Journal of Coastal Research	39	4	722–739	Charlotte, North Carolina	July 2023

The Impacts of a Restoration Dredging Project and Storm Events on Water Quality in a Northeast Florida Barrier Island Estuary

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ABSTRACT

Brown, M.T.; Dix, N.; Osborne, T.; Castle, A.; Woodward, W.; Hartnett, A.; Dunnigan, S.; Tanner, S., and McGinley, E., 2023. The impacts of a restoration dredging project and storm events on water quality in a northeast Florida barrier island estuary. *Journal of Coastal Research,* 39(4), 722–739. Charlotte (North Carolina), ISSN 0749-0208.

Previous studies in estuarine systems have demonstrated that both dredging projects and storm events can have profound impacts on water quality, particularly increases in nutrients, sediment loading, and shifts in phytoplankton community biomass. This study provided the opportunity to assess the potential relative impacts of a 9-month-long barrier island restoration dredging operation, in comparison to storm events that shortly followed, on water quality in a well-flushed, subtropical NE Florida estuary. Twice-a-month monitoring of turbidity, total suspended solids, chlorophyll-*a*, and major nutrients (ammonium, nitrate+nitrite, and orthophosphate) was carried out at six sites near the dredging operation area from November 2016 through October 2018. This timeframe sampled the dredging project from January to September 2017 associated with the reopening of the Summer Haven River on the barrier island side of the Matanzas River estuary. In addition, two major storm events, Hurricane Irma in September 2017 and a 5-day nor'easter in October 2017, took place during the water quality monitoring. Although the dredging project did result in notable impacts on total suspended solids and ammonium concentrations, the results highlight how natural storm events can have more significant and lasting impacts on water quality in a well-flushed estuary, particularly with regard to decreases in salinity, input and remineralization of dissolved organic carbon, and increases in dissolved nutrients.

ADDITIONAL INDEX WORDS: *Nutrients, ammonium, nitrate, chlorophyll-*a*, phytoplankton, turbidity, dissolved organic carbon, hurricane, Matanzas River.*

INTRODUCTION

In addition to being incredibly productive and diverse, estuaries offer natural protection from coastal storms and sea-level rise, ecosystem services such as improved water quality, and storage of carbon in coastal vegetated systems such as salt marshes, mangroves, and seagrasses, also called blue carbon (Barbier *et al*., 2008, 2011; Bromberg-Gedan, Silliman, and Bertness, 2009; Duarte *et al*., 2013; Macreadie, Hughes, and Kimbro, 2013; Mcleod *et al*., 2011). However, increasing coastal development is putting anthropogenic pressure on estuaries and will likely inhibit the ecosystem services mentioned. According to the U.S. Census Bureau, approximately 95 million, or 29%, of the total 325.7 million people in the United States lived in coastal counties in 2017. Between 1970 and 2010, the coastal population of the United States increased by 40% (NOAA, 2013). An estimated 1355 building permits were issued daily in coastal shoreline counties from 2000 to 2010 (NOAA, 2013). Undoubtedly, this coastal population increase and development will have profound impacts on water quality.

This study was associated with a 2017 restoration dredging project and two storm events that shortly followed, Hurricane Irma and a 5-day nor'easter, in the Matanzas River estuary (MRE) in NE Florida. The MRE is a lagoonal system with abundant salt marsh, mangrove, and oyster habitats (Dix *et al*., 2017, 2019; Gray *et al*., 2021). In addition, the MRE is the southern portion of the Guana Tolomato Matanzas National Estuarine Research Reserve (GTMNERR) and is characterized by strong tidal flushing, relatively short water residence times, relatively low amounts of freshwater input, and low phytoplankton biomass (Dix, Phlips, and Suscy, 2013; Frazel, 2009; Phlips *et al*., 2004; Sheng *et al*., 2008). Pellicer Creek is the significant source of freshwater to the MRE. Matanzas Inlet is the major connection between the Atlantic Ocean and the southern portion of the MRE. South of Matanzas Inlet, in St. Johns County, Florida, the MRE historically forked into the Summer Haven River east of the main estuary system and east of State Road A1A (Figure 1A,B). The restoration dredging project reopened a portion of this barrier island estuary that

DOI: 10.2112/JCOASTRES-D-22-00070.1 received 27 July 2022; accepted in revision 20 December 2022; corrected proofs received 6 March 2023.
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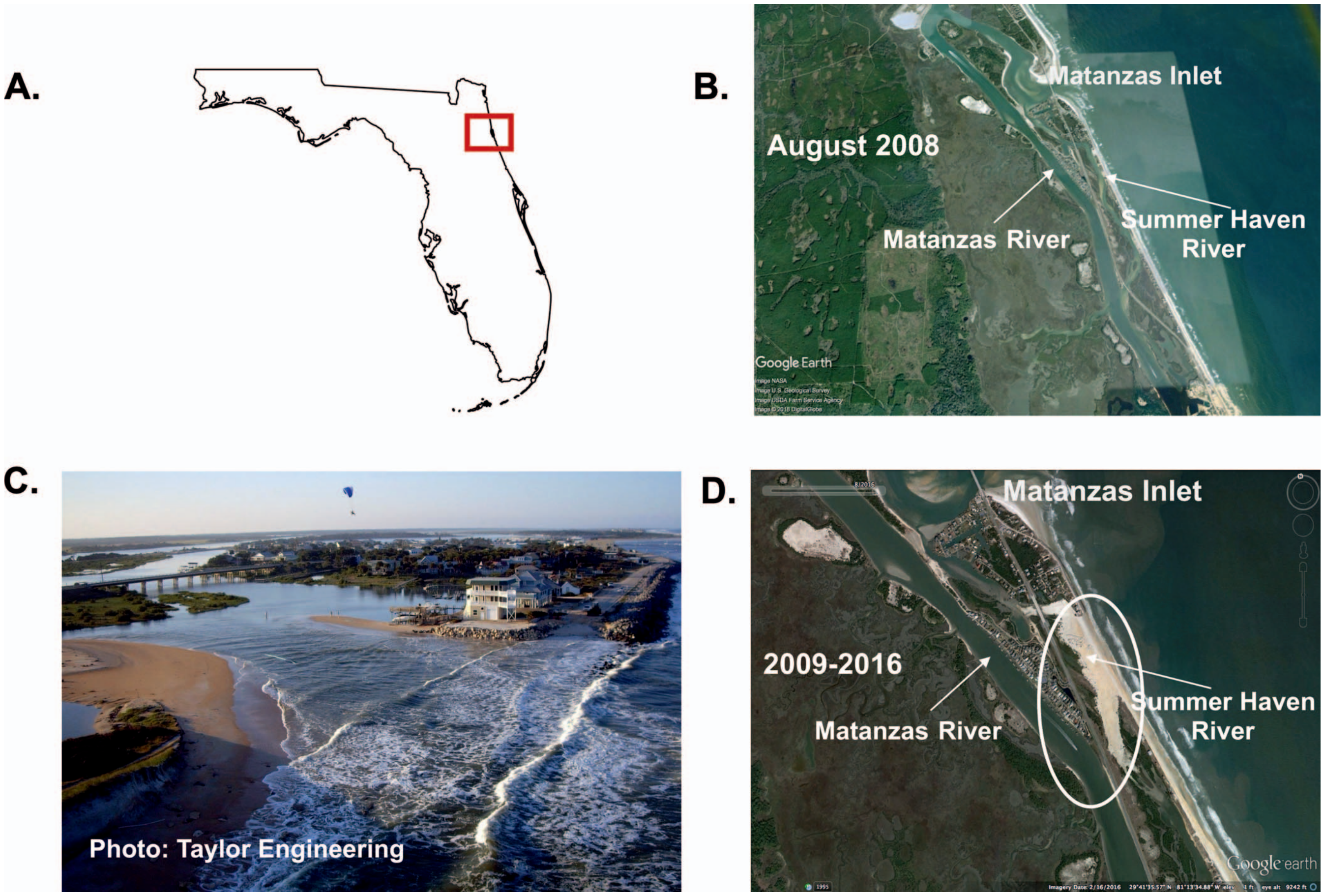

Figure 1. (A) Geographical location of the study site in NE Florida as indicated by a red square (top left). (B) Prebreach conditions of the Summer Haven River south of Matanzas Inlet in August 2008 (top right). (C) Aerial photograph of the Summer Haven River breach in November 2008, looking NW from the Atlantic Ocean into the Summer Haven River (bottom left; courtesy of Taylor Engineering). (D) Postbreach and infill of the Summer Haven River with sand post-2008 breach (bottom right). The river was almost filled in with sand south of Matanzas Inlet. The white circle in the bottom-right picture indicates the area of sediment excavation.

naturally filled in with sand after a breach and washover event of the coastal dune line years prior. Beginning in autumn 2008, a series of tropical storms created and maintained a breach through a narrow stretch of primary dune south of Matanzas Inlet and into the Summer Haven River (Figure 1C). Over the next few months, with storm and tide mixing, sand gradually filled in the Summer Haven River until the breach closed naturally in 2009 (Figure 1D). The infilling of the Summer Haven River buried more than half its 3.2-km length, expanding the coastal beach and dunes westward and covering acres of preexisting salt marsh and coastal strand that once defined the river's western shoreline (Figure 1D). The restoration dredging project, beginning in January 2017 and ending in September 2017, removed 210,250 m^3 (275,000 cubic yards) of sand and reopened the Summer Haven River to its prebreach flow.

Dredging can have both positive and negative environmental impacts on estuarine systems. Benefits can include flood control, removal of excessive nutrients stored in sediments, and improvement in fish habitat (Boerema and Meire, 2017; Maglio *et al*., 2016). In a review of anthropogenic environmental impacts on estuaries, Kennish (2002) noted that the loss and alteration of estuarine habitat can result in compromised water quality, including eutrophication and an accumulation of chemical contaminants. Specifically, dredging and dredged material disposal related to the physical alteration of estuaries can pose significant environmental problems (van Maren *et al*., 2015; Wilbur and Clarke, 2001). Van den Berg *et al*. (2001) and Eggleton and Thomas (2004) outlined how dredging disturbances can result in both particulate matter resuspension and pore water release into the overlying water column, potentially increasing metal and other chemical contaminant concentrations. It is important to understand the impacts that these projects might have on water quality, specifically potential eutrophication and changes in salinity, turbidity, and suspended sediment loads.

Storm events can also have major impacts on water quality in estuarine systems. Previous work on the impacts of storm events on water quality in SE U.S. estuaries has shown significant changes in salinity, nutrient concentrations, dissolved organic matter (DOM), and fecal coliform bacteria (Coulliette and Noble, 2008; Dix, Phlips, and Gleeson, 2008; Paerl *et al.*, 2006; Paerl *et al.*, 2018; Peierls, Christian, and Paerl, 2003; Reyna, Hardison, and Liu, 2017). A recent study by Asmala *et al.* (2020) highlighted significantly increased dissolved organic carbon (DOC) concentrations persisting for nearly 3 months and subsequently higher partial pressure of carbon dioxide (pCO_2) after hurricanes in the Neuse River estuary in North Carolina. In addition, fall and winter nor'easters, although not named and not receiving the storm coverage and media attention that hurricanes do, can have significant impacts on water quality due to prolonged wind and precipitation events. These storm impacts are of significant interest when taken in the broader context of projected climate change impacts in the study region. The Fourth National Climate Assessment highlights an increase in both frequency and intensity of extreme precipitation events in the SE United States (Carter *et al.*, 2018). Under a high carbon emissions scenario, climate models predict a doubling of heavy precipitation events and a more than 20% increase in rainfall totals falling during heavy precipitation events (Allan and Soden, 2008; Easterling *et al.*, 2017). Thus, it is important to continue to monitor for and understand the water quality changes associated with these storm events.

The initial objective of this study was to investigate potential water quality changes associated with the restoration dredging project. Although the GTMNERR conducts monthly water quality sampling at the Fort Matanzas site approximately 2.5 miles north of Matanzas Inlet, this study established several sites in and around the dredging area for the monitoring of potential changes in water quality, as well as more frequent sampling during the dredging period (Figure 2). However, because of the passage of two major storm events (Hurricane Irma and a following nor'easter in September and October 2017) after the dredging project and within the study period, a unique opportunity was presented to compare the impacts on water quality of the restoration sediment dredging and the storm events that followed.

METHODS

Water quality sampling was a collaborative effort between Dr. Matthew Brown, GTMNERR staff, and undergraduate research assistants from Flagler College in St. Augustine, Florida. With the exception of the organic carbon analyses detailed later, all laboratory analyses were carried out by Brown and Flagler College undergraduate research assistants at Flagler College. Organic carbon analyses were carried out at the Whitney Laboratory for Marine Bioscience at the University of Florida under the guidance of Dr. Todd Osborne.

Sampling, Filtration, and Turbidity and Total Suspended Solids Analysis

Water quality sampling was conducted monthly in November 2016 and December 2016 and then continued bimonthly (every 2 weeks) from January 2017 through October 2018. In total, 45 sampling dates were completed over the 2-year study period. Sampling took place at six sites near the location of sediment excavation (Figure 2) and always occurred on the ebb tide beginning approximately 1 hour after high tide. This timeframe of sampling is the same as that of the GTMNERR systemwide monitoring program (Buskey *et al.*, 2015) and allows a more robust data comparison to historical data collected by the GTMNERR. The sediment excavation work associated with the Summer Haven River restoration project began in January 2017 and continued through September 2017. Ideally, a greater number of pre-excavation water quality samplings would have been carried out to establish baseline data before study commencement. However, because of a short timeline between the notification of grant funding and the study start, only two predredge samplings were carried out.

During each sampling, hydrographic parameters (temperature, salinity, and dissolved oxygen) were collected with a calibrated YSI Quattro Pro multimeter. Duplicate water column–integrated samples were collected using either a clean 3-m polyvinylchloride (PVC) pole sampler or, if too shallow, a clean 1-m PVC pole sampler. The duplicate water samples were collected and emptied into separate, acid-washed polyethylene buckets. From the two buckets, duplicate discrete 500- and 125-mL water samples were collected in acid-cleaned bottles for the analysis of turbidity, total suspended solids (TSS), chlorophyll-*a* (chl-*a*), and the major nutrients ammonium (NH_4), nitrate+nitrite (NO_x), and orthophosphate (PO_4). Samples were immediately placed on ice in a dark cooler for transport to the laboratory. The preceding sampling protocols are the same protocols used by the National Estuarine Research Reserve System (NERRS) Wide Monitoring Program for water quality (Buskey *et al.*, 2015).

For turbidity analysis, well-mixed, unfiltered aliquots from the 125-mL bottles were analyzed using a turbidity module (Turner Designs Application note S-0072) on a Turner Designs Trilogy laboratory fluorometer. For TSS, water samples were filtered through preweighed, 47-mm glass fiber filters (Fisher Scientific). These filters were then rinsed with a small volume of deionized water to remove salts, heated to 105°C in a drying oven for 3 days, and then dried for 1 week in a desiccator, after which the final filter weights were taken.

Within 6 to 8 hours of sample collection, discrete water samples were filtered through 47-mm, 0.45-μm, acid-cleaned polyethersulfone (PES) membrane filters (Sterlitech) for chl-*a* and nutrient analysis. For nutrients, the filtrate was poured off into acid-cleaned 125-mL polyethylene bottles and frozen at –20°C before analysis. It has been shown that freezing is an acceptable means of sample storage for nutrient analysis (Dore *et al.*, 1996). PES membrane filters were stored at –80°C before chl-*a* extraction. Size-fractionated chl-*a* filtrations were carried out monthly on water samples by filtering sequentially through the following PES filter (Sterlitech) pore sizes: 20, 5, and 0.45 μm. The size fractions measured were 0.45 to 5 μm, 5 to 20 μm, and greater than 20 μm.

Chl-*a*, Nutrient, and Organic Carbon Analysis

The chl-*a* analysis was carried out within 1 month of sample collection according to the method of Welschmeyer (1994). A Turner Designs laboratory fluorometer was used to measure fluorescence after 90% acetone extraction and sonication using a QSonica Q125 sonicator to ensure cell lysis.

NH_4 analyses were completed according to the fluorometric method of Holmes *et al.* (1999). Sample and working reagents

Figure 2. Map of the study region. The six sites regularly sampled for this study are indicated in yellow. The GTMNERR Fort Matanzas water quality sampling site (GTMFMWQ), the Pellicer Creek water quality sampling site (GTMPCWQ), and the Pellicer Creek meteorological station (GTMPCMET) are indicated in white. The location of the Summer Haven River sediment excavation is shown within the light blue dashed rectangle.

were combined in a 4:1 volume ratio and incubated for 3 hours in the dark, and fluorescence was measured using a Turner Designs Trilogy fluorometer with an NH_4 optical module (PN 7200-067). NH_4 standards were made by making small volume standard additions of a stock ammonium chloride (Sigma-Aldrich) solution to low-NH_4 concentration filtered seawater.

NO_x analyses were carried out using a modification of the Schnetger and Lehners (2014) method. First, 3.6 mL of a filtered seawater sample was mixed with 3.0 mL of a mixed reagent composed of vanadium chloride reduction solution, N-1 napthylenediamine dihydrochloride solution, and sulfanilamide solution (Fisher Scientific), and then the mixture was heated to 50°C for 1 hour. The absorbance of the mixture was measured at 540 nm using a Turner Designs nitrate absorbance module (Turner Designs PN 7200-074). Standards were made in a manner similar to the NH_4 standards outlined earlier.

PO_4 concentrations were determined using an adaptation of Strickland and Parsons (1972). A phosphate mixed reagent solution composed of ammonium molybdate, sulfuric acid, ascorbic acid, and potassium antimonyl-tartrate was prepared and added to filtered samples in a 10:1 sample/reagent ratio. After 15 minutes, the absorbance of the solution was measured at 885 nm using a phosphate absorbance module (Turner PN 7200-070).

DOC and particulate organic carbon (POC) determinations were made on water samples collected during the first year of the study from December 2016 through December 2017, according to the methodology outlined in Osborne, Inglett, and Reddy (2007). DOC was measured on a Shimadzu TOC-5050 analyzer. Analysis of total carbon was conducted on a Carlo-Erba CN analyzer.

Data Access and Statistical Analysis

The St. Johns River Water Management District (SJRWMD) maintains a surface water sampling site (Station MRT) close to Site 5 from this study (Figure 2), and monthly data since 1986 are publicly available (SJRWMD, 2023). SJRWMD Station MRT data from 2010 to 2016 were used as an index of water quality in the region before the dredging project. Rainfall and water quality data from the GTMNERR Pellicer Creek meteorological and water quality stations (Figure 2) were downloaded from the National Oceanic and Atmospheric Administration (NOAA) NERRS Centralized Data Management Office (2023).

Statistical analyses using JASP open-source statistical software (JASP Team, 2022; Love *et al.*, 2019) were used to test for differences in the means of water quality parameters collected during the 2017 dredge period (January to September 2017; 18 sampling events) compared with a 2018 postdredge period over the same months (January to September 2018; 18 sampling events). Water quality parameters were initially evaluated for normality using the Shapiro-Wilk test. A similar number of normal distributions among water quality parameters was found in the data compared with nonnormal distributions. However, because of the relatively small number of samples ($n = 18$) being compared in each period, the nonparametric Wilcoxon sign-rank test was used. If the Wilcoxon sign-rank test revealed a significant ($p < 0.05$) difference between dredge and postdredge periods for a particular parameter and the parameter revealed a normal distribution over either the dredge or the postdredge period, a paired t test was also run for comparison.

To examine whether the sites close to the restoration project showed a difference from other sites farther removed during the dredge period in terms of collective water quality, an enhanced k-means cluster analysis was performed on the water quality data from the dredge project period and postdredge project period using the statistical program R (R Core Team, 2020). Water quality data (salinity, turbidity, chl-a, TSS, dissolved inorganic nitrogen [DIN], and PO_4) were scaled, ensuring that data with a larger range of values did not exert more influence on the results than values with smaller ranges. The scaling was performed in the R package *factoextra*. Enhanced k-means clustering was performed on the dataset to determine the optimal number of clusters based upon the gap statistic using the function *eclust* in R. The gap statistic was calculated by comparing the within-cluster sum of squares to k, the number of clusters. The error/k pairing is then compared with a null distribution to find the k with the largest gap between the expected error in a null distribution and the dataset in question (Tibshirani, Walther, and Hastie, 2001).

RESULTS

The results of the 2-year study are divided into three sections: basic hydrography (temperature, salinity, and dissolved oxygen) and storm events, turbidity and TSS, and phytoplankton biomass as expressed by chl-a and associated major nutrient concentrations. Project results are also divided into a during-dredge period and a postdredge period, and these results are compared with historical data collected by the SJRWMD.

Hydrography

The trends in water temperature were generally the same at the six sampling sites (Figure 3). The minimum water temperature, ~9.9°C, observed in late January 2018 was nearly 7°C colder than the minimum temperature observed in January 2017. The maximum temperature observed over the 2-year study was 30.8°C at Site 6 in early July 2017. At that same time, the temperature at Site 1, which is more influenced by higher-salinity water from the open Atlantic Ocean, was nearly 7°C colder at 24.1°C. Dissolved oxygen concentrations ranged from 3.6 mg O_2 L^{-1} in late August 2017 to 10.5 mg O_2 L^{-1} in early January 2018 and showed a negative correlation with water temperature ($R^2 = 0.68$).

Salinity showed a significant degree of temporal and spatial variation over the 2-year study period (Figure 3). Ranging from a maximum of 37.95 psu at Site 2 to a minimum of 11.36 psu at Site 6, salinity showed some degree of variation with rainfall depending on the station sampled. For example, a significant increase in rainfall totals associated with Hurricane Irma in mid-September 2017 and a following 5-day nor'easter in early October 2017 was correlated with a significant decrease in salinity at Sites 4, 5, and 6, the three southernmost sites and the sites most physically removed from Matanzas Inlet and the mixing of higher-salinity water from the Atlantic Ocean. In contrast, Sites 1, 2, and 3, even with the greatly increased

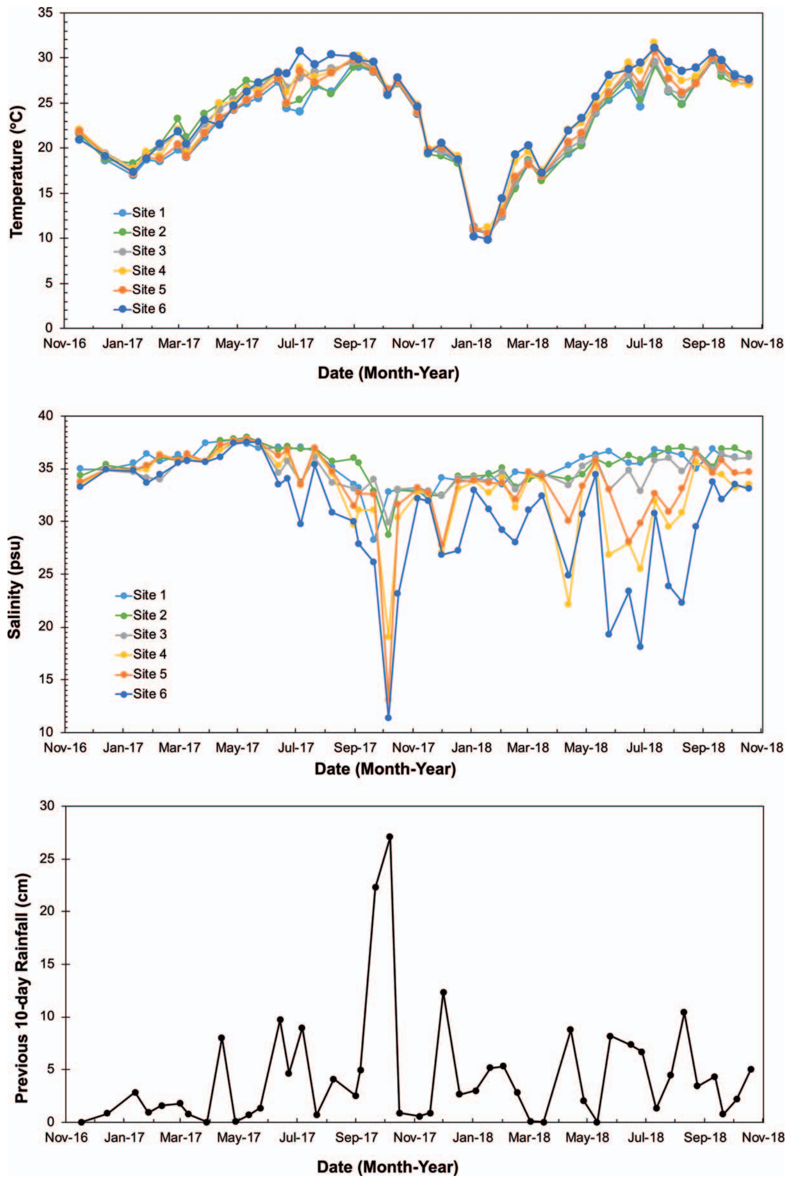

Figure 3. Temperature (top), salinity (middle), and rainfall totals (bottom) at the project sampling sites. The rainfall totals are cumulative and for the 10 days before a given sampling date. Rainfall totals were calculated using data from the GTMNERR Pellicer Creek meteorological station shown in Figure 2.

rainfall totals mentioned previously, showed only relatively small decreases in salinity to ~30 psu. From the beginning of the study in November 2016 through July 2017, rainfall totals were relatively low and salinity at all sites was largely greater than 33 psu.

Just as the dredging project was completed in September 2017, NE Florida experienced the impacts of two major storms: Hurricane Irma from 10 to 11 September 2017 and a 5-day nor'easter in late September to early October 2017. The total precipitation associated with Hurricane Irma was approximately 22.3 cm (8.78 in) in less than 30 hours, along with wind gusts in excess of 22 m s^{-1} (~50 mph). Less than 3 weeks later, a multiple-day nor'easter (30 September to 5 October 2017) with onshore winds blowing ~9 to 14 m s^{-1} (~20–30 mph) dumped 26.7 cm (10.5 in) of rain on the region. Along with the excessive precipitation, large decreases in salinity were observed. Salinities at the GTMNERR Pellicer Creek water quality sampling site decreased to 0 psu from 12–17 September and then again from 1–16 October 2017 (Figure 4). The salinity did not rise above 20 psu in Pellicer Creek until early November 2017. These prolonged periods of low-salinity water exiting Pellicer Creek are indicative of the significant freshwater input that occurred with these two storm events. Across both periods when salinity dropped to 0 psu mentioned previously, dissolved oxygen also decreased significantly (Figure 4). From 12–17 September, dissolved oxygen values fell from ~6 to 2 mg L^{-1}.

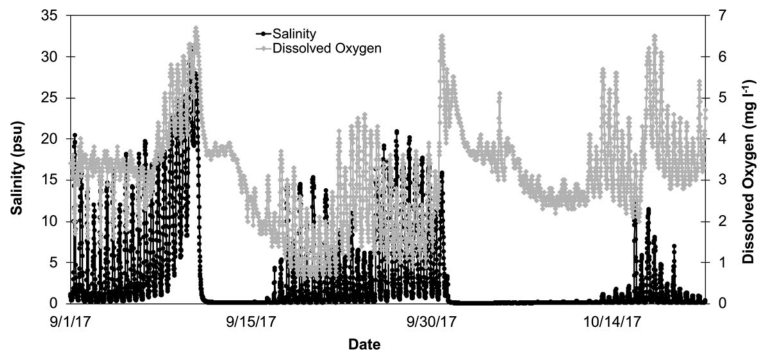

Figure 4. Salinity (left y-axis) and dissolved oxygen (right y-axis) at the GTMNERR Pellicer Creek water quality station from September to October 2017. This site was not a regularly sampled site over the duration of the study but is used to show the storm impacts of Hurricane Irma and a following nor'easter on water quality in the region.

Similarly, dissolved oxygen values decreased from ~6 mg L^{-1} on September 30 to ~2.5 mg L^{-1} on October 9.

Turbidity and TSS

Over the 2-year study period, turbidity values at the six sites ranged from a minimum of 2.7 NTU at Site 1 in mid-October 2017 to a maximum of 52.4 NTU at Site 2 in early August 2017 (Figure 5). Notable increases in turbidity at the six sites sampled relative to the SJRWMD predredge data were observed in early February 2017, early August 2017, and early January 2018, with the first two of these sampling dates occurring during the sediment excavation project. Further inspection of the turbidity data revealed that Site 3, the Summer Haven Bridge site, had elevated turbidity values from mid-April through mid-September 2017 relative to the long-term baseline SJRWMD turbidity data.

TSS values over the study period were quite variable, ranging from 0.8 mg L^{-1} at Site 5 in late August 2018 to 114 mg L^{-1} at Site 4 in late June 2017 (Figure 5). Long-term baseline values from 2010 to 2016 determined by the SJRWMD at Site 5 show average monthly values ranging from 21.5 to 38.5 mg L^{-1}, with a significant degree of yearly variability. It appears that average TSS values determined at the six sites during the excavation project showed no significant difference relative to the long-term average monthly values and their variation collected by the SJRWMD from 2010 to 2016. However, short-term spikes in TSS were observed during the dredge period at multiple sites that were not observed during the postdredge period (Figure 5).

Major Nutrients, Chl-*a*, and DOC

DIN concentrations, the sum of dissolved NO_x and dissolved NH_4 concentrations ranged from 0.1 to 6.5 μm over the course of the 2-year study period (Figure 6). From November 2017 through late May 2017, when rainfall totals were relatively low (Figure 3), DIN concentrations were largely less than 1 μm at all sites. DIN concentrations showed significant increases both at Site 6 in early August 2017 and at Sites 4 and 5 in late August 2017. These increases in DIN came before the dramatically increased rainfall totals in late September 2017 and early October 2017 associated with Hurricane Irma and the following nor'easter, respectively. DIN increased significantly at all sites from October 2017 through January 2018, ranging largely from 2 to 5 μm. Concentrations decreased to values largely less than 1 μm from February 2018 through November 2018 with the exception of elevated values in early August 2018.

Figure 7 shows the relative contribution of dissolved NO_x and dissolved NH_4 to the DIN pool for Sites 1, 2, and 6. These particular sites were selected for the following reasons. Site 1 represents a higher-salinity, more oceanic endmember, whereas Site 6 represents a lower-salinity endmember site physically removed from Matanzas Inlet and more influenced by freshwater runoff. In addition, Site 6 is nearly 7 km from the sediment excavation site within the Summer Haven River, possibly too far removed to the south to see an impact on water quality from the excavation project. Site 2 was selected because it is just north of the location of the sediment excavation project and would likely experience the most significant impact on nutrient concentrations because of sediment excavation among the sites. From November 2016 through June 2017 at Sites 1, 2, and 6, DIN concentrations at all three sites were relatively low (Figure 7) and, on average, NH_4 accounted for 69 ± 24, 70 ± 18, and 66 ± 18% of the DIN pool, respectively. From early August 2017 through mid-October 2017, DIN increased substantially at Site 6, and over this period, the NH_4 contribution to the DIN pool increased to 89 ± 7% at Site 6, whereas values remained largely unchanged at Site 1 (60 ± 24%) and Site 2 (74 ± 26%). A secondary peak in DIN occurred from mid-November 2017 through mid-January 2018. Over this period, the percent contribution of NH_4 to the DIN pool decreased remarkably at Sites 1, 2, and 6 to 19 ± 17, 23 ± 12, and 30 ± 24%, respectively.

PO_4 concentrations (data not shown) were largely less than 0.5 μm from November 2016 through July 2017. PO_4 concentrations began to increase at Site 6 in early August 2017 and at Sites 1 to 5 by late August 2017. PO_4 concentrations remained elevated between 1 and 2 μm through mid-October 2017. By mid-November 2017, concentrations largely decreased to less than 0.5 μm for the remainder of the study period with the exception of elevated values periodically at Site 6 from late May 2018 through early August 2018.

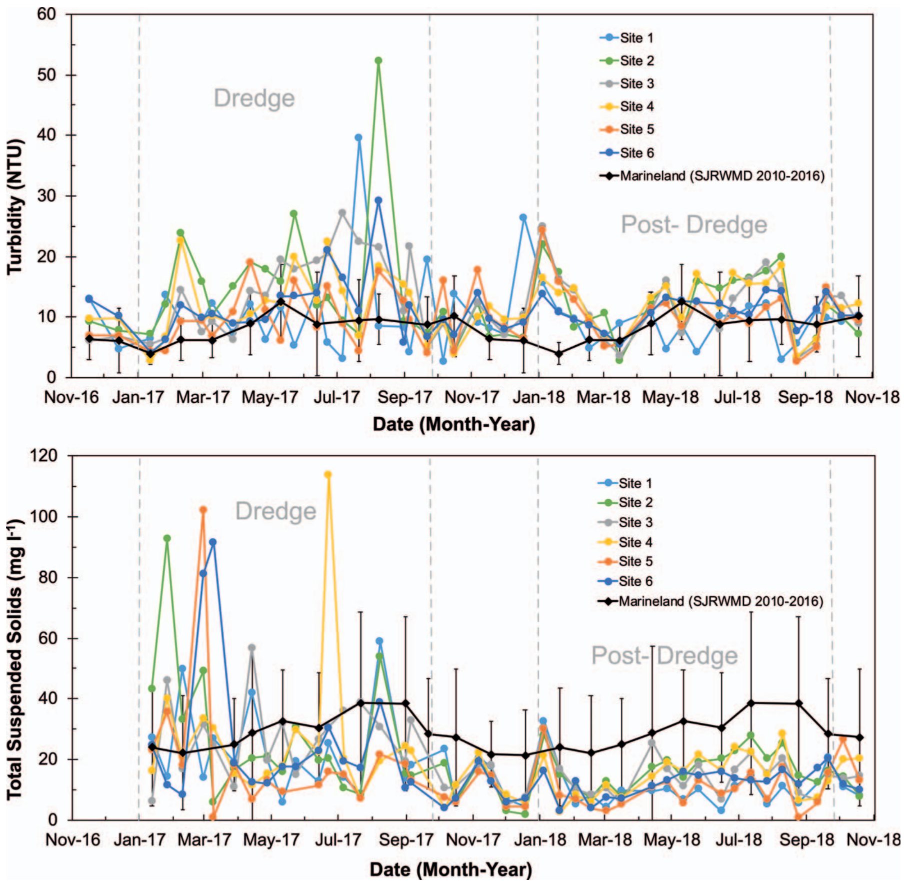

Figure 5. Turbidity (top) and total suspended solids (bottom) concentrations at the six study sites. The January–September 2017 dredge project period and the January–September 2018 postdredge period for comparison are denoted by vertical gray lines. The data in black are the monthly averages collected from 2010 to 2016 by the St. Johns River Water Management District site in Marineland, which is close to Site 5 from this study.

$DIN:PO_4$ ratios (mole:mole) at the six sites from November 2016 through late August 2017 are generally 5 or less, indicative of a strongly nitrogen-limited system relative to the Redfield stoichiometry. The $DIN:PO_4$ ratios do increase from September 2017 through late January 2018 to values from 5 to 10, still indicative of a nitrogen-limited system.

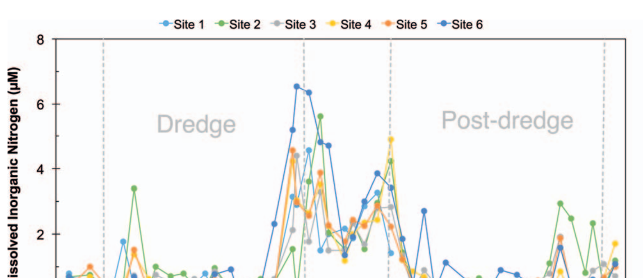

Figure 6. Dissolved inorganic nitrogen (DIN) concentrations expressed in micromoles of nitrogen per liter for the six sites sampled during the study period. The January–September 2017 dredge project period and the January–September 2018 postdredge period for comparison are denoted by vertical gray lines. Comparison of this study with 2010–16 SJRWMD DIN data was not possible because of differences in sample treatment (non-acidification *vs*. acidification).

The chl-*a* concentrations varied by roughly an order of magnitude, with a minimum of 2.3 $\mu g\ L^{-1}$ at Site 1 in early July 2017 to a maximum value of 24.8 $\mu g\ L^{-1}$ at Sites 1 and 2 in late July 2017 and early August 2017, respectively (Figure 8). A general seasonal trend is shown at all sites with lower concentrations in the winter months ($\sim$4–6 $\mu g\ L^{-1}$) increasing to a summer maximum of $\sim$10 to 20 $\mu g\ L^{-1}$ in July and August. An unusual wintertime increase in chl-*a* was observed at all sites in early February 2017. The chl-*a* concentrations at the six sites associated with this study appear to be generally similar to the 2010–16 SJRWMD data also shown in Figure 8. This is with the exception of increases in chl-*a* at the six study sites in February 2017 and at select sites from June to August 2017. There were no significant statistical correlations between chl-*a* concentrations and NH_4, NO_x, or DIN (data not shown).

Although both the GTMNERR and the SJRWMD regularly collect samples for chl-*a* analysis in the region, the size-fractionated chl-*a* measurements from December 2016 to

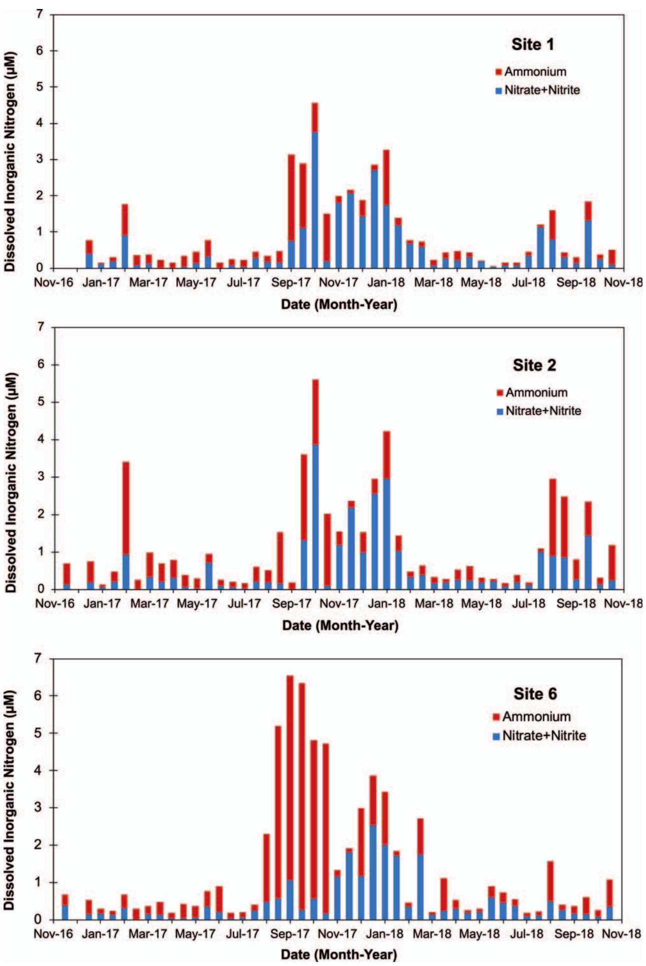

Figure 7. Dissolved inorganic nitrogen concentrations expressed as the sum contribution of nitrate+nitrite (NO_x) and ammonium (NH_4) concentrations for Site 1 (top), Site 2 (middle), and Site 6 (bottom). The relative contributions of NO_x and NH_4 are plotted in blue and red, respectively.

December 2017 are the first reported in the MRE system. At all sites and during all months, the large majority (>85%) of chl-*a* was found in the size fraction of more than 20 µm, indicative of relatively large phytoplankton (*e.g.*, diatoms and larger dinoflagellates) contributing to most of the phytoplankton biomass. The only significant change observed from this trend was in mid-February 2017 when, during the unusual wintertime increase in chl-*a*, the size fraction of greater than 20 µm decreased to 72% of the total chl-*a* and the 0.45- to 5-µm size fraction and 5- to 20-µm size fraction combined to account for 28% of the observed total chl-*a*.

DOC and POC concentrations showed the same general trends at all sites (Figure 9). From December 2016 through April 2017, when rainfall totals were relatively low and salinities were relatively high, total organic carbon concentrations were largely 10 mg C L^{-1} or less at all sites, with the DOC contribution as 75 to 100% of the total. Between May and August 2017, total organic carbon concentrations increased to ~35 to 45 mg C L^{-1}, the DOC contribution decreased to ~60 to 70%, and an increase in POC concentrations was noted at all sites. This was coincident with the summer 2017 increase in chl-*a* concentrations (Figure 8). However, the most remarkable feature is the dramatic increase in DOC concentrations in September and October 2017. Coincident with a significant increase in rainfall and a decrease in salinity associated with the storm events, DOC concentrations increased by nearly an order of magnitude. This increase in DOC was not accompanied by a significant increase in POC concentrations. Lastly, there was no major difference in the POC and DOC concentrations at Sites 1 and 2, closer to the sediment excavation site, compared with Site 6, which was well removed to the south and likely not as affected by the dredging operation.

Statistical Comparison of the Dredging Period and the Postdredge Period

The Wilcoxon sign-rank test results indicated that TSS values at all sites were significantly greater during dredging—in some cases nearly twice as great, such as TSS during the postdredge period—with the exception of Site 2 (Table 1). No significant differences in chl-*a* values between the dredging

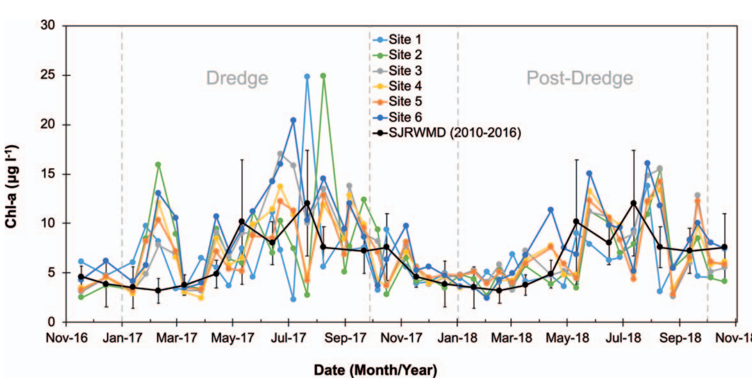

Figure 8. Chlorophyll-a (chl-*a*) concentrations for the six sites sampled during the study period. The January–September 2017 dredge project period and the January–September 2018 postdredge period for comparison are denoted by vertical gray lines. The data in black are the monthly average chl-*a* values from 2010 to 2016 as determined at the St. Johns River Water Management District site in Marineland, which is close to Site 5 from this study.

Table 1. *Water quality parameter averages and standard deviations and results of Wilcoxon sign-rank tests in comparing the January to September 2017 dredging project period vs the January to September 2018 nondredge period for each of the six sampling sites. Each period had 18 samplings, and the Wilcoxon p values in columns 4 and 8 marked with an asterisk indicate significant differences at the 95% confidence level.*

	Site 1			Site 2			Site 3		
	Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value	Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value	Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value
Salinity (psu)	36.16 ± 1.33	35.32 ± 1.12	0.025*	36.25 ± 1.24	35.34 ± 1.17	0.028*	35.16 ± 1.56	34.68 ± 1.18	0.347
TSS (mg L^{-1})	22.3 ± 14.5	11.0 ± 7.0	0.002*	27.8 ± 21.8	17.3 ± 6.1	0.075	28.0 ± 12.8	14.7 ± 6.9	0.004*
Turbidity (NTU)	10.2 ± 8.1	9.8 ± 5.3	0.711	15.8 ± 10.7	12.7 ± 5.5	0.293	14.6 ± 6.9	12.0 ± 5.7	0.107
chl-*a* (μg L^{-1})	7.3 ± 5.0	6.2 ± 2.7	0.285	8.6 ± 5.3	6.8 ± 3.4	0.103	9.2 ± 4.3	7.6 ± 4.0	0.184
NH_4^+ (μm)	0.46 ± 0.62	0.26 ± 0.37	0.062	0.58 ± 0.71	0.49 ± 0.59	0.395	0.58 ± 0.85	0.26 ± 0.34	0.020*
NO_x (μm)	0.30 ± 0.34	0.52 ± 0.51	0.074	0.31 ± 0.37	0.60 ± 0.71	0.087	0.27 ± 0.29	0.42 ± 0.43	0.177
DIN (μm)	0.71 ± 0.92	0.78 ± 0.82	0.646	0.86 ± 1.0	1.1 ± 1.2	0.52	0.86 ± 1.03	0.68 ± 0.71	0.667
PO_4^{3-} (μm)	0.20 ± 0.17	0.17 ± 0.09	0.757	0.62 ± 1.47	0.21 ± 0.12	0.19	0.33 ± 0.30	0.21 ± 0.13	0.177

avg. = average, SD = standard deviation, Wilcox. = Wilcoxon

project period and the postdredge project period were observed (Table 1). Notable differences in salinity were observed in the 2018 postdredge project period compared with the 2017 dredging project period. At all sites with the exception of Site 3, salinities during the postdredge project period were significantly lower than those observed during the dredging project (Table 1). At Sites 4, 5, and 6, the average salinities during the postdredge project period were 3.52, 2.42, and 5.46 psu less than during the dredging project. Finally, significantly higher NH_4 concentrations were measured at Sites 3, 4, and 5 during the dredging project period (Table 1).

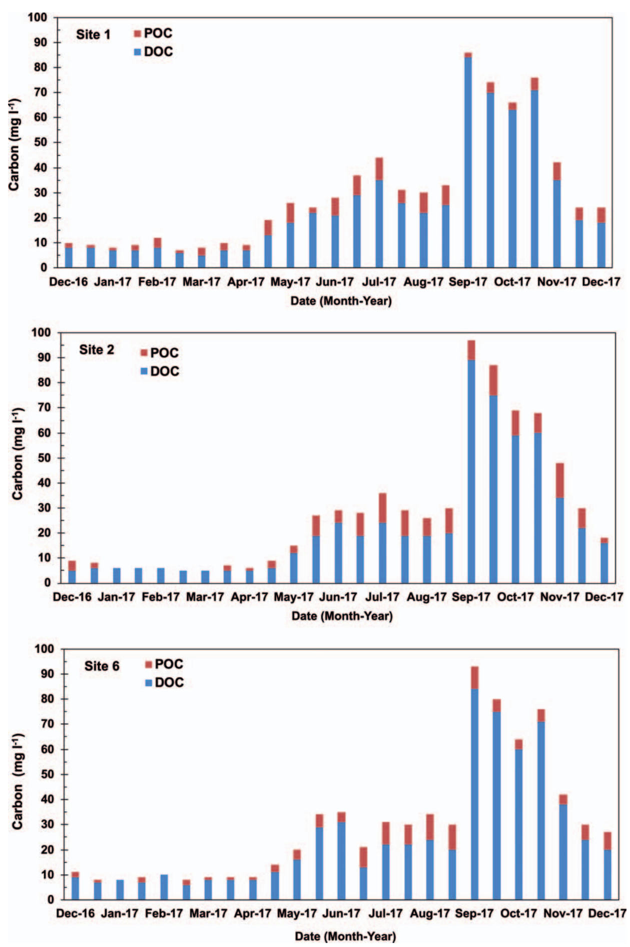

Figure 9. Dissolved organic carbon (blue) and particulate organic carbon (red) concentrations at Sites 1, 2, and 6 for the first year of the study period, December 2016–December 2017.

Using the salinity, turbidity, TSS, chl-*a*, DIN, and PO_4 data from both the dredging project period and the postdredge project period, the *k*-means clustering analysis determined the optimal number of clusters among sites, based on the gap statistic, to be five. The distribution of specific sites and dates across the clusters was highly uneven, with group memberships of 52, 58, 20, 10, and 121. There was no clear clustering of Sites 2 and 3 (closest to the restoration project site) during the dredging project period relative to other sites during the dredging project or among sites during the postdredge project period. The within-cluster sum of squares was higher than the between-cluster sum of squares, indicating high variation within each cluster. This was corroborated by a clusters' silhouette plot (data not shown), in which the average silhouette width of the clusters was 0.23. The silhouette width considers the compactness of a cluster and its separation from other clusters. The values range from –1 to 1, with 1 indicating perfect separation and –1 indicating a high level of misclassification (Roesseeuw, 1987). The overall silhouette width for this cluster analysis was 0.23, indicating the cluster structure is not strong and inferences about differences between sites in terms of earlier collective water quality parameters cannot be made.

DISCUSSION

Although the main focus of this study was to investigate potential changes in water quality associated with the 2017 dredging project in the Summer Haven River, the September and October 2017 storm events showed dramatic impacts on water quality relative to the dredging project. The subsequent discussion is separated into three sections: dredging impacts, storm event impacts, and observations of size-fractionated phytoplankton biomass over the project period in the MRE.

Restoration Dredging Impacts

TSS values during the dredging project period at all sites were nearly twice as great as the postdredge project period,

Table 1. *(extended).*

Site 4			Site 5			Site 6		
Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value	Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value	Dredge Period (avg. ± SD)	Nondredge Period (avg. ± SD)	Wilcox. *P* Value
35.08 ± 2.38	31.56 ± 3.81	0.016*	35.51 ± 1.88	33.09 ± 2.24	0.018*	33.64 ± 3.38	28.19 ± 5.02	0.005*
27.2 ± 23.8	14.5 ± 7.5	0.032*	20.7 ± 23.2	10.1 ± 6.8	0.026*	26.4 ± 23.9	12.6 ± 4.6	0.005*
12.5 ± 5.7	12.4 ± 4.5	0.741	9.9 ± 4.7	10.7 ± 5.0	0.984	11.9 ± 6.0	11.1 ± 2.6	0.741
8.2 ± 3.6	7.0 ± 3.6	0.184	7.8 ± 3.2	7.2 ± 3.5	0.589	10.3 ± 4.5	7.9 ± 3.9	0.093
0.69 ± 1.07	0.22 ± 0.26	0.014*	0.63 ± 1.07	0.21 ± 0.27	0.035*	1.22 ± 1.98	0.38 ± 0.41	0.098
0.22 ± 0.27	0.51 ± 0.90	0.162	0.30 ± 0.35	0.34 ± 0.37	0.944	0.25 ± 0.26	0.53 ± 0.62	0.124
0.86 ± 1.19	0.67 ± 1.10	0.881	0.84 ± 1.25	0.55 ± 0.60	0.535	1.45 ± 2.17	0.91 ± 0.92	0.646
0.31 ± 0.36	0.37 ± 0.21	0.165	0.26 ± 0.34	0.26 ± 0.15	0.267	0.50 ± 0.56	0.58 ± 0.34	0.211

with no significant differences in phytoplankton biomass as estimated by chl-*a* concentrations. This provides evidence that the dredging project likely was responsible for sediment resuspension and increased TSS values in the region. The increases in both TSS and NH_4 during the dredging project period are remarkable given that the Matanzas Inlet region is modeled to have flushing times on the order of 3 to 4 days (Sheng *et al.*, 2008) and a watershed-scale residence time of 2.16 d km^{-2} in the Pellicer Creek-Matanzas Inlet region, according to modeling by Gray *et al.* (2021). With these short flushing and residence times, it might be expected that resuspended sediment would be quickly transported out of the estuary. With regards to the increased NH_4 concentrations, particularly at Sites 3, 4, and 5, during the dredge period (Table 1), no sediment particle size analysis was carried out on the sediment removed from the Summer Haven River. However, it is possible that some removed sediment was fine-grained silt and clay with a high organic matter content that could have remobilized dissolved NH_4 upon resuspension. A significant amount of previous research has been conducted on the effects of dredging operations on planktonic and benthic organisms (DeCoursey and Vernberg, 1975; Fraser *et al.*, 2017; Harrell and Hall, 1991; Hedge, Knott, and Johnston, 2009; Thrush and Dayton, 2002; Wulff *et al.*, 1997). Less seems to be known about the impacts of dredging on water quality parameters. Certain water quality impacts of sediment excavation and dredging on estuarine systems have been documented in previous studies. Balchand and Rasheed (2000) showed short-term increases in the light extinction coefficient because of increased turbidity, an increase in water column nutrient concentrations, and increases in chl-*a* during a dredging operation in the Cochi estuary in India. These authors also note a relatively quick return to normal conditions postdredging. Lohrer and Wetz (2003) investigated the impact of a small-scale dredging operation on TSS and dissolved nutrient concentrations in a salt marsh tidal creek. The study found increased NH_4 concentrations in dredging plumes in the North Inlet estuary in South Carolina yet noted these concentrations were roughly 50% lower than during a natural annual peak. Similarly, although still greater than the postdredge project period, the range of average NH_4 concentrations observed over the 9-month dredging project period within this study at the six sampling sites, 0.46 to 1.22 μm, is significantly lower than the ~2- to 6-μm NH_4 concentrations measured during and after the major natural storm events in September and October 2017. Hossain, Eyre, and McKee (2004) showed how dredging in the Brisbane River estuary system led to the maintenance of two distinct turbidity maximum zones within the estuary and significantly increased the flushing time of the system. In contrast, this study showed no significant statistical difference in turbidity values between the dredging period and the postdredge period. Although sediment grain size analysis was not performed on the sediment removed, because of the breach of the Summer Haven River and infill with mostly sand, a large portion of the sediment removed was likely refractory and did not contribute significantly to increases in nutrient concentrations compared with whether the sediment removed was more fine silt or clay organic material.

Statistical analysis revealed that salinity was significantly lower during the postdredge period at all sites with the exception of Site 3 (Table 1). Boxplots of salinities illustrate the larger decreases in salinity at Sites 4, 5, and 6 during the postdredge period compared with the dredge period (Figure 10). Each of these sites is relatively close to the influence of Pellicer Creek, the main freshwater source to the MRE (Figure 2). The total precipitation during the 2018 postdredge project period (104.4 cm) was significantly less than that measured during the 2017 dredging project period (155.2 cm; data from the Pellicer Creek Meteorological Station). Thus, the lower salinities do not result from increased rainfall. It is possible that the watershed was still actively draining following these heavy storm events into 2018. However, if this was the case, the Pellicer Creek water discharge rates would likely be elevated, and they were not. Although it is beyond the scope of this paper, it is possible that the opening of the Summer Haven River and associated hydrodynamic changes in the estuary increased the outflow of fresher water from Pellicer Creek.

Although the dredging project period in 2017 did show increased TSS and increased NH_4 relative to the postdredge period in 2018, as mentioned earlier, it is worthwhile to examine how the increases in TSS and NH_4 compare with previously collected long-term data. Monthly averaged SJRWMD data from 2010 to 2016 showed no significant difference in chl-*a* and higher TSS relative to the 2017 dredging project period data collected at Site 5 (Wilcoxon sign-rank tests; data not shown). Although the TSS concentrations at Site 5 were greater during the dredge project period compared with the postdredge period, the long-term variability observed by

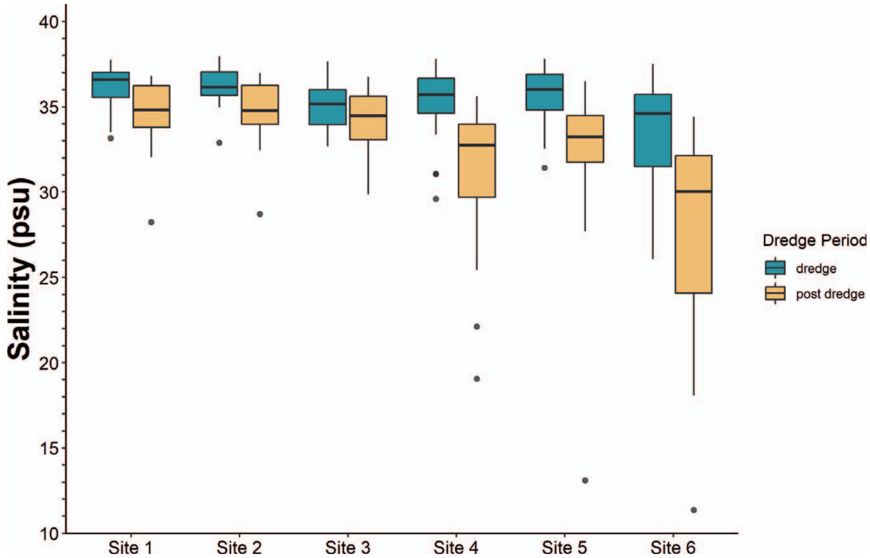

Figure 10. Boxplots showing the differences in salinity at the six sites during the dredge project period compared with the postdredge period. The boxplots show the minimum value, 25% quartile value, median, 75% quartile value, and maximum. Black dots above or below the boxplots indicate statistical outliers and were not included in the boxplots.

the 2010–16 SJRWMD was greater than these differences for TSS (Figure 11). A comparison of during-dredge and post-dredge NH_4 concentrations between this study and the 2010–16 SJRWMD NH_4 data is not possible because of differences in sample pretreatment (nonacidification *vs*. acidification of filtered samples).

The U.S. Environmental Protection Agency (EPA) and Florida Department of Environmental Protection (FDEP) have set forth surface water quality standards for various estuarine systems in the state of Florida. For the southern Matanzas River region, which is where this study took place, surface water quality standards exist for chlorophyll, turbidity, total phosphorus, and total nitrogen. The range of average chl-*a*

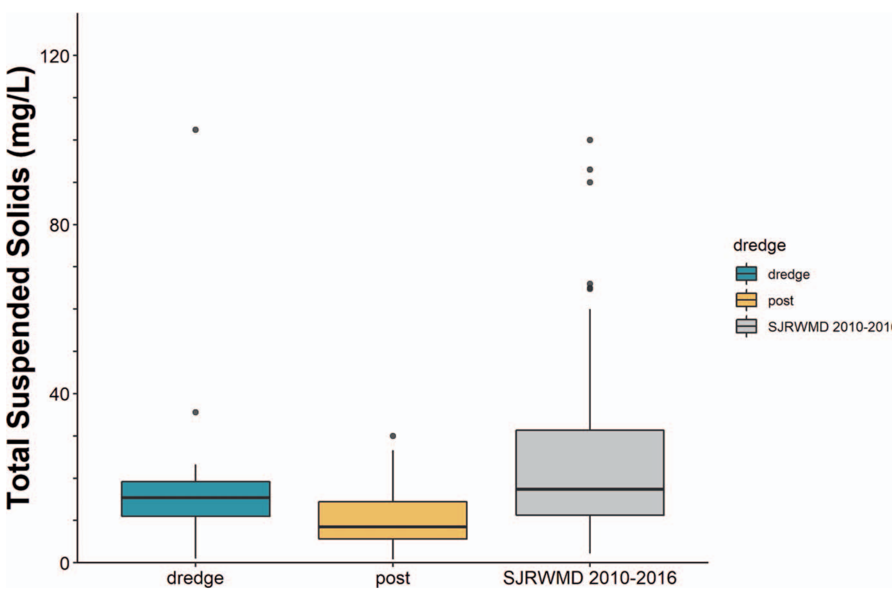

Figure 11. TSS concentration boxplots for the following periods: dredge (2017), postdredge (2018), and SJRWMD data collected monthly over the period 2010–16. The boxplots show the minimum value, 25% quartile value, median, 75% quartile value, and maximum. Black dots above or below the boxplots indicate statistical outliers and were not included in the boxplots.

values at the six sites (7.2–10.3 μg L^{-1}) during the dredging project period is above the 5.5 μg L^{-1} recommended annual geometric mean set forth by the EPA/FDEP. The EPA/FDEP surface water quality standard for turbidity for the southern Matanzas River designates as acceptable turbidity values less than 29 NTU above background values. From 2010 to 2016, the average monthly turbidity value from grab samples collected by the SJRWMD was 8.0 ± 2.4 NTU. The average monthly turbidity value collected over the January 2017 to September 2017 dredging project period was 9.9 ± 4.7 NTU. The dredging project period turbidity levels were in accordance with acceptable levels set forth by the EPA/FDEP. With regard to nutrients, whereas the DIN and PO_4 concentrations during the dredging project were an order of magnitude less than the recommended total nitrogen and total phosphorus surface water quality standards for the region, it is not possible to estimate whether surface water nutrient levels during the project were in accordance with EPA/FDEP standards, because total (dissolved + particulate) nutrient concentrations were not measured as part of this study.

Storm Impacts

Previous research has highlighted impacts from storm events on water quality in Pellicer Creek. Dix, Phlips, and Gleason (2008) found significantly decreased salinity and an approximate doubling of total nitrogen concentrations yet lower chl-*a* concentrations associated with four tropical storms during 2003 and 2004 as opposed to prestorm conditions. More recently, Schafer *et al*. (2022) investigated changes in both DOC and major nutrients in Pellicer Creek during Hurricane Irma and the 5-day nor'easter sampled in this study. The authors estimated that DOC fluxes out of Pellicer Creek increased more than 50-fold from less than 5 to 250 kg d^{-1} after Hurricane Irma and that the two storms accounted for nearly 67% of the average annual export of DOC from Pellicer Creek. Although increases in DOC at the MRE sites sampled here (Figure 10) during the storm events were significant, increasing by a factor of ~4, the reduction in the factor increases in DOC at Sites 1, 2, and 6 relative to Pellicer Creek points to a removal of DOC within the MRE, likely because of dilution of seawater with lower organic matter concentration, flocculation, and sedimentation. Both Sholkovitz (1976) and Bauer and Bianchi (2011) demonstrated and discussed the conversion of DOC upon estuarine mixing to particulate organic flocs.

It is possible to determine the DIN export contribution from Pellicer Creek during the two storm events relative to a typical year. Using U.S. Geological Survey (USGS) Pellicer Creek average discharge data from 2002 to 2021 (Site 02247222; USGS, 2023) and long-term average monthly DIN data collected at the GTMNERR Pellicer Creek sampling site from 2002 to 2021, a yearly average DIN export from Pellicer Creek of 4.35×10^4 kg N y–^{1} is estimated. Using DIN concentrations measured in Pellicer Creek approximately 1 week following each storm event within Pellicer Creek, as reported by Schafer *et al*. (2022), and USGS Pellicer Creek hourly water discharge rates during the storm events, a combined storm DIN export of 7.52×10^3 kg N was estimated. This combined DIN storm export is more than 17% of the average annual DIN discharge delivered in less than 3 weeks. This estimate is likely on the low end, because nutrient measurements were not made directly during the storm. Regardless, stemming from this increased, abrupt DIN export from Pellicer Creek, DIN concentrations at all sites within the MRE increased significantly from prestorm values of ~0.5 μm to values ranging from 2 to 5 μm, an increase of a factor of ~5 to 10, during and after the two storm events.

The shift from NH_4^+ to NO_x as the major contributing species to the DIN pool that occurred around mid-October 2017 to early November 2017 (Figure 7) following the storm events is noteworthy, because the major form of inorganic nitrogen in certain estuaries has been shown to affect phytoplankton community structure and phytoplankton growth (Glibert *et al*., 2016). Elevated NH_4^+ concentrations relative to NO_x have been shown to limit phytoplankton growth in San Francisco Bay (Dugdale *et al*., 2007) and Delaware Bay (Yoshiyama and Sharp, 2006). However, following the storm events when NO_x increased relative to NH_4^+ to become the dominant inorganic nitrogen species, the lack of increase in chl-*a* suggests inhibitory concentrations of NH_4^+ were not limiting production in the MRE. The concentrations of NH_4^+ were lower before the storm events than in urbanized estuaries such as San Francisco Bay and Delaware Bay. It is possible that the storm events led to increased nitrification and production of nitrate (NO_3^-) from NH_4^+. Hampel *et al*. (2020) observed decreases in NH_4^+ concentrations, increases in NO_3^- concentrations, and increases in rates of nitrification by two to three orders of magnitude in the St. Lucie estuary shortly following Hurricane Irma in September 2017. Those authors partly attributed the increased nitrification to wind-driven sediment resuspension and a sediment source of NH_4^+, along with increased oxygen in surface waters and light limitation because of high turbidity. Although increased turbidity was not observed following the storm events here, it could be that increased DOM played a role in light limitation of phytoplankton in the MRE.

The DIN values remain elevated in the MRE into January 2018, nearly 3 months after these storm events occurred (Figure 5). In addition, the fourfold increased DOC concentrations induced by the storm events remained elevated for nearly 2 months before decreasing in December 2017, whereas POC concentrations remained largely unchanged during this time. This time span of elevated DOC and DIN in the estuary system is quite remarkable and unexpected given the relatively short flushing times (~3–4 days; Sheng *et al*., 2008) and watershed residence times (2.16 d km^{-2}; Gray *et al*., 2021) in the southern MRE. The significantly elevated DOC observed 2 months after the storm events was similar to recent work by Asmala *et al*. (2020) that found storm-induced increases in DOC persisting for 50 to 200 days in the Neuse River estuary in North Carolina. In addition, Traving *et al*. (2017) showed that significant release of inorganic nutrients from bacterial processing of DOM occurs with increased organic matter loading in estuaries. It is plausible that the increased DOC loading from Pellicer Creek, upon mixing with the higher-salinity water of the MRE system, results in flocculation and sinking of this terrestrially derived DOC, providing a benthic source of DIN that persists longer than it would if maintained in the water column.

Although no measure of bacterial decomposition of DOM such as biochemical oxygen demand (BOD) or a time series of

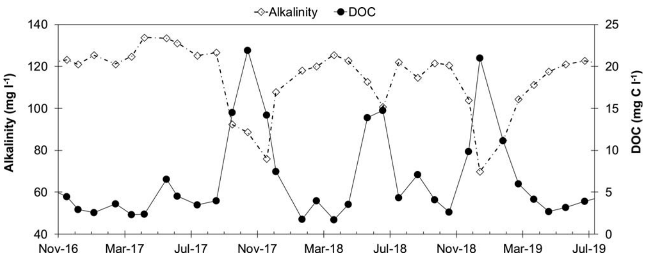

Figure 12. Alkalinity (left) and dissolved organic carbon (right) concentrations reported by the St. Johns River Water Management District from a surface water quality site close to Site 5 (Marineland) from this study.

DOM concentrations was made on water samples before, during, or after the storm events at any site, the decrease in oxygen concentrations accompanying the freshwater input from Pellicer Creek (Figure 4) was likely indicative of enhanced bacterial decomposition of DOM. Raymond and Bauer (2000) showed relatively low rates of DOC utilization by bacteria in the York River estuary (Virginia), with low temperatures as a limiting factor for much of the year. However, in the MRE following the storm events, water temperatures were greater than 20 to 28°C through most of fall 2017, and this could have facilitated increased bacterial respiration of DOC. McCabe *et al*. (2021) highlighted increases in BOD that were significantly correlated with increased POM concentrations following storm events and freshwater runoff in a South Carolina coastal plain wetland. These authors also noted that DOM, being the larger fraction of the organic matter pool at all sites sampled and the primary ingredient of bacterial decomposition, also contributed significantly to the increased BOD (McCabe *et al*., 2021).

Data collected by the SJRWMD close to Site 5 in September and October 2017 show a fivefold increase in DOC concentrations, an increase similar to those observed at the sites in this study, accompanied by a ~40% reduction in alkalinity (Figure 12) and a decrease from approximately pH 7.8 to pH 7.2. In addition, Figure 12 illustrates increases in DOC accompanied by decreases in alkalinity in June 2018 and January 2019 in the MRE. Most DOM emanating from rivers and creeks into estuarine systems are terrigenous substances such as humic and fulvic acids extracted from soils (Bauer and Bianchi, 2011). It might be suspected that the decrease in pH and alkalinity is associated with the input of these organic acids from Pellicer Creek. However, using data from three Georgia estuaries, Cai, Wang, and Hodson (1998) showed that a large portion of humic substances are either fully protonated or completely deprotonated during estuarine mixing and thus do not contribute to alkalinity changes or acidity changes. Although DOM can undergo various transformations once delivered from rivers into estuaries, two of the most significant are bacterial decomposition (respiration) of DOM and photochemical degradation of DOM, both of which release carbon dioxide (CO_2) in estuaries. Following storm events in the Neuse River estuary in North Carolina, Paerl *et al*. (2018) showed elevated pCO_2 accompanied by decreases in pH following wet and windy storm events. Paerl *et al*. (2018) also noted that in estuarine waters with a reduced buffering capacity (alkalinity), more of the microbially produced CO_2 from the processing of DOM would be lost to the atmosphere as CO_2, rather than being transported toward the ocean as bicarbonate ion or carbonate ion. Although measurement of the dissolved inorganic carbon species was not part of this study, the decreases in pH and alkalinity measured by the SJRWMD suggest that increased CO_2 from biological respiration and/or photodegradation of DOM following the storm events affected the normal biogeochemical cycling of carbon in the MRE. Future water quality studies associated with storm events in the MRE could benefit greatly from incorporation of carbon system parameters and BOD relative to baseline conditions, particularly with regard to the abundant oyster population in the region.

Previous work by Dix, Phlips, and Gleeson (2008) examined storm impacts on water quality in Pellicer Creek during a series of tropical storms in 2004. This research showed that increased rainfall was positively correlated with increased DIN in Pellicer Creek, similar to this study. The authors pointed out that because of the limited spatial extent of their study, conclusions could not be made about the spatial extent of nutrient dispersion from Pellicer Creek following storm events. The multiple sites sampled during this study (Figure 2) showed that the nutrient input from Pellicer Creek increases nutrient concentrations within the MRE both north and south of Pellicer Creek 3 miles or more for a period of 2 months following these two major storm events. It is somewhat surprising that during this period of elevated DIN following these storm events in September and October of 2017, there was no significant increase in phytoplankton biomass, as indicated by chl-*a* in this nitrogen-limited system. It seems that chl-*a* concentrations decreased over the September to October 2017 storm event period from the summer 2017 maximum (Figure 8). This is similar to the previous work of Dix, Phlips, and Gleeson (2008) that showed no significant phytoplankton biomass response to doubling of total nitrogen concentrations in Pellicer Creek. This is in contrast to the observations of Paerl *et al*. (2006), which showed that phytoplankton biomass increased nearly fivefold following major storm events in the Pamlico Sound in North Carolina. In addition, Paerl *et al*. (2014) found that significant rainfall associated with a tropical storm led to significant nutrient input, enhanced vertical stratification, and the subsequent bloom of a toxic dinoflagellate in the New River estuary, North Carolina. Reyna, Hardison, and Liu (2017) observed significant increases in phytoplankton biomass, along with increases in DIN, following storm events in the Mission-Aransas estuary system in Texas.

The contrasting results relative to this study bring forth the question of why a significant increase in DIN concentrations after the September and October 2017 storm events did not result in a significant increase in phytoplankton biomass in the nitrogen-limited MRE system. The storm events in September and October 2017 in the MRE resulted in a significant increase in DOC (and DOM), yet turbidity values remained somewhat low and agreed well with the 2010–16 SJRWMD data (Figure 5). Although colored DOM (CDOM) was not measured as part of this study, significant browning of surface waters (Figure 13) was noted in the month following these storm events, a result of the significant input of terrestrially derived DOM from the surrounding salt marsh and forest ecosystems. Tester *et al*.

Figure 13. Brownification of Matanzas River estuary surface water associated with a plume front of low salinity and high dissolved organic carbon near Site 3 (Summer Haven Bridge) on 6 October 2017 following two storm events (photo: Shannon Dunnigan, GTMNERR).

(2003) found an inverse relationship between CDOM concentrations and both depth of the 1% light level and chl-*a* concentrations following the passage of Hurricane Dennis and Hurricane Floyd in North Carolina estuaries. Branco and Kremer (2005) observed a strong positive correlation between CDOM absorption and the photosynthetically available light attenuation coefficient in three shallow NE U.S. estuaries. It is possible that although the input of significant DOM was not correlated with a significant increase in TSS or turbidity, high concentrations of CDOM led to light limitation of the phytoplankton community in the MRE during and after the storm events. More recently, Hart *et al*. (2015) found a negative relationship between high rainfall periods and phytoplankton biomass in the MRE system, possibly because of increased flushing rates during periods of high rainfall. It is likely that some combination of light limitation and increased flushing rate of the system led to the suppression of significant phytoplankton biomass accumulation after the enhanced nutrient supply because of these storms. However, an increased flushing rate would not support the nearly 2-month elevation of both DOC and DIN in the MRE system over these storm events.

Size-Fractionated Chl-*a*

As mentioned previously, this study provides the first reported size-fractionated chl-*a* measurements made in the MRE system. Although size fractionated chl-*a* determinations were only made during the first year of the study, this includes the January to September 2017 dredging period and the September to October 2017 storm events. Over both of these periods, the outstanding feature was the large percentage of chl-*a* found in the size fraction of greater than 20 μm. At Sites 1, 2, and 6, the size fraction of greater than 20 μm of chl-*a* represented 69 to 98% of the total chl-*a*, likely indicative of large pennate and centric diatoms and, to a lesser extent, dinoflagellates dominating the system. In a 2005–2009 analysis of phytoplankton biovolume in the MRE, Hart *et al*. (2015) showed increased biovolumes of large diatoms relative to dinoflagellates at a site near Matanzas Inlet. There was no significant change in relative chl-*a* size fractionations because of either the dredging operation or the storm events.

The dominant size class fraction found earlier is in contrast to a 2005–2009 analysis of phytoplankton biovolume by Hart *et al*. (2015) at the GTMNERR Fort Matanzas site, approximately 2.5 miles north of Matanzas Inlet (Figure 2). They found that spherical picocyanobacteria (0.7–2 μm) and cryptophytes (5–20 μm) represent 56% of the total phytoplankton biovolume at this site. Although an investigation of differences in phytoplankton community composition were not an objective of this study, this contrast in the 2005–2009 data at Fort Matanzas (Hart *et al*., 2015), where diatom biovolume greater than 30 μm accounted for only approximately 20% of total phytoplankton biovolume, and this study, where most, if not all, of the chl-*a* concentration was associated with a size fraction of greater than 20 μm, is noteworthy. This difference could partly result from measuring size-fractionated chl-*a* as opposed to biovolumes. In Apalachicola Bay, Putland and Iverson (2007) found that as the chl-*a* fraction of less than 5 μm increased, the carbon/chl-*a* ratios increased dramatically, indicating that smaller cells had less chl-*a*. In addition, Putland and Iverson (2007) show that as total chl-*a* in various estuaries increases, the percentage of chl-*a* less than 8 μm decreases.

One major factor that separates the GTMNERR Fort Matanzas site and the sites associated with this study is Matanzas Inlet. The sites within this study are south of Matanzas Inlet, whereas the GTMNERR Fort Matanzas site is north of the inlet. In a comparison of 2010–16 GTMNERR Fort Matanzas average monthly chl-*a* values with 2010–16 SJRWMD average monthly chl-*a* values collected in Marineland near Site 5, it was found that the chl-*a* data collected south of Matanzas Inlet near Site 5 was 34 to 83% greater between May and October than the Fort Matanzas site north of Matanzas Inlet. Recent work by Gray *et al*. (2021) calculated a watershed residence time of $\sim$2.2 d km^{-2} in the Marineland region and $\sim$1.1 d km^{-2} around the GTMNERR Fort Matanzas site. Although both of these residence times are relatively short, it is possible that increased residence times farther south of Matanzas Inlet, along with increased nutrient loading from Pellicer Creek, might lead to enhanced phytoplankton biomass and a different phytoplankton community structure south of the inlet. This highlights a potential future area of research in examining phytoplankton community assemblages N and S of Matanzas Inlet in relation to nutrient concentrations.

CONCLUSIONS

This study had the unique opportunity to observe the potential impacts of both a dredging project and major storm events on water quality in a SE U.S. estuary system. The results indicate that the barrier island maintenance dredging, compared with a postdredge period the following year, showed elevated TSS and NH_4 concentrations. However, these changes are not likely to be ecologically significant over the long term compared with natural annual variability and when taking into account the relatively rapid flushing times of the MRE. As coastal populations and development increase, there will likely be an increase in the number of dredging operations and dredging permit applications. The water quality changes mentioned earlier can be taken into consideration when trying to assess the impacts that these dredging operations might have. The decreased salinities during the postdredge period provide an area of additional research investigating potential hydrodynamic changes in the region.

The natural storm events hold more significance and impact with regard to water quality. In addition to the dramatic decreases in salinity, the significantly increased and relatively long-lasting (~2 month) DOC and DIN concentrations compared with the flushing times of the region are remarkable and merit further study with regard to microbial activity, the fate of the organic matter input, and potential changes in water residence time. With the projected increase in frequency and intensity of extreme precipitation events in the SE United States, it is important to further understand the impacts these storm events have on water quality, particularly in the areas of nutrient cycling, increased bacterial activity, and the fate of the increased DOC loads to estuarine waters as they relate to both nutrient sources and potential increases in pCO_2.

ACKNOWLEDGMENTS

Funding for this research was provided to the GTMNERR from the Florida Coastal Management Program *via* NOAA Award NA1SNOS4l90120. In addition, the authors thank two anonymous reviewers for their constructive feedback in improving the paper. The authors also thank Dr. Chris Kao at Flagler College for his valuable assistance and insight with statistical analyses. Finally, the authors thank the following Flagler College students who were responsible, under the mentorship of Dr. Matthew Brown, for the chl-*a* and nutrient analyses presented herein: Amanda Aydlett, Christopher Byrd, Breona Green, Trinity Lundeen, and Alexis Morris.

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