The Boston Harbor Project and the Reversal of Eutrophication of Boston Harbor

Massachusetts Water Resources Authority
Environmental Quality Department
Report 2013-07

Citation:

Taylor DI. 2013. **The Boston Harbor Project and the Reversal of Eutrophication of Boston Harbor.** Boston: Massachusetts Water Resources Authority. Report 2013-07. 33p.

THE BOSTON HARBOR PROJECT AND THE REVERSAL OF EUTROPHICATION OF BOSTON HARBOR

Prepared by

David I Taylor

MASSACHUSETTS WATER RESOURCES AUTHORITY
Environmental Quality Department and
Department of Laboratory Services
100 First Avenue
Charlestown Navy Yard
Boston, MA 02129
(617) 242-6000

June 2013

Report No: 2013-01

ACKNOWLEDGEMENTS

This report draws on data collected by a number of monitoring projects. Grateful thanks are extended to the following Principal Investigators (PI) of these projects:

Nancy Maciolek and James A. Blake
AECOM Environment, Marine & Coastal Center, 89 Water Street, Woods Hole, MA 02543, USA

Anne E. Giblin and Jane Tucker
The Ecosystems Center, Marine Biological Laboratory, Woods Hole, MA 02543, USA

Robert J. Diaz
Virginia Institute of Marine Science, College of William and Mary, Gloucester Pt., VA 23061, USA

Charles T. Costello
Division of Watershed Management, Massachusetts Department of Environmental Protection, 1 Winter Street, Boston MA 02108, USA

Kelly Coughlin, Wendy Leo, Ken Keay, Laura Ducott
ENQUAD, Massachusetts Water Resources Authority, 100 First Ave, Charlestown Navy Yard, MA 02129

TABLE OF CONTENTS

EXECUTIVE SUMMARY

During the past 25 years the Massachusetts Water Resources Authority (MWRA) has conducted a series of monitoring projects to track changes to environmental conditions in Boston Harbor. The projects were conducted in support of the Boston Harbor Project (BHP), and one of the aspects addressed by the projects was the eutrophication of the harbor ecosystem. Eutrophication is the anthropogenic organic enrichment of an ecosystem, and is caused by elevated nutrient and organic matter inputs. Changes attributed to eutrophication of other coastal bays and estuaries have included the development of dense nuisance algae blooms, decreases in water transparency, development of anoxia or hypoxia, fish kills, declines in benthic invertebrate diversity, and loss of important seagrass habitats.

This report tracks changes to eutrophication-related conditions in Boston Harbor during a ~20 year period that spanned the BHP. It addresses standing stocks of N and P in the harbor water-column, phytoplankton biomass and production, suspended particulate material and water transparency, and bottom-water dissolved oxygen concentrations. It also addresses rates of sediment oxygen uptake and the nutrient fluxes from the sediments. Additional aspects addressed by the report include the diversity of the benthic invertebrate communities, and the areal coverage by seagrasses. Boston Harbor has likely received anthropogenic nutrient and organic matter inputs since the 1600's and 1700's when the harbor watersheds were first colonized by Europeans.

During the 20 years spanned by this study, the N, P and organic C inputs to the harbor declined by between 65% and 90%; the decreases in the wastewater inputs brought about by the BHP contributed the bulk of the decreases. The patterns of changes in the harbor were in broad agreement with the Nixon-Vollenweider model of the eutrophication of phytoplankton-based coastal systems. The nature of the changes indicates the historic eutrophication of the harbor has been reversed. The correlations between the changes in the harbor and inputs, and the match between the observed and modeled harbor changes suggests the input decreases brought about by the BHP were responsible for the eutrophication reversal.

1.0 INTRODUCTION

Between 1991 and 2001, the Massachusetts Water Resources Authority (MWRA) undertook a large engineering project, the Boston Harbor Project (BHP), to better collect, treat and dispose of the wastewater discharged from the City of Boston and surrounding communities to Boston Harbor. Between 1990 and 2009 a series of monitoring programs were implemented by the MWRA to track the environmental effects of the project. Much of the monitoring focused on the changes to Massachusetts Bay, the coastal system to which the better-treated wastewater discharges were diverted in 2000. Programs of monitoring were however also conducted of Boston Harbor, the system that originally received the discharges. The degraded condition of the harbor before the BHP was one of the primary reasons the BHP was implemented.

Boston Harbor is one of the coastal systems associated with the large urban areas along the eastern USA that has been heavily impacted by human activities. One of the impacts of these activities has been the eutrophication, or anthropogenic enrichment (Nixon 1995, 2009) of the systems (Cloern 2001, Nixon 2009). Some of the changes that have been attributed to the anthropogenic nutrient and organic matter inputs responsible for eutrophication have included the development of dense and often nuisance phytoplankton and macroalgae (or seaweed) blooms (Valiela et al. 1992, Anderson et al. 2002), anoxia and hypoxia (Diaz and Rosenberg 2008), loss of seagrass habitats (Short and Wylie-Echeverria 1996), fish kills (Breitburg et al. 2009), and changes to the benthic fauna of the systems (Caddy 1993, Dauer et al. 2000).

Before the start of the BHP the human population in the harbor watersheds exceeded 2.0 million, and the total inputs of N and P to the harbor were among the highest reported for bays or estuaries in the USA (Kelly 1997, Boynton et al. 2008). The two wastewater treatment facilities (WWTF's) that discharged wastewater from the City of Boston and surrounding communities to the harbor contributed >90% of the elevated inputs (Alber and Chan 1994, Taylor 2010). Prior to a series of studies conducted in support of the BHP in the late 1980's and early 1990's (Robinson et al. 1990, Adams et al. 1992, Kelly 1997), little was known of the eutrophication of the harbor ecosystem.

The purpose of this report is to document the changes to the eutrophication of Boston Harbor over a 20-year period (2000-2009) that spanned the BHP. The changes in the nutrient and organic matter inputs brought about by the BHP have been described in detail by Taylor (2010). Some of the changes that occurred to the harbor ecosystem during the BHP have been reported by Giblin et al. (1997), Oviatt et al. (2007), Diaz et al. (2008), Tucker et al. (2010), Maciolek et al. (2010), Costello and Kenworthy (2011), Taylor et al. (2011) and Connor et al. (2012). In this report we correlate the harbor changes to the changes in inputs, to quantify the extent to which the BHP was responsible for the harbor changes.

2.0 THE BHP AND THE DECREASES IN INPUTS TO BOSTON HARBOR

2.1 Background on the BHP

Details of the BHP and of the changes in loadings of nutrients, solids and particulate organic matter brought about by the BHP, have been provided in Taylor (2010), so only an overview is provided here. Details of the engineering involved in the BHP have been provided by Breen et al. (1994), Grace (2009) and Roberts et al. (2010). Prior to the BHP the harbor received primary-treated effluent, and after anaerobic digestion the sludge generated by the primary-treatment process, from two wastewater-treatment facilities (WWTF) (the locations of the facilities and of the discharges from the two facilities are provided in Figure 1). The Deer Island (DI) WWTF, the larger of the facilities,

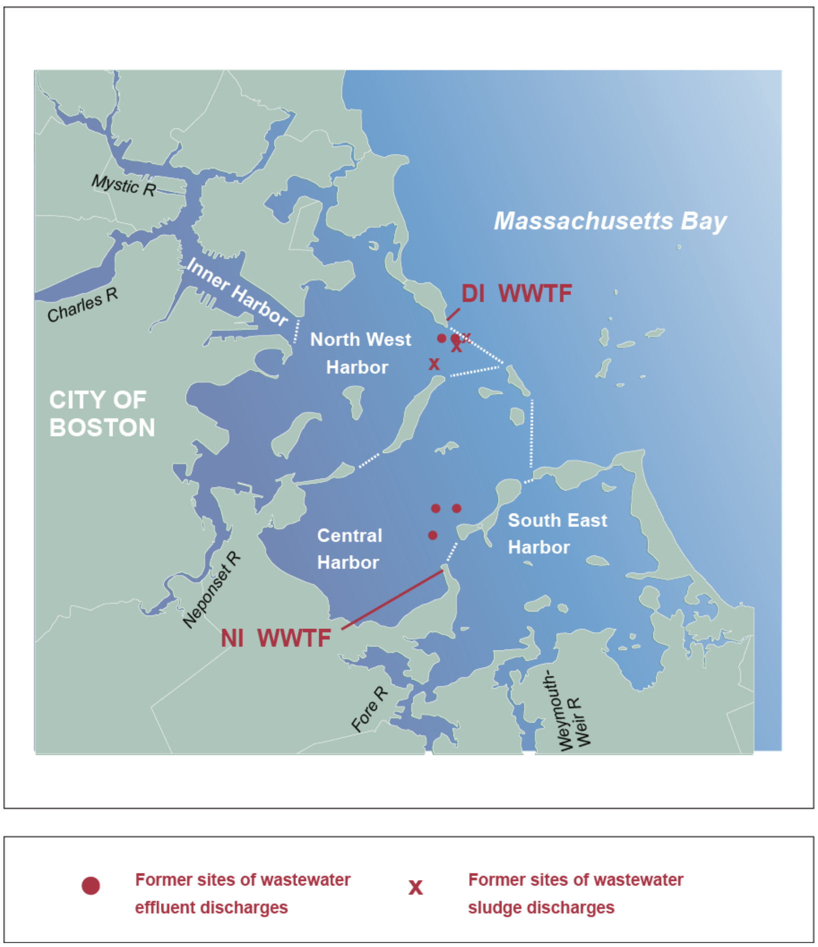


Fig. 1. Locations of the Deer Island (DI) and Nut Island (NI) wastewater treatment facilities (WWTF's), the locations at which the effluent and sludge from the two facilities were discharged before the BHP, and the four regions of Boston Harbor.

discharged its effluent to the mouth of the NWH; the Nut Island (NI) WWTF discharged its effluent to the mid-Central Harbor region (mid-CH). The sludge from both facilities was discharged on the outgoing tide, to the mouth of the North West Harbor (NWH) region.

The BHP involved five construction milestones. Figure 2 provides a schematic of the milestones and of the changes to the locations of the WWTF effluent and sludge discharges during the BHP. Based on the changes in locations of the wastewater effluent

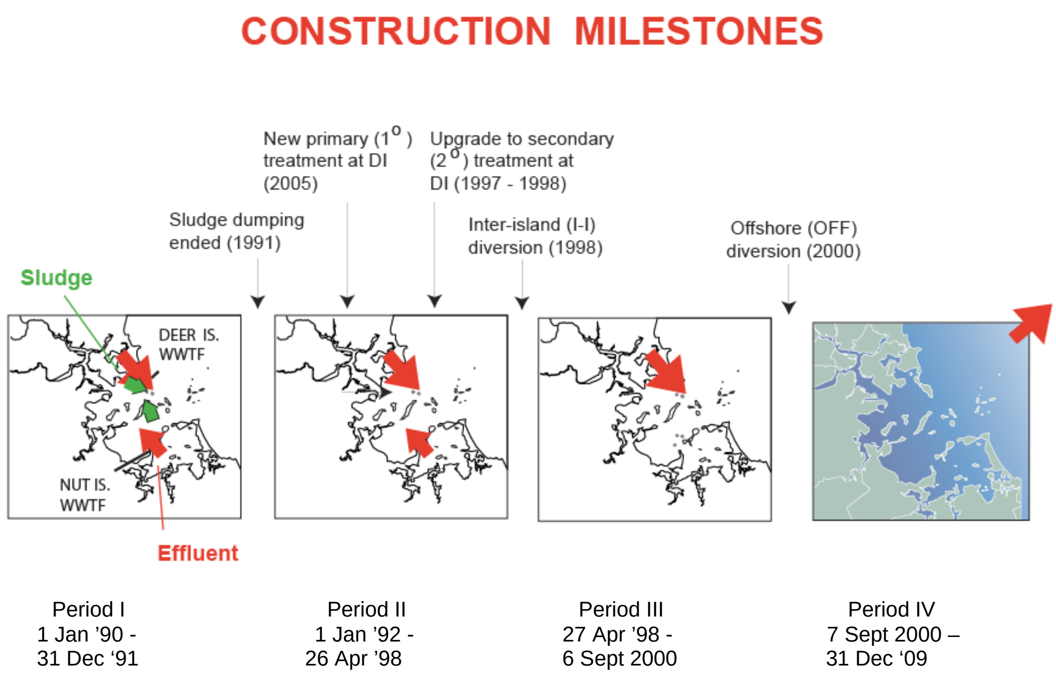


Fig. 2. Schematic of the changes in the locations of the wastewater effluent and sludge discharges to the harbor through the BHP. Also shown are the timing of the five construction milestones of the BHP, and the durations of each of the four loading periods..

and sludge discharges, the period encompassed by the BHP could be partitioned into four 'Loading Periods' (Periods I – IV). The sludge discharges to the harbor were ended in December 1991, after a sludge pelletization/fertilizer plant was completed (Milestone 1). In July 1995 a new, more efficient primary (1^o) treatment facility was constructed at the DI facility (Milestone 2). From 1997 through 2001, treatment at the DI WWTF was upgraded from primary to secondary- (2^o) treatment (Milestone 3). In April 1998, the effluent formerly treated at the NI facility was diverted through the upgraded DI facility for discharge to the mouth of the NWH (Milestone 4). In September 2000, the now combined and secondary-treated wastewater from the DI facility was diverted 15-km offshore for discharge through a multi-port diffuser system located at 30-m depth on the

Massachusetts Bay seafloor (Milestone 5). The ocean outfall-diffuser system has been described in detail by Grace (2009) and Roberts et al. (2010).

2.2 Changes to the nutrient and organic matter inputs to the harbor

During the 20 years spanned by the study, the external (wastewater + river + non-point source) inputs of TN and TP to the harbor were decreased from 83 g N m^{-2} y^{-1} to 16 g N m^{-2} y^{-1} (or by 81%) and from 17 g P m^{-2} y^{-1} to 1 g P m^{-2} y^{-1} (or by 94%), respectively (Fig. 3, Taylor 2010). Decreases in wastewater inputs accounted for ~80% to~95% of the TN and TP input decreases. The combined (external + internal) inputs of organic C were decreased from 1,047 g C m^{-2} y^{-1} to 384 g C m^{-2} y^{-1}, or by 63%. Decreases in internal inputs (pelagic primary production, measured by Oviatt et al. 2007) accounted for ~354-g C m^{-2} y^{-1} (or 53%) of the decrease; decreases in wastewater inputs contributed the remaining 309-g C m^{-2} y^{-1} or 47%. The molar TN:TP input ratios to the harbor were increased from ~11:1 to ~38:1. The molar C:N input ratios (combined organic C: external TN) were increased from ~15:1 to 28:1. The bulk of the TN, TP and combined organic C input decreases occurred during three to four years (1997/1998 to 2001) that spanned the upgrade to secondary treatment at the DI WWTF and the two diversions of the wastewater discharges, first to the harbor mouth and then offshore. The increases in molar TN:TP and C: N input ratios were also focused during the same 3-4 years.

The contributions made by the different milestones of the BHP to the decreases in the external inputs of TSS, POC, TN and TP are shown in Figure 4 (this Figure is from Taylor 2010). For TSS and POC, the first milestone, the ending of the sludge discharges in 1991, contributed ~40% of the total decrease in loadings. The second milestone, the completion of the new primary-treatment facility at the DI WWTF in 1995, contributed 1.6% and 0.08% of the total TSS- and PC-loading decreases, respectively. Milestone 3, the upgrade to secondary-treatment at the DI facility (1997-2001) contributed 5% of the decrease in TSS-loadings, and 19% of the decrease in PC-loadings. The inter-island diversion in 1998 (Milestone 4) contributed 15% and 23% of the decreases in the TSS- and PC-loadings, respectively. The offshore diversion in September 2000, contributed 31% and 19% of the TSS- and PC-loading decreases. For TN, milestones 1 through 5 contributed 13%, 0.5%, 8%, 23% and 63% of the total decreases. For TP, the respective contributions were 13%, 0.5%, 19%, 28% and 40%.

3.0 CHANGES TO THE EUTROPHICATION OF BOSTON HARBOR

3.1. What impact did the BHP have on eutrophication-related conditions in the harbor?

Changes to the harbor water-column. Details of the changes to the harbor water-column have been provided by Oviatt et al. (2007), Libby et al. (2011), and Taylor et al. (2011). Figure 5 shows the locations at which changes were monitored in the harbor. Between

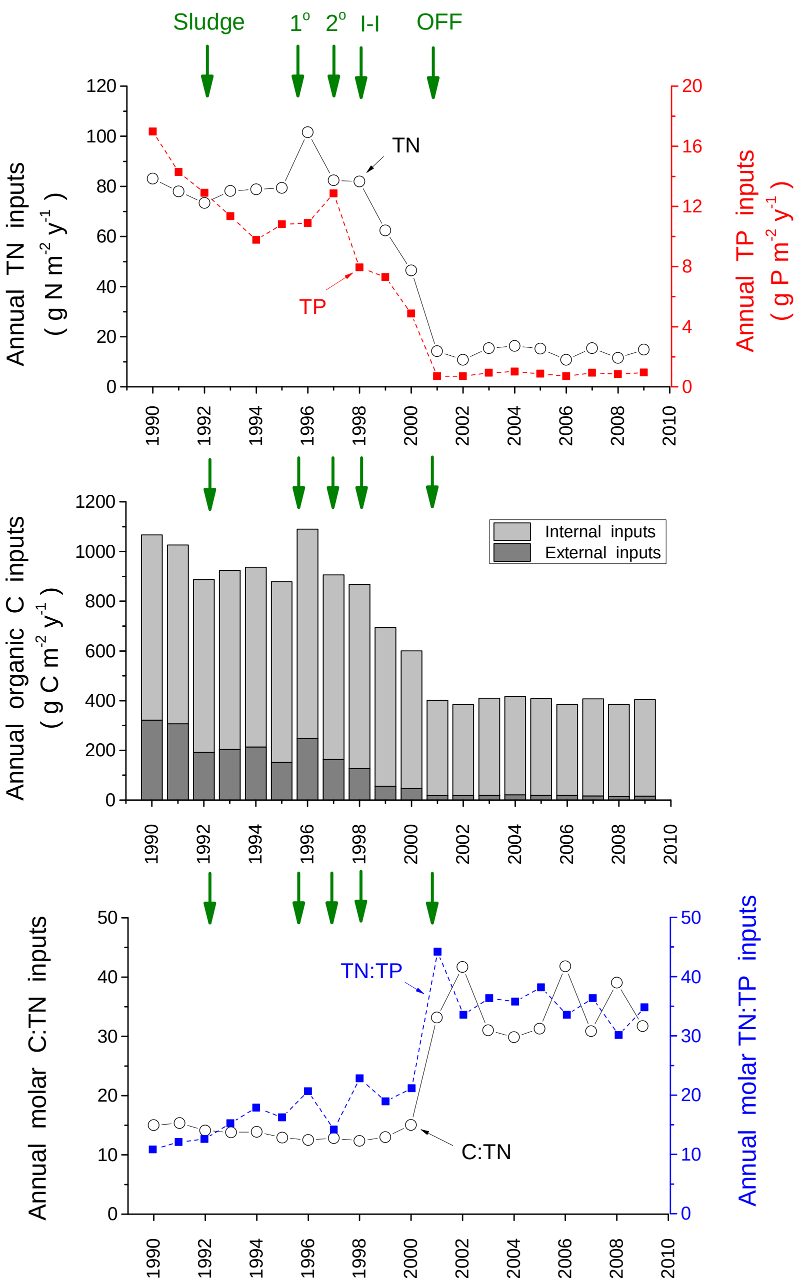


Fig. 3. Annual average loadings of TN, TP, organic C, and molar TN:TP and POC:TN input ratios, 1990 through 2010. Vertical arrows indicate dates the five milestones of the BHP were completed. Sludge = sludge dumping ended; 1° = new primary treatment facility at DI WWTF; 2° = start of phase-in of secondary treatment at DI; I-I = inter-island diversion; OFF = diversion of the DI WWTF flows offshore.

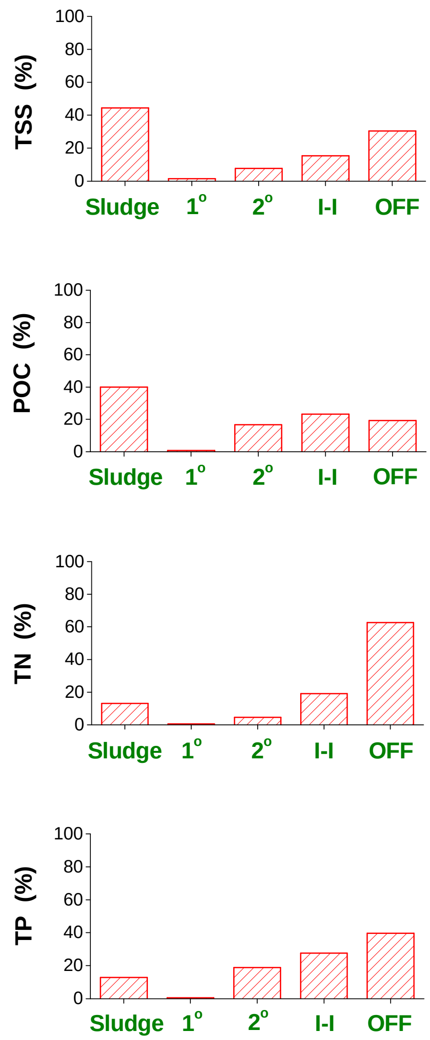


Fig. 4. Percent contributions of the five construction milestones of the BHP to the total decreases in the WWTF loadings of TSS, POC, TN and TP to Boston Harbor (from Taylor 2010).

1995 (the first year for which harbor-wide water-column data were available) and 2009, the average water-column TN- and TP- concentrations declined from ~33-µmol l^{-1} to ~22.5-µmol l^{-1} (or ~32%), and from ~2.0-µmol l^{-1} to ~1.5-µmol l^{-1} (or ~25%), respectively (Fig. 6). The decreases in the average TN- and TP- concentrations were observed at all 10 stations (Fig. 7). Decreases in the dissolved inorganic N and P fractions accounted for about two thirds of the respective decreases. Concentrations of particulate N and P also decreased, but the concentrations of the dissolved organic fractions were unchanged (Taylor et al. 2011).

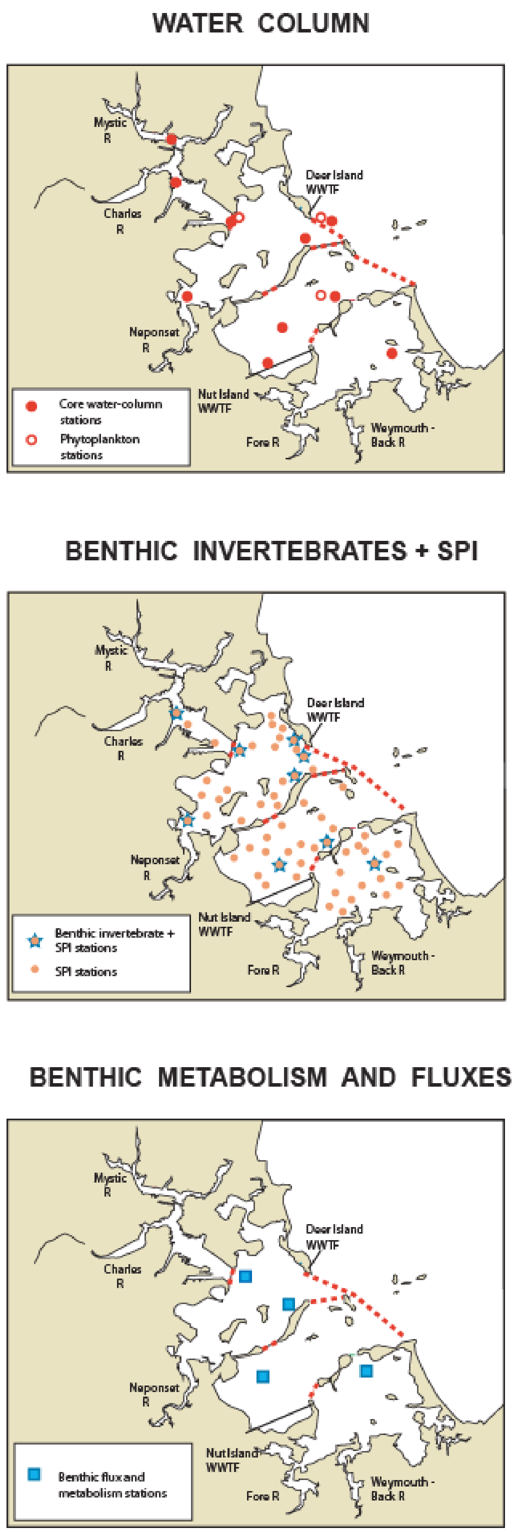


Fig. 5. Monitoring stations used to track changes to the water-column (top), benthic infauna (middle), and the benthic metabolism (bottom) of the harbor. SPI = sediment profile imaging.

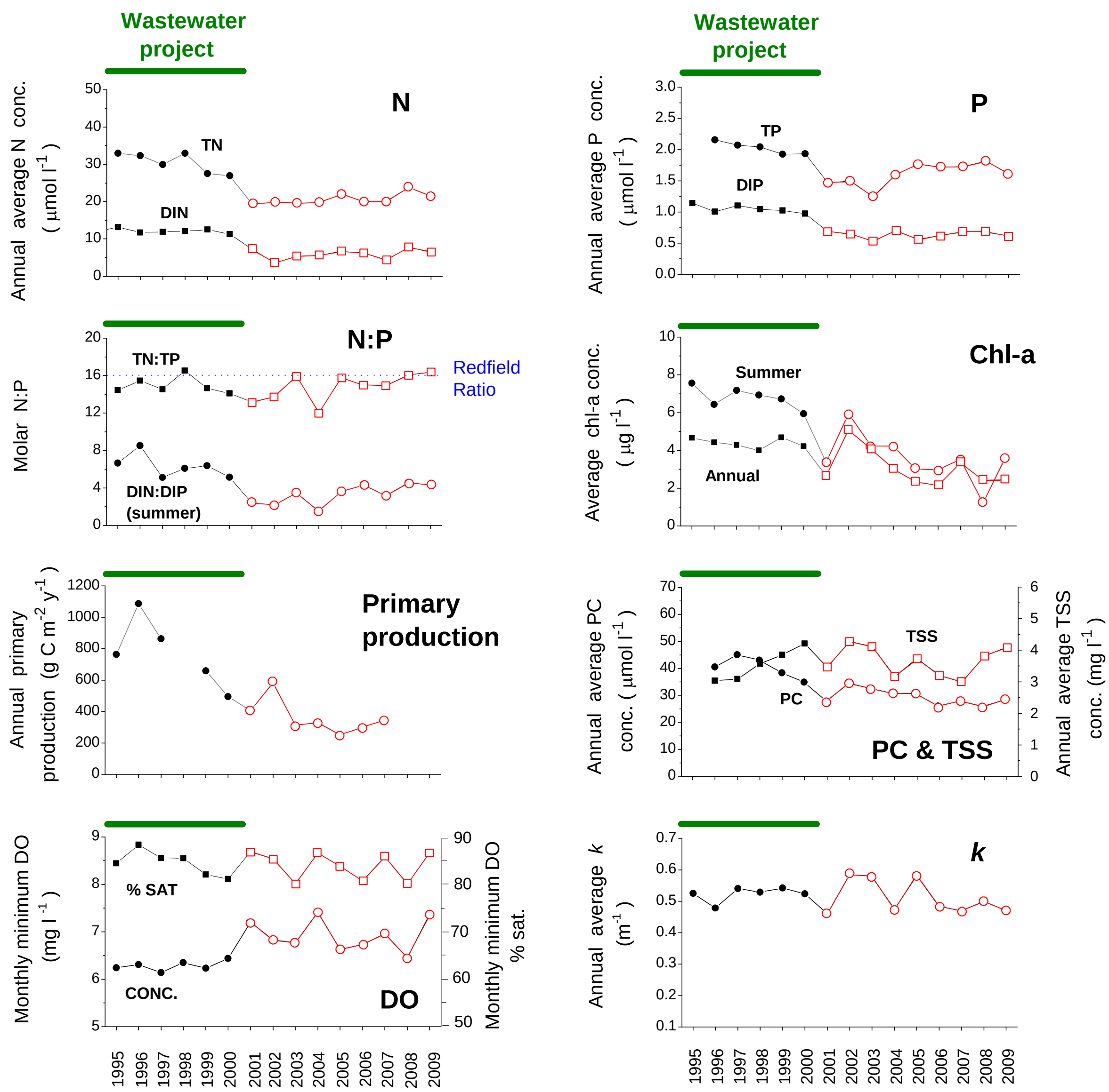


Fig. 6. Changes to the harbor water-column,1995-2009. The horizontal line at the top of each panel shows the period encompassed by the BHP (plot adapted from Taylor et al. submitted).

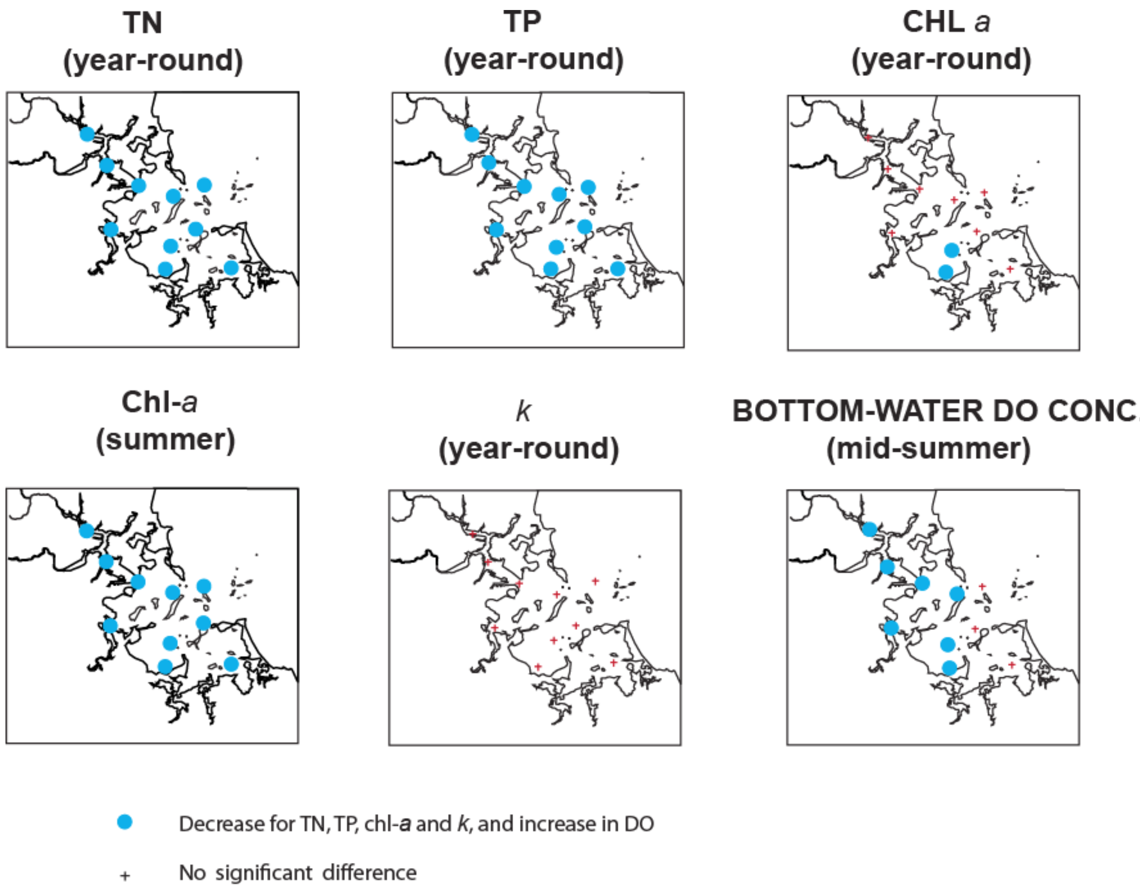


Fig. 7. Stations at which average conditions in the water-column after the wastewater discharges were diverted offshore (Loading Period IV) were significantly different (p <0.05) from conditions during Loading Period II (the period when the harbor received discharges of effluent alone (no sludge) from both WWTF's). Figure adapted from Taylor et al. (2011).

Annual molar TN:TP concentrationratios showed no trend, and during all years, remained between ~12:1 and 16:1, and similar to the molar Redfield ratio of 16:1. The mid-summer DIN:DIP ratios (J, J, A, S) decreased from between ~5:1 and 8:1 before the discharges to the harbor were discontinued, to between 1.5:1 and 4.5: 1 after. Both before, but especially after the discharges were ended, the ratios during mid-summers were well below the Redfield Ratio. N:P (and especially DIN:DIP) ratios less than 16:1, and especially below~10:1 are considered indicative of potential N limitation. Average DIN:DIP ratios during winters (D, J, F, M) were >10:1 during all except one of the years before or after discharges were ended (and specifically during 2001) (MWRA unpublished data), indicating excess DIN:DIP relative to demand.

Annual average phytoplankton biomass, measured as annual average chl-a-concentrations, declined from ~4.5-µg l^{-1} to ~3.2-µg l^{-1} (or by 29%); the average concentrations during summers (J, J, A, S) declined from ~7.5-µg l^{-1} to ~4.0-µg l^{-1} (or by 47%). The decreases in the annual average concentrations were significant at 2 of the 10 stations; the decreases in the concentrations during summers were significant at all 10 stations. Diatoms and micro-flagellates were the two most abundant phytoplankton taxa in the harbor both before and after the wastewater discharges to the harbor were discontinued (Taylor et al. 2011). Both the average micro-flagellate and diatom counts declined during the BHP, and by 0.20 $x10^6$ cell l^{-1} (or 21%) and 0.24 $x10^6$ cell l^{-1} (or 31%), respectively. Average ^{14}C pelagic primary production, measured at the harbor mouth, decreased from 504 g C m^{-2} y^{-1} in 1992-1994 (Oviatt et al. 2007) to 292 g C m^{-2} y^{-1} in 2006-2008 (Libby et al. 2009).

The transparency of the harbor water-column was relatively high, and showed no trend. Secchi depths averaged 3.2 m, and vertical PAR attenuation coefficients, 0.5 m^{-1}. Average TSS-concentrations in the harbor also showed no trend. Average particulate organic carbon (POC) concentrations, however, declined from 43.5-µmol l^{-1} to 29.6-µmol l^{-1}, or by 13.9-µmol l^{-1} (or 32%). Concentrations at all 10 stations declined (figure not shown). Average TSS:POC concentrations (by weight) increased, suggesting the amount of suspended particulate material remained the same, but the organic content of the material declined. Summer average DO concentrations tended to be high, and averaged between 6.9 mg l^{-1} and 7.9 mg l^{-1} during the study. They showed no trend. The minimum bottom-water DO- concentrations observed in the harbor each summer, however, increased, from 6.0 - 6.5 mg l^{-1}, to 6.5 - 7.0 mg l^{-1}. The increase was small, but was observed during eight of the nine summers after the diversion. It was also seen at 7 of the ten stations.

Changes to benthic invertebrate communities. Diaz et al. (2008) and Maciolek et al. (2010) have described the changes to the harbor benthic infauna in detail. The diversity of the benthic infauna, measured as Shannon-Weiner H', increased from 2.1 to 3.0; and as log-series *alpha*, from 4 to 9 (Fig. 8). The total numbers of benthic invertebrate taxa increased from 20-30 taxa $sample^{-1}$ in 1991-1992, to 40-50 taxa $sample^{-1}$ in 2007-2008. Details of the actual taxa that showed changes in the harbor have been provided by Maciolek et al. (2010) The percent of the 61 stations that showed *Ampelisca* amphipod tubes on the sediment surface increased from ~40% in the early 1990's, to a peak of 60% in 1995, and then declined to zero in 2005. The densities of the invertebrates (#individuals per sample) showed a general decrease over the study, but as for the total # taxa, the variability year-to-year was large.

Changes to benthic metabolism and nutrient fluxes- Details of the changes to the benthic metabolism and net fluxes of nutrients from the harbor sediments have been provided by Giblin et al. (1997) and Tucker et al. (2010). The average rates of sediment oxygen uptake (SOD) declined from an average of ~160-mmol O_2 m^{-2} d^{-1} in 1993-1995, to ~50-mmol O_2 m^{-2} d^{-1} in 2007-2009 (Fig. 9). The average net fluxes of DIN from the sediments to the water-column decreased from ~7.5-mmol N m^{-2} d^{-1} to ~4.1-mmol N m^{-2} d^{-1}. The average rates of sediment denitrification decreased from 3.2-mmol N m^{-2} d^{-1} to

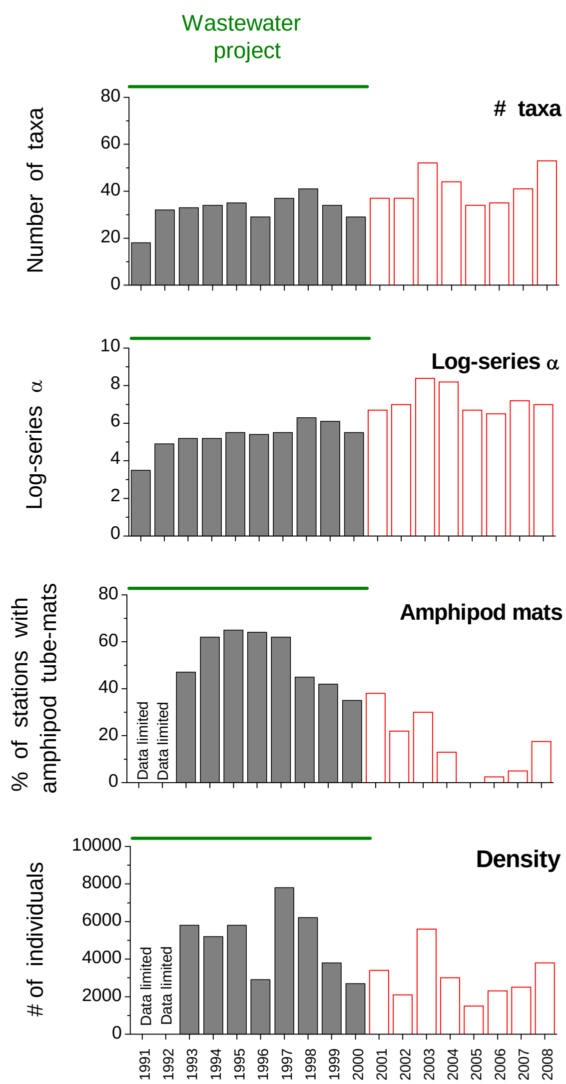


Fig. 8. Changes to the harbor benthic invertebrate communities,1991-2009. The sold bars show the years during which one or both WWTF's discharged to the harbor.

2.0-mmol N m^{-2} d^{-1} (Tucker et al. 2010), but a change in methods likely accounted for at least part of the decrease.

***Zostera* seagrass beds** – The seagrass beds in the harbor were not tracked as intensively as the water-column or sediments, but based on surveys conducted by Costello and

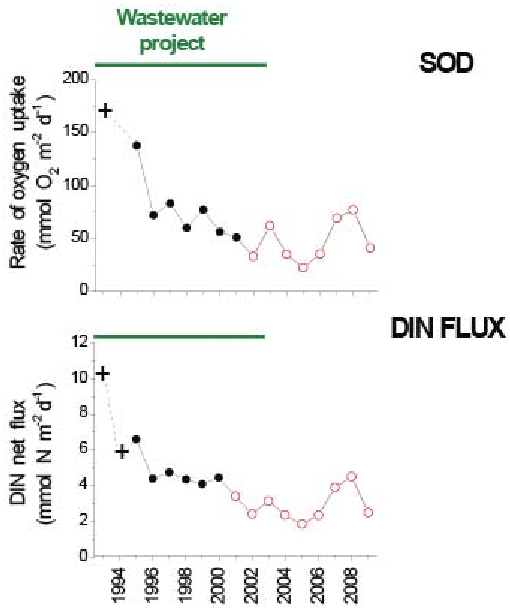


Fig. 9. 'Summer' rates of sediment oxygen uptake (top) and seasonal (May through October) net fluxes of DIN from the harbor sediments (bottom), 1993-2009. (Figure from Taylor et al. submitted).

Kenworthy (2011), the total area covered by the seagrass beds in the harbor also changed during the BHP (Fig. 10). Between 1994-1996 and 2000-2002, the first two periods *Zostera* was monitored, the beds declined from 82 ha to 27 ha, or from ~0.08% to 0.02% of the harbor area. The area of the beds then increased by a factor of 1.7, to 47 ha, (or 0.04% of the harbor area) between 2000-2002 and 2006-2007, a period that encompassed the first 6-7 years after the wastewater discharges to the harbor were discontinued. In a separate study *Zostera* seagrass shoots were transplanted in four ~10 - 20 m^2 plots in the harbor in 2005 and 2006 (Leschen et al. 2010). By 2009, the area covered by the shoots in the plots had increased 2.0- to 4.6- fold; shoot density in the plots (shoots per unit area) had increased between 1.05- and 4.2- fold. Existing beds contributed the bulk of the expansion we observed after the harbor discharges to the harbor were discontinued.

3.2 Relationships between harbor changes and changes in loadings

For most of the variables that showed changes in the harbor, the changes were significantly correlated either with the decreases in the nutrient inputs or the decreases in

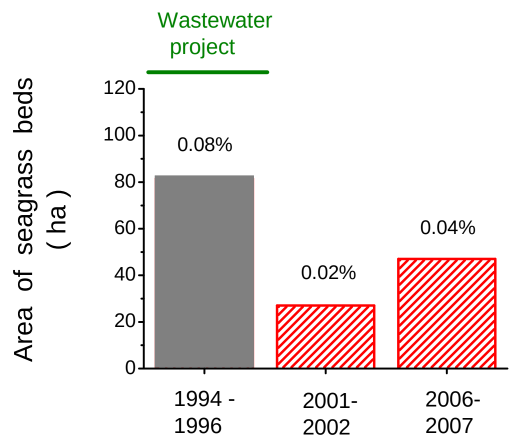


Fig. 10. Areas covered by seagrass beds during three periods. The % values above the bars are the percentages of the total harbor area covered by seagrasses (these data are derived from Costello and Kenworthy 2011).

the organic C inputs (Fig. 11, Table 1). The decreases in the annual average TN-concentrations ($r^2 = 0.92$), the average summer chl-a-concentrations ($r^2 = 0.61$) and the annual rates of pelagic primary production ($r^2 = 0.77$), all occurred in linear proportion to the annual external inputs of N. The annual average PC-concentrations decreased ($r^2 = 0.81$), and the minimum bottom-water DO-concentrations observed each summer increased ($r^2 = 0.45$) in linear proportion to the decreases in the combined external + internal inputs of C. No correlations existed between annual average *k* values, and the loadings of N or organic C. Average TN:TP concentrations showed no correlation with annual TN:TP inputs (Taylor et al. 2011).

The decrease in the percent incidence of the amphipod mats ($r^2 = 0.77$) and the increase in the diversity of the benthic faunal community measured as log-series α ($r^2 = 0.66$) both occurred in linear proportion to the decrease in external + internal inputs of C (Fig. 12). The same applied for the increase in diversity measured as Shannon-Weiner H' (Taylor et al. submitted). Both the rates of sediment oxygen uptake and the net fluxes of DIN from the sediments declined in linear proportion to the internal + external inputs of C; the r^2 values were relatively low (Taylor et al. submitted). Thus, both the water-column and sediments of the harbor were changed, and for both of these components of the harbor ecosystem the changes were rapid and occurred in linear proportion to the inputs.

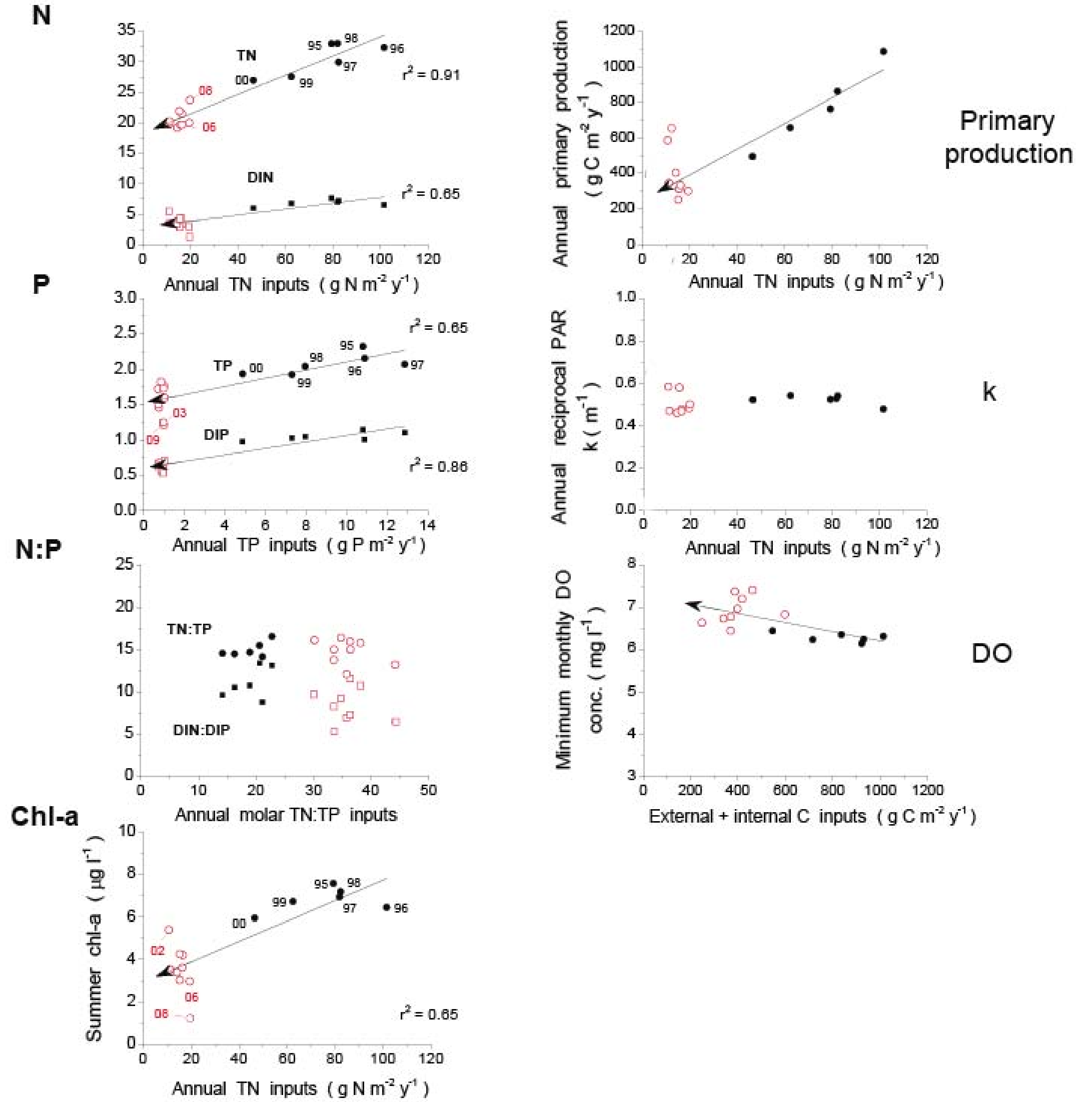


Fig. 11. Correlations between various water-column variables, and loadings of N or PC to the system (Figure adapted from Taylor et al. submitted). Solid circles show the years the harbor received WWTF discharges; open circles show the years after the discharges were diverted offshore.

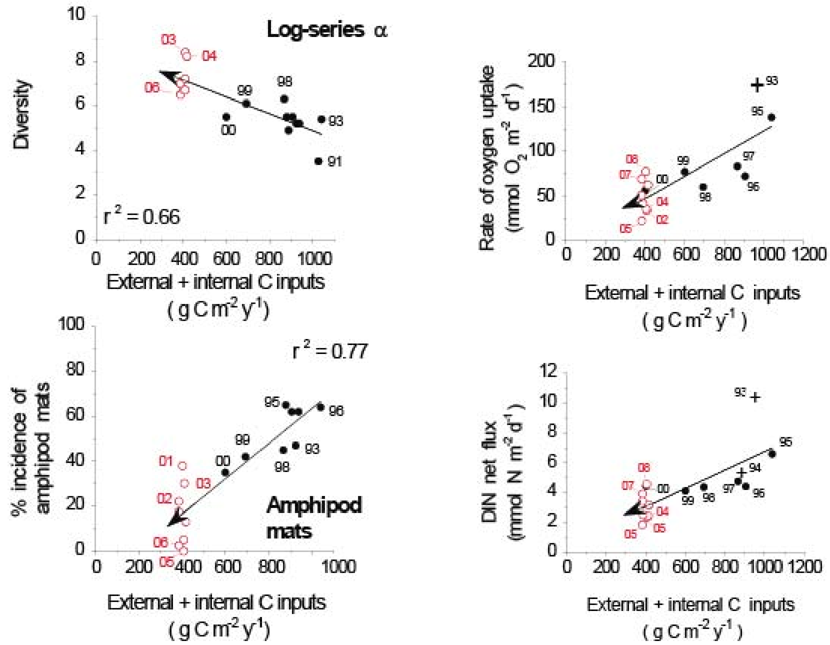


Fig. 12. Correlations between various sediment variables, and loadings of PC to the system (Figure adapted from Taylor et al. submitted). Solid circles show the years the harbor received WWTF discharges; open circles show the years after the discharges were diverted offshore. '+' show years when SOD was only measured at the two north harbor stations (Figure adapted from Taylor et al. submitted).

3.3 Changes to the biogeochemical C and N budgets

Significant changes were also observed to the C and N budgets of the harbor (Fig. 13 and 14). At the end of the study (represented by the period 2006-2009), the total (external + internal) inputs of C to the water-column were 145.6 x 10^3 kg C d^{-1} or 60% smaller than at the start (1990-1994). The total (external + internal) N- inputs were, in turn, 23.3 x 10^3 contributed 58% of the decrease in inputs; the decrease in pelagic primary production contributed 40%. The decrease in wastewater inputs contributed 80% of the decrease in N-inputs; the decreases in sediment DIN-fluxes contributed an additional 16% of the decrease in inputs to the water-column.

Both before and after the wastewater discharges were diverted offshore, most of the C- and N- inputs were advected offshore. During the period the harbor received its elevated loadings, only 24% of the C inputs and 11% of the N inputs were 'retained' by the harbor; 'retained', as used here, refers to burial + mineralization within the harbor. Our estimate of 11% of the N inputs 'retained' was similar to the estimates of 12% and 14% computed, using data from the early 1990's, by Adams et al. (1992) and Nixon et al. (1996), respectively. After the loadings to the harbor were decreased, the proportions of the C- and N-inputs retained by the harbor were increased to 30% and 38%, respectively.

Table 1. Relationships between aspects of the harbor ecosystem and the inputs of N, P and organic carbon to the harbor, 1990 to 2009. Asterisks denote r^2 values $\geq$ 0.65.

Dependent variable	Independent variable	Equation	r^2
Water column			
TN	TN load	$TN_{annual} = 18.3 + (0.16 \times TN_{load})$	0.91*
DIN	TN load	$DIN_{annual} = 2.93 + (0.047 \times TN_{load})$	0.65*
TP	TP load	$TN_{annual} = 1.52 + (0.05 \times TN_{load})$	0.65*
DIP	TP load	$DIP_{annual} = 0.61 + (0.045 \times TP_{load})$	0.86*
Chl-a (annual)	TN load	$Chl_{annual} = 2.87 + (0.02 \times TN_{load})$	0.30
	TP load	$Chl_{annual} = 3.01 + (0.13 \times TP_{load})$	0.35
Chl-a (summer)	TN load	$Chl_{summer} = 2.93 + (0.047 \times TN_{load})$	0.65*
	TP load	$Chl_{summer} = 3.33 + (0.356 \times TP_{load})$	0.71*
Primary production (annual)	TN load	$Prod_{annual} = 365 + (5.76 \times TN_{load})$	0.65*
	TP load	$Prod_{annual} = 340 + (33.6 \times TP_{load})$	0.51
PC (annual)	TN load	$PC_{annual} = 26.95 + (0.17 \times TN_{load})$	0.75*
	Total org. C load	$PC_{annual} = 20.2 + (0.24 \times C_{load})$	0.81*
DO conc. (min.)	TN load	$DO_{min} = 7.05 + (-0.01 \times TN_{load})$	0.55
	Total org. C load	$DO_{min} = 7.3 + (-0.001 \times C_{load})$	0.45
Benthic fauna			
Total # taxa	Total org. C load	$\# taxa = 48.7 + (-0.01 \times C_{load})$	0.32
	TN load	$\# taxa = 42.78 + (-0.13 \times TN_{load})$	0.26
Diversity index	Total org. C load	$\text{Log ser } \alpha = 8.7 + (-0.01 \times C_{load})$	0.66*
	TN load	$\text{Log ser } \alpha = 7.49 + (-0.027 \times TN_{load})$	0.57
Amphipod mats	Total org. C load	$Mats = -14.9 + (0.079 \times C_{load})$	0.77*
	TN load	$Mats = 8.12 + (0.58 \times TN_{load})$	0.76*
Benthic metabolism/fluxes			
SOD	Total org. C load	$SOD = 3.84 + (0.126 \times C_{load})$	0.59
	TN load	$SOD = 38.2 + (0.715 \times TN_{load})$	0.35
DIN flux	Total org. C load	$DIN_{flux} = 0.55 + (0.006 \times C_{load})$	0.52
	TN load	$DIN_{flux} = 2.49 + (0.039 \times TN_{load})$	0.38

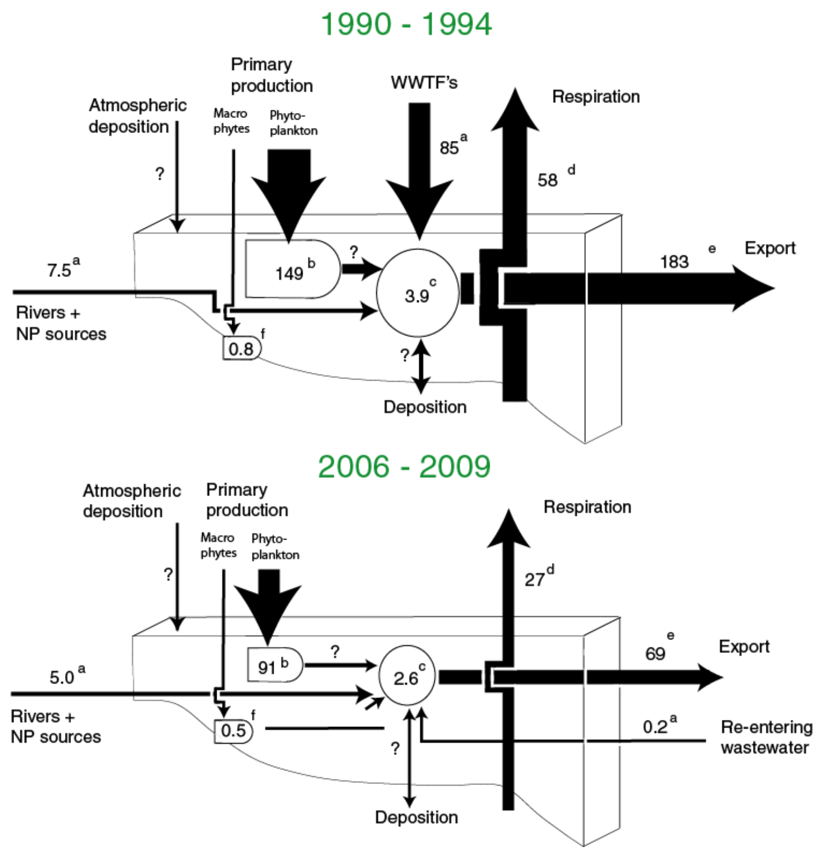


Fig. 13. Preliminary C budget of Boston Harbor, 1990 - 1994 (top) versus 2006 – 2009 (bottom). Arrows show net fluxes (x 10^3 kg d^{-1}); circles show standing stocks (ton). [a] inputs of PC from Taylor (2010), [b] ^{14}C primary production from Oviatt et al. (2007) and Libby et al. (2009), [c] PC standing stocks from Taylor et al. (2011), [d] DIC net flux for 1993-1994, estimated from rates of sediment oxygen uptake assuming RQ = 1.0 (Tucker et al. 2010), and assuming soft sediments accounted for 51% of harbor bottom (Knebel et al. 1991), [e] computed by difference.

4.0 DISCUSSION

4.1 How heavily loaded was Boston Harbor before the BHP?

Figure 15 compares the TN- and TP- loadings to Boston Harbor before and after the BHP, with the loadings to select other coastal aquatic ecosystems. Also shown in the Figure are the loadings to five other coastal aquatic ecosystems for which decreases in loadings have been reported; Hillsborough Bay and Tampa Bay (Greening and Janicki 2006), the Danish Straits (Carstensen et al. 2006), Kaneohe Bay (Smith et al. 1981) and Lajaalahti Bay (Kauppila et al. 2005). Before the BHP the total TN- (804 g m^{-2} y^{-1}) and

TP- (155 g m^{-2} y^{-1}) loadings to the harbor ranked third or fourth among the systems shown in the Figure.

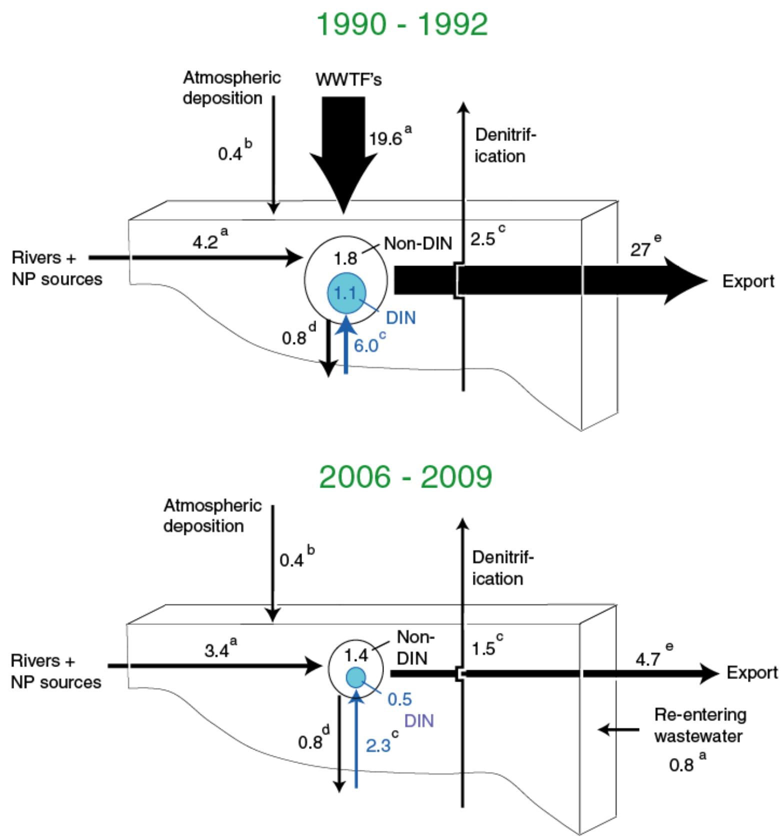


Fig. 14. Preliminary N budget of Boston Harbor, 1990 – 1994 (top) versus 2006 – 2009 (bottom). Arrows show net fluxes (x 10^3 kg d^{-1}), circles show standing stocks (ton). Black arrows and symbols refer to TN; blue arrows and symbols refer to DIN. [a] inputs from Taylor (2010), [b] estimate of 12.5 kg N ha^{-1} y^{-1} from Bowen and Valiela (2001), and assumed to be same during both periods, [c] from Tucker et al. (2010) for 1993-1994, calculated assuming depositional sediments accounted for 51% of harbor bottom, [d] from Kelly (1997), and assumed to be same during both periods, [e] computed by difference.

Among the six systems in the Figure that experienced decreased loadings, Boston Harbor showed the highest initial loadings of both TN and TP. Boston Harbor also showed the largest absolute decreases in loadings of both nutrients. Expressed as percent of initial loadings, the decreases in the inputs of both TN and TP to Boston Harbor ranked second only to Lajaalahti Bay. Before the BHP the TN and TP loadings were in the order of the

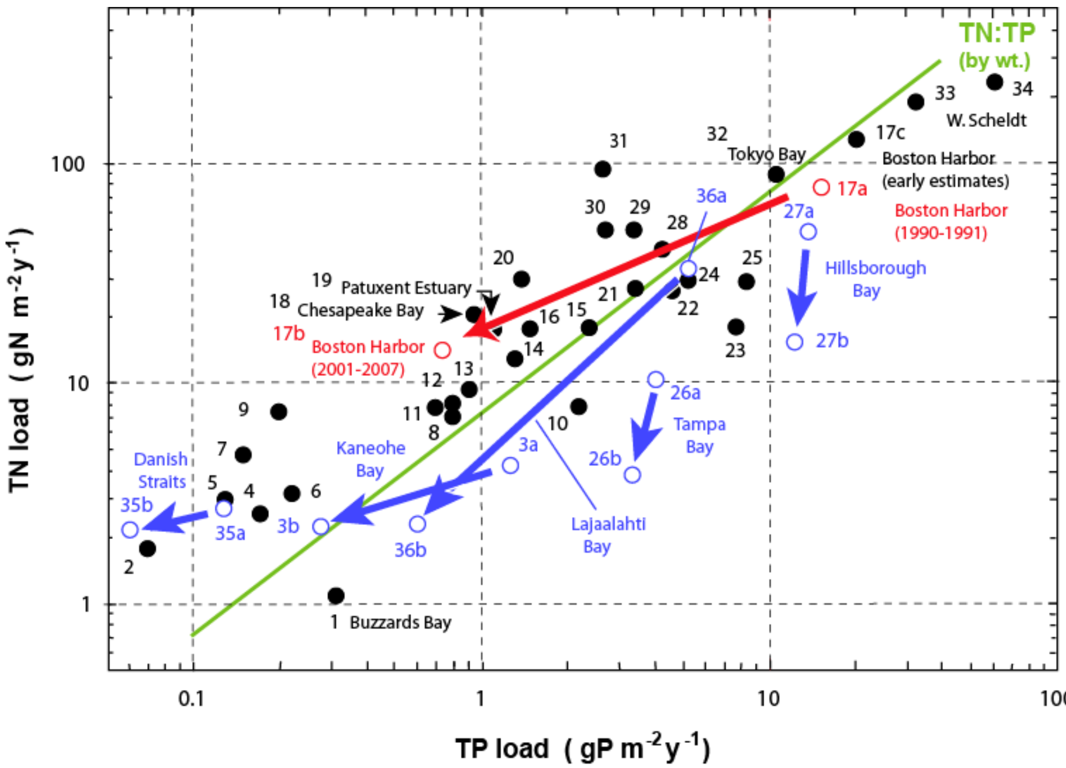


Fig. 15. Comparison of the TN- and TP-loadings to Boston Harbor before and after the BHP, with the loadings to 35 other coastal aquatic ecosystems (adapted from Boynton et al. 2008). Also shown in the Figure are the loadings to five other systems that have experienced decreases in nutrient loadings. 1 Buzzards Bay; 2 Sinepuxent Bay; 3a Kaneohe Bay, pre-diversion (Smith et al. 1981); 3b Kaneohe Bay, post-diversion (Smith et al.); 4 Isle of Wight Bay: 5 Baltic Sea; 6. Chincoteague Bay; 7 Gulf of Riga; 8 Albermarle Sound; 9 Himmerfjarden; 10 Guadalupe Bay, dry year; 11 Buttermilk Bay; 12 Moreton Bay; 13 Seto Inland Sea; 14 Taylorville Creek; 15 Newport Bay; 16 Adriatic Sea; 17a Boston Harbor, pre-diversion (Taylor 2010); 17b Boston Harbor, post-diversion (Taylor 2010); 17c Boston Harbor (early estimates, pre-diversion) (Alber and Chan (1994); 18 Chesapeake Bay; 19 Patuxent Estuary; 20 Potomac Estuary, 21 Narragansett Bay; 22 Mobile Bay; 23 Delaware Bay; 24 Guadelaupe Bay, wet year; 25 N San Francisco Bay; 26a Tampa Bay, before loading reductions (Zarbock et al. 1994); 26 b Tampa Bay, after (Poe et al. 2005); 27a Hillsborough Bay, before loading reductions (Zarbock et al. 1994); 27b Hillsborough Bay, after (Poe et al. 2005); 28 St Martens River; 29 Patapsco Estuary; 30 Apalachicola Bay; 31 Back River; 32 Tokyo Bay; 33 Westerschelde; 34 Charles River Basin (Breault et al. 2002); 35a Danish Straits, before loading reduction (Carstensen et al. 2006); 35b Danish Straits, after (Carstensen et al.); 36a Lajaalahti Bay, before loading reductions, Kauppila et al. (2005); 36b Lajaalahti Bay, after, Kauppila et al. The solid diagonal line represents the Redfield ratio of TN:TP loadings (by weight).

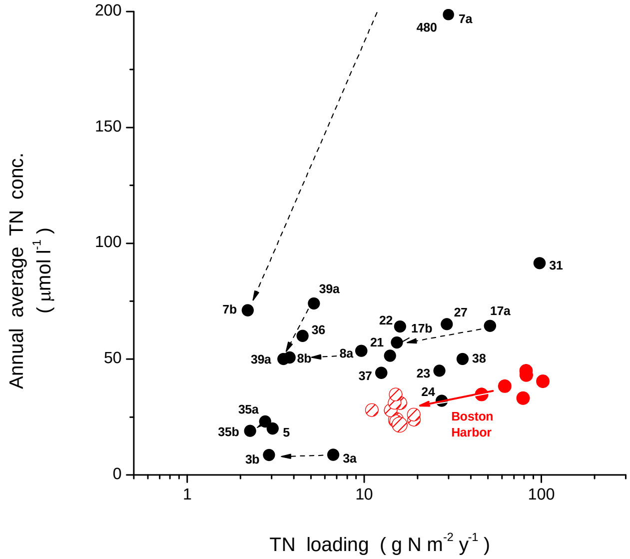
loadings received by other highly urbanized bays and estuaries such as Tokyo Bay and the Scheldte Estuary. After the completion of the BHP, the harbor TN- and TP- loadings ranked between those reported for the Chesapeake Bay and Patuxent River Estuary, and the Seto Inland Sea and Moreton Bay.

4.2 Comparison of the changes in Boston Harbor and other systems to which nutrient or organic matter inputs have been decreased

Boston Harbor is a highly-flushed, tidally dominated bay-estuary, of the type common along the northeast coast of the USA. Little is known of the eutrophication of these types of systems, but the evidence available suggests that their symptoms of eutrophication are quite different from those of the better-studied, less-flushed and often river-dominated bays and estuaries that occur further south in the USA. During the period the harbor received its very elevated loadings, it did not show the dense phytoplankton blooms, or anoxia or hypoxia typical of hyper-eutrophic, less-flushed systems. Based on its changes during the BHP however (Taylor et al. submitted), the harbor did show a series of characteristics that might be viewed as symptoms of eutrophication of this type of coastal system.

At the start of the BHP annual- and summer- average phytoplankton biomasses in the harbor (measured as chl-a) were elevated by factors of 1.4 and 1.9, but were not as high as in other systems that received similar N and P loadings (Fig. 16). Kelly (1997) and Kelly and Doering (1997), using the data available for the harbor in the early 1990's, showed a similar phenomenon for average DIN-concentrations and rates of ^{14}C primary production plotted against loadings of N-alone. When they scaled the loadings by the hydraulic residence times of the systems, the harbor data fell in line with the other systems. Annual average concentrations of TN and TP before the BHP were elevated by factors of 1.5 and 1.3, respectively. Annual average primary production measured at the mouth of the harbor was elevated by a factor of 1.7. The annual average PC-concentrations were elevated by a factor of 1.5, and the minimum DO- concentrations in the harbor bottom-waters each year were lowered by a factor of 0.92. The once extensive *Zostera* seagrass beds in the harbor (Addy and Aylward 1944), covered < 1% of the harbor area. The average (summer) rates of sediment oxygen uptake, and the (May-Oct) average net fluxes of DIN from the sediments, were both elevated by factors of 4.4.

During the period the harbor received its elevated loadings, the diversity of its benthic invertebrate communities was greater than in other systems that received similar loadings (Fig. 17), but its diversity was (depending on the index of diversity employed) 0.4 or 0.7 of the diversity after the discharges were ended. The percent incidence of *Ampelisca* mats on the sediment surface was also greater during the wastewater discharges than after The proportion of the N-inputs retained by the harbor was ~one-fourth of the proportion retained at lower loadings. Thus, at elevated loadings, the efficiency with which the harbor was able to retain the added inputs, was decreased.



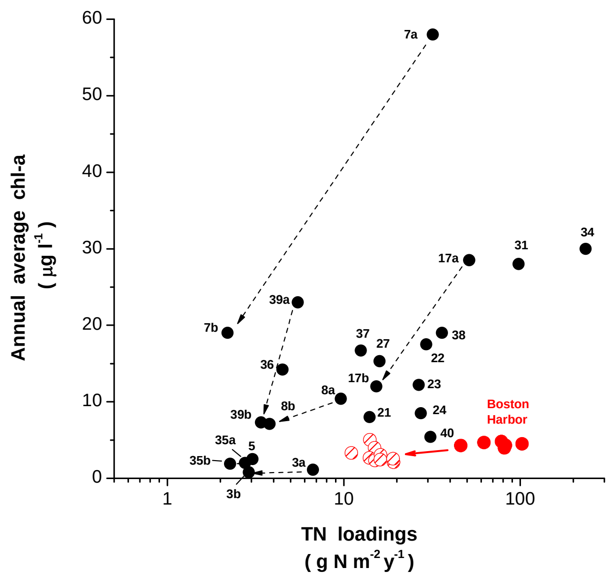


Fig. 16. Relationships between average TN- (top) and chl-a-concentrations (bottom), and the average TN-loadings to Boston Harbor and other coastal ecosystems. Numbering is as in Fig. 3, except: 36 Choptank River estuary (N concentration data from Boynton and Kemp 2000, chl-a- data from Whitall et al. 2010), 37 South San Francisco Bay (N concentration data are DIN data, from Smith and Hollibaugh 2006, chl-a from USGS), 38 Dutch Wadden Sea (Philippart et al 2007), 39 a, b New River estuary before and after (Mallin et al. 2005), 40 Tagus River Estuary (Gameiro et al. 2004), 41 North San Francisco Bay (N concentration data are DIN, Wilkerson et al. 2006).

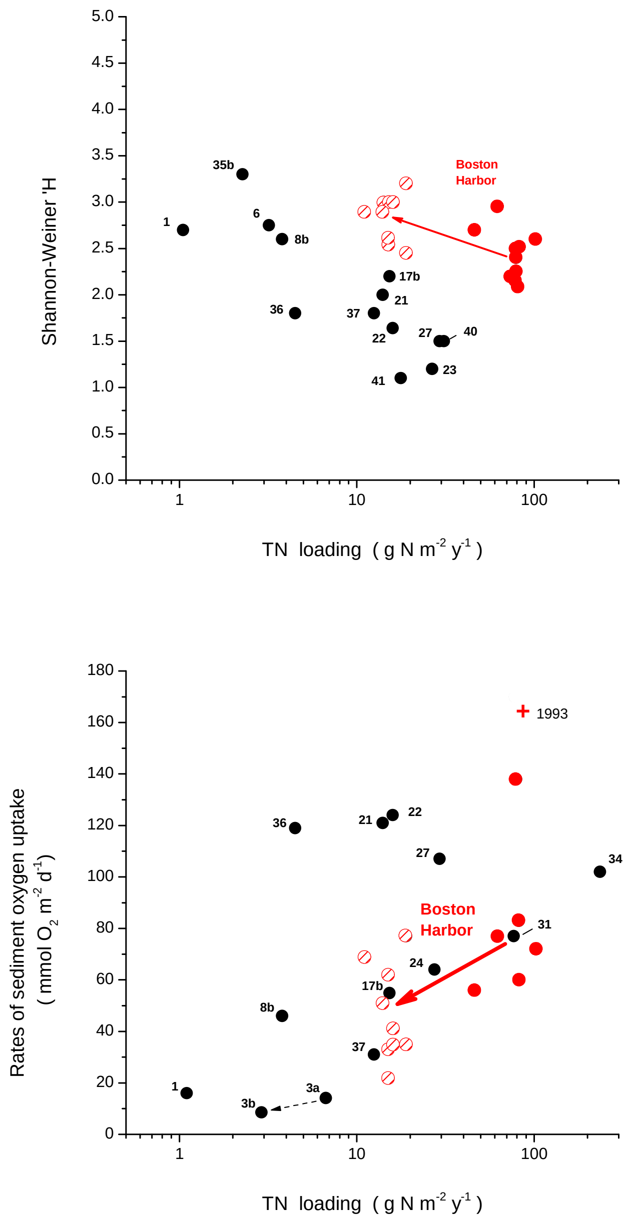


Fig. 17. Relationships between Shannon-Weiner H' (top) and rates of sediment oxygen uptake (bottom), and the TN-loadings to Boston Harbor and other coastal ecosystems. Numbering is as in Figures 3 and 15. The 1993 and 1995 harbor SOD data, which are averages for only the two stations in the NWH, are denoted by '+' symbols.

4.3 What processes were involved in the reversal of the harbor eutrophication?

We cannot be certain of the complex interactions and feed-backs that occurred within the harbor during the study, but the loading-response trajectory and budget data together indicate the overall changes to the harbor were cause by the decreased C and especially N inputs brought about by the BHP. The large decreases in the N-inputs brought about largely by the two wastewater diversions, caused phytoplankton and biomass production in the harbor to decrease. The decrease in C inputs brought about by the wastewater treatment- upgrades and diversions were augmented by the decrease in pelagic primary production in the harbor. Based on the limited area covered by seagrass beds in the harbor, the decrease in pelagic primary production has to date not been compensated for by an increase in production by these benthic macrophytes.

The decrease in the combined (external + internal) inputs of C apparently in turn caused decreases in PC-concentrations in the water-column, and an increase in the monthly minimum bottom-water DO-concentrations. The decreases in PC concentrations and presumably, in turn decreased rates of PC-inputs to the sediments, caused the harbor sediments to change significantly. Changes to the sediments included decreases in the area covered by amphipod tube mats, increases in the diversity of the fauna associated with the sediments, increased occurrence of benthic species typically seen in the less-enriched Massachusetts Bay (Maciolek et al. 2010), an increase in the redox potential discontinuity depth (Tucker et al. 2010), and decreased rates of sediment oxygen uptake and nutrient fluxes.

4.4 How do the harbor changes compare with other systems?

Boston Harbor is one of the relatively few systems in the USA and Western Europe that have experienced decreases in loadings of the materials responsible for eutrophication. The sizes of the decreases in inputs were also larger than the decreases experienced by these other systems. Table 2 compares the responses of Boston Harbor and nine of these other systems. The decreases in the concentrations of both N and chl-a were as for most, but not all of the other systems. Two systems, the Danish coastal waters and the upper Patuxent River Estuary, showed decreases in N but no decrease in chl-a. In the Danish coastal waters, a relatively open system that received low base N-inputs (Fig. 14), the decreases in loadings may not have been sufficient to cause measurable changes to phytoplankton biomass. In the upper Patuxent River Estuary, phytoplankton biomass was not decreased, apparently because primary production in this turbid estuary was limited by water clarity and not by nutrient availability (Testa et al. 2008).

The absence of an increase in water clarity in the harbor was unlike for most of the other systems. In five of the six other systems for which water clarity data were available clarity increased. Only in the upper Patuxent River, and for probably different reasons than in the harbor, was clarity not increased. In the upper Patuxent, where factors other

Table 2. Comparison of the responses of Boston Harbor and nine other systems that have experienced decreases in nutrient- and/or organic matter- inputs.

System	System characteristics	Loadings decreased	TN or DIN (umol l^{-1})	Chl-a (ug l^{-1})	DO (mg l^{-1} or % sat.)	Shannon-Weiner 'H	Sediment O_2 uptake mmol $O_2 m^{-2} d^{-1}$)	Water clarity Secchi or k	References
Boston Harbor, USA	Shallow (6.5 m), well-flushed (5-7d) high-salinity (31 psu) bay	N, P, C	Decrease 32 to 21	Decrease 4.5 to 3.2	Increase 6.2 to 6.9	Increase 2.1 to 3.0	Decrease 150 to 50	No change ~0.5	Present study, water clarity as k, DO values are minimum summer values
Danish Coastal waters, Denmark	Deep (51 m), moderately flushed, medium salinity (10- 20 psu) coastal waters	N, P (C not reported)	Decrease 23 to 12	No change ~2	-	-	-	-	N conc. data from HELCOM (2009), chl-a data from Hjorth and Josefson (2010)
Hillsborough Bay, USA	Shallow (3.2 m), moderately flushed estuary (40 d), medium salinity (15 - 30 psu)	N, P (C not likely decreased)	Decrease 64 to 57	Decrease 29 to 12	-	-	-	Increase 0.6 to 1.0	Johansson (2000), clarity as sechhi depth
Kaneohe Bay, USA	Shallow (8.5 m), well-flushed (8 d), high salinity (35 psu) bay	N, P, C	Decrease 8.63 to 8.57	Decease 1.1 to 0.8	-	-	Decrease 14 to 9	Increase 0.32 to 0.26	Smith et al. (1981), clarity is k
Laajalahti Bay, Finland	Very shallow (2.4 m), moderately flushed (40 d), low salinity (5 psu) inlet	N, P, C	Decrease 640 to 160	Decrease 70 to 20	Increase 28 to 80	-	-	Increase 0.4 to 0.8	Kauppila et al. (2005), water clarity is secchi depth, DO data are % sat. data, all data summer averages, except for N which are winter averages. N data are DIN values.
New River Estuary, USA	Shallow (1-2 m), poorly flushed (65 d), mesohaline to polyhaline lagoon	WWTF N, P (C likely decreased but not reported	Decrease 74 to 50	Decrease 23 to 16	Increase (?) 5.5 to 6.4	-	-	Increase 1.8 to 1.4	Mallin et al. (2005), water clarity data are k values, N conc. data are DIN data, increase in bottom-water DO not significant, data are annual averages
Upper Patuxent River Estuary, USA	Shallow (1.1 m), moderately flushed (30-65 d), low salinity (< 5 psu) region of estuary	WWTF N, P (C not reported)	Decrease 64 to 46	No change 17 to 16	Increase 2.5 to 3.75	-	-	No change ~0.35	Testa et al. (2008), Kemp et al. (2009), N conc. data are DIN, water clarity is secchi depth, DO is summer concentrations
Tagus River Estuary, Portugal	5-30 m deep, moderately flushed (10 d), moderate salinity (15-28 psu) estuary	C, N, (P not reported likely decreased	No change	Decrease 9 to 4	-	Increase 0.7 to 1.4	-	-	Benthic infauna data from Chainho et al. (2010) , N and chl-a from Gameiro and Brotas (2010)
Tampa Bay, USA	Shallow (3.5 m), poorly flushed (165 d), high salinity (28-30 psu) estuary	N, P, (C not reported but likely decreased)	Decrease 54 to 51	Decrease 10 to 7	-	-	-	Increase 1.25 to 2.0	Greening and Janicki (2006), clarity is secchi depth
Thames River Estuary, UK	Low salinity portion of estuary	C, (N and P not reported but P likely decreased)	-	-	Increase 5 to 40	-	-	-	Andrews and Rickard (1980), DO is summer % saturation

than chl-a regulated clarity, the decreases in nutrient loadings did not cause the relatively elevated chl-a concentrations (~17 µg l^{-1}) to decrease. In the harbor, a much more loaded, but more much more rapidly flushed system, the decreases in the low chl-a-concentrations were apparently not sufficient to increase clarity.

Increases in DO were seen in three of the four other systems for which DO- data were measured. Only in the New River estuary, a very shallow (1-2 m) lagoon system, did system-wide bottom-water DO not increase (Mallin et al. 2005). The increase in Shannon-Weiner H' in the harbor agreed with the inverse correlation that appears to exist between Shannon-Weiner H' and TN-loadings for all systems combined (Fig. 15). As in the harbor, Shannon-Weiner H' in the Tagus River Estuary increased after wastewater inputs to the estuary were decreased. Rates of SOD decreased in Kaneohe Bay after wastewater discharges to the bay were diverted offshore (Smith et al. 1981). Seagrass expansions have also been observed in other formerly-enriched, shallow systems subjected to decreases in wastewater inputs (e.g. Hillsborough Bay, Johansson 2000; Mumford Cove, Vaudrey et al. 2010).

A number of studies have suggested that the recovery of coastal systems subjected to decreases in nutrient- or organic- inputs may be delayed by internal stores of nutrients or organic matter within especially the sediments of the systems (Kauppila et al. 2005, Soetaert and Middelburg, 2009). The linear loading-response trajectories we observed for the harbor, and the small contributions made by the sediments to the harbor C- and N-budgets (Tucker et al.), together indicate that this was not the case for Boston Harbor. The changes entailed in the reversal of eutrophication of the harbor were rapid and occurred in linear proportion to the inputs.

4.5 Comparison with model predictions

During the design phase of the BHP, a three-dimensional, time-variable water quality model (the Bays Eutrophication Model, BEM) was used to predict the effects various wastewater engineering scenarios would have on the harbor ecosystem. The model was run using boundary conditions and estimates of loadings for two periods; 1989 to April 1990, and January through December 1992. Three modeled scenarios are relevant here (Table 3). The first scenario aimed to simulate pre-existing conditions, and involved primary treatment with continued wastewater discharges to the harbor. Scenario II entailed an upgrade to secondary treatment, but with continued discharge to the harbor. The third scenario, which was the scenario ultimately implemented, entailed both the upgrade to secondary treatment and the diversion of the discharges offshore.

For the three variables shown in the Table, the directions of the changes in the harbor were as predicted by modeled Scenario III. The 3.8-µg l^{-1} (or 50%) decrease in August chl-a was similar to the ~4.4 µg l^{-1} (or 60%) predicted by the BEM (Scenario III). The 2.2 µmol l^{-1} decrease in summer DIN was slightly greater than the 0.9 µmol l^{-1} decrease predicted by

Table 3. Comparison of the changes observed in Boston Harbor and the changes predicted using the Bays Eutrophication Model (BEM) run to predict the effects of two engineering scenarios. The estimated changes are derived from regression plots using estimates of the decreases in inputs caused by different levels of wastewater treatment. Modeled or simulated data are from HydroQual and Normandeau (1995).

ECOSYSTEM COMPONENT	SCENARIO					
	I		II		III	
	Existing condition		Upgrade to secondary treatment, but continued discharge to harbor		Upgrade to secondary treatment + diversion of discharges offshore	
	Simulated [a, e]	Observed	Simulated [e]	Estimated [d]	Simulated	Observed [e]
Chl-a ($\mu g\ l^{-1}$) [b]	6.9	7.6	-0.6 (-9%)	-0.4 (-6%)	-4.4 (-60%)	-3.8 (-50%)
DIN ($\mu mol\ l^{-1}$) [b]	1.4	5.6	0 (0%)	-0.5 (-9%)	-0.9 (-64%)	-2.2 (-40%)
DO ($mg\ l^{-1}$) [c]	6.9 [a, c]	6.2	+0.9 (+13%)	+0.1 (+2%)	+1.3 (+18%)	+0.7 (+11%)

[a] Pre-existing condition was simulated using data from two periods, October 1989 to April 1990, and January through December 1992.
[b] Modeled chl-a and DIN data are August surface values; observed values are averages of surface + bottom during summers (J,J,A,S). [c] Modeled DO data are August minimum values, observed DO data are monthly minimum, bottom-water values. [d] estimated using loading-response models determined in this study. [e] values are area-weighted harbor-wide averages computed from contour plots generated by the BEM.

by the BEM; in percent terms the decreases were similar, 50% and 64%, respectively. The 0.7-mg l^{-1} increase in minimum DO was smaller than the predicted increase, 1.25 mg l^{-1}. Again, in percent terms, the observed (+12%) and predicted changes (18%) were similar. The fact that the model simulations were conducted using boundary conditions 10 or more years before the discharges were diverted offshore, may account for (at least part) of the small differences between the observed and predicted changes.

Comparison of Scenarios III and II provided an opportunity to quantify the effects that the diversion of the secondary-treated discharges offshore had on the changes to the harbor. The model predicted the diversion would have a large impact on chl-a and DIN in the harbor, but that the bulk of the increases in bottom-water DO would occur with the upgrade to secondary treatment. The comparison of the increase in bottom-water DO observed in the harbor (equivalent to Scenario III) and the estimates of the increase in DO that might have followed the upgrade to secondary-treatment alone, suggests that the diversion of the discharges offshore (and perhaps the decrease in primary production in the harbor caused by the diversion) was largely responsible for the increase in bottom-water DO.

4.6 Overview of the reversal of harbor eutrophication

Figure 18 provides a schematic model of the changes that hypothetically follow the decreases in nutrient loadings to moderately- to poorly- flushed systems (left panels) and highly-flushed tidally-dominated systems such as Boston Harbor (right panels). In Boston Harbor, the decreases in nutrient inputs caused phytoplankton biomass, which was low relative to loadings, to decline. The bottom-water DO-concentrations, which were high relative to loadings, were increased. Unlike in the schematic model, we were unable to detect the expected small increase in clarity in the harbor. The start of the expansion of the seagrass beds in the harbor was as for both sets of scenarios.

4.7 Are the changes brought about by the BHP complete?

The BHP has reversed the historic eutrophication of the Boston Harbor ecosystem. The reversal was rapid, and has been sustained through the first nine years since the wastewater discharges to the harbor were discontinued. The expansion of the seagrass beds in the harbor may still be underway. If these beds continue to expand, and cover the extensive areas of the harbor they did in the 1930's, then additional changes to the system might be expected. Extensive seagrass beds can significantly impact the structure and function of a shallow coastal ecosystem. The total inputs of N to the harbor have likely been decreased to levels the harbor received in the 1800's. The changes to the harbor were rapid, but may not be complete.

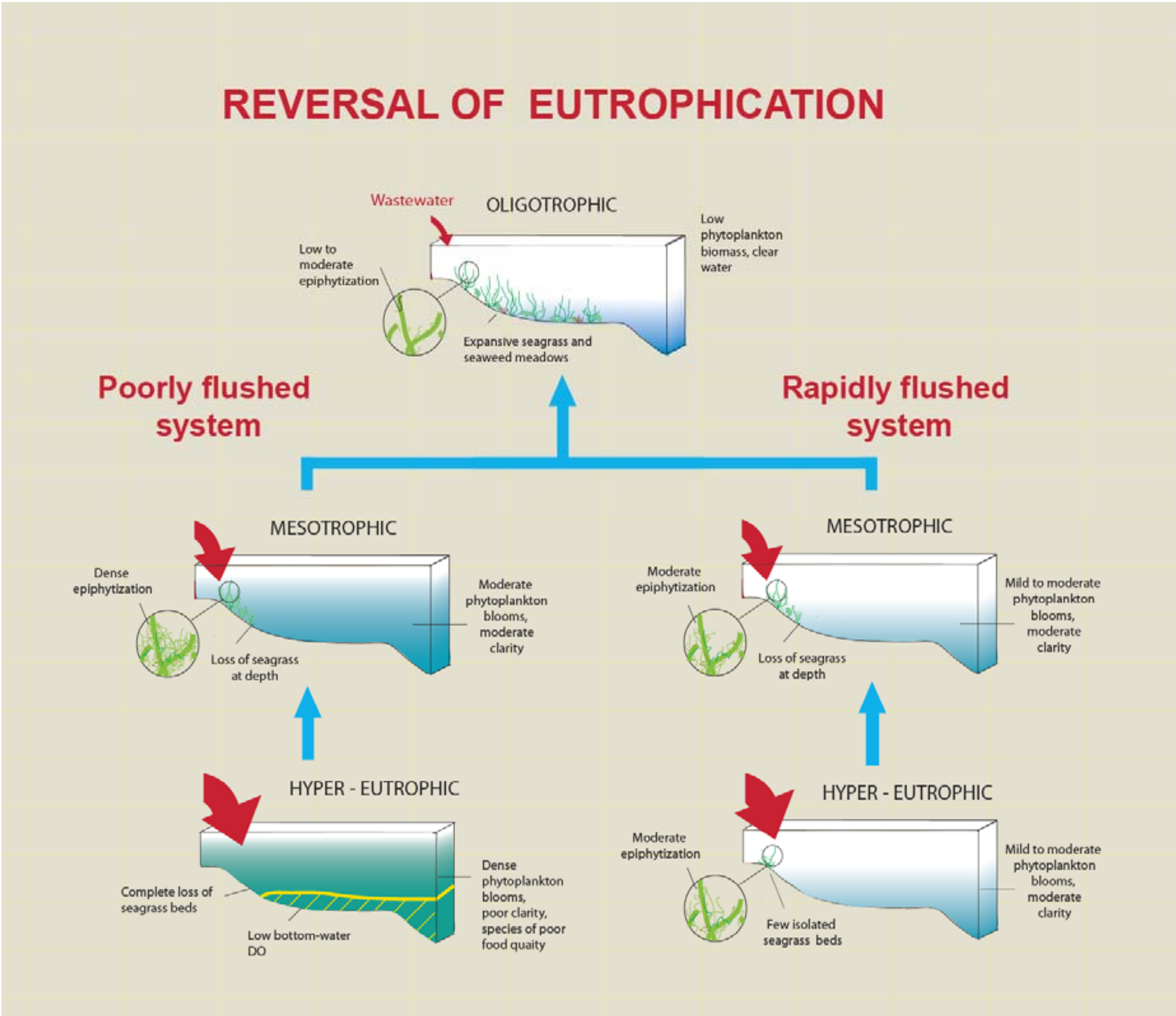


Fig. 18. Conceptual model of the changes that occur to the plant communities of shallow, moderately- (left) and well- flushed (right) coastal aquatic ecosystems following decreases in nutrient loadings.

LITERATURE CITED

Adams, E.E., J.W. Hansen, R.F. Lago, P. Clayton and X. Zhang. 1992. A simple box model of the nitrogen cycle in Boston Harbor and the Massachusetts Bays. Civil Engineering Practice. Fall 1992, 91-103.

Addy, C.E., Aylward, D.A., 1944. Status of eelgrass in Massachusetts during 1944. Journal of Wildlife Management 8, 269-275.

Alber, M., and A.B. Chan. 1994. Sources of contaminants to Boston Harbor: revised loading estimates. Boston: Massachusetts Water Resources Authority. Report 1994-01. 93 p.

Anderson D.A., Glibert P.M., Burkholder J.M., 2002. Harmful algal blooms and eutrophication: Nutrient sources, composition and consequences. Estuaries 25, 562-584.

Andrews, M.J., Rickard, D.G., 1980. Rehabilitation of the inner Thames Estuary. Marine Pollution Bulletin 11, 327-332.

Bowen, J.L., Valiela, I., 2001. The ecological effects of urbanization of coastal watersheds: historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. Canadian Journal of Fisheries and Aquatic Science 58, 1489-1500.

Boynton, W.R., Kemp, W.M., 2000. Influence of river flow and nutrient loads on selected ecosystem processes: a synthesis of Chesapeake Bay. In: Hobbie JE (Ed) Estuarine research: a synthetic approach to research and practice, Island Press, Washington DC, p 269-298.

Boynton, W.R., Hagy, J.D., Cornwell, J.C., Kemp, W.M., Greene, S.M., Owens, M.S., Baker, J.E., Larsen, R.K., 2008. Nutrient budgets and management actions in the Patuxent River Estuary, Maryland. Estuaries and Coasts 31, 623-651.

Breault, R.F., Sorenson, J.R., Weiskel, P.K., 2002. Streamflow, water quality, and contaminant loads in the lower Charles River Watershed, Massachusetts, 1999-2000. USGS Water-Resources Investigations Report 2002-4137, 131 p.

Breitburg, D.L., Hondorp, D.W., Davias, L.A., Diaz, R.J. 2009. Hypoxia, nitrogen and fisheries; integrating effects across local and global landscapes. Annual Review of Marine Science 1, 329-349.

Breen, C., Vittands, J., O' Brien, D., 1994. The Boston Harbor Project: history and planning. Civil Engineering Practice 9, 11-32

Caddy J.F., 1993. Toward a comparative evaluation of human impacts on fishery ecosystems of enclosed and semi-enclosed seas. Rev Fish Sci 1:57-95

Carstensen, J., Conley, D.J., Anderson, J.H., Ærtebjerg, G., 2006. Coastal eutrophication and trend reversal: A Danish case study. Limnology and Oceanography 51, 398-408.

Chainho, P., Silva, G., Lane, M.F., Costa, J.L., Pereira, T., Azeda, C., Almeida, P.R., Metelo, I., Costa, M.J., 2010. Long-term trends in intertidal and subtidal benthic communities in response to water quality improvement measures. Estuaries and Coasts 33, 1314-1326.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223-253.

Connor, M.S., Werme, C., Rex, A.C. 2012. Management Case Study: Boston Harbor/Massachusetts Bay. In: Treatise on Estuarine and Coastal Science, 1st Edition. Academic Press.

Costello, C.T., Kenworthy, W.J., 2011. Twelve-year mapping and change analysis of eelgrass (*Zostera marina*) areal abundance in Massachusetts (USA) identifies statewide declines. Estuaries and Coasts 34, 232-242.

Dauer, D.M., Ranasinghe, J.A., Weisberg, S.B. 2000. Relationships between benthic community condition, water quality, sediment quality, nutrient loads and land use patterns in Chesapeake Bay. Estuaries 23, 80-96.

Díaz, R.J., Rosenberg, R. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321, 926-928.

Diaz, R.J., Rhoads, D.C., Blake, J.A., Kropp, R.K., Keay, K.E., 2008. Long-term trends in benthic habitats related to reduction in wastewater discharges to Boston Harbor. Estuaries and Coasts 31, 118-1197.

Gameiro, C., Cartaxana, P., Cabrita, M.T., Brotas, V., 2004. Variability in chlorophyll and phytoplankton composition in an estuarine system. Hydrobiologia 525, 113-124.

Giblin, A.E., Hopkinson, C.S., Tucker, J., 1997. Benthic metabolism and nutrient cycling in Boston Harbor and Massachusetts Bay, Massachusetts. Estuaries 20, 46-364.

Grace, R., 2009. Marine Outfall Construction: Background, Techniques and Case Studies. ASCE Press, Reston, 403 pp.

Greening, H., Janicki, A., 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. Environmental Management 38, 163-178.

HELCOM, 2009. Eutrophication in the Baltic Sea – an integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region. Baltic Sea Environmental Proceedings No. 115B.

HydroQual and Normandeau, 1995. A water quality model for Massachusetts and Cape Cod Bays: calibration of the Bays Eutrophication Model (BEM). Boston: Massachusetts Water Resources Authority. Report No. 1995-08.

Johansson, J.O.R., 2000. Historical overview of Tampa Bay water quality and seagrass: issues and trends. *In* H.S Greening (ed.). Seagrass management: it's not just nutrients! Tampa Bay Estuary Program Publication. St. Petersburg, Florida pp 1-10.

Kauppila, P., Weckström, K., Vaalgamaa, S., Korhola, A., Pitkänen, H., Reuss, N., Drew, S., 2005. Tracing pollution and recovery using sediments in an urban estuary, northern Baltic Sea: Are we far from ecological reference conditions? Marine Ecology Progress Series 290, 35-53.

Kelly, J.R., 1997. Nitrogen flow and the interaction of Boston Harbor with Massachusetts Bay. Estuaries 20, 365-380.

Kelly, J.R., Doering, P.H., 1997. Monitoring and modeling primary production in coastal waters: studies in Massachusetts Bay 1992-1994. Marine Ecology Progress Series 148, 155-168.

Kemp, W.M., Testa, J.M., Conley, D.J., Gilbert, D., Hagy, J.D., 2009. Temporal responses of coastal hypoxia to nutrient loading and physical controls. Biogeosciences 6, 2985–3008.

Knebel H.J., Rendigs, R.R., Bothner, M.H., 1991. Modern Sedimentary Environments in Boston Harbor, Massachusetts. Journal of Sedimentology and Petrology 61, 791-804.

Leschen, A.S., Ford, K.H., Evans, N.T., 2010. Successful eelgrass (*Zostera marina*) restoration in a formerly eutrophic estuary (Boston Harbor) supports the use of a multifaceted watershed approach to mitigating eelgrass loss. Estuaries and Coasts 33, 1340-1354.

Libby, P.S.A., Anderson, D.M., Borkman, D.G., Geyer, W.R.K., Oviatt, A.A., Oviatt, C.A., Turner, J.T., 2009. 2008 Water column monitoring results. Boston: Massachusetts Water Resources Authority. Report 2009-12. 31 p. + appendices.

Libby, P.S., Borkman, D.G., Geyer, W.R., Turner, J.T., Keller, A.A., McManus, M.C., Oviatt, C.A.. 2011. 2010 Water column monitoring results. Boston: Massachusetts Water Resources Authority. Report 2011-12. 36 p. + appendices.

Maciolek, N.J., Diaz, R.J., Dahlen, D.T., Doner, S.A., 2010. 2009 Boston Harbor Benthic Monitoring Report. Boston: Massachusetts Water Resources Authority. Report No. 2010-18.

Mallin, M.A., Mcivor, M.R., Wells, H.A., Parsons, D.C., Johnson, V.L., 2005. Reversal of eutrophication following sewage treatment upgrades in the New River Estuary, North Carolina. Estuaries 28, 750-760.

Nixon, S.W., 1995. Coastal eutrophication: a definition, social causes, and future concerns. Ophelia 41, 199-219.

Nixon, S.W. 2009. Eutrophication and the macroscope. Hydrobiologia 629, 5-19

Nixon, S.W., Arnrnerman, J.W., Atkinson, L.P., Berounsky, W.M., Billen, G., Boicourt, W.C., Boynton, W.R., Church, T.M., Ditoro, D.M., Elmgren, R., 1996. The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. Biogeochemistry 35, 141-180,

Oviatt, C.A., Hyde, K.J.W., Keller, A.A., 2007. Production patterns in Massachusetts Bay with outfall relocation. Estuaries and Coasts 30, 35-46.

Philippart, J.J.M, Beukema, J.J., Cadée, G.C., Dekker, R., Goedhart, P.W., van Iperen, J.M., Leopold, M.F., Herman, P.M.J. 2007. Impacts of nutrient reduction on coastal communities. Ecosystems 10, 95-118

Poe, A., Hackett, K., Janicki, S., Pribble, R., Janicki, A., 2005. Estimates of Total Nitrogen, Total Phosphorous, Total Suspended Solids, and Biochemical Oxygen Demand Loadings to Tampa Bay, Florida: 1999-2003. Tampa: Tampa Bay Estuary Program. Report No. #02-05.

Roberts, P.J.W., Salas, H.J., Reiff, F.M., Libhaher, M., Labbe, A., and Thomson, J.C., 2010. Marine Wastewater Outfalls and Treatment Systems. IWA Publishing, London, 491 pp.

Robinson, W.E., Coffey, T.J., Sullivan, P.A., 1990. New England Aquarium's Ten Year Boston Harbor Monitoring Program. First Report (March 1987 - July 1989). New England Aquarium Report Series. 108 pp. + appendices.

Short, F.T., Wyllie-Echeverria, S., 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation 23, 17-27.

Smith S.V., Kimmerer, W.J., Laws, E.A., Brock, R.E., Walsh, T.W., 1981. Kaneohe Bay sewage diversion experiment: perspectives on ecosystem responses to nutrient perturbation. Pacific Science 35, 279-395.

Soetaert, K., Middelburg, J.J., Heip, C., Meire, P., Van Damme, S., Maris, T., 2006. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldte estuary (Belgium, The Netherlands). Limnology and Oceanography 5, 409-423.

Taylor, D.I., 2010. The Boston Harbor Project, and decreases in loadings of eutrophication-related materials to Boston Harbor. Marine Pollution Bulletin 60, 609-619.

Taylor, D.I., Oviatt, C.A., Borkman, D.G., 2011. Non-linear responses of a coastal aquatic ecosystem to large decreases in nutrient and organic loadings. Estuaries and Coasts 34, 745-757.

Testa, J.M., Kemp, W.M., Boynton, W.R., Hagy, J.D., 2008. Long-term changes in water quality and productivity in the Patuxent River estuary: 1985 to 2003. Estuaries and Coasts 31, 1021-1037.

Tucker, J., Kelsey, S., Giblin, A.E., 2010. 2009 benthic nutrient flux annual report. Boston: Massachusetts Water Resources Authority. Report 2010-10. 27 p.

Valiela, I., Foreman, K., LaMontagne, M., Hersh, D., Costa, J., Peckol, P., DeMeo-Andreson, B., D_Avanzo, C., Babione, M., Sham, C., Brawley, J., Lajtha, K., 1992. Couplings of watersheds and coastal waters: sources and consequences of nutrient enrichment in Waquoit Bay, Massachusetts. Estuaries 15, 443-457.

Vaudrey, J.M.P., Kremer, J.N., Branco, B.F., Short, F.Y., 2010. Eelgrass recovery after nutrient enrichment reversal. Aquatic Botany 93, 237-243.

Whitall, D., Hively, W.D., Leight, A.K., Hapeman, C.J., McConnell, L.L., Fisher, T., Rice, C.P., Codling, E., McCarty, G.W., Sadeghi, A.M., Gustafson, A., Bialek, K., 2010. Pollutant fate and spatio-temporal variability in the Choptank River Estuary; factors influencing water quality. Science of the Total Environment 408, 2096-2108.

Wilkerson, F. P., Dugdale, R. C., Hogue, V. E., Marchi, A., 2006. Phytoplankton blooms and nitrogen productivity in San Francisco Bay. Estuaries and Coasts, 29, 401-416.

Zarbock, H.W., Janicki, A.J., Wade, D.L., Heimbuch, D.G., 1994. Estimates of Total Nitrogen, Total Phosphorus and Total Suspended Solids Loadings to Tampa Bay, Florida. Tampa: Tampa Bay National Estuary Program Report No. 04-94.

Massachusetts Water Resources Authority
Charlestown Navy Yard
100 First Avenue
Boston, MA 02129
(617) 242-6000
http://www.mwra.state.ma.us