RESEARCH ARTICLE

Wastewater input reductions reverse historic hypereutrophication of Boston Harbor, USA

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Abstract This paper documents the changes that followed large nutrient (N and P) and organic matter input reductions to a major metropolitan marine bay, Boston Harbor (USA). Before input reduction, its N and P inputs fell in the upper range of the $< 1 \rightarrow 300 \text{ gN m}^{-2} \text{ year}^{-1}$ and $< 0.1 \rightarrow 40 \text{ gP m}^{-2} \text{ year}^{-1}$ for coastal systems. Elevated nutrient and organic matter inputs are recognized causes of coastal eutrophication. Treatment upgrades and then diversion of its wastewater discharges offshore, lowered its N, P, and organic C inputs by 80–90%. The input decreases lowered its trophic status from hypereutrophic to eutrophic–mesotrophic. With the reversal of hypereutrophication, pelagic production and phytoplankton biomass decreased, and the nitrogen limitation relative to phosphorus limitation increased. Benthic metabolism and dissolved inorganic N fluxes decreased, and benthic–pelagic coupling was altered. Bottom-water dissolved oxygen, already at healthy levels, increased, and seagrass expanded. Coastal management requires that the changes, following the nutrient and organic matter input reductions implemented to address eutrophication, be understood. Boston Harbor’s recovery, because its water column was vertically well mixed and marine, was more pronounced than in many other systems.

Keywords Eutrophication · Nitrogen · Phosphorus · Recovery · Wastewater

INTRODUCTION

During the past 30–40 years, considerable effort has been expended, especially in developed countries, on policies to reduce the anthropogenic nutrient and organic matter inputs responsible for widespread degradative eutrophication of coastal aquatic ecosystems (Cloern 2001; Nixon 2009; Conley 2012). Excess inputs of nitrogen (N) and phosphorus (P) and organic matter (organic carbon, C) have been the recognized causes of eutrophic conditions, including increased pelagic primary production and dense phytoplankton blooms, increased anoxia, or hypoxia, changes to benthic–pelagic coupling, and loss of seagrass habitats. Much of the policy focus has been on N, the primary limiting nutrient in most temperate coastal marine systems (Howarth and Marino 2006). Because P can be limiting in some systems (Burson et al. 2016; Berthold et al. 2018), and P decreases to lakes and impoundments can cause N exports from these systems to coastal bays and estuaries to increase (Paerl 2009), management of coastal eutrophication on global and regional scales requires both N and P inputs be addressed (Conley et al. 2009). Coastal management requires that the responses ecosystems show to policy-driven nutrient and organic matter input reductions be understood. To help meet this need, this paper documents the changes that followed large N, P, and organic C input reductions to Boston Harbor, a major metropolitan marine bay in the northeast USA, and then compares the changes using both the N and P inputs as references, with other systems to which inputs of these materials have been lowered.

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MATERIALS AND METHODS

Study site—Boston Harbor

Boston Harbor is a shallow, vertically well-mixed marine bay, completely surrounded by metropolitan Boston, with a metropolitan statistical area population of 4.3 million (Fig. 1). In the early 1990s, its N and P total inputs per unit waterbody surface area were among the highest for major bays or estuaries (Boynton et al. 2008), with 80–90% contributed by two metropolitan wastewater-treatment facilities, WWTFs (Alber and Chan 1994; Taylor 2010). During 1991–2001, the harbor was the site of a major wastewater-improvement project that upgraded treatment at these two facilities, and then in two phases diverted their discharges offshore (Fig. 2). Boston Harbor at mid-tide has an area of 108 km², a volume of 643×10^6 m³, and an average depth of 5.5 m (Signell and Buttmann 1992). Its average tidal range is 2.7 m, and tidal exchanges with Massachusetts Bay average $\sim 3500\text{--}\sim 4300$ m³ s⁻¹ (Kelly 1998). Hydraulic residence time is 6–7 days (Ketchum 1951). Its water column is well mixed

throughout the year. Soft sediments cover 51% of its sea-floor. Salinity averages 31 psu.

Overview of measurements of inputs and ecosystem changes, and comparisons with other systems

Details of all statistical tests have been provided in the primary publications cited for each topic below. Land-based total N, P and organic C inputs to Boston Harbor were measured from its two wastewater-treatment facilities, four largest tributary rivers (the Charles, Mystic, Neponset and Weymouth-Weir rivers), and nonpoint sources (combined sewer overflows + storm water + groundwater + atmosphere) (Taylor 2010). The water column was sampled at 13 locations (Fig. S1, Taylor et al. 2011). ¹⁴C pelagic primary production was determined at one location, using ¹⁴C production versus irradiance incubations, from curves from five depths (Oviatt et al. 2007). Pelagic primary production prior to 1995 was determined from measured N input data, using this relation between measured annual production (g C m⁻² year⁻¹) and annual N inputs (g N m⁻² year⁻¹) for the period 1995–2010; pelagic production = $312 + (6.6 \times \text{total N})$

Fig. 1 Boston Harbor and the two wastewater-treatment facilities that formerly discharged to the harbor

Fig. 2 The Boston Harbor wastewater-improvement project and changes to locations of wastewater sludge and effluent discharges

inputs), $r^2 = 0.76$. Pelagic production N and P demand were determined stoichiometrically using the Redfield Ratio (C:N:P = 41:7:1, by mass). All water column data, except for primary production, have been reported as harbor-wide averages, volume weighted by region. Eelgrass (*Zostera marina*) production was estimated from bed area measured by Costello and Kenworthy (2011) and Massachusetts Department of Marine Fisheries (unpublished data), assuming aboveground production to be $350 \text{ gC m}^{-2} \text{ bed year}^{-1}$.

Sediments were profiled at 61 locations using a sediment profile-imaging camera (Diaz et al. 2008). Benthic organic C mineralization rates and dissolved inorganic nutrient fluxes were measured on duplicate cores, collected at 2–4 locations, from March through October (Tucker et al. 2014). Benthic organic C mineralization rates were determined from measured sediment oxygen uptake, assuming a Respiratory Quotient of 0.8. Benthic organic C mineralization rates and dissolved inorganic N (DIN) flux have been expressed per unit area entire harbor, assuming soft sediments covered 51% of the harbor bottom. Annual estimates were computed by averaging the March through October values, and then estimating the average for the remaining November through March period using the differences in water temperatures between the two periods, assuming a Q_{10} of 2.

Table 1 lists the 11 systems in Western Europe and the USA with which Boston Harbor was compared. Loading estimates for the 11 systems include wastewater plus river, and where available, atmospheric plus nonoceanic non-point sources. System data are reported per unit waterbody surface area. All data for these systems have been published by others, with sources cited below.

RESULTS

Upgrades to wastewater treatment and diversion of wastewater discharge from Boston Harbor reduced inputs of organic C by 94%, N by 82%, and P by 94% (Fig. 3, Taylor 2010). Concurrent with the input reductions, annual pelagic primary production rates fell by 56%, from 790 to $350 \text{ gC m}^{-2} \text{ year}^{-1}$ (Fig. 4, Oviatt et al. 2007), and summer phytoplankton biomass (chlorophyll-*a*) 42%, from 6.5 to $3.8 \mu\text{g l}^{-1}$ (Table 2, Taylor et al. 2011). Pelagic production rates before inputs were lowered, were in the upper range for coastal systems worldwide (Cloern et al. 2014). Phytoplankton biomass was moderate, but because of short residence time (6–7 days) and flushing with less enriched coastal waters, was lower than in other systems receiving similar N inputs (Kelly 1998). High production rates with moderate biomass have been documented in the marine portions of other highly N enriched estuaries where transparency is greatest (Howarth et al. 2006).

Boston Harbor's N and P input decreases were each sufficient to account fully for the pelagic production decrease, and to shift the primary source of nutrient inputs away from land-based point sources (Table S1). Before the wastewater-improvement project, inputs from WWTFs supported 60–70% of primary production. After decreases in those inputs, most of the primary production was supported by nonpoint coastal inputs. Based on the decrease in the dissolved inorganic N: dissolved inorganic P concentration ratios (DIN:DIP), which were low relative to Redfield, the N relative to P limitation of its pelagic production was increased. The 1% photosynthetic compensation depth was unchanged, remaining at an average of 9.4 m (Taylor et al. 2011), 1.7 times mid-tide water depth, suggesting pelagic production was not light limited.

Table 1 Characteristics of Boston Harbor and 11 other systems to which nutrient and organic C inputs have been lowered

System	System type	Sources of input reductions	System characteristics (depth, salinity, flushing, stratification)
Boston Harbor, USA ^a	Marine bay	Wastewater upgrades, diversions	Shallow (5.5 m), marine (31 psu), rapidly flushed (5–7 days), vertically well mixed
Baltic Sea proper, Scandinavia ^b	Inland sea	Agricultural	Deep (62 m), 6–8 psu, permanently stratified, residence time (20–33 years)
Chesapeake Bay, USA ^c	Large bay/estuary	Agricultural + wastewater	Depth (6.4 m), 90–180 days residence time, 0.5–30 psu, seasonally stratified
Danish coastal systems, Denmark ^d	Shallow bays, estuaries	Agricultural + wastewater	Shallow, 7–31 psu, residence time (days to months), diverse stratification
Hillsborough Bay, USA ^e	Marine bay	Wastewater upgrades	Depth (3 m), 15–30 psu, 20 days residence time
Kaneohe Bay, USA ^f	Marine bay	Wastewater upgrades, diversions	Depth (8.4 m), 34–35 psu, residence time 10 days, well mixed
Laajalahti Bay, Finland ^g	Small estuary	Wastewater upgrades, diversion	Shallow (2.4 m), residence time 40 days, salinity (4.6 psu), vertically well mixed
Narragansett Bay, USA ^h	Bay/estuary	Wastewater upgrades	Depth (10 m), 20 days residence time, 29–30 psu, seasonally stratified
New River Estuary, USA ⁱ	Coastal lagoon	Wastewater upgrades, diversions	Shallow (1–2 m), 64 days residence time, 5–25 psu, vertically well mixed
Patuxent Estuary, USA ^j	Large estuary	Agricultural + wastewater	Depth (5.4 m), 5–18 psu, 6–68 days residence time, seasonally stratified
Roskilde Fjord, Denmark ^k	Shallow estuary	Wastewater treatment + diffuse sources	Shallow (3.0 m), 90 days residence time, vertically well mixed, 10–20 psu
Tampa Bay, USA ^l	Marine bay	Wastewater upgrades	Depth (4 m), 15–38 psu, 26–53 days residence time, vertically well mixed

^aTaylor et al. (2011), ^bHELCOM (2009), ^cKemp et al. (2005), ^dReimann et al. (2016), ^eGreening et al. (2014), ^fSmith et al. (1981), ^gKaupilla et al. (2005), ^hOviatt et al. (2017), ⁱMallin et al. (2005), ^jTesta et al. (2008), ^kStaeher et al. (2017)

Fig. 3 **a** Total (external + internal) organic C inputs and **b** land-based N and P inputs (updated from Taylor 2010). Organic C input bars are additive. Internal inputs include pelagic plus aboveground eelgrass production

Total organic C inputs from external sources and internal production decreased from 1050 to 320 gC m⁻² year⁻¹ during 1991–2010. Decreased pelagic production, which had accounted for 70% of total organic C inputs prior to input reductions, was responsible for 60% of the decrease, while decreases in the wastewater inputs accounted for the remainder. Summer average bottom-water dissolved oxygen (DO) concentration was unchanged, ranging from 7.0 to 9.4 mg l⁻², but minimum monthly-average DO, increased slightly (Fig. 4), coinciding with the final wastewater diversion in 2000 and evident in eight of the first nine succeeding summers. Because its water column was vertically well mixed, and rapidly flushed with surface coastal waters, it did not exhibit the anoxia or hypoxia traditionally associated with hypereutrophy.

Eelgrass (*Zostera marina*), as in other highly N-loaded systems (Latimer and Rego 2010), covered very little (73 ha) of the harbor (Costello and Kenworthy 2011). Its production, expressed m⁻² entire harbor, increased from 2.6 gC m⁻² year⁻¹ in 1994–1995 to 4.9 gC m⁻² year⁻¹ in 2016–2017. The increase, because eelgrass covered so little of the harbor, was equivalent to less than 1% of the pelagic decline. Boston Harbor's largely unvegetated benthic habitat was typical of enriched but oxic coastal systems;

Fig. 4 **a** Pelagic primary production, **b** phytoplankton biomass (chl *a*), **c** minimum monthly-average bottom-water dissolved oxygen (DO) concentrations, **d** sediment organic C mineralization rates, **e** sediment–water dissolved inorganic N (DIN) fluxes, and **f** the rates of benthic organic C mineralization and DIN flux expressed relative to total organic C inputs and pelagic production N demand. Trophic classification in top panel from Nixon (2009). Vertical bars = 1 × SD. Open circles in **d**, **e** denote years when the two benthic processes were measured only at the two stations in the northwest part of the harbor

most of its 61 sediment-profile stations were oxic (Diaz et al. 2008). Its benthic organic C mineralization rates (measured as rates of oxygen uptake) and sediment–water DIN flux were elevated (Fig. 4), but when averaged over the entire harbor, were mid-range for coastal systems (Tucker et al. 2014). Its sediments too did not serve as a large P source.

As a result of input reductions, both the benthic organic C mineralization rates and DIN flux were significantly lowered (Tucker et al. 2014). The declines, however, did not greatly contribute to eutrophication reversal; the reduction in DIN flux, for example, could account for only 3% of primary production decrease. The benthic organic C mineralization rate decline was equivalent to 13% of the total organic C input decrease. The percent equivalence of its benthic mineralization rates to total organic C inputs, and DIN flux to pelagic production N demand, increased,

but were insufficient to counteract its eutrophication reversal. DIP and DIN:DIP flux ratios, both of which were highly variable, were unchanged relative to before input decreases. Percent sediment organic C decreased from 2.4 to 1.6%.

DISCUSSION

Insights into how coastal systems respond to input reductions

Boston Harbor's large reductions in inputs of nutrients and organic C reversed its historic trophic status from hyper-eutrophic to borderline eutrophic–mesotrophic (categorization by Nixon 2009). Of 22 comparable coastal systems where nutrient and organic C inputs have been studied,

Table 2 Comparison of pelagic primary production and phytoplankton biomass, benthic organic C mineralization rates, and DIN flux, and water column nutrient chemistry before and after inputs were lowered

Variable	Period	Mean		Difference
		1995–1997	2001–2010	
Pelagic primary production ($\text{gC m}^{-2} \text{ year}^{-1}$)	Annual	790 ± 890	350 ± 260	$-440 (-56\%)^*$
Phytoplankton biomass ($\mu\text{g l}^{-1}$)	Summer	6.5 ± 1.5	3.8 ± 1.2	$-2.7 (-42\%)^*$
Benthic organic C mineralization rates ($\text{gC m}^{-2} \text{ year}^{-1}$)	Annual	152 ± 100	71 ± 31	$-81 (-53\%)^*$
Benthic DIN flux ($\text{gN m}^{-2} \text{ year}^{-1}$)	Annual	9.8 ± 11	5.4 ± 2.5	$-4.4 (-45\%)^*$
Total N ($\mu\text{g l}^{-1}$)	Annual	437 ± 92	295 ± 52	$-142 (-32\%)^*$
DIN ($\mu\text{g l}^{-1}$)	Annual	166 ± 86	79 ± 54	$-87 (-52\%)^*$
Total P ($\mu\text{g l}^{-1}$)	Annual	56 ± 12	39 ± 11	$-17 (-30\%)^*$
DIP ($\mu\text{g l}^{-1}$)	Annual	43 ± 15	28 ± 5	$-15 (-35\%)^*$
Total N:total P (by mass)	Annual	$8:1 \pm 2$	$7.6:1 \pm 1.6$	$-0.5 (-6\%)$
DIN:DIP (by mass)	Annual	$4:1 \pm 1.8$	$3:1 \pm 1.6$	$-1 (-25\%)^*$
DIN:DIP (by mass)	Summer	$2.8:1 \pm 0.5$	$2.1:1 \pm 0.4$	$-0.7 (-25\%)^*$

* $p < 0.05$, Mann–Whitney U test

Fig. 5 N and P total inputs per waterbody surface area, for Boston Harbor and 22 other coastal aquatic ecosystems (adapted from Boynton et al. 2008). Inputs include wastewater plus river inputs, and in some cases, also land-based nonpoint source plus atmospheric loadings. Solid diagonal line represents the Redfield N:P input ratio (by mass). Green labels indicate the 11 systems for which pre- and post-input reduction data are available, which are compared with Boston Harbor in this analysis. Data sources are from Boynton et al. (2008), except where other citations are given: 1 Boston Harbor (Taylor 2010), 2 Kaneohe Bay (Smith et al. 1981); 3 Baltic Sea (HELCOM 2009); 4 Laajalahti Bay (Kauppila et al. 2005); 5 Hillsborough Bay (Tampa Bay Estuary Program, unpublished); 6 Tampa Bay (Tampa Bay Estuary Program, unpublished); 7 Danish coastal systems (Reimann et al. 2016); 8 Gulf of Riga; 9 Chesapeake Bay; 10 Patuxent Estuary; 11 Narragansett Bay (before, Nixon et al. 2008; after, Oviatt et al. 2017); 12 Roskilde Fjord (Staehr et al. 2017); 13 New River Estuary (Mallin et al. 2005), 14 Moreton Bay; 15 Delaware Bay; 16 Tokyo Bay; 17 Westerschelde; 18 saline Hudson River Estuary (Howarth et al. 2006), 19 N San Francisco Bay; 20 Potomac Estuary; 21 Apalachicola Bay; 22 Mobile Bay

input reductions have been implemented for 12 (Fig. 5, Table S2). In all systems for which N inputs were lowered, so was P. Among the 12, the N decreases per unit waterbody surface area ranged from $< 1 \text{ gN m}^{-2} \text{ year}^{-1}$ for the

Baltic Sea proper and Danish coastal systems, to $66 \text{ gN m}^{-2} \text{ year}^{-1}$ for the harbor. P decreases spanned from $< 0.1 \text{ gP m}^{-2} \text{ year}^{-1}$ for the Baltic Sea and Danish coastal systems, to $15 \text{ gP m}^{-2} \text{ year}^{-1}$ for the harbor. In all four

Fig. 6 Pelagic primary production per waterbody surface area (diameters of circles, legend at lower right) for Boston Harbor and other coastal systems, shown on the same axes as in Fig. 6 for total N and total P inputs per waterbody surface area. Numbering as in Fig. 5. Sources for the production data are Baltic Sea (HELCOM 2009), Chesapeake Bay (Malone et al. 1988), Danish coastal waters (HELCOM 2009), Hillsborough Bay (Greening and Janicki 2006), Hudson River Estuary (Howarth et al. 2006), Kaneohe Bay (Smith et al. 1981), Narragansett Bay (Oviatt et al. 2002, 2017), Patuxent River Estuary (Flemer et al. 1970; Testa et al. 2008), Westerschelde (Soetaert et al. 1994)

systems for which pre- and post- data were available for production, it decreased (Fig. 6). Hillsborough Bay, like Boston Harbor, shifted from hypereutrophic to eutrophic–mesotrophic (Greening et al. 2014). Narragansett Bay changed from eutrophic to mesotrophic (Oviatt et al. 2017), and Kaneohe Bay, the least nutrient loaded of the four estuaries, from mesotrophic to oligotrophic (Smith et al. 1981). Among the four systems, a positive relationship existed between the sizes of the pelagic production decreases, and the sizes of especially the N input declines.

Phytoplankton biomass was lowered in nine of the 12 systems (Fig. 7). The decreases ranged in size from $< 5 \mu\text{g chl-}a \text{ m}^{-2}$ for Kaneohe Bay and Roskilde Fjord, to $> 80 \mu\text{g chl-}a \text{ m}^{-2}$ for the much more heavily nutrient-loaded Hillsborough and Lajaalahti bays. The three systems in which biomass averaged over the entire system did not decline included the Baltic Sea proper (HELCOM 2009), Chesapeake Bay (Harding et al. 2016) and Patuxent River Estuary (Testa et al. 2008). These are all large stratified systems, to which the N decreases were also much smaller than to Boston Harbor or Hillsborough Bay. The increases in N relative to P limitation seen in the harbor also have been observed in other estuaries where wastewater discharges have been reduced (Wood and Bukaveckas 2014). In Kaneohe Bay, like in the harbor, benthic organic C mineralization rates and DIN flux declined (Smith et al. 1981).

Seagrass coverage increased in six of the 12 systems, including Boston Harbor, Chesapeake Bay (Lefcheck et al. 2018), Danish coastal systems (Reimann et al. 2016), Hillsborough Bay and Tampa Bay (Greening et al. 2014), and Roskilde Fjord (Staehr et al. 2017). In both systems for which pre- and post-pelagic and pre- and post-seagrass production data were available, the increased seagrass production was small relative to the decrease in pelagic production. In Hillsborough Bay, the increase in seagrass production from > 5 to $50 \text{ gC m}^{-2} \text{ year}^{-1}$, was equivalent to 13% of its $350 \text{ gC m}^{-2} \text{ year}^{-1}$ pelagic decline. In Boston Harbor, where eelgrass bed expansion might have been dampened by a regional eelgrass decline (Costello and Kenworthy 2011), the $2.3 \text{ gC m}^{-2} \text{ year}^{-1}$ eelgrass production increase was equivalent to $< 1\%$ of the large pelagic decrease.

Management implications

For effective management of eutrophication, the changes that coastal ecosystems undergo following managed nutrient and organic matter input reductions must be understood. The systems to which inputs have been lowered have been diverse, as have been the sizes and the proportions in which the N and P inputs have been decreased. Despite these differences, in, by far, the majority of the 12 systems we examined, phytoplankton

Fig. 7 Phytoplankton biomass (chl-a) per waterbody surface area (diameter of circles), for Boston Harbor and other coastal systems, shown on the same total N and total P input axes as Fig. 5. In systems labeled blue, biomass was lowered at specific locations, but not for biomass averaged system-wide. System numbering as in Fig. 5. Chl *a* sources include Baltic Sea (HELCOM 2009), Chesapeake Bay (Kemp et al. 2005), Danish coastal systems (HELCOM 2009), Hillsborough Bay (Greening and Janicki 2006), Kaneohe Bay (Smith et al. 1981), Laajalahti Bay (Kauppila et al. 2005), Narragansett Bay (Oviatt et al. 2017), New River Estuary (Mallin et al. 2005), Patuxent River Estuary (Flemer et al. 1970; Testa et al. 2008), Tampa Bay (Johansson 2000; Greening and Janicki 2006), Tokyo Bay (Ramaiah and Furuya 2002), Westerschelde (Kromkamp and Peene 2005)

biomass decreases were observed that are consistent with trophic status changes indicating reversal of eutrophication. The 12 systems included many of the systems in which responses to lowered inputs have been most studied. Lowered inputs caused biomass to decline in most systems, but no correlation existed between the sizes of the biomass declines and the magnitudes of the N and/or P input decreases, suggesting that the responses depended on other factors in addition to input reduction magnitude. Policy development will benefit from improved understanding of the roles these additional factors play.

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