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Production Patterns in Massachusetts Bay with Outfall Relocation

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ABSTRACT: On September 6, 2000, the sewage outfall at the mouth of Boston Harbor was relocated 15 km offshore to alleviate pollution in Boston Harbor. The expected responses for nutrients and water column productivity parameters were for values to decrease at Boston Harbor, to increase at the outfall location, and to remain unchanged at the boundary of the near field region within 5 km of the outfall. Average values for water column measurements from 1992 to 2004 indicated these general responses with some exceptions. At the Harbor, nutrients decreased and chlorophyll increased but ^{14}C primary productivity remained statistically unchanged. At the new outfall location and at the boundary of the near field, nutrients increased and chlorophyll increased but primary productivity remained unchanged. Physical factors affecting primary production, such as spring water temperature, stratification, and wind, had more affects on productivity patterns than nutrients from outfall relocation in the near field. A decrease in summer zooplankton after outfall relocation appeared to be due to a region-wide factor rather than outfall effluents.

Introduction

Long-term monitoring in Massachusetts Bay has allowed an environmental quality evaluation of sewer relocation from Deer Island at the mouth of Boston Harbor to 15 km into the Bay. Comprehensive monitoring was initiated by the Massachusetts Water Resources Authority (MWRA) in 1992 and continues to the present. These data are publicly available in electronic form from the MWRA (www.mwra.state.ma.us) and in the MWRA Environmental Quality Technical Report Series from 1992 to 2004 (e.g., Alber et al. 1993). Outfall relocation occurred on September 6, 2000. Following the relocation, nutrients that formerly entered the Harbor, now flow into western Massachusetts Bay at a depth of 32 m (Fig. 1). This paper summarizes patterns of primary production and factors regulating production prior and since outfall relocation.

Factors controlling productivity in Massachusetts Bay include light, nutrients, temperature, and grazing (Kelly 1997; Keller et al. 2001). Light can be limiting mainly in the colder months when mixing depth exceeds critical depth (Sverdrup 1953; Townsend and Spinard 1986; Siegel et al. 2002). Inshore light can be limiting in any month when runoff or wind and tidal mixing increase turbidity and mixing. Nutrients begin to limit productivity in late spring and through the warmer months when a decreased mixing depth and stratification cap nutrient exchange with deeper waters (Keller et al. 2001). Surface nutrients can also be available in nearshore waters from rivers and sewage outfalls. In the past Boston Harbor exported 88–90% of its nitrogen input to western Massachusetts Bay resulting in a gradient of decreasing nutrient conditions into the Bay (Kelly 1997). Precipitation, runoff, and wind become important factors in regulating the degree of stratification, upwelling, and downwelling nutrient exchange in warmer months (Geyer et al. 1992). During warmer months higher temperatures result in increased phytoplankton growth rates. Phytoplankton biomass and production are typically affected by zooplankton grazing, which is also regulated by temperature. Recently we have discovered that temperature stimulated grazing can regulate the expression of the winter-spring diatom bloom (Deason 1980; Keller et al. 2001). During warmer winters in the early 1990s the winter-spring bloom often had a reduced intensity and duration or failed altogether (Keller et al. 2001). The magnitude of the winter-spring bloom in Massachusetts Bay from 1995 to 1999 was significantly correlated with warmer winter water temperatures and zooplankton abundance over the bloom period February to April. Mesocosm experiments have confirmed that zooplankton grazing stimulated by the warmer temperatures could limit the expression of the winter-spring bloom (Keller et al. 1999). This paper will focus on the question of how much did primary productivity decrease at the old Boston Harbor outfall and increase in the waters of the western Bay as a result of outfall nutrient relocation in view of

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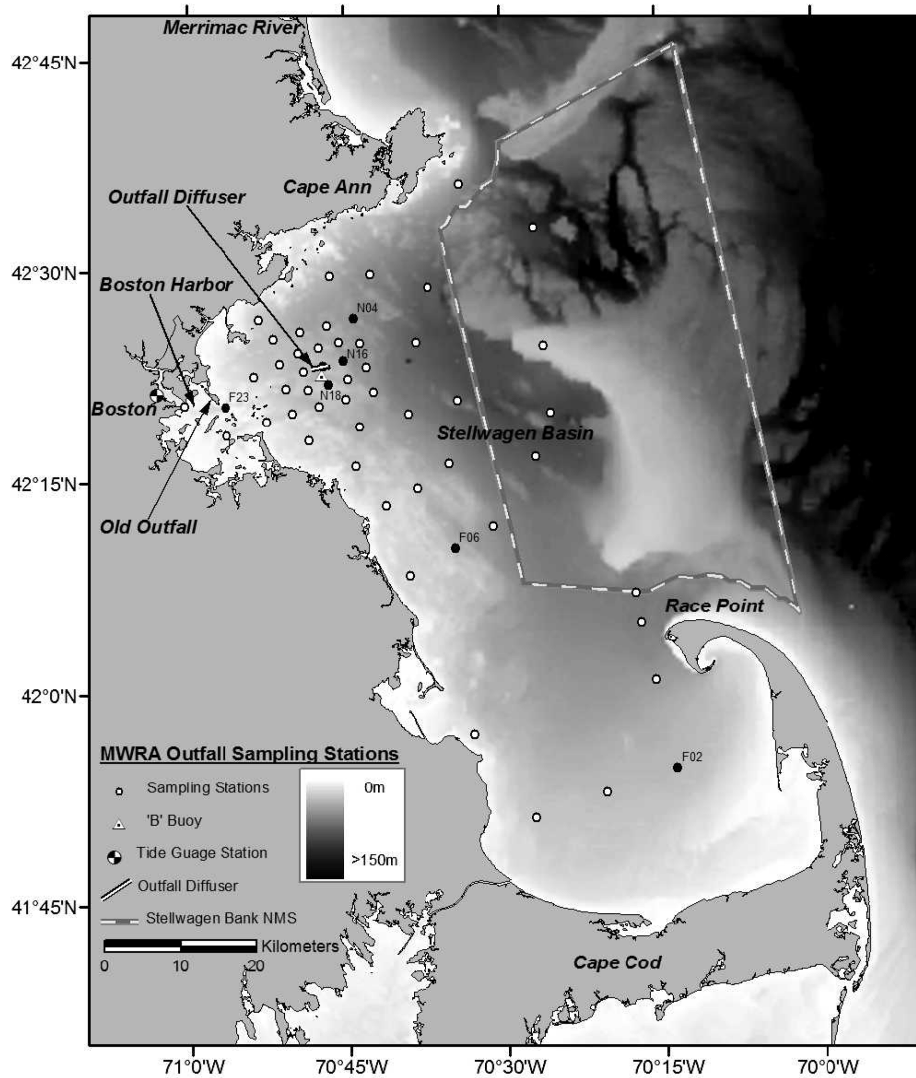

Fig. 1. Massachusetts Bay indicating the location of sampling stations (bold) and features referred to in the text. The dashed line outlines the Stellwagen Bank National Marine Sanctuary (NMS). The depth shading west of the Bank indicates the deeper Stellwagen Basin. Note the location of the NOAA buoy station: 'B'Buoy.

seasonal factors regulating primary productivity. Most of the effort will be in detecting changes in the near field of the new outfall location.

Study Sites

Boston Harbor

Boston Harbor is relatively small, shallow, nutrient-enriched, and dominated by depositional sediments compared to Massachusetts Bay. Before outfall relocation Boston Harbor, with an average depth of 10 m, received 1.5 million m^{-3} of sewage flow per day from the MWRA Deer Island sewage treatment plant and up to 2.7 million m^{-3} during wet weather (Alber et al. 1993), resulting in an average 8 mol N M^{-2} yr^{-1} in the Harbor (Nowicki et al. 1997). Rapid flushing controlled the concentration of dissolved inorganic nitrogen (DIN) in the outer Boston Harbor (annual mean of 10 μM). Flushing also accounted for most of the nitrogen export with denitrification accounting for less than 10% of the annual land derived nitrogen input (Kelly 1997; Nowicki et al. 1997). A flushing time of days to weeks diluted the nutrients to Massachusetts Bay and southward along the coast. Around the periphery of the inner Harbor the concentration of nutrients in Boston Harbor water remained greater indicating relatively poor flushing in these areas due to weaker tidal currents. Knebel et al. (1991) indicated that Harbor sediments were 51% depositional, 29% reworked sediment, and 20% erosional and nondepositional. Sediments from Boston Harbor metabolize about 46% of the organic inputs from phytoplankton production and sewage inputs (Giblin et al. 1997). These authors note that nutrient regeneration from the benthos is equivalent to 40% of the nitrogen, 29% of the phosphorus, and more than 60% of the silicon demand of the phytoplankton. A MWRA monitoring program has been documenting the recovery of Boston Harbor since outfall relocation. This Harbor success story is not the topic of this paper except for the monitoring conducted at the mouth of the Harbor at the old outfall location at Station F23 (Fig. 1). Station F23 was the only location in or near the Harbor where primary productivity studies were conducted.

MASSACHUSETTS BAY

The currents in Massachusetts Bay are driven by tides, wind, changes in water density resulting from the local input of fresh water into Boston Harbor and from the Merrimack River, and low salinity water from the Gulf of Maine (Butman et al. 1992). The area of Massachusetts and Cape Cod Bays is large at 4274 km^2 (Fig. 1). The annual volume of river discharge from the Merrimack River to the north is about 215 m^3 s^{-1} compared to river discharge through Boston Harbor of about 10 m^3 s^{-1}. A counterclockwise flow starts from Cape Ann and across Stellwagen Bank flowing southerly along the coast and northeast out past Race Point on Cape Cod at 20 cm s^{-1}. In the western part of Massachusetts Bay waters move typically 6 to 12 cm s^{-1}. Near bottom current speeds at a depth of 23 m remain between 2 and 4 cm s^{-1} and do not show a seasonal trend. Fluctuations are usually stronger than the mean flow. The area of the new outfall has generally weak currents compared to outer Massachusetts Bay–Stellwagen Bank area. Material at the outfall area is mixed and transported by a variety of processes rather than being swept in a consistent direction (Butman et al. 1992; Geyer et al. 1992).

Maps of the bottom at the outfall area indicate hills on the sea floor have the orientation and elliptical shape of drumlins and are typically covered by glacially derived gravel and boulders (Fig. 1; Butman et al. 1992). Coarse gravel and boulders (2–4 m in size) on these hills cover 23% of the area. Surface sediments on the western side of the area contain fine-grained sediment and represent 6% of the area. About 42% of the area is coarse sand and gravel on the slopes between hills. The remaining 29% of the area is highly variable patches of sand and gravel to smaller grain sediment. Sand ripples with crests oriented north to south may have wave lengths of 4 m and heights of 0.5 m.

The new outfall discharges at a depth of about 32 m through 55 diffuser heads each with 8 outlets creating a diffuser that altogether is 2000 m long with an initial dilution of effluent of about 150:1 (Rex and Connor 1997). In comparison, the Deer Island diffuser in Boston Harbor had a dilution ratio of 14:1. In winter the effluent plume is mixed through the water column to the surface. In summer effluent is trapped below the pycnocline. After initial mixing the effluent is transported horizontally by currents. While surface waters have a residence time of 20–45 d, the deep water in Stellwagen Basin may not be renewed between the onset of stratification and the fall cooling period (Geyer et al. 1992). In the western portion of the Bay in the region of the new outfall, where the near field area is roughly 12 km on a side, upwelling processes may be the most important mechanism of vertical exchange during the stratified portion of the annual cycle (Fig. 1; Geyer et al. 1992).

In anticipation of outfall relocation a monitoring program was initiated by the MWRA in 1992 in Massachusetts Bay although there were pilot studies prior to the regular monitoring program (MWRA 1991). The purpose of the program was to identify ambient conditions in Massachusetts and Cape Cod Bays and to document negative effects if any, on the ecosystem once the outfall came on line by comparison with baseline studies. Over time the monitoring plan has changed and different contractors have conducted the sampling. MWRA has managed the data and made it available to all interested parties. An effort has been made to keep the data, stations, number of samples, and methods consistent over time with more or less success depending on the parameters in question. The types of monitoring included effluent monitoring, water-column monitoring of nutrients and plankton (including primary production in the near field) both in the near field and far field, near field and far field benthic monitoring, monitoring fish and shellfish for contamination and physiological condition, and special studies of water circulation and particle fate, effluent tracers, benthic nutrient flux, and denitrification, effluent plume tracking, and others (MWRA 1991). This paper will focus on water-column monitoring and include the primary production data starting in 1992.

Two previous efforts have examined the productivity patterns in the Boston Harbor-Massachusetts Bay area prior to outfall relocation as part of the MWRA monitoring program. Kelly and Doering (1997) modeled annual rates of primary production using biomass, depth, and light (BzI_o) models developed by Cole and Cloern (1987). Photosynthesis versus irradiation incubations (P versus I) using oxygen served as ground truth for the modeled estimations. Their values ranged from 435 to 546 g C m^{-2} yr^{-1} for the mouth of Boston Harbor and 386 to 620 g C m^{-2} yr^{-1} for the outfall region from 1992 to 1994. Keller et al. (2001) measured primary production rates at the mouth of Boston Harbor and two stations in the outfall area. They used ^{14}C in P versus I incubations, and from the curves at five different depths at each station estimated rates of primary production. Their values ranged from 224 to 1087 g C m^{-2} yr^{-1} at the Harbor and 191 to 612 g C m^{-2} yr^{-1} in the near field region of the outfall from 1995 to 1999. This paper will examine rates of primary production associated with outfall relocation to Massachusetts Bay and the variability coupled with bottom-up and top-down factors affecting productivity rates.

Methods

Primary Production

Primary production rates have been measured using ^{14}C at the Harbor Station and two stations in the outfall region since 1995 to 2005. Data for several years prior to outfall transfer and after will be used to assess the effect of outfall relocation to 15 km offshore. Station F23 is located in 24 m depth at the entrance to Boston Harbor; stations N16 and N18 in 41 and 27 m depths, respectively, are located adjacent to the outfall, and station N04 is located in 49 m depth at the northeast corner of the near field region around the outfall (Fig. 1). The near field was defined by a series of stations located in a square roughly 12 km by 12 km located over the outfall region. Production measurements at station N16 were shifted to N18 in 1997 to better represent the outfall location. Primary production was measured approximately every 3 wk from February to December at the near field stations and at about half that frequency at the Harbor station until 2005 when the frequency of sampling was decreased. The sampling frequency was often dictated by weather, and dates of sampling were highly variable in all seasons.

P versus I incubations using ^{14}C were conducted in temperature-controlled light gradient boxes with water from each station collected at 5 different depths using the procedures of Strickland and Parsons (1972) and Lewis and Smith (1983). Water samples from surface, mid surface, mid depth, mid bottom, and bottom were filtered through 300-μm mesh to remove mesozooplankton grazers and stored in l-l dark bottles and kept cool until returned to the laboratory (< 8 h) for incubation. In the laboratory, 5-ml subsamples in 20-ml borosilicate scintillation vials were inoculated with 1 μCi $NaH^{14}CO_3$. The vials (16 light, 2 dark) from each depth were incubated in a light (250 W metal halide lamps attenuated with neutral density filters) and temperature controlled (within $\pm 2°C$ of in situ temperature) incubator for 1 h. Light intensities ranged to 2000 μE m^{-2} s^{-1}. After incubation, samples were acidified with 100 HCl (0.05 N), loosely capped, and shaken overnight to allow degassing of residual ^{14}C. The following morning 15 ml of scintillation fluid (Universol, ICN Biomedicals, Irving, California) was added to each vial, and samples were placed in the scintillation counter (Beckman model 3801 and since June 2002 Packard 2900 TR) for dark adaptation for 12 h prior to counting. From 1995 to 1997 a different method of incubation was used but the measurements were not found to be significantly different with the more efficient, current method (Keller et al. 2001). Samples for dissolved inorganic carbon (DIC) were taken at each depth and analyzed on a total carbon analyzer (O. I. Corporation model 1010). Primary production (mg C m^{-3} h^{-1}) was calculated following Strickland and Parsons (1972).

P versus I curves were modeled for each station and depth as described by Keller et al. (2001). Briefly in cases where photoinhibition occurred, the model of Platt et al. (1980) was used; when photoinhibition did not occur, the model of Webb et al. (1974) was used with the equivalence equation of Lohrenz et al. (1994). The parameters of the P versus I curves were obtained by least squares using the NLIN procedure in SAS (1988) for each incubation. Hourly production rates were calculated over the daylight period based on the best fit parameters of the P versus I curves, the in situ light intensity at each depth determined from the exponential decay equation, and the photoperiod incident light time series obtained from Deer Island on the nearest sunny day to the incubation. Since in situ intensity was measured only once to estimate the extinction coefficient at each station during a survey, the 15-min light time series from Deer Island was used to estimate primary production over the day light period. Volumetric daily production (mg C m^{-3} d^{-1}) was determined for each depth by summing calculated hourly rates. Daily production (mg C m^{-2} d^{-1}) was obtained by trapezoidal integration of these values over the five different depths. Annual productivity (mg C m^{-2} yr^{-1}) was obtained by integration of daily values over time and extrapolating results to 1 yr.

Hydrographic Surveys

Hydrographic surveys have been conducted about every 3 wk from February to December by MWRA since 1992 to monitor water quality in Boston Harbor and Massachusetts and Cape Cod Bays (Fig. 1). Continuous vertical profiles of the water column and discrete water samples (5 depths) were collected using a Sea-Bird SBE-9 CTD/Bottle Rosette system. Standard profiles included measurements of temperature, salinity, density, fluorescence, and photosynthetically active irradiation (PAR). Depth (m) of the mixed layer was determined from vertical profiles of density. Stratification intensity was estimated as the difference in surface and bottom densities. Light profiles (PAR) were measured using a Biospherical QSP-200L spherical quantum scalar sensor mounted on top of the hydrocast rosette and a Biospherical QSR-240 hemispherical quantum scalar sensor for simultaneous on-deck measurement of incident irradiance (PAR). In situ fluorescence profiles were measured with a WetLabs WetStar fluorometer. At each station, discrete depths were sampled for DIC and nutrients, chlorophyll *a* (chl *a*), phytoplankton concentration, and productivity. Zooplankton samples were obtained by vertical-oblique tows of the upper two-thirds of the water column (maximum tow depth 30 m). Additional temperature data were available from the National Oceanic and Atmospheric Administration (NOAA) #44013 buoy near station N18, with sensors located at the surface, 5 m, and 20 m (Fig. 1).

Nutrients and Chlorophyll

Samples for the determination of dissolved inorganic nutrients (NH_4, NO_3, NO_2, SiO_4, PO_4) were filtered through 47-mm Nuclepore membrane fiber filters (0.4 µm pore size) and frozen until analysis (< 1 mo) in 60-ml polyethylene bottles. Samples were analyzed on a Technicon Autoanalyzer using standard procedures (Strickland and Parsons 1972). Primary and secondary standards in seawater were used to calculate nutrient concentrations (Oviatt and Hindle 1994). Standard deviations were 0.03 µM for NH_4, 0.01 µM for NO_2 and NO_3, and 0.05 µM for SiO_4. Chlorophyll concentrations (µg l^{-1}) at productivity stations have been determined by acetone extraction (without grinding filters in 2003 only) and analyzed on a Turner Designs model 10AU fluorometer (Yentsch and Menzel 1963). From 1995 to 1997, chl *a* concentrations at productivity stations were calculated by regression from in situ fluorescence.

Phytoplankton Composition

Whole water samples were obtained from the surface Rosette sampling bottle for phytoplankton identification and enumeration. The 850 ml samples were preserved in Utermöhl's solution and allowed to settle in a cylinder with a 5 to 1 height to width ratio until analysis with an Olympus BH-2 compound microscope with phase contrast optics. Phytoplankton abundance was estimated from the volume examined in a Sedgwick Rafter cell.

Zooplankton Abundance

Vertical oblique zooplankton tows were conducted through the upper 30 m of the water column with a 0.5 m diameter 102 µm mesh net equipped with a flow meter. Zooplankton were preserved in 10% buffered formalin, transferred to 70% ethanol, and reduced to aliquots of at least 300 organisms with a Folsom plankton splitter. Zooplankton were identified to stage and species and were counted (Albro et al. 1998; Libby et al. 2002).

Results

Annual Primary Production Rates

Estimated annual rates of primary production from 1992 to 2004 ranged from 191 to 664 g C m^{-2} yr^{-1} in the region adjacent to the new outfall and from 224 to 1087 g C m^{-2} yr^{-1} at the Harbor Station (Table 1). After relocation of the outfall in late 2000 no apparent increase occurred in annual values at the new outfall location. No statistically significant decrease occurred at the Harbor mouth. Annual values prior to and after outfall relocation were not significantly different (0.05 level). The lowest values occurred in 1998, the year of no winter spring bloom as discussed in Keller et al. (2001). No year showed consistently high values for all stations although 1996 and 1997 both had higher than average values. The average values were 439 g C m^{-2} yr^{-1} at the near field and 557 g C m^{-2} yr^{-1} at the Harbor mouth.

Average Patterns for Parameters Prior and Post Relocation

The expected responses to outfall relocation for nutrients and water column productivity parameters were for values to decrease at Boston Harbor (F23), to increase at the outfall location (N16, N18) and to remain unchanged at the boundary of the near field region (N04; Fig. 1). Average values for water column measurements from 1992 to 2004 indicated this general response with some exceptions (Fig. 2 and Table 2). At the Harbor station DIN, PO_4, and SiO_4, and particulate organic carbon (POC) decreased. Chlorophyll increased at the Harbor station while primary production and zooplankton did not change. At the new outfall stations N16 and N18, chlorophyll, DIN, PO_4, and POC increased. Zooplankton abundance and SiO_4 decreased and primary production did not change significantly at the outfall. In the northeast corner of the near field at station N04, small average increases were mea-

TABLE 1. Integrated rates of annual primary production (g C m^{-2} yr^{-1}) and daily range (g C m^{-2} d^{-1}) measured in Massachusetts Bay (F23 Harbor Station, N16, N18 near new outfall, N04 northeast corner of the near field around the outfall, see Fig. 1) using ^{14}C tracer incubations and using light from the nearest sunny day. *Data from 1992 to 1994 were modeled and derived from Kelly and Doering (1997). Values from 1995 to 1999 are slightly different than Keller et al. (2001) due to an improved annual integration method. ** Outfall relocation. na – not available.

Year	N04	N16, N18	F23	Average
1992*	na (0.09–2.63)	386 (0.14–2.58)	546 (0.12–2.65)	466
1993*	na (0.18–4.31)	527 (0.17–5.36)	527 (0.16–3.44)	527
1994*	na (na)	440 (0.29–3.98)	440 (0.14–2.44)	440
1995	390 (0.10–1.90)	544 (0.09–3.85)	763 (0.18–7.58)	566
1996	533 (0.07–2.95)	482 (0.13–3.05)	1,087 (0.22–5.20)	701
1997	480 (0.27–3.12)	612 (0.22–5.02)	862 (0.02–2.48)	651
1998	191 (0.15–1.65)	213 (0.19–2.51)	224 (0.11–1.12)	209
1999	395 (0.48–2.31)	503 (0.29–3.71)	658 (0.27–3.23)	519
2000	511 (0.42–3.12)	664 (0.51–5.03)	494 (0.14–4.38)	556
2001**	569 (0.50–3.50)	559 (0.39–3.48)	404 (0.20–2.00)	511
2002	532 (0.30–3.69)	607 (0.30–4.86)	587 (0.52–3.17)	575
2003	295 (0.16–1.53)	293 (0.11–2.52)	311 (0.15–1.18)	300
2004	247 (0.32–2.24)	207 (0.19–1.40)	332 (0.25–1.39)	262
2005	343 (0.26–1.49)	244 (0.16–1.28)	251 (0.04–1.36)	279

sured in DIN, PO_4, and POC but not in primary production, chlorophyll, zooplankton, or SiO_4 (Table 2).

Plankton Composition

Seasonal patterns of plankton community composition did not change in the near field prior and post outfall relocation (Figs. 3 and 4). Throughout the year, the phytoplankton community was dominated by microflagellates and diatoms, accounting for 45% to 70% and 20% to 35% of the total, respectively (Fig. 3). In the spring *Phaeocystis* sp. was a major component of the community, whereas dinoflagellates were only a minor component throughout the year. A principle components analysis found no significant differences in zooplankton community composition prior and post outfall relocation to the near field (Cropp et al. 2003). Throughout the year, the macrozooplankton community was dominated by cyclopoid copepods with *Oithona* sp. the most dominant at 15% to 30% of the total over all seasons and years. Other dominant copepods included *Calanus* sp., *Pseudocalanus* sp., and *Centropages* sp. The other zooplankton category included mainly meroplankton with abundances ranging from 8% to 30% of the total while other cyclopoid copepods, pelagic tunicates, rotifers, etc. were present in usually low abundance. High interannual seasonal variability in all plankton components occurred and no changes in patterns of composition were apparent prior and post outfall relocation (Figs. 3 and 4).

Bottom-up Factors Affecting Primary Production in the Near Field

Natural seasonal variability in mixing and stratification generally governs rates of primary production in Massachusetts Bay. An example year 2002 was used to illustrate the general pattern of primary production and stratification (Fig. 5). During deep winter mixing productivity remained low. Stratification usually began in April with high nutrients in the surface water and extended to October when dissolved nutrients were mixed to surface waters. In April and May stratification intensified allowing initiation of the winter-spring bloom. With continuing stratification during summer, nutrients and productivity dropped to low levels. In late summer early fall mixing events brought nutrients to the surface fueling two and even three productivity peaks before deep mixing decreased productivity to winter levels. These patterns of mixing and stratification resulted in seasonal nutrient pulses and productivity repeating on an annual basis.

Spring and fall blooms did not necessarily exceed the rates of productivity during the summer months (Fig. 6). The winter-spring bloom occurred in most years with prominent exceptions, e.g., in 1998, 2003, and 2004. If summer stratification remained undisturbed, summer productivity became low (Figs. 5 and 6). During some summers, such as 1997, 1999, 2000, and 2002, summer productivity equaled or exceeded the productivity of other seasons (Fig. 6). Years such as 1998 and 2003 with extreme stratification tended to have lower annual values of primary production (Figs. 6 and 7). The stratification versus primary production regression relationship was significant at the 0.01 level according to a SAS regression model ($R^2 = 0.32$, $n = 28$, $F = 12.12$, $p > 0.0018$). Wind can be a factor in summer mixing events. Years with strong wind gusts (wind speeds > 5 m s^{-1} for more than 8 s) such as 1997 tended to have higher productivity compared to years with weak wind gusts such as 1998 and 2004

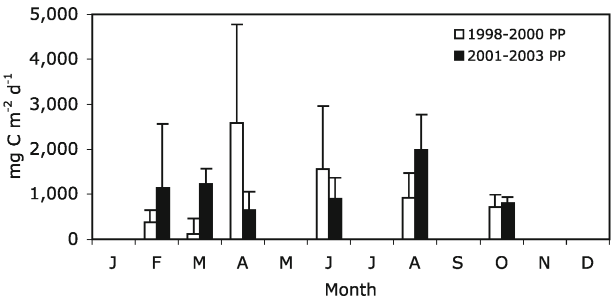

Fig. 2. Primary production (PP) rates at the mouth of Boston Harbor at station F23 from 1998 through 2000 prior and 2001 through 2003 subsequent to outfall relocation. Error bars are standard deviation.

(Fig. 8). This relationship was significant at the 10% level according to SAS regression model 1 (DF = 9, F = 3.48, p = 0.0992). Large fall blooms or a series of blooms resulted after breakdown of the thermocline by wind mixing. Every year a fall bloom occurred that often equaled or exceeded the intensity of the spring bloom, as in 1997, 2001, and 2003 (Fig. 6).

Top-down Control on Primary Production

High summer zooplankton concentrations might be expected to correlate with low levels of summer phytoplankton (Figs. 6 and 9). Zooplankton achieved highest levels of abundance during the warmer months of stratification during 5 of the 7 years from 1997 to 2003 at N18 (Fig. 9). Summer concentrations ranged from 40,000 to 100,000 individuals m^{-3}. During warm months a slight positive correlation ($R^2 = 0.38$, n = 7, F = 3.11, p > 0.14) occurred between levels of primary production and levels of copepod abundance (Fig. 10). Zooplankton peak abundances of about 40,000 individuals m^{-3} apparently controlled phytoplankton productivity in the warm winter of 1998 (Keller et al. 2001). No correlation between primary production and zooplankton abundance was evident in colder months. Spring abundance of zooplankton never exceeded summer abundance but exceeded fall abundance twice during 1999 and 2002.

During 2001 to 2003 zooplankton decreased in Massachusetts and Cape Cod Bays. Adjacent to the new outfall at N18 zooplankton abundance decreased significantly after outfall relocation (Table 2). This decrease occurred in all seasons but was most dramatic in the summer months (Fig. 11). The same trend (not statistically significant) of decreased zooplankton abundance after outfall relocation, particularly in summer, also occurred at N04 in the northeast corner of the near field (Fig. 12). At this location primary production remained the same while chlorophyll and POC increased after outfall relocation (Table 2). Reduced zooplankton grazing and nutrient increase

TABLE 2. Average values compared for years prior and after relocation: 1992 to 2000 and 2001 to 2004. Dissolved inorganic nitrogen (DIN, μM), dissolved inorganic phosphate (PO_4, μM), dissolved inorganic silicate (SiO_4, μM), primary production (PP, mg C m^{-2} d^{-1}), chlorophyll *a* (Chl *a*, μg l^{-1}), particulate organic carbon (POC, μM), and zooplankton (Zoopl, no. m^{-3}; prior and post outfall data from 1998 to 2000 and 2001 to 2003) are shown. PP and Chl *a* were converted to logarithms to normalize the data prior to testing. *t* test significant for the 95% confidence interval (*) and 99% confidence interval (**).

		Relocation						
		Prior			After			
Station	Parameter	n	Standard Error	Mean	n	Standard Error	Mean	
F23	DIN	521	0.221	3.91	167	0.102	1.13	**
	PO_4	519	0.017	0.87	178	0.023	0.60	**
	SiO_4	519	0.2	6.91	178	0.2	5.42	**
	PP	59	0.069	1,422	24	0.073	1,054	
	Chl *a*	243	0.03	2.49	115	0.028	3.74	**
	POC	521	0.47	24.59	167	0.48	1,9.35	**
	Zoopl	51	6,971	32,688	17	6,726	28,793	
N16, N18	DIN	1,198	0.026	0.65	436	0.144	2.34	**
	PO_4	1,198	0.009	0.58	436	0.017	0.66	**
	SiO_4	1,198	0.098	5.18	436	0.135	4.37	**
	PP	116	0.039	1,453	63	0.042	1,069	
	Chl *a*	491	0.023	2.19	313	0.02	2.51	**
	POC	1,198	0.061	3.26	436	0.103	4.04	**
	Zoopl	111	4,041	45,762	50	3,138	30,267	*
N04	DIN	1,156	0.025	0.57	436	0.057	1.05	**
	PO_4	1,127	0.009	0.61	464	0.013	0.67	*
	SiO_4	1,127	0.095	5.05	464	0.141	4.98	
	PP	115	0.051	1,066	62	0.044	1,073	
	Chl *a*	544	0.023	1.54	313	0.028	1.9	
	POC	1,156	0.044	2.83	436	0.089	3.31	**
	Zoopl	102	4,415	48,797	51	3,901	33,123	

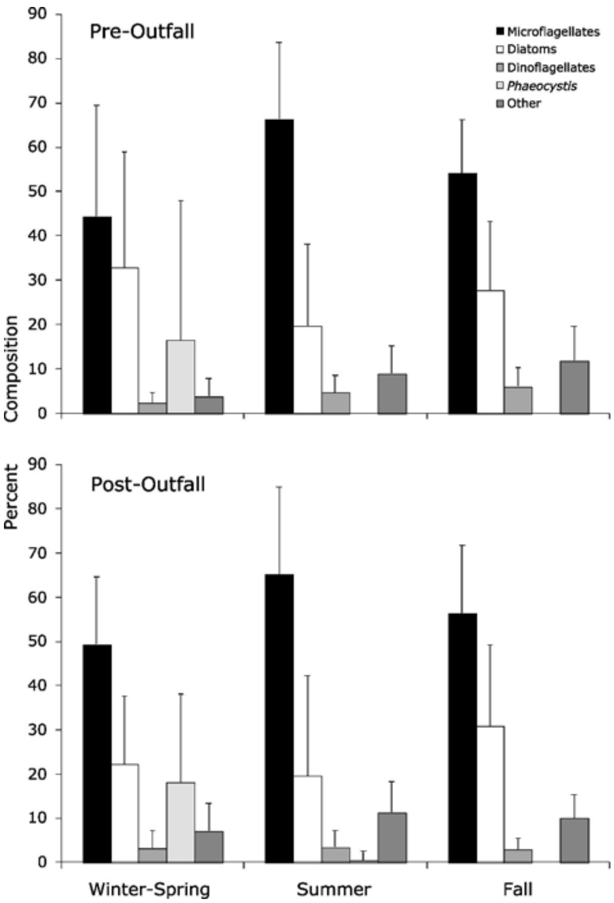

Fig. 3. Phytoplankton community composition prior and post outfall relocation displayed seasonally from 1998 to 2000 and 2001 to 2004. The data were from stations N16, N18, and N04 in the near field. Composition was averaged by season: spring = February through April, summer = May through September, and fall = October through December. Error bars are standard deviation.

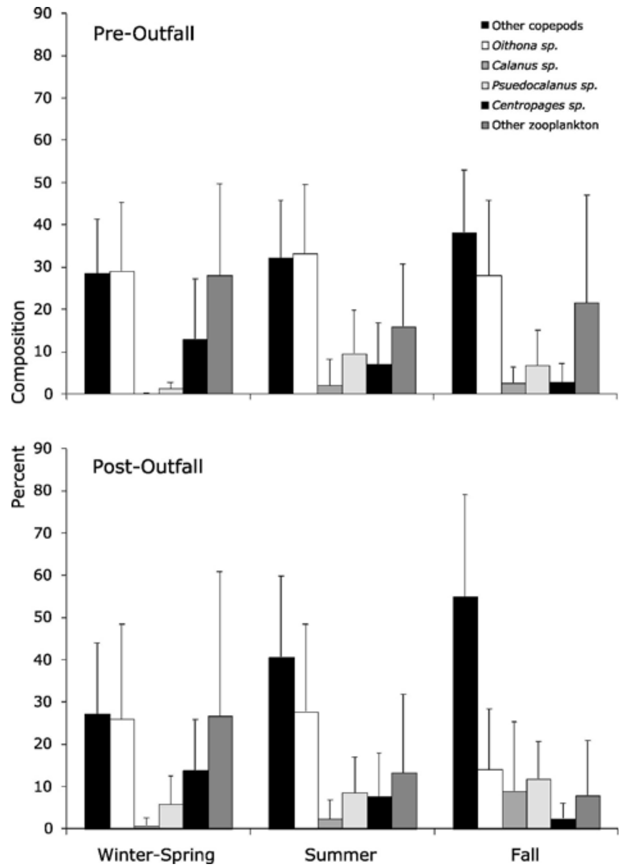

Fig. 4. Zooplankton community composition prior and post outfall relocation displayed seasonally from 1998 to 2000 and 2001 to 2004. The data were from stations N16, N18, and N04 in the near field. Composition was averaged by season: spring = February through April, summer = May through September, and fall = October through December. Error bars are standard deviation.

may explain this pattern after outfall relocation. At all stations from the near field (N16, N18, N04)) to Cape Cod Bay (F23, F06, F02) summer levels of zooplankton were significantly reduced after outfall relocation (analysis of variance $F = 10$, $p = 0.0025$ level; Fig. 13).

Discussion

Outfall Relocation

The effect of outfall relocation on individual measurements of water column parameters was not remarkable compared to the natural variability in Massachusetts Bay. The effect of outfall relocation was most dramatic at the Harbor mouth where nutrients throughout the water column significantly decreased by 72%. Nutrients throughout the water column did increase significantly at the outfall site and this increase extended to the northeast corner and probably other stations in the near field area (Table 2). The increase in nutrients in the water column did not lead to an increase in primary production in the near field either on an annual basis or in individual measurements. Plankton community composition did not change in the near field with outfall relocation (Figs. 3 and 4). The decreased macrozooplankton abundance and reduced grazing at the outfall in conjunction with increased nutrients may explain slightly increased chlorophyll and POC concentrations after outfall relocation. Summer stratification tended to restrict the nutrients beneath the euphotic zone in warm months, while winter mixing dispersed the nutrients in cold months as the outfall designers intended.

Seasonal Variability

While light and deep mixing may limit production in winter (no measurements were made in

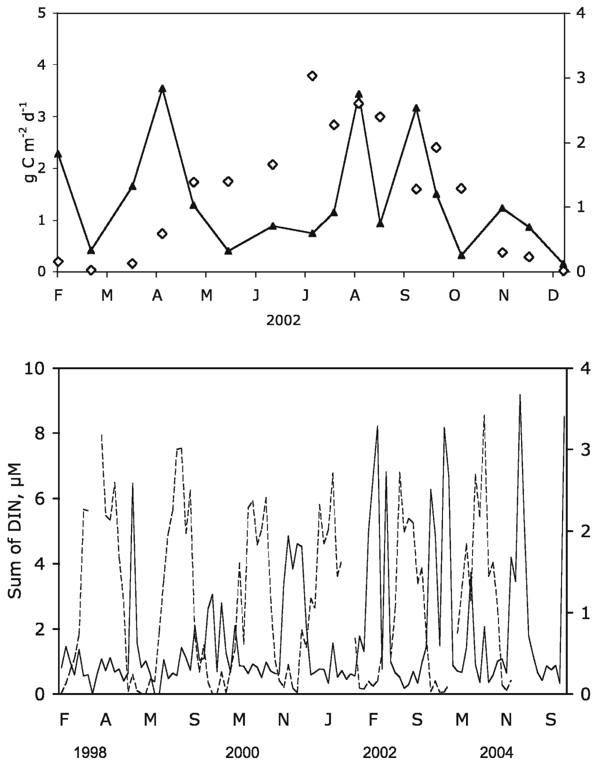

Fig. 5. Primary production rates in 2002 in Massachusetts Bay near the new outfall at station NO4 (triangles) and the intensity of stratification. Stratification intensity was estimated as the difference in surface and bottom densities (diamonds). Dissolved inorganic nitrogen (DIN) concentrations (bold line) and the intensity of stratification (light line) from 1995 to 2003 at station N18 adjacent to the outfall. Outfall relocation occurred on September 6, 2000.

January), seasonal peaks of productivity primarily reflected nutrient availability during the spring initiation and fall loss of stratification, respectively (Fig. 5). The onset of stratification in early April, in combination with available nutrients brought to the surface by deep winter mixing, controlled the initiation of the spring bloom. The winter-spring bloom occurred in most years with prominent warm

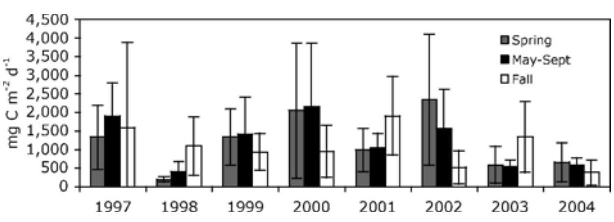

Fig. 6. Seasonal rates of primary production in Massachusetts Bay in the near field of the new outfall from 1997 to 2004. Daily rates were averaged by season: spring = February through April (n = 4–6), summer = May through September (n = 8–10), and fall = October through December (n = 4–6). Outfall relocation occurred in September 6, 2000.

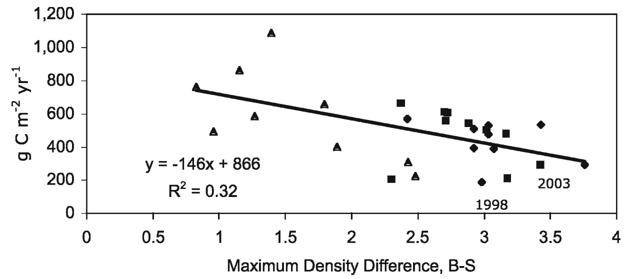

Fig. 7. Annual primary production in Massachusetts Bay in the near field at stations F23 (triangles), N18 (squares), and N04 (diamonds) as a function of maximum annual stratification (bottom density minus surface density) from 1995 to 2004.

winter exceptions, e.g., in 1998 (Keller et al. 2001). During warm winters the loss of the winter spring bloom has been attributed to zooplankton grazing (Keller et al. 2001). In 1998 the lack of a spring bloom was followed by rain in June and extremely strong stratification. This cap limited nutrients and productivity all summer (Fig. 6). These spring blooms use the available nutrients in the euphotic zone so that during the warm months of stratification, productivity tended to level off at lower rates (Fig. 5). Any strong wind mixing resulted in elevated productivity (Fig. 8). Southwest winds that predominate in summer may cause up-welling along the coast; alternatively northwest winds may cause down-welling and mixing along the coast (Geyer et al. 1992). Both of these mixing events may bring deeper waters and nutrients to surface water during summer. Large fall blooms or a series of blooms resulted after breakdown of the thermocline by wind mixing and the increase of nutrients into surface waters. An infrequent sampling schedule may have missed some of these fall blooms in, e.g., 1998 when satellite remote sensing indicated

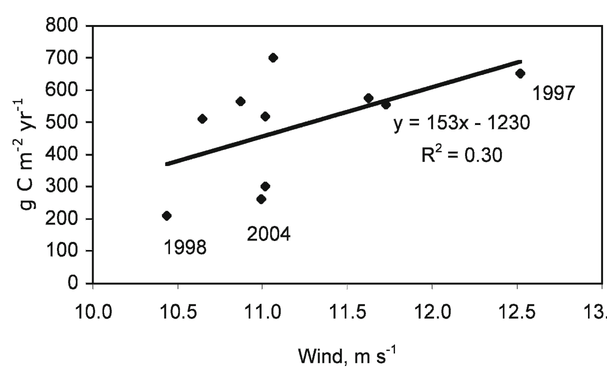

Fig. 8. Annual primary production in Massachusetts Bay in the near field at station N18 as a function of wind gusts from 1995 to 2004. Wind gusts were standard measurements defined as > 5 m s^{-1} for more than 8 s and a daily mean of 6 or more measurements at NOAA buoy station 44013.

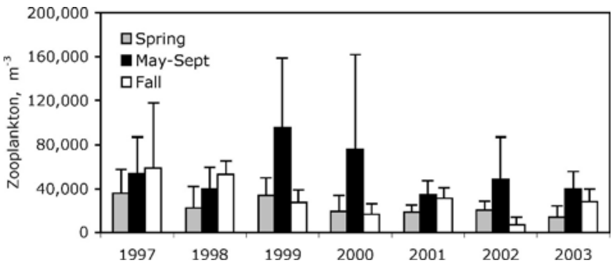

Fig. 9. Seasonal mean total (excluding fish larvae and medusa) zooplankton in Massachusetts Bay at the near field station N18 from 1997 to 2003. Spring is February to May (n = 17). Summer is May through September (n = 27). Fall is September to December (n = 16). Outfall relocation occurred on September 6, 2000.

elevated chlorophyll levels during an unsampled period (Hyde 2006).

Zooplankton Decrease

One explanation for the decrease in summer zooplankton after outfall relocation might be the movement north and increased abundance of *Ctenophora*, predators on zooplankton, with the recent warming of coastal waters (Link and Ford 2006). The ctenophore, *Mnemiopsis ledyi*, was formally noted in Massachusetts Bay during a workshop meeting in 2000 with the MWRA monitoring program. Limited data (below) has suggested that they may have been most abundant in the fall rather than the summer period when the zooplankton decrease occurred. A search of the zooplankton data from the MWRA monitoring program revealed their presence recorded in June 1993, October 1997, October 2003, and August 2004 (Libby personal communication). Only in 2002 were the ctenophores abundant enough to be a nuisance and screened from the zooplankton samples: August 2002 (13 stations), October 2002 (2 stations), and November 2002 (1 station; Libby personal communication). The evidence for a warming trend north

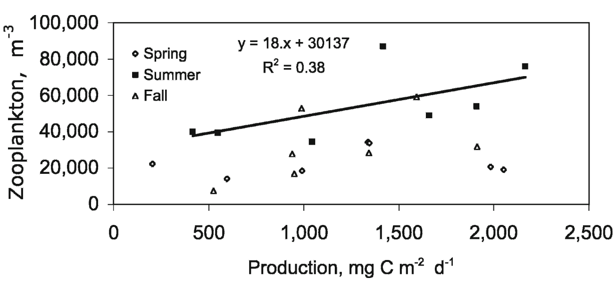

Fig. 10. Seasonal mean total zooplankton as a function of seasonal mean primary production from 1997 to 2003. Spring is February to May. Summer is May through September. Fall is September to December. The regression was for May to September data. The R^2 for all points was 0.006.

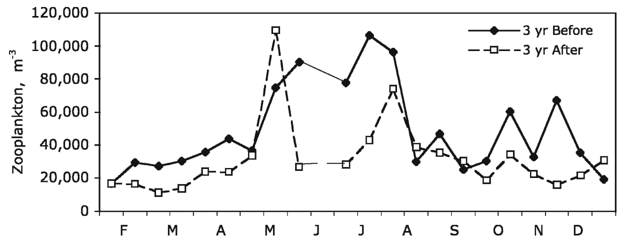

Fig. 11. Zooplankton prior and subsequent to outfall relocation in Massachusetts Bay station N18.

of Cape Cod was not as strong as south of Cape Cod where mean annual temperatures have increased by a little over 1°C from the 1950s to the late 1990s (Oviatt 2004). Our regression analysis has indicated that annual temperatures at the NOAA tide gauge in Boston (station # 8443970) have increased about 0.8°C from the 1950s to the mid 1980s (data not shown). With recent cooler winters from the mid 1990s to the present, our regression analysis indicated no temperature trend at the NOAA buoy (station #44013) in Massachusetts Bay (data not shown). It is possible that warmer winters like 1998 (Keller et al. 2001) may have allowed the northward movement of the ctenophore, *M. ledyi*. It is also possible that the ctenophore was moved north of Cape Cod in ballast water and has managed to survive the temperature change (Travis 1993).

Evidence that predation by this ctenophore on zooplankton could have recently changed summer cycles of zooplankton abundance in Massachusetts Bay has strengthened. The frequency of *Ctenophora* including *Manemiopsis leidyi* has increased in the Massachusetts Bay area by up to 8 times since the early 1980s (Link and Ford 2006). The authors attribute the persistent and widespread increase to local warming and over fishing. To the southwest of Cape Cod in Narragansett Bay, *M. ledyi*, which used to be abundant in fall, has become most abundant in early summer with the climate warming trend (Oviatt 2004; Costello et al. 2006). High levels of summer ctenophore abundance have been correlated with dramatically reduced levels of zooplankton abundance in Narragansett Bay (Costello et al.

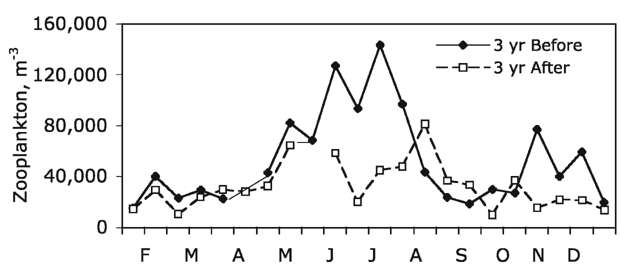

Fig. 12. Zooplankton prior and subsequent to outfall relocation in Massachusetts Bay station N04.

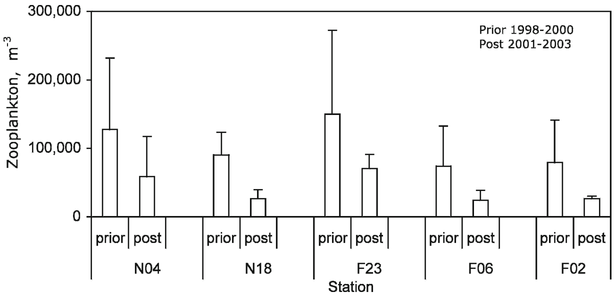

Fig. 13. June zooplankton abundance in Massachusetts and Cape Cod Bays prior and post outfall relocation in fall 2000.

2006). The summer decline in zooplankton coincided with the outfall relocation and occurred at all stations in the region. Only at the outfall station N18 did the zooplankton decline occur over the entire seasonal cycle and not just the summer period as at all the other sites (Fig. 11). Some of the decline may be attributable to outfall effluents or changes in circulation due to the outfall. Alternative explanations for the region-wide decrease might be increased ctenophores and/or the recent low summer primary productivity levels (Figs. 6, 9, and 10). Unfortunately, no microzooplankton data were collected on abundance or grazing and this community might have been more strongly correlated with reduced productivity, particularly since microflagellates (3–8 μm) dominated the phytoplankton community (Fig. 3). With the strong evidence for increased ctenophores, the summer decline in zooplankton might be attributed to increased predation (Link and Ford 2006). The broad spatial coverage of the monitoring program has suggested an interpretation of a regional factor rather than outfall effluent effect that would not have been possible if the monitoring program had been limited to the outfall near field.

Primary Production in Massachusetts Bay

The long monitoring record in Massachusetts Bay has placed potential changes due to outfall relocation in perspective with natural variability. During the period prior to outfall relocation, annual rates of primary production varied by a factor of about three in the near field and a factor of 5 at the Harbor mouth (Table 1). The near field rates of production ranged from 200 to 600 g C m^{-2} yr^{-1} and averaged 440 g C m^{-2} yr^{-1}, These rates of production on average generally slightly exceed the values of 280 to 470 g C m^{-2} yr^{-1} reported from an extensive data set by O'Reilly and Busch (1984) for the Mid-Atlantic Bight to the Gulf of Maine probably due to the more coastal location and access to land nutrients. The lowest rates of primary production occurred in 1998 prior to relocation at all three sampling stations. Without this low year of annual production, low rates of productivity in 2003 and 2004 might have been viewed as outfall related (Table 1). Annual rates of productivity since relocation have statistically not exceeded nor been less than the levels prior to relocation.

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