Marine Pollution Bulletin 178 (2022) 113598

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Initial estuarine response to inorganic nutrient inputs from a legacy mining facility adjacent to Tampa Bay, Florida

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ARTICLE INFO

Keywords:
Macroalgae
Nitrogen
Phosphate mining
Piney Point
Seagrass
Tampa Bay

ABSTRACT

Legacy mining facilities pose significant risks to aquatic resources. From March 30th to April 9th, 2021, 814 million liters of phosphate mining wastewater and marine dredge water from the Piney Point facility were released into lower Tampa Bay (Florida, USA). This resulted in an estimated addition of 186 metric tons of total nitrogen, exceeding typical annual external nitrogen load estimates to lower Tampa Bay in a matter of days. An initial phytoplankton bloom (non-harmful diatoms) was first observed in April. Filamentous cyanobacteria blooms (*Dapis* spp.) peaked in June, followed by a bloom of the red tide organism *Karenia brevis*. Reported fish kills tracked *K. brevis* concentrations, prompting cleanup of over 1600 metric tons of dead fish. Seagrasses had minimal changes over the study period. By comparing these results to baseline environmental monitoring data, we demonstrate adverse water quality changes in response to abnormally high and rapidly delivered nitrogen loads.

1. Introduction

Wastewater byproducts from mining are a global threat to the quality of surface and groundwater resources (Hudson-Edwards et al., 2011; Tayibi et al., 2009). The production of phosphate fertilizer generates large amounts of phosphogypsum waste ($CaSO_4{\cdot}H_2O$) that is typically stored on-site in large earthen stacks (gypstacks) capable of holding hundreds of millions of liters of process water. Water quality in gypstacks can vary depending on processing method used at the mining facility, background geological characteristics of the region, and on-site practices for managing stormwater or other activities that can introduce additional materials to the holding ponds (Henderson, 2004; Pérez-López et al., 2010). In addition to elevated phosphorus concentrations, other nutrients, contaminants, and radionuclides may be present at values much higher than natural surface waters (Beck et al., 2018a; Burnett and Elzerman, 2001). Many of these gypstacks no longer support active mining and aging infrastructure combined with climate change and seasonal stressors (e.g., heavy precipitation events) have reduced the capacity of these facilities to maintain water on site. Numerous studies have documented the environmental and human health risks

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https://doi.org/10.1016/j.marpolbul.2022.113598
Received 9 February 2022; Received in revised form 17 March 2022; Accepted 20 March 2022
Available online 30 March 2022

associated with these stacks (Beck et al., 2018a; El Zrelli et al., 2015; Pérez-López et al., 2016; Sanders et al., 2013; Tayibi et al., 2009).

The geology of central Florida is rich in phosphates that have supported a multi-billion dollar mining industry for fertilizer to support agricultural production (Henderson, 2004). By 2001, an estimated 36 million metric tons of phosphogypsum were created each year in northern and central Florida (Burnett and Elzerman, 2001). Effective management and final closure of these facilities are imperative to reduce threats to prior ecosystem recovery efforts and investments. The Piney Point facility located in Palmetto, Florida is a large, remnant gypstack with three holding ponds located 3 km from the shore of Tampa Bay and near two Florida Aquatic Preserves [see supplement for a history of the facility; Henderson, 2004]. Holding capacity of the ponds has decreased over time from seasonal rain events, tropical storms, and storage of dredging material from nearby Port Manatee. Releases from the stacks occurred in the early 2000s and in 2011 to nearby Bishop Harbor connected to Tampa Bay. Those releases resulted in spatially-restricted, ecosystem responses including localized harmful algal blooms and increased macroalgal abundance (Garrett et al., 2011; Switzer et al., 2011).

In March 2021, leakages were detected from a tear in the plastic liner of the southern holding pond (NGS-S) at Piney Point. At that time, approximately 1.8 billion liters of mixed legacy phosphate mining wastewater and seawater from port dredging operations were being held in the failing gypstack. Piney Point historically produced Diammonium Phosphate ($(NH_4)_2HPO_4$) and the remnant stackwater has very high concentrations of total nitrogen (TN), in addition to total phosphorus (TP). Water quality parameters of NGS-S measured in 2019 showed TP (160 mg/L) and TN (230 mg/L) were approximately three orders of magnitude higher than typical concentrations in Tampa Bay. From March 30th to April 9th, approximately 814 million liters (215 million gallons) of stack water were released to lower Tampa Bay following an emergency order authorized by the Florida Department of Environmental Protection (FDEP). Over this ten day period, an estimated 186 metric tons (205 tons) of nitrogen were delivered to the bay, exceeding contemporary annual estimates of external nutrient loads to lower Tampa Bay in a matter of days (Janicki Environmental, Inc., 2017).

This paper provides an initial assessment of environmental conditions in Tampa Bay over the six month period after the release of legacy phosphate mining wastewater from the Piney Point facility in 2021. The goal is to describe the results of monitoring data of surface waters collected in response to the event to assess relative deviation of current conditions from long-term, seasonal records of water quality, phytoplankton, and seagrass/macroalgae datasets available for the region. Numerous studies, as well as the successful nutrient management paradigm, have demonstrated nitrogen-limitation in Tampa Bay and the system is generally considered phosphorus enriched (Greening et al., 2014; Greening and Janicki, 2006; Wang et al., 1999). As such, we focus on nitrogen in our analyses as the identified limiting nutrient for Tampa Bay and its potential to create water quality conditions unfavorable for seagrass growth due to enhanced algal production. Our analysis evaluated datasets that are descriptive of the vulnerability of seagrasses to nutrient pollution though cascading negative effects of nitrogen, phytoplankton growth and persistence, and water clarity on seagrass growth and survival (Beck et al., 2018b; Dixon and Leverone, 1995; Greening and Janicki, 2006; Kenworthy and Fonseca, 1996). A timeline of events is provided, which is supported by the quantitative results from 2021 routine and response-based monitoring of conditions in and around Port Manatee, FL – the focal point of emergency releases from the Piney Point facility. The results from this paper provide an unprecedented chronology of short-term estuarine response to acute nutrient loadings from legacy mining facilities, where context would not have been possible without the long-term monitoring datasets available for the region.

2. Methods

2.1. Simulation modeling

Monitoring of the natural resources of Tampa Bay in response to the release from Piney Point began in April 2021 and continued for six months through September. These data were collected through a coordinated effort under the guidance of a plume simulation by a numerical circulation model run by the Ocean Circulation Lab at the University of South Florida (USF), College of Marine Science. The plume evolution from Piney Point was simulated using the Tampa Bay Coastal Ocean Model (TBCOM) nowcast/forecast system (Chen et al., 2018, 2019), with an embedded tracer module that included realistic release rates. Normalized tracer distributions were automatically updated each day, providing 1-day hindcasts and 3.5-day forecasts throughout the period of discharge and subsequent Tampa Bay distribution. The modeled plume evolution web product (http://ocgweb.marine.usf.edu/~liu/Tracer/) served as the principal guidance for coordinating the data collection during the event. Preliminary model results for Piney Point are reported in Liu et al. (2021) and previous model veracity testing was described in Chen et al. (2018) and Chen et al. (2019) (and references therein).

2.2. Monitoring response to the emergency release

Monitoring agencies and local partners that collected data using standardized protocols included FDEP, Environmental Protection Commission (EPC) of Hillsborough County, Parks and Natural Resources Department of Manatee County, Pinellas County Division of Environmental Management, Fish and Wildlife Research Institute (FWRI) of the Florida Fish and Wildlife Conservation Commission (FWC), City of St. Petersburg, Tampa Bay Estuary Program (TBEP), Sarasota Bay Estuary Program, Environmental Science Associates, University of South Florida, University of Florida, and New College of Florida. Monitoring efforts focused on a suite of parameters expected to respond to increased nutrient loads into the bay, including water quality sampling, phytoplankton identification, and seagrass and macroalgae transect surveys (Fig. 1).

Water quality parameters included discrete, laboratory-processed and in situ samples for TN (mg/L), total ammonia nitrogen (NH_3 + NH_4^+, mg/L, hereafter referred to as ammonia), nitrate/nitrite (NO_3^- + NO_2^-, mg/L), TP (mg/L), orthophosphate (PO_4^{3-}, mg/L), chlorophyll-a (chl-a, μg/L), pH, salinity (ppt), temperature (°C), and dissolved oxygen saturation (%). Most samples were surface collections by boat, with sample frequency approximately biweekly for locations around Piney Point, although effort varied by monitoring group and was more consistent during the first three months after the release. Established laboratory and field sample protocols for all survey methods were based on an Interagency Monitoring Project Plan maintained by the TBEP and those of the inter-agency partners. Data quality objectives followed guidelines outlined in the USEPA-approved TBEP Data Quality Management Plan (Sherwood et al., 2020). Many of the local partners also participate in the Southwest Florida Regional Ambient Monitoring Program (RAMP) that ensures similar standards and protocols are followed in the collection and processing of monitoring data, including routine cross-reference of split samples between laboratories to check precision of measured values. Samples requiring laboratory analysis (e.g., nutrient assays) were obtained primarily from bottle collection at the surface, whereas in situ measurements were available for many parameters (e.g., dissolved oxygen, Secchi depth, etc.). In situ measurements were collected using common monitoring equipment, such as YSI sondes or Seabird CTD casts, depending on monitoring agency. Laboratory methods used to process samples were based on accepted procedures promoted through the Southwest Florida RAMP. Additionally, the Sentinel-3 satellites were used to derive chl-a maps, which were subsequently calibrated using field-measured chl-a in surface waters.

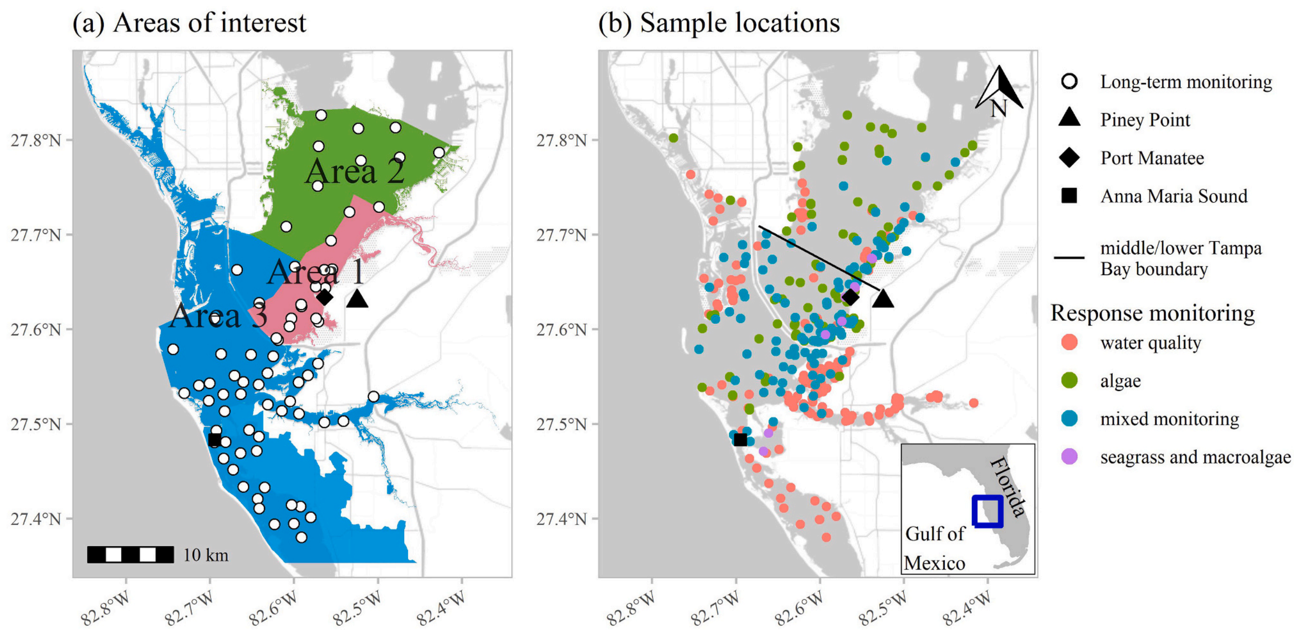

Fig. 1. Areas of interest and long-term monitoring stations (a) for evaluating status and trends in response-based monitoring data and sample locations from March through September 2021 by monitoring data type (b) in response to release from Piney Point. Data types include algae sampling, seagrass and macroalgae, water quality (field-based and laboratory samples), and mixed monitoring (algae, seagrass and macroalgae, water quality). Inset shows location of Tampa Bay on the Gulf coast of Florida, USA.

Phytoplankton samples included a mix of quantitative (cells/L) and qualitative (presence/absence) samples for major taxa at similar frequency and spatial distribution as the water quality samples. Harmful Algal Bloom (HAB) data for *Karenia brevis* were obtained from event-based monitoring samples from the FWC-FWRI HAB Monitoring Database. HAB sampling typically occurs in response to bloom events or fish kills with extensive quality control of cell counts conducted by FWC-FWRI (additional details in Stumpf et al., 2022). HAB data were restricted to Tampa Bay boundaries and over 90% of the samples were collected within one meter of the surface. Bloom sizes for *K. brevis* were described qualitatively as low/medium/high concentrations based on FWC breakpoints at 10,000/100,000/1,000,000 cells/L. Fish kill reports were obtained from the FWC online database. Seagrass and macroalgae sampling occurred approximately biweekly at 38 transects using a modified rapid assessment design, where species were identified and enumerated using Braun-Blanquet abundances in a 0.25 m^2 quadrat at 10 m distances along each 50 m transect (see supplement). Finally, precipitation and wind data were from Albert Whitted Airfield at St. Petersburg, Florida and inflow estimates to Tampa Bay were based on summed hydrologic loads of major tributaries from US Geological Survey gaged sites (similar to Janicki Environmental, Inc., 2012). Additional details of the sampling methods and data sources are provided in supplement.

2.3. Data analysis

Long-term water quality monitoring data from Hillsborough and Manatee counties (accessible at https://wateratlas.usf.edu/, Hillsborough County collected monthly, Manatee County collected quarterly) were used to establish baseline conditions for major areas of interest in Fig. 1a to compare with the response monitoring data described above. These areas (Area 1: closest to Piney Point; Area 2: north of Piney Point; Area 3: south of Piney Point including northern Sarasota Bay) were identified based on anticipated impacts from expected plume patterns following the TBCOM simulations and other prominent bay boundaries relative to Piney Point (i.e., the main shipping channel in the bay, inflow boundaries, location of the Skyway Bridge at the mouth of Tampa Bay, and major bay segments used by TBEP for assessing annual water quality targets). Observations at each long-term monitoring station were averaged for each month across years from 2006 to 2020. This period represents a "recovery" stage for Tampa Bay where water quality conditions were much improved from historical conditions during a more eutrophic period and when seagrass areal coverage was trending toward and above a 1950s benchmark target of 15,378 ha (38,000 acres, Greening et al., 2014; Sherwood et al., 2017). For each month, the mean values $+/-1$ standard deviation for each parameter at each station were quantified and used as reference values relative to results at the closest water quality monitoring station that was sampled in response to Piney Point. This comparison was made to ensure that the response data were evaluated relative to stations that were spatially relevant (e.g., long-term conditions near the mouth of Tampa Bay are not the same as those in the middle of the bay) and seasonally-specific (e.g., historical conditions in April are not the same as historical conditions in July). In some cases, the nearest long-term station did not include data for every monitoring parameter at a response location and the next closest station was used as a reference. The average distance from a monitoring location in 2021 to the long-term sites was 1.6 km (see https://shiny.tbep.org/piney-point/ for a map of the matches).

The historical monitoring data were also used to model an expected seasonal pattern for water quality parameters from April to October in 2021. This was done by estimating smoothed annual and seasonal splines with Generalized Additive Models (GAMs) using data only from the "recovery" stage of Tampa Bay (2006 to 2020). GAMs were used to model time series of water quality parameters as a function of a continuous value for year (i.e., decimal year) and as an integer value for day of year. The continuous year value was modeled with a thin plate regression spline and the day of year value was modeled with a cyclic spline (following similar methods as Murphy et al., 2019). The modeled results provided an estimate of the expected normal seasonal variation that takes into account a long-term annual trend. Differences in the observed values sampled in the April to October time periods from the "forecasted" predictions of the baseline GAMs through 2021 provided an assessment of how the current data may have deviated from historical and normal seasonal variation.

Statistical assessments were conducted only on TN, chl-a, and Secchi disk depth as a general analysis of potential patterns in eutrophication in nitrogen-limited systems. Spatial comparisons were based primarily on the three areas identified in Fig. 1a. Variables with log-normal distributions were $\log_{10}$-transformed (i.e., nutrients, chl-a) prior to analysis. Only the water quality data from FDEP were used for statistical analysis given the consistency of sample location and collection dates. Secchi observations that were visually identified on the bottom (71 of 431 observations in the FDEP data) were removed from analysis. Observations for other parameters that were below laboratory standards of detection were evaluated with methods described below.

Differences in observations between months for April to September for water quality, seagrass, and macroalgae within each area (Fig. 1a) were evaluated using a Kruskal-Wallis one-way analysis of variance (ANOVA) followed by multiple comparisons using 2-sided Mann-Whitney *U* tests (Hollander et al., 2013). These tests were used to statistically characterize the temporal progression of changes in the bay following release from Piney Point, e.g., were July conditions significantly different from April? Probability values were adjusted using the sequential Bonferroni method described in (Holm, 1979) to account for the increased probability of Type I error rates with multiple comparisons. An adjusted p-value $< 5\%$ ($\alpha = 0.05$) was considered a significant difference between months. For water quality variables, monthly averages from long-term monitoring data were subtracted from 2021 observations to account for normal seasonal variation not attributed to potential effects from Piney Point. Similar corrections were not done for monthly comparisons of seagrass and macroalgae data because comparable long-term seasonal data do not exist. Frequency occurrence estimates were used to evaluate macroalgae and seagrasses as a standard metric used in previous analyses in Tampa Bay (Johansson, 2016; Sherwood et al., 2017). Methods used to accommodate measured concentrations of water quality variables that were below detection included summary statistics (e.g., median, mean, and standard deviation) following estimates of the empirical cumulative distribution functions for each parameter using the Kaplan-Meier method for censored data (Helsel, 2005; Lee, 2020).

The R statistical programming language (v4.0.2) was used for all analyses (R Core Team, 2021). We imported data using the googlesheets4 (Bryan, 2020) and googledrive (D'Agostino McGowan and Bryan, 2020) R packages and used tidyverse (Wickham et al., 2019) packages to format data for analysis. The tbeptools R package (Beck et al., 2021b) was used to import and summarize long-term monitoring data (EPC water quality data and seagrass transect data). The NADA R package (Lee, 2020) was used for analysis of censored data. All spatial analyses were done using the simple features (sf) R package (Pebesma, 2018). The mgcv R package (Wood, 2017) was used to create the GAMs for water quality parameters. All datasets used in this study are available from an open access data archive hosted on the Knowledge Network for Biocomplexity (Beck, 2021). Materials for reproducing the analyses, figures, tables, and other content in this paper are provided in a GitHub repository. Finally, the Piney Point Environmental Monitoring Dashboard can be used to view all data included in this paper through an interactive, online application (Beck et al., 2021a). Links and details are provided in supplement.

3. Results

3.1. Water quality trends

Water quality conditions in the northern gypstack measured in 2019 and measured directly at the point of discharge in 2021 showed concentrations that were generally much higher for key water quality parameters as compared to baseline conditions in Tampa Bay (Table 1). Notably, total ammonia nitrogen was measured at 210 mg/L at Piney Point and in the discharge, compared to a long-term median of 0.02 mg/L in lower Tampa Bay. Similar differences for total phosphorus, TN, and chl-a were observed when comparing stack conditions with those of the ambient conditions in Tampa Bay.

Table 1
Measured concentrations from the phosphogypsum stack (NGS-S) at Piney Point from a 2019 sample and samples from April 2021 for relevant water quality variables. Values are compared to normal annual medians (min, max) for concentrations in lower Tampa Bay. Normal medians are based on data for a baseline period from 2006 to 2020 from long-term monitoring stations in lower Tampa Bay (Fig. 1a). The 2021 samples are from the NGS-S stack on April 13th and directly from the outflow site at Port Manatee on April 6th. Missing values were not measured in the stack water or release water.

Water quality variable	2019 stack value	2021 stack value	2021 pipe value	2006–2020 lower Tampa Bay median (min, max)
Nitrate/Nitrite (mg/L)	0.004	0.292	0.004	0.012 (0.007, 0.014)
NH3, NH4+ (mg/L)	210	–	210	0.019 (0.007, 0.039)
TN (mg/L)	230	–	220	0.288 (0.226, 0.385)
TP (mg/L)	160	161	140	0.082 (0.058, 0.145)
Ortho-P (mg/L)	150	155	140	0.049 (0.029, 0.055)
DO (% sat.)	107.5	–	–	90.7 (86, 92)
pH	4	–	–	8.1 (8, 8.1)
Chl-a (μg/L)	–	105	–	3.1 (2.3, 3.5)

Samples collected in the bay between April through September 2021 indicated that water quality conditions were outside of normal values expected for each month. A total of 7831 samples were collected and analyzed for chl-a, dissolved oxygen, TN, total phosphorus, total ammonia nitrogen, nitrate/nitrite, pH, salinity, Secchi depth, and temperature (Table 2). The percentage of observations outside of the normal range (mean $+/-1$ standard deviation from long-term data) varied by location and parameter. For chl-a, 50% of the observations from April through September were above the normal range for Area 1 located closest to the discharge point, whereas only 6% and 22% were above for Areas 2 (to the north) and 3 (to the south), respectively. TN concentrations were above the normal range for 37% of observations in Area 1, whereas concentrations were above for 22% of observations in Area 2 and 22% in Area 3. Secchi observations were below the normal range for 41% of observations in Area 1 and for 18% and 36% of observations in Areas 2 and 3. Notable differences were also observed for dissolved oxygen (e.g., 53% were above in Area 1, 44% in Area 2). Physical parameters (salinity, temperature) and inorganic nitrogen (ammonia, nitrate/nitrite) were more often in normal ranges, although initial time series showed much higher concentrations for ammonia in April near Area 1. Ammonia concentrations near the point of discharge were observed in excess of 10 mg/L in April, about three orders of magnitude above baseline (Figs. S2, S3), similar to the discharge measurements in Table 1. Inorganic nitrogen did not persist at high concentrations past April as it was likely rapidly utilized by phytoplankton (see below). Spatial variation among the parameters showed that values were generally above the normal range (or below for Secchi depth) for many locations near Piney Point (Area 1), Anna Maria Sound (Area 3), and the northern mouth of Tampa Bay (Area 3, Fig. 2).

TN, chl-a, and Secchi depth followed temporal progressions in 2021 that were distinct from long-term seasonal trends estimated from historical data (Fig. 3). For Area 1, TN and chl-a concentrations were frequently above normal ranges during April. Chl-a concentrations were observed in excess of 50 μg/L, although median concentrations for each week in April were $<$10 μg/L. The initial chl-a peak was associated with a localized phytoplankton bloom generally dominated by diatoms. The initial diatom bloom did not persist past April. Chl-a concentrations decreased slightly until June and July when values increased again above the seasonal expectation, coincident with an increase in *K. brevis* concentrations to bloom levels. Many Secchi observations in Area 1 were

Table 2

Summary of water quality variables collected in Tampa Bay from April through September 2021 following the release of water from Piney Point. Variables are grouped by major areas of interest for evaluating status and trends shown in Fig. 1a. Summaries are median, minimum, and maximum values. Total observations (N obs.) and the percentage of observations in range, above, or below normal ranges are also shown. Normal ranges are defined as within ±1 standard deviation of the mean for the month of observation from 2006 to 2020 for values collected at the nearest long-term monitoring site to each sample location. The final column shows the percentage of total observations that were outside of detection, defined as minimum laboratory detection limits for all parameters and values on the bottom for Secchi observations. Medians denoted by "–" could not be calculated due to insufficient values above detection.

Area	Water quality variable	Med. (min., max.)	N obs.	% in range	% above	% below	% outside detection
1	Chl-a (μg/L)	4.3 (1.1, 265.01)	485	44	50	6	0
	DO (% sat.)	97.9 (28.3, 215.3)	430	30	53	17	0
	NH3, NH4+ (mg/L)	0.005 (0, 14.86)	495	66	18	17	26
	Nitrate/ Nitrite (mg/L)	0 (0, 0.14352)	517	63	19	18	70
	pH	8.1 (6.8, 9.1)	476	58	29	14	0
	Sal (ppt)	30.2 (12.9, 34.6)	441	83	4	13	0
	Secchi (m)	2.4 (0.4, 9.5)	350	37	22	41	25
	Temp (C)	25.5 (19.6, 32.9)	442	66	15	19	0
	TN (mg/ L)	0.41 (0.178, 5.6)	429	59	37	4	4
	TP (mg/ L)	0.12 (0.019, 3.9)	485	81	15	4	1
2	Chl-a (μg/L)	2.7 (1.08, 42)	78	60	6	33	0
	DO (% sat.)	95 (60.6, 153.3)	73	42	44	14	0
	NH3, NH4+ (mg/L)	0.004 (0.002, 0.071)	76	86	1	13	21
	Nitrate/ Nitrite (mg/L)	– (0.00078, 0.037)	87	63	18	18	79
	pH	8 (7.3, 8.6)	92	72	16	12	0
	Sal (ppt)	27.3 (18.1, 32.3)	73	90	0	10	0
	Secchi (m)	2 (0.5, 3.5)	44	41	41	18	39
	Temp (C)	25.3 (19.9, 31.6)	73	73	7	21	0
	TN (mg/ L)	0.344 (0.068, 1.13)	63	65	22	13	14
	TP (mg/ L)	0.1 (0.05, 0.235)	67	60	12	28	0
3	Chl-a (μg/L)	2.9 (0.93, 25.9)	254	69	22	9	0
	DO (% sat.)	98.7 (42.4, 229.9)	223	53	26	21	0
	NH3, NH4+ (mg/L)	0.003 (0.002, 0.041)	248	55	0	45	50
	Nitrate/ Nitrite (mg/L)	– (0.00078, 0.046)	267	60	9	31	89
	pH	8.1 (6.2, 9.8)	245	70	21	9	0
	Sal (ppt)	31.8 (1.4, 36.5)	294	81	8	11	0
	Secchi (m)	1.9 (0.2, 5.5)	225	46	17	36	11
	Temp (C)	27 (19.6, 32.1)	294	64	13	24	0
	TN (mg/ L)	0.33 (0.152, 1.78)	249	73	22	5	10
	TP (mg/ L)	0.06 (0.019, 0.589)	256	78	11	12	17

lower than normal in April and July. Observations in Areas 2 and 3 were more often within the normal seasonal range, with some exceptions for TN and chl-a in Area 3 in April, May, and July. These field-based observations were in line with remotely-estimated chl-a using satellite observations. These observations showed an initial bloom on April 5, which peaked on April 9 with a bloom area of about 25 km^2 (about 10 km alongshore and 2.5 km cross-shore) in Area 1 of Fig. 1a, with chl-a ranging between 5 and 40 μg/L. The bloom disappeared on April 12 but reappeared on April 15 at the same location, then disappeared after April 22. Notably, similar blooms at this location were not observed from satellite in the month of April since Sentinel-3 satellite data became available in 2016. Clearly, the bloom was induced by the wastewater discharge, but localized and also short lived.

Statistical comparisons between months for seasonally-corrected observations of TN, chl-a, and Secchi depth (Table 3) supported the results in Fig. 3. Kruskal-Wallis tests that assessed if at least one of the months had significantly different observations for each parameter were significant ($p < 0.05$) for TN, chl-a, and Secchi depth for Areas 1 and 3 and for TN and chl-a for Area 2 (Table 3). Further analysis with multiple comparison tests generally showed that April/May were different from June/July depending on Area and parameter, such that observations in the later months were generally higher (or lower for Secchi) corresponding to increasing *K. brevis* abundances by mid-summer.

3.2. Macroalgae and seagrass trends

A total of 38 transects were sampled for macroalgae and seagrass from April through September, each visited on average 1.7 times per month. Macroalgae observed along the transects varied in coverage, with red macroalgae groups having the highest frequency occurrence of 57%. Common taxa in the red group included genera *Gracilaria* and *Acanthophora*. Green macroalgae and filamentous cyanobacteria were less common, with frequency occurrences of 7% and 13%. Common taxa in the green group included genera *Ulva* and *Caulerpa*, whereas cyanobacteria biomass was dominated by the benthic filamentous genus *Dapis*. Brown macroalgae (primarily in the genus *Feldmannia*) were only observed at one transect in April (2% frequency occurrence). For seagrasses, turtle grass (*Thalassia testudinum*) was the dominant species with frequency occurrence of 50% across all locations and sample dates. Manatee grass (*Syringodium filiforme*) and shoal grass (*Halodule wrightii*) had similar coverage across all transects, with frequency occurrences of 31% and 33%, respectively. The frequency occurrences of seagrasses near Piney Point were similar to the long-term record of seagrass transect data available for Tampa Bay (Sherwood et al., 2017, also see https://shiny.tbep.org/seagrasstransect-dash), with turtle grass being the dominant species in more euhaline waters closer to the Gulf. There is no historical macroalgae record for Tampa Bay that is comparable to the spatial and temporal resolution of the 2021 samples. Only annual historical data are available for seagrasses, with no seasonal data comparable to the results herein.

A typical temporal pattern for macroalgae and seagrass observed at

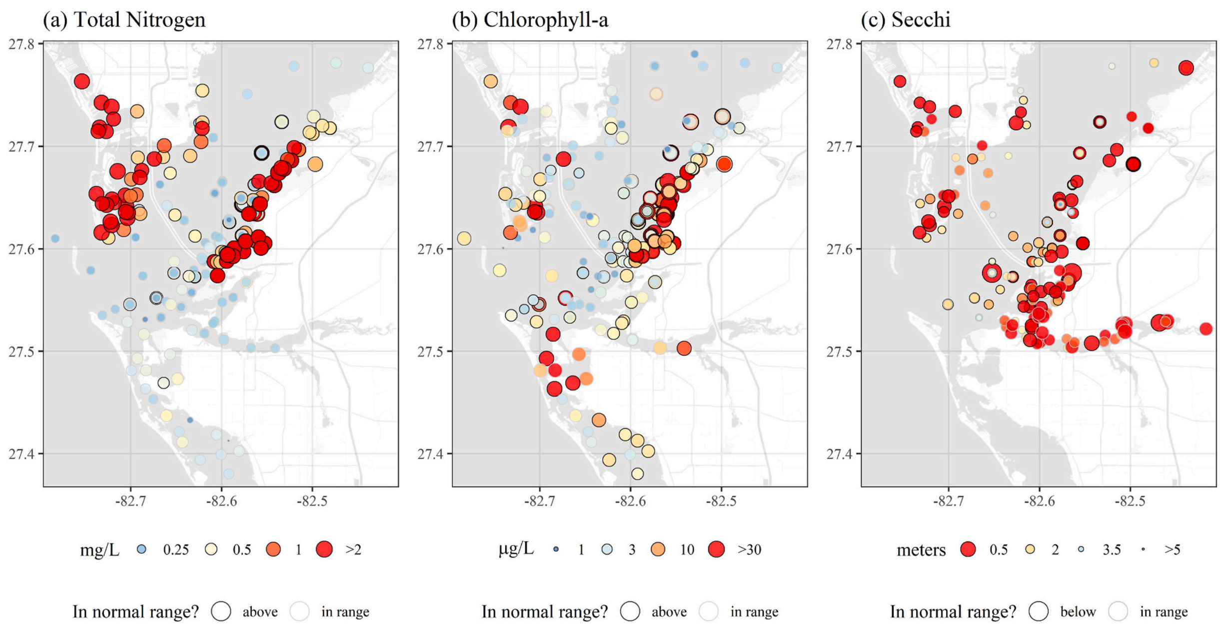

Fig. 2. Water quality data (raw observations) for April through September 2021 following the release from Piney Point for (a) total nitrogen (mg/L), (b) chlorophyll-a (μg/L), and (c) Secchi disk depth (meters). Values outside of the normal range (above for total nitrogen and chlorophyll-a, below for Secchi) are outlined in black and those in normal range are outlined in light grey. Color ramps and point sizes show relative values (reversed for Secchi). Normal ranges are defined as within ±1 standard deviation of the mean for the month of observation from 2006 to 2020 for values collected at the nearest long-term monitoring site to each sample location (Fig. 1a). Values below detection limits (or Secchi on bottom) are not shown. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

many of the transects is shown in Fig. 4, using transect S3T6 near Port Manatee as an example. Macroalgal abundances changed over the course of sampling similar to the remainder of transects sampled during the study. Red macroalgae were present in high abundances from April to May. Filamentous cyanobacteria (*Dapis* spp.) mats were first observed on May 24th and was present at all of the sample locations along this transect on June 4th and 15th. Filamentous cyanobacteria persisted through June and July, but was not observed in abundance after July 20th. Green macroalgae taxa were first observed in July, although at generally low abundances. Red macroalgae were the dominant taxa by the end of September. Overall abundance of seagrass did not change from April 22nd through September. The site is dominated by manatee grass that was observed at nearly all of the sample points along the transect at varying coverages.

Monthly summaries in frequency occurrence by area (Fig. 5) provided an indication of macroalgae and seagrass trends in 2021 across all transects. No transects were sampled in Area 2 to the north of Piney Point and no transects were sampled past September in Area 1 given allocated sampling effort following projected dispersal patterns of the discharge from the TBCOM simulations. Red macroalgae was the dominant group across all months and areas, with the highest frequency occurrences observed in April (81% in Area 1, 95% in Area 3). Reductions in red macroalgae frequency occurrence were observed in June when cyanobacteria frequency occurrence peaked, with greater coverage of cyanobacteria in Area 3 (43%) compared to Area 1 (36%). Notable blooms of the filamentous cyanobacteria (*Dapis* spp.) were observed in Anna Maria Sound (Area 3) and near Port Manatee (Area 1) (Fig. 1), typically observed covering benthic and seagrass habitats, in addition to large floating mats on the surface. Green macroalgae had the second lowest frequency occurrence, although it increased slightly by the end of the study period (9% in September in Area 1, 31% in October in Area 3). For seagrass, both areas had generally stable total frequency occurrence. Turtle grass (*T. testudinum*) occurred in higher frequency occurrence in both areas (45% overall in Area 1, 58% overall in Area 3), compared to shoal grass (*H. wrightii*, 31% Area 1, 38% Area 3) and manatee grass (*S. filiforme*, 30% Area 1, 31% Area 3). Slight changes in frequency occurrence in Area 3 were observed for all species starting in July, with a slight reduction in frequency occurrence of turtle grass and an increase in shoal grass and manatee grass. Statistical analyses with multiple comparison tests confirmed the general trends described above, with significant changes observed over time only for macroalgae (Tables S1, S2). Tests using Braun Blanquet cover estimates confirmed the results from the frequency occurrence estimates (Tables S3, S4).

3.3. Red tide impacts

On April 20th, the HAB species *Karenia brevis* was observed near Anna Maria Sound at the southern edge of the mouth of Tampa Bay. This first Tampa Bay influx likely originated from an ongoing coastal bloom in the Gulf of Mexico, as is common when red tide is observed in the bay (Flaherty and Landsberg, 2011; Steidinger and Ingle, 1972). By May 23, bloom concentrations of *K. brevis* were observed in lower Tampa Bay (lower/middle bay boundary Fig. 1b), with concentrations peaking (10^6 to 10^7 cells/L) by the week of July 4th in middle Tampa Bay, after which concentrations declined (Fig. 6b). The increase in *K. brevis* from April to July was an anomaly in 2021 that is not regularly observed in Tampa Bay. The historical record from 1953 to present (Fig. 6a) shows cell concentrations sampled in Tampa Bay between April and September, with only a few years having cell concentrations $>10^5$ cells/L, notably 1963, 1971, 2005, 2018, and 2021. Median cell concentrations for most years were well below 1000 cells/L. The two highest concentrations in the long-term record were observed in 1971 (20 million cells/L) and 2021 (17.6 million cells/L), both being over an order of magnitude above the high category. Cumulative rainfall and associated inflow from the main rivers entering Tampa Bay in 2021 were below historical values (2006–2020) in the months preceding the highest bloom concentrations (i.e., January to June, Fig. 6c, d). This likely contributed to elevated salinity in lower and middle Tampa Bay that created conditions favorable for *K. brevis* growth in 2021 (Figs. S2f, S3f), in addition to the elevated nutrient concentrations from the Piney Point discharge.

Fish kill reports attributed to *K. brevis* at the cities of Tampa and Saint Petersburg, FL closely tracked cell concentrations during June and July

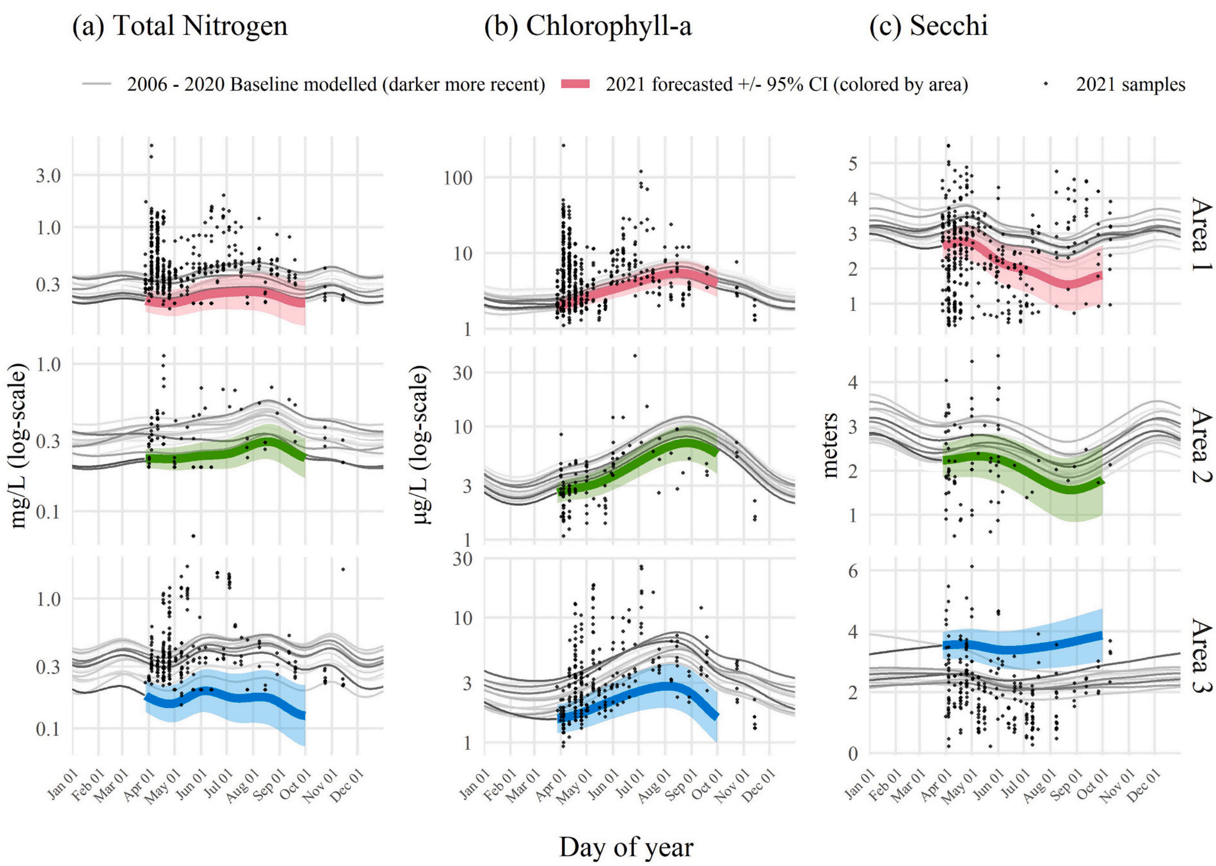

Fig. 3. Expected 2021 (a) total nitrogen (mg/L), (b) chlorophyll-a (μg/L), and (c) Secchi disk depth (meters) by area based on historical seasonal models. Predictions (expected values) from the historical models for dates during and after the Piney Point release are shown in thick lines (+/−95% confidence), with observed samples overlaid on the plots to emphasize deviation of 2021 data from historical seasonal estimates. Expected values are based on Generalized Additive Models fit to historical baseline data from 2006 to early 2021, where historical predictions are shown as thin grey lines, with darker lines for more recent years. Results are grouped by assessment areas shown in Fig. 1a.

2021 (Fig. 6e). In total, 331 reports were made in Saint Petersburg and 65 in Tampa. The combined weekly reports in 2021 for Tampa and Saint Petersburg peaked the week of July 4th, the same week as the peak of *K. brevis* cell concentrations (Fig. 6b). Notably, all of the fish kill reports occurred within a 1.5 month period when *K. brevis* cell concentrations were consistently above the medium threshold (10^4 cells/L). The center of Tropical Storm Elsa (Fig. 6f, pre-, post-storm wind roses) also passed through the bay area on July 5th, causing a shift in winds that likely disturbed the water column and altered the spatial distribution of *K. brevis* in the bay. Strong southeasterly winds also likely moved dead fish closer to heavily populated areas of Tampa Bay, specifically near St. Petersburg and Tampa, contributing to an increase in fish kill reports. It is important to note that high cell concentrations (>10^6 cells/L) were observed in middle Tampa Bay (Fig. 6b) and fish kills were reported both before and after storm passage (Fig. 6e). By August, cleanup efforts removed over 1600 metric tons of dead fish near public and private shoreline areas (K. Hammer Levy, Pinellas County, pers. comm. Aug. 2021).

4. Discussion

The observed conditions in Tampa Bay in 2021 following releases from Piney Point provide multiples lines of evidence for an adverse environmental response to a large pulse of inorganic nitrogen into the system. Collectively, these observations show that conditions in 2021 were anomalous when compared to long-term monitoring data for Tampa Bay, although some of the anomalies may not be related to the Piney Point release. These anomalous events (Fig. 7) included 1) a large diatom bloom (~25 km^2, chl-a between 5 and 40 μg/L) in April in the vicinity of the release at Port Manatee, 2) high abundance of filamentous cyanobacteria in Anna Maria Sound and near Port Manatee, 3) medium to high bloom concentrations of the ride tide organism *K. brevis* in lower and middle Tampa Bay from June through July, and 4) high incidence of fish kill reports prompting local governments to remove over 1600 metric tons of dead fish from shoreline areas. The water quality conditions observed during the study period, particularly for TN, chl-a, and Secchi depth, were outside of normal seasonal ranges for many of the observations (Fig. 2, Table 2). The Piney Point event also represented an anomalous volume and load of labile nitrogen released directly into lower Tampa Bay. Spill events reported to FDEP (e.g., industrial spills, service line failures, sanitary sewer overflows) provide additional context for Piney Point relative to other potential anomalous releases to Tampa Bay. An assessment of over 800 reports to FDEP for the Tampa Bay watershed over the last five years showed spill volumes for these events are small (median volume 13.7 thousand liters TBEP unpublished analysis) compared to the 814 million liters released from Piney Point. Moreover, the estimated nutrient load of 186 metric tons of nitrogen to Tampa Bay from Piney Point over the ten day period, exceeded current annual estimates of all external loading sources into lower Tampa Bay (Janicki Environmental, Inc., 2017). External nitrogen loads to lower Tampa Bay averaged 164 metric tons per year for the baseline period of 2006 to 2020 (https://tbep-tech.github.io/load-estimates/).

4.1. Potential nutrient cycling

The events of 2021 can be considered together to develop a narrative

Table 3
Comparison of total nitrogen, chlorophyll-a, and Secchi depth by areas of interest (Fig. 1a) and month. Overall significance of differences of concentrations between months for each water quality variable and area combination are shown with Chi-squared statistics based on Kruskall-Wallis rank sum tests. Multiple comparisons with Mann-Whitney U tests (Comp. column) were used to evaluate pairwise monthly concentrations for each water quality variable in each area. Rows that share letters within each area and water quality variable combination have concentrations that are not significantly different between month pairs. All statistical tests were performed on the seasonally-corrected water quality values that were based on observations with the long-term monthly median subtracted (observed medians are shown for comparison). $^{**}p < 0.005$, $^{*}p < 0.05$, blank is not significant at $\alpha = 0.05$.

Area	Water quality variable	Chi-Sq.	Comp.	Month	N obs.	Observed median	Seasonally-corrected median
1	TN (mg/L)	25.01**	a	Apr	135	0.390	0.008
			b	May	32	0.360	0.110
			ab	Jun	38	0.430	0.112
			b	Jul	24	0.520	0.178
			ab	Aug	25	0.470	0.065
			ab	Sep	8	0.390	0.075
	Chl-a (μg/L)	61.84**	a	Apr	144	3.300	1.010
			b	May	32	2.400	−0.870
			a	Jun	38	6.600	1.960
			a	Jul	24	5.600	0.310
			c	Aug	27	3.300	−3.590
	Secchi (m)	47.47**	a	Apr	118	2.900	0.000
			b	May	28	3.000	−0.600
			b	Jun	34	2.000	−0.900
			b	Jul	18	2.000	−0.700
			c	Aug	15	3.500	0.400
			c	Sep	12	3.600	0.900
2	TN (mg/L)	20.85**	a	Apr	18	0.390	−0.002
			b	May	4	0.390	0.160
			ab	Jun	3	0.500	0.113
			ab	Jul	3	0.510	0.097
			ab	Aug	3	0.540	0.174
			ab	Sep	1	0.570	0.049
	Chl-a (μg/L)	10.76*	a	Apr	22	2.500	−1.390
			a	May	4	2.150	−2.590
			a	Jun	4	6.000	−1.050
			a	Jul	3	7.200	−0.940
			a	Aug	3	5.200	−4.940
	Secchi (m)	3.82	a	Apr	17	2.000	0.200
			a	May	1	2.000	0.500
			a	Jun	3	2.100	0.700
			a	Jul	1	1.400	−0.100
3	TN (mg/L)	22.13**	a	Apr	48	0.330	−0.010
			b	May	16	0.335	0.079
			ab	Jun	10	0.350	−0.087
			ab	Jul	12	0.365	0.043
			ab	Aug	4	0.435	0.126
			ab	Sep	7	0.380	0.023
	Chl-a (μg/L)	33.62**	ab	Apr	48	1.900	−0.900
			ac	May	16	2.350	−0.450
			b	Jun	12	2.800	−1.580
			cd	Jul	8	4.150	0.770
			bd	Aug	4	3.200	−3.100
			abcd	Sep	8	3.600	−1.500
	Secchi (m)	8.77	a	Apr	41	2.700	0.000
			a	May	16	2.200	−0.500
			a	Jun	12	2.200	−0.400
			a	Jul	12	2.200	−0.100
			a	Aug	3	2.000	−0.800
			a	Sep	11	2.200	0.000

of the temporal shift of nutrient pools between ecosystem components of the bay from April through September, starting with the influx of inorganic nitrogen from Piney Point. TN concentrations first peaked in April (Fig. 8a), as did chl-a concentrations (Fig. 8b). The initial peak in water quality parameters suggested a rapid response of the phytoplankton community as an increase in diatoms (e.g., centric species, such as *Skeletonema* sp., and also *Asterionellopsis* sp., Fig. 8c) that can readily utilize inorganic forms of nitrogen that were present in the initial discharge (Bates, 1976; Domingues et al., 2011). These results were evidenced by taxonomic enumeration of phytoplankton samples collected near Port Manatee. Water quality indicators improved slightly following the decrease in diatoms in late April, as noted by relatively lower concentrations of TN and chl-a as the bloom dispersed. However, filamentous cyanobacteria biomass increased after the initial diatom bloom and peaked in June (Fig. 8d), suggesting a shift of nutrients from phytoplankton to drift macroalgae communities or changing availability of nutrient ratios creating favorable conditions for macroalgae growth (Cohen and Fong, 2006; Valiela et al., 1997). During peak macroalgae growth, TN and chl-a concentrations remained relatively low as nutrients were likely retained in macroalgae, until late June and early July when *K. brevis* concentrations peaked (Fig. 8e). The co-occurring decline in macroalgae and increase in *K. brevis* suggests a release of nutrients from the former that could have stimulated growth of the latter, although residual nutrients from the initial release from Piney Point were likely still available (Liu et al., 2021). Finally, conditions were relatively stable in August and September with relatively improved water quality conditions and no dominant algal blooms.

Our quantitative results provide some evidence to support the progression of events outlined above as a flow of nutrients over time. The distinct temporal progression can be readily identified through an

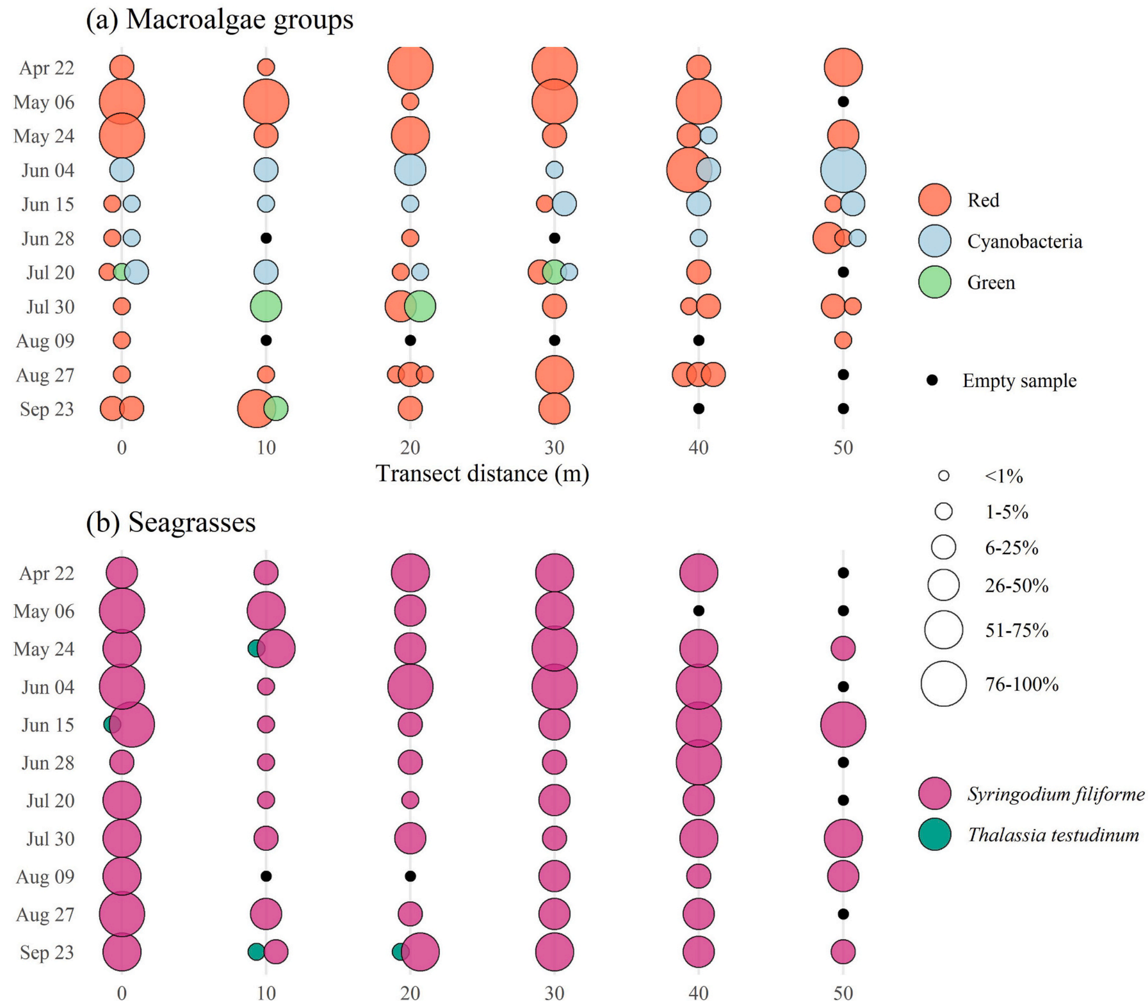

Fig. 4. Results for (a) macroalgae and (b) seagrass rapid response transect surveys at a site (S3T6, −82.55866 W longitude, 27.64483 N latitude) near Piney Point. Sample dates in 2021 are shown in rows with transect meter results shown in columns (0 m nearshore, 50 m offshore). Results show dominance of manatee grass (*Syringodium filiforme*) and red macroalgae groups, with abundances of *Dapis* spp. (cyanobacteria) peaking in June and green macroalgae (*Ulva* spp.) increasing in July. Abundances are Braun-Blanquet coverage estimates. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

ordination plot (Fig. S7) for the observed data in Fig. 8. Weekly summaries of the data are clearly separated in the ordination into monthly groups where different communities were dominant and is partially explained by orientation of the water quality vectors relative to cyanobacteria, diatoms, and *K. brevis*. For example, TN and chl-a are strongly aligned with the *K. brevis* axis as nutrients were likely available in organic form during the peak of the red tide event. However, this simple analysis only demonstrates an association in the observed data and cannot be verified without additional information. Additional data to support these results could include explicit load-based estimates for all sources entering the bay through 2021 and these estimates are forthcoming. Laboratory-based methods, such as isotopic analyses of nutrient signatures found in biological tissues (e.g., macroalgae) compared to those from the release, could provide a more comprehensive description of the recycling of nitrogen from Piney Point. Additional confounding variables can also obscure the association between water quality and community changes. Bay conditions preceding the 2021 events, as well as the passage of tropical storm Elsa, could obscure these associations (described below).

Several of the water quality responses are consistent with observations of nutrient loading in other shallow Gulf Coast estuaries (Caffrey et al., 2013; Doering et al., 2006; Greening et al., 2014). The relationship between nutrients, chl-a, and water transparency followed expectations of reduced water quality with increased nutrient loads. Temporally, these changes were observed at different times and for different species of phytoplankton. The initial increase in chl-a was first associated with a diatom bloom in April. The red tide species *K. brevis* was also first introduced to Tampa Bay from the Gulf of Mexico in April, but was not observed at high densities in the Bay until June and July. Peaks in dissolved oxygen saturation were also observed as an indicator of elevated phytoplankton production (Kemp and Boynton, 1980), particularly in July with the peak *K. brevis* bloom (Figs. S2d, S3d). Of note is that inorganic species of nitrogen, mainly ammonia, were only present at high concentrations in early April. Management concerns of the negative impacts of nutrients on water quality focused primarily on the high concentrations of ammonia in the discharge (Table 1), which can be utilized rapidly by many phytoplankton taxa (Bates, 1976; Domingues et al., 2011). Low concentrations of ammonia after April may be explained by quick uptake by the initial diatom bloom, where TN that included particulate and dissolved organic sources was at high concentrations through April and again peaked in July. Variation in observed concentrations of nutrients is complex given that high concentrations may suggest availability to support phytoplankton growth, whereas low concentrations may imply cycling of available nitrogen in organic forms already utilized by different taxa, including macroalgae (Cohen and Fong, 2006; Valiela et al., 1997).

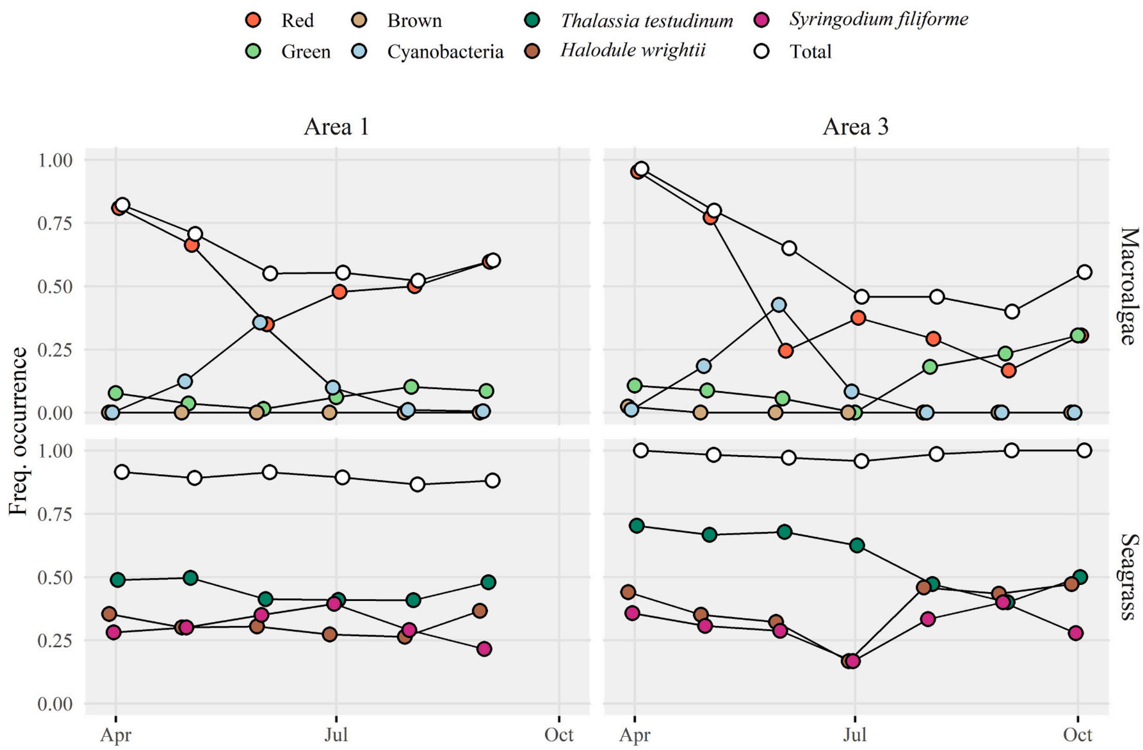

Fig. 5. Frequency occurrence estimates for (a) Area 1 and (b) Area 3 (see map Fig. 1a for locations) for macroalgae (top) and seagrass (bottom) rapid response transect surveys across all transects (n = 38). Estimates are grouped by sample months in 2021. Frequency occurrences are absolute for each taxon based on presence/absence, whereas the total frequency occurrence applies to any taxa observed on each transect. Points are offset slightly for readability. No transects were sampled in Area 2 to the north of Piney Point and no transects were sampled past September in Area 1 given allocated sampling effort following projected dispersal patterns of the plume from model simulations.

4.2. Additional interpretation of impacts

Previous research for Tampa Bay has identified water quality conditions that are likely to promote seagrass growth (Greening et al., 2014, and references therein; Greening and Janicki, 2006). Water quality results in 2021 suggested that conditions may have been light-limiting for seagrass growth (e.g., high chl-a concentrations, low Secchi observations), although the conditions likely did not persist long enough to impact seagrasses. The long-term effects of the Piney Point discharge on the seagrass community remains uncertain. From 2018 to 2020, seagrass coverage declined by 16% in Tampa Bay, with similar losses observed in Sarasota Bay (18%), Lemon Bay (12%), and Charlotte Harbor (23%) to the south (Southwest Florida Water Management District, unpublished results). These broader trends suggest regional drivers are affecting seagrass communities (e.g., variation in precipitation, Tomasko et al., 2020), yet local issues specific to individual bays also pose challenges to managing water quality and subtidal habitats. Recent seagrass losses in Sarasota Bay may be linked to decreased light availability from a persistent *K. brevis* bloom in 2018. Although the 2021 red tide in Tampa Bay was short-lived, potential long-term effects on seagrasses remain a concern (e.g., alteration of sediment geochemistry, Eldridge et al., 2004). Ecosystem shifts from seagrass to macroalgae dominated communities are also a concern, both in 2021 and as observed at some locations in recent years from annual transect monitoring results for Tampa Bay. In particular, increasing abundance in recent years of the green algae *Caulerpa* sp. has been observed at long-term transects that were previously dominated by seagrass. These changes may be indicative of broader ecosystem shifts concurrent with alteration of nutrient loads or system resilience at the expense of seagrass communities (Lloret et al., 2005; Stafford and Bell, 2006). Acute stressors from short-term events, such as unanticipated releases from Piney Point, create additional and often preventable challenges to managing seagrass health.

Macroalgae trends across the study period were much more dramatic than the minimal changes observed in the seagrass community. This was expected given both the documented changes from past releases from Piney Point (Switzer et al., 2011) and the more rapid response of macroalgae to changing water quality conditions relative to seagrasses (Valiela et al., 1997). In Tampa Bay, red macroalgae groups (e.g., *Gracilaria* spp., *Acanthophora* sp.) are more common than green macroalgae (e.g., *Ulva* spp., *Caulerpa* spp.) and occur earlier in the growing season. The dominance of the red groups early in the summer followed by an increase in the green alga *Ulva* spp. may reflect a natural phenology in Tampa Bay. The most notable change in the macroalgal community in 2021 was a high abundance of filamentous cyanobacteria (i.e., *Dapis* spp.) in May and June. High abundances of *Dapis* spp. were observed in Anna Maria Sound near the mouth of Tampa Bay and near Port Manatee at the release site, which is uncommon at these locations. Long-term monitoring data describing normal seasonal variation in macroalgae are unavailable and we cannot distinguish between seasonal and interannual changes and those in potential response to the Piney Point release. Filamentous cyanobacteria has been observed during routine annual transect monitoring in Tampa Bay and it has previously been documented in public reports to the Florida Department of Environmental Protection. However, these communities can respond rapidly to external nutrient inputs (Ahern et al., 2007; Albert et al., 2005), often exhibiting lagged responses with characteristic growth/decay periods similar to observations herein (Estrella, 2013), and it is not unreasonable to expect these trends to be related to nutrients from Piney Point. Although long-term seasonal data are unavailable for comparison, anecdotal reports suggested that the observed biomass in 2021 was very unusual (R. Woithe, Environmental Science Associates, pers. comm. Dec. 2021).

There were also concerns that the release from Piney may have contributed to the persistence and intensity of *K. brevis*, having negative

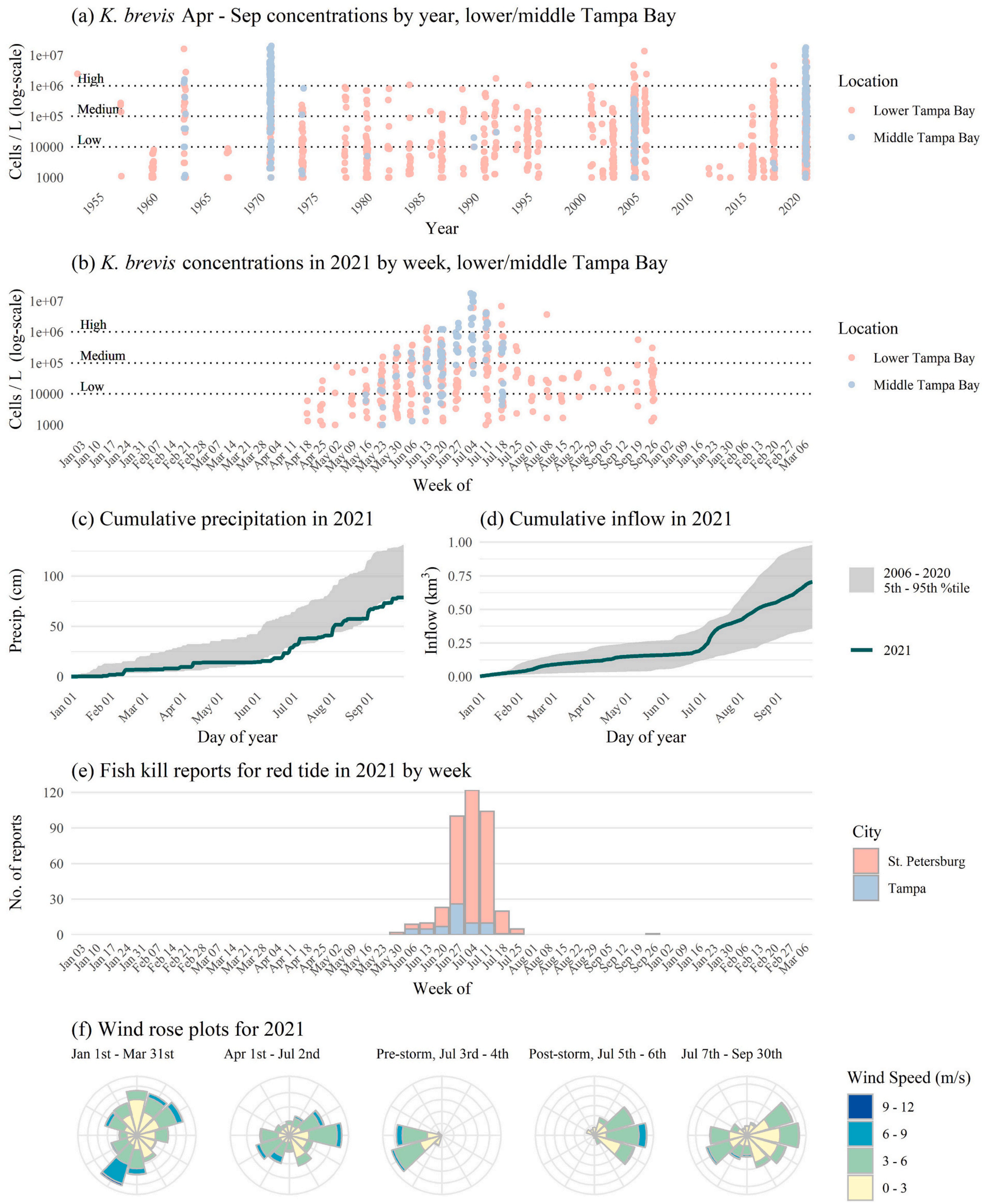

Fig. 6. *Karenia brevis* concentrations (cells/L) (a) by year and (b) by week in 2021, (c) cumulative precipitation in 2021 compared to past years, (d) cumulative inflow in 2021 compared to past years, (e) fish kill reports in 2021, and (f) wind rose plots for 2021 with notable breaks before/after Piney Point release and tropical storm Elsa. Wind roses show relative counts of six minute observations in directional (30 degree bins, north is vertical) and speed (m/s) categories.

effects on fisheries resources in June and July (Fig. 6). Fisheries resources in Tampa Bay have previously been negatively affected by red tide (e.g., in 2005, Flaherty and Landsberg, 2011; Schrandt et al., 2021). For past Piney Point events, Switzer et al. (2011) evaluated nekton communities in Bishop Harbor from November 2003 to October 2004 following discharge to this subembayment. Fish community structure and species composition did not differ compared to a pre-impact period, although HAB species (*Prorocentrum minimum*, *Heterosigma akashiwo*), including *K. brevis* and diatoms, were observed in Bishop Harbor during this time (Garrett et al., 2011). Prior blooms in Tampa Bay were more localized and *K. brevis* was at lower abundances in comparison to the 2021 bloom event, potentially mitigating exposure of fishes to related harmful conditions. In Sarasota Bay to the south, fish activity measured by passive acoustic methods was significantly lower during a 2018 red

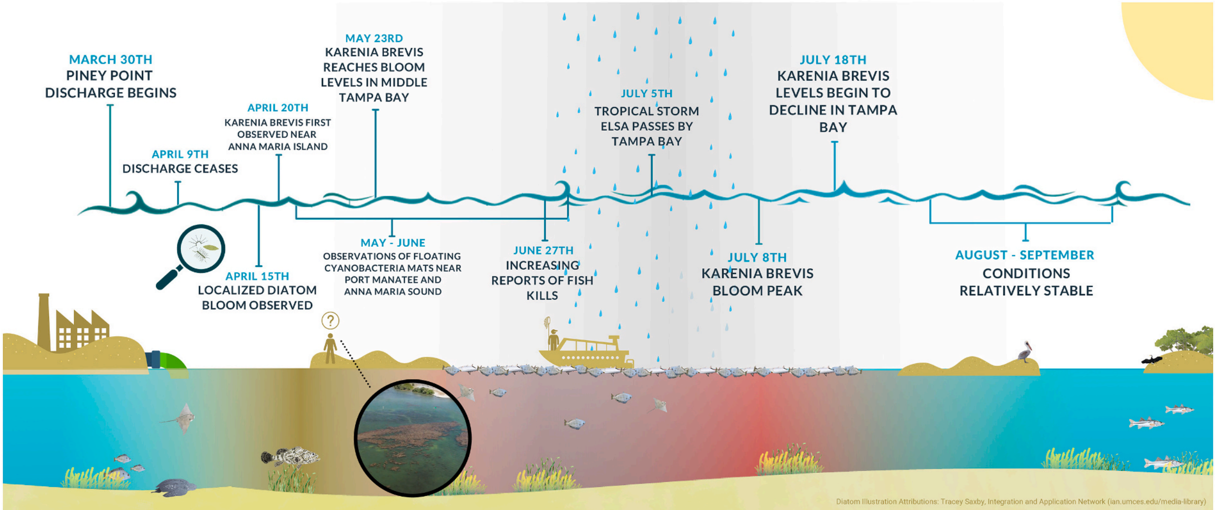

Fig. 7. Graphical timeline of events in Tampa Bay from March 30th through September 2021 following the release from Piney Point. Inset image shows blooms of filamentous cyanobacteria (*Dapis* spp.).

tide event as compared to pre-bloom levels (Rycyk et al., 2020). Water quality conditions before and after passage of tropical storm Elsa may have also contributed to fish kills by reducing bottom-water dissolved oxygen. Stevens et al. (2006) documented impacts of a category 4 storm on fish resources in the Charlotte Harbor estuary, although tropical storm Elsa was much smaller and fish kills were documented prior to and after arrival of the storm. Lack of continuous monitoring data for bottom waters in Tampa Bay prevents a more detailed assessment of impacts of the storm on water quality.

Establishing causal linkages between the nutrient inputs from Piney Point and the severity of the *K. brevis* bloom observed in Tampa Bay this year is difficult in the absence of more quantitative results or mechanistic tools to support understanding. Occurrence of this species has historically been spatially distinct, with blooms originating in subsurface water offshore on the West Florida Shelf (Liu et al., 2016; Steidinger, 1975; Weisberg et al., 2014, 2019) and occasionally occurring at bloom concentrations in lower and middle Tampa Bay. Although bloom concentrations in 2021 were extreme, historical blooms have been observed in Tampa Bay with notable events occurring in 1971 (Steidinger and Ingle, 1972), 2005 (Flaherty and Landsberg, 2011), and recently in 2018 (Skripnikov et al., 2021). Seasonal persistence in Gulf waters in southwest Florida can vary between years, with some blooms lasting as short as a few weeks, while others have been present for longer than a year (the 2018 bloom lasted sixteen months, Skripnikov et al., 2021). Severe *K. brevis* blooms are rarer in estuaries because high abundances are most common at higher salinities typical of coastal or oceanic waters (Steidinger et al., 1998; Villac et al., 2020). Contributing factors in 2021, such as low rainfall preceding the bloom and varying wind patterns, created conditions that were favorable for growth of *K. brevis* in Tampa Bay. However, the results suggest a likely scenario that residual nutrients from the Piney Point release, or indirectly through nutrients made available from the growth and decomposition of other primary producers (e.g., diatoms, macroalgae) stimulated by inputs from Piney Point, were sufficiently available to allow growth of *K. brevis* to the concentrations observed in July (also see Medina et al., 2020). Daily simulation results from the Tampa Bay Coastal Ocean Model (Chen et al., 2018, 2019) suggested that the plume was widespread throughout the bay and persisted for many months after the release ceased at Port Manatee. Plume dispersal also suggested that both open-water and back-bay habitats were exposed to nutrient concentrations sufficient to stimulate phytoplankton production. Although Piney Point did not cause red tide (i.e., it originates in the Gulf of Mexico), the events of 2021 may have created conditions in Tampa Bay conducive for the extreme bloom concentrations observed in July. Similarly, recent studies have highlighted the role of anthropogenic forcing in increasing bloom intensity in southwest Florida (Medina et al., 2020, 2022).

In the broader context of mining impacts to surface waters, these results reinforce the understanding that legacy pollutants from phosphate mining can negatively affect environmental resources. In addition to Tampa Bay (Garrett et al., 2011; Switzer et al., 2011), other Gulf Coast estuaries have been affected by pollutants from unanticipated gypstack releases. For example, two spills have occurred in Grand Bay, Mississippi, the first in 2005 following failure of the retaining walls after a heavy rain event and the second in 2012 after passage of Hurricane Isaac when the holding capacity of the local gypstack was exceeded again with heavy rainfall (Beck et al., 2018a; Dillon et al., 2015). The historical context of Grand Bay is similar to Piney Point and other international examples, e.g., Huelva estuary in Spain (Pérez-López et al., 2010, 2016). Legacy wastewater from fertilizer production has been poorly maintained at some facilities and long-term plans are insufficient to safely dispose of remnant pollutants that pose a risk of significant impacts to coastal resources that increases over time. These are not isolated examples and enhanced regulatory oversight is needed to safely and effectively close these types of facilities (Nelson et al., 2021). Local, regional, and state partners should continue to pursue management and policy actions that can mitigate the continued threats of these facilities to the health of coastal resources. These efforts are critical to managing Gulf of Mexico ecosystems given past successes and the need to address ongoing threats of climate change, human population growth, habitat loss, severe weather events, and recurring pollutant sources.

CRediT authorship contribution statement

Marcus W. Beck: Conceptualization, Data curation, Formal analysis, Methodology, Writing – original draft, Writing – review & editing. **Andrew Altieri:** Data curation, Writing – review & editing. **Christine Angelini:** Conceptualization, Writing – review & editing. **Maya C. Burke:** Project administration, Supervision, Writing – review & editing. **Jing Chen:** Data curation, Writing – review & editing. **Diana W. Chin:** Data curation, Writing – review & editing. **Jayne Gardiner:** Data curation. **Chuanmin Hu:** Data curation, Writing – review & editing. **Katherine A. Hubbard:** Data curation, Validation, Writing – review & editing. **Yonggang Liu:** Data curation, Writing – review & editing. **Cary Lopez:** Data curation, Validation, Visualization, Writing – review & editing. **Miles Medina:** Formal analysis, Methodology, Visualization, Writing – review & editing. **Elise Morrison:** Data curation, Funding

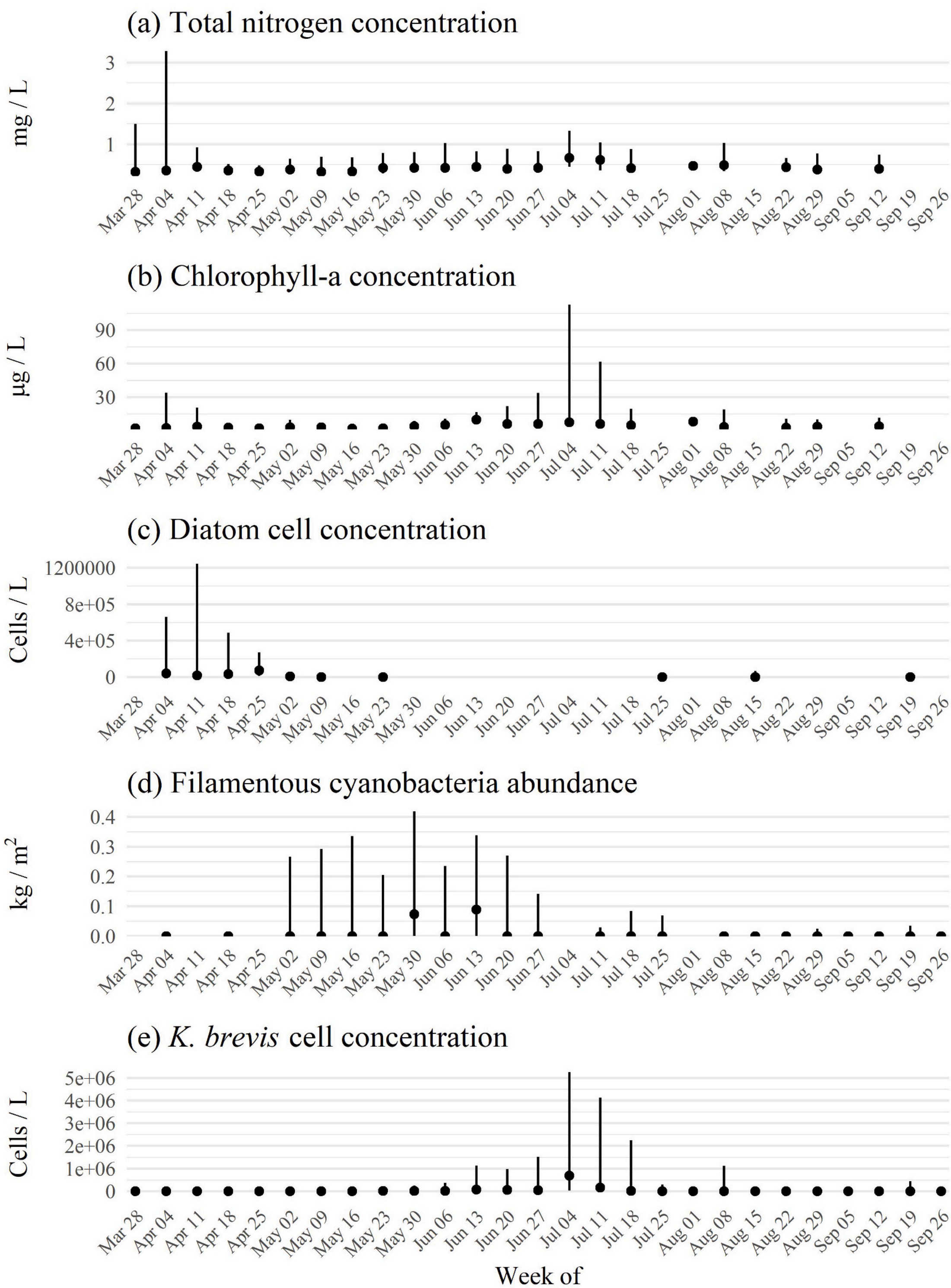

Fig. 8. Weekly summarized observations (medians, 2.5th to 97.5th percentiles) across all sampled locations for (a) total nitrogen concentrations, (b) chlorophyll-a concentrations, (c) diatom cell concentrations, (d) filamentous cyanobacteria abundances, and (e) *Karenia brevis* cell concentrations. Values are summarized for all samples within each week. The values suggest nutrient cycling between water column phytoplankton in the initial April diatom bloom, then to filamentous cyanobacteria in May to June, and then to *K. brevis* peaking in early July. The upper limit of the y-axis on (e) is truncated to emphasize trends. Quantitative cell counts for diatoms are missing for several weeks, but see Fig. S6 for frequency occurrence estimates across all dates. Diatom concentrations are based on combined cell counts from *Asterionellopsis* sp. and *Skeletonema* sp.

acquisition, Writing – review & editing. **Edward J. Phlips:** Data curation, Funding acquisition, Writing – review & editing. **Gary E. Raulerson:** Data curation, Writing – review & editing. **Sheila Scolaro:** Data curation, Writing – review & editing. **Edward T. Sherwood:** Project administration, Supervision, Writing – review & editing. **David Tomasko:** Conceptualization, Data curation, Writing – review & editing. **Robert H. Weisberg:** Data curation, Writing – review & editing. **Joseph Whalen:** Conceptualization, Visualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

We thank the many TBEP partners and collaborators for their continuing efforts to restore and monitor Tampa Bay. We thank the administrative staff, field crews, and laboratory staff from the Florida Department of Environmental Protection, Environmental Protection Commission of Hillsborough County, Parks and Natural Resources Department of Manatee County, Pinellas County Division of Environmental Management, Fish and Wildlife Research Institute of the Florida Fish and Wildlife Conservation Commission, City of St. Petersburg, Tampa Bay Watch, Sarasota Bay Estuary Program, Environmental Science Associates, University of South Florida, University of Florida, and New College of Florida. Funding was provided to the University of Florida from the Ocean Conservancy and the National Science Foundation (project ID 2130675). The progress achieved in restoring the Tampa Bay ecosystem over recent decades would not be possible without the collaborative partnerships fostered in the region. Our partners' willingness to adapt and implement innovative monitoring and management actions in response to Piney Point and the ever-evolving challenges threatening Tampa Bay is greatly appreciated.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2022.113598.

References

Ahern, K.S., Ahern, C.R., Udy, J.W., 2007. Nutrient additions generate prolific growth of Lyngbya majuscula (cyanobacteria) in field and bioassay experiments. Harmful Algae 6, 134–151. https://doi.org/10.1016/j.hal.2006.08.004.

Albert, S., O'Neil, J.M., Udy, J.W., Ahern, K.S., O'Sullivan, C.M., Dennison, W.C., 2005. Blooms of the cyanobacterium Lyngbya majuscula in coastal Queensland, Australia: disparate sites, common factors. Mar. Pollut. Bull. 51, 428–437. https://doi.org/10.1016/j.marpolbul.2004.10.016.

Bates, S.S., 1976. Effects of light and ammonium on nitrate uptake by two species of estuarine phytoplankton. Limnol. Oceanogr. 21, 212–218. https://doi.org/10.4319/lo.1976.21.2.0212.

Beck, M.W., 2021. 2021 piney point sampling in Tampa Bay. In: Knowledge Network for Biocomplexity. https://doi.org/10.5063/F1959G05 urn:node:KNB.

Beck, M.W., Cressman, K., Griffin, C., Caffrey, J., 2018a. Water quality trends following anomalous phosphorus inputs to Grand Bay. GulfCaribb.Res. 29, 1–14. https://doi.org/10.18785/gcr.2901.02.

Beck, M.W., Hagy III, J.D., Le, C., 2018b. Quantifying seagrass light requirements using an algorithm to spatially resolve depth of colonization. Estuar. Coasts 41, 592–610. https://doi.org/10.1007/s12237-017-0287-1.

Beck, M.W., Burke, M.C., Raulerson, G.E., Scolaro, S.C., Sherwood, E.T., Whalen, J.D., 2021. Piney Point Environmental Monitoring Dashboard. tbep-tech/piney-point: v1.0.0. Zenodo. https://doi.org/10.5281/zenodo.4666494.

Beck, M.W., Schrandt, M.N., Wessel, M.R., Sherwood, E.T., Raulerson, G.E., Prasad, A.A. B., Best, B.D., 2021b. Tbeptools: an R package for synthesizing estuarine data for environmental research. J.Open Source Softw. 6, 3485. https://doi.org/10.21105/joss.03485.

Bryan, J., 2020. googlesheets4: Access Google Sheets using the Sheets API V4. R package version 0.2.0. https://CRAN.R-project.org/package=googlesheets4.

Burnett, W.C., Elzerman, A.W., 2001. Nuclide migration and the environmental radiochemistry of Florida phosphogypsum. J. Environ. Radioact. 54, 27–51. https://doi.org/10.1016/S0265-931X(00)00164-8.

Caffrey, J.M., Murrell, M.C., Amacker, K.S., Harper, J., Phipps, S., Woodrey, M., 2013. Seasonal and inter-annual patterns in primary production, respiration and net ecosystem metabolism in 3 estuaries in the northeast Gulf of Mexico. Estuar. Coasts 37, 222–241. https://doi.org/10.1007/s12237-013-9701-5.

Chen, J., Weisberg, R.H., Liu, Y., Zheng, L., 2018. The Tampa Bay Coastal Ocean Model performance for hurricane Irma. Mar. Technol. Soc. J. 52, 33–42. https://doi.org/10.4031/MTSJ.52.3.6.

Chen, J., Weisberg, R.H., Liu, Y., Zheng, L., Zhu, J., 2019. On the momentum balance of Tampa Bay. J.Geophys.Res.Oceans 124, 4492–4510. https://doi.org/10.1029/2018JC014890.

Cohen, R.A., Fong, P., 2006. Using opportunistic green macroalgae as indicators of nitrogen supply and sources to estuaries. Ecol. Appl. 16, 1405–1420. https://doi.org/10.1890/1051-0761(2006)016[1405:UOGMAI]2.0.CO;2.

<collab>Janicki Environmental, Inc.</collab>, 2012. Development of a Screening Level Tool for Estimating Annual Hydrologic Loadings to Tampa Bay (No. 05-12). Tampa Bay Estuary Program, St. Petersburg, Florida.

<collab>Janicki Environmental, Inc.</collab>, 2017. Estimates of Total Nitrogen, Total Phosphorus, Total Suspended Solids, And Biological Oxygen Demand Loadings to Tampa Bay, Florida: 2012-2016 (No. 04-17). Tampa Bay Estuary Program, St. Petersburg, Florida.

D'Agostino McGowan, L., Bryan, J., 2020. googledrive: An interface to Google Drive. R package version 1.0.1. https://CRAN.R-project.org/package=googledrive.

Dillon, K., Caffrey, J., Carmichael, R.H., Cressman, K., Woodrey, M., 2015. Water Quality in Bangs Lake: Effects of Recurrent Phosphate Spills to a Coastal Estuary. Mississippi Water Resources Research Institute, Mississippi State, US Geological Survey, Starkville, Mississippi.

Dixon, L.K., Leverone, J.R., 1995. Light Requirements of Thalassia testudinum in Tampa Bay, Florida. Number 425. Mote Marine Lab, Sarasota, Florida.

Doering, P.H., Chamberlain, R.H., Haunert, K.M., 2006. Chlorophyll a and its use as an indicator of eutrophication in the Caloosahatchee Estuary,Florida. Florida Sci. 51–72 https://www.jstor.org/stable/24322132.

Domingues, R.B., Barbosa, A.B., Sommer, U., Galvão, H.M., 2011. Ammonium, nitrate and phytoplankton interactions in a freshwater tidal estuarine zone: potential effects of cultural eutrophication. Aquat. Sci. 73, 331–343. https://doi.org/10.1007/s00027-011-0180-0.

El Zrelli, R., Courjault-Radé, P., Rabaoui, L., Castet, S., Michel, S., Bejaoui, N., 2015. Heavy metal contamination and ecological risk assessment in the surface sediments of the coastal area surrounding the industrial complex of Gabes city, gulf of Gabes, SE Tunisia. Mar. Pollut. Bull. 101, 922–929. https://doi.org/10.1016/j.marpolbul.2015.10.047.

Eldridge, P.M., Kaldy, J.E., Burd, A.B., 2004. Stress response model for the tropical seagrass Thalassia testudinum: the interactions of light, temperature, sedimentation, and geochemistry. Estuaries 27, 923–937. https://doi.org/10.1007/BF02803419.

Estrella, S.M., 2013. Effects of Nutrient Enrichment And Toxic Lyngbya Blooms on Benthic Invertebrates And Migratory Shorebird Communities of Roebuck Bay Ramsar Site, Final Report to the NRM Office. University of Western Australia, Perth, Australia, WA.

Flaherty, K.E., Landsberg, J.H., 2011. Effects of a persistent red tide Karenia brevis bloom on community structure and species-specific relative abundance of nekton in a Gulf of Mexico estuary. Estuar. Coasts 34, 417–439. https://doi.org/10.1007/s12237-010-9350-x.

Garrett, M., Wolny, J., Truby, E., Heil, C., Kovach, C., 2011. Harmful algal bloom species and phosphate-processing effluent: field and laboratory studies. Mar. Pollut. Bull. 62, 596–601. https://doi.org/10.1016/j.marpolbul.2010.11.017.

Greening, H., Janicki, A., 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. Environ. Manag. 38, 163–178. https://doi.org/10.1007/s00267-005-0079-4.

Greening, H., Janicki, A., Sherwood, E., Pribble, R., Johansson, J.O.R., 2014. Ecosystem responses to long-term nutrient management in an urban estuary: Tampa Bay, Florida, USA. Estuar. Coast. Shelf Sci. 151, A1–A16. https://doi.org/10.1016/j.ecss.2014.10.003.

Helsel, D.R., 2005. Nondectects And Data Analysis: Statistics for Censored Environmental Data. John Wiley & Sons, New Jersey, USA.

Henderson, C.S., 2004. Piney Point Phosphate Plant: An Environmental Analysis. University of South Florida St, Petersburg, Florida.

Hollander, M., Wolfe, D.A., Chicken, E., 2013. Nonparametric Statistical Methods. John Wiley & Sons.

Holm, S., 1979. A simple sequentially rejective multiple test procedure. Scand. J. Stat. 6, 65–70.

Hudson-Edwards, K.A., Jamieson, H.E., Lottermoser, B.G., 2011. Mine wastes: past, present, future. Elements 7, 375–380. https://doi.org/10.2113/gselements.7.6.375.

Johansson, R., 2016. Seagrass Transect Monitoring in Tampa Bay: A Summary of Findings From 1997 Through 2015 (No. 08-16). Tampa Bay Estuary Program, St. Petersburg, Florida.

Kemp, W.M., Boynton, W.R., 1980. Influence of biological and physical processes on dissolved oxygen dynamics in an estuarine system: implications for the measurement of community metabolism. Estuar. Coast. Mar. Sci. 11, 407–431. https://doi.org/10.1016/S0302-3524(80)80065-X.

Kenworthy, W.J., Fonseca, M.S., 1996. Light requirements of seagrasses Halodule wrightii and Syringodium filiforme derived from the relationship between diffuse light attenuation and maximum depth distribution. Estuaries 19, 740–750.

Lee, L., 2020. NADA: nondetects and data analysis for environmental data. R package version 1.6-1.1. https://CRAN.R-project.org/package=NADA.

Liu, Y., Weisberg, R.H., Lenes, J.M., Zheng, L., Hubbard, K., Walsh, J.J., 2016. Offshore forcing on the "pressure point" of the West Florida Shelf: anomalous upwelling and its influence on harmful algal blooms. J.Geophys.Res.Oceans 121, 5501–5515. https://doi.org/10.1002/2016JC011938.

Liu, Y., Weisberg, R.H., Zheng, L., Sun, Y., Chen, J., 2021. Nowcast/forecast of the Tampa Bay, Piney Point effluent plume: a rapid response. In: Abstract (Os35b-1036) Presented at AGU Fall Meeting, December, 2021. New Orleans, Louisiana.

Lloret, J., Marin, A., Marin-Guirao, L., Velasco, J., 2005. Changes in macrophytes distribution in a hypersaline coastal lagoon associated with the development of intensively irrigated agriculture. OceanCoast.Manag. 48, 828–842. https://doi.org/10.1016/j.ocecoaman.2005.07.002.

Medina, M., Huffaker, R., Jawitz, J.W., Muñoz-Carpena, R., 2020. Seasonal dynamics of terrestrially sourced nitrogen influenced Karenia brevis blooms off Florida's southern Gulf Coast. Harmful Algae 98, 101900. https://doi.org/10.1016/j.hal.2020.101900.

Medina, M., Kaplan, D., Milbrandt, E.C., Tomasko, D., Huffaker, R., Angelini, C., 2022. Nitrogen-enriched discharges from a highly managed watershed intensify red tide (Karenia brevis) blooms in southwest Florida). Sci. Total Environ. 154149 https://doi.org/10.1016/j.scitotenv.2022.154149.

Murphy, R.R., Perry, E., Harcum, J., Keisman, J., 2019. A generalized additive model approach to evaluating water quality: Chesapeake Bay case study. Environ. Model Softw. 118, 1–13. https://doi.org/10.1016/j.envsoft.2019.03.027.

Nelson, N.G., Cuchiara, M.L., Hendren, C.O., Jones, J.L., Marshall, A.M., 2021. Hazardous spills at retired fertilizer manufacturing plants will continue to occur in the absence of scientific innovation and regulatory enforcement. Environ.Sci. Technol. https://doi.org/10.1021/acs.est.1c05311.

Pebesma, E., 2018. Simple features for R: standardized support for spatial vector data. R J. 10, 439–446. https://doi.org/10.32614/RJ-2018-009.

Pérez-López, R., Nieto, J.M., López-Coto, I., Aguado, J.L., Bolívar, J.P., Santisteban, M., 2010. Dynamics of contaminants in phosphogypsum of the fertilizer industry of Huelva (SW Spain): from phosphate rock ore to the environment. Appl. Geochem. 25, 705–715. https://doi.org/10.1016/j.apgeochem.2010.02.003.

Pérez-López, R., Macías, F., Cánovas, C.R., Sarmiento, A.M., Pérez-Moreno, S.M., 2016. Pollutant flows from a phosphogypsum disposal area to an estuarine environment: an insight from geochemical signatures. Sci. Total Environ. 553, 42–51. https://doi.org/10.1016/j.scitotenv.2016.02.070.

R Core Team, 2021. R: A Language And Environment for Statistical Computing, v4.1.2. R Foundation for Statistical Computing, Vienna, Austria.

Rycyk, A.M., Moore, R.B.T., Wells, R.S., McHugh, K.A., McCabe, E.J.B., Mann, D.A., 2020. Passive acoustic listening stations (PALS) show rapid onset of ecological effects of harmful algal blooms in real time. Sci. Rep. 10, 1–12. https://doi.org/10.1038/s41598-020-74647-z.

Sanders, L., Luiz-Silva, W., Machado, W., Sanders, C.J., Marotta, H., Enrich-Prast, A., Bosco-Santos, A., Boden, A., Silva-Filho, E., Santos, I.R., et al., 2013. Rare earth element and radionuclide distribution in surface sediments along an estuarine system affected by fertilizer industry contamination. Water Air Soil Pollut. 224, 1–8. https://doi.org/10.1007/s11270-013-1742-7.

Schrandt, M.N., MacDonald, T.C., Sherwood, E.T., Beck, M.W., 2021. A multimetric nekton index for monitoring, managing and communicating ecosystem health status in an urbanized Gulf of Mexico estuary. Ecol. Indic. 123, 107310 https://doi.org/10.1016/j.ecolind.2020.107310.

Sherwood, E., Greening, H., Johansson, J.O.R., Kaufman, K., Raulerson, G., 2017. Tampa Bay (Florida, USA): documenting seagrass recovery since the 1980's and reviewing the benefits. Southeast. Geogr. 57, 294–319. https://doi.org/10.1353/sgo.2017.0026.

Sherwood, E.T., Raulerson, G., Beck, M., Burke, M., 2020. Tampa Bay Estuary Program: Quality Management Plan (No. 16-20). Tampa Bay Estuary Program, St. Petersburg, Florida.

Skripnikov, A., Wagner, N., Shafer, J., Beck, M., Sherwood, E., Burke, M., 2021. Using localized Twitter activity to assess harmful algal bloom impacts of Karenia brevis in Florida, USA. Harmful Algae 110, 102118. https://doi.org/10.1016/j.hal.2021.102118.

Stafford, N.B., Bell, S.S., 2006. Space competition between seagrass and Caulerpa prolifera (Forsskaal) Lamouroux following simulated disturbances in Lassing Park, FL. J. Exp. Mar. Biol. Ecol. 333, 49–57. https://doi.org/10.1016/j.jembe.2005.11.025.

Steidinger, K.A., 1975. Implications of dinoflagellate life cycles on initiation of Gymnodinium breve red tides. Environ. Lett. 9, 129–139. https://doi.org/10.1080/00139307509435842.

Steidinger, K.A., Ingle, R.M., 1972. Observations on the 1971 summer red tide in Tampa Bay, Florida. Environ. Lett. 3, 271–278. https://doi.org/10.1080/00139307209435473.

Steidinger, K.A., Vargo, G.A., Tester, P.A., Tomas, C.R., 1998. Bloom dynamics and physiology of Gymnodium breve with emphasis on the Gulf of Mexico. In: Anderson, D.M., Cembella, A.D., Hallegraeff, G. (Eds.), Physiological Ecology of Harmful Algal Blooms. Springer-Verlag, Berlin, Heidelberg, pp. 133–151.

Stevens, P.W., Blewett, D.A., Casey, J.P., 2006. Short-term effects of a low dissolved oxygen event on estuarine fish assemblages following the passage of Hurricane Charley. Estuar. Coasts 29, 997–1003. https://doi.org/10.1007/BF02798661.

Stumpf, R.P., Li, Y., Kirkpatrick, B., Litaker, R.W., Hubbard, K.A., Currier, R.D., Harrison, K.K., Tomlinson, M.C., 2022. Quantifying Karenia brevis bloom severity and respiratory irritation impact along the shoreline of Southwest Florida. PLOS ONE 17, 1–31. https://doi.org/10.1371/journal.pone.0260755.

Switzer, T.S., Tyler-Jedlund, A.J., Rogers, K.R., Grier, H., McMichael Jr., R.H., Fox Jr., S., 2011. Response of estuarine nekton to the regulated discharge of treated phosphate-production process water. In: Florida Fish; Wildlife Conservation Commission, Fish. Wildlife Research Institute, St. Petersburg, Florida.

Tayibi, H., Choura, M., López, F.A., Alguacil, F.J., López-Delgado, A., 2009. Environmental impact and management of phosphogypsum. J. Environ. Manag. 90, 2377–2386. https://doi.org/10.1016/j.jenvman.2009.03.007.

Tomasko, D., Alderson, M., Burnes, R., Hecker, J., Iadevaia, N., Leverone, J., Raulerson, G., Sherwood, E., 2020. The effects of Hurricane Irma on seagrass meadows in previously eutrophic estuaries in Southwest Florida (USA). Mar. Pollut. Bull. 156, 111247 https://doi.org/10.1016/j.marpolbul.2020.111247.

Valiela, I., McClelland, J., Hauxwell, J., Behr, P.J., Hersh, D., Foreman, K., 1997. Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. Limnol. Oceanogr. 42, 1105–1118. https://doi.org/10.4319/lo.1997.42.5_part_2.1105.

Villac, M.C., Hoeglund, A., Tilney, C., Garrett, M., Lopez, C., Hubbard, K.A., Steidinger, K.A., 2020. Ecophysiology and bloom dynamics of Karenia with emphasis on Karenia brevis in Florida waters. In: Rao, D.V.S. (Ed.), Dinoflagellates: Classification, Evolution, Physiology And Ecological Significance. Nova Science Publishers, New York, pp. 261–301.

Wang, P., Martin, J., Morrison, G., 1999. Water quality and eutrophication in Tampa Bay, Florida. Estuar. Coast. Shelf Sci. 49, 1–20. https://doi.org/10.1006/ecss.1999.0490.

Weisberg, R.H., Zheng, L., Liu, Y., Lembke, C., Lenes, J.M., Walsh, J.J., 2014. Why no red tide was observed on the West Florida Continental Shelf in 2010. Harmful Algae 38, 119–126. https://doi.org/10.1016/j.hal.2014.04.010.

Weisberg, R.H., Liu, Y., Lembke, C., Hu, C., Hubbard, K., Garrett, M., 2019. The coastal ocean circulation influence on the 2018 West Florida Shelf K. brevis red tide bloom. J.Geophys.Res.Oceans 124, 2501–2512. https://doi.org/10.1029/2018JC014887.

Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L.D., François, R., Grolemund, G., Hayes, A., Henry, L., Hester, J., Kuhn, M., Pedersen, T.L., Miller, E., Bache, S.M., Müller, K., Ooms, J., Robinson, D., Seidel, D.P., Spinu, V., Takahashi, K., Vaughan, D., Wilke, C., Woo, K., Yutani, H., 2019. Welcome to the tidyverse. J.Open Source Softw. 4, 1686. https://doi.org/10.21105/joss.01686.

Wood, S.N., 2017. Generalized Additive Models: An Introduction With r, 2nd ed. Chapman; Hall, CRC Press, London, United Kingdom.