



Deoxygenation, Acidification and Warming in Waquoit Bay, USA, and a Shift to Pelagic Dominance

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Abstract

Coastal nutrient pollution, or eutrophication, is commonly linked to anthropogenic influences in terrestrial watersheds, where land-use changes often degrade water quality over time. Due to gradual changes, the management and monitoring of estuarine systems often lag environmental degradation. One example can be found at the Waquoit Bay National Estuarine Research Reserve, where we developed an analysis framework to standardize and analyze long-term trends in water quality and submerged vegetation data from monitoring programs that began in the 1990s. These programs started after the nearly complete loss of historically extensive *Zostera marina* (eelgrass) meadows throughout the estuary. Recently, eelgrass only persisted in small, undeveloped sub-embayments of the estuary, with conservative declines of over 97% in areal coverage. Over the past 2 decades, the average deoxygenation, acidification, and warming were $-24.7 \mu\text{mol O}_2 \text{ kg}^{-1}$ (-11%), $0.006 \mu\text{mol H}^+ \text{ kg}^{-1}$ ($+34\%$), and $1.0 \text{ }^\circ\text{C}$ ($+4\%$), respectively. Along with the loss of eelgrass, there was also a decline in macroalgal biomass over 3 decades, resulting in a system dominated by pelagic metabolism, indicated by a 71% increase in water column chlorophyll *a* concentrations since 2009. This recent increase in phytoplankton biomass, which is highly mobile and transported throughout the estuary by tides, has resulted in recent degradation of isolated embayments despite their lower nutrient loads. This shift toward pelagic dominance in Waquoit Bay may indicate that other eutrophic and warming estuaries may also shift toward pelagic dominance in the future, as the Northeastern US is one of the fastest warming regions across the world.

Keywords Deoxygenation · Acidification · Warming · Eutrophication · Seagrass · Monitoring

Introduction

Coastal estuaries are productive ecosystems which provide numerous services to humans (Costanza et al. 1997) from sequestering carbon (Waycott et al. 2009; Novak et al. 2020) to sustaining commercial fish stocks (Nixon et al. 1986) among many other beneficial functions. However, excess nutrients from leaching septic systems, agricultural runoff, and other anthropogenic sources continue to degrade

the water quality and overall health of estuaries worldwide (Valiela et al. 1992; Bricker et al. 2008). Additionally, warming water temperatures from climate change, particularly in the northeastern USA as seen in Woods Hole, Massachusetts (Nixon et al. 2004) and Narragansett Bay in Rhode Island (Melrose et al. 2009), exacerbate the nitrogen loading problem by further reducing dissolved oxygen levels and accelerating respiration rates (Peckol and Rivers 1996, Bauman and Smith 2018). Previous research points to the combined impacts of eutrophication and climate variables in explaining the changing benthic biogeochemistry of northeastern USA estuaries (Nixon et al. 2009) and reduced benthic metabolism and ecosystem productivity (Fulweiler and Nixon 2009; Foster and Fulweiler 2014).

Seagrass meadows are one of the most threatened ecosystems worldwide with estimated losses of nearly 30% of their historical extent with accelerating losses of about 7% annually (Waycott et al. 2009). Previous research has shown that nitrogen loading from developed watersheds of estuaries

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has led to declines in seagrasses due to prolific algal growth that reduces the light available for seagrass growth (Short and Burdick 1996; Valiela et al. 1997; Burkholder et al. 2007; Costello and Kenworthy 2011; Cebrian et al. 2014). Increased macroalgal growth has also been linked to heightened night-time respiration and low oxygen (hypoxic) events due to intensified organic matter remineralization (Hauxwell et al. 1998; D'Avanzo and Kremer 1994; D'Avanzo et al. 1996; Burkholder et al. 2007; Baumann and Smith 2018). In eutrophic estuaries, the complex interaction of macroalgal growth, epiphytic algal growth, light reduction, hypoxia, sediment resuspension, and other benthic-pelagic biogeochemical feedbacks can lead to the loss of seagrasses as well as ecosystem function (Burkholder et al. 2007; Cebrian et al. 2014). For example, in the well-studied Waquoit Bay on Cape Cod, USA, research has revealed that the increased rates of hypoxia and loss of *Zostera marina* (eelgrass) habitat resulted in declines in bay scallop populations (Taylor and Capuzzo 1983; Costa et al. 1992), a reduction in species richness and abundance of invertebrates (D'Avanzo and Kremer 1994; Hauxwell et al. 1998), reduced fish diversity (Deegan et al. 2002), and the potential for the system to switch from benthic to pelagic dominance (Cebrian et al. 2014).

The range of nitrogen loading across the Waquoit Bay sub-watersheds is comparable to the range of nitrogen loading across 90% of the world's estuaries (Valiela et al. 2021) making it an ideal site for investigating eutrophication impacts. The estuary was historically productive supporting a high diversity and abundance of finfish and shellfish species and has been home to the Mashpee Wampanoag Tribe since before European settlers colonized the area in the seventeenth century (Geist 1996). Although land use in the watershed was dominated by agriculture for approximately 250 years following European immigration (Geist 1996), residential development grew exponentially from the late 1930s to the 1990s (Bowen and Valiela 2001). This rapid increase in the urbanized and residential area resulted in increased nitrogen discharge to the estuary through the groundwater, due to leaching septic systems and fertilizer runoff (Valiela et al. 1992). Since individual home septic systems are the primary form of wastewater management on Cape Cod (Valiela et al. 1997), the variation in residential development density across the sub-watersheds results in varying nitrogen loads with corresponding degrees of degradation in water quality (Valiela et al. 2000).

Nitrogen loading studies in Waquoit Bay asserted that while nutrient loads remain high, primary production would continue to be dominated by macroalgae and that phytoplankton concentrations would always be limited by the tidal residence times (D'Avanzo et al. 1996; Havens et al. 2001; Valiela and Bowen 2002). These conclusions were derived from the fact that there were still small, persistent eelgrass beds in a few of the sub-embayments within the estuary

where nitrogen loads were lower or tidal flushing with Vineyard Sound regularly reintroduced lower-nutrient, highly oxygenated water (Fox et al. 2008; Costello and Kenworthy 2011; Howes et al. 2012). However, more recent research efforts have uncovered several new developments regarding the hydrological and metabolic dynamics of Waquoit Bay, including longer residence times in some parts of the estuary (Tomasky-Holmes et al. 2013) and reduced sediment fluxes and ecosystem productivity levels relative to the 1990s (Foster and Fulweiler 2014). Also, research conducted in the southern New England region suggests that water column chlorophyll *a* concentrations are responding to climate change variables, such as warmer temperatures and increased precipitation (Rheuban et al. 2016; Williamson et al. 2017).

Using long-term monitoring data collected over roughly 3 decades, we investigated the connection between chronic eutrophication and warming temperatures on long-term water quality and changes in eelgrass, macroalgae, and phytoplankton, from two long-term monitoring programs coordinated by the Waquoit Bay National Estuarine Research Reserve (WBNERR). Based on our observations in the field and previous studies, we surmised that the increased thermal stress from warming summer temperatures and higher frequency and duration of heat waves had resulted in (1) eelgrass mortality, (2) a reduction in oxygen concentrations and increased duration and frequency of hypoxic events, and (3) increasing concentrations of phytoplankton throughout the estuary. The presented analyses revealed recent and unexpectedly large increases in chlorophyll *a* concentrations throughout the estuary, which coincided with ongoing decreases in macroalgal density. These results point to a transition from benthic to pelagic dominance that may be the future for other eutrophic estuaries throughout the world due to climate change, as the northeastern USA is one of the fastest warming regions in the world (Walsh et al. 2014; Seidov et al. 2021).

Methods

Study Site

The Waquoit Bay estuary is a shallow, micro-tidal estuary (603 hectares) located on the south side of temperate Cape Cod with a hydrologic connection to Vineyard Sound (Fig. 1). The majority of the tidal area is ~1–2 m deep at low tide with a < 1 m tidal range. The estuary is made of multiple sub-watersheds and -embayments (Fig. 1, Valiela et al. 1992). The western peninsula bordering Childs River and Eel River has high residential density while the eastern area surrounding Hamblin and Jehu Ponds has been less densely developed. Lastly, the sub-watershed of Sage Lot Pond which drains into the southeastern portion of the estuary has been largely protected as part of the Massachusetts State Park system (see

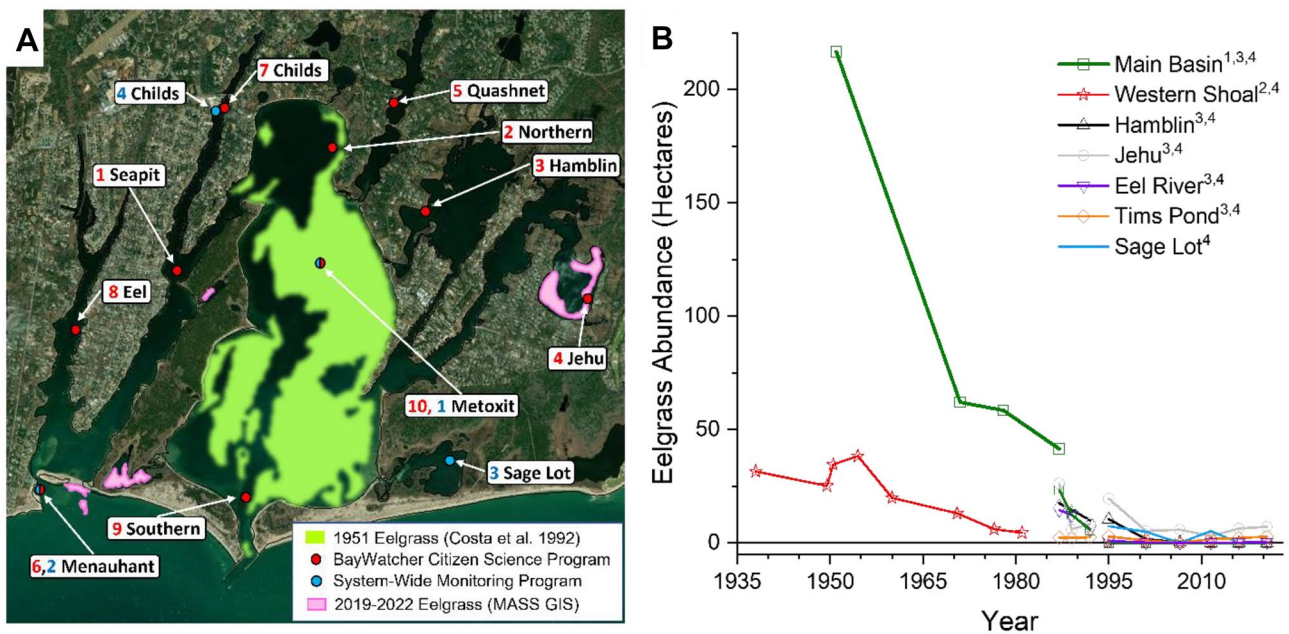


Fig. 1 Map of Waquoit Bay showing locations of the BayWatcher Citizen Science and System-Wide Monitoring Programs run by the Waquoit Bay National Estuarine Research Reserve (A). The green shading (A) represents the 1951 eelgrass extent adapted from Costa et al. (1992), and the pink shading (A) represents the 2019–2022 eelgrass extent determined by the Massachusetts Department of Environmental Protection Eelgrass Mapping Project, both determined from aerial imagery. Imagery in A is from the Massachusetts Bureau of

Geographic Information (MassGIS). The eelgrass abundance (B) was adapted from Costa et al. (1992)¹, Costa (1988)², Short and Burdick (1996)³, and MassGIS⁴ as denoted in the figure legend with superscripts. The Western Shoal represents only the western portion of the main basin, and both Costa (1988) and Costa et al. (1992) did not analyze any of the tributaries or sub-estuaries that are the only places where eelgrass has been detected in the last 30 years by MassGIS

Fig. 1 for satellite imagery). The landscape surrounding Sage Lot Pond consists of mixed-pitch pine forest and back-barrier dunes with little to no influence of septic systems on the sub-watersheds nitrogen load (Howes et al. 2012).

Monitoring Programs

Because of the varying degrees of nitrogen leaching into the groundwater, Waquoit Bay offers an interesting site for comparing nitrogen impacts on coastal water quality and related habitats. In an effort to protect the area from development and promote its research potential, the Waquoit Bay National Estuarine Research Reserve (WBNERR) was established in 1988. The National Estuarine Research Reserve (NERR) System is composed of 30 reserves located across the USA and administered by the National Oceanographic and Atmospheric Administration. As part of the NERR System, water quality and submerged aquatic vegetation (including eelgrass and macroalgae) have been monitored within the Waquoit Bay estuary as core components of the System-Wide Monitoring Program (SWMP). Four sites were selected as long-term monitoring stations: Childs River, Metoxit Point (located in the middle of Waquoit Bay), Sage Lot Pond, and Menauhant (Fig. 1). Metoxit Point and Sage Lot are both open water sites, while Childs River

and Menauhant are dockside sites. Menauhant is located at one of the tidal inlets to the estuary where water conditions are dominated by Vineyard Sound water on rising tides and changes to the size of the inlet opening and location occurred over time (Fig. S1). Menauhant also suffered from interference with macroalgae that led to unreliable chlorophyll *a* data. Menauhant was therefore excluded from our main analyses. A citizen science water quality monitoring program, the Waquoit BayWatchers (hereafter referred to as BayWatchers), was initiated in 1993 by the Waquoit Bay NERR to provide supplemental data for research and monitoring while also engaging the local community. The BayWatchers started with five sites and expanded to ten over the life of the program.

SWMP includes three different monitoring modules: (1) monthly water sampling for nutrient and chlorophyll *a* analyses, (2) continuous 15-min basic water quality data logging, and (3) periodic inter-annual submerged aquatic vegetation and macroalgae (SAV/MAC) surveys. The monthly water samples are collected at 1 m below the water surface within 3 h of low tide and are analyzed for orthophosphate, nitrate, nitrite, ammonium, total dissolved nitrogen, particulate organic nitrogen, particulate organic carbon, and algal pigments (chlorophyll *a* and pheophytin). For the purposes of this study, we will be focusing on the total nitrogen and chlorophyll *a* data from

the monthly sampling dataset. Chlorophyll *a* samples were filtered (Whatman glass fiber filter GF/F, 0.7 μ m pore size) and processed at the Waquoit Bay NERR Laboratory following the EPA standard fluorescence method for marine algae (Arar and Collins 1997). The dissolved inorganic nutrients were filtered using a syringe filter with polypropylene housing (0.45 μ m and 25 mm diameter) and stored frozen at -20°C until processed following EPA standards (EPA No. 841-R-09-002). Total nitrogen was derived from laboratory analyses of total dissolved nitrogen and particulate organic nitrogen and was performed using standard EPA-approved procedures (O'Dell 1993; Zimmermann et al. 1997) by UMASS Boston until 2012 and by the UMASS Dartmouth School for Marine Science and Technology Coastal Systems Program from 2013 to 2020.

The 15-min (or 30-min data in early years) time series data include temperature, salinity, dissolved oxygen (% and mg/L), pH, relative depth (corrected for barometric pressure), turbidity, and chlorophyll fluorescence (the latter implemented in 2009). The NERR SWMP protocols are standardized across all reserves and require frequent calibration and redeployment of YSI sondes (Porter et al. 2004). Sonde models were upgraded from the 6600-series to the EXO-series starting in the summer of 2016 and were all upgraded by the winter of 2017. The sondes are deployed in permanent PVC stations situated such that readings are collected at roughly 25–50 cm above the bottom. Sondes are exchanged with recently calibrated units every 3 to 4 weeks (to prevent data loss due to battery failure and biofouling) for most of the year (typically about April through December) but deployment periods varied throughout the years. Additionally, the sonde data undergo rigorous quality control checks composed of automated primary data review, quarterly and annual secondary review by WBNERR staff, and a final tertiary review by the NERRS Centralized Data Management Office (CDMO). Sonde and nutrient data that fail to meet standard diagnostics are flagged and receive specific codes within the dataset. All the NERR System data are available for download at <http://cdmo.baruch.sc.edu/>.

The SAV/MAC survey, repeated every 3–4 years, was conducted at the Waquoit Bay NERR in September/October of the following years: 2004, 2007, 2011, 2016, and 2020. Due to staff and funding limitations, the number of sites surveyed varied across the sampling years, but several sub-embayments were consistently sampled across all years: Childs River, Quashnet River, Jehu Pond, and Sage Lot Pond. Sampling methods follow those employed by previous studies (Hersh 1996; Fox et al. 2008; Valiela et al. 2016). Benthic grab samples were collected using a Ponar sampler (22.86×22.86 cm). These samples were saltwater rinsed in a bucket with $\frac{1}{4}$ in holes in situ, drained with a salad spinner, and stored cold (4°C) in Ziploc bags until sorted by species and weighed dry (dried 48 h at 60°C). Samples were sorted

and dried within 2 weeks of sampling before decay impacts identification and sample weights.

The BayWatchers water quality data included measurements of temperature, salinity, and dissolved oxygen (in mg/L and %) using a YSI handheld device (2007–2016: YSI 85; 2016–2020: YSI Pro2030) at 25 cm below the water surface. Handheld devices are calibrated for specific conductivity and dissolved oxygen every month by WBNERR staff following YSI calibration standards. Prior to 2007, the BayWatchers collected these parameters using a thermometer, hydrometer, and the Winkler titration method. Measurements were collected every 2 weeks between 6 and 9 am and during low tide from late May to early October. Volunteers also collected a water sample at 25 cm below the water surface which was filtered within 3 h of sampling and used to measure concentrations of total dissolved nitrogen, particulate organic nitrogen, and chlorophyll *a* using the same methods as described above for the SWMP samples. The Woods Hole Oceanographic Institution analyzed the BayWatcher samples until 2014, and starting in 2015, samples were sent to the Center for Coastal Studies for analysis. Data were entered into an electronic database and quality control checked by WBNERR staff. Suspicious or rejected data were removed from the long-term dataset and noted in the metadata records. The BayWatcher data are available for download via the Cape Cod Commission's Water Quality Data Portal: <https://waterquality.capecodcommission.org/#/>.

Data Filtering

Data from the WBNERR SWMP was first filtered by using flags that were supplied with the quality-controlled data downloaded from the CDMO. Descriptions of the quality-control procedures, codes, and standards are outlined in the Quality Assurance Project Plan for Waquoit Bay NERR Water Quality Monitoring Programs (2019), and the only data that were retained had either passed quality checks, were from before quality checks were implemented, or were corrected data. The remaining data were then evaluated for data density on each day of the year (Julian day) to determine a threshold for comparisons across sites and years. The remaining data were then further refined by removing outliers that fell outside of 3 standard deviations from the 2-week running mean for each measured parameter. All data analyses were conducted using MATLAB where all removed data were replaced with NaNs, and the final filtered data were converted into timetables for subsequent analyses. Data from the BayWatcher Citizen Water Quality Monitoring program and the SAV/MAC surveys did not contain data flags but had separate quality-control procedures. These data were not continuous time series and were therefore not able to be filtered using deviations

from the running mean. These programs also evaluated different numbers of sites across the years of each program, and only long-term data from common sites across all years were evaluated (i.e., Fig. 2, Table 1).

During the early years of the WBNERR SWMP, the data sondes were applied throughout most months of the year and would have led to bias in long-term trends due to later years that did not sample through the first 3 months of the year. Therefore, a threshold of 84% was established (representing the minimum coverage of mid-summer data of 16 out of 19 years on any given Julian day) and resulted in continuous data at all 3 sites from Julian day 135 to 288 (see Fig. S2). All data outside of this mid-May to mid-October range were removed from all subsequent analyses, including from the BayWatcher and monthly SWMP water sample data.

Data Analyses

The filtered data were then analyzed using a variety of time steps to evaluate trends. Data were averaged to hourly, daily, weekly, and monthly scales to investigate trends. To estimate daily anomalies, a 2nd order polynomial model was fit to the Julian day average data across all years to estimate the seasonal trends for each parameter, where the anomalies were defined as the deviation from this seasonal model on any given Julian day. The pH data were converted to hydrogen ion (H^+) concentration. Both the H^+ and oxygen data were corrected for temperature and salinity, resulting in concentration units of $\mu\text{mol kg}^{-1}$, based on the methods of Gill (1983). Statistics generated using linear regression models were the slope, coefficient of determination (R^2), and F and p values. Trends with a p value greater than 0.05 were reported as not significant when compared with a constant slope model using analyses of variance tests. The upper 90% and lower 10% thresholds were

also calculated for each parameter, and the duration of time where each daily-averaged parameter was above or below these thresholds was determined. All error estimates are standard deviations unless stated otherwise.

Seagrass and algal biomass data were also extracted from previous studies by (1) digitizing figures and extracting data from Costa (1988), Short and Burdick (1996), Valiela et al. (1997), Deegan et al. (2002), and Fox et al. (2008) using OriginPro software, and (2) importing figures into Google Earth, aligning map figures to estuary boundaries, and tracing historical eelgrass extents using the polygon tool to estimate areas from the eelgrass maps of Costa et al. (1992) (e.g., Fig. 1). Current eelgrass extent data and aerial imagery were directly available from the Massachusetts Department of Environmental Protection Eelgrass Mapping Project and the Massachusetts Bureau of Geographic Information (MassGIS 2022).

Results

Macrophyte Surveys

The decline of eelgrass (*Zostera marina*) in Waquoit Bay is well-established, and eelgrass was only recently found in isolated sub-embayments in less-densely populated areas of Waquoit Bay (Fig. 1). Eelgrass extent has decreased by > 97% from 1951 aerial imagery estimates (i.e., Costa et al. 1992), but this early estimate did not consider the small sub-embayments that represent the last locations that eelgrass has been found in Waquoit Bay. The only location where the most recent (2020) SWMP SAV/MAC survey found physical evidence of eelgrass was in Jehu Pond with a density of $0.23 \text{ g dry weight m}^{-2}$, which represents a 98% decline since 2004 (Fig. 2). Sage Lot lost 100% of its

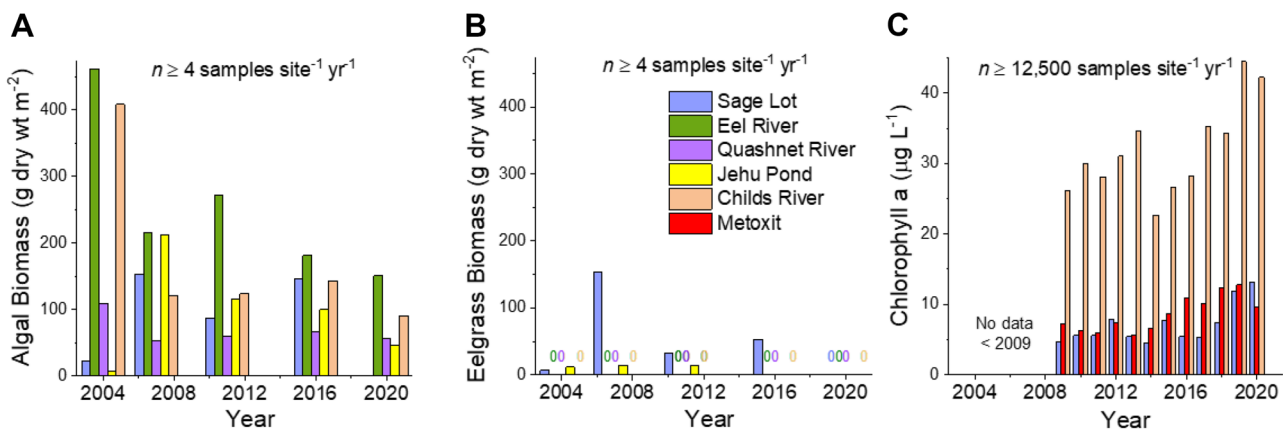


Fig. 2 Algal (A) and eelgrass (B) biomass collected by WBNERR using a Ponar sampler in 2004, 2007, 2011, 2016, and 2020. Only sites that were evaluated in each sampling year were analyzed, and the number of samples evaluated each year varied, but was always > 4 samples

site⁻¹ year⁻¹, and the Metoxit site was not sampled. Zeros indicate that no eelgrass was found, and zero colors correspond to site colors. Chlorophyll *a* was determined with WBNERR data sondes beginning in 2009 (note the much larger number of samples)

Table 1 Waquoit Bay Monitoring Programs, Sites, and Parameters

<i>System-Wide Monitoring Program (SWMP) - Continuous In Situ Sensors and Monthly Water Samples</i>			
Site #	Site Name	Parameters Analyzed	Dates (continuous, monthly)
1	Metoxit	Temperature, O ₂ , pH, salinity, chlorophyll a, nutrients	2001–present
2	Menauhant	Temperature, O ₂ , pH, salinity, chlorophyll a, nutrients	2001–present
3	Sage Lot	Temperature, O ₂ , pH, salinity, chlorophyll a, nutrients	2002–present
4	Childs River	Temperature, O ₂ , pH, salinity, chlorophyll a, nutrients	2002–present
<i>BayWatcher Citizen Water Quality Monitoring Program (BW) - Water Samples and Handheld Sensors</i>			
Site #	Site Name	Parameters Analyzed	Dates (biweekly)
1	Seapit	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1993–present
2	Northern Bay	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1993–present
3	Hamblin	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1993–present
4	Jehu Pond	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1993–present
5	Quashnet	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1993–present
6	Menauhant	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1995–present
7	Childs River	Temperature, O ₂ , salinity, chlorophyll a, nutrients	1996–present
8	Eel River	Temperature, O ₂ , salinity, chlorophyll a, nutrients	2003–present
9	Southern Bay*	Temperature, O ₂ , salinity, chlorophyll a, nutrients	2007–present
10	Metoxit*	Temperature, O ₂ , salinity, chlorophyll a, nutrients	2016–present
<i>Submerged Aquatic Vegetation Surveys (Only Showing Common Sites Through All Years)</i>			
Site #	Site Name	Parameters Analyzed	Dates (Sept/Oct of)
143–170	Childs River	Ponar dredge (aquatic vegetation, macroalgae)	2004, 2007, 2011, 2016, 2020
7–12, 180–184	Jehu Pond	Ponar dredge (aquatic vegetation, macroalgae)	2004, 2007, 2011, 2016, 2020
171–175	Quashnet	Ponar dredge (aquatic vegetation, macroalgae)	2004, 2007, 2011, 2016, 2020
121–135	Eel River	Ponar dredge (aquatic vegetation, macroalgae)	2004, 2007, 2011, 2016, 2020
195–199	Sage Lot	Ponar dredge (aquatic vegetation, macroalgae)	2004, 2007, 2011, 2016, 2020

The * represents sites not analyzed as part of this study due to their short duration

eelgrass since 2004 based on both aerial imagery (Fig. 1) and surveys (Fig. 2). Since 2004, there have also been general declines in macroalgal abundance at the Childs River, Eel River, Quashnet River, and Sage Lot sites revealed by the fall SAV/MAC surveys conducted every 3–4 years (Fig. 2). Combining the SAV/MAC data with historical data from previous studies (Valiela et al. 1997; Deegan et al. 2002; Fox et al. 2008; Cebrian et al. 2014, Valiela 2015–16 pers. comm.) revealed a general decline in macroalgae at the two commonly evaluated sites (Childs River and Sage Lot, Fig. 3). While there is substantial variability in macroalgal biomass at each site through time, over the last 30 years macroalgae have declined by approximately 78 and 57% at Childs River and Sage Lot, respectively (Fig. 3). Due to the high error and sporadic sampling of macroalgae over the last 3 decades further analyses (e.g., anomalies, seasonal patterns) were not conducted.

Chlorophyll *a*

Beginning in 2009, there have been steady increases in the amount of chlorophyll *a* in the water column with increases

of 62%, 34%, and 181% at the Childs River, Metoxit, and Sage Lot sites, respectively (Fig. 2) with an average increase of 71%. All sites with available chlorophyll *a* data from both monitoring programs showed significant increases in chlorophyll *a* (Tables 2 and 3). Since 2009, the SWMP program has shown increases between 4.0% and 7.3% per year (Table 2), and since 1998, the BayWatchers program has shown increases between 2.0 and 4.0% per year (Table 3). The quality-controlled SWMP chlorophyll *a* sonde data and the SWMP monthly water samples were also highly correlated ($R^2 = 0.78$), and the relationship between them was not significantly different from a 1:1 slope line indicating adequate sensor performance over the study period and across sites (Fig. S3).

Temperature

At all sites of the SWMP, there was a significant increase in the mid-May to mid-October temperature that was very consistent between sites at a rate of $0.05\text{ }^{\circ}\text{C y}^{-1}$ or $1.0 \pm 0.0\text{ }^{\circ}\text{C}$ (+4.4%) over the past 20 years (Table 2, Fig. 4, Fig. S4). The BayWatcher data generally showed the same increases

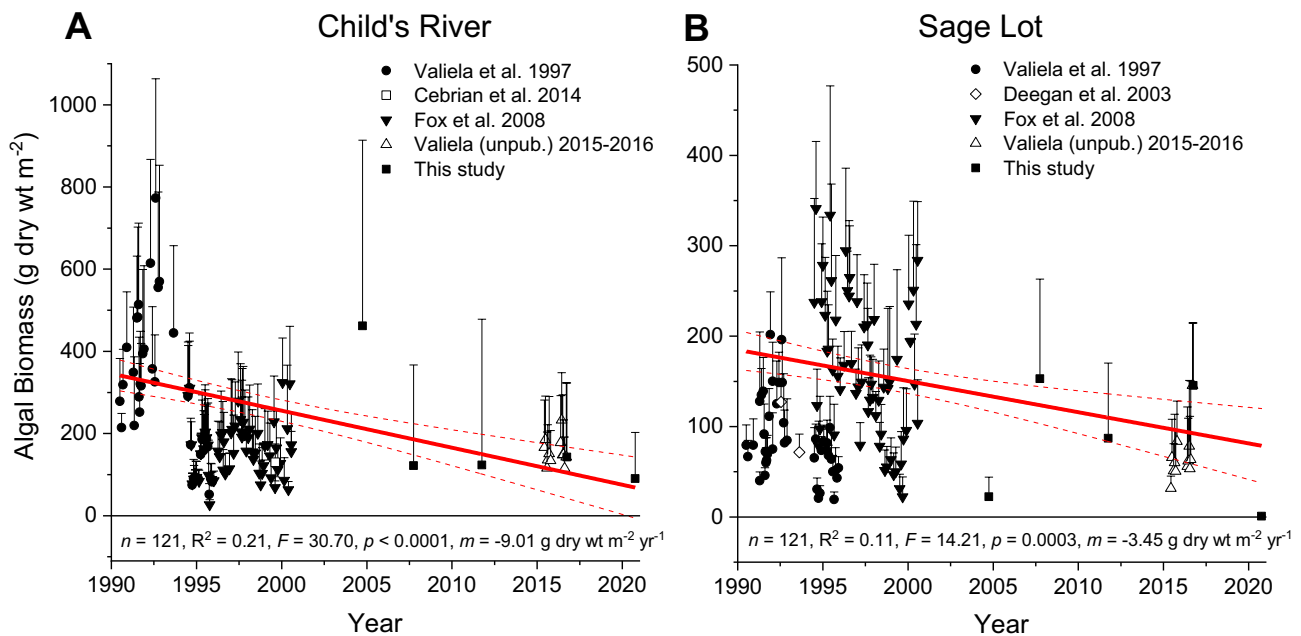


Fig. 3 Macroalgal biomass in Childs River (A) and Sage Lot (B) over the last 30 years. Data were digitized from published figures (Valiela et al. 1997; Deegan et al. 2002; Fox et al. 2008), taken from the text (Cebrian et al. 2014), unpublished (Valiela pers. comm.), or were from this study. Macroalgal data that reported seasonal, annual, or inter-annual means

were not included (e.g., Hersh 1996; Foster and Fulweiler 2014; Valiela et al. 2016). Bold lines indicate the least-squares fit that was weighted by the sample error, dashed lines indicate the 95% confidence interval, and statistics represent ANOVA compared to a constant model

Table 2 System-Wide Monitoring Program (SWMP) - Continuous In Situ Sensor Anomaly Trends

	Site	1st year	Slope				R^2	n	Constant Model ANOVA	
			°C y ⁻¹	μmol kg ⁻¹ y ⁻¹	μg L ⁻¹ y ⁻¹	% y ⁻¹			F	p
Temperature										
	1, Metoxit	2001	0.049			0.22%	0.031	2831	90.862	<0.0001
	2, Menauhant*	2001	0.050			0.24%	0.052	3030	166.425	<0.0001
	3, Sage Lot	2002	0.049			0.22%	0.023	2726	65.034	<0.0001
	4, Childs River	2002	0.049			0.21%	0.030	2811	86.341	<0.0001
Oxygen										
	1, Metoxit	2001		-0.785		-0.32%	0.011	2602	30.181	<0.0001
	2, Menauhant*	2001		0.024		0.01%	0.000	2535	0.168	0.6816
	3, Sage Lot	2002		-1.566		-0.66%	0.078	2094	176.292	<0.0001
	4, Childs River	2002		-1.354		-0.77%	0.015	2008	29.896	<0.0001
[H ⁺]=10 ^{-pH}										
	1, Metoxit	2001		4.98E-05		0.46%	0.003	2753	9.3406	0.0023
	2, Menauhant*	2001		3.62E-05		0.32%	0.005	2781	14.8841	0.0001
	3, Sage Lot	2002		2.72E-04		1.97%	0.061	2535	164.2812	<0.0001
	4, Childs River	2002		5.46E-04		2.09%	0.019	2494	47.3659	<0.0001
Chlorophyll <i>a</i>										
	1, Metoxit	2009			0.442	5.60%	0.185	1096	247.746	<0.0001
	3, Sage Lot	2009			0.482	7.28%	0.189	1172	272.811	<0.0001
	4, Childs River	2009			1.175	3.99%	0.072	1058	82.263	<0.0001

Menauhant* - This site is located at one mouth of Waquoit Bay and the inlet has changed over time (see Fig. S1, 4)

Table 3 BayWatcher Citizen Water Quality Monitoring Program (BW) - Anomaly Trends

Site	1st year	Slope				R^2	n	Constant Model ANOVA	
		$^{\circ}\text{C y}^{-1}$	$\mu\text{mol kg}^{-1} \text{ y}^{-1}$	$\mu\text{g L}^{-1} \text{ y}^{-1}$	$\% \text{ y}^{-1}$			F	p
Temperature									
1, Seapit	1994	0.038			0.22%	0.032	535	17.51028	<0.0001
2, Northern Bay	1994	0.007			0.04%	0.001	529	0.48808	0.4851
3, Hamblin	1994	0.036			0.19%	0.022	420	9.53457	0.0022
4, Jehu Pond	1994	0.057			0.30%	0.048	358	17.92671	<0.0001
5, Quashnet River [^]	1994	0.007			0.05%	0.000	424	0.18956	0.6635
6, Menauhant*	1994	0.013			0.09%	0.005	541	2.74176	0.0983
7, Childs River	1996	0.124			0.79%	0.149	403	69.98499	<0.0001
8, Eel River	2003	0.055			0.35%	0.014	305	4.28924	0.0392
Oxygen									
1, Seapit	1994		−1.493		−0.67%	0.113	527	67.09321	<0.0001
2, Northern Bay	1994		−1.544		−0.75%	0.049	512	26.31451	<0.0001
3, Hamblin	1994		−0.540		−0.30%	0.011	410	4.34091	0.0378
4, Jehu Pond	1994		−1.110		−0.57%	0.036	349	12.95536	0.0004
5, Quashnet River [^]	1994		0.287		0.12%	0.001	412	0.54263	0.4618
6, Menauhant*	1994		−1.697		−0.65%	0.201	532	133.5031	<0.0001
7, Childs River	1996		−0.982		−0.57%	0.018	392	7.00011	0.0085
8, Eel River	2003		−1.544		−0.69%	0.028	295	8.58781	0.0037
Chlorophyll <i>a</i>									
1, Seapit	1998			0.130	2.07%	0.030	177	5.38306	0.0215
2, Northern Bay	1998			0.231	4.13%	0.072	177	13.55442	0.0003
3, Hamblin	1998			0.207	4.03%	0.071	171	13.01052	0.0004
4, Jehu Pond	1998			0.142	3.50%	0.050	164	8.57363	0.0039
5, Quashnet River [^]	1998			0.493	4.08%	0.038	176	6.8636	0.0096
6, Menauhant*	1998			0.065	2.92%	0.077	196	16.16947	0.0001
7, Childs River	1998			0.672	3.64%	0.038	185	7.25828	0.0077
8, Eel River	2003			0.138	2.64%	0.028	149	4.15692	0.0433

Menauhant* - Site is located at one mouth of Waquoit Bay and the inlet has changed over time (Fig. S1, 4)

Quashnet River[^] - Site had a −2.5% year^{−1} decline in salinity due to dam removal (Table S1)

in temperature (Table 3) but there was much more variability in the rates (0.007 to 0.124 $^{\circ}\text{C y}^{-1}$) and some sites were not significantly different from a constant rate model. However, there are substantial limitations to examining long-term trends in the bi-weekly BayWatcher data including the limited number of samples (as few as two samples per year) and because most samples were collected at low tide in the morning hours or had no sample time recorded (Fig. S5). Upstream dams were removed above the Bay-Watcher Quashnet River site that led to changes in water input (Howes et al. 2017) as evident in the large decrease in salinity (−2.5% y^{-1} , Table S1). The variability and difficulty in interpreting the BayWatcher long-term trends are also apparent from the two overlapping sites with the SWMP, where the Menauhant site was not different from a constant slope model and the Childs River site showed much higher rates of warming (0.12 $^{\circ}\text{C y}^{-1}$) compared to the consistent

rates from the higher frequency (sub-hourly) data of the SWMP (Tables 2 and 3).

Oxygen

At all SWMP and BayWatchers sites, there were significant declines in oxygen concentrations when excluding the unstable and problematic sites of the Menauhant and Quashnet Rivers. Across the remaining sites, there were declines of between −0.30 to −0.77% per year (Tables 2 and 3, Fig. 4). Over the last 2 decades, the deoxygenation was $-24.7 \pm 6.6 \mu\text{mol O}_2 \text{ kg}^{-1}$ or a decline of −11.2%.

pH/ H^+

At all the SWMP sites, there were significant decreases in pH, which are shown as increases in the concentration of

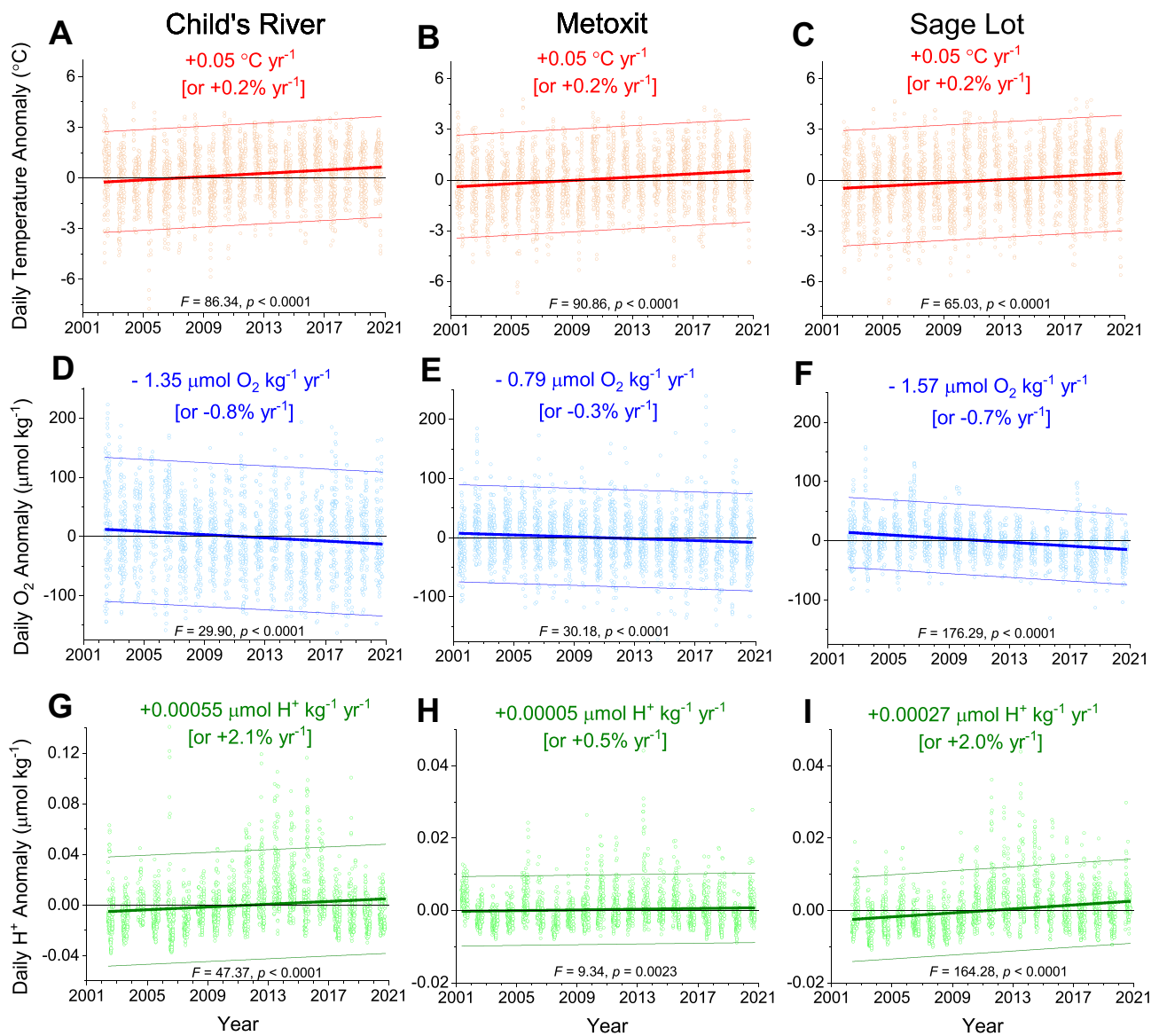


Fig. 4 Temperature (A, B, C), oxygen (D, E, F), and H^+ (G, H, I) anomalies from individual Julian-day seasonal models over the entire study period at the Childs River (A, D, G), Metoxit (B, E, H), and Sage Lot (C, F, I) sites of the WBNERR SWMP. Bold lines indi-

cate the linear fit, light lines indicate the 95% prediction band, and the slopes are represented in text as rate changes and percent per year (details and significant differences from a constant slope model are found in Table 2)

H^+ ions (i.e., increases in acidity) (Table 2, Fig. 4, Fig. S3). Excluding Menauhant, the increase in acidity was between 0.5% and 2.1% per year, which results in an increase of acidity of $0.006 \pm 0.005\text{ }\mu\text{mol H}^+\text{ kg}^{-1}$ or 34.2% over the last 20 years.

The oxygen and H^+ anomalies were also significantly correlated and exhibited different slopes across sites (Fig. 5). The correlations between temperature anomalies and the H^+ and oxygen anomalies were one and two orders of magnitude lower, respectively, for each site (Table S2).

Duration and Variability

While long-term trends in warming, acidification, and deoxygenation were apparent, there were also increased durations of extreme conditions across the SWMP sites (Fig. 6). Heat waves, defined as week-long periods where the daily maximum temperatures were greater than the 90th percentile for each site, increased across the last 2 decades. Similarly, low oxygen conditions, defined as week-long periods where daily minimum oxygen concentrations were less than the 10th

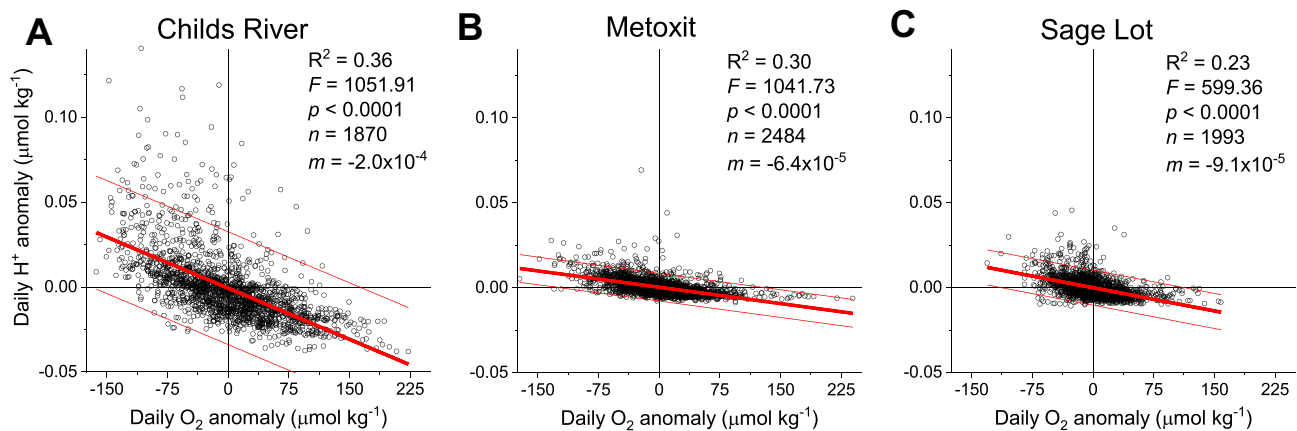


Fig. 5 Correlations between daily oxygen and H^+ anomalies at the Childs River (A), Metoxit (B), and Sage Lot (C) sites of the WBNER SWMP. Bold lines indicate the linear fit and light lines indicate the 95% prediction band. Statistics represent ANOVA compared to a con-

stant model. The correlations between temperature and H^+ and oxygen anomalies were one and two orders of magnitude lower, respectively, for each site (Table S2)

percentile for each site, became more common. Over the last decade, there were also increased phytoplankton blooms, defined as week-long periods where the daily maximum chlorophyll *a* was greater than the 90th percentile for each site.

There were also changes to the daily or diel cycles across the study period (Fig. 7). At Childs River and Sage Lot, there were general decreases in the oxygen concentrations across all hours of the day which was particularly apparent in some recent years with afternoon maxima substantially reduced (Fig. 7A, C). At Metoxit, a different diel trend was apparent with more variability in recent years, evident in higher afternoon maxima and lower oxygen levels at dawn (Fig. 7B). Diel trends in temperature exhibited the general warming trend across years and highlight the different daily temperature variability at each site (Fig. 7D–F).

Nutrients

The BayWatcher and SWMP water sample data were used to examine changes in total nitrogen over time as well as the relationship between total nitrogen and chlorophyll *a* concentration throughout the estuary (Fig. 8A–D). There was a significant increase in total nitrogen in Waquoit Bay that ranged from $0.67 \mu\text{mol N L}^{-1} \text{ y}^{-1}$ at Sage Lot from 2006 to 2020 (Fig. 8C) and up to $2.32 \mu\text{mol N L}^{-1} \text{ y}^{-1}$ from 2016 to 2020 at all sites across Waquoit Bay from the BayWatcher data (Fig. 8a). Due to limitations in the BayWatcher data quality, the intermediate years were rejected, so the analysis was completed with and without the data from 2004 to 2006. The fit to the 2004–2020 BayWatcher data was still positive and significant (Fig. 8A) when the 2004–2006 data were included. There was also a significant positive correlation between total nitrogen and chlorophyll *a* concentration across the bay (Fig. 8B) as well as at the individual sites of the SWMP (Fig. 8D).

Discussion

A diversity of water quality monitoring programs (e.g., the 30 sites of the NERR program) have been making excellent progress in monitoring coastal ecosystems in recent decades due to the wide availability of commercial sensor systems. However, these programs are usually focused on collecting and curating data with limited time and resources applied to analyzing and synthesizing the collected data. Motivated by the recent demise of the last *Zostera marina* meadows in Waquoit Bay, we analyzed water quality data collected by the Waquoit Bay NERR and local citizen science BayWatcher programs over the last 30 years. One of the main objectives of this study was to apply time-series analysis techniques and substantial knowledge about the history of the monitoring programs (including the strengths and limitations of the data) to reveal long-term trends in water quality. These methods can be applied to other monitoring data to advance the knowledge gained from similar monitoring programs, enhance our understanding of estuarine biogeochemistry, and investigate estuarine responses to long-term change. These analyses and results point to the ongoing plight of coastal estuaries across the world that are impacted by the combined effects of local (eutrophication, the loss of seagrasses, intensifying algal blooms) and global (deoxygenation, acidification, and warming) anthropogenic stressors.

Analysis Framework

A major challenge with analyzing this type of monitoring program data is ensuring that comparisons across sites and years are equitable and valid. While many data programs and repositories have robust quality assurance and quality control procedures (e.g., the NERR system), there are still

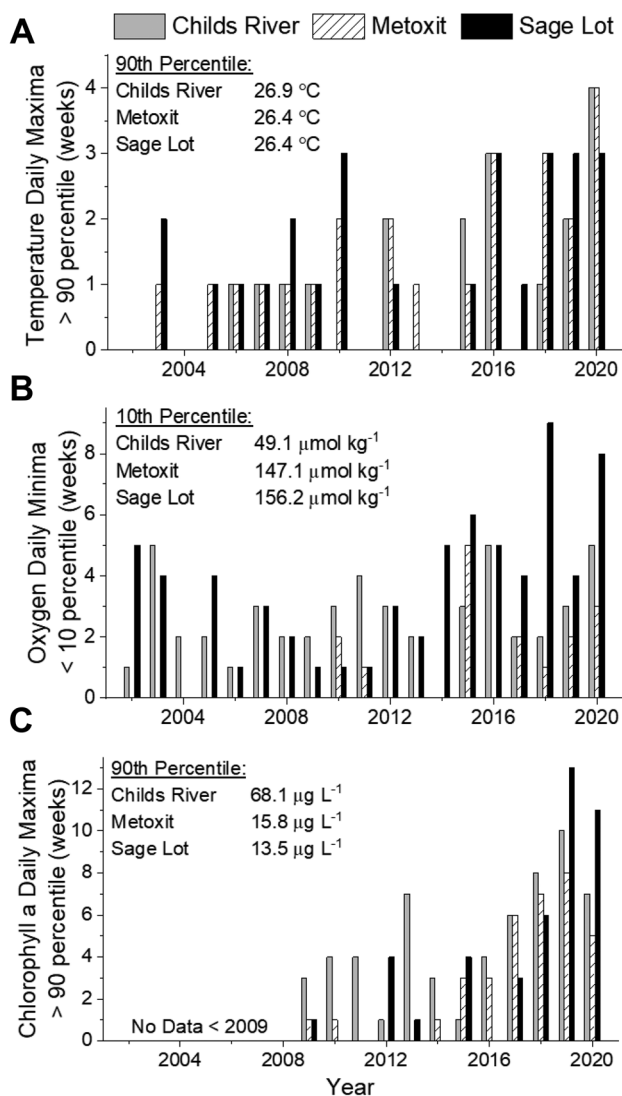


Fig. 6 The number of weeks in each year when **A** the daily temperature maxima were greater than the 90th percentile for 7 consecutive days, **B** the daily oxygen minima were less than the 10th percentile for 7 consecutive days, and **C** the daily chlorophyll *a* maximum was greater than the 90th percentile for 7 consecutive days. The 10th and 90th percentiles (inset text) were determined from all filtered SWMP data for each site

site-specific factors that must be considered, which may also be opaque or difficult to discern from metadata files. For example, the WBNERR staff collected data year-round in the early years of the program, but later started to avoid the winter months due to the potential of occasional freezing and instrument loss. This, combined with other periods that had reduced data coverage across years, led to the exclusion of a large portion of the calendar year (mid-October to mid-May) as data would be biased by early years when there was year-round sampling. The comparison across different sites should also be considered to ensure that anomalous or non-comparable sites are removed to prevent unintended biases.

For example, one SWMP site, Menauhant, was located at an inlet to Waquoit Bay and was dominated by Vineyard Sound water on incoming tides, had changed to the inlet over time, and had chlorophyll sensor interferences, so this site was not directly compared in this study which focuses on the interior of the estuary. Additionally, one site in the BayWatcher program, Quashnet River, had significant declines in salinity that were caused by upstream dam removal (Howes et al. 2017). These examples demonstrate that knowledge of local site characteristics and program history is vital to accurate analyses of similar long-term monitoring data. Other program- and site-specific factors, such as changes in sensors, platforms, locations, land-use, anti-fouling efforts, and sampling rates, should also be investigated to limit bias in data and long-term trend analyses.

Many of the long-term trends (i.e., the slope of concentration vs. time) reported here have statistically significant correlations, compared to a constant or zero slope model, but have low coefficients of determination (R^2 , Tables 2 and 3) due to the use of daily anomalies from the long-term seasonal trend. On any given day, variability was driven by multiple environmental parameters that were not encompassed in the date axes. For example, daily anomalies in temperature can be driven by short-term changes in solar radiation, wind directions, cloud cover, weather, precipitation, and tides. Similar environmental factors drive daily anomalies in other factors examined here including oxygen, pH, and chlorophyll *a* concentrations, such as changes in solar radiation, temperature, weather, nutrient input, dominant primary producers, and tides. Daily anomalies were used in this study, as opposed to weekly or monthly anomalies, as the daily anomalies produced the best fits for the seasonal model due to the increased data density (i.e., daily = 153 data points y^{-1} , as opposed to weekly = ~ 22 data points y^{-1} or monthly = ~ 5 data points y^{-1}). Thus, the presented analyses represent a trade-off for best parameterizing the Julian-day seasonal model while being detrimental to the long-term anomaly R^2 values. Regardless, there were statistically significant trends across multiple decades at multiple sites, which were poorly described by the correlated date axes, but were still indicative of significant long-term changes.

Macrophyte Declines

The global decline of seagrass meadows is well documented, placing them among the most threatened ecosystems worldwide (Waycott et al. 2009). At the same time, they have become valued for a variety of ecosystem services (Costanza et al. 1997), including their important contributions to blue carbon cycling and their potential for climate change mitigation (Fourqurean et al. 2012). From the perspective of seagrass decline, macroalgae are typically viewed as a competitive species exacerbated by eutrophication; however, many macroalgal

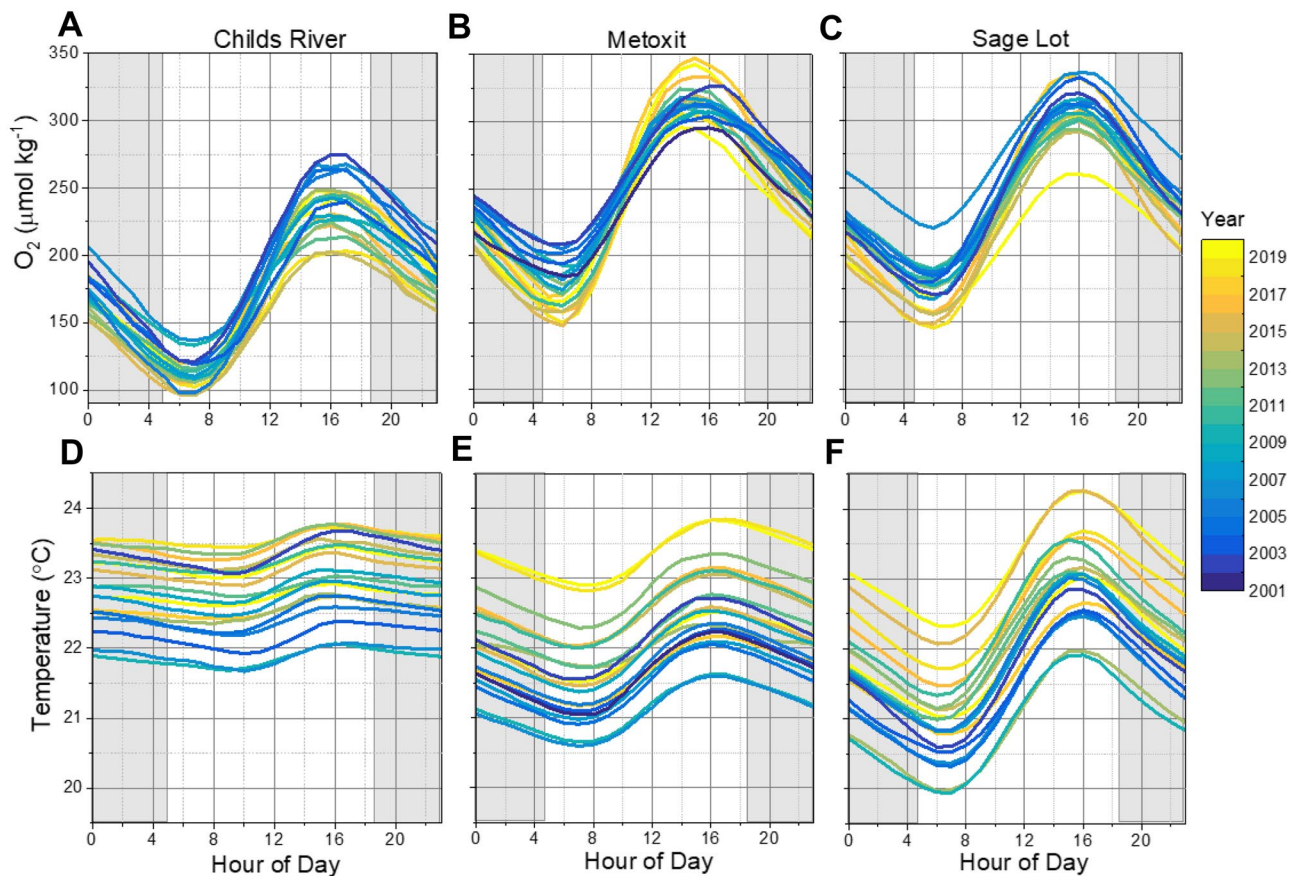


Fig. 7 Diel trends of oxygen (A, B, C) and temperature (D, E, F) during each year at the Childs River (A, D), Metoxit (B, E), and Sage Lot (C, F) sites of the WBNERR SWMP. Time-of-day averages are for each year during the filtered period of mid-May to mid-October

species are also in large-scale decline due to the same set of local and global stressors (Wahl et al. 2015). Whether macroalgae are defined as an intermediate benefactor in eutrophication or a group of macrophytes that are in global decline, it is readily apparent that most benthic macrophyte communities are being adversely affected by anthropogenic stressors.

Waquoit Bay has a rich history of research and monitoring that allowed for the investigation of seagrass abundances over the last ~70 years (Costa 1988; Costa et al. 1992; Short and Burdick 1996, MassGIS) and macroalgal abundances over the last 30 years (Valiela et al. 1997, 2016; Deegan et al. 2002; Fox et al. 2008; Cebrian et al. 2014). Based on these historical estimates eelgrass had declined by 95% from 1951 aerial imagery estimates (Costa et al. 1992) at the start of the WBNERR monitoring program in 2001. By 2020, eelgrass had declined by >97% which was similar to the results found by Howes et al. (2021). However, early estimates of eelgrass coverage were extremely conservative as they did not consider the smaller sub-embayments that represent the last locations that eelgrass has been found in Waquoit Bay (Short and Burdick 1996). Additionally, while the trends of

deoxygenation, warming, acidification, and chlorophyll *a* presented here represent changes in water quality, most eelgrass losses had happened prior to these monitoring efforts. In contrast, significant declines of macroalgae occurred during the SWMP program and historical data from the last 30 years support this. While individual sites were highly variable across the years, it is apparent that macroalgae, which replaced eelgrass as the dominant macrophyte, are now also in decline. This is supported by Hersh (1996) and Fox (2008) who saw decreases in macroalgae in Waquoit Bay from the early 1990s to the early 2000s of about ~50%. However, the substantial interannual and seasonal variability in macroalgae, as noted in Fox et al. (2008), in addition to macroalgal bloom and drift dynamics, make periodic macroalgal biomass sampling difficult to interpret. For example, Valiela et al. (2016) found no changes in macroalgae between the mid-1990s and 2015 with average biomasses of about 160 ± 15 , 106 ± 20 , and 54 ± 7 at the sites of Childs River, Quashnet River, and Sage Lot, respectively (all numbers g dry wt m^{-2} and \pm standard errors). In comparison, this study found similar average biomass and variability across

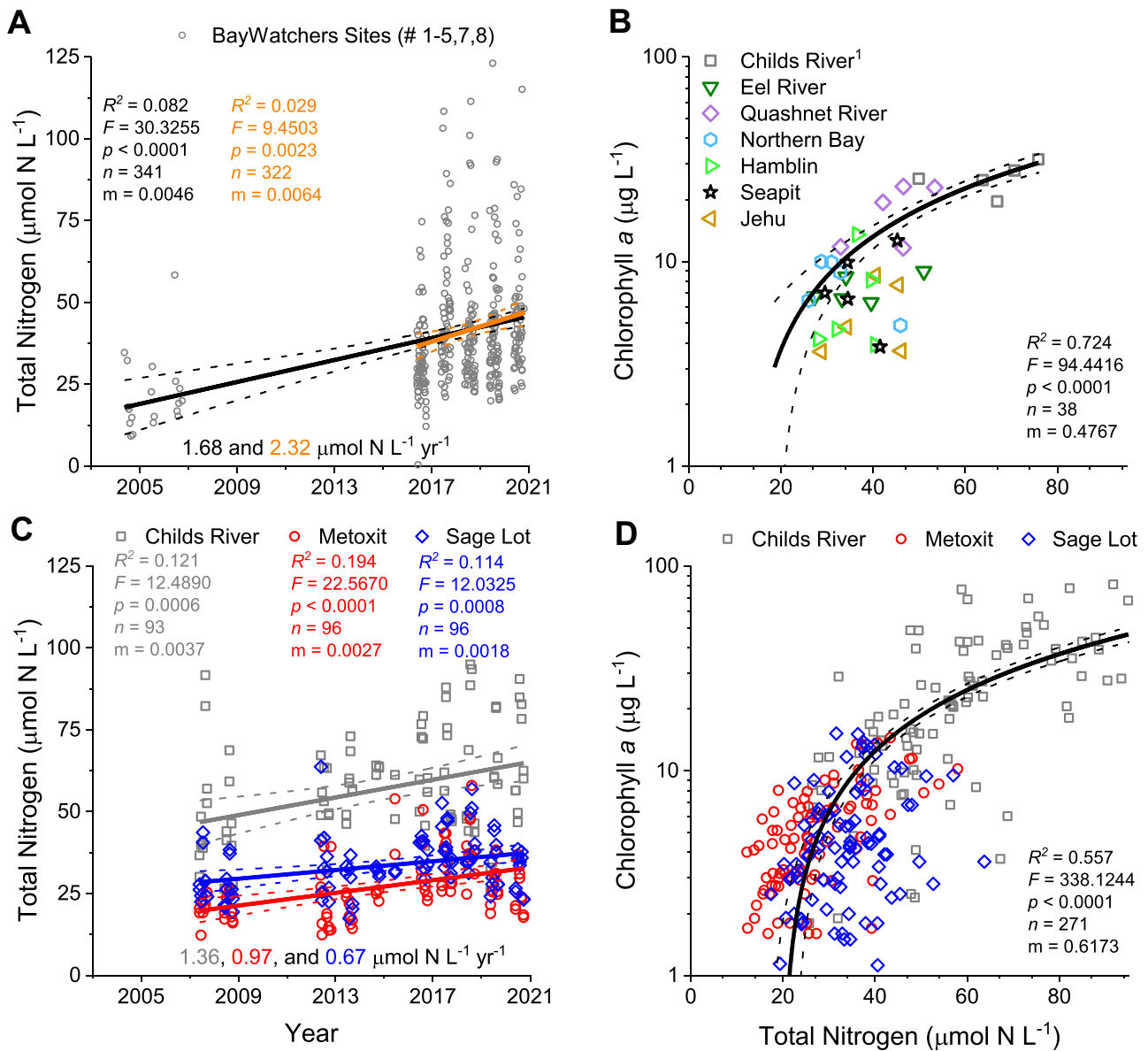


Fig. 8 BayWatcher total nitrogen for sites 1–5, 7, and 8 in Waquoit Bay from mid-May to mid-October (**A**) and the yearly-averaged (from mid-May to mid-October) total nitrogen plotted against the yearly-averaged chlorophyll *a* (error bars omitted for clarity) from BayWatcher water samples or the SWMP sonde data, with the latter indicated with a superscript (**B**). Similarly, the SWMP total nitrogen for the Childs River (grey), Metoxit (red), and Sage Lot (blue) sites in Waquoit Bay from mid-May to mid-October from SWMP monthly water samples (**C**), and

the sample total nitrogen plotted against the chlorophyll *a* from individual SWMP water samples (**D**). Bold lines indicate the least-squares fit, dashed lines indicate the 95% confidence interval, and statistics represent ANOVA compared to a constant model. Without the BayWatcher data from 2004 to 2006 (from only a few of the sites), the fit to the 2016 to 2020 BayWatcher data is still positive and significant (**A**, orange)

the WBNERR SWMP surveys (every 3–4 years) with average biomasses of about 188 ± 16 , 69 ± 4 , and 82 ± 12 at the sites of Childs River, Quashnet River, and Sage Lot, respectively (all numbers g dry wt m^{-2} and \pm standard errors). However, substantial declines over the last 30 years are evident when combining data from multiple studies.

Shift to Pelagic Dominance

The dominance of phytoplankton communities in estuarine systems due to intense nutrient loading has been proposed as the ultimate response to eutrophication (Duarte 1995; Valiela et al. 1997; Cebrian et al. 2014; Qin and Shen 2019).

As phytoplankton are able to reside and reproduce near the water surface, they can grow under nutrient-replete conditions with minimal light limitation, as opposed to seagrasses or macroalgal mats that reside on the bottom and decline due to shading by increased phytoplankton biomass (Duarte 1995; Valiela et al. 1997; Cebrian et al. 2014). While the generalized response of the loss of benthic communities due to eutrophication-linked increases in phytoplankton has been proposed, complicating factors such as water residence times, grazing of phytoplankton, and light reduction by sediment resuspension have been proposed as factors that may prevent or limit pelagic dominance (Cebrian et al. 2014). The study by Cebrian et al. (2014) also points to the presence of hypoxic conditions as a major factor in the ultimate shift to a pelagic-dominated system. Furthermore, recent evidence of increased pelagic metabolism and reduced benthic metabolism along a eutrophication gradient in the York River estuary of the lower Chesapeake Bay, USA, by Qin and Shen (2019), points to incremental changes in the ecosystem state. These gradual changes were challenging to detect and required continuous monitoring to distinguish them from natural variability.

By tracking the decline of eelgrass and the ongoing decline of macroalgae in Waquoit Bay, it is now evident that pelagic phytoplankton are becoming the dominant group of primary producers. While macroalgal abundance has varied substantially, it is apparent from the historical and SWMP data that macroalgae have decreased in the last 3 decades and that pelagic chlorophyll *a* concentrations and phytoplankton have increased more recently (Howes et al. 2021). Fox et al. (2008) suggested that macroalgae abundance would become light limited and decrease, though they suggested this would be due to intra-canopy shading and not shading by pelagic phytoplankton. Since 2009, when chlorophyll *a* sensors were added to the SWMP, chlorophyll *a* has increased by 71%, and longer-term trends in the BayWatcher chlorophyll *a* samples show an increase of ~2–7% y^{-1} . The average annual increase in Waquoit Bay of 2–7% is consistent with a recent study by Rheuban et al. (2016) that examined many sites in nearby Buzzards Bay and found an average annual increase in chlorophyll *a* of 4%. The increase in phytoplankton in the water column may have led to less light transmission to the benthos and could be partially responsible for the decline in macroalgal abundance (Cebrian et al. 2014). Other possible factors that may have led to decreases in macroalgae biomass include intra-canopy shading (Fox et al. 2008), changes in phytoplankton assemblages due to warming and changes in precipitation (Rheuban et al. 2016; Williamson et al. 2017), and changes in nutrient cycling due to hypoxia, warming, and freshwater input (Foster and Fulweiler 2014, 2019). In comparison to macroalgae, chlorophyll *a* concentrations have increased in Waquoit Bay despite the addition of oyster aquaculture to

help remove excess nutrients and phytoplankton (Mara et al. 2021) and short tidal residence times that were thought to limit pelagic productivity (D'Avanzo et al. 1996; Havens et al. 2001; Valiela and Bowen 2002). However, our results indicate that ongoing eutrophication has ultimately led to a shift to pelagic dominance.

Many studies have investigated the nutrient inputs and the terrestrial watersheds from which they derive in Waquoit Bay (Valiela et al. 1992, 2016, 2021; Valiela and Bowen 2002; Martinetto et al. 2006; Howes et al. 2012, 2017, 2021). Additionally, many studies used the difference in watershed nitrogen loads across the estuary to study the impacts of eutrophication, often utilizing Sage Lot as a low-load or oligotrophic site due to its relatively low nutrient inputs from a less-developed and more forested watershed (e.g., Peckol and Rivers 1996; Fox et al. 2008; Foster and Fulweiler 2014). In an isotopic study by Martinetto et al. (2006), they found that benthic primary producers (macroalgae and seagrass), benthic taxa, and fish were isotopically coupled to their watersheds, but that planktic organisms were not because of short-residence time waters that uncoupled them from their terrestrial watersheds. The shift to pelagic dominance in Waquoit Bay may explain why relatively low nutrient input locations, such as Sage Lot, are now degraded and absent of seagrass, as the pelagic organisms and associated particulate organic nutrients are readily transported across all of the hydraulically connected sub-embayments (Howes et al. 2021). With the loss of sessile seagrass and the ongoing decline in semi-sessile macroalgal mats, nutrients are not as efficiently captured in the sub-embayments but rather are now readily distributed throughout the entire estuary by planktic organisms in conjunction with the tidal exchange. For example, the total groundwater discharge of the entire Waquoit Bay estuary is ~10,000 $m^3 d^{-1}$, but the daily tidal exchange is ~20,000,000 $m^3 d^{-1}$ (Howes et al. 2012) showing that the tidal exchange is three orders of magnitude larger than groundwater inputs. Therefore, if this nutrient-rich groundwater is no longer being captured by sessile organisms in the local sub-embayments, it can be efficiently transported, along with pelagic organisms, throughout the estuary leading to estuary-wide decline associated with a shift to pelagic dominance.

Long-term Trends

Across the SWMP sites, there were significant and similar trends of increasing temperature, increasing acidity, and decreasing oxygen concentrations. In general, these trends were also consistent across the BayWatchers data, but were generally more variable, likely due to the drastically lower number of samples (as few as two per year in the filtered data) and because sampling was only done during low tide in the morning. Averaging across the three SWMP sites over

20 years revealed a temperature increase of $+1.0 \pm 0.0$ °C ($+4.4\%$), an acidity increase of $+0.006 \pm 0.005$ $\mu\text{mol H}^+$ kg^{-1} ($+34.2\%$), and an oxygen decrease of -24.7 ± 6.6 $\mu\text{mol O}_2$ kg^{-1} (-11.2%). These trends are consistent with the results of Baumann and Smith (2018) who found Waquoit Bay to have significant trends of increasing temperature, decreasing pH (or increased H^+), and decreasing oxygen. Our results are also consistent with the general trends of Baumann and Smith (2018) who found at 16 NERRS sites across the USA that coastal acidification is linked to warming temperatures and that elevated nutrients were largely correlated with higher H^+ and lower oxygen levels. The average annual temperature increases in Waquoit Bay ($+0.05$ °C y^{-1}) were also consistent with recent rates from the study of NERRS sites along the US Atlantic coast ($+0.05$ °C y^{-1} , Baumann and Smith 2018), with rates from the Atlantic Ocean near Bermuda from the same period ($+0.05$ °C y^{-1} , Bates and Johnson 2020) and slightly smaller than rates from nearby Buzzards Bay ($+0.08$ °C y^{-1} , Rheuban et al. 2016). Bates and Johnson (2020) found rates of deoxygenation of 2% per decade, while Rheuban et al. (2016) found no trends or slight deoxygenation in Buzzards Bay, highlighting the deleterious effects of eutrophication in Waquoit Bay. Similarly, Bates and Johnson (2020) found an increase in acidity of only about $+0.001$ $\mu\text{mol H}^+$ kg^{-1} over the last 20 years, again suggesting that in Waquoit Bay eutrophication is driving enhanced acidification.

Increases in temperature can lead to both deoxygenation and acidification (Baumann and Smith 2018); however, much of the deoxygenation and some acidification in Waquoit Bay can be attributed to eutrophication or temperature-enhanced respiration. For example, a $+1.0$ °C change in temperature results in a reduction of only ~ 4 $\mu\text{mol O}_2$ kg^{-1} due to reduced solubility or about 16% of the observed change. Likewise, a $+1.0$ °C change in temperature results in a change of ~ 0.0005 $\mu\text{mol H}^+$ kg^{-1} or about 9% of the observed change (determined using CO2SYS (Van Heuven et al. 2011) assuming 28 PSU, 1850 $\mu\text{mol kg}^{-1}$ total alkalinity, and 1750 $\mu\text{mol kg}^{-1}$ dissolved inorganic carbon as observed in Waquoit Bay by Liu et al. (2017)) with most of the changes in pH due to the combined effects of ocean acidification and eutrophication. While our results cannot specifically distinguish between the effects of temperature-induced increases in respiration, eutrophication, and ocean acidification, it is apparent that water quality in Waquoit Bay, and likely in other similar estuaries, will continue to degrade due to the combined effects of climate change and eutrophication.

The SWMP and BayWatchers water sampling program data indicated that total nitrogen has increased in recent years (Fig. 8A, C). Much of the increase in total nitrogen is due to increases in particulate organic nitrogen and is directly related to increases in phytoplankton (Fig. 8B, D;

Rheuban et al. 2016; Howes et al. 2021). Increases in total nitrogen are also shown in a nitrogen loading model that indicated a 17% increase in nitrogen loads, despite decreases in atmospheric deposition, across the bay from 1990 to 2014 (Table 3 in Valiela et al. 2016) although Valiela et al. (2016) did not find the model results or dissolved inorganic nitrogen measurements to be significantly different across this period. Howes et al. (2017) found that total and dissolved nitrogen increased in the Quashnet River between 2003 and 2016 but noted this was complicated by upstream dam removal. In the following study, Howes et al. (2021) found that nutrient concentrations remained relatively steady from 1997 to 2015 (consistent with the findings of Valiela et al. 2016) but that from 2016 to 2020 nutrients have increased and water quality has declined, which is supported by our results (i.e., Fig. 8). Whether this increase in nutrients represents the shift to pelagic biomass or the results of ongoing nutrient loading require further investigation.

The oxygen and H^+ anomalies were negatively correlated with R^2 of 0.36, 0.30, and 0.23 (for Childs River, Metoxit, and Sage Lot, respectively) with the highest correlation at the most impacted site of Childs River and the lowest at the least impacted site of Sage Lot. This is similar to the different slopes of increasing total nitrogen at these sites (1.36, 0.97, and 0.67 $\mu\text{mol N L}^{-1} \text{y}^{-1}$, respectively). This negative correlation between oxygen and H^+ is common in coastal systems due to daytime primary production that produces oxygen and reduces H^+ and respiration that consumes oxygen and produces H^+ (e.g., Cai et al. 2011; Wallace et al. 2014; Baumann and Smith 2018), and the relationship between these parameters may be useful in evaluating the trophic status, tidal influence, or residence time. The correlations between H^+ and oxygen anomalies also suggest that long-term trends in H^+ are at least partially driven by eutrophication. The correlations between temperature anomalies and the oxygen and H^+ anomalies were 1–2 orders of magnitude lower, respectively (Table S2), highlighting the potentially larger role of eutrophication over warming in impacted estuarine systems.

The duration of high temperature, low oxygen, and high chlorophyll *a* conditions also increased across the study period. These waves (e.g., heat waves) led to stressful conditions that likely helped to accelerate the loss of eelgrass and macroalgae, as well as other organisms (Pulido and Borum 2010; Rivers and Peckol 1995). Increases in pelagic phytoplankton blooms, along with increases in the occurrence of low oxygen conditions, as seen in Waquoit Bay over the last 20 years, have been shown to exacerbate declines in macrophytes leading to systems dominated by pelagic metabolism (Cebrian et al. 2014). The diel variability of conditions also changed across the study period with lower oxygen levels in the early morning in addition to higher temperatures in recent years. Interestingly, the afternoon oxygen maximum

increased in recent years at Metoxit, which may reflect the increases in pelagic primary production at this deeper site, where surface waters may have become more productive than the previous (and deeper) benthic communities. While the diel temperature curves showed temperature increases across the study period, they also illustrate the differences between the sites. Childs River is narrow, largely shaded, and has more groundwater inputs generating the smallest diel variability. Metoxit is deeper and is near the center of the main estuary and exhibited more variability. Sage Lot showed the greatest diel variability that was likely driven by being the shallowest site with the largest surface area to volume ratio.

Conclusions

This research identified trends in key water quality parameters in Waquoit Bay that coincided with the loss of macrophytes and the recent shift to pelagic dominance. The distinct trends of warming, acidification, and deoxygenation are partially driven by global trends, but local eutrophication has also likely contributed to increased rates of acidification and deoxygenation. The shift from benthic sessile primary production by seagrasses and macroalgae to a system now dominated by pelagic primary production and respiration has led to a mobile biomass that is efficiently transported throughout the estuary system, leading to degradation even at sites with low nutrient loads. In 1950s seagrass surveys, the Sage Lot site was too small to matter, then later became an ideal site for examining systems with low nutrient loads but is now an unvegetated degraded pond dominated by pelagic metabolism. In general, water quality has declined substantially in the last 2 decades since the WBNERR SWMP and BayWatchers began their monitoring programs, and there is an urgent need to address wastewater handling to improve the estuary, especially in the face of global changes. As demonstrated here, we expect that the presented analysis framework for monitoring program data can be applied to other sites to enable the evaluation of data quality, gradual ecosystem changes (e.g., warming, acidification, and deoxygenation), and the past conditions that put the current trends into a historical perspective. Ultimately, these results in the rapidly warming region of the Northeastern USA indicate that other similar eutrophic estuaries may become dominated by pelagic metabolism with ongoing global climate change. These results cannot disentangle the contributions of global change or eutrophication to estuary decline. However, they do point to a potential combined effect that may result in other similar estuaries becoming dominated by pelagic metabolism in the future, and the resulting deleterious effects of harmful algal blooms, hypoxia, and the loss of species diversity and ecosystem function.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s12237-022-01166-7>.

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Data Availability The WBNERR SWMP data is available at <http://cdmo.baruch.sc.edu/>. The BayWatcher data are available at <https://waterquality.capecodcommission.org/#/>. The macrophyte survey data is available from the WBNERR on request.

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