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Spatial and temporal trends in summertime climate and water quality indicators in the coastal embayments of Buzzards Bay, Massachusetts

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Abstract

Degradation of coastal ecosystems by eutrophication is largely defined by nitrogen loading from land via surface and groundwater flows. However, indicators of water quality are highly variable due to a myriad of other drivers, including temperature and precipitation. To evaluate these drivers, we examined spatial and temporal trends in a 22 year record of summer water quality data from 122 stations in 17 embayments within Buzzards Bay, MA (USA), collected through a citizen science monitoring program managed by Buzzards Bay Coalition. To identify spatial patterns across Buzzards Bay's embayments, we used a principle component and factor analysis and found that rotated factor loadings indicated little correlation between inorganic nutrients and organic matter and chlorophyll *a* (Chl *a*) concentration. Factor scores showed that embayment geomorphology in addition to nutrient loading was a strong driver of water quality, where embayments with surface water inputs showed larger biological impacts than embayments dominated by groundwater influx. A linear regression analysis of annual summertime water quality indicators over time revealed that from 1992 to 2013, most embayments (15 of 17) exhibited an increase in temperature (mean rate of 0.082 ± 0.025 (SD) $^{\circ}\mathrm{C\,yr^{-1}}$) and Chl *a* (mean rate of 0.0171 ± 0.0088 $\log_{10}$ (Chl *a*; $\mathrm{mg\,m^{-3}}$) $\mathrm{yr^{-1}}$, equivalent to a 4.0 % increase per year). However, only 7 embayments exhibited an increase in total nitrogen (TN) concentration (mean rate 0.32 ± 0.47 (SD) $\mathrm{\mu M\,yr^{-1}}$). Average summertime $\log_{10}$ (TN) and $\log_{10}$ (Chl *a*) were correlated with an indication that yield of Chl *a* per unit total nitrogen increased with time suggesting the estuarine response to TN may have changed because of other stressors such as warming, altered precipitation patterns, or changing light levels. These findings affirm that nitrogen loading and physical aspects of embayments are essential in explaining observed ecosystem response. However, climate-related stressors may also need to be considered by managers because increased temperature and precipitation may worsen water quality and partially offset benefits achieved by reducing nitrogen loading.

1 Introduction

Long-term monitoring of coastal water quality is critical to identifying environmental degradation, quantifying the effects of management strategies, and improving our understanding of coastal ecosystem dynamics. Extensive monitoring of estuaries in the United States reveals that many US coastal waters are impacted by eutrophication (Bricker et al., 2007, 2008). In the US Northeast, increases in population density, coastal development, and heavy reliance on residential septic systems have elevated nutrient loading rates (e.g. Valiela et al., 1997; Howarth et al., 2002). Many coastal waters show higher levels of phytoplankton biomass, greater variation in water column dissolved oxygen, loss of submerged aquatic vegetation, and decreases in fish and shellfish abundance associated with eutrophication (Ryther, 1954; Nixon, 1995).

In addition to nutrient enrichment, coastal waters may be influenced by climate change, which has the potential to exacerbate the effects of eutrophication (Rabalais et al., 2009, 2010; Doney et al., 2012). For example, recent studies documenting climate-related changes to coastal waters suggest that increasing precipitation or warming may initiate phytoplankton blooms by increasing the delivery of nutrients (Najjar et al., 2010), changing phytoplankton growth rates (Eppley, 1972), or changing the phenology of regularly occurring phytoplankton blooms, causing a trophic mismatch between primary and secondary producers (Brander, 2010). Warming or increased freshwater inputs to coastal waters can also alter circulation patterns and enhance vertical stratification, which may exacerbate bottom-water hypoxia or anoxia as a result of eutrophication (Testa et al., 2008; Kemp et al., 2009; Rabalais et al., 2010; Murphy et al., 2011; Lennartz et al., 2014).

In this work, we focus on Buzzards Bay, Massachusetts (MA, USA) (41.55 N, 70.80 W), a shallow, elongated estuary ($\sim$ 11 m deep and covering $\sim$ 600 km^2) bordered by southern Massachusetts and Cape Cod. The coastline of Buzzards Bay consists of small, river-fed and groundwater-fed embayments distributed along the mainland (west) and Cape Cod (east) side of the estuary, respectively. Many of the embayments of Buzzards Bay are classified by the Massachusetts Department of Environmental Protection as nutrient impaired pursuant to Federal Clean Water Act Sect. 303d, because they do not meet surface water quality standards due to high levels of nitrogen loading (Sullivan et al., 2013). Many of these embayments also show signs of nutrient-related declines in water quality such as widespread losses in submerged aquatic vegetation (Costello and Kenworthy, 2011).

Century-scale records of climate indicators in the US Northeast suggest changes in the major physical drivers controlling productivity in Buzzards Bay. The 2014 National Climate Assessment (NCA) identified the US Northeast as one of the regions with the largest increases in temperature, precipitation, and high intensity rainfall events (Walsh et al., 2014). Between 1894 and 2011, the NCA reports an average land-based temperature increase of ~ 1.1 °C (Walsh et al., 2014) for the Northeastern US, and long-term records of coastal ocean water temperature from Woods Hole, MA suggest that water temperatures have increased by 1.9 °C since the late 1800s (Nixon et al., 2004). Recent studies have found oceanic warming at rates in the Northwest Atlantic of up to 0.26 °C yr^{-1} since 2004 (Mills et al., 2013), and ocean temperature during the first half of 2012 was one of the warmest on record in the past 150 years. This event was attributed to less cooling from the atmosphere (Chen et al., 2014), and provides a glimpse into what the future may hold under future climate warming. In addition to temperature change, the Northeast US has had a ~ 15 % increase in overall rainfall, and a 71 % increase in heavy precipitation events since the late 1800s (Walsh et al., 2014; see also Spierre and Wake, 2010) that may drastically change freshwater and nutrient delivery rates to coastal waters.

Short term studies documenting water quality changes in Buzzards Bay are limited. Turner et al. (2009) analyzed trends in water quality between 1987 and 1998 at eight sites primarily in central Buzzards Bay. Although Turner et al. (2009) did not observe an overall trend in annual temperature across their 11 year record from 1987–1998, they did find a significant increase in May temperatures, suggesting that spring warming may have occurred during that period. They also found that water column nutrients and

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chlorophyll either remained the same, or declined over their study period that was related to improvements to a large municipal wastewater treatment facility that discharges into the open waters of Buzzards Bay (New Bedford, MA). Climate driven changes to Buzzards Bay, coupled with limited analyses of coastal nutrient trends, justifies the need to revisit long-term water quality datasets to investigate for climate-related signals and biogeochemical–climate synergies.

Here, we present the results of an analysis of 22 years of summertime water quality data collected through a citizen-science monitoring program in the coastal embayments across Buzzards Bay. The Buzzards Bay Coalition (www.savebuzzardsbay.org), a local non-profit organization, maintains an innovative, long-term, citizen-science water quality monitoring program. Since, 1992, volunteers through this program have collected summertime water samples from approximately 200 sites across Buzzards Bay each year, with station coverage concentrated primarily in the coastal embayments (www.savebuzzardsbay.org). This program has engaged hundreds of local citizen scientists, and data from the monitoring program is used for education, outreach, and management actions to protect the waters of Buzzards Bay (Buzzards Bay Coalition, 2011). This monitoring program has provided baseline data spurring nutrient reduction efforts such as expansion of sewering and upgrading of wastewater treatment facilities in several subwatersheds.

This water quality dataset provides a unique perspective across both space (regionally) and time with coverage of coastal embayments along a gradient of nitrogen loads and different geomorphologies while also having multi-decadal coverage over a period of rapid climatic change. Using this dataset, we show: (1) spatial patterns in water quality are driven by both nutrient loading and the geomorphology of the embayment, (2) significant summertime warming has occurred across the coastal embayments of Buzzards Bay which equated to nearly 2 °C over 22 years, (3) temporal trends in water quality indicators suggest chlorophyll is increasing at a faster rate than inorganic and organic nitrogen species, and (4) the yield of Chl *a* per unit TN has increased over

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the time series. We conclude by describing possible factors that may be driving the broader-scale patterns observed.

2 Methods

2.1 Data collection

The data for this analysis were collected through a long-term citizen-science program to understand the water quality of coastal Buzzards Bay. Water samples were collected by citizen volunteers who measured temperature (calibrated thermometers), salinity (field hydrometer), and dissolved oxygen (DO, modified Winkler titration, Hach OX2P test kits) weekly from late May through September typically between 6–9 a.m. LT. In July and August, water samples were collected during 2 to 4 sampling events from a subset of sites during the last three hours of an outgoing tide. Water samples were either filtered on site or after immediate transport to a laboratory and were kept on ice in the dark while transported for further analysis (1992–1996 Woods Hole Oceanographic Institution; 1997–2008 University of Massachusetts Dartmouth; 2009–2013 Marine Biological Laboratory). Inorganic nutrients nitrate and nitrite (NO_3^- + NO_2^-) were analyzed spectrophotometrically by automated Cd reduction (Johnson and Petty, 1982) and orthophosphate (PO_4^{-3}) by the molybdenum blue method (Johnson and Petty, 1983). Ammonium (NH_4^+) was measured using the phenol hypochlorite method (Strickland and Parsons, 1972), and total dissolved nitrogen was measured as nitrate following persulfate digestion (D'Elia et al., 1977). Particulate organic carbon (POC) and nitrogen (PON) were measured by elemental analysis (Sharp, 1974). Chlorophyll *a* (Chl *a*) and phaeopigments were measured following acetone extraction and standard spectrophotometric methods (Parsons et al., 1989). Beginning in 2002 at a select few sites, point measurements of temperature, salinity, DO, pH, and Chl *a* were collected using a YSI6600 datasonde (YSI). The program's methods are outlined in a Quality Assurance Project Plan that has been approved by the Massachusetts Department of Envi-

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ronmental Protection and the US Environmental Protection Agency (Williams and Neill, 2014).

2.2 Auxiliary Data

We compared the water quality variables measured here to nitrogen loading estimates from the Massachusetts Estuary Project reports and the Buzzards Bay National Estuary Program (BBNEP) loading estimates using comparable loading assumptions as summarized by Costa (2013). We compare the trends from this citizen science monitoring program to other publically available datasets from the same time period collected locally including: hourly air temperatures from the NOAA National Data Buoy Center (buoy BUZM3, 41°23′48″ N, 71°2′0″ W) and daily water temperature recorded at the Woods Hole, MA dock (Nixon et al., 2004, buoy BZBM3, 41°31′25″ N 70°40′16″ W). We also compare the water quality dataset to monthly precipitation records collected by the Massachusetts Rainfall Program from April–June from 3 rain gages located around Buzzards Bay from the towns of Dartmouth, East Wareham, and Falmouth which all contained no missing data from years 1992–2013.

2.3 Data analysis

The purpose of this analysis was to examine embayment-scale patterns in water quality rather than within-embayment patterns. The monitoring program expanded over time and many sites were only sampled transiently, and to minimize spatio-temporal aliasing, we chose to include in this analysis only 122 sites (Fig. 1) chosen as follows: first, sites were grouped into 27 spatially separate embayments, and second, sites were included in the analysis only if they were sampled both during the first and last five years of the time series and had at least 15 years of data of data collected. Embayments were classified as “river-fed” (Fig. 1, triangles) if the standard deviation in embayment mean salinity over the entire record was greater than 5 ppt (Table S1 in the Supplement) and were otherwise classified as “groundwater-fed” (Fig. 1, circles). Only data collected

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during July and August were considered for the analysis of mean summertime water quality because, for the majority of the dataset, nutrient parameters were only available from these months. Samples within each embayment and across sampling events were then averaged in order to calculate each embayment's summertime mean water quality. Chl *a* data were log-10 transformed (Campbell, 1995) prior to averaging and further analysis.

We performed principal component analysis (PCA) and factor analysis (FA) to examine the spatial patterns among water quality variables. Identifying patterns in water quality can be complicated by covariance between forcings (nutrients) and ecosystem responses (Chl *a*). PCA develops new, uncorrelated variables from linear combinations of the measured observations. In addition to using PCA/FA to reduce the number of dimensions in the data space, the resulting FA variable loadings onto each factor and factor scores for each embayment can help explain some of the relationships between measurements. PCA has been used in other shallow coastal systems to quantify spatial patterns in water quality (Boyer et al., 1997; Caccia and Boyer, 2005) and develop eutrophication indicators from the eigenvector that explains the largest amount of variance in the dataset (Primpas et al., 2010; Fertig et al., 2014). PCA and FA were applied to the summertime averages from all embayments. All years with missing data in any variable were discarded and data were standardized prior to PCA to have zero mean and unit variance. Principal components with eigenvalues < 1 were discarded for the FA. The remaining principal components were then rotated using an iterative, orthogonal rotation (VARIMAX) using Matlab code from Glover et al. (2011).

To determine if water quality indicators have changed over time, linear regression through time was then applied to the summertime embayment means using the associated standard error to weight the data points accordingly (Glover et al., 2011). Linear regression was also applied to water and air temperatures and precipitation data. Normality of residuals was tested with Shapiro–Wilk tests. Type-II regression was performed on property–property correlations with uncertainty in both independent and dependent variables (York, 1966; Matlab code from Glover et al., 2011). We test dif-

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ferences between regression characteristics using a paired t test with pooled error variance.

3 Results and discussion

Buzzards Bay, MA represents a natural laboratory to observe the combined impacts of nutrient loading, management action, and climate change on coastal waters. The coastal waters of Buzzards Bay exhibited considerable spatial (Fig. 2) and temporal variability (Fig. S1 in the Supplement) in both physical (Table S1) and water quality parameters (Table S2). Our regional view of Buzzards Bay documents the changes in water quality of the coastal embayments over time. As such, the purpose of this analysis is not to characterize in great detail the fine-scale, local changes in water quality within each embayment; thus our aggregating, embayment-scale perspective may not reflect changes in localized water quality drivers at the individual sampling stations. Although this analysis only includes measurements from July and August, summertime water quality is relevant to managers because that is when many adverse impacts are likely to be observed, for example, hypoxia, benthic algae accumulations, or eelgrass stress due to temperature (Moore and Jarvis, 2008; Moore et al., 2011).

3.1 Spatial patterns in water quality

The PCA/FA of the embayment summertime means revealed substantial correlations among water quality indicator variables (Fig. 3a). There were 4 factors with eigenvalues ≥ 1 that explained 75 % of the total variance in the dataset. Many variables typically considered telltale signs of eutrophication, e.g. elevated POC, PON, and Chl *a* loaded heavily on the first factor (Primpas et al., 2010; Fertig et al., 2014), suggesting that the first factor may represent a particulate biological signal in the dataset (Table 1). Temperature also loaded heavily onto the first factor, suggesting positive correlation between biological response variables and T. The second factor grouped inorganic nutrients, $NO_3^- + NO_2^-$, NH_4^+, and PO_4^{-3}, and the variability in the inorganic nutrient species were approximately orthogonal to Chl *a* (Fig. 3a, Table 1), indicating little correlation between DIN and Chl *a* concentration. Salinity loaded most heavily onto the third factor which likely reflects relative inputs of freshwater while fourth factor likely reflects the strong thermodynamic relationship between temperature and dissolved oxygen saturation (DOsat) (Table 1).

There are several possible explanations for the lack of correlation between DIN and Chl *a*. First, shallow, high productivity coastal lagoons may be dominated by other primary producers such as macroalgae, submerged aquatic vegetation, or microphytobenthos, resulting in rapid uptake and biological conversion of inorganic N to organic N by benthic producers which would not be part of the sampled water column N pool (McGlathery et al., 2007; Glibert et al., 2010). Second, this pattern would also be seen if production was not N limited; however, this is an unlikely explanation as most embayments had low inorganic N/P ratios (calculated from Table S2 as DIN/PO_4). The mean N/P ratio was lower than 7 in every embayment, well below the Redfield ratio of 16 : 1 (Redfield et al., 1963), suggesting that primary production may be limited by N during this time of year (Howarth and Marino, 2006; Howarth et al., 2014; Hayn et al., 2014). Third, in N limited systems, primary producers can draw down DIN to low levels even under high TN (Hayn et al., 2014). For example, trends in N species varied with increasing, decreasing, and stable concentrations over time (described in more detail below, Sect. 3.2). Of the 7 embayments with increasing trends in TN, only two embayments also had increasing trends in DIN (#s 1 and 7). In both cases, the rate of TN increase was 2.88 and 6.46 times larger than the rate of DIN increase, suggesting that changes in DIN may not reflect changes in TN. If this water quality monitoring program measured only summertime DIN as an indicator of water quality, the program would not capture the degree of eutrophication in Buzzards Bay (Souchu et al., 2010; Glibert et al., 2014).

Comparing factor 1 and factor 2 scores shows a clear spatial pattern among embayments that reflects a combination of the dominant morphology and relative nutrient

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loadings (normalized by estuarine area), which varied by over an order of magnitude across the embayments (Fig. 4). Embayments with both positive factor 1 and factor 2 scores were largely riverine dominated, located on the northwestern side of Buzzards Bay (Figs. 1 and 3b, triangles), and tended to have lower salinities. These embayments also had the largest watersheds, were generally the most urbanized, and had larger nutrient inputs per unit embayment area. In contrast, embayments that were lagoonal systems with little riverine input and were more saline tended to have lower factor 1 and 2 scores (Figs. 1 and 3b circles).

We found strong evidence of a larger Factor 1 score, or biological signal (and reduced water quality), in embayments with higher total nitrogen loads (organic plus inorganic) (Fig. 4), similar to other shallow coastal estuaries and other studies in the Buzzards Bay region (e.g. Benson et al., 2013). Using the factor 1 score rather than Chl *a* or TN as an ecosystem indicator incorporates nutrient availability, organic matter, and Chl *a* into a combined variable that would be sensitive to changes in any of these variables. Further, the relationship we found is consistent with estuaries which are not saturated with nutrients along a conceptual nutrient loadecosystem response curve (Glibert et al., 2010). The separation of the river dominated and lagoonal systems along the nutrient loading gradient (Fig. 4, Glibert et al., 2010) further supports the pattern of embayments in factor space (Fig. 3b). Over the observational time span, embayments with consistently large negative values of factor 1 had good water quality, agreeing well with other ecosystem indicators such as eelgrass extent (Latimer and Rago, 2010; Costello and Kenworthy, 2011; Benson et al., 2013). Five of the six embayments with normalized factor 1 scores ≥ 0 are federally listed as nutrient impaired waters, further supporting using factor 1 as an indicator of the biological response to nutrient loading.

3.2 Temporal trends in water quality

We found long-term, statistically significant positive trends in temperature in 15 of the 17 embayments (Fig. 5a, Table 2), and on average across all embayments, Buzzards Bay warmed at a rate of 0.082 ± 0.025 (SD) $^{\circ}$C yr^{-1} (Table 1) during sum-

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mer. Temperature trends from the citizen science database agreed well with both July/August air temperature trends (NDBC buoy BUZM3, $0.091 \pm 0.069\,^{\circ}\mathrm{C\,yr^{-1}}$, $r^2 = 0.422$, $p = 0.004$, data not shown) and water temperature recorded at the Woods Hole dock ($0.12 \pm 0.005\,^{\circ}\mathrm{C\,yr^{-1}}$ SD, $r^2 = 0.596$, $p < 0.0001$). The regional warming observed across Buzzards Bay embayments may have substantial implications for biological processes that are strongly temperature dependent (see below, Sect. 3.3).

There were also statistically significant increasing trends in Chl *a* in 15 of the 17 embayments, with a mean slope of 0.0171 ± 0.0088 $\log_{10}$ (Chl *a* ($\mathrm{mg\,m^{-3}}$)) $\mathrm{yr^{-1}}$ (SD, Fig. 5b, Table 1), or increasing at a mean rate of $4.0\,\%\,\mathrm{yr^{-1}}$ (95 % confidence interval, 3.0–5.0). The increase in Chl *a* alone is symptomatic of degrading summer water quality across Buzzards Bay, and the widespread increase in Chl *a* is consistent with declines in bay-wide eelgrass extent (loss of $3.5\,\%\,\mathrm{yr^{-1}}$, Costello and Kenworthy, 2011). However, the corresponding long-term trends in summer nutrient concentrations (Figs. 5c, 6 and 7) suggest that nutrient levels may not be the sole factor driving the degradation in water quality (see Sect. 3.3 on climate impacts, below). Of the 17 embayments, only 7 had statistically significant increasing trends in TN with a bay-wide mean trend of 0.236 ± 0.523 (SD) $\mathrm{\mu M\,yr^{-1}}$, and no embayments had declining trends (Fig. 5c, Table 1). Ten embayments exhibited no statistical trends in TN (Fig. 5c) possibly due to high interannual variability in individual nitrogen species that may reflect local sources or changes in nutrient loading.

Inorganic (Fig. 6) and organic (Fig. 7) nutrient temporal trends varied considerably across the embayments with increasing, decreasing, and consistent concentrations over time (Figs. 6 and 7). NH_4 and $NO_3 + NO_2$ declined in 5 and 1 embayments, increased in 2 and 5 embayments, and remained the same in 10 and 11 embayments, respectively, while PO_4 declined in 2 embayments and remained the same in 15. Increases in DON, PON, and POC occurred in a few embayments (Fig. 7), and the observed trends in PON and POC could reflect the increase in Chl *a* (and live phytoplankton biomass) as PON and POC were correlated with Chl *a* ($r = 0.62$, $p < 0.0001$ and $r = 0.61$, $p < 0.0001$, respectively, not shown). However, although nearly all embayments exhibited strong increasing trends in Chl *a*, only 3 embayments increased in POC and PON (Fig. 7a, 7B), suggesting that live phytoplankton biomass may not be the only factor driving the increase in Chl *a*. West Falmouth Harbor (Fig. 1, #16) is notable as the only embayment where $NO_3^- + NO_2^-$, DON, PON, TN, and Chl *a* all increased significantly through time (Figs. 6b and c, and 7b and c), consistent with changes in nitrogen loads, which increased by a factor of about three between the mid- to late-1990s and 2003 and have since remained constant (Hayn, 2012; Howarth et al., 2014; Hayn et a. 2014). As a result of the considerable declines in water quality, efforts to better manage nitrogen loads into West Falmouth Harbor have led to the recent addition of nitrogen removal to a wastewater facility that discharges into the groundwater upstream of the embayment.

3.3 Climatic impacts on water quality

Although changes in Chl *a* concentration over time (Fig. 5b) may be partially driven by increases in total nitrogen (Fig. 5c), the inconsistency of trends among different embayments suggests that other factors such as changing physical drivers may also be influencing water quality. Salinity declined in 4 of 6 embayments that were river-fed (Fig. 8b), and mean summer salinity in these embayments and 6 others (#s 1, 2, 3, 4, 5, 10, 12, 13, 14, and 17) was strongly correlated to the April–June rainfall total ($p < 0.05$). However, there was no significant change in average April–June rainfall totals from 1992–2013 ($p > 0.05$, not shown). Combined with significant warming, these altered drivers may change hydrodynamics (e.g. flushing time) and lateral or vertical stratification. Vertical stratification has influenced water quality in a number of deeper coastal systems by exacerbating hypoxic and anoxic events (Kemp et al., 2009; Rabalias et al., 2009; Murphy et al., 2011). Buzzards Bay is a relatively shallow estuary, however, with an average depth of ∼ 2 m in the embayments and a maximum depth of ∼ 11 m in open waters. From wind shear and tidal cycling, the open waters of Buzzards Bay are rarely and only weakly stratified (Turner et al., 2009), so enhancement of vertical stratification is unlikely to influence water quality. This is further supported by trends in early morning dissolved oxygen saturation (DOsat) in the coastal embayments because although declining water quality was observed in most embayments, only 4 displayed decreasing trends in DOsat (Fig. 8a) suggesting that the coastal embayments are also well-mixed environments.

The warmer summertime temperatures we observed over the time series can also influence biological responses. Warmer temperature can enhance phytoplankton production (Eppley, 1972; Reay et al., 1999; Finkel et al., 2009). For example, for a typical phytoplankton growth Q10 (the increase in growth rate over 10 °C) of 1.88 (Eppley, 1972), the warming of 0.57 to 3.19 °C across the embayments would increase growth rates by ~3–22 % if no other factors are limiting. Increased summer temperatures may also stimulate nitrogen uptake rates, and thus primary productivity, as phytoplankton affinity for inorganic nitrogen has been found to increase with temperature (Reay et al., 1999). Ecosystem respiration can be correlated to temperature in coastal systems (Caffery, 2004), and thus warming can increase potential nutrient availability by enhancing recycling of organic matter and nutrients.

Increases in Chl *a* concentration over time may also be related to changes in the timing of seasonal phytoplankton blooms. Although the open waters of Buzzards Bay show sporadic blooms with no apparent seasonal cycle (Turner et al., 2009), measurements from inside New Bedford Harbor (Fig. 1, #6) suggest that blooms occur from March–October (Turner et al., 2009). If blooms occur in the other embayments similarly to New Bedford Harbor, changes in timing of blooms caused by summer warming or changes to freshwater delivery may alias our measurements in any number of different ways as our analysis is limited to only July and August. It is interesting to note that while summertime chlorophyll in Buzzards Bay has increased, in nearby Narragansett Bay, RI (USA), there has been a 70 % decline in water column chlorophyll since 1960 and changes to the timing of phytoplankton blooms. These changes have been attributed to climate warming, changes in wind shear, and reduced benthic/pelagic coupling (Oviatt et al., 2002; Oviatt, 2004; Fulweiler and Nixon, 2009; Nixon et al., 2009) rather than modification of nitrogen inputs to the system (Nixon et al., 2008).

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Higher primary production rates or changes in the timing of phytoplankton blooms in a warmer climate may explain the trends in Chl *a* (Fig. 5b), but not the fact that Chl *a* is increasing at a faster rate than DIN, DON, PON and POC. Across the embayments, we found strong correlations between TN and Chl *a*, which differed during the first half and last half of the dataset (Fig. 9), with an increased offset ($t = -2.113$, df = 30, $p = 0.0431$) and no change in slope ($t = 0.685$, df = 30, $p = 0.498$), a result remarkably similar to that found in Cartensen et al. (2011) for coastal systems across North America and Europe. This pattern was largely driven by a changing correlation between PON and Chl *a* (not shown, 1992–2002, $r^2 = 0.836$ $p < 0.0001$; 2003–2013 $r^2 = 0.720$, $p < 0.0001$) that may have been caused by several possibilities. First, increasing Chl *a* per unit PON suggests that the ratio of live phytoplankton biomass to detritus has increased although more information would be needed to test this hypothesis. Second, increased eutrophication may also cause a more light limited system, where phytoplankton increase Chl *a* : carbon ratio at the cellular level to compensate for increased shading through the water column (Geider et al., 1997). Third, an increase in the ratio of Chl *a* : PON could be due to warming influences on phytoplankton community composition. Blooms of the red-tide forming dinoflagellate *Cochlodinium polykrikoides* were first reported in 2005, and have since become a regular feature of late summer across Buzzards Bay (BBNEP, www.buzzardsbay.org). If new phytoplankton communities are present compared to historical data, they may have higher Chl *a* per unit N or be able to utilize the existing PON pool in addition to DIN (Mulholland et al., 2009) and outcompete species requiring more labile N. Regardless of the cause, the increase in yield of Chl *a* per unit nitrogen over time has implications for management because in order to restore water clarity to some threshold level, TN in many locations may need to be further reduced than would be predicted based on the assumption of a static relationship between TN and Chl *a* (Cartensen et al., 2011).

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4 Conclusions

The analysis described here highlights the value of long-term monitoring of coastal systems, and the wealth of data across large temporal and spatial scales obtained by a well-managed citizen science monitoring program. We found many of the embayments of Buzzards Bay had poor or declining water quality, suggesting that further nutrient reduction efforts will be critical to improving the coastal waters of Buzzards Bay. The combined biological response to nitrogen loading, summarized by Factor 1 (Fig. 4), suggests that reductions in nitrogen load by nearly fivefold may be required to mitigate the impacts of nitrogen enrichment in some embayments. We found that Chl *a* was increasing in more embayments than nutrients, which may be a result of multiple drivers of change such as temperature (Fig. 5a) and precipitation patterns (Walsh et al., 2014), in addition to anthropogenic watershed nutrient loading. To test these speculations of changing patterns of Chl *a* and nutrients, further investigations into spatial datasets with higher temporal coverage such as satellite ocean color data and investigations into other historical community composition datasets is warranted. An understanding of how nutrient loads to the embayments of Buzzards Bay and embayment net community production has changed over time will clarify whether the changes in Chl *a* are a result of changes in nutrient flux to the system, or due to changes in physical drivers. If climate drivers alter the ecosystem response to nutrient availability, water quality management targets such as total maximum daily loads, calculated from historic ecosystem response curves, may not reach the desired improvements under future climate scenarios. Long-term stewardship of Buzzards Bay will likely require adaptive management that takes into account changing water quality baselines (e.g. Duarte et al., 2009) and forward looking climate trends in order to maintain or restore ecosystem health.

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Table 1. Factor loadings from principal component and factor analysis. Of the 12 original factors, the four with eigenvalues > 1 were retained for rotation. Values closer to 1 or −1 indicate strong correlation between the factor and the original data for a variable, while values closer to 0 indicate little or no correlation. Loadings > 0.5 are bold for visibility.

Variable	Factor 1	Factor 2	Factor 3	Factor 4
PON	**−0.86**	0.31	0.17	0.07
POC	**−0.86**	0.30	0.14	0.16
Chl*a*	**−0.85**	0.08	0.18	−0.03
T	**−0.54**	−0.12	−0.30	0.50
PO_4	−0.26	**0.81**	−0.29	0.15
NH_4	−0.03	**0.81**	0.26	−0.01
$NO_3 + NO_2$	−0.22	**0.76**	0.20	−0.10
Salinity	0.22	−0.10	**−0.86**	−0.09
DOsat	0.04	−0.07	−0.20	**−0.90**
DON	−0.16	0.44	0.43	0.18

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Table 2. Linear fits for temperature, TN, and Chl *a*. Significant fits are bolded. Embayment numbers correspond to locations shown in Fig. 1.

Embayment Name	#	Temperature (°C yr^{-1})					TN (µM yr^{-1})					$\log_{10}$ (Chl*a* (mg m^{-3})) yr^{-1}				
		slope	SD	*r*2	*p*	*n*	slope	SD	*r*2	*p*	*n*	slope	SD	*r*2	*p*	*n*
Westport River	1	**0.096**	**0.006**	**0.373**	**0.003**	**22**	**0.97**	**0.09**	**0.23**	**0.023**	**22**	**0.015**	**0.001**	**0.40**	**0.002**	**22**
Slocums River	2	**0.092**	**0.009**	**0.273**	**0.013**	**22**	−0.02	0.19	0.00	0.934	21	0.004	0.003	0.05	0.346	21
Apponagansett Bay	3	**0.082**	**0.007**	**0.279**	**0.011**	**22**	**0.63**	**0.16**	**0.47**	**< 0.001**	**22**	**0.015**	**0.002**	**0.41**	**0.001**	**22**
Clarks Cove	4	**0.080**	**0.011**	**0.280**	**0.016**	**20**	**0.63**	**0.08**	**0.42**	**0.002**	**20**	**0.014**	**0.002**	**0.26**	**0.026**	**19**
New Bedford Harbor	5	**0.115**	**0.008**	**0.448**	**0.001**	**20**	−0.81	0.24	0.15	0.090	20	**0.014**	**0.004**	**0.53**	**< 0.001**	**20**
Nasketucket Bay	6	**0.108**	**0.016**	**0.231**	**0.024**	**22**	**1.25**	**0.09**	**0.49**	**< 0.001**	**21**	**0.044**	**0.001**	**0.79**	**< 0.0001**	**21**
Mattapoisett Harbor	7	**0.096**	**0.007**	**0.349**	**0.004**	**22**	0.25	0.07	0.16	0.068	21	**0.014**	**0.001**	**0.36**	**0.004**	**21**
Aucoot Cove	8	**0.065**	**0.007**	**0.222**	**0.027**	**22**	**0.38**	**0.05**	**0.33**	**0.005**	**22**	**0.019**	**0.001**	**0.52**	**< 0.001**	**22**
Sippican Harbor	9	**0.087**	**0.009**	**0.202**	**0.036**	**22**	0.23	0.08	0.13	0.113	21	**0.021**	**0.001**	**0.68**	**< 0.0001**	**21**
Wareham River	10	**0.055**	**0.005**	**0.230**	**0.024**	**22**	0.12	0.04	0.03	0.444	22	**0.022**	**0.001**	**0.76**	**< 0.0001**	**22**
Onset Bay	11	**0.082**	**0.007**	**0.369**	**0.003**	**22**	0.06	0.04	0.02	0.520	22	**0.009**	**0.001**	**0.27**	**0.013**	**22**
Buttermilk Bay	12	0.008	0.009	0.002	0.852	21	−0.04	0.06	0.00	0.769	21	**0.019**	**0.001**	**0.73**	**< 0.0001**	**21**
Eel Pond/Phinneys Harbor	13	**0.096**	**0.010**	**0.339**	**0.006**	**21**	0.20	0.06	0.14	0.089	22	**0.018**	**0.002**	**0.61**	**< 0.0001**	**22**
Red Brook Harbor/Pocasset River	14	**0.090**	**0.006**	**0.326**	**0.005**	**22**	**0.58**	**0.06**	**0.37**	**0.004**	**21**	**0.018**	**0.001**	**0.62**	**< 0.0001**	**21**
Megansett Harbor	15	**0.076**	**0.006**	**0.257**	**0.016**	**22**	0.17	0.06	0.13	0.114	21	**0.019**	**0.002**	**0.59**	**< 0.0001**	**21**
West Falmouth Harbor	16	0.068	0.009	0.169	0.057	22	**0.78**	**0.05**	**0.64**	**<0.0001**	**22**	**0.019**	**0.002**	**0.41**	**0.006**	**17**
Quissett Harbor	17	**0.101**	**0.008**	**0.406**	**0.001**	**22**	0.06	0.05	0.02	0.5650	21	0.0037	0.0021	0.02	0.5239	21

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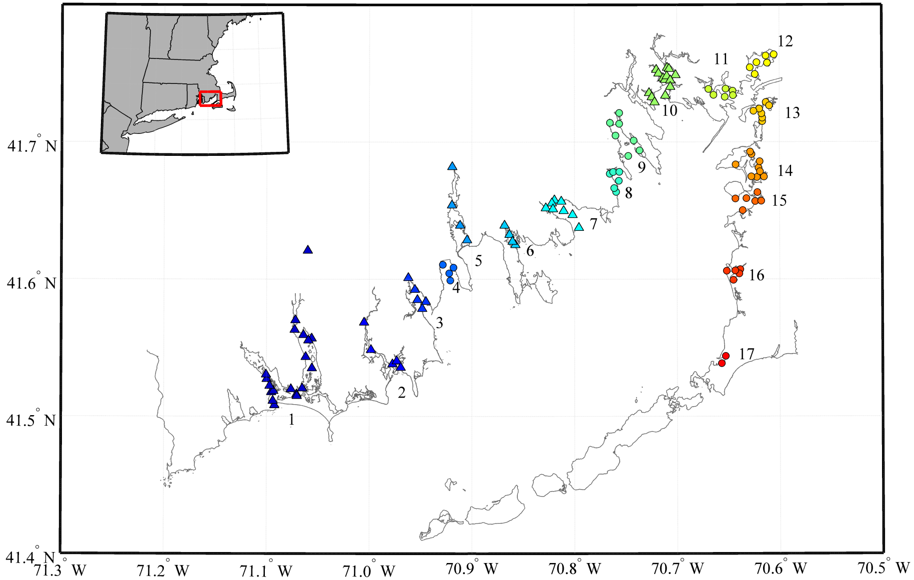

Figure 1. Study area of Buzzards Bay, MA (USA). Dots show individual sampling locations, different colors and numbers correspond to the 17 embayments across the estuary where the gradient from red to blue indicates geographical location around Buzzards Bay. Circle symbols indicate the embayment is classified as "groundwater-fed", while triangles indicate the embayment is classified as "river-fed" based on variability in mean summer salinity. Embayment numbers correspond to data in Tables 2, S1 and S2.

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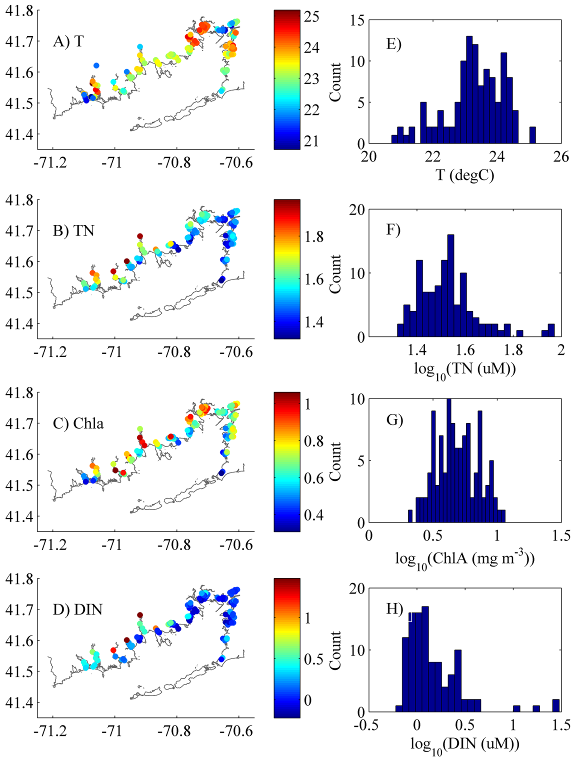

Figure 2. Spatial variability and distributions of mean summer **(a, e)** temperature (T), **(b, f)** total nitrogen (TN), **(c, g)** chlorophyll *a* (Chl *a*), and **(d, h)** dissolved inorganic nitrogen (DIN), respectively from 2012 as an example year. TN, Chl *a*, and DIN have been log10-transformed. The units are the same for the corresponding left and right panels.

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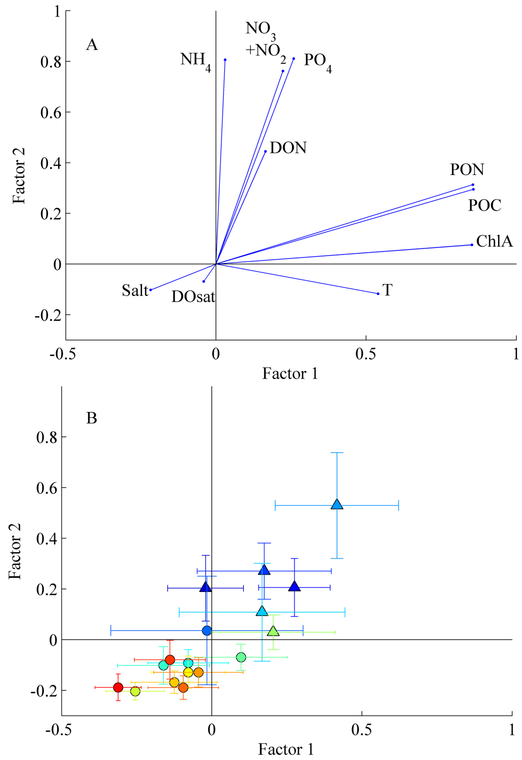

Figure 3. Factor analysis biplots show rotated **(a)** factor loadings and **(b)** mean factor scores across all years for the individual embayments. Factor scores have been normalized by the factor loadings to plot on the same axes, and the factor loadings have been scaled such that the largest loading in the first factor is given a positive magnitude. Colors in factor scores correspond to coloring of embayments shown in Fig. 1. Error bars in **(b)** are the mean ±1 standard deviation. Triangles are embayments "river-fed", circles are "groundwater-fed".

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Long-term trends in Buzzards Bay

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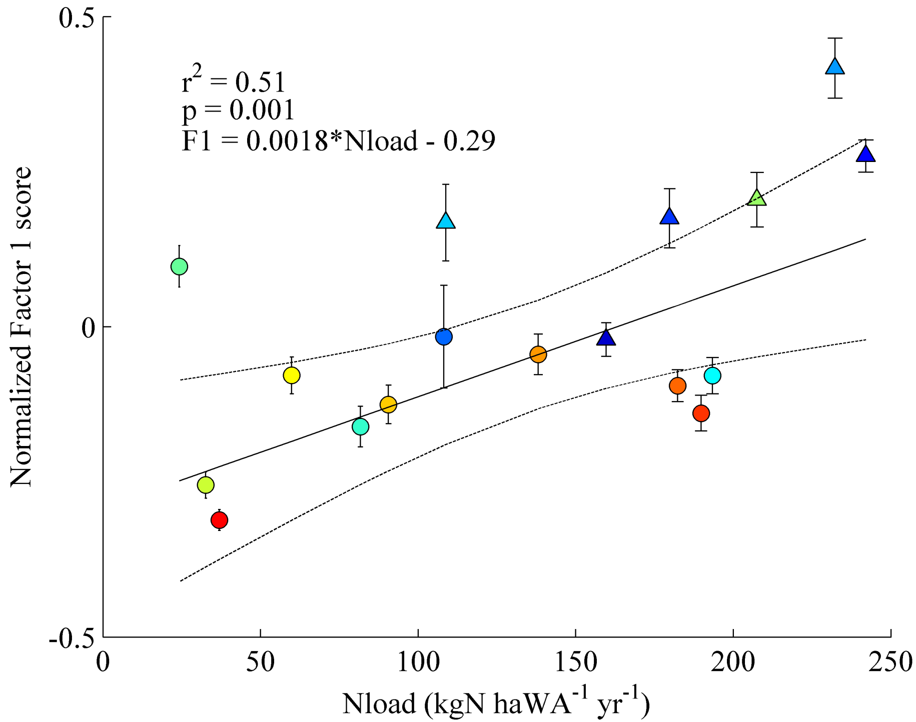

Figure 4. Average factor 1 score from each embayment fitted to nitrogen load estimated by the Buzzards Bay National Estuary Program (Costa, 2013) normalized by estuarine area. Colors correspond to embayments in Fig. 1. Dotted lines are 95 % confidence intervals for the fit and error bars are standard error. Triangles are embayments "river-fed" and circles are "groundwater-fed".

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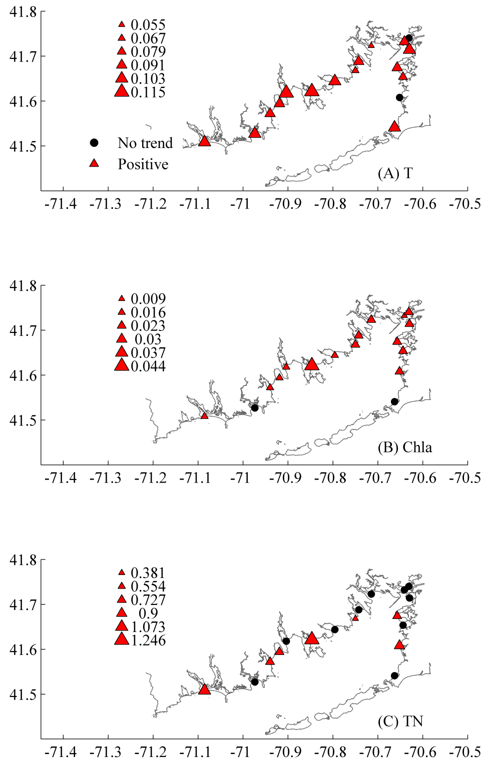

Figure 5. Slopes from long-term trend analysis for **(a)** temperature (°C yr^{-1}), **(b)** chlorophyll (Chl *a*) ($\log_{10}$ (mg m^{-3}) yr^{-1}), and **(c)** total nitrogen (TN) (µM yr^{-1}). Colors and symbols indicate statistically significant trends and direction while the size of the points indicates the magnitude of the slopes.

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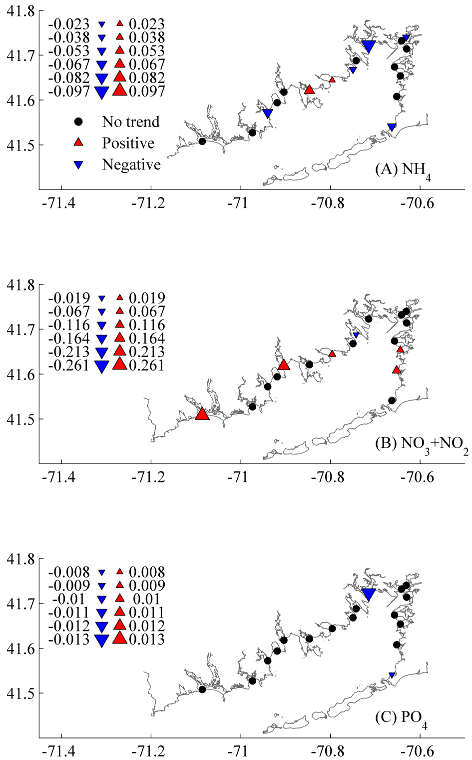

Figure 6. Slopes from long-term trend analysis for **(a)** ammonium (NH_4^+) ($\mu M\,yr^{-1}$), **(b)** nitrate + nitrite ($NO_3^- + NO_2^-$) ($\mu M\,yr^{-1}$), and **(c)** phosphate (PO_4^{-3}) ($\mu M\,yr^{-1}$). Colors and symbols indicate statistically significant trends and direction while the size of the points indicates the magnitude of the slopes.

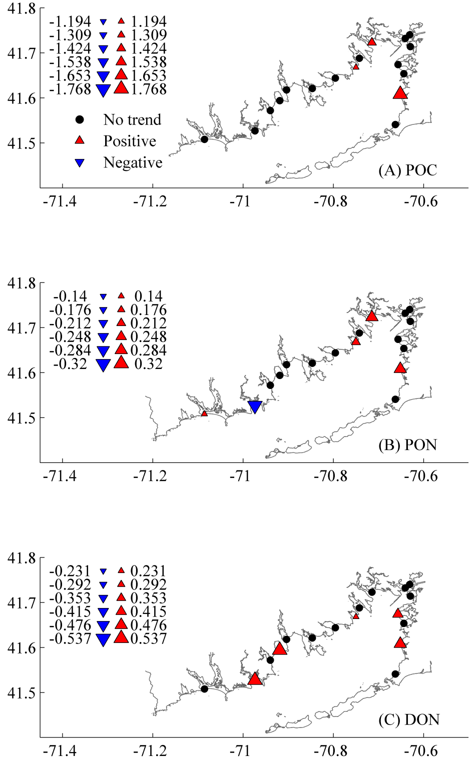

Figure 7. Slopes from long-term trend analysis for **(a)** particulate organic carbon (POC) ($\mu M\,yr^{-1}$), **(b)** particulate organic nitrogen (PON) ($\mu M\,yr^{-1}$), and **(c)** dissolved organic nitrogen (DON) ($\mu M\,yr^{-1}$). Colors indicate statistically significant trends and direction while the size of the points indicates the magnitude of the slopes.

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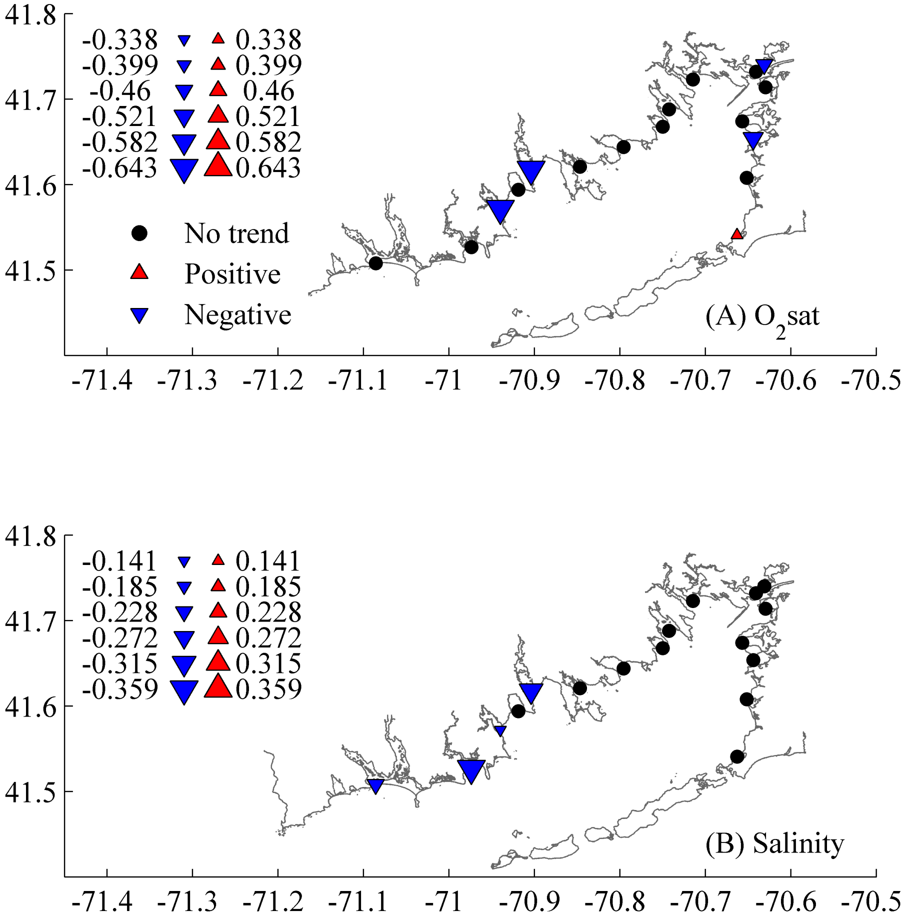

Figure 8. Slopes from long-term trend analysis for **(a)** oxygen saturation (O_2 sat) (% yr^{-1}) and **(b)** salinity (ppt yr^{-1}). Colors and symbols indicate statistically significant trends and direction while the size of the points indicates the magnitude of the slopes.

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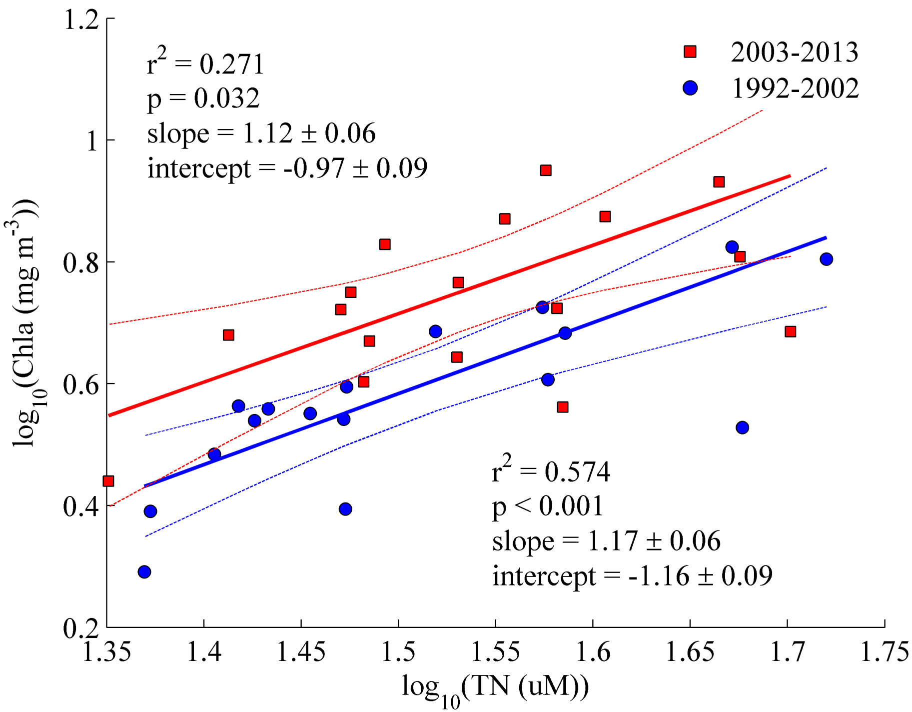

Figure 9. 1992–2002 and 2003–2013 average log-Chl *a* fitted to log-TN. Type-II regression accounted for uncertainty in both dependent and independent variables. Dotted lines are 95 % confidence intervals.

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