

Multivariate Analysis of Water Quality and Plankton Assemblages in an Urban Estuary

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Received: 5 April 2013 / Revised: 14 August 2013 / Accepted: 14 September 2013 / Published online: 3 October 2013
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Abstract Raritan Bay, located between the states of New York and New Jersey, has a long history of cultural eutrophication and associated harmful algal blooms (HABs). Despite striking chemical and biological alterations occurring in Raritan Bay, publications in the early 1960s were the last to report consecutive measurements of *both* water quality parameters and plankton species composition in this system. The objectives of this study were to characterize water quality trends and plankton composition in a eutrophic estuary, compare current environmental conditions to those documented in Raritan Bay 50 years ago (i.e., at the same six sampling sites), and to further clarify the relationship among nutrients, secondary consumers, and algal bloom generation in this system using ordination techniques. This study (monthly data collected from April 2010–October 2012) indicates that Raritan Bay continues to exhibit numerous symptoms of eutrophication, including high algal biomass, high turbidity, violations of the dissolved oxygen standard to protect fish health, and blooms of potentially harmful phytoplankton species. Altered spatial and temporal patterns for nitrate and soluble reactive phosphorus (SRP) over the past 50 years may suggest new, changing, or expanding sources of nutrients. A total of 14 HAB species have been identified, including *Heterosigma akashiwo*, which formed a bloom in the upper Raritan Bay during summer 2012 in association with hypoxic conditions. Multivariate analyses indicate that abundance of this species is positively

associated with high temperature, salinity, nitrate, and SRP and negatively associated with spring river discharge rates and total zooplankton abundance in Raritan Bay.

Keywords Raritan Bay · Water quality · Eutrophication · Phytoplankton · Algal blooms · Ordination

Introduction

Environmental monitoring and assessments of nearshore coastal ecosystems have become critical as human populations expand and intensify the strain on these ecologically and economically important systems. Concentrated human activities have accelerated fluxes of nutrients to coastal waters and led to cultural eutrophication and associated increases in productivity, simplification of aquatic communities (e.g., increased abundance of generalist species and decreased overall diversity), and ecosystem instability (National Research Council 2000). Of particular concern in eutrophic coastal ecosystems is the stimulation of harmful algal blooms (HABs), which cause environmental damage by depleting available oxygen, reducing light availability, disrupting recreational uses, and/or producing toxins harmful to other organisms including humans (Glibert et al. 2005). In recent decades, there has been a global increase in the reported number of HABs (Glibert et al. 2005; Heisler et al. 2008; Schindler and Vallentyne 2008). Few coastal environments have been left unaltered by the effects of anthropogenic eutrophication (National Research Council 2000), and it is clearly one of the major environmental problems facing coastal resource managers.

The Hudson–Raritan estuary (HRE) of New York and New Jersey is one of the most urbanized estuaries, and busiest ports, in the world. Not surprisingly, this system has a long history of environmental degradation. Public policy debates regarding waste disposal and water pollution began as early as

Communicated by Dennis Swaney

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the seventeenth century, and by the turn of the twentieth century, shoreline development, discharges of industrial waste, and inadequate waste disposal methods led to collapse of the oyster industry, reduced fish abundance and diversity, beach closures, nutrient enrichment, and severe oxygen depletion (Environmental Protection Agency (EPA) 2007). The present study aims to build upon current understanding of the coastal eutrophication process by evaluating linkages among nutrient concentrations and phytoplankton and zooplankton assemblage composition in the Raritan Bay ecosystem, located at the southern portion of Lower New York Bay between the states of New York and New Jersey.

Although historical water quality data for this system exist, current information on water quality characteristics in relation to plankton dynamics is lacking. In addition, more recent studies in this system focus primarily on sediment contamination, heavy metals, and bioaccumulation of persistent organic pollutants (POPs; e.g., Meador et al. 1995; Wolfe et al. 1996; Reiser 2004). To our knowledge, publications by Jeffries (1962, 1964) in the early 1960s were the last to report on the results of *both* nutrient and plankton dynamics in Raritan Bay for a period of longer than 1 year. Even then, Jeffries (1962) reported that rich nutrient supplies and a sluggish circulation were contributing to extremely dense plankton populations and low summer dissolved oxygen concentrations. Interestingly, when Jeffries conducted his study in Raritan Bay, cultural eutrophication was not yet perceived as a widespread problem in the coastal marine environment (Nixon 1995). However, it was around this time (i.e., mid-1950s to the mid-1970s) that scientists first made the connection between nutrient loading from the land and intensive coastal algal blooms and began expressing serious concern about eutrophication after conducting research in this region of the Long Island Sound and the New York Bight, which includes Raritan Bay (Ryther and Dunstan 1971; Nixon 1995). Therefore, our intent was not only to complete a multiyear study of water quality and plankton dynamics in a eutrophic estuary but also to collect environmental data from the same six sampling sites in Raritan Bay monitored in this system by Jeffries (1962), so that current environmental conditions could be compared to those documented in this system more than 50 years ago.

Another indicator of the long-term water quality degradation of the Raritan Bay is the large phytoplankton blooms that have occurred annually in this system for decades (Olsen and Mahoney 1986). Eutrophication problems, such as HABs, cannot be diagnosed by studies and monitoring programs that focus on algal chlorophyll biomass alone (Hallegraeff 1993). Yet, previous studies in Raritan Bay do not include plankton identifications to the genus or species level or analysis of changes in phytoplankton species composition in relation to both “bottom up” (nutrient concentrations and ratios) and “top down” (zooplankton abundance and composition) effects. As

a result, this study also aims to further clarify the relationships among habitat characteristics, nutrient concentrations, plankton species composition, and algal bloom generation. This satisfies an urgent need in this system and in general to better understand the environmental conditions contributing to the success of individual species and leading to the generation of phytoplankton blooms in estuaries (Livingston 2001; Cloern and Dufford 2005). Here, we address variation in phytoplankton species composition using ordination, which has been described as a group of methods for data reduction leading to hypothesis generation (Kent and Coker 1992). Ordination techniques are especially useful for analyzing community data because they reveal the strongest patterns in species composition by representing complex ecological relations in low-dimensional space (McCune and Grace 2002) and provide interpretations of species–environment relationships. This represents one of the few available studies, and the first for the Raritan Bay ecosystem, to include accurate and consistent data over a 3-year period on phytoplankton assemblage composition in relation to both nutrients and zooplankton composition and to use ordination techniques to evaluate linkages among these parameters. This approach, which can be adapted to examining the eutrophication process in other coastal systems, will aid in predicting biological changes under certain circumstances and identifying where corrective measures are most necessary.

Study Area

The Raritan River flows ~26 km through the central New Jersey Highlands and Piedmont before slowing in tidewater at New Brunswick, and its estuary extends another ~23 km before entering Raritan Bay at South Amboy. The Raritan River Basin (RRB), the largest river basin in NJ, comprises approximately 2,850 km² and contains approximately 1.2 million people. Land use is primarily woodlands, suburban and agricultural in the upper RRB, and the lower RRB is dominated by urban/suburban land use with industrial and commercial centers throughout (Newcomb et al. 2000).

Water quality studies within the basin have indicated that total phosphorus (TP) and fecal coliform bacteria are two of the water quality parameters that most commonly fail to meet New Jersey reference levels (Reiser 2004). Permitted point source dischargers in the RRB, which include >575 wastewater treatment facilities (WWTPs) concentrated in Middlesex County surrounding the mouth of the Raritan River (data acquired from New Jersey Department of Environmental Protection (NJDEP)), account for more than one third of the TP and one third of the nitrate load at low- and median-flow conditions (Reiser 2004). Most municipal WWTPs were upgraded to full secondary treatment during the 1970s and 1980s (Shallcross et al. 2002). In the earlier part of the century, however, both New York and New Jersey communities were

discharging raw sewage into the HRE (Brosnan and O'Shea 1996; EPA 2007). Incremental constructions and upgrades of sewage treatment plants over the past few decades have yielded long-term improvements in water quality (Brosnan and O'Shea 1996). However, one of the remaining water pollution problems in this system results from combined sewer overflows following storms that discharge raw sewage and street debris (EPA 2007). For example, damage caused by hurricane Sandy on 29 October 2012 took several WWTPs offline in New Jersey, and raw sewage carrying high levels of fecal coliform bacteria, nutrients, and pollutants emptied into New Jersey's rivers and estuaries (NJDEP 2013). Nevertheless, nonpoint sources continue to be the largest source of nutrient loads in the RRB, and studies have shown that these nonpermitted yields of nutrients are most highly correlated with septic system density and other factors related to urban areas (Reiser 2004).

Water quality data in Raritan Bay indicate that, despite significant progress following abatement measures, the system is still polluted. The water and sediments are contaminated with a variety of pollutants such as heavy metals, PCBs, polynuclear aromatic hydrocarbons (PAH), and pesticides (Breteler 1984). Excessive nutrient loads also continue to degrade water quality by contributing to algal blooms, hypoxia, and fish kills (Olsen and Mulcahy 1991; Reid et al. 2002). Chronic phytoplankton blooms of various species, including the dinoflagellates *Ceratium tripos* (Müller) Nitzsch, *Prorocentrum micans* Ehrenberg, and *Heterocapsa rotundata* (Lohmann) Hansen, and the raphidophyte *Heterosigma akashiwo* (Hada) Hada ex Hada et Chihara (formerly misidentified as *Olisthodiscus luteus* Carter, Smayda 1998; Olsen and Mahoney 2001) have occurred in the HRE and New Jersey coastal waters for over three decades (Gastrich 2000). These blooms have been associated with moderate bather discomfort and/or illness, diminished esthetic value of beaches, hypoxia, and fish mortality (Gastrich 2000).

Methods

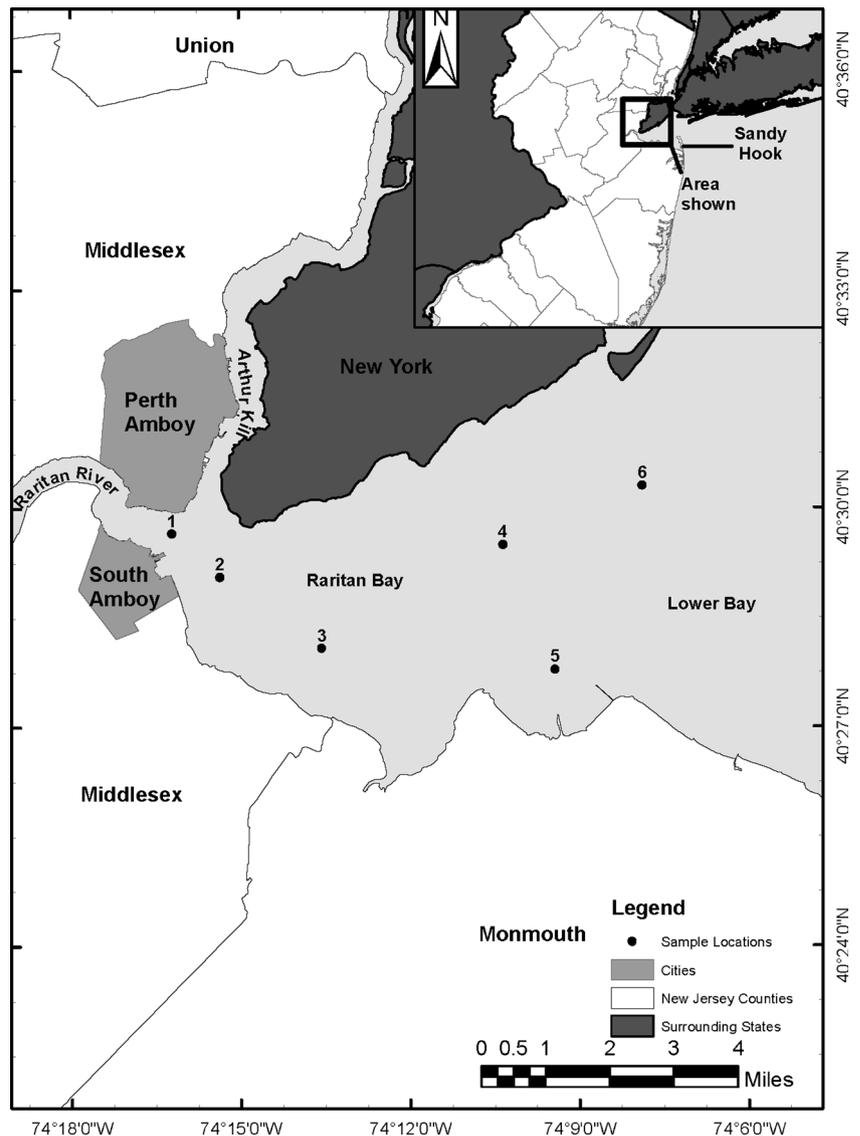
Water Sample Collection Six sites in Raritan Bay (Fig. 1) were sampled on a monthly basis from April through November. The estuarine water quality study extended from April 2010 through October 2012. It was not possible to sample in November 2012 and complete three full years of sampling due to damage caused by hurricane Sandy to the Atlantic Highlands marina on October 29, 2012. Water temperature, salinity, pH, and dissolved oxygen (DO) were determined using a YSI 6820 V2 multiparameter meter, which was calibrated prior to each sample period. Water transparency was assessed using Secchi depth data (Wetzel and Likens 2001). A vertical polycarbonate Van Dorn water sampler (2.2 L) was thoroughly rinsed with site water and used to collect water

samples near the surface (~0.5 m), at mid-depth (~1.5 m), and low depth (~3.0 m; average depth at the six sampling sites is ~4.8 m) between 1000 and 1200 hours local time for biological and chemical analyses. Samples for nutrient analyses were poured into acid-cleaned bottles, maintained in darkness on ice for transport to the laboratory, and refrigerated or frozen as appropriate until analysis (EPA 1993). Phytoplankton samples were poured into amber bottles and preserved with acidic Lugol's solution on site. Zooplankton samples were collected at a depth of 1.5 m using a 12-l Schindler–Patalas trap and preserved with 4 % buffered formalin. All plankton samples were held at 4 °C until analysis. A Nalgene hand vacuum pump and gridded 0.45- μ m membrane (Whatman WCN filters) was used on site to filter 200 mL of sample water for analysis of chlorophyll *a*, and filters were placed in opaque canisters and maintained on ice until analysis.

Nutrient and Plankton Analyses Nutrient analyses were performed using standard methods (APHA 2012). Concentrations of all nutrients, except for total iron (Fe_t), were determined colorimetrically with a HACH (Loveland, CO, USA) DR/2500 spectrophotometer. Accuracy checks, including the use of reagent blanks and standard solution adjusts, followed HACH procedures (Hach Manual for DR/2500 spectrophotometer). Nitrate (NO_3^-) was determined within 12 h of collection using the method of cadmium reduction, and the Nessler method was used within 24 h for determination of ammonium (NH_4^+). Samples for the analysis of soluble reactive phosphorus (SRP) were first filtered in the field with a Whatman Puradisc 0.45- μ m syringe filter and analyzed within 48 h using the ascorbic acid-molybdate blue method. Silicon (Si) and ferrous iron (Fe_f) were also determined colorimetrically using the silicomolybdate and 1,1,10 phenanthroline methods, respectively. Samples for the analysis of ferrous iron were analyzed as soon as possible after collection (i.e., within 8 h) to prevent oxidation of ferrous to ferric iron. Samples for analysis of total iron were preserved with concentrated nitric acid to adjust the pH to ~2 and sent to Penn State Agricultural Analytical Services laboratory for analysis via inductively-coupled plasma–atomic emission spectrometry as per EPA method 200.7. All chlorophyll extractions were performed using 90 % basic acetone (Wetzel and Likens 2001), and spectrophotometric analyses (using the UV–VIS spectrophotometer) were conducted within 48 h of sample collection (Lorenzen 1967).

Phytoplankton counts were conducted using a Palmer Maloney cell (X400) under a Leica DM 1000 phase contrast microscope, and cells were identified and counted until at least 100 specimens of the most common species were counted (LeGresley and McDermott 2010). Taxa abundances, including colonial and filamentous forms, were quantified by enumerating single cells, and identifications were taken to the lowest taxonomic level. Many identifications were taken to genus or

Fig. 1 The Raritan River Basin and Bay, showing the six sampling stations for temperature, salinity, pH, dissolved oxygen, Secchi depth, nutrient concentrations, phytoplankton biomass as chlorophyll *a*, phytoplankton and zooplankton assemblages. Samples were collected from 2010–2012 on a monthly basis (April–November). These are six of the same sampling sites monitored for nutrients and zooplankton assemblages by Jeffries (1962, 1964)



species level, but for some flagellates, individual genera were difficult to consistently differentiate from one another in preserved samples and were mostly identified to the class level (i.e., Cryptophyceae, Prymnesiophyceae, Prasinophyceae, Raphidophyceae). *Pfiesteria piscicida* Steidinger and Burkholder, *Pfiesteria shumwayae* Glasgow and Burkholder, cryptoperidiniopsoids, and other physically similar taxa were classified as “*Pfiesteria*-like” under light microscopy (Seaborn et al. 2006). In the case of obvious phytoplankton blooms and water discoloration, fresh samples were brought back to the laboratory for easier identification of bloom-forming species. For the purpose of this study, a bloom is defined as two standard deviations above the 3-year mean phytoplankton biomass (threshold based on means from this study = 4.5×10^4 cells ml^{-1} and $22 \mu\text{g L}^{-1}$ chlorophyll *a*). For enumeration of zooplankton, a subsample of ~10 % of the total sample was examined (Goswami 2004). Counts were

conducted using a Sedgewick Rafter counting cell ($\times 100$), also under a Leica DM 1000 phase contrast microscope. Identifications were taken to genus or species level according to Todd et al. (2006) and Johnson and Allen (2005). Juvenile or nauplii stages of zooplankton were counted separately from their adult stage because of differential feeding patterns (Johnson and Allen 2005) and difficulty in identifying them to genus level under light microscopy. *Mnemiopsis leidyi* Agazzi were roughly quantified at each sampling site by counting the number of individuals in Schindler–Patalas samples prior to preservation.

Data Preparation and Statistical Analyses Paired, two-sample *t* tests were used for between-site and between-year comparisons of water quality and plankton abundance. All analyses were performed at an alpha level of 0.05. Phytoplankton and zooplankton data were compiled into separate matrices of cell

number by site and sample date, resulting in a final matrix of 78 columns of phytoplankton cell counts and 40 columns of zooplankton counts by 138 sample rows (3 year \times 8 months \times 6 sites, minus November 2012 data). To reduce the “noise” (variability) in the plankton datasets and enhance detection of assemblage patterns in relation to environmental parameters, rare taxa (defined as present in <5 % of samples) were removed prior to analysis (McCune and Grace 2002), and abundances of the remaining 41 phytoplankton species and 32 zooplankton species were log transformed after 1 was added as a constant. Every plankton sample used in the analysis had a corresponding suite of physical and chemical measurements (described above) that could be related to ordinations by species composition. Dissolved N/P and Si/N ratios, precipitation data and Raritan River discharge rates were also included in the analyses because previous studies have indicated that they are important factors governing phytoplankton species composition and bloom development (e.g., Riegman 1998; Anderson et al. 2002, Rothenberger et al. 2009). Precipitation data were acquired from the New Jersey Weather and Climate Network, and precipitation was determined for each sample by summing the total amount of rainfall at the New Brunswick station (i.e., the nearest municipal area) for the week prior to the sampling date. Raritan River discharge rates were acquired from US Geological Survey (USGS) at Bound Brook, which is approximately 20 km upstream from the upper edge of the study area at site 1. Season (April–May=spring; June–September=summer; October–November=fall), site, and year were all included in the nonmetric multidimensional scaling (NMDS) analyses (described below) as categorical variables.

Ordination techniques, which order samples along axes expressing the main trends or gradients in the data, were used to investigate potential environmental predictors of phytoplankton and zooplankton assemblage patterns. NMDS, considered the most effective ordination method for ecological data (McCune and Grace 2002), was used to establish seasonal and site-to-site differences in plankton assemblage structure and to interpret those differences in terms of environmental conditions. All NMDS ordinations (Table 1) were carried out using PC-ORD version 5.0 software (MjM software 2010).

Results

Climatic Conditions During this study (2010–2012), the RRB was affected by two tropical cyclones and record-breaking late winter and early spring precipitation totals in 2010 and 2011, all associated with high flooding in the region (Fig. 2; National Oceanic and Atmospheric Administration (NOAA) National Hurricane Center (NHC) 2013, Office of the New Jersey State Climatologist (ONJSC) 2010; 2011). The study began in April 2010, the warmest April on record since 1895 in New Jersey, following the wettest March on record (ONJSC 2010). The

result of large amounts of rain falling on soils still wet from recently melting snows was high river discharge and near-record flooding in the RRB in spring 2010. In marked contrast, drought conditions developed in the coastal counties of the RRB during summer 2010 (Fig. 2). Temperature, precipitation, and river discharge were again well above the state average in April 2011 following major precipitation events and flooding in March (Fig. 2; Office of the New Jersey State Climatologist 2010). In August 2011, tropical storm Irene caused catastrophic flooding in the New Jersey and New York regions with 1.0 to 1.5 m storm surges (NOAA NHC 2013). At Sandy Hook, NJ, wind speeds were recorded at 40 kt with storm surges of 1.4 m (NOAA NHC 2013). Overall, 2011 was an extremely high-precipitation year, setting a 100-year record for the state of New Jersey (Office of the New Jersey State Climatologist 2010). In contrast to the first 2 years of the study, spring precipitation was below average and summer precipitation was near average in 2012 (Fig. 2; Office of the New Jersey State Climatologist 2012). However, hurricane Sandy (29 October 2012), which made landfall along the coast of New Jersey, caused major damage to both the New Jersey and New York coastlines. Although precipitation totals associated with the storm were less than predicted, hurricane Sandy arrived close to spring high tide and created a record storm surge in Raritan Bay at Sandy Hook of >3.0 m.

Water Quality Trends Seasonal, interannual, and site-specific variability in temperature and salinity occurred in this ecosystem (Tables 2 and 3), and both continue to reflect the circulation patterns in Raritan Bay described by Jeffries (1962). Flood tides carry high-salinity water northward past Sandy Hook and sites 6 and 4 toward an area of extensive eddies where it mixes with freshwater from the river, and freshwater enters the bay from the Raritan River and flows southward past site 3 (Fig. 1). As a result, site 6 still has significantly lower temperature (all samples and depths considered; $M=18\pm 1$ °C) than site 5 ($M=19\pm 1$ °C, $p=0.002$). Site 6 also has a significantly higher salinity (all samples and depths considered; $M=28\pm 1$) than site 5 ($M=25\pm 1$, $p=0.001$), particularly during spring months (Table 3). Surface salinities, which ranged from 15 to 34 (grand mean for surface waters 24 ± 0.3), were dependent on seasonal precipitation patterns and river discharge rates. In 2010 and 2012, salinity throughout the bay was lowest in June following the high river flow months of spring and gradually increased through summer and fall as precipitation and discharge declined (Table 2). In contrast, during the record-wet year of 2011 with high-precipitation events continuing through summer, salinity remained low through summer (Table 2). Salinity stratification (i.e., the difference between surface and bottom salinities) was generally greatest at site 1 and least pronounced at site 6 where water stratification is reduced by tidal mixing. Overall, salinity levels and patterns are similar to those

Table 1 Descriptions of the five datasets used for NMDS ordinations, and the questions targeted by each of those ordinations. The various data groupings were chosen to better extract and compare phytoplankton species variation between seasons and years and develop hypotheses about long-term relationships among phytoplankton assemblage structure, individual nuisance or harmful species, environmental factors, and zooplankton composition. The 16 quantitative environmental variables

were Raritan River discharge rates, precipitation, water temperature, salinity, pH, DO, Secchi depth, nitrate, ammonium, SRP, silica, ferrous iron, total iron, chlorophyll *a*, N/P ratio, and Si/P ratio, and the three categorical values were year, season, and site. Rare species (present in <5 % of the samples) were removed and phytoplankton cell numbers were log transformed prior to the analysis

Dataset	Phytoplankton species matrix dimensions	Environmental matrix dimensions	Targeted questions
Full dataset	78 species × 144 samples	19 parameters × 144 samples	How does phytoplankton composition vary from year-to-year?
Full dataset, monthly averages	78 species × 48 samples	18 parameters × 48 samples	How does phytoplankton composition vary from seasonally?
Spring samples	57 species × 36 samples	18 parameters × 36 samples	How does spring phytoplankton composition and bloom-forming species relate to year-to-year changes in environmental conditions and zooplankton composition?
Summer samples	46 species × 72 samples	18 parameters × 72 samples	How does summer phytoplankton composition and bloom-forming species relate to year-to-year changes in environmental conditions and zooplankton composition?
Fall samples	41 species × 36 samples	18 parameters × 36 samples	How does fall phytoplankton composition and bloom-forming species relate to year-to-year changes in environmental conditions and zooplankton composition?

reported for this system 50 years ago (Jeffries 1962) except during spring months when salinity was $\cong 1.5$ times higher than reported by Jeffries (Table 3).

Dissolved oxygen ranged from 4 to 20 mg L⁻¹ (grand mean for surface waters 10±0.2 and bottom waters 9±0.2) with concentrations generally increasing with distance away from the mouth of the river. Oxygen concentrations were lowest during late summer in all 3 years (Table 2) and violated state standards for fish health (≥ 5 mg DO L⁻¹; NJDEP 2011) following dry periods in the lower two thirds of the water column at sites 1–3 in June and August 2010 and in bottom water at site 1 in August 2012. During the unusually wet year of 2011, DO never fell below 5 mg L⁻¹. Hypoxic conditions in summer 2010 and 2012 occurred in association with mixed flagellate blooms ($5\text{--}6 \times 10^4$ cell ml⁻¹; chlamydomonad-like

species, *Cryptomonas* spp., *Euglena* spp., *Heterocapsa rotundata*, *Heterosigma akashiwo*, *Prorocentrum micans* were most abundant). Spatial distribution of summer hypoxia from 2010–2012 is similar to that reported by previous studies with concentrations increasing along the southern shore as river water becomes diluted with bay water (Table 3; Jeffries 1962; Reid et al. 2002). However, DO concentrations were $\cong 1.8$ times higher at site 1 and $\cong 1.4$ times higher at site 6 than reported by Jeffries (Table 3).

Secchi depth ranged from 0.5–4.0 m and generally peaked during the fall season (Table 2). However, when precipitation and river discharge were unusually high in fall 2011, the seasonal increase in light penetration in Raritan Bay was less pronounced. Low Secchi depths reflected both sediments in runoff and algal blooms. For example, the low Secchi depths (i.e., <0.8 m) recorded at all sites on 17 April 2010 followed the wettest March on record in New Jersey and another significant rain event occurring from April 16–17 (ONJSC 2010). In late spring and early summer 2011, low Secchi depths (i.e., <1.0 m) coincided with both high river discharge and blooms of diatoms and/or flagellates (cell densities $>5 \times 10^4$ cell ml⁻¹ and >22 $\mu\text{g L}^{-1}$ chlorophyll *a*).

Nitrate and soluble phosphorus concentrations (795 ± 31 $\mu\text{g NO}_3^{-1}\text{L}^{-1}$, range of 70–3,600 $\mu\text{g NO}_3^{-1}\text{L}^{-1}$; 47 $\mu\text{g SRP L}^{-1}$, range of 0–157 $\mu\text{g SRP L}^{-1}$) continue to indicate eutrophic conditions, as expected (Table 2; Burkholder 2000; Wetzel 2001). Overall, nitrate concentrations are $\cong 20$ times higher at site 1 and $\cong 50$ times higher at site 6 than reported by Jeffries (1962), and SRP concentrations are $\cong 20$ times higher at site 1 and $\cong 10$ times higher at site 6 (Table 3). Spatial and temporal patterns for N and P concentrations in Raritan Bay are also different from that reported by Jeffries. In 1958, nitrate

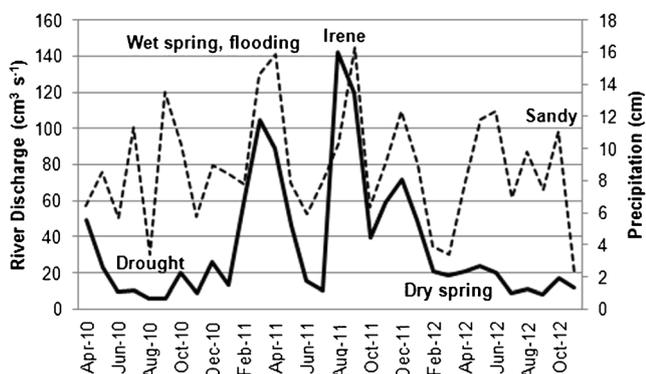


Fig. 2 Monthly mean Raritan River discharge rates (solid line) measured by US Geological Survey at Bound Brook (about 20 km upstream from study area at site 1), and monthly precipitation totals (dashed line) for New Brunswick (i.e., the nearest municipal area) acquired from the New Jersey Weather & Climate Network from April 2010 through October 2012

Table 2 Seasonal mean (in boldface, $n = 36$ for spring and fall and $n = 72$ for summer) \pm 1SE and median levels for each water quality variable measured in Raritan Bay for 2010, 2011, and 2012. Seasonal categories are spring, April–May; summer, June–September; and fall, October–November. Water temperature is reported in degrees Celsius; dissolved oxygen is reported in milligrams per liter; chlorophyll *a*, NO_3^- , NH_4^+ , soluble reactive phosphorus, silica, ferrous iron, and total iron are reported in micrograms per liter. Analysis of ferrous and total iron began in July 2010

Year/Season	Water Temp	Salinity	DO	Secchi	Chl <i>a</i>	NO_3^-	NH_4^+	SRP	Si	Fe_f	Fe_t
2010											
Spring	15±0.5, 14	22±0.4, 22	11±0.5, 12	1±0.1, 1	6±1, 5	487±68, 400	71±13, 40	18±2, 10	2,896±346, 1,800	N/A	N/A
Summer	24±0.3, 25	26±0.3, 26	8±0.3, 8	2±0.1, 2	2±0.2, 2	987±113, 700	112±16, 53	40±2, 37	3,228±248, 2,800	29±6, 30	135±39, 105
Fall	12±0.4, 12	32±0.2, 32	10±0.2, 10	3±0.1, 3	10±1, 10	1428±148, 1,200	80±15, 28	47±1, 46	2,794±265, 2,850	60±7, 60	98±9, 90
2011											
Spring	17±1, 15	22±1, 23	10±0.3, 10	1±0.1, 1	11±2, 4	470±37, 400	62±7, 47	33±3, 29	2,161±250, 1,600	20±2, 20	234±40, 160
Summer	24±0.4, 25	22±1, 24	9±0.5, 9	1±0.1, 1	7±1, 5	677±47, 600	83±14, 43	54±4, 56	3,898±531, 3,150	40±8, 20	82±10, 65
Fall	11±0.1, 11	29±3, 29	10±0.1, 11	3±0.1, 2	13±1, 14	689±71, 700	52±8, 58	59±3, 57	3,986±371, 3,950	50±9, 45	296±37, 245
2012											
Spring	18±0.4, 16	22±0.5, 22	10±0.2, 10	1±0.1, 2	8±1, 4	650±59, 500	105±15, 56	54±3, 50	3,502±348, 2,700	45±6, 35	82±12, 65
Summer	26±0.1, 26	30±0.3, 30	8±0.3, 8	1±0.1, 1	3±0.6, 2	1143±95, 1,050	173±19, 58	72±3, 71	1,994±504, 245	71±11, 60	177±23, 155
Fall	16±0.1, 16	30±0.2, 30	10±0.1, 10	2±0.2, 2	1±0.1, 1	806±63, 800	54±6, 47	63±3, 62	9,150±1087, 7,900	29±5, 25	143±82, 135

DO dissolved oxygen, Chl *a* chlorophyll *a*, SRP soluble reactive phosphorus, Si silica, Fe_f ferrous iron, Fe_t total iron

concentrations were highest in spring and decreased with distance away from the mouth of the Raritan River, and when river flow and nitrate loading decreased during summer, nitrate concentration decreased throughout the bay and station differences in nitrate concentrations were less pronounced (Table 3; Jeffries 1962). These patterns indicated that the Raritan River was the major source of nitrate entering the Bay (Jeffries 1962). In this study, nitrate concentrations still generally decreased with distance away from the mouth of the Raritan River during spring and summer (Table 3). However, the trend reverses itself during fall when average nitrate concentrations are higher in the Lower Bay (Table 3). In addition, nitrate concentrations are now highest during late summer and early fall months (Tables 2 and 3) rather than during spring as reported by Jeffries. Spatial patterns for SRP concentrations in Raritan Bay are also different from patterns described by Jeffries (1962). In 1957–58, SRP values were maximal at station 6 (Table 3), suggesting that Lower Bay water flooding into Raritan Bay was an important source of P to the system. During this study, however, SRP values were significantly higher at site 2 (all samples and depths considered; $M = 57 \pm 3 \mu\text{g L}^{-1}$) and site 3 ($M = 53 \pm 3 \mu\text{g L}^{-1}$) than at all the other stations ($M = 48 \pm 3 \mu\text{g L}^{-1}$ for site 1, $M = 43 \pm 3 \mu\text{g L}^{-1}$ for site 4, $M = 43 \pm 3 \mu\text{g L}^{-1}$ for site 5, and $M = 40 \pm 3 \mu\text{g L}^{-1}$ for site 6, $p \leq 0.01$ for all pairwise comparisons).

Although not studied by Jeffries in 1958, ammonium and total iron have also been identified as important nutrients in determining overall algal biomass and associated adverse effects of eutrophication (Anderson et al. 2002). Ammonium concentrations in Raritan Bay ranged from <20 (i.e., the detection limit) to $500 \mu\text{g L}^{-1}$ (grand mean for surface waters 88 ± 9 and bottom waters 96 ± 10). Ammonium concentrations peaked during summer months (Table 2) with highest values in the lower third of the water column. Total iron concentrations in Raritan Bay ranged from 40 to $630 \mu\text{g L}^{-1}$ (grand mean, 163 ± 12) and ferrous iron concentrations ranged from <20 (i.e., the detection limit) to $290 \mu\text{g L}^{-1}$ (grand mean, 40 ± 2). Dissolved silicon, which is particularly important for the growth and reproduction of diatoms, ranged from 10 to $19,500 \mu\text{g L}^{-1}$ (grand mean, $3,378 \pm 147$).

Plankton Dynamics Phytoplankton biomass as chlorophyll *a* ranged from 0.1 to $43 \mu\text{g L}^{-1}$ over the 3-year study (grand mean, $6 \pm 0.4 \mu\text{g L}^{-1}$; Table 2). Phytoplankton biomass generally peaked in late spring each year (Table 2). A total of 78 phytoplankton taxa were identified in samples from Raritan Bay. These include 7 cyanobacteria, 34 diatoms, 24 dinoflagellates, 1 raphidophyte, 3 euglenoids, 6 green algae, 1 prasinophyte, and various chrysophytes. Cryptophytes and prymnesiophytes were included as well but were difficult to consistently distinguish beyond class level in preserved samples. Of the 78 phytoplankton taxa identified in Raritan Bay, 14 have been identified as capable of causing harmful algal

Table 3 Comparison of seasonal mean \pm 1SE ($n=12$ for spring and fall and $n=24$ for summer) water temperature, salinity, dissolved oxygen, NO_3^- , and soluble reactive phosphorus (surface samples, 0.5 m) between 1958 (Jeffries 1962) and 2010–2012. Jeffries did not provide seasonal mean values for these parameters at sites 2, 3, and 4. Seasonal categories

are spring, April–May; summer, June–September; and fall, October–November. Water temperature is reported in degrees Celsius; DO is reported in milligrams per liter; NO_3^- and SRP are reported in micrograms per liter

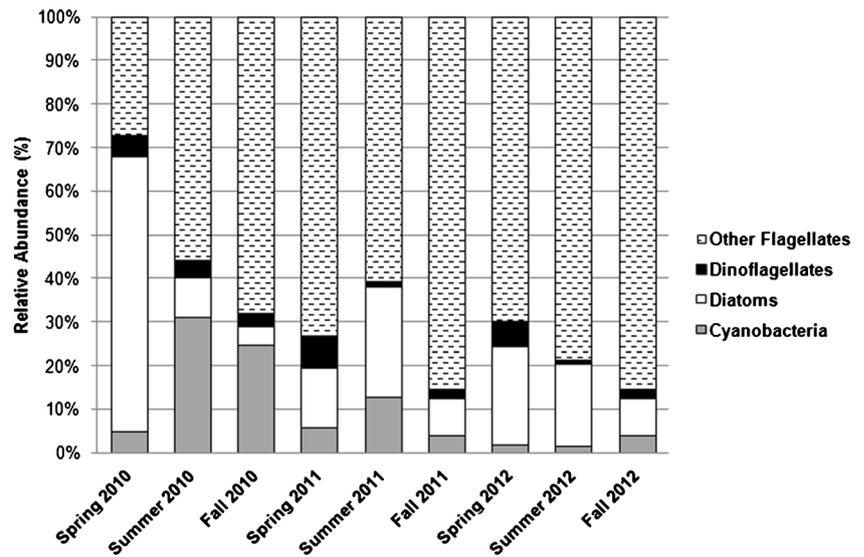
Water temperature									
	1958			2010–2012					
	1	5	6	1	2	3	4	5	6
Spring	14 \pm 6	14 \pm 6	12 \pm 5	15 \pm 1	14 \pm 1	15 \pm 1	14 \pm 1	15 \pm 1	14 \pm 1
Summer	24 \pm 3	23 \pm 3	23 \pm 3	24 \pm 1	24 \pm 1	24 \pm 1	23 \pm 1	23 \pm 1	23 \pm 1
Fall	12 \pm 5	10 \pm 4	11 \pm 4	11 \pm 1	12 \pm 1	11 \pm 1	11 \pm 1	11 \pm 1	11 \pm 1
Salinity									
	1958			2010–2012					
	1	5	6	1	2	3	4	5	6
Spring	12 \pm 7	18 \pm 5	19 \pm 6	21 \pm 3	24 \pm 2	24 \pm 2	27 \pm 3	25 \pm 2	27 \pm 2
Summer	25 \pm 2	26 \pm 1	27 \pm 1	20 \pm 2	22 \pm 2	25 \pm 4	24 \pm 1	26 \pm 4	26 \pm 2
Fall	22 \pm 2	26 \pm 1	27 \pm 1	26 \pm 4	29 \pm 3	31 \pm 2	31 \pm 2	31 \pm 2	31 \pm 1
DO									
	1958			2010–2012					
	1	5	6	1	2	3	4	5	6
Spring	6 \pm 2	8 \pm 2	8 \pm 3	9 \pm 1	10 \pm 1	11 \pm 1	12 \pm 1	12 \pm 1	12 \pm 1
Summer	5 \pm 1	7 \pm 3	8 \pm 4	8 \pm 1	8 \pm 1	9 \pm 1	9 \pm 1	10 \pm 1	11 \pm 1
Fall	4 \pm 1	9 \pm 2	8 \pm 1	10 \pm 1	10 \pm 1	10 \pm 1	11 \pm 1	11 \pm 1	10 \pm 1
NO_3^-									
	1958			2010–2012					
	1	5	6	1	2	3	4	5	6
Spring	96 \pm 42	35 \pm 14	26 \pm 12	711 \pm 367	717 \pm 151	467 \pm 131	667 \pm 304	283 \pm 48	478 \pm 70
Summer	28 \pm 16	14 \pm 9	7 \pm 10	1,000 \pm 262	1017 \pm 290	756 \pm 207	1050 \pm 260	875 \pm 214	500 \pm 107
Fall	N/A	N/A	N/A	800 \pm 158	800 \pm 308	700 \pm 191	1025 \pm 95	825 \pm 239	940 \pm 352
SRP									
	1958			2010–2012					
	1	5	6	1	2	3	4	5	6
Spring	2 \pm 1	2 \pm 1	3 \pm 3	35 \pm 10	43 \pm 11	41 \pm 13	23 \pm 10	33 \pm 15	28 \pm 6
Summer	3 \pm 1	3 \pm 2	3 \pm 2	52 \pm 8	69 \pm 9	64 \pm 8	55 \pm 8	46 \pm 9	49 \pm 9
Fall	4 \pm 2	4 \pm 1	4 \pm 1	55 \pm 9	57 \pm 1	66 \pm 14	53 \pm 8	64 \pm 9	49 \pm 5

DO dissolved oxygen, SRP soluble reactive phosphorus

blooms (i.e., they cause harm either through toxin production, cell physical structure, or accumulated biomass (Anderson et al. 2002). Total cell density ranged from 4×10^2 to 7×10^4 cells ml^{-1} (grand mean, $2 \times 10^4 \pm 1 \times 10^3$ cells ml^{-1}), and species richness ranged from 3 to 24. Overall, flagellates were the most dominant functional group in this system, accounting for $>50\%$ of phytoplankton cells, except for spring 2010 and fall 2012 when diatoms were dominant (Fig. 3). Although dinoflagellates comprised $>30\%$ of the taxa identified in this study, they never accounted for more than 8% of phytoplankton cells (Fig. 3). Dinoflagellates were most abundant and diverse during spring months, and cyanobacteria were most abundant during summer and early fall (Fig. 3), particularly along the southern shore (sites 1, 2, 3, and 5).

A total of 40 zooplankton taxa were identified in plankton tows from the Raritan Bay. Total density of zooplankton ranged from 1 to 2×10^3 organisms L^{-1} (grand mean, 1×10^2 organisms L^{-1}), and species richness ranged from 2 to 12 species. Both zooplankton abundance and species richness was greatest in late spring and early summer. Relative zooplankton abundances varied seasonally and from year-to-year; however, rotifers, copepods, and copepod nauplii generally comprised the greatest proportion of the zooplankton taxa identified. *Mnemiopsis leidyi* was most abundant in July (i.e., mean, 0.3 individuals L^{-1}) at sites 3, 4, and 5. The population of *M. leidyi* was in decline (i.e., mean, 0.1 individuals L^{-1}) by September and virtually absent from samples in October and November.

Fig. 3 Relative abundance (calculated from mean cell number, $n = 12$ for spring and fall and $n = 24$ for summer) of dinoflagellates, diatoms, cyanobacteria, and other flagellates in Raritan Bay from spring 2010 through fall 2012



NMDS of phytoplankton species abundance also indicates that species composition varied seasonally (Fig. 4). As evidenced by the clear separation of spring samples from summer and fall samples in the ordination diagram (Fig. 4), spring phytoplankton assemblages in Raritan Bay have a distinct species composition. A variety of diatoms and dinoflagellates, particularly *H. rotundata* and *Heterocapsa triquetra* (Ehrenberg) Stein (Fig. 4b), were abundant during spring months when precipitation, river discharge, and Si/N ratios were high (Fig. 4a). The copepods, *Acartia tonsa* Dana, *Oithona* spp., and *Eurytemora affinis* Poppe, were also more abundant during spring in Raritan Bay (Fig. 4c). Lower river discharge rates and higher temperature, salinity, nitrate, and SRP concentrations were associated with increased abundance

of flagellates, including green flagellates, cryptomonads, some dinoflagellates (i.e., *Prorocentrum minimum* (Pavillard) Schiller, *Pfiesteria*-like dinoflagellates), and raphidophytes (i.e., *H. akashiwo*, Fig. 4a, b). Secchi depth was greatest in fall when river discharge and phytoplankton cell number and biomass were lower (Fig. 4a).

NMDS of phytoplankton species abundance also indicated that species composition varied from year-to-year (Fig. 5), and these year-to-year differences were even more pronounced when separate ordinations were run with samples from a single season (Figs. 6, 7, and 8). As mentioned, flagellates, particularly chlamydomonad-like species and cryptomonads, were always abundant in Raritan Bay, but year-to-year variations in river flow were often highly correlated with shifts in

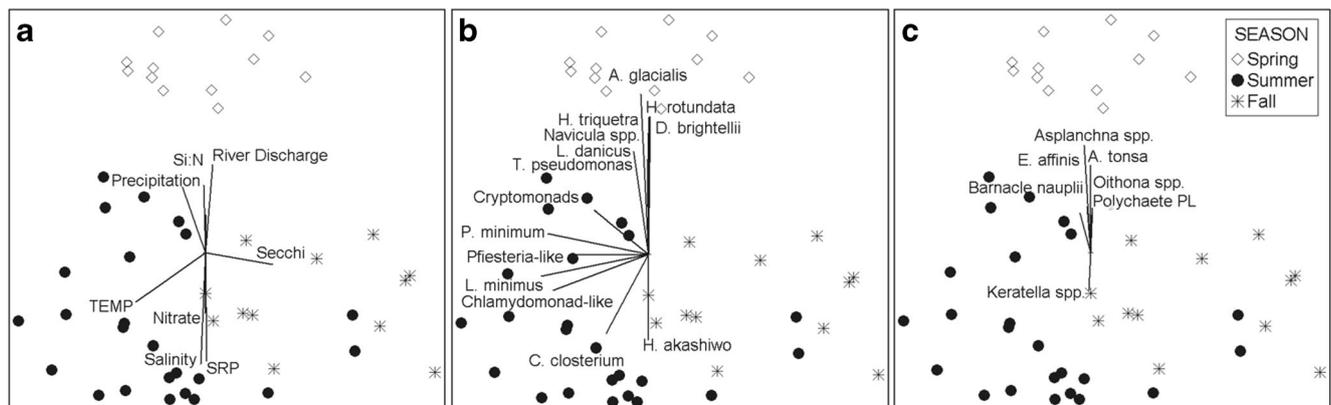


Fig. 4 Ordination of all samples by phytoplankton species composition showing seasonal differences in environmental conditions, phytoplankton species composition, and zooplankton species composition. In this set of diagrams, data from the 3 years of sampling was averaged so that each point in the diagram represents monthly mean abundance ($n = 3$) of all phytoplankton species from a given site and month. **a** Vectors indicate strength and direction of environmental gradients (r^2 cutoff value=0.15; Si/N dissolved Si/N ratio, TEMP water temperature, SRP soluble reactive

phosphorus, SECCHI Secchi depth). **b** Vectors indicate strength and direction of phytoplankton species gradients (r^2 cutoff value=0.40). **c** Vectors indicate strength and direction of zooplankton species gradients (r^2 cutoff value=0.20). Abbreviated taxa are *Ditylum brightwellii*, *Heterocapsa rotundata*, *Asterionellopsis glacialis*, *Heterocapsa triquetra*, *Leptocylindrus danicus*, *Thalassiosira pseudonana*, *Prorocentrum minimum*, *Leptocylindrus minimus*, *Cylindrotheca closterium*, *Heterosigma akashiwo*, *Eurytemora affinis*, and *Acartia tonsa*

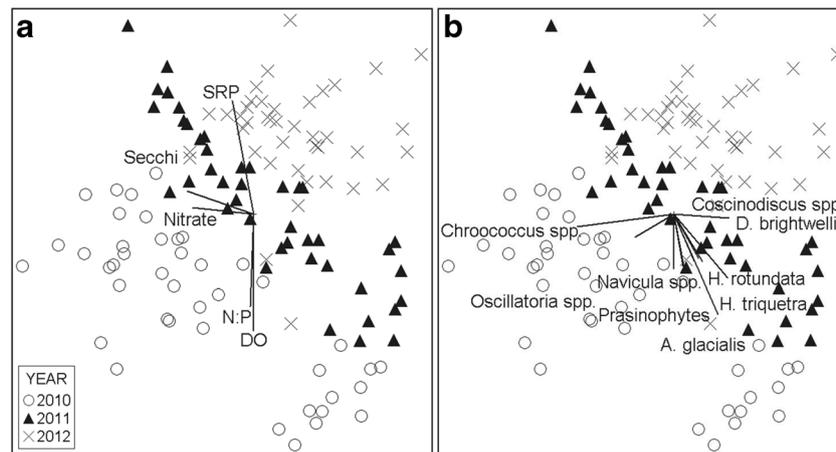


Fig. 5 Ordination of all samples by phytoplankton species composition showing year-to-year differences in environmental conditions and phytoplankton species composition. Each point represents a single sample from one of the six sites in a particular season and year (April 2010–October 2012). The relative distance between samples reflects relative similarity in

phytoplankton species composition. **a** Vectors indicate strength and direction of environmental gradients (r^2 cutoff value=0.10; *SRP* soluble reactive phosphorus, *Secchi* Secchi depth, *N:P* dissolved N/P ratio, *DO* dissolved oxygen). **b** Vectors indicate strength and direction of phytoplankton species gradients (r^2 cutoff value=0.40)

phytoplankton composition. When winter and spring river discharge rates, oxygen concentrations, and N/P ratios were higher in 2010 and 2011, diatoms and *Heterocapsa* spp. reached greater densities in Raritan Bay during April and May (Figs. 6a, b). During 2010 and 2011, higher spring river discharge rates lead to lower salinity and water temperature and higher N/P ratio during summer and fall (Figs. 7a and 8a) and a summer/fall phytoplankton assemblage composed of cyanobacteria (*Chroococcus* spp., *Oscillatoria* spp.) and certain

dinoflagellates (*P. minimum*) and diatoms (*Coscinodiscus* spp., *Chaetoceros* spp.; Figs. 7b and 8b). Rotifers (i.e., *Asplanchna*, *Keratella*, *Brachionus*, and *Synchaeta*) and barnacle nauplii were also more abundant in Raritan Bay during the first 2 years when spring freshwater discharge was high and salinity was lower (Figs. 7 and 8).

In contrast, lower winter and spring river discharge rates lead to higher salinity and SRP concentration and lower N/P ratio in Raritan Bay (Fig. 6a). These spring

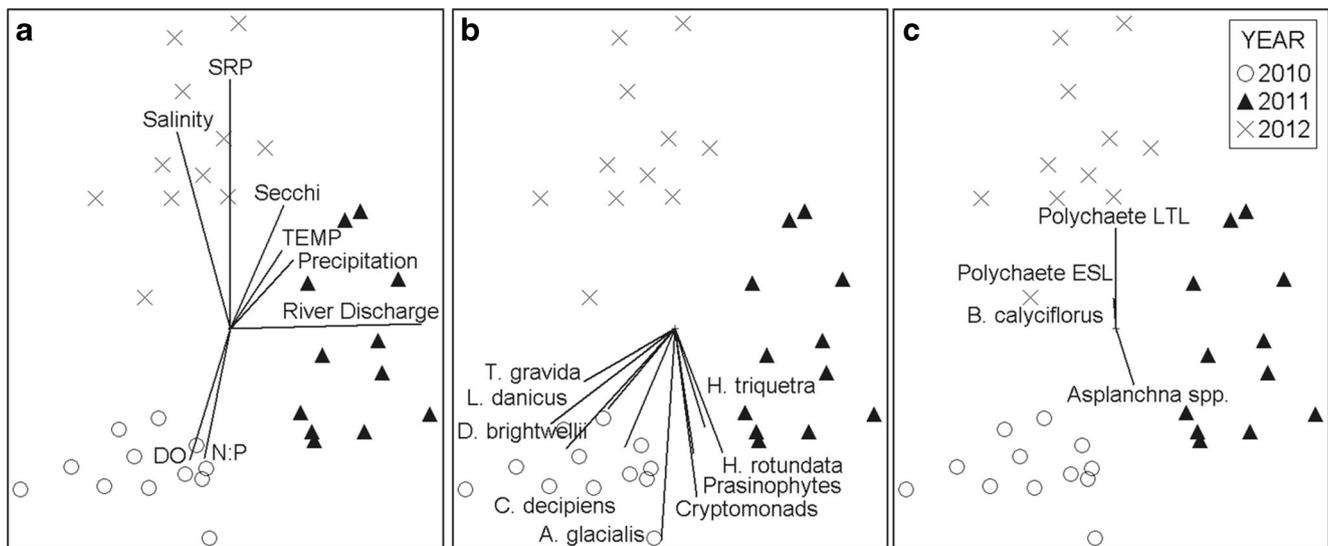


Fig. 6 Ordination of spring samples by phytoplankton species composition showing yearly differences in environmental conditions, phytoplankton species composition, and zooplankton species composition. Each point represents a single spring sample (i.e., April and May) from one of the six sites in a particular year (2010–2012). The relative distance between samples reflects relative similarity in phytoplankton species composition. **a** Vectors indicate strength and direction of environmental gradients (r^2 cutoff value=0.10; *SRP* soluble reactive phosphorus,

DO dissolved oxygen, *N:P* dissolved N/P ratio, *TEMP* temperature, *Secchi* Secchi depth). **b** Vectors indicate strength and direction of phytoplankton species gradients (r^2 cutoff value=0.40). **c** Vectors indicate strength and direction of zooplankton species gradients (r^2 cutoff value=0.20). Additional abbreviated taxa are *Thalassiosira gravida*, *Chaetoceros decipiens*, polychaete late trochophore larvae, polychaete early segmented larvae, *Brachionus calyciflorus*, and polychaete presettlement larvae

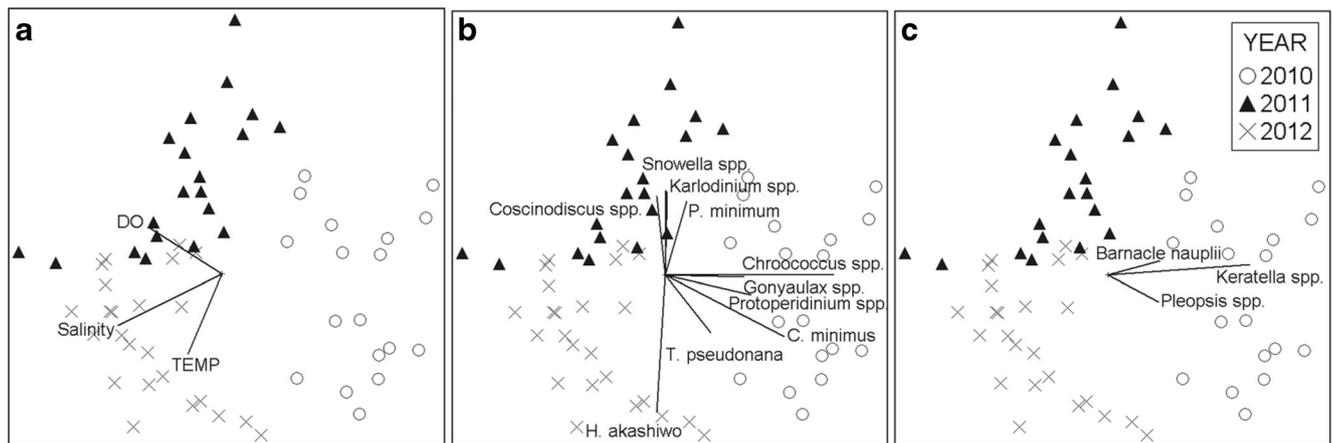


Fig. 7 Ordination of summer samples by phytoplankton species composition showing yearly differences in environmental conditions, phytoplankton species composition, and zooplankton species composition. Each point represents a single summer sample (i.e., June–September) from one of the six sites in a particular year (2010–2012). The relative distance between samples reflects relative similarity in phytoplankton

species composition. **a** Vectors indicate strength and direction of environmental gradients (r^2 cutoff value=0.20; DO dissolved oxygen, TEMP water temperature). **b** Vectors indicate strength and direction of phytoplankton species gradients (r^2 cutoff value=0.40). **c** Vectors indicate strength and direction of zooplankton species gradients (r^2 cutoff value=0.20). Additional abbreviated taxa are *Chaetoceros minimus*

conditions were associated with relatively high biomass (mean, 2×10^2 organisms L^{-1}) and diversity (>6 taxa) of zooplankton as well as a greater number of polychaete larvae. In addition, 2012 was characterized by increased abundance of flagellates, especially the raphidophyte *H. akashiwo*, which reached its maximum density in July 2012 (Fig. 7b). During summer 2012, total zooplankton abundance was significantly lower ($M=79 \pm 15$ organisms L^{-1}) than total abundance during

summer of 2010 ($M=1 \times 10^2 \pm 19$ organisms L^{-1} , $p=0.01$) and summer of 2011 ($M=2 \times 10^2 \pm 93$ organisms per L^{-1} , $p=0.05$). When large amounts of precipitation fell in fall 2012, high river discharge rates and higher Si/N ratios in the bay (Fig. 8a) were associated with declining *H. akashiwo* cell numbers and increasing numbers of the diatoms, *Chaetoceros decipiens* P.T. Cleve, *Coscinodiscus* spp., *Cylindrotheca closterium* (Ehrenberg) Reiman, and *Rhizosolenia* spp. (Fig. 8b). In fact,

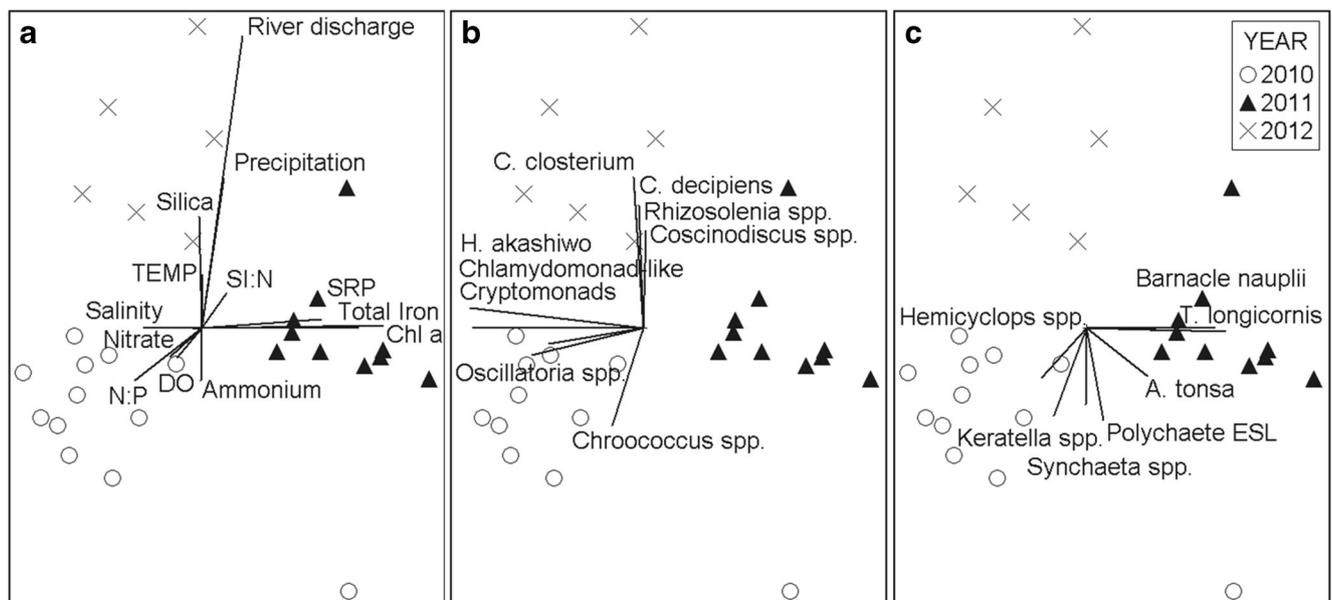


Fig. 8 Ordination of fall samples by phytoplankton species composition showing yearly differences in environmental conditions, phytoplankton species composition, and zooplankton species composition. Each point represents a single fall sample (i.e., October and November) from one of the six sites in a particular year (2010–2012). The relative distance between samples reflects relative similarity in phytoplankton species composition. **a** Vectors indicate strength and

direction of environmental gradients (r^2 cutoff value=0.10; TEMP temperature, N/P dissolved N/P ratio, Chl a chlorophyll a, SRP soluble reactive phosphorus, Si/N dissolved Si/N ratio). **b** Vectors indicate strength and direction of phytoplankton species gradients (r^2 cutoff value=0.40). **c** Vectors indicate strength and direction of zooplankton species gradients (r^2 cutoff value=0.20). Additional abbreviated taxa are *Temora longicornis*

diatoms comprised more than 50 % of the phytoplankton cells in October 2012 (Fig. 3). The zooplankton assemblage in fall 2012 was composed almost entirely of copepod nauplii and adults (i.e., *A. tonsa* and *E. affinis*).

Discussion

The Raritan Bay is both ecologically and economically significant, supporting numerous recreational and commercial fisheries and providing habitat for waterfowl, shellfish, and marine, estuarine, and anadromous fish (Kane and Kerlinger 1994). However, the water quality and ecological integrity of this system has long been threatened by a dense human population and urban and industrial overdevelopment, and full utilization of these waters for commercial fishing and recreational activities is restricted by poor water quality. Our multiyear study of Raritan Bay indicates that this system continues to exhibit numerous symptoms of eutrophication, including high algal biomass, high turbidity, seasonal hypoxia and violations of the dissolved oxygen standard to protect the health of bottom-dwelling fishes, and blooms of potentially harmful phytoplankton species.

The continued nutrient enrichment of this system likely reflects the numerous municipal discharges to the bay and the sensitivity of water quality variables to alterations in land use. A number of studies in the Raritan Bay and Hudson River estuary have demonstrated some improvements to water quality in this system such as increases in DO and decreases in fecal coliforms, biological oxygen demand (BOD), and total suspended solids (TSS) following passage of the Clean Water Act in 1972 and improvements to sewage treatment (Brosnan and O'Shea 1996; O'Shea and Brosnan 2000). However, efforts to decrease nutrient loading by constructing and upgrading sewage treatment plants throughout the RRB are likely being offset by human population growth (i.e., the RRB now contains $\sim 1 \times 10^6$ people; Shallcross et al. 2002) and a history of high-nutrient loading (Jeffries 1962). In addition, urban land cover and municipal discharges are concentrated in the lower Raritan River Basin in Middlesex County around the mouth of the Raritan River (data to be presented elsewhere), and sewer overflows and the discharge of raw sewage and street debris following storms continues to be a major pollution problem in this system (EPA 2007).

Human activity has profoundly changed the landscape in the RRB over the past few decades. The construction of shore management structures, highways, and landfills and the discharge of industrial waste and oil have all contributed to the growing pollution in the bay (EPA 2007). The RRB is now 36 % urban, and most of the housing development and conversion to urban land use occurred between 1986 and 1995 (Shallcross et al. 2002). The construction of housing and roads along with runoff from the suburban landscape and storm

drains are thought to be major sources of nonpoint nutrients and pollutants to Raritan Bay (Shallcross et al. 2002). Much like other coastal river basins (Howarth et al. 1996; Carpenter et al. 1998), N and P loads to the Raritan River are contributed primarily by nonpermitted sources, such as urban land cover and septic tanks, particularly during median and high-flow periods (Reiser 2004). Yet, relatively little attention has been given to controlling nonpoint sources of pollutants. Water quality monitoring studies in other systems have also demonstrated that targeted goals to reduce nutrient loading are not realized because the importance of controlling nonpoint loading has been overlooked (e.g., NRC 2000, Burkholder et al. 2006).

Considering the magnitude of the nutrient loads to this system, the increase in nutrient concentrations in Raritan Bay over the last 50 years is not all that surprising. However, ambient nutrient concentrations and ratios are key variables supporting phytoplankton assemblage shifts and the formation of HABs (Hecky and Kilham 1988).

Analysis of the detailed, 3-year dataset from this study indicates that climatic conditions and nutrient concentrations and ratios interacted to influence the origin and succession and composition of phytoplankton blooms in Raritan Bay. The abundance of flagellates in Raritan Bay follows the general trend observed in other eutrophic systems where increased nutrient enrichment promotes overall shifts in dominance from diatoms to flagellates (Burkholder 2000). Seasonal changes in the composition of both phytoplankton and zooplankton were mostly consistent with descriptions from previous studies of Raritan Bay and other coastal systems of New Jersey (Jeffries 1964; Sage and Herman 1972; Olsen and Mahoney 2001; Gastrich 2000, NJDEP Marine Water Monitoring 2013). As in other systems (e.g., Cloern et al. 1983; Howarth et al. 2000; Arhonditsis et al. 2007), hydrologic forcing (i.e., changes in river discharge) is an important factor influencing water quality and seasonal shifts in plankton composition in Raritan Bay. Increases in river discharge and Si/N ratios in spring were associated with increases in faster-growing diatom taxa, which take up dissolved silica and use it to construct their cell walls (frustules). As phytoplankton assemblages became dominated by diatoms during spring months, the relative abundance of copepods also increased, particularly during the first 2 years of the study. Many near-shore calanoid copepods (e.g., *A. tonsa*) produce resting eggs that sink to the bottom and remain viable until increasing temperatures induce hatching and initiate the next generation (Johnson and Allen 2005). The increase in copepods that generally occurs several weeks after the spring diatom bloom is considered a classic characteristic of coastal food webs (Johnson and Allen 2005). The late spring and early summer maxima in total zooplankton density recorded in this study was also reported for Raritan Bay by Jeffries (1964) and by Sage and Herman (1972).

Spring plankton assemblages generally give way to summer/fall assemblages with a greater relative abundance of small flagellates, cyanobacteria, and microzooplankton such as copepod nauplii, rotifers (e.g., *Keratella* spp.), and ciliates (e.g., tintinnids). Because they can regulate their vertical position in the water column and have a lower demand for silica (Sommer 1994), flagellates and cyanobacteria gain the competitive advantage over diatoms during summer when decreasing freshwater discharge rates result in water column stratification and N and P concentrations increase relative to Si. Certain dinoflagellates such as *Pfiesteria*-like species were also more abundant during summer in association with higher salinity, nitrate, and SRP concentrations. Studies in other systems have also documented increases in *Pfiesteria*-like dinoflagellates in association with high salinities and P concentrations during summer (Rothenberger et al. 2009). These field observations about potentially toxic *Pfiesteria* spp. are supported by laboratory experiments that have shown that increased cell production of *P. piscicida* and/or *P. shumwayae* has occurred under phosphate or N enrichment (Burkholder and Glasgow 1997; Glibert et al. 2006). The greater relative abundance of microzooplankton, particularly copepod nauplii, during summer also corresponds with previous studies of zooplankton annual cycles in the New York Bight (e.g., Turner 1982).

This study also documented a number of phytoplankton blooms and the presence of several potentially harmful species. Recurring algal blooms associated with hypoxia and other negative ecological impacts have been documented in New Jersey coastal waters for the past three decades (Mahoney and Steimle 1979; Gastrich 2000). Past phytoplankton surveys have indicated that the most common bloom formers in this area are the dinoflagellates, *Ceratium tripos*, *Prorocentrum* spp., and *Heterocapsa* spp. and the raphidophycean species *H. akashiwo* (formerly misidentified as *Olisthodiscus luteus*; Gastrich 2000). This study indicated that these species are still present and seasonally abundant in Raritan Bay. NMDS of phytoplankton species abundance indicated that the majority of these bloom-forming species are most abundant during summer months, except for *Heterocapsa* spp., which were most abundant during early spring over the last 3 years. The growth rate of *Heterocapsa* seems to be constrained by low water temperatures to a lesser degree than other phytoplankton species (Mallin et al. 1991; Litaker et al. 2002; Rothenberger et al. 2009; Baek et al. 2011).

The 14 potentially harmful species identified in Raritan Bay as part of this study were mostly present at low densities (i.e., $<2 \times 10^3$ cells ml^{-1} for diatoms and dinoflagellates and $<2 \times 10^4$ cells ml^{-1} for cyanobacteria). For example, subbloom concentrations of the potentially toxic diatom *Pseudo-nitzschia* spp. (associated with amnesic shellfish poisoning in humans) were found at site 6 near Sandy Hook

(density of 2×10^3 cells ml^{-1}) in October 2012. The NJDEP's Bureau of Marine Monitoring screens for *Pseudo-nitzschia* spp., along with several other potentially toxic species, to ensure that shellfish harvested in NJ waters are not toxic for consumption. Prior to 2000, the NJDEP reported that *Pseudo-nitzschia* spp. was abundant in NJ coastal waters during fall and winter months, but over the past 15 years, *Pseudo-nitzschia* spp. has only been reported by the NJDEP near Sandy Hook (i.e., site 6 of this study) one other time and at a concentration insufficient to pose a threat to human health (NJDEP 2013). Although *Pseudo-nitzschia* spp. were present in samples too rarely during this study to identify patterns in their abundance to environmental gradients, continued monitoring for *Pseudo-nitzschia* spp. throughout the year and the use of additional identification techniques to distinguish between toxic and nontoxic representatives of this genus are recommended, especially considering that there has been an increase in their abundance in other coastal systems in association with increased nutrient loading (Parsons et al. 2002; Rothenberger et al. 2009).

The largest monospecific blooms that occurred during this study were caused by *H. akashiwo*. This species reached high densities throughout Raritan Bay beginning in June 2012 with densities exceeding 3×10^4 cells ml^{-1} at site 1 (i.e., *H. akashiwo* cells accounted for >90 % of the total cell number in samples) and becoming less dense with distance away from the mouth of the Raritan River. *H. akashiwo* persisted at high densities in Raritan Bay, particularly at site 1 (highest density $\sim 5 \times 10^4$ cells ml^{-1} in July), for the remainder of the 2012 sampling year and was among the most dominant species in spring, summer, and fall 2012. This species, formerly called *O. luteus*, has been considered a dominant species, often forming late spring/early summer blooms in association with anthropogenic nutrient enrichment, in New Jersey coastal waters since the 1960s (Mahoney and McLaughlin 1977; Gastrich 2000). Summer blooms of *H. akashiwo* and the dinoflagellates, *Prorocentrum* spp. and *Heterocapsa rotundata*, have long been associated with hypoxia and fish kills in the Hudson–Raritan estuary (Olsen and Mulcahy 1991). As mentioned, oxygen concentrations in bottom water at site 1 (<5 mg L^{-1}) violated New Jersey standards for fish health in August 2012 as the *H. akashiwo* bloom was beginning to decompose.

NMDS analyses indicated that *H. akashiwo* abundance was associated with warmer temperatures, decreased river discharge, higher salinity, higher nitrate and SRP concentrations, and reduced total zooplankton abundance. Year-to-year variations in *H. akashiwo* abundance in Raritan Bay suggest that both winter/spring river discharge rates and water temperature are interacting to determine the density of this species. Previous studies have shown that cysts, both vegetative cells suspended in the water column and benthic cysts, play an

important role in the survival and abundance of *H. akashiwo* during summer seasons (Imai and Itakura 1999). Germination of *H. akashiwo* cysts is optimal at 15 °C, and population growth is greatest when surface water temperatures reach 18 °C (Imai and Itakura 1999). During 2010 and 2011, average water temperatures in the bay did not reach 15 °C until May. By this time, high winter/spring precipitation and river discharge rates created conditions more favorable to diatoms and lowered the competitive ability of *H. akashiwo*. High river discharge may also play a role in flushing vegetative cysts out of the system in spring. In 2012, however, the combination of lowered winter/spring river discharge and earlier onset of optimal germination temperature (water temperatures reached 15 °C in April 2012) increased the rate of *H. akashiwo* excystment and population growth. After this initial growth period, the allelopathic activity of *H. akashiwo* against competing phytoplankton and ability to migrate within the water column to obtain nutrients likely played a role in maintaining the bloom (Smayda 1998; Edvardsen and Imai 2006).

The inverse relationship between *H. akashiwo* and total zooplankton abundance indicated by NMDS may suggest that the *H. akashiwo* bloom that formed in late spring 2012 throughout Raritan Bay produced adverse effects for some zooplankters (Turner et al. 1998). A number of laboratory studies aimed at quantifying deleterious effects of *H. akashiwo* on zooplankton grazers have been conducted (e.g., Tomas and Deason 1981; Verity and Stoecker 1982; Clough and Strom 2005; Graham and Strom 2010; Yu et al. 2010). These studies have generally shown that *H. akashiwo* is either avoided as a food item when part of a mixed prey assemblage, or if ingested, decreases growth rate and increases the mortality rate of a number of protist grazers (e.g., *Acartia tonsa*, tintinnids, and oligotrichs). Interestingly, Verity and Stoecker (1982) reported that *H. akashiwo* was lethal to tintinnids at concentrations above 5×10^3 cells ml^{-1} in multi-algal treatments. In this study, *H. akashiwo* densities approached 5×10^4 cells ml^{-1} in July 2012 at the same time that tintinnids were the most abundant group of zooplankton in the bay. However, by August, no tintinnids were found in samples, and copepod nauplii dominated. It is possible that the seemingly delayed effect of *H. akashiwo* on tintinnids was due to tintinnid selective feeding on nontoxic prey. Graham and Strom (2010) suggested that preferential feeding on alternate prey can reduce toxic effects of *H. akashiwo* on microzooplankton. Selective zooplankton grazing in favor of potential phytoplankton competitor species may have contributed to the persistence of the *H. akashiwo* bloom and is worth further investigation. At the end of this 3-year study, fall storms lead to increased river discharge rates and the decline of the *H. akashiwo* bloom throughout the bay. Diatoms regained dominance, and relative abundance of adult copepods increased once again.

Similar to Jeffries (1962) assessment of Raritan Bay water quality more than 50 year ago, the detailed 3-year dataset from this study indicates that this system is still rich in plant nutrients arising from natural and domestic sources and capable of supporting extremely dense plankton populations. However, this study suggests that there have been some changes in water quality since 1958. As mentioned, water column DO concentrations throughout Raritan Bay appear to be higher now than they were 50 years ago. Salinity also appears to be higher throughout the bay during spring months. Increased salinization in spring when precipitation and river discharge are high could be related to increases in impervious surface coverage in the RRB and the use of deicer salt on roadways (Kaushal et al. 2005).

In addition, N and P concentrations have increased in Raritan Bay, and spatial and temporal patterns for these nutrients are different from that reported by Jeffries. For example, current trends indicate that, although the Raritan River is still an important source of N during spring and episodic increases in river discharge in association with storms and hurricanes, recycled sources of N become important during late summer and fall when river discharge is lower. Other studies have indicated that N regeneration from the sediments during seasonal periods of oxygen depletion in the lower water column can play an important role in nutrient loading in eutrophic systems (Seitzinger and Nixon 1985; Howarth et al. 1996). Nitrate concentrations also appear to have increased to a greater extent in the Lower Bay near site 6, particularly during low-flow periods. The Lower Bay receives riverine flow from the Hudson River, which has one of the highest N inputs per area of estuary in the USA (Bricker et al. 1999). In contrast to nitrate, SRP concentrations appear to have increased to a greater extent at the head of Raritan Bay (near the confluence of the Raritan River) when compared with Jeffries findings. A suspected major source of soluble P to the Bay, especially during periods of low flow, is the Middlesex County wastewater treatment plant (WWTP), which is the second-largest WWTP in the state of New Jersey (i.e., treats ~110 million gallons per day [mgd]) and has its outfall located at site 2. During this study, SRP values were significantly higher at sites 2 and 3 than at all the other stations.

In order to understand the full magnitude of human influence on the functioning and stability of coastal ecosystems, it is important to intensify study of the coastal eutrophication problem and to generate ecological data in multiple systems (Cloern 2001). The 3-year dataset from this study contributes new information about how climatic factors and nutrients may be affecting the abundance and distribution of phytoplankton taxa, including some potentially toxic and bloom-forming species as well as how phytoplankton are related to associated food webs. The ordination results of this investigation have also identified several important questions for future research directions. First, blooms of *H. akashiwo* were associated with

increases in both nitrate and SRP in Raritan Bay but not with iron concentrations as indicated in other systems (Yamochi 1983). However, it is still unclear how altered nutrient supply ratios are affecting phytoplankton composition in Raritan Bay. As a result, it will be important to supplement field observations with controlled experiments to investigate the relationship between specific forms and supplies of nutrients and bloom formation. Another question regards the importance of selective zooplankton grazing in contributing to the persistence of algal blooms (e.g., *H. akashiwo*) in Raritan Bay. How might manipulations of zooplankton abundance affect phytoplankton biomass and species composition? One final question concerns the impact of high cell densities of potentially toxic phytoplankton species such as *H. akashiwo* on the health of reintroduced eastern oysters (*Crassostrea virginica*). Although *C. virginica* is ecologically extinct in Raritan Bay, the New York/New Jersey Baykeeper has been conducting studies to determine whether existing conditions in Raritan Bay can support long-term oyster survival (Ravit et al. 2012). Research in other systems has indicated that even short-term exposures of oysters to *H. akashiwo* blooms could have long-term physiological effects (e.g., Keppler et al. 2005). Since this study has indicated that *H. akashiwo* can form large monospecific blooms in this system, more research on the lethal and sublethal effects of HABs on *C. virginica* is needed in order to develop a productive long-term strategy for reestablishing this species into Raritan Bay.

Long-term monitoring (i.e., >10 years) of coastal ecosystem processes (e.g., climatic factors, watershed land use change, water quality) and community characteristics (plankton biomass and species composition, food web structure) are vital for improving our understanding of the mechanisms involved in the development of eutrophication. As a result, we intend to continue collection of data on water quality and plankton assemblages. With further long-term observations, greater confidence in the processes that lead to altered phytoplankton composition and food web structure will be realized. This information will strengthen the ability of scientists, public health officials, and resource managers to predict the likelihood of phytoplankton blooms and bring us closer to the design of appropriate management strategies to mitigate the impact of cultural eutrophication on coastal ecosystems.

Acknowledgments We thank Captain Mick Trzaska, captain and owner of the CRT II, for transporting us to our sampling sites and for technical support in the field. We thank Andrew Chun for Geographic Information Systems (GIS) support, Phil Auerbach for technical support in the laboratory and with field equipment, Penn State Analytical laboratories for total iron analysis, and Jeffrey Hollander, Raphael Cuomo, Danielle Sobol, Carly Feiro, Shane Foye, Alexander Pong, Ryan Hughes, Sam Rosen, and Elizabeth Cole for assistance with water sampling and nutrient analyses. We thank two anonymous reviewers for counsel on the manuscript. Funding support for this research was provided by the Lafayette College Department of Biology.

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