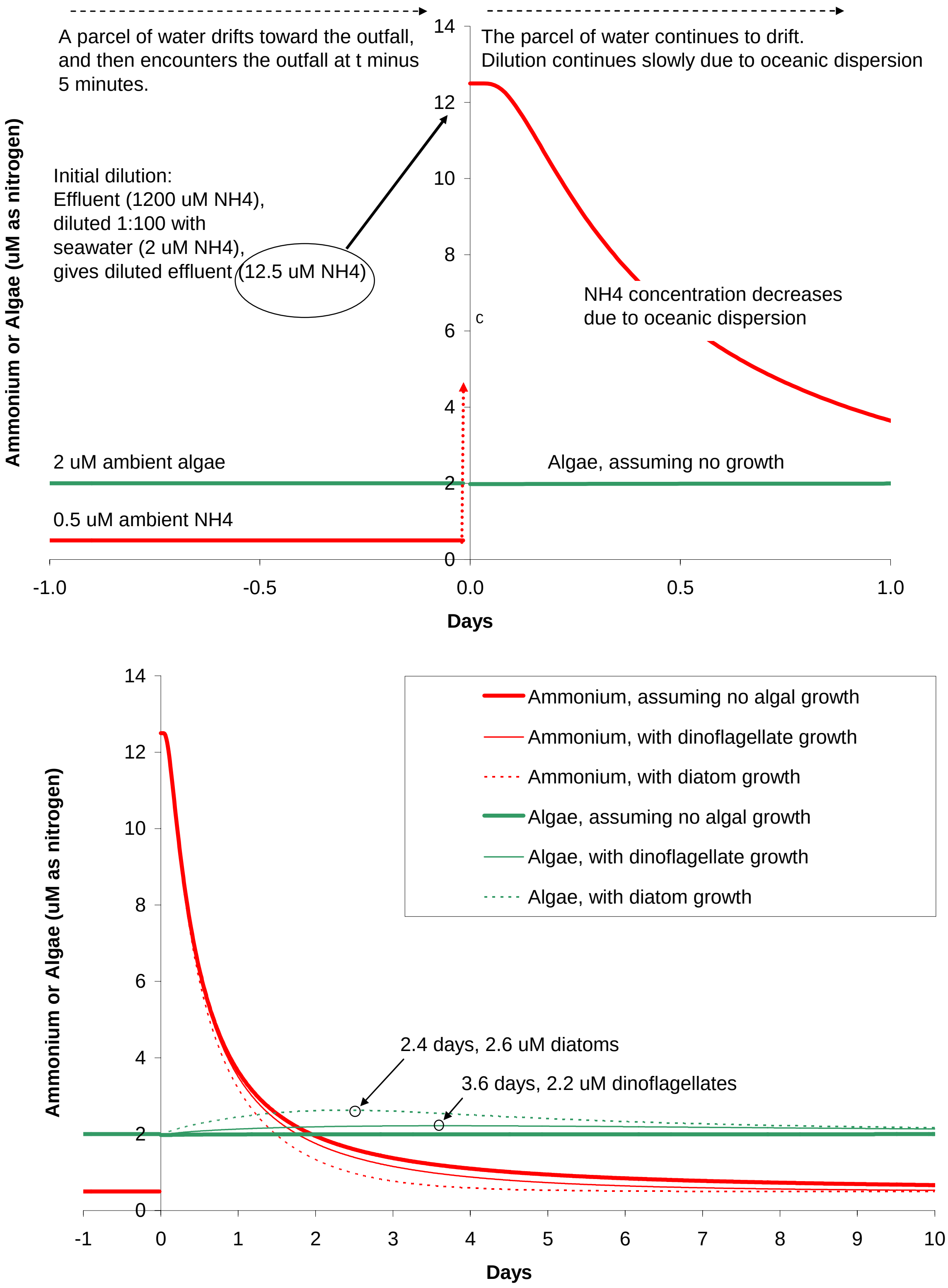

Figure 2-33. Hypothetical oceanic dispersion of the effluent plume, calculated for three scenarios of algal growth and nutrient uptake.

2.7.4 Correlations between nutrients and *Alexandrium*

The distributions of nutrients and *Alexandrium* were presented for each of the surveys in Section 2.4 and are included in Appendix B for all depths sampled. There were no clear spatial associations between nutrients and *Alexandrium*; this section revisited those data and in particular looks for temporal associations. It is a well-accepted truism, however, in phytoplankton ecology that ambient nutrients and phytoplankton biomass seldom correlate because in many cases, cells would have already taken up the nutrients prior to the measurements. Rather, it is the change in cell concentration that should inversely correlate with the change in nutrient concentration; such changes, however, are difficult to observe directly. In the case of the 2005 bloom, *Alexandrium* was never the dominant phytoplankton species and even peak abundances were an order of magnitude lower that those for other species/groups (see **Figure** 2-24). Since nutrient dynamics (i.e. uptake) were not driven by *Alexandrium*, it seemed plausible that there might be a discernable positive relationship between nutrient availability and abundance for this species.

To take a closer look at changes in nutrients and *Alexandrium* over time, the data were examined on a per-station basis from April to July (**Figures** 2-34 to 2-36). The results at three stations located to the south of Cape Ann (F22/F26), in the nearfield (N18), and off of Marshfield (F06) exhibit the basic trends observed across the stations. There are a number of patterns that are evident. Nutrient concentrations were higher in April/May than in June and there was a progressive decrease in nutrient concentrations from the north (**Figure** 2-34) to the nearfield (**Figure** 2-35) and then further south (**Figure** 2-36). Silicate concentrations were high in April and early May (surface and 10-m depths), but decreased by mid to late May in the surface waters during the diatom blooms. Nitrate was relatively high (2 µM) in comparison of NH_4 in the surface waters at F22/F26 in April and early May and at 10-m depth at each of the stations from April to late May/early June. Ammonium was generally quite low (<1 µM). It peaked at F22/F26 in the 10-m waters in late May/early June at concentrations slightly above 1 µM. At station N18, NH_4 concentrations approaching 2 µM were observed at 10-m in mid May and early June. This station is located only 2 km to the south of the outfall and is often a location where the outfall plume can be detected due to elevated NH_4 concentrations of 5-10 µM. These low NH_4 data highlight the greater level of dilution during this time period. Similar levels of NH_4 were observed at 10-m at station F06 in late May/early June, but surface water levels of NH_4 at this station also approached 2 µM in late May and were comparable to the concentrations of NO_3 and SiO_4 (**Figure** 2-36).

The patterns in *Alexandrium* abundance from each of the stations showed peak bloom values in late May (as seen for F22/F26 and N18 in **Figure** 2-34 and **Figure** 2-35, respectively) or early June (e.g. stations N04, F13, AF10 data not shown). The other clear pattern was that abundance in the 10-m depth samples peaked after it had peaked in the surface waters. Usually the 10-m depth peak abundances were lower than the peak levels observed in the surface waters, but at station F06 the 10-m depth peak was about 4 times higher (~20,000 cells L^{-1}) than any of the surface water measurements (all <5,000 cells L^{-1}; **Figure** 2-36). One other observation based on these figures is that the higher *Alexandrium* abundances tended to occur when nutrient concentrations were generally depleted (<0.5 µM) in both the surface and 10-m waters. One exception was at station F22/F26 in late May when surface SiO_4 and NO_3 were present in concentrations >1 µM (low, but not depleted).

No linear relationships could be detected between nutrient concentrations and *Alexandrium* abundance over all of the May-early July surveys (**Figure** 2-37). These plots show DIN, PO_4, SiO_4 and the N:P ratio versus cell abundance for each of the *Alexandrium* samples collected in Massachusetts and Cape Cod Bays (note that the number of data points in the DIN and N:P ratio plots is lower since NH_4 data from the WHOI surveys are suspect and not included). The samples were primarily from surface and 10-m depths with some samples from ≥20 m.

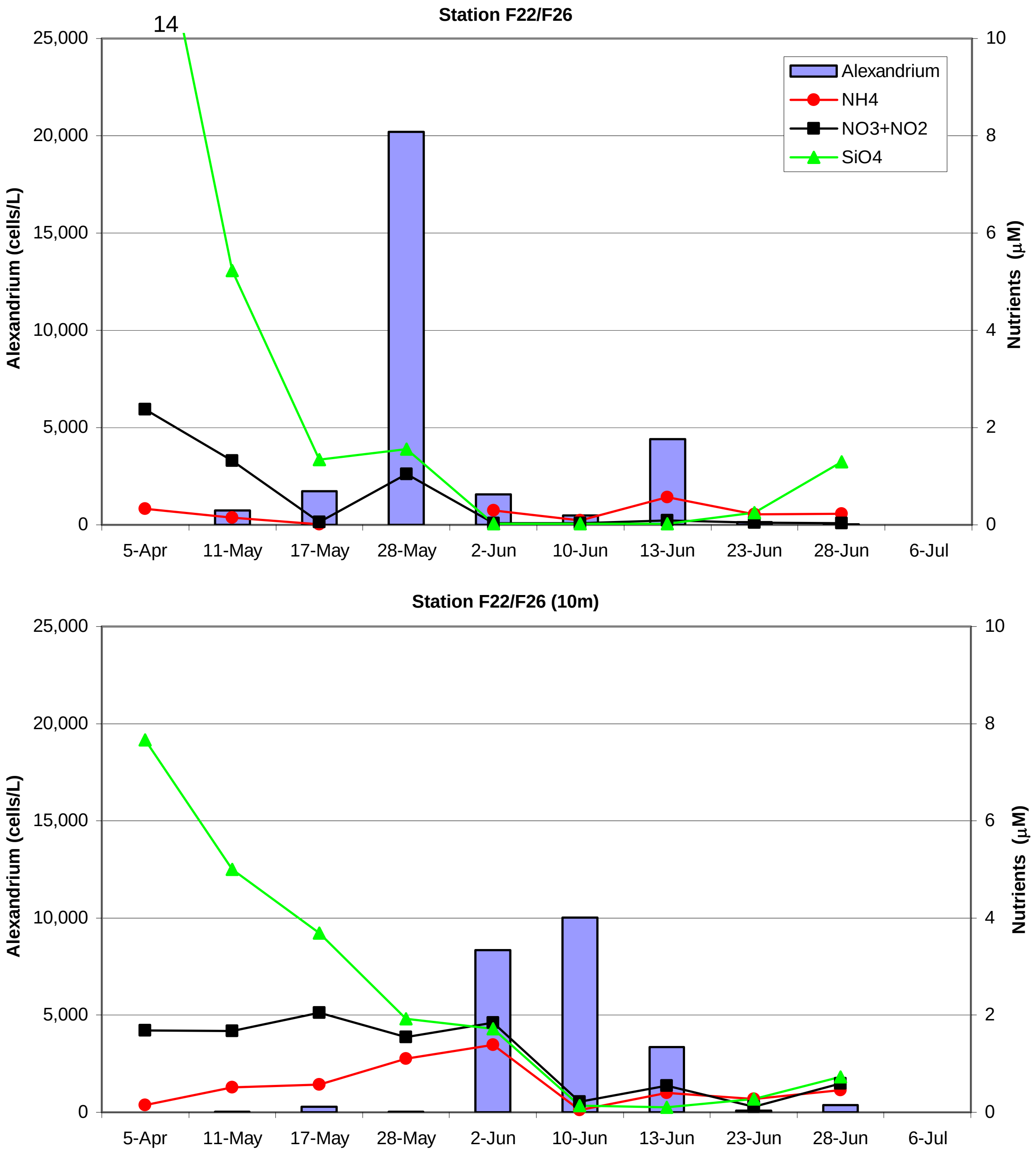

Figure 2-34. Station F22/F26 nutrient concentrations and *Alexandrium* abundance at the surface and 10-m depth (upper and lower panels, respectively).

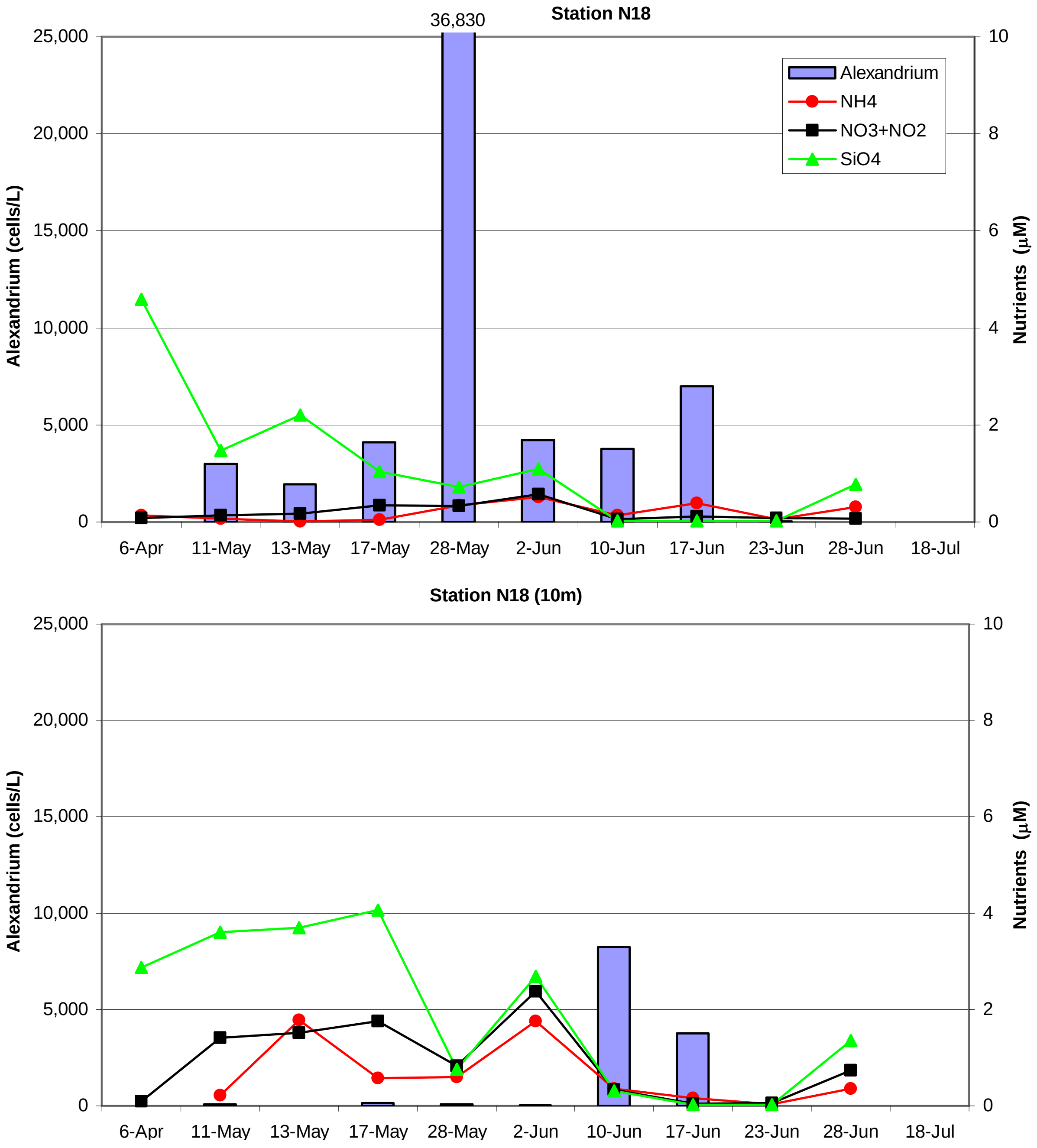

Figure 2-35. Station N18 nutrient concentrations and *Alexandrium* abundance at the surface and 10-m depth (upper and lower panels, respectively)

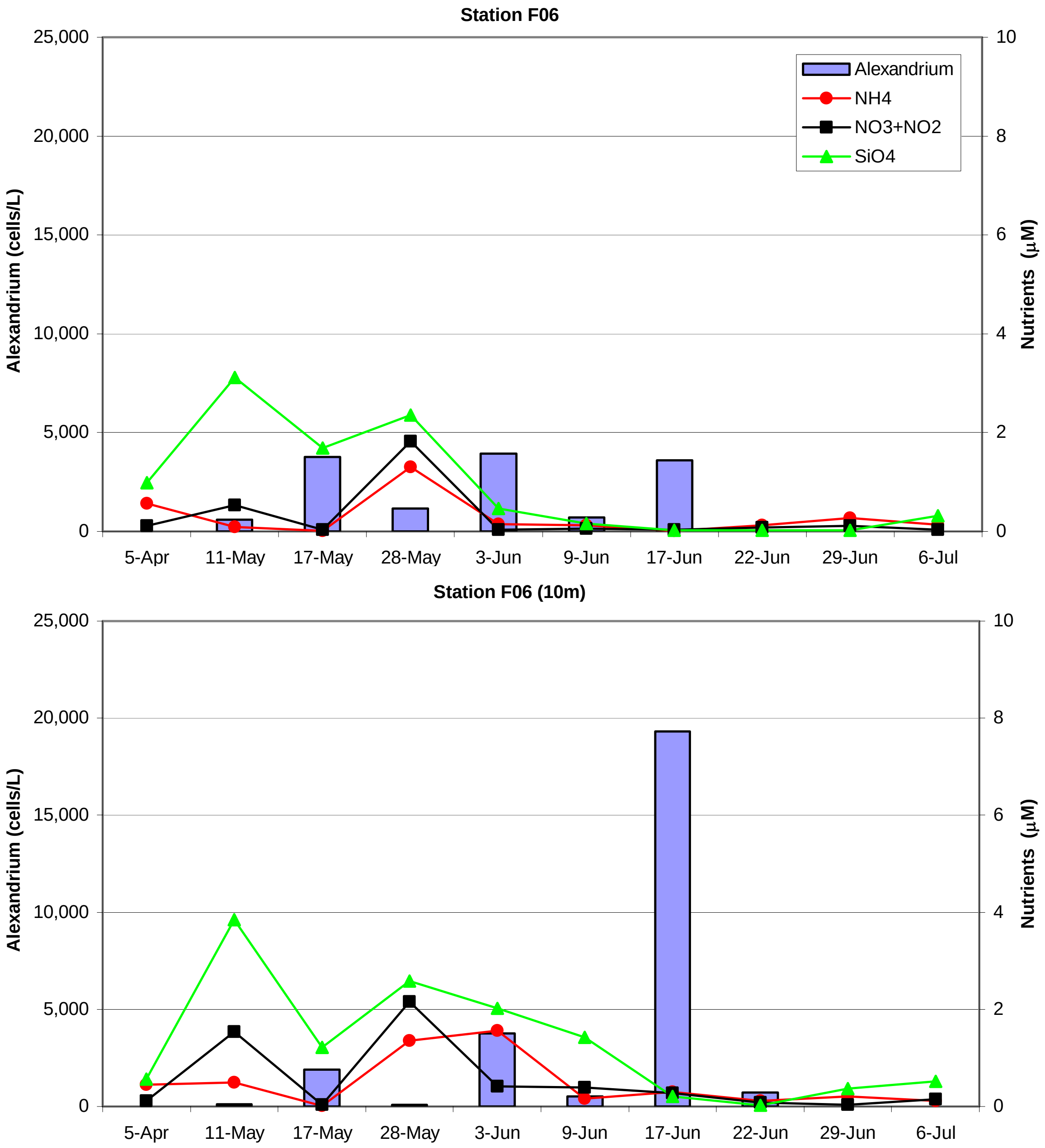

Figure 2-36. Station F06 nutrient concentrations and *Alexandrium* abundance at the surface and 10-m depth (upper and lower panels, respectively)

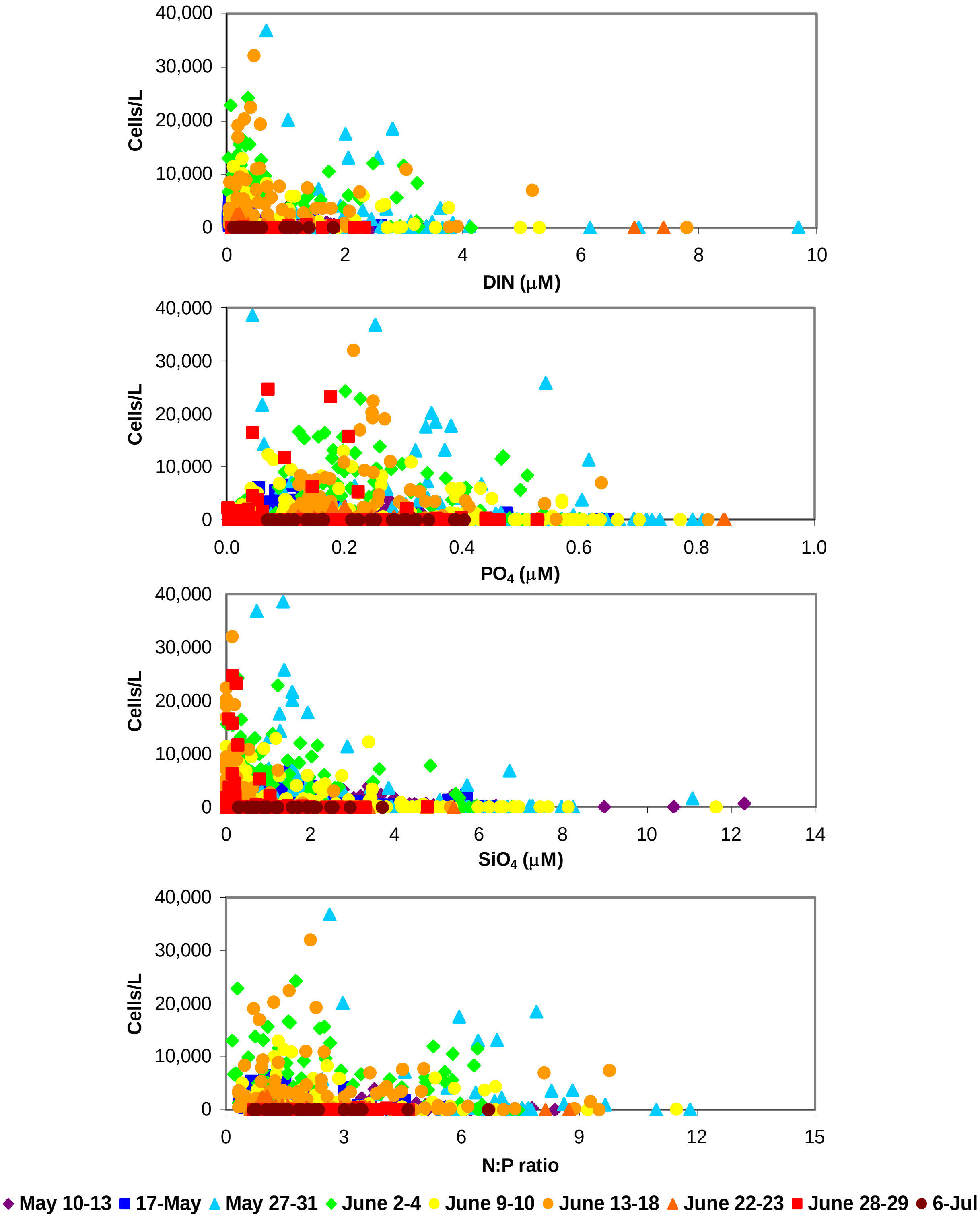

Figure 2-37. Nutrient concentrations and N:P ratios for all depths sampled for *Alexandrium* (surface, 10-m and 20-m) in Massachusetts and Cape Cod Bays.

The early and mid-May data are not clearly visible in **Figure** 2-37, but *Alexandrium* abundances were relatively low and nutrients were available at relatively high concentrations. During the peak of the bloom in late May, high abundances (>10,000 cells L^{-1}) occurred over a range of nutrient concentrations – DIN from 0.5 to 3 µM, PO_4 from 0.05 to 0.6 µM, and SiO_4 from 1-3 µM (**Figure** 2-37). A closer look at the data shows that most of these data points were from surface samples and the range in nutrient concentrations associated with these high *Alexandrium* abundances is consistent with the overall range in concentrations observed over the bays during that survey period. The availability of nutrients during the late May peak in the bloom may also have contributed to the overall toxicity impact of the 2005 *Alexandrium* bloom as it has been noted in lab and field studies that *Alexandrium* tend to be more toxic under nutrient (particularly nitrogen) replete conditions (Poulton *et al.* 2005).

The N:P ratio during the May 27-31 period for the high *Alexandrium* abundance samples ranged from about 3 to 8 with the highest abundances occurring at the lower N:P ratios. This general trend was also observed during the three early to mid-June surveys. During the June 2-4, June 9-10 and June 13-18 surveys, the highest *Alexandrium* abundances occurred in waters with low DIN concentrations (<1 µM). These waters also had PO_4 concentrations of about 0.2-0.3 µM and low SiO_4 concentrations. In early June, SiO_4 concentrations associated with *Alexandrium* abundances of >10,000 cells L^{-1} ranged from 0-2 µM. By the mid-June survey, SiO_4 as well as DIN was essentially depleted in the samples with higher *Alexandrium* abundances (**Figure** 2-37). There were some samples during these early- to mid-June surveys that had relatively high abundances (5,000 to 10,000 cells L^{-1}) that were coincident with elevated DIN concentrations (>2 µM). These samples were either from within Boston Harbor or were collected from deeper depths likely near or below the nutricline.

Overall, high *Alexandrium* abundances occurred over a range of DIN (0-4 µM) and PO_4 (0-0.6) concentrations, but primarily at low SiO_4 levels (<2µM). This does not necessarily suggest an effect of SiO_4 on *Alexandrium*, but rather as noted earlier may be a reflection of how phytoplankton ecological processes led to decreasing SiO_4 concentrations due to the diatom blooms in May and June. Townsend (unpublished) has hypothesized that there is an allelopathic relationship between *Alexandrium* and diatoms, whereby *Alexandrium* blooms cannot occur until diatoms have disappeared from the water, typically as a result of silicate limitation. Scatter plots of DIN versus PO_4 and SiO_4 highlight trends in these ratios compared to *Alexandrium* abundance (**Figure** 2-38). The data points are color coded to note *Alexandrium* abundance and include the same data shown in **Figure** 2-37. The system is clearly depleted in DIN relative to PO_4 (nearly all the points are below the Redfield ratio line of 16:1) and only slightly depleted in DIN vs. SiO_4 though there are nearly as many high *Alexandrium* abundance data points above the line as below. The most obvious trend is that as abundance increases there are decreases in nutrient concentrations. For DIN vs. PO_4 it appears many of the >5,000 cells L^{-1} abundances occur when DIN is depleted relative to PO_4. The DIN vs. PO_4 trends are similar to those observed in the Gulf of Maine during previous *Alexandrium* blooms (Love *et al.* 2005). The DIN vs. SiO_4 trends, however, are different in that in our data these nutrients are present in close to a 1:1 ratio over the entire range of abundances, whereas Love *et al.* (2005) found that *Alexandrium* abundances increase as DIN was depleted relative to SiO_4. Two possible explanations for this are that the 2005 bloom occurred concurrently with centric and pennate diatom blooms (drawing down SiO_4) and that the 2005 bloom abundances were an order of magnitude higher than the 1998 and 2000 bloom data examined by Love *et al.* (2005).

Spearman's Rank Correlations between *Alexandrium*, *in situ* parameters, and nutrient concentrations showed no strong relationships between *Alexandrium* abundance and nutrient concentrations. There were weak, but significant ($P<0.01$) inverse relationships with NO_3, NH_4, PO_4 and SiO_4 concentrations when all data were examined, but this was likely due to the confounding influence of depth (nutrients lower in surface waters and *Alexandrium* tended to be higher). An examination using only surface data showed no relationships between *Alexandrium* and nutrient concentrations, but weak, significant ($P<0.0001$) inverse relationships with both salinity and temperature ($r_s = -0.24$ and -0.42, respectively).

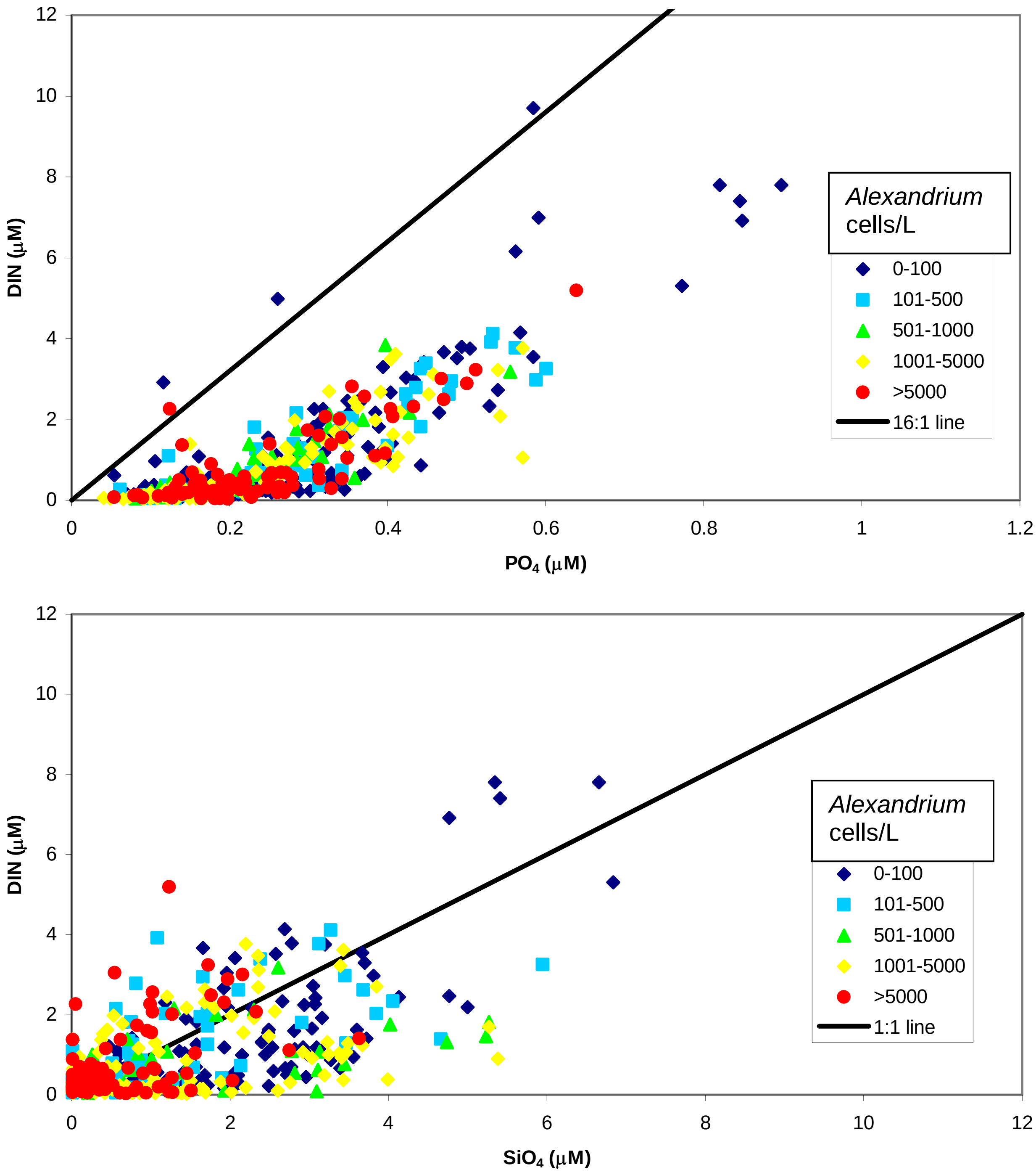

Figure 2-38. DIN versus PO_4 (upper panel) and SiO_4 (lower panel) concentrations for all depths sampled for *Alexandrium* (surface, 10-m and 20-m) in Massachusetts and Cape Cod Bays.

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3 NUMERICAL MODELING

Numerical modeling efforts during the 2005 *Alexandrium* bloom were conducted by multiple researchers and institutions. Modelers from the University of Maine provided forecasts of currents that were invaluable for planning MWRA surveys in Massachusetts Bay (H. Xue, pers. comm.; Xue *et al.* 2000). Modelers from the University of Massachusetts at Boston provided **Figure** 2-4 and **Figure** 2-5 as part of their focus on oceanographic processes within Massachusetts Bay (M. Zhou and M. Jiang, pers. comm.; Jiang *et al.* 2004). Previous runs of the Bays Eutrophication Model showed that the MWRA outfall typically provides only a very minor (~3%) portion of the total nitrogen load to the bays, while the Gulf of Maine provides more than 90% (Hunt *et al*. 1999).

Here we present the results of WHOI scientists working on a coupled physical/*Alexandrium* population dynamics model of the Gulf of Maine (Stock *et al.* 2005, McGillicuddy *et al.* 2005).

3.1 The WHOI *Alexandrium* model

The WHOI model was created to help understand the spatial disconnect between inshore *Alexandrium* blooms and offshore cyst beds (McGillicuddy *et al.* 2003; Hetland *et al*. 2002). The present formulation of the biological model also includes germination, growth, mortality, and nutrient limitation (Stock *et al*., 2005; McGillicuddy *et al*. 2005).

The model uses observed distributions of cysts overwintering from the preceding year, and calculates cyst germination and vegetative cell growth rates based on ambient conditions of salinity, temperature, light, and NO_3 concentration. You have to tell the model what these ambient conditions are; it does not model them (for example, the model does not attempt to account for uptake or transport of NO_3). Below is a list of sources of information for the ambient data:

submodel	parameter	source of information
biology	cysts	2004 cyst map from WHOI surveys.
physics	wind	2005 wind and heat fluxes from NOAA/NCEP EDAS (6-hourly; 40 km resolution)
physics	river runoff	2005 data from USGS
biology	ambient NO_3	"Petrie99": Petrie *et al*. (1999) calculated the long-term average at many locations throughout the gulf for 4 seasons.
biology	light	2005 data from NOAA/NCEP EDAS
biology	salinity, temperature	physical model

The model was run for the period March to July 2005. **Figure** 3-1 (He *et al.*, submitted) shows the observed *Alexandrium* abundance on four dates in 2005 alongside the model output for comparison. The model performs reasonably well in mid-May but then under-predicts *Alexandrium* abundance by mid-June.

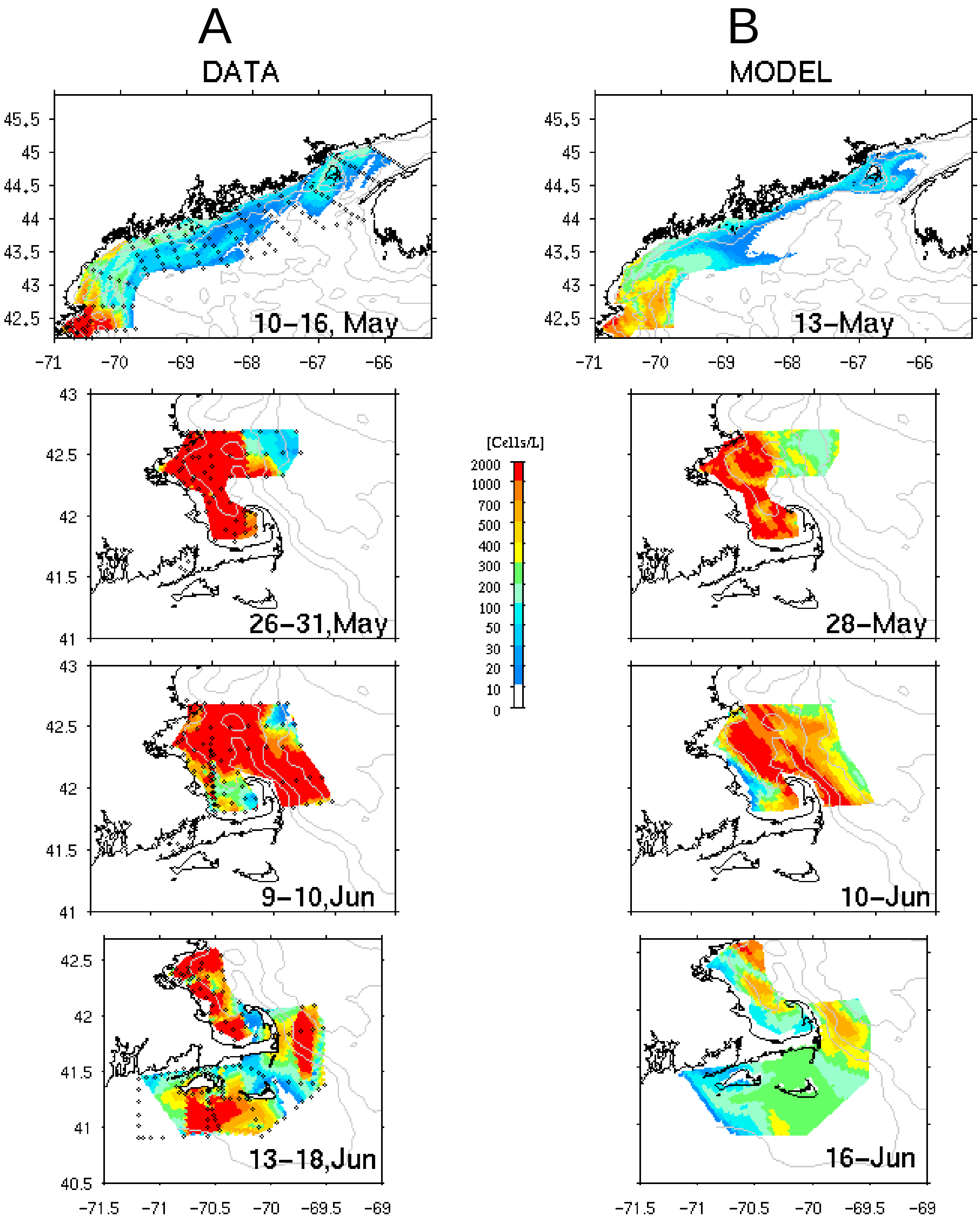

Figure 3-1. Observed versus simulated surface *Alexandrium* cell concentration (left panels versus right), for 4 dates. *In-situ* cell counts were collected by ship surveys in May and June.

3.2 Model Sensitivity Tests

The observations presented in Section 2 suggest that cysts, wind, and river runoff contributed to the 2005 bloom. Model sensitivity tests were conducted with the WHOI *Alexandrium* model to evaluate the importance of those three factors, and where and when they were important. The model sensitivity approach was also used to evaluate the effect of the outfall on the bloom. The questions underlying the four sensitivity runs were:

- How important was overall cyst abundance?
- How important was wind forcing?
- What was the impact of river runoff?
- What was the effect of MWRA effluent nutrients on the bloom?

For comparison with the sensitivity results, we call the run of **Figure** 3-1B the "control run." The sensitivity test for cysts involved using a cyst map from 1997. The sensitivity test for wind used 2004 wind data (2004 was chosen because it was a "typical" year with respect to wind). The sensitivity test for river runoff used 2004 river runoff data (again, a typical year). The sensitivity test for outfall effects blended the Petrie99 NO_3 data with either our 2005 NO_3 data or our 2005 NO_3 + NH_4 data; note that summing NO_3 and NH_4 this way treats them as equivalent for growth in the model.

The following table summarizes what was changed for each sensitivity run:

parameter	control run	**model sensitivity tests**			
		cysts	wind	river	outfall
cysts	2004	**1997**	2004	2004	2004
wind	2005	2005	**2004**	2005	2005
river runoff	2005	2005	2005	**2004**	2005
ambient NO_3	Pietre99	Petrie99	Petrie99	Petrie99	**Petrie99 blended with our 2005 observations of NO_3, and Petrie99 blended with our 2005 observations of NO_3 + NH_4**
see Figure:	3-1 B 3-3 A	3-3 B	3-3 C	3-3 D	3-6 to 3-8

3.3 How important was overall cyst abundance?

The 2005 *Alexandrium* bloom may have been given a head start by the high abundance of newly deposited cysts in western GOM sediments, as documented during a fall 2004 survey (**Figure** 3-2). The only prior cyst survey was in 1997, and concentrations observed at that time were nearly 10-fold lower than in 2004 (**Table** 2-1; Anderson *et al.* 2005b). The source of the high cyst abundance is unknown, as no surveys were conducted between 1997 and 2004. There had been an unusually large *Alexandrium* bloom in September 2004, however, that may have contributed to the overall high abundance of cysts observed in October 2004 (recently deposited and not subject to typical benthic burial over the course of the summer). Likewise, high cell concentrations of *Alexandrium* and high levels of toxicity in shellfish were observed in the Bay of Fundy and eastern Maine in fall, 2003 (Martin *et al.* 2006), perhaps contributing to the high cyst abundance observed in October, 2004.

The model results for the sensitivity run using the 1997 cyst map were only about one fourth of the control run, which uses the 2004 cyst map. To see this, compare the left panels of **Figure** 3-3 to the corresponding right panels of **Figure** 3-1. This suggests that the high 2004 cyst abundance was a primary factor in the 2005 bloom.

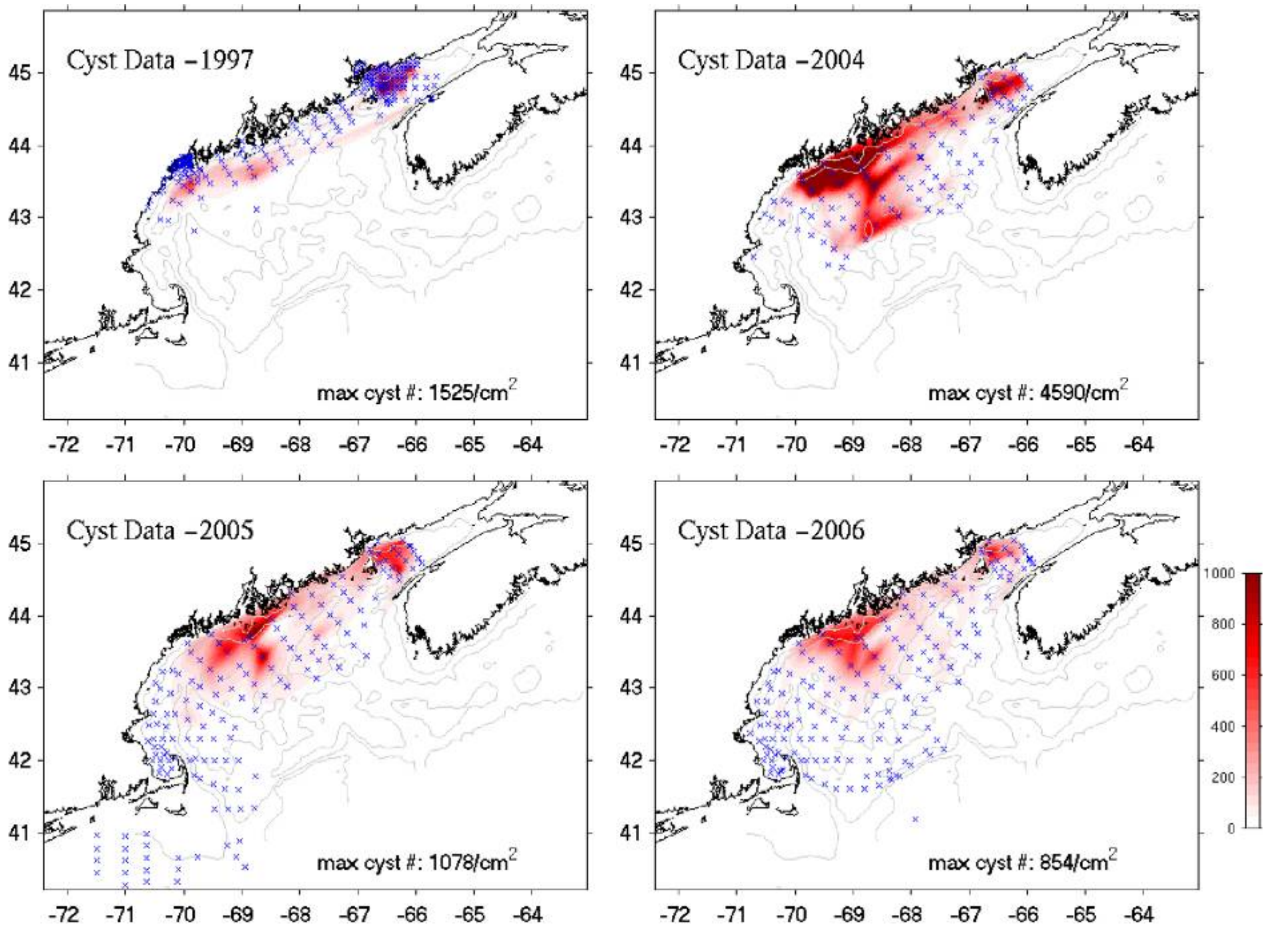

Figure 3-2. *Alexandrium* cyst distributions from 1997 and 2004 as used in the model sensitivity tests. For comparison the 2005 and 2006 maps are also shown. (Source: Anderson *et al.* 2005b, and D.M. Anderson, unpublished data).

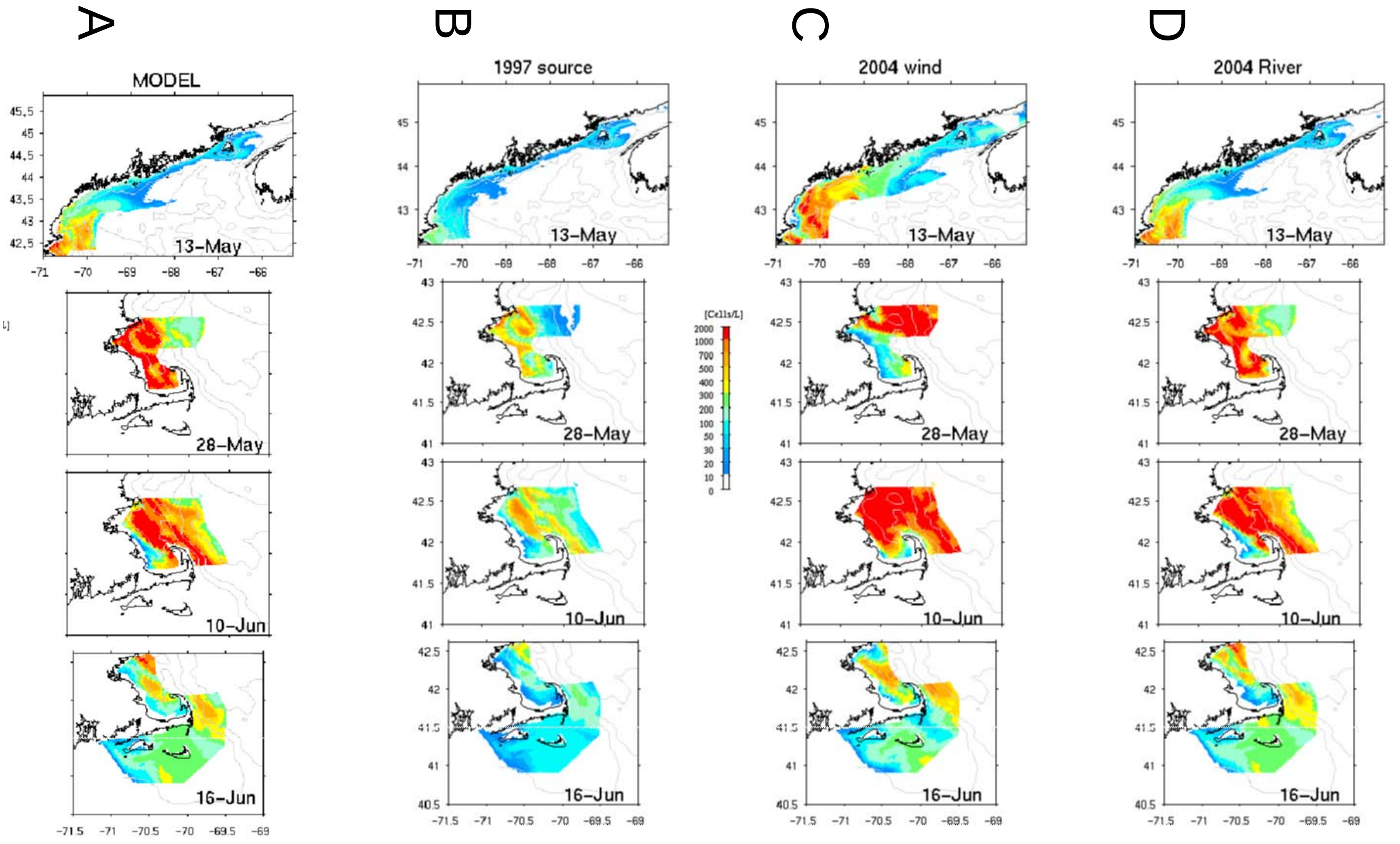

Figure 3-3. Simulated surface cell concentrations from three sensitivity model experiments compared to the control run from Figure 3-1B. The first column of panels shows the control run, the second column shows results based on the 1997 benthic cyst distribution (the control run used the 2004 cyst map). The third column of panels shows the results based on 2004 surface wind forcing (the control run used 2005 winds). The last column of panels shows the results based on 2004 river runoff data (the control run used 2005 river flow).

3.4 How important was wind forcing?

The model results for the sensitivity run using the 2004 surface winds show subtle differences from the control run, which uses 2005 winds (**Figure** 3-3). On May 13 2005 the sensitivity actually showed higher cell abundance than the control run, a result not expected. Since 2004 was a "typical" year for winds this suggests that a strong bloom can occur in the Gulf of Maine regardless of the wind conditions. Closer to the coast, however, there was one notable difference between the sensitivity and the control run - western Massachusetts Bay was impacted earlier in the control run. This suggests that coastal toxicity does depend on particular wind conditions, a premise qualitatively consistent with the results and interpretation of Section 2.1, which emphasized the importance of the unusual 2005 winds to PSP toxicity in Massachusetts Bay. But the absence of PSP toxicity in Massachusetts Bay in 2004 (**Figure** 1-3) reminds us that regardless of winds the main factor for Massachusetts Bay is whether or not there is a major bloom off the coast of Maine.

3.5 What was the impact of river runoff?

The model results for the sensitivity run using the 2004 river flow data look similar to the control run, which uses 2005 river flow data (**Figure** 3-3). This suggests that although the very high flow of Maine rivers in 2005 increased local buoyancy and accelerated the near coastal flow and along-coast transport in the hydrodynamic part of this coupled model, it did not stimulate or concentrate the bloom compared to the more typical river flow of 2004. A caveat to this conclusion is that the sensitivity used the same ambient NO_3 as the control run, as we lacked sufficient river nutrient data to model riverine input of nutrients, whether they are NO_3, other macro nutrients, or micronutrients. Increased riverine nutrient loading would be expected to enhance coastal blooms, but gulf-wide the nutrients entering from offshore via the Northeast Channel far exceed riverine sources.

It is worth noting that the high river flow of 2005 did affect the physical hydrodynamic part of this coupled model, causing increased local buoyancy and accelerated the near coastal flow and along-coast transport. This transport was further accelerated and pushed toward the shore by the strong downwelling-favorable winds of 2005.

3.6 What was the effect of MWRA effluent nutrients on the bloom?

The preceding sensitivity runs of *Alexandrium* model proved useful for evaluating the importance of possible factors causing the gulf-wide bloom. Within Massachusetts Bay, the model was used to assess the relative importance of nutrients from MWRA's outfall. Ambient NH_4 was used as a surrogate for the MWRA discharge, as most (about 80%) of the DIN (dissolved inorganic nitrogen) in the effluent is NH_4. This is a highly conservative assumption, as background DIN also contributes NH_4 (typically about 0.1-2 μM). The estimated effect of NH_4 on the bloom approximates an upper bound of the effect of the outfall.

One caveat about this modeling exercise is that, in its present form, the model uses a fairly coarse nutrient field that is smooth compared to the variability seen between observations. **Figure** 3-4 compares, for May and for August, observations of NH_4 to the smoothed field (calculated as the difference between two other smoothed fields: "Petrie99 blended with $NO_3 + NH_4$" minus "Petrie99 blended with just NO_3"). The smoothed field does not seem to do a good job of matching the observations, but that could result from the fact that the objective analysis used blends in observations from other months not shown in the figure. This may warrant further investigation, but let us proceed with that caveat in mind.

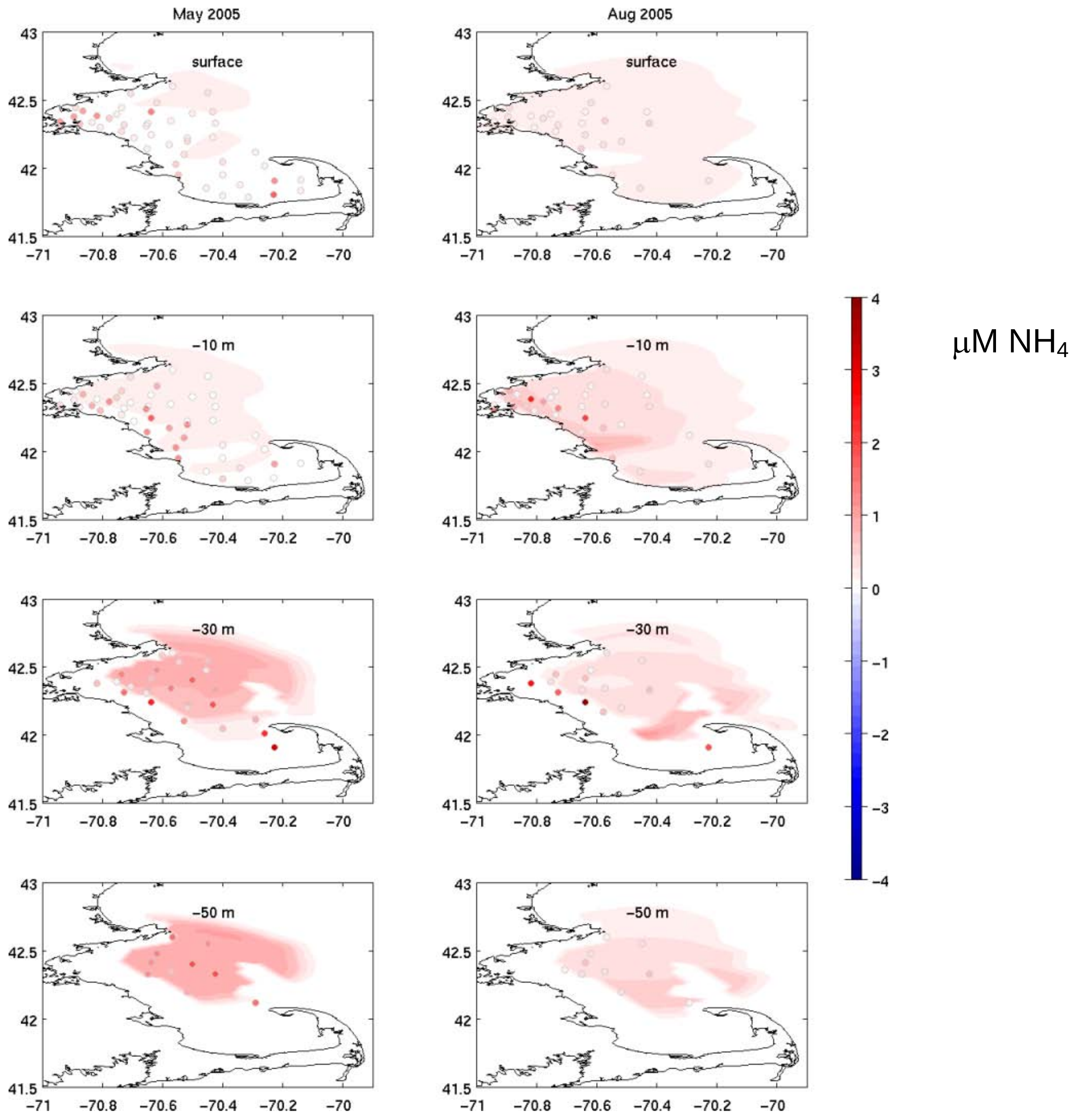

Figure 3-4. May and August 2005 comparison of NH_4 observations (dots) with the smoothed NH_4 field (reddish cloud). The field is used by the model and was calculated using objective analysis.

We modeled *Alexandrium* abundance resulting from growth in two nutrient fields: "with NO_3 + NH_4", and "with NO_3" (i.e. "Petrie99 blended with NO_3 + NH_4", and "Petrie99 blended with just NO_3"). **Figure** 3-5 shows snapshots of the modeled increase in abundance (log scale) due to NH_4. The largest differences of up to ~700 cells L^{-1} were observed in the surface waters on June 3 while subsurface (-10 m) differences peaked at about 100 cells L^{-1} on May 28. Survey measurements indicated that stratification was beginning to strengthen by early June (see **Figure** 2-31) and may have trapped most of the outfall plume below the pycnocline. Low salinity surface waters were also present in northern Massachusetts Bay during this time period due to input of the riverine plume from the Gulf of Maine. Additionally, survey measurements on June 2-4 showed very low NH_4 in the surface waters (whether from the outfall or the Gulf of Maine).

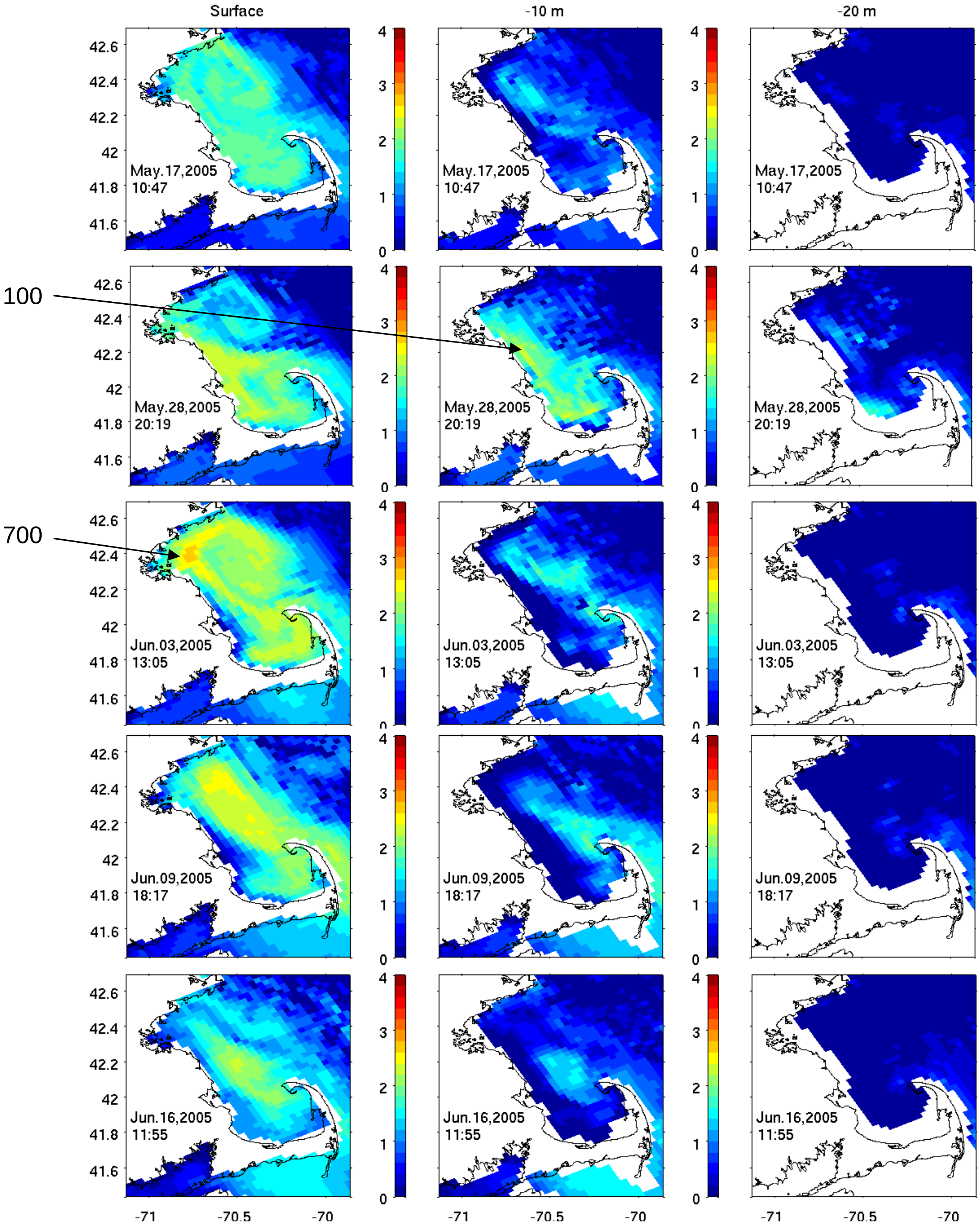

Figure 3-5. Modeled log *Alexandrium* response to NH_4 at surface, 10m, and 20m on 5 dates. The response to NH_4 is calculated as the response to $NO_3 + NH_4$ minus that with just NO_3.

We explored the model results by graphing the average and the maximum abundance for an area. We progressively focused in on the outfall by graphing the results for three regions: the Gulf of Maine, Massachusetts Bay (a subset of the Gulf of Maine), and a “downstream” area near the outfall (a subset of Massachusetts Bay). The “downstream” area is the location that could reasonably be expected to be potentially affected by the effluent plume (**Figure** 3-6). The outfall plume (as characterized by typical distribution of NH_4) is generally observed within 10-20 km of the outfall; the assumed downstream area in the analysis is skewed further to the south than 20 km because circulation during May and June 2005 was on average counterclockwise due to the predominant north and northeast winds. The selection of this area as the "downstream area" for this analysis is problematic, however, because the 2005 storms dispersed the plume more quickly than normal to undetectable levels.

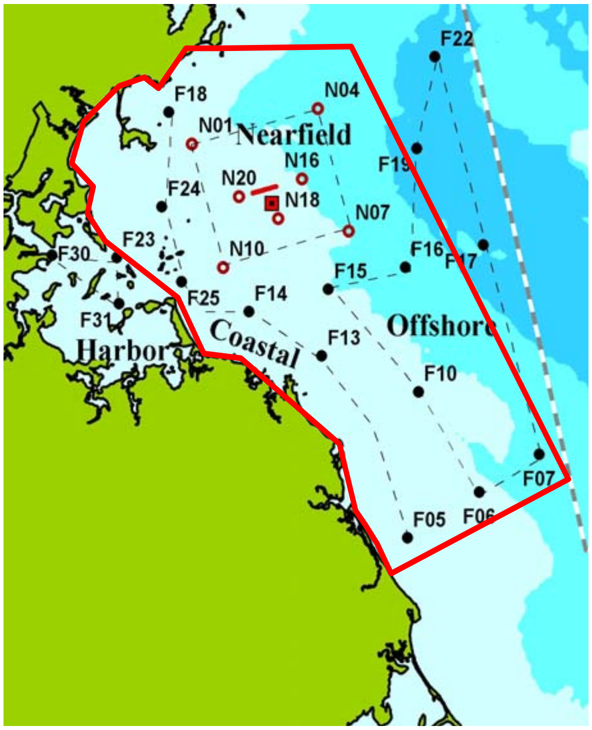

Figure 3-6. “Downstream area" selected for model results evaluation.

Figure 3-7 shows that NH_4 gave a small but noticeable increase in the average cell abundance (the blue line is higher than the red line). The most visible difference in the figure was at the peak on June 2 with an increase of 2, 50, and 100 cells/L (3%, 17%, and 14%) for GOM, Massachusetts Bay, and the "downstream area". **Figure** 3-8 compares the maximum cell abundance. The plots for Massachusetts Bay and the "downstream area" look similar because the maximum was often found within the "downstream area", a subset of Massachusetts Bay. On June 2 there was an increase of 300 cells/L (18%) for those areas. Overall, it seems fair to summarize by saying that overall, the run with NH_4 seems to favor an increase of 10-20% in maximum *Alexandrium* abundance in Massachusetts Bay.

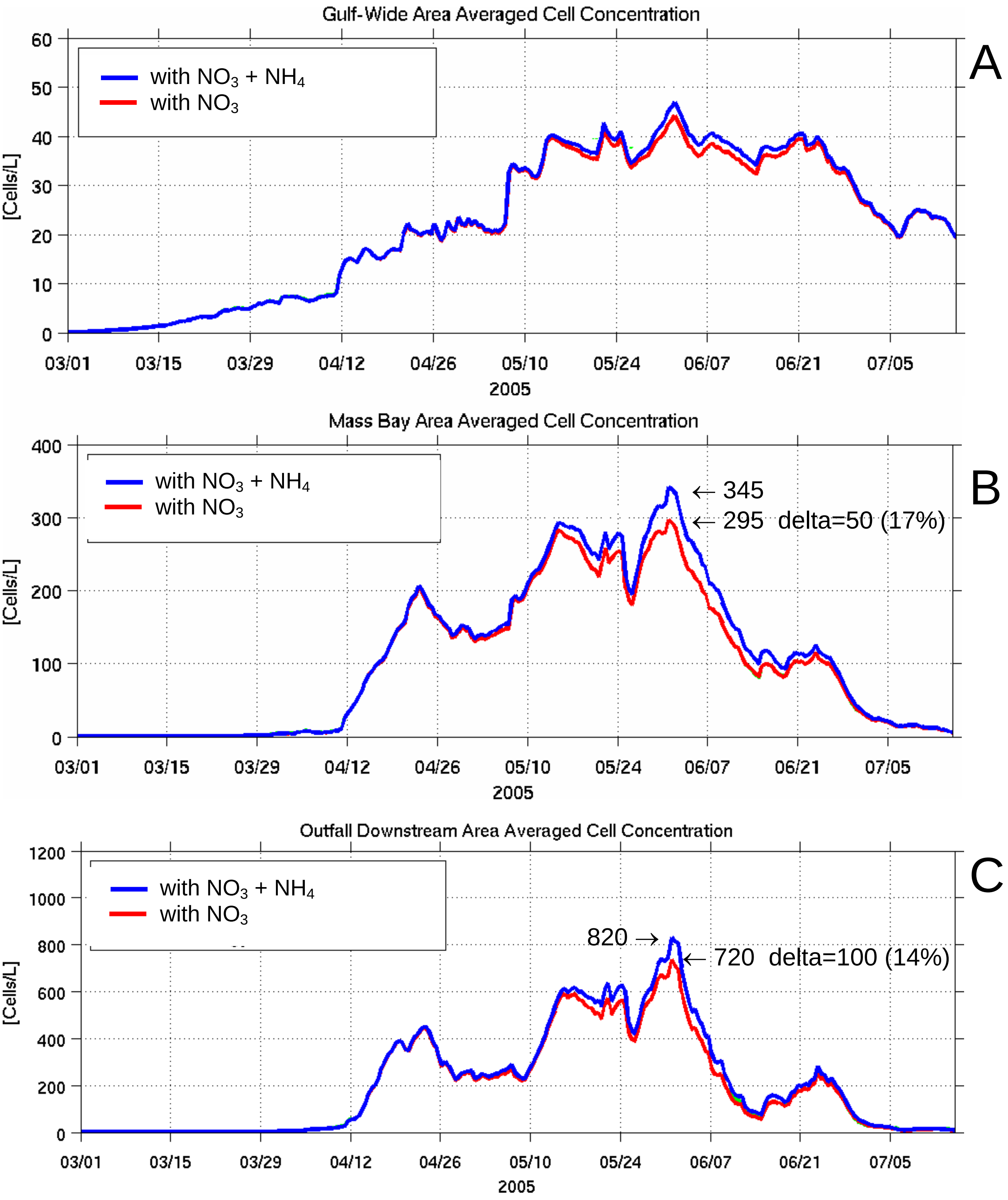

Figure 3-7. Simulation of average abundance with NO_3 + NH_4 versus that with just NO_3. (A) Gulf-wide, (B) Massachusetts Bay, and (C) "downstream area" average *Alexandrium* cell concentrations.

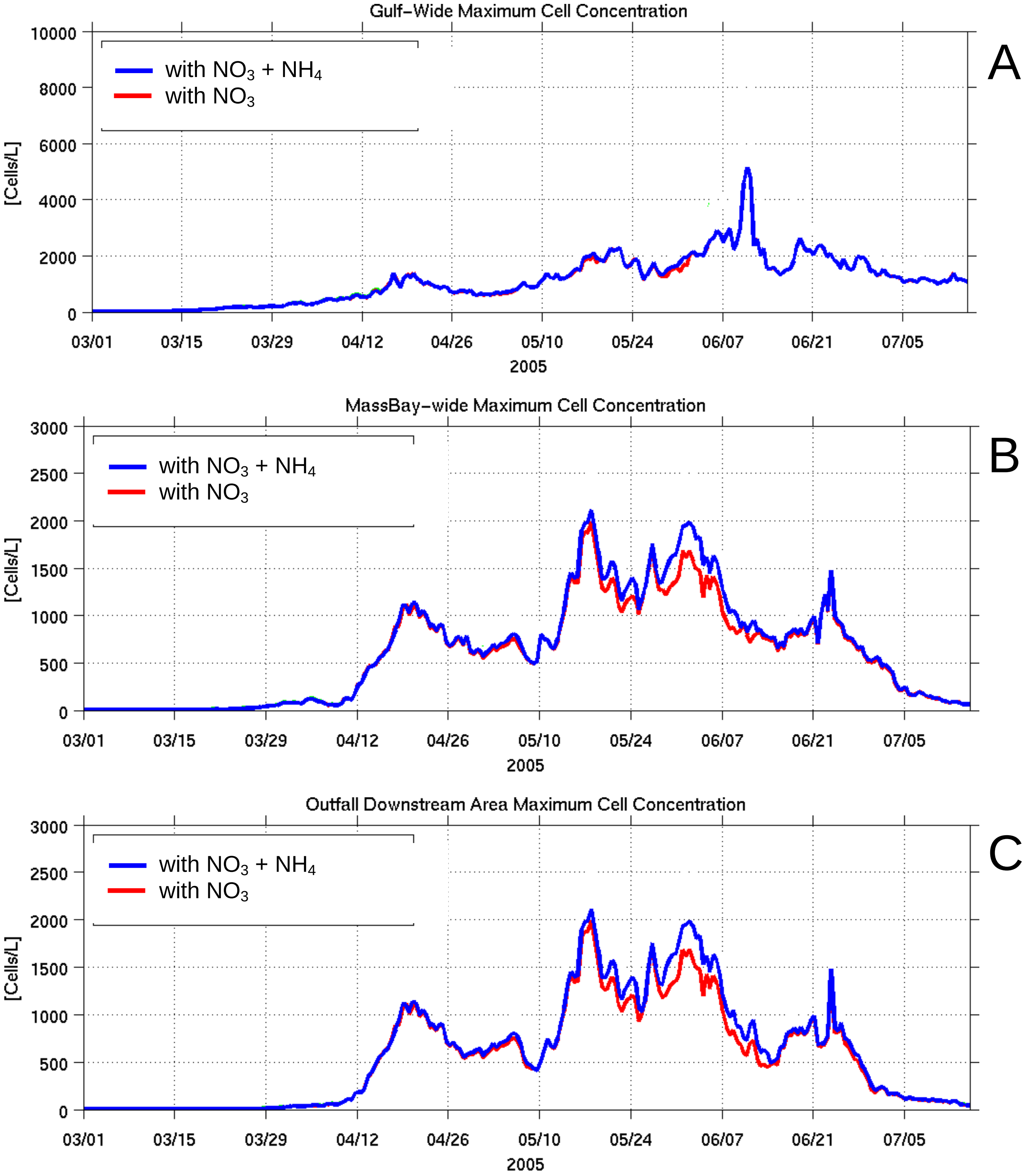

Figure 3-8. Simulation of maximum abundance with NO_3 + NH_4 versus that with just NO_3. (A) Gulf-wide, (B) Massachusetts Bay, and (C) "downstream area" maximum *Alexandrium* cell concentrations.

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4 CONCLUSIONS & RECOMMENDATIONS

The following conclusions and recommendations follow from this study:

- The dynamic physical oceanographic conditions spread the bloom throughout the bay much faster than the gradual progression seen in 1993.
- The storms that brought the *Alexandrium* into Massachusetts Bay also greatly increased the dilution of MWRA effluent. The plume was at times undetectable.
- The 2005 Massachusetts Bay *Alexandrium* bloom occurred within the context of a diverse phytoplankton community that was about 1,000-fold to 400-fold more abundant than *Alexandrium*. The sequence of storm events and synchronous changes in the rest of the phytoplankton community indicate that the bloom in Massachusetts Bay was likely the result of water mass changes rather than phytoplankton species succession within a single water mass.
- The regional bloom was caused by high cyst abundance, according to a numerical model sensitivity study. Closer to the coast, river flow and winds contributed somewhat to the alongshore transport of the bloom.
- The effect of outfall nutrients on the bloom in Massachusetts bay was estimated to be:
  - less than 14% in the outfall “downstream,” according to a numerical model sensitivity study where measured ambient ammonium, (which in reality included ammonium from other sources) was used as a highly conservative surrogate for outfall nutrients.
  - ~3%, according to previous modeling (>90% of nutrients enter the bay from the Gulf of Maine).
  - less than 10% in the centerline of the plume, according to a simple model of oceanic dispersion.
- The lack of a prolongation of the bloom near the outfall when cell abundances declined throughout the bay also argues against any appreciable stimulation by outfall nutrients.
- The WHOI *Alexandrium* model could be improved by
  - using nutrient fields generated by the Bays Eutrophication Model.
  - using nutrient fields that are not so smooth.
  - conducting a sensitivity test in relation to the assumed nutrient uptake and growth kinetics, including possible differences for NO_3 versus NH_4.
- The <u>2006</u> *Alexandrium* bloom was significant in size, and extended into Massachusetts Bay, though the cell abundances and toxicity were not as extensive or high as in 2005. The data from that bloom could be evaluated for outfall effects, including model runs with and without outfall nutrients. Since the procedures for this type of analysis have already been established, and a 2005 cyst map exists, this would not be a major undertaking. It would provide valuable information on possible outfall effects, particularly since the wind and riverine forcings were reduced compared to 2005, and thus the outfall effect might be more visible against this less-dynamic background.
- The conditions that trigger a decision to conduct *Alexandrium* rapid response surveys could be revisited. Presently it is very sensitive (100 cells L^{-1}) and may cause a false alarm.

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