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Managed Nutrient Reduction Impacts on Nutrient Concentrations, Water Clarity, Primary Production, and Hypoxia in a North Temperate Estuary.

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Abstract

Except for the Providence River and side embayments like Greenwich Bay, Narragansett Bay can no longer be considered eutrophic. In summer 2012 managed nitrogen treatment in Narragansett Bay achieved a goal of reducing effluent dissolved

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inorganic nitrogen inputs by over 50%. Narragansett Bay represents a small northeast US estuary that had been heavily loaded with sewage effluent nutrients since the late 1800s. The input reduction was reflected in standing stock nutrients resulting in a statistically significant 60% reduction in concentration. In the Providence River estuary, total nitrogen decreased from 100µm to about 40µm, for example. We tested four environmental changes that might be associated with the nitrogen reduction. System apparent production was significantly decreased by 31% and 45% in the upper and mid Bay. Nutrient reductions resulted in statistically improved water clarity in the mid and upper Bay and in a 34% reduction in summer hypoxia. Nitrogen reduction also reduced the winter spring diatom bloom; winter chlorophyll levels after nutrient reduction have been significantly lower than before the reduction. The impact on the Bay will continue to evolve over the next few years and be a natural experiment for other temperate estuaries that will be experiencing nitrogen reduction. To provide perspective we review factors effecting hypoxia in other estuaries with managed nutrient reduction and conclude that, as in Narragansett Bay, physical factors can be as important as nutrients. On a positive note managed nutrient reduction has mitigated further deterioration in most estuaries.

Keywords: Nitrogen reduction, System Responses, Hypoxia

Introduction

As the expensive engineering efforts reduce nutrients in coastal systems, the documentation toward restoration becomes increasingly relevant to justify. Some systems fail to follow a simple trajectory from degradation to restoration (Duarte et al. 2009, Conley 2012). In large stratified Chesapeake Bay, inorganic nutrients were a primary driver of hypoxia through growth, sinking and decomposition of algal cells, but the removal of bivalve filter feeders, climate change and changing physical factors have continued to contribute to hypoxia (Kemp et al. 2009). However, other systems indicate improvement with reduced chlorophyll levels, improved water clarity and in some cases the re-growth of seagrass beds (Staehr et al. 2017, Taylor et al. 2011, Greening and Janicki 2006). Here, we offer one more case where the effort may be evaluated.

In the late 1990s and early 2000s, studies in Narragansett Bay revealed incidences of low oxygen concentrations during the summer in the Providence River area, the upper Bay and coves such as Greenwich Bay (Bergondo 2004, Bergondo et al. 2005, Deacutis et al. 2006, Melrose et al. 2007, Codiga et al. 2009). The Providence River and Seekonk River estuaries at the head of the Bay have low oxygen reports dating back to 1923 (Desbonnet and Lee 1991, US Public Health Service 1960). Estimates of prehistoric nutrient concentration suggest that reactive nitrogen and phosphorus had been increased 5 fold and 2 fold, respectively, from human activities by the 1990s (Nixon 1997).

Eutrophication and low oxygen resulting in the 2003 fish kill in the Greenwich Bay portion of Narragansett Bay caught the public's attention and provided impetus for managed nitrogen reduction (Deacutis et al. 2006). The Rhode Island Department of Environmental Management (RI DEM) implemented nitrogen reductions in waste water treatment facilities (WWTF) beginning in 2005 (RIDEM 2005). By summer 2012 the overall goal of a 50% reduction of the WWTF effluent DIN load was achieved in Narragansett Bay (Liberti, A. 2014 RI DEM, personal communication). Management regulations have mandated tertiary treatment at WWTF year round but threshold limits of 5 ppm nitrogen only apply in warmer months when the process is most efficient.

This study presents the change in nutrient standing stocks in Narragansett Bay and four environmental changes associated with oligotrophication: apparent production, water clarity, frequency of summer hypoxia and intensity of the winter-spring diatom bloom. The variables were examined Bay-wide and presented in a north-south format for comparisons to nutrient concentrations and provide an estimation of gradient changes within the Bay ecosystem. This manner of presentation has the advantage of making outlier areas evident. Since our study began after the initial nutrient reduction, two annual surveys (1979-1980 and 1997-1998) conducted before this study were mined for data for the before treatment in this study.

The Study Area – Narragansett Bay

Narragansett Bay, with a length of 45 km and mean depth of 8.3 m, lies in a north-south orientation on the coast of Rhode Island and opens into Rhode Island Sound (Figure 1). The 4660 km^2 watershed extends from Rhode Island into Massachusetts and provides an average freshwater input of 37 $m^3 s^{-1}$ (Kremer and Nixon 1978). The Bay includes several distinct regions: the northern Seekonk - Providence estuaries, upper Bay, and the lower Bay divided by islands into the West and East Passages. The East Passage borders the entrance to the Mt Hope Bay-Sakonnet Passage complex.

Tidal waters account for about 13% of the volume of the Bay. Offshore waters enter from Rhode Island Sound mainly into the deeper East Passage of the Bay. They flow north to the Providence area along the bottom and slowly mixing to the surface. The range of salinity in the Bay is small, about 22 to 30 ppt, owing to the low fresh water input compared to the large tidal volume. Water in the upper Bay and East Passage moves west due to prevailing winds and Coriolis force. The resulting flow then propagates south on the surface in the West Passage (Kincaid et al. 2008). Average residence time of water in the Bay is about 40 days (Pilson 1985).

Historically WWTF located in the watershed and the urban northern portion of the Bay have contributed high nutrient loads to the Bay resulting in a gradient of high nutrient concentrations in the upper Bay to low nutrients in Rhode Island Sound (Oviatt 2008). Primary production, chlorophyll-a, zooplankton abundances and frequency of hypoxia events have followed a similar gradient of higher values up Bay and decreasing values down Bay (Oviatt 2008).

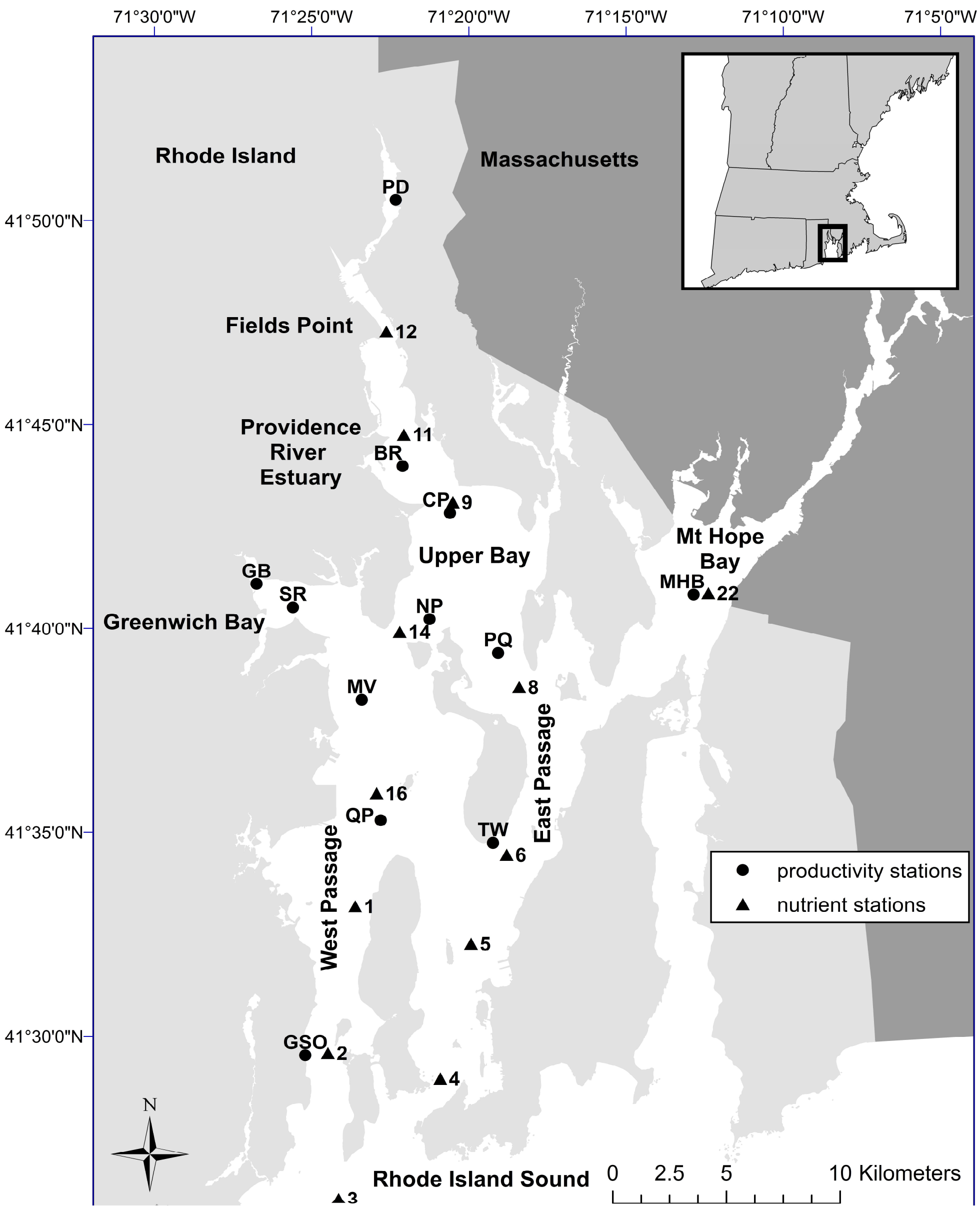

Figure 1. Narragansett Bay station locations are numbered nutrient sampling stations and labeled fixed and buoy monitoring sites: PD is Phillipsdale, BR is Bullocks Reach; CP is Conimicut Point; NP is North Prudence; GB is Greenwich Bay; SR is Sally Rock; PQ is Poppasquash; MHB is Mount Hope Bay; MV is Mount View; T-W is T-Wharf; QP is Quonset Point; GSO is Graduate School of Oceanography Dock.

Methods

Nutrients

From late summer 2005 through 2014, 13 surface stations around Narragansett Bay (Figure 1) were sampled to measure ammonia, nitrate, nitrite, phosphate, total phosphorus and total nitrogen. Sampling was conducted monthly during cold months and bi-monthly during June, July, August. These surveys were compared to a dissolved inorganic nutrient survey conducted in 1979-80 and a total nitrogen and total phosphorus survey in 1997-1998 over annual cycles (Oviatt et al. 1984, Oviatt et al. 2002).

Samples were collected using an acid cleaned plastic container within 0.5 m of the surface and stored on ice until return to the laboratory. Upon return during the afternoon of the same day 40 ml dissolved inorganic nutrients were filtered using 0.45-micron Nucleopore filters. A second aliquot of 40 ml whole water was collected for TN and TP. All samples were frozen at -4 ^{o}C until analysis.

Samples were analyzed with a colorimetric method on a Technicon auto-analyzer before 2008 and an Astoria SFA auto-analyzer after 2008. The two instruments were inter-calibrated with samples over a three-year period (Krumholz 2012). Total nutrient samples were mineralized with alkaline persulfate before being analyzed (Patton and Kryskalla 2003). Ammonia, nitrate and nitrite were analyzed using methods from Astoria Pacific (2005), Scott et al. (2005) and Schmidt and Clement (2009). Phosphate was analyzed using methods from Scott et al. (2005).

Field Monitoring Network for Oxygen and Apparent System Production

Since the early 2000s, a summer monitoring network of fixed and buoy sites was developed to observe and record water quality variables at several station throughout the Bay (Figure 1). The Rhode Island Department of Environmental Management (RI DEM) developed the program in partnership with the Narragansett Bay Commission, the Narragansett Bay National Estuarine Research Reserve and the University of Rhode Island, Graduate School of Oceanography. The RI DEM has used the data from this network to assess levels of low oxygen conditions for the upper Bay.

For this study the network data were used to estimate primary production. System apparent production is defined as water column net community production during the day (Oczkowski et al. 2016). Surface and bottom sensors were deployed at each site, with the exception of GSO and CP winter stations that had only surface sensors deployed. Sensors took measurements every 15 minutes for depth, temperature, salinity, oxygen, chlorophyll-a fluorescence (surface only), and pH using Yellow Springs Incorporated (YSI) brand instruments at all stations. Stations are serviced on a 2-week interval to remove bio-fouling (for water quality procedures see RI DEM 2014). The buoy stations are operational from May through October (PD, BR, CP, SR, MV, QP, PQ, MHB). Four of the stations operated year round (CP, GB, T-W, GSO). Daily averages of the data are stored and available online: http://www.narrbay.org/d_projects/buoy/buoydata.htm (accessed May 2015).

Model for System Apparent Production

Net apparent system metabolism estimates were calculated from oxygen data (RIDEM 2007, 2014) from each station in the network from 2006 to 2015 using a Dawn-Dusk (DD) estimation (Odum and Hoskin 1958). System apparent production was estimated from surface sensors using dawn and dusk measurements daily June through August (92 d):

$$PP = [(O_{2(dusk)} - O_{2(dawn)}) - D],$$

where PP is apparent daytime production, $(O_{2(dusk)})$ is oxygen concentration at sunset and $(O_{2(dawn})$ *is* oxygen at sunrise, and D is the air-sea gas exchange correction coefficient. *D* is estimated from an empirical exchange coefficient (*K*) using wind speed at 10 m above the water surface (Kremer et al. 2003):

$D = K^*$ *Saturation deficit* O_2, where $K = 0.55e^{(015*U10)}$; U_{10} = wind speed at 10 m.

Wind data were obtained from the National Oceanic and Atmospheric Administration (NOAA) buoys in Narragansett Bay at QP and CP and averaged together for the day estimate of diffusion (Figure 1). Diffusion corrected metabolism data were averaged from June through August period for each summer period (92 days).

Hypoxic Event Estimation

The monitoring network was also used to estimate hypoxia duration for the agency and this study. Low oxygen events were defined as the number of days from June 1- August 31 that DO levels did not meet RI state water quality criteria for dissolved oxygen for salt waters (RIDOCS) below the pycnocline. The criterion was 2.9 mg O_2 l^{-1} for a 24 h period or 1.4 mg l^{-1} for 1 hour and designed to protect 95 % of larval recruitment from the cumulative effects of exposure to low dissolved oxygen (RI DEM

2009). Monitoring data were summed to determine the total seasonal exposure. All buoy data have gaps that may create a temporal bias in the data.

Chlorophyll Estimation

Chlorophyll–a fluorescence readings were taken every 15 minutes at all surface monitoring locations using YSI 6 –Series sondes (RIDEM 2007, RIDEM 2014). Water samples were collected for chlorophyll-a analysis every two weeks during sonde swaps. Water was filtered using a 25 mm Whatman GF/F filters for chlorophyll extraction and frozen until analysis to verify/adjust sonde readings (Yentsch and Menzel 1963, Lorenzen 1966, Oviatt and Hindle 1994). Stations from previous surveys and operating year round were used to assess the winter-spring bloom chlorophyll at the CP, GB, TW and GSO stations before, during and after nutrient reduction.

Precipitation Data

Rainfall data were obtained from the Kingston, RI weather station (Kingston Weather 2015). Daily precipitation data for June, July and August were summed for summer totals.

PAR Attenuation Measurements

PAR attenuation was measured bi-weekly to monthly during summer from 2007 through 2009, 2013 and 2014. In addition PAR values were available from an earlier survey in 1997 for comparison. A Li Cor Light meter Data Logger (LI1000 or LI1400) with deck (Quantum LI-190) and depth (Spherical LI-193) sensors (factory calibrated annually) was used to measure the decrease in light at one-meter intervals throughout the water column. The extinction coefficient (k) was estimated from logarithm transformed light data through the water column using Beers' Law.

Statistical Analyses

Analysis of covariance ANCOVA was conducted using SAS PROC GLM (SAS v. 9.3, SAS Institute, Inc., Cary, North Carolina) to examine variation relative to time period, station location and an interaction term (Sokal and Rohlf 1981). The null hypothesis tested was that no difference in nutrient concentration, water column metabolism, chlorophyll concentration or days of hypoxia occurred between periods ($P > 0.05$ or 0.0001). The purpose of the ANCOVA is to compare two or more linear regression lines. Coefficients of determination (R^2) and *P*-values were reported in linear regression analyses to indicate the model fits and tests of significances for changes with time and distance. Station location was incorporated in the model as a covariate as distance and as a proxy for nutrient concentration. Either the slopes are different, and the results are considered different based on a significant *P*-value for the interaction term, or if the slopes are not different, the next step in an ANCOVA is to draw a regression line through each group of points, all with the same slope. This common slope is a weighted average of the slopes of the different groups and the final test in the ANCOVA is to test the null hypothesis that all of the Y-intercepts of the regression lines with a common slope are the same. Because the lines are parallel, saying that they are significantly different at one point (the Y-intercept) means that the lines are different at any point. The main factor in the ANCOVA was year or period, which had various levels within the study (one year: 1979-80 (inorganic nutrients-before), one year: 1998 (total nutrients-before), 5 years continuous data: 2006-2010 (inorganic and total nutrients-during) and after years of continuous data (inorganic and total nutrients-after). Nutrients, system apparent production and chlorophyll were transformed to natural logarithm before analysis as appropriate.

Analysis of variance (ANOVA) was used to test for significant differences in mean winter-spring chlorophyll levels for periods before, during and after nitrogen reduction. The null hypothesis was no difference in environmental conditions between periods. We used ANOVA based on daily values and thus high degrees of freedom rather than ANCOVA based on means because there were only 3 to 4 stations for each treatment. *F* levels, *P* values and coefficients of determination, R^2 were estimated.

Results

Nutrient Concentrations

Yearly concentrations of nitrogen and phosphorus were compared in surveys before, during and after nutrient reduction along a north-south distance gradient in Narragansett Bay (Table 1, Figure 1). An ANCOVA test showed that stations for DIN and DIP from earlier surveys were significantly higher than the survey for the 2014 (Table1). Stations in the upper Bay decreased over the nutrient reduction period more than stations in the lower Bay for most nutrient species. Nitrite/nitrate showed the opposite pattern as ammonia oxidized with distance down Bay. During early surveys, before implementation of tertiary treatment, ammonia was higher than nitrate/nitrite concentration (Figure 2). TN and TP decreases were also statistically significant between 1997-1998 and 2014 (Figure 2, Table 1). All data for all surveys are not shown in Figure 2 to simplify the figure. Including more data does not change conclusions. Most of the phosphorus decrease occurred in the early years (TP -56%) rather than in recent years

(TP -11%) (Table 1). In general, higher nutrient concentrations occurred in the Providence River Estuary, which received inputs from rivers and large WWTF, and lower concentrations were found towards the mouth of Narragansett Bay (Figure 2). With lower nutrient concentrations, a reduced north–south nitrogen gradient in the Bay persisted (Figure 2). Bay wide concentrations of forms of nutrients decreased on the order of 50 to 60% (Figure 2f).

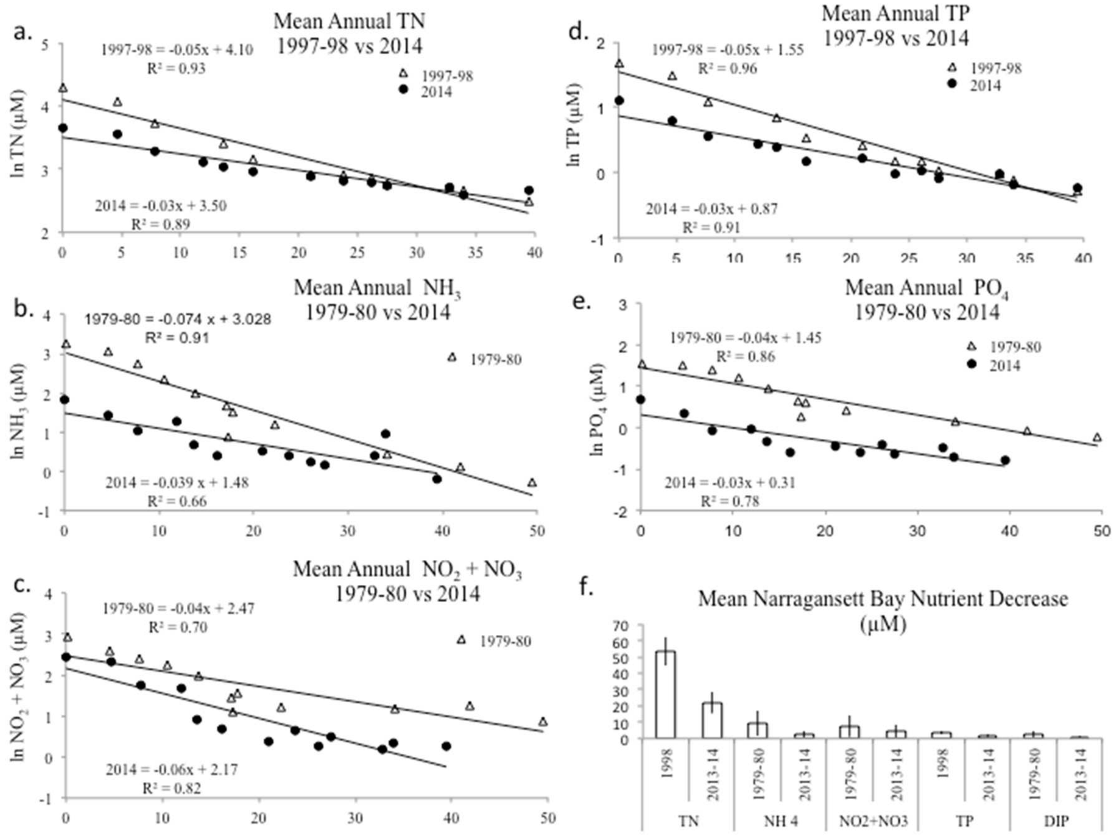

Figure 2. ANCOVA regressions using the natural logarithm of yearly nutrient concentration along the north-south distance axis of Narragansett Bay before and after nutrient reduction. a –e, Mean surface concentrations of nutrients (*TN, NH_4, $NO_{2,3}$, TP,* and *DIP*) before and after nutrient reduction survey periods plotted as kilometers from

Fields Point in the Providence River to the mouth of the bay; f, mean concentration of nutrients for all stations before and after nutrient reduction.

Table 1. Nutrient reduction *P* value tests of significance for all stations comparing periods before to after and during to after. ANCOVA tests were performed on regressions of the natural logarithm of nutrient concentration with north to south distance during the successive surveys before, during and after managed nutrient reduction (Figure 2).

Dissolved Inorganic Nutrient	1979-80 vs 2014	2006-10 vs 2014
DIN	*<0.0001*	*0.05*
NH_3	*0.005*	*0.003*
$NO_2 + NO_3$	*<0.0001*	*0.003*
PO_4	*<0.0001*	*0.005*
Total Nutrient	**1997-98 vs 2014**	**2006-10 vs 2014**
TN	*0.0004*	*0.02*
TP	*0.0004*	*0.03*

Notes:
NS - not significant
P-value shown for significantly different sampling periods.
If significant 2014 typically < other years but underlined cell indicate 2014 > than other years.

Chlorophyll and Water Clarity Response

As summer chlorophyll decreased, extinction coefficients decreased and water clarity increased (Figures 3, 4). Before nutrient reduction mean summer chlorophyll values were 58 $\mu g\ l^{-1}$ in the Providence River and 6 $\mu g\ l^{-1}$ near the mouth of the Bay (Figure 3). After nutrient reduction summer chlorophyll values ranged from 22 $\mu g\ l^{-1}$ in the Providence River to 5 $\mu g\ l^{-1}$ near the mouth. ANCOVA tests between before and after nitrogen reduction were statistically significant ($P<0.0001$ level). The mean values for during and after nitrogen reduction were not significantly difference (Figure 3). An increase change in water clarity was statistically significant before to after nutrient reduction (0.05 level). Measurement of light extinction coefficients after nitrogen reduction showed similar water clarity Bay wide in 2014 (Figure 4). Lower Bay values did not change over the periods.

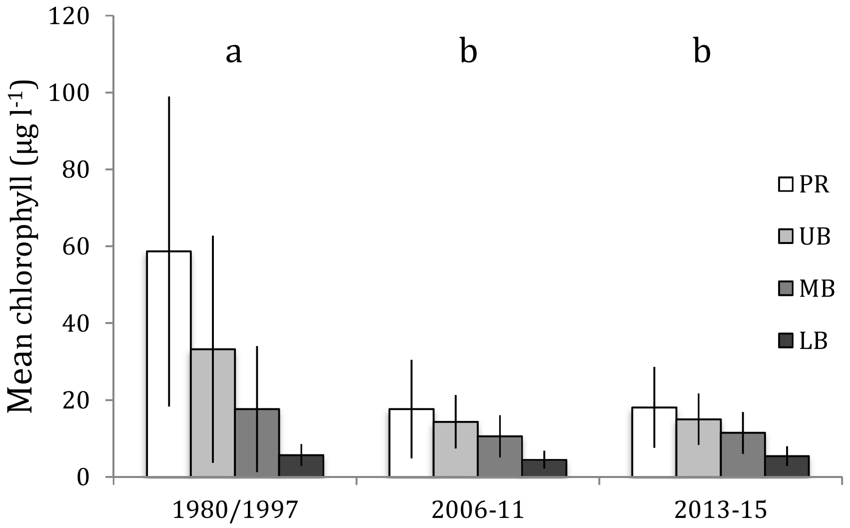

Figure 3. Summer surface chlorophyll before (mean 1980 and 1997), during and after nutrient reduction along the north south axis of the Bay (Providence River (PD, BR), Upper Bay (CP, MHB, NP), mid Bay (MV, PQ), Lower Bay (QP, TW, GSO) ± standard deviation. Letters refers to ANCOVA tests of significance before, during and after nitrogen reduction at 0.0001 level.

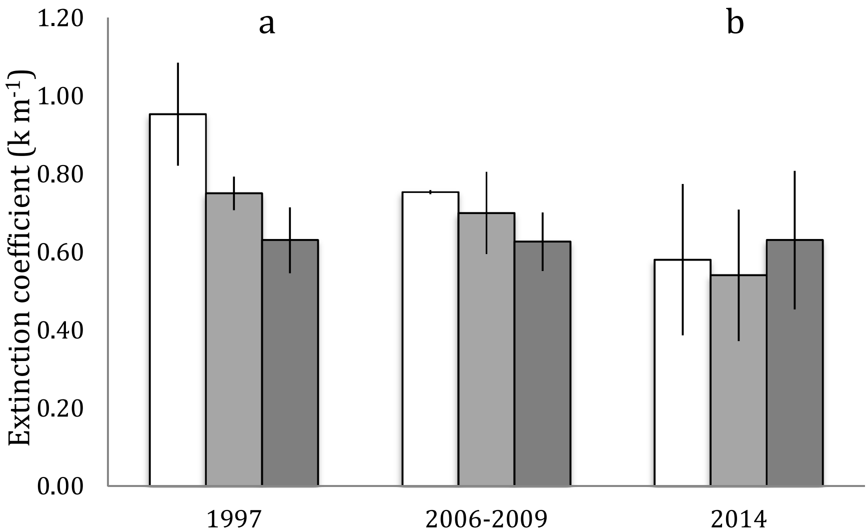

Figure 4. Water clarity before, during and after nutrient reduction in the Providence River, upper Bay and mid Bay $\pm$ standard deviation. Letters a and b indicate ANCOVA significant difference at the *0.05* level between before and after but not during. Region code is white Providence River Estuary, light grey Upper Bay and dark grey Mid Bay.

Surface Production Response

Summer apparent production Bay-wide decreased by 33% after nutrient reduction (Figure 5, Table 2). The largest reductions occurred in the upper Bay area between North Prudence, Mt View (MV) and Poppasquash (PQ) rather than in areas of lower or higher production. Surface summer production in 2004, before nitrogen reduction, ranged from over 2 g O_2 m^{-3} d^{-1} in the Providence River estuary to 1.0 g O_2 m^{-3} d^{-1} in the upper Bay and decreased to 0.6 g at the mouth of Narragansett Bay. After nitrogen reduction in 2013 to 2015 production was less than 0.7 g O_2 m^{-3} d^{-1} in the upper Bay (Figure 5). Lower Bay values were slightly lower compared to upper Bay values at 0.5 g O_2 m^{-3} d^{-1} after nutrient reduction. Areas with additional sources of nutrients such as Greenwich Bay do not fit the general pattern of reduced production, remaining higher than the Providence Rives estuary, the upper Bay and Mt Hope Bay values (Figure 5, Table 3).

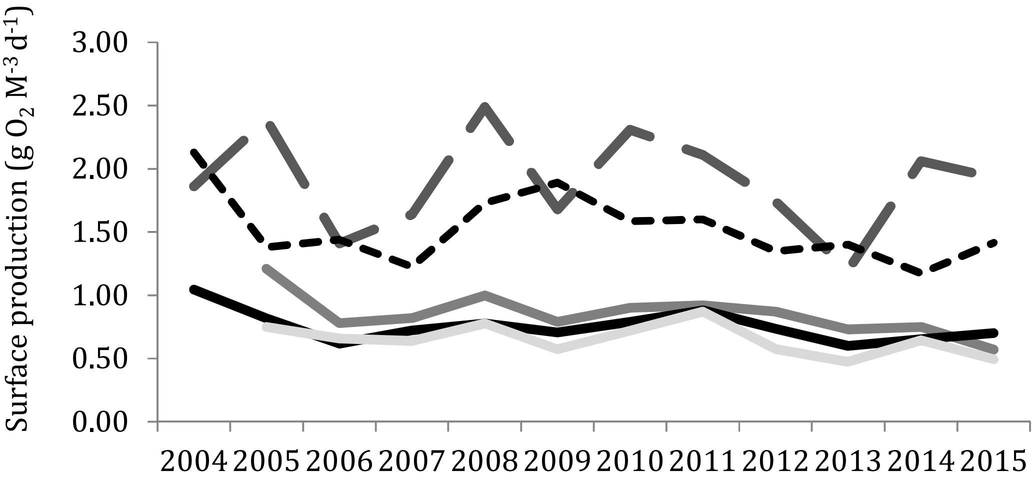

Figure 5. Summer mean system apparent production in Greenwich Bay (long dash), Providence River estuary (BR, CP, short dash), Mt Hope Bay (dark grey), the upper to mid Bay (NP, PQ, MV, black) and the lower Bay (TW, QP, GSO, light grey) from 2004 before to after nutrient reduction in 2013.

Table 2. Percent reduction in surface system apparent production along the north-south stations in Narragansett Bay over three time intervals: from before to 30% nitrogen reduction, from 30% to 50% nitrogen reduction and from before to 50% nitrogen reduction. Regressions of production with distance for each time period comparison were tested for significant differences by ANCOVA.

Station	Prod % Difference 2004 to 2006-2011	Prod % Difference 2006-2011 to 2013-2015	Prod % Difference 2004 to 2013-2015
BR	-13	-21	-31
CP		-15	

MHB		-15	
NP	-29	-22	-44
GB	4	-16	-12
SR		-22	
MV	-28	-24	-45
PQ	-29	-3	-31
QP		-18	
T-W		-29	
GSO		2	
Average	-21	-17	-33
Statistical Significance:	**	N.S.	**

** = *0.01* level of significance
N.S. - not significant at the *0.05* level of significance

Summer Hypoxia Response

Summer hypoxia in all stations decreased on average by 34% after the 60% nitrogen reduction (Figure 6). An ANCOVA intercept test indicated the decreases in summer hypoxia were statistically significant for open Bay stations ($p<0.001$). MHB, SR and GB were eliminated as outliers for this test where distance down Bay was a proxy for nutrient concentration. A separate t-test of decreased mean values during and after nutrient reduction for GB was significant at the 0.05 level. Both Sally Rock and MHB showed increases in hypoxia after nutrient reduction (Figure 6).

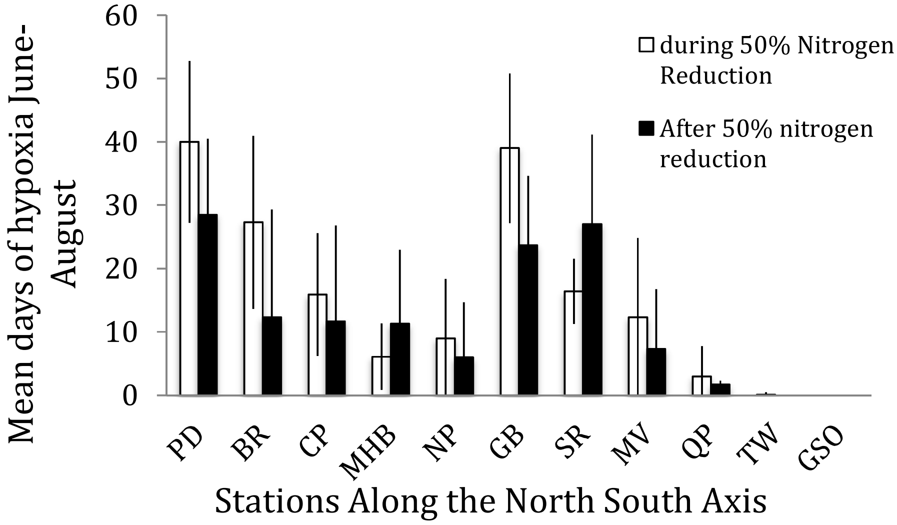

Figure 6. Mean number of summer hypoxic days $\pm$ standard deviation during (2006-2012) and after (2013-2015) the 50% nitrogen reduction at stations along the north to south axis of the Bay. Sally Rock (SR) data was only available after 2008 and before 2015.

Hypoxia was enhanced under stratified conditions in 2013 after nitrogen reduction (Figure 7). A regression analysis of summer precipitation versus summer days of hypoxia was significant (0.01 level) although the R^2 is lower than

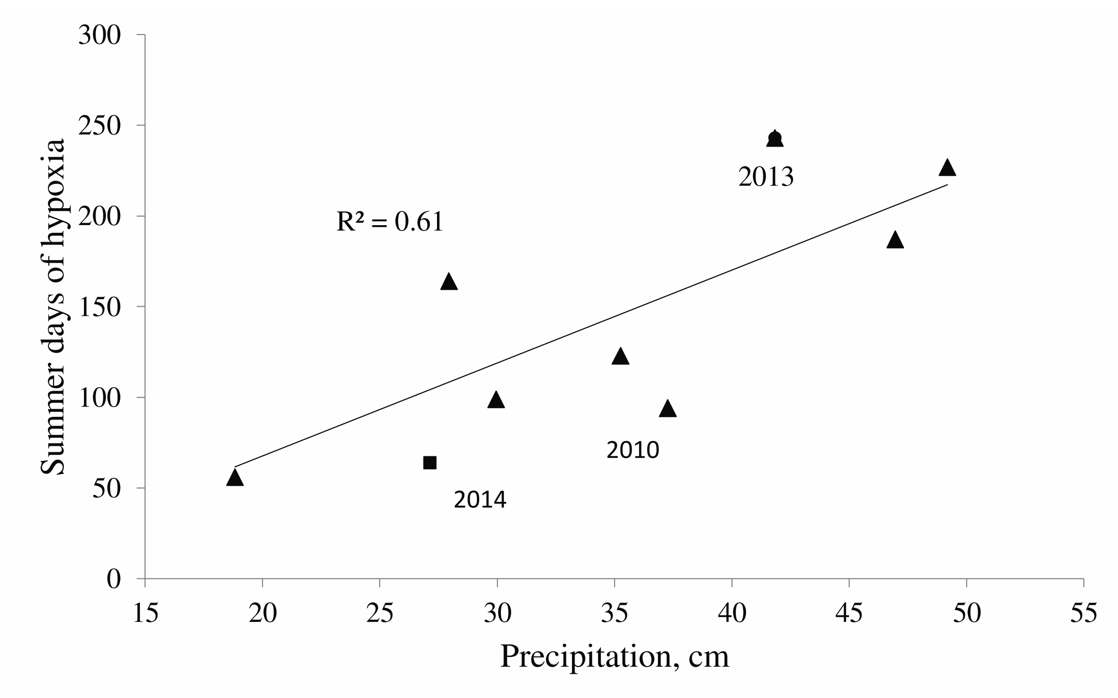

Figure 7. Hypoxia as a function of rainfall. Summer precipitation versus days of summer hypoxia (all sites summed) in June, July and August from 2006 to 2014. The flood year 2010, and 2013 and 2014, after 50% nitrogen reduction are separately labeled.

the Prairie criteria for predictive value (Figure 7, Prairie 1996). While the relationship may have low predictive capability at $R^2 < 0.65$, it is higher than the $R^2=0.6$ needed to distinguish between a low and high value for the dependent variable. The data show that after the 50% nitrogen reduction, the wet summer 2013 was at the upper end of the hypoxia record with 42 cm of rain and 242 days of hypoxia for all stations compared to 2014 at the lower end of the relationship with 25 cm of summer precipitation and 50 days of hypoxia (Figure 7). At the lower extreme, summer precipitation of ~20 cm resulted in ~50 days of hypoxia in 2007 the dry summer for the record.

The winter-spring bloom and other spring blooms do not apparently contribute to summer hypoxia. The average 40-day residence time of the Bay (Pilson 1985) prevented an April flood in the spring of 2010 (100 year flood) from increasing hypoxia during the 2010 summer (Figure 7). High concentrations of nutrients flowed into the system from extreme runoff and the submergence of a WWTF, resulting in a large phytoplankton bloom but not summer hypoxia (Figure 7).

Winter-Spring Diatom Bloom

The winter-spring diatom bloom has been reduced significantly in Narragansett Bay during and after nitrogen reduction compared to historical levels (Figure 8). In Greenwich Bay (GB) and other coves where nitrogen reduction has not yet been successful, blooms still reach historical proportions and durations (Figure 8). The bloom has become variable or non-existent in recent warm winters like 1998 (Oviatt et al 2002). However, with the return of some cold winters in the 2000s, the bay has bloomed, but not to levels prior to nutrient reduction or currently observed in Greenwich Bay. The upper Bay at Conimicut Point (CP) has significantly reduced chlorophyll values compared to before nitrogen reduction. Currently CP has higher winter chlorophyll values than down Bay locations at T-Wharf (TW) and the Graduate School of Oceanography dock (GSO) (Figure 8). T-Wharf had significantly greater blooms before nitrogen reduction (Figure 8). The GSO dock had significantly greater blooms before and during nitrogen reduction than after (Figure 8).

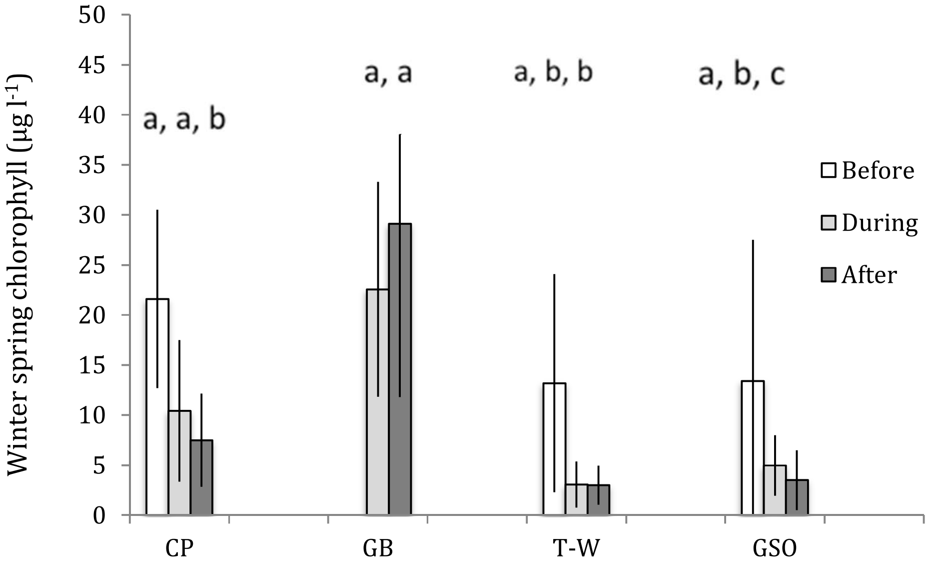

Figure 8. Impact of nitrogen reduction on the winter-spring bloom in Narragansett Bay. Winter-spring mean chlorophyll during January, February and March from before nitrogen reduction (1980) (no data for GB) during at CP (2009-2013) GB (2007-2010 & 2012-2013), T-W (2011-2013), GSO (2005-2013) and after at all stations (2014-2015). The mean values include + standard deviations of the means. The letters indicate ANOVA tests of significance on the daily values from January through March; different letters indicate significance levels greater than 0.0001; similar letters indicate no significant difference between time periods.

Discussion

Managed Nitrogen Reduction

The final nutrient reduction for large WWTF in the Providence River occurred in summer 2012 achieving about 60% reduction in nitrogen compared to the 1990s. The managed nitrogen reductions began in 2005 and achieved over a 30% decrease of direct wastewater input to the Bay by 2006-10 (Krumholz 2012). The effort of reducing nitrogen inputs from sewage treatment plants in Rhode Island were helped by nitrogen and phosphorus WWTF reductions being implemented in the entire Narragansett Bay watershed extending into Massachusetts (Liberti, A. 2014 RI DEM, personal communication). Thus the reductions also occurred in all the rivers entering Narragansett Bay affecting a greater reduction than initially hoped. Final tertiary treatment was implemented at large WWTF in the Providence River area in summer 2012 with TN, DIN reductions achieving 58%, 60% and TP, DIP reductions achieving 54%, 61% Bay wide. During this study dissolved organic nitrogen (DON) and phosphorus (DOP) were not measured so we cannot directly indicate how much these forms may have declined. Based on bay-wide surveys conducted four times over 1985 to 1986 we can provide an annual estimate of the percentage of DON and DOP as 45% and 47% respectively, of TN and TP (Pilson and Hunt 1988).

Our use of annual survey data in 1979-1980 and 1997-1998, before managed nitrogen reduction in 2005, should attribute the decrease in phosphorus as happening before managed nitrogen reduction. Nutrient budgets compiled in the mid-80s and 90s and updated in 2003 recognized the large reduction in phosphate and total phosphorus due to regulation of detergents by the 1990s (Nixon et al. 2008). Before 2005 TN discharged to the Bay had apparently not changed since the 1980s and although ammonia has decreased and nitrate has increased, DIN had also not changed since the 1980s (Nixon et al. 2008).

Sewage based inputs to the Bay represent the largest sources of nitrogen in the Bay budget. Direct sewage nitrogen inputs have been reduced by about 85 $x10^6$ moles per year (updated from Krumholz 2012). This sewage source can be compared to atmospheric deposition, which was about 30 x10^6 moles per year in the nutrient budget as of 2003 (Nixon et al. 2008). Recent estimates suggest atmospheric input has decreased on the order of 25% or 7.5 x 10^6 moles per year (Latimer and Charpentier 2010) making this source a minor input of nitrogen to the Bay. Sewage and river inputs (primarily sewage) total to about 182 x 10^6 moles N per year (updated from Krumholz 2012).

In the Providence River estuary light, and in the lower Bay nutrients, have limited primary production. In consequence the largest percentage reductions in summer apparent production occurred in the upper and mid Bay region (Table 2). Values in this large region have converged with lower Bay values (Figure 5). This region has also experienced decreased summer chlorophyll and improved water clarity. These changes should improve habitat for fauna but perhaps, not flora like eelgrass. Local eelgrass strains have failed to thrive in increasingly warm Bay waters (Bintz et al. 2003).

The gradient presentation reveals current problem areas (Figure 6). Two stations Sally Rock (SR) in Greenwich Bay and Mt Hope Bay (MHB) had increases in hypoxia after 50% nitrogen reduction. Managed nitrogen reduction is incomplete in both areas. Both have additional sources of nutrients from ground water, brook or river inputs and WWTF which do not originate from the Providence River area. Greenwich Bay has an additional problem of long residence time during southwest sea breeze periods (Rogers 2008). All these factors enhance the frequency of hypoxia and are not related to the open Bay gradient of nutrients.

Controlling Hypoxia in Estuaries

Nutrients and Light

In many estuaries including Narragansett Bay, nutrients have increased several fold since the pre-industrial era and may no longer be strongly limiting to primary production. The relationship between nutrients and primary production is a hyperbolic tangent where at high nutrients productivity continues to increase a small amount for large increases in nutrients (Oviatt et al. 1986). The water column productivity range in estuaries is constrained to a low range of usually less than 700 g C m^{-2} y^{-1}, compared to the terrestrial environment, due to self shading (Oviatt 1994). Thus large reductions in nitrogen, usually the limiting nutrient in marine systems, will be needed to reach the exponential portion of the curve where small changes in nitrogen cause large reductions in primary production. For example, in the North Carolina New River DIP was reduced by 71% and DIN was reduced by 57% from WWTF sources resulting in a decrease of 69% in chlorophyll-a; however, oxygen concentrations in bottom water have generally failed to improve (Mallin et al. 2005). Greater decreases in nutrients from the watershed livestock production facilities would be required to improve hypoxia in the estuary.

In some estuaries where nutrients alone control environmental conditions, regime changes can be accomplished by nutrient reduction. In Tampa Bay, Florida beneficial improvements occurred when total nitrogen was reduced by 60%; chlorophyll-a concentrations declined by 50% improving water clarity and over several years, seagrass beds increased by 25% (Greening and Janicki 2006). In this case a turbid phytoplankton system of the 1980s shifted back with nutrient reduction to a clear, seagrass system characteristic of the 1950s (Greening et al. 1914). A long-term study of nutrient reduction in a shallow Danish estuary provides a trajectory of improvement over time (Staehr et al. 2017). Nutrient reductions resulted in initial fast improvements followed by slower changes over more than three decades. Hypoxia decreased, chlorophyll decreased by 50% in the inner estuary and water clarity improved (Staehr et al. 2017). Unlike Narragansett Bay, the winter-spring bloom increased in the Danish estuary probably due to increased water clarity. Over time the Danish system became more autotrophic and less heterotrophic.

Not all estuaries will exhibit a relationship between nitrogen and primary productivity if they are highly turbid, strongly mixed and exchanged or if they have large organic inputs from the watershed agriculture, livestock production, WWTFs or other sources (Cloern 2001, Kemp et al. 2009). Estuaries with high particulates and/or tannins may have low productivity but still sufficient organic carbon inputs to become hypoxic. Light limited low productivity will limit the oxygen input to the water column. Water contains only a small amount of oxygen compared to air. During summer oxygen saturation may be only 5 to 7 g O_2 m^{-3}. In a eutrophic estuary with primary production or organic inputs of greater than 0.5 g C m^{-2} d^{-1} only a few days of sinking organic matter will be needed to utilize all the oxygen available where residence times are on the order of weeks or months. Some highly enriched estuaries with high tidal exchange and water column mixing do not experience hypoxia. In Boston Harbor with nutrient and organic carbon reduction, a more aerated later successional stage benthos resulted (Diaz et al. 2008). With sewage treatment upgrades and re-location of the outfall 8 km offshore into Massachusetts Bay, organic carbon inputs declined by 90% and DIN declined by 56%; after two to three years chlorophyll concentrations declined by 29% and oxygen in bottom waters increased by 12% (Taylor et al. 2011). Over six years of monitoring after waste-water reduction, the benthic habitat in the harbor became a more oxygenated environment colonized by deeper burrowing fauna (Diaz et al. 2008). In other highly enriched river estuaries rate of fresh water flow dictates eutrophic conditions. When fresh water discharged large volumes in the 1970s, the Hudson River Estuary, where nitrogen loads have increased 12 fold and phosphorus loads have increased 50 fold over the pre-industrial era, experienced low rates of primary production; during low flows of the 1990s the estuary became hypereutrophic (Howarth et al. 2000)

Stratification and Residence Time

Large river flows and/or high summer precipitation stratifies estuarine waters with surface fresh water preventing re-aeration and causing hypoxia due to higher summer respiration of organic matter (Diaz and Breitburg 2009). In some systems like Narragansett Bay, hypoxia is ephemeral lasting hours, days or weeks; in others it is seasonal lasting months. During a rainy summer, river flows of fresh waters cap the bottom waters in the Providence River estuary and the upper portions of Narragansett Bay leading to stronger degrees of stratification. The decrease in exposure to the atmosphere and decrease in vertical mixing results in a higher frequency, duration and intensity of hypoxia (Figure 7). In the Susquehanna River spring floods cause seasonal stratification and hypoxia/anoxia in the bottom waters of Chesapeake Bay (Wang et al. 2015, Kemp et al. 2009). A large complex and variable riverine system with strong stratification and long residence time on the order of months have stymied efforts of controlling low oxygen by nutrient reduction in main stem Chesapeake (Boesch and Goldman 2009). The main stem has also continued to be a detrimental source of nutrients to some tributaries like the Patuxent. In the Patuxent River TP was reduced by 54% and TN by 55% at WWTFs but chlorophyll failed to decline and hypoxia was not reduced (Testa et al. 2008). In the Neuse River and Pamlico Sound estuary long residence times, stratification and warmer temperatures exacerbate hypoxic conditions even with nitrogen reduction (Pearl 2006).

Biota

In some eutrophic systems a large filtering benthos may control a high primary production system to low biomass of phytoplankton and result in aerobic conditions. San Francisco Bay with a high biomass of clam filter feeders had a nitrogen input greater than Chesapeake Bay but a chlorophyll concentration of only 2.7 mg m^{-3} compared to the Chesapeake Bay value of 13.6 mg chl m^{-3} and no hypoxia (Cloern 2001). Chesapeake Bay researchers have long proposed that in the past, the vast biomass of oysters reduced phytoplankton biomass and prevented hypoxia (Newell 1988, Kemp et al. 2009). Pomeroy et al. (2006) refuted that idea with calculations apparently showing that the oysters could not have reduced the spring bloom biomass to the point that eliminated hypoxia and anoxia. This controversy continues (Pomeroy 2007, Newell et al. 2007) with oysters being restored to some 60 sites in the Bay tributaries (Boesch and Goldman 2009).

A biota change under the rubric of unintended consequences has to do with the winter-spring diatom bloom in Narragansett Bay. In northern temperate estuaries the winter-spring bloom has historically been the signature ecological event of the year (Pratt 1965). The winter spring diatom bloom has been renowned for duration and intensity during cold winters of the past. During warm winters (positive North Atlantic Oscillation) from the late 1970s to early 2000's zooplankton grazing appeared to control the expression of the bloom and the bloom often failed to occur during warm winters

(Oviatt et al. 2002, Keller et al. 1999). The lack of winter-spring blooms during warm winters caused a significant reduction in measures of benthic metabolism and nutrient flux compared to historically higher values during periods of strong blooms (Nixon et al. 2008, Fulweiler et al. 2008, Fulweiler et al. 2009). During and after the nutrient reduction period the winter-spring bloom occurred during cold winters (negative NAO) maintaining to a significantly lesser extent the north to south gradient in chlorophyll (Figure 8). Thus the larger winter spring diatom bloom of the past relied not only on cold winter temperatures but also sewage effluent nitrogen. Likely the continually warming climate will eliminate cold winters and winter diatom blooms in the near future.

Climate

In some systems climate trends have overwhelmed efforts of estuarine restoration and nutrient reduction. In Chesapeake Bay since the 1990s decrease in nutrients, hypoxic volume of water per unit nitrogen input has doubled (Kemp et al. 2009). A numerical simulation model has demonstrated that wind-driven lateral circulation and enhanced vertical mixing in shoal regions is a dominant mechanism for providing oxygen to deeper waters and thus wind direction may explain much of the variability in hypoxia in Chesapeake Bay (Scully 2010). A change in wind direction and a decrease in wind speed and mixing with a shift in the North Atlantic Oscillation may be the cause of increased hypoxia (Wang et al. 2015). In Long Island Sound WWTF have decreased nitrogen inputs by about 28% but this change did not decrease in the volume of hypoxic water in the western portion of the Sound (O’Donnell et al. 2014). Prevailing wind direction may have changed over the last several decades, increasing the tendency for stratification and reducing the renewal of oxygen in bottom waters in LIS. In San Francisco Bay the biomass of filter feeders has recently declined by a factor of 20 due to increased predation by shrimp, sole, and crab from offshore and chlorophyll has increased to 10 mg m^{-3} (Cloern et al. 2007). A negative oscillation cold upwelling phase has apparently introduced offshore waters and predators into the Bay and stimulated phytoplankton blooms. In shallow coastal Danish systems hypoxia has increased after nutrient reductions of over 50% due to decreased wind, increased stratification and warmer temperatures (Riemann et al. 2016). In several cases climate oscillation and trends have resulted in decreased wind mixing and changes to circulation exacerbating low oxygen conditions.

Conclusion

The nutrient reduction in Narragansett Bay will continue to evolve over the next few years and serves as a possible outcome for other temperate estuaries that will be experiencing nitrogen reduction. With nutrient reductions of near 60%, surface primary production in the Bay declined from 323 g C m^{-2} y^{-1} before nutrient reduction to about 224 g C m^{-2} y^{-1}, water clarity increased and hypoxia tended to decrease. Much greater nutrient reductions would be needed to decrease hypoxia during rainy summer conditions raising concerns about decreased secondary productivity for aquaculture and estuarine fishes.

Elimination of hypoxia/anoxia by sufficient reduction of nutrient sources will remain difficult for many systems including Narragansett Bay: (1) large nutrient reductions in eutrophic systems decrease primary production only slightly, (2) hypoxia in some systems is driven by diffuse watershed inputs, (3) stratification greatly exacerbates hypoxia, (4) biotic controls on hypoxia may be ephemeral and (5) climate changes in temperature and wind tend to increase hypoxia. Boesch and Goldman (2009) suggest the way forward for Chesapeake Bay will be to focus on individual tributaries with researchers and stakeholders developing a strategy that works for the individual case. Similarly each estuarine restoration will need to take into account the factors most important for the particular site. On the positive side nutrient reduction in some systems has effected a positive regime change to a previous ecosystem state and in most other systems nutrient reduction mitigated deterioration.

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Table 1. Nutrient reduction *P* value tests of significance for all stations comparing periods before to after and during to after. ANCOVA tests were performed on regressions of the natural logarithm of nutrient concentration with north to south distance during the successive surveys before, during and after managed nutrient reduction (Figure 2).

Table 2. Percent reduction in surface system apparent production along the north-south stations in Narragansett Bay over three time intervals: from before to 30% nitrogen reduction, from 30% to 50% nitrogen reduction and from before to 50% nitrogen reduction. Regressions of production with distance for each time period comparison were tested for significant differences by ANCOVA.

Figure 1. Narragansett Bay station locations are numbered nutrient sampling stations and labeled fixed and buoy monitoring sites: PD is Phillipsdale, BR is Bullocks Reach; CP is Conimicut Point; NP is North Prudence; GB is Greenwich Bay; SR is Sally Rock; PQ is Poppasquash; MHB is Mount Hope Bay; MV is Mount View; T-W is T-Wharf; QP is Quonset Point; GSO is Graduate School of Oceanography Dock.

Figure 2. ANCOVA regressions using the natural logarithm of yearly nutrient concentration along the north-south distance axis of Narragansett Bay before and after nutrient reduction. a –e, Mean surface concentrations of nutrients (*TN*, NH_4, $NO_{2,3}$, *TP*, and *DIP*) before and after nutrient reduction survey periods plotted as kilometers from Fields Point in the Providence River to the mouth of the bay; f, mean concentration of nutrients for all stations before and after nutrient reduction.

Figure 3. Summer surface chlorophyll before (mean 1980 and 1997), during and after nutrient reduction along the north south axis of the Bay (Providence River (PD, BR), Upper Bay (CP, MHB, NP), mid Bay (MV, PQ), Lower Bay (QP, TW, GSO) $\pm$ standard deviation. Letters refers to ANCOVA tests of significance before, during and after nitrogen reduction at 0.0001 level.

Figure 4. Water clarity before, during and after nutrient reduction in the Providence River, upper Bay and mid Bay ± standard deviation. Letters a and b indicate ANCOVA significant difference at the *0.05* level between before and after but not during. Region code is white Providence River Estuary, light grey Upper Bay and dark grey Mid Bay.

Figure 5. Summer mean system apparent production in Greenwich Bay (long dash), Providence River estuary (BR, CP, short dash), Mt Hope Bay (dark grey), the upper to mid Bay (NP, PQ, MV, black) and the lower Bay (TW, QP, GSO, light grey) from 2004 before to after nutrient reduction in 2013.

Figure 6. Mean number of summer hypoxic days ± standard deviation during (2006-2012) and after (2013-2015) the 50% nitrogen reduction at stations along the north to south axis of the Bay. Sally Rock (SR) data was only available after 2008 and before 2015.

Figure 7. Hypoxia as a function of rainfall. Summer precipitation versus days of summer hypoxia (all sites summed) in June, July and August from 2006 to 2014. The flood year 2010, and 2013 and 2014, after 50% nitrogen reduction are separately labeled.

Figure 8. Impact of nitrogen reduction on the winter-spring bloom in Narragansett Bay. Winter-spring mean chlorophyll during January, February and March from before nitrogen reduction (1980) (no data for GB) during at CP (2009-2013) GB (2007-2010 & 2012-2013), T-W (2011-2013), GSO (2005-2013) and after at all stations (2014-2015).

The mean values include $\pm$ standard deviations of the means. The letters indicate ANOVA tests of significance on the daily values from January through March; different letters indicate significance levels greater than 0.0001; similar letters indicate no significant difference between time periods.