Muck Removal Efficiency plus Biological and Chemical Responses/Improvements after Dredging (Subtask 2)

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Impacts of Environmental Muck Dredging 2017–2018

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Final Project Report Submitted to
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Executive Summary

Removal of fine-grained, organic-rich sediments is an integral part of restoring the Indian River Lagoon (IRL) to a healthy ecosystem. This sediment, often referred to as IRL muck, is a concern because it can increase turbidity, consume oxygen, create an inhospitable benthic habitat and is an internal source of dissolved nitrogen (N) and phosphorus (P) that diffuse from muck into the overlying water of the lagoon. Dredging is being used as a method for removing large reservoirs of muck with associated N and P; however, there are challenges to dredging and few data are available to quantify the efficacy of environmental dredging in the IRL or elsewhere.

As the primary emphasis of this study, extensive surveys of sediment and water biogeochemistry, seagrass, benthic infauna and fisheries in Turkey Creek were carried out before dredging (April 2015–January 2016), during two separate phases of dredging (Phase I, February 20, 2016–April, 22, 2016 and Phase II, September 6, 2016–January 11, 2017), and after dredging during May 2016–April 2017. The goals of the Turkey Creek study were to track changes in (1) the distribution and composition of muck, (2) benthic fluxes of N and P from muck to the overlying water column, (3) sediment and water quality, (4) seagrass distribution and growth, (5) abundance, diversity and richness of benthic infauna and (6) abundance, diversity and microhabitat use of juvenile fishes and (7) feeding habits of fishes that could be impacted by dredging. Additional surveys were started before and during dredging at the Mims Boat Ramp and before dredging in Sykes Creek.

Pre-dredging surveys in Turkey Creek (February 2015) identified up to 3 m of muck in the study area between the Florida East Coast Railroad Bridge and the mouth of Turkey Creek with little to no muck in the adjacent IRL. Within the established dredge area, the efficacy of muck removal by conventional hydraulic dredging was ~63% based on removal of 52,000 of 83,000 m^3 of muck, which contained ~300 metric tons of N and ~70 metric tons of P. Dredging increased water depth and bay volume, a potential benefit to fishes, benthic fauna and seagrass. Benthic fluxes of N and P were ~50% lower at 3 months after dredging, which if continued would decrease annual releases of dissolved N and P from IRL muck by ~3 tons and ~1 ton, respectively, within the ~0.10 km^2 of Turkey Creek that were dredged. Monthly water quality surveys in Turkey Creek (April 2015 to April 2017) showed that the 1- to 2-m deeper water column after dredging contained twice the dissolved oxygen. Increase in oxygen may enhance resiliency to oxygen depletion events. Before, during and after dredging, values for dissolved ammonium and phosphate were highest in bottom water, consistent with sediments as a continuing source of these nutrients to the overlying water.

Surveys for seagrasses, drift algae and benthic infauna were conducted quarterly in Turkey Creek during the time period of muck removal listed above. Data for the occurrence of seagrasses (primarily *Halodule wrightii*) and drift algae (DA), included % cover, canopy heights, %

occurrence, and biomass. *H. wrightii* was not present within transects sampled in Turkey Creek nearest the dredge site. *H. wrightii* was most abundant (32% coverage), when present, in the shallower nearshore portions of transects (40–70 cm depth) within the adjacent IRL; abundance generally declined in winter to 0-5% visual percent cover. In contrast, drift algae, mainly *Gracilaria* spp. plus one or two other species, were more abundant in Turkey Creek than in the adjacent IRL. During 2017, lesser amounts of drift algae were observed in the IRL; however, large amounts were seen in Turkey Creek during winter and spring of 2017 and summer 2018. No statistically significant impact from dredging or muck removal was found in the abundance of seagrasses and drift algae. Rather, the small population changes observed in seagrass communities were more likely tied to simple seasonal changes, with winter months having the least abundances. Seagrasses are populations we expect to respond to longer term, general improvements in regional estuarine water quality, but not necessarily to local dredging.

The abundances and distributions of 83 species of invertebrate benthic infauna were determined via sediment grabs, and community parameters correlated with sediment geochemistry, especially percent organic matter. Richness and diversity of infaunal invertebrate communities were greatest at sandier IRL sites, almost nil within muck, and intermediate in Turkey Creek (TC) adjacent to where dredging occurred. Sediments in these areas displayed a gradient of fine-grained, organic-rich sediment characteristics, which co-varied with the occurrence of certain species and with diversity and richness patterns. Infaunal diversity and abundance were greater in sandier sediments. Dredged muck sites showed increases in infauna diversity, richness, and abundances in the months following dredging. Dredged sites with mixed sand and muck sometimes showed a decrease in these same metrics during 2017, likely from dredge-removal of infauna. A year later, however, infauna abundances and diversity at dredged intermediate stations were more like undredged intermediate stations.

Sampling with a 17-m seine net within the Turkey Creek region captured 59 taxa. These samples were dominated by small pelagic and demersal fishes, and did not include larger predatory fishes moving through the area. Catches were heavily dominated by schools of small pelagic fishes, particularly anchovies (*Anchoa* spp.), whose distributions reflected rapid movement of schools in and around the region without any direct association with dredging activities. Of the demersal fishes, the most abundant taxa were juvenile mojarras (*Eucinostomus* spp. and *Diapterus* spp.) and juvenile drums (Atlantic croaker, silver perch, red drum, juvenile seatrout and kingfish). All of these taxa showed large variations in abundance that reflected seasonal patterns in reproduction and interannual variation in lagoon-wide recruitment as well as responses to environmental perturbations in Turkey Creek such as periods of high freshwater discharge through the creek following major rain events. These taxa also exhibited variable patterns of microhabitat utilization: juvenile mojarras were most abundant in the hard sandy habitat along the western edge of the Turkey Creek embayment, silver perch and red drum were most common in the northern portion of the habitat characterized by oyster shells and softer sediments, and sea trout and kingfish were most common in the hard sandy habitat outside the mouth of Turkey Creek. Gut content analysis of these juveniles showed that most fed on epibenthic and infaunal crustaceans, mollusks and polychaetes. Although dredging did not have a detectable impact on the distribution and abundance of these juveniles, removal of muck appeared to have increased the abundance of some of these prey taxa. An increase in the prey base of these fishes indicates that the region could support increased numbers of juvenile fish, but the lack of complex habitat structure (e.g. seagrasses and oyster beds) that provides small fish with some protection against predators may mitigate against positive impacts of increasing food resources for the fish.

Dredged material from Turkey Creek was transported ~2 km north to a Dredge Material Management Area (DMMA) for settling and dewatering. Clarified water was discharged into the adjacent IRL. The overall retention efficiency for solids in the DMMA was >99.9%. Values for Total Suspended Solids (TSS) in the outfall from the DMMA to the IRL averaged ~28 mg/L during Phase I and 12 mg/L during Phase II, with four brief episodes of higher values. Background TSS values were 10–20 mg/L in the adjacent IRL. Chemical treatments effectively reduced concentrations of dissolved phosphate from as high as 10,000 µg P/L in the incoming dredged material to <40 µg P/L in clarified water released to the lagoon during Phase II, relative to <50 µg P/L in the lagoon. Total dissolved nitrogen in water released from the DMMA to the IRL was >5 mg N/L throughout the dredging process, relative to <0.8 mg/L in the lagoon. Additional efforts are now underway to decrease concentrations of dissolved nitrogen, mostly as ammonium, during future uses of DMMAs. Nutrient concentrations were at baseline values for the IRL at ~100 m from the outfall. We estimate that ~6 tons of N (~90% dissolved) and ~0.1 ton of P (~30% dissolved) were released from the DMMA during this one-year dredging project. Although unique to this particular IRL area, freshwater discharges annually release ~80 and ~5 tons of N and P, respectively, to Turkey Creek, far more than released from the DMMA.

Monitoring at the Mims Boat Ramp and Sykes Creek was initiated during 2017 and 2018, respectively. Dredging at the Mims Boat Ramp was carried out from April 23, 2018 to June 26, 2018 and then from August 29, 2018 to the conclusion on December 4, 2018. About 27,000 m^3 of wet muck sediment were removed from the Mims area in dredging that was completed in December 2018. About 80% of the muck, at thicknesses of 1–2 m, was found in a small basin at the southern end of the Mims dredge area. Benthic fluxes of N and P before dredging at the Mims area were 20–30% greater than in Turkey Creek. Mims seagrasses were primarily *Halodule wrightii* and had some of the densest populations observed in IRL during this study for the Environmental Muck Dredging (EMD) Project, but were absent from the densest muck areas of Mims. In adjacent sandier areas in summer, the high growth season, seagrass coverage reached as high as 100% and seagrass blades reached a length of 16.8 cm. Mims also had some of the highest observations of drift algae towards the end of the sandy transects, in the summer of 2018.

Regarding infauna, 45 species of benthic fauna, mostly sediment infauna of the phyla Arthropoda, Mollusca and Annelida, were observed at Mims. In the summer, infauna abundances reached a high of 2.5×10^4 animals per m^2 and richness was as great as 17 species per sample. Fishes were less abundant than in Turkey Creek, but the populations were again dominated by small pelagic species. Demersal juveniles were primarily species that spawn inside the lagoon, unlike at Turkey Creek where off-shore spawned larvae that recruit through Sebastian Inlet are dominant members of the community. At the completion of the third year of sampling (EMD3), Mims dredging was still underway. For this reason, before and after dredging comparisons of seagrasses and infauna, respectively, were not possible for Mims and associated sites. Biological sampling at Mims will continue for the near future, and the data herein will serve as baselines for comparison with post-dredging data yet to be collected.

At Sykes Creek, where dredging has not yet occurred, pre-dredging data provide a baseline for water/sediment quality and biological health (seagrasses, infauna, and fishes) for future comparisons after dredging occurs.

Measured Turkey Creek dredging outcomes during the timeframe of this study were as follows:

- Muck volume reduced by >60%
- Muck surface area reduced by ~20%
- Water volume increase due to increased column depth, of 160,000 m^3, accompanied by increased DO
- Composition of persistent muck was unchanged
- N and P fluxes reduced by 50% within 3 months after dredging
- Variable seagrass trends were indistinguishable from seasonal trends at control sites during the course of this study. Seagrasses are slow responding and should continue to be followed for possible long-term dredging responses, as it is hoped they will still respond to more general, regional improvements in water quality
- Some infauna indicators improved when dredging was carried out on muck sediments with the highest organic content, where metazoan life was largely absent prior to dredging. With intermediate sediments, however, dredging activity reduced some infauna
- In cases where infauna community indicators responded to dredging in the short term, continued monitoring showed that the biological community tracked sediment conditions, even in cases where they reverted to a pre-dredging state
- Fish captured in Turkey Creek seining were dominated by pelagic fishes (e.g., anchovies) and some small benthic species (e.g., juvenile mojarras and juvenile drums), whose occurrence is likely a function of typical, regional spawning cycles and microhabitat use.
- In summary, most chemical and biological factors improved after muck dredging; however, results were compounded/confounded by natural seasonal and yearly variation.

Recommendations include continued monitoring and refinement of muck removal techniques. Continued monitoring of seagrasses at Turkey Creek, Mims, Sykes Creek and associated sites on a quarterly schedule will allow comparison with historical data in different annual growth phases, and will be useful because seagrasses are expected to eventually respond to general regional improvements, though this may take some years. Quarterly monitoring of benthic invertebrates and sediment conditions at the same sites is important because these are the first environmentally dredged sites of many planned under the SOIRLPP; successes and failures related to environmental muck dredging have only begun to be observed and measured. Future dredging stands to benefit from a more full knowledge of early outcomes, including long-term effects. Improvement in the completeness of muck removal, or a softening of the angle of repose of dredged borders, could help rehabilitated sediments keep low organic content and host diverse and abundant benthic communities for longer after dredging.

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1.0 Introduction

Eutrophication continues to be a leading environmental challenge to coastal communities worldwide (Diaz and Rosenberg, 2008). Virtually all excess nutrients in the environment originate from anthropogenic nitrogen (N) fixation and phosphorus (P) mining (Howarth and Marino, 2006). Nutrients are released to coastal waters through a variety of mechanisms including, but not limited to: (1) excess fertilizer application and runoff, (2) animal and human waste and improper treatment of wastewater, (3) decomposition of organic matter (OM) from stormwater runoff and yard or agricultural trimmings and (4) air emissions that contribute to excess atmospheric deposition. Nutrients also are released to the overlying water column when sediment OM decomposes.

The Indian River lagoon (IRL) extends ~250 km along the central east coast of Florida. In the most restricted, northern section of the lagoon, no inlets are present along a 140-km section from Sebastian Inlet to Ponce de Leon Inlet. Since the 1950s, >4 million m^3 (~5.2 x 10^6 yd^3) of fine-grained, organic-rich sediment have accumulated in areas of the northern IRL that once were covered with sand and shell (Fox and Trefry, 2018a). Locally called IRL muck, this sediment contains 10–30% OM and >60% silt/clay with a high water content (porosity >0.9). IRL muck (1) is easily resuspended to block light from seagrass, (2) consumes oxygen, (3) is characterized by lack of biota and (4) continually releases large quantities of dissolved, bioavailable N and P.

Management of IRL muck requires a multifaceted approach that addresses both control of upland inputs of N, P and suspended sediments and removal of muck that has already accumulated. Muck removal is an integral part of the ongoing restoration process for the IRL. Although dredging has been proposed as a method for removing large reservoirs of N and P associated with fine-grained silt and clay, numerous challenges and limited data constrain present-day assessments of environmental dredging in the IRL. For example, a study of dredging in Crane Creek identified ~82,000 m^3 of muck in 2002, even though the creek had been dredged four years earlier (Trefry et al., 2004). However, previous studies such as this only documented volumetric changes and did not quantify flux reduction or biotic responses.

The primary focus of our study was in Turkey Creek with the following goals: (1) determine the efficiency of muck removal, (2) quantify any improvements in sediment and water quality after dredging, (3) track changes in species abundance and diversity for seagrasses, drift algae, benthic infauna and fishes throughout the dredging process, and (4) assess the effectiveness of sediment and nutrient retention in the Dredge Material Management Area (DMMA). Turkey Creek was a challenging setting due to its proximity to a large freshwater inflow and the presence of numerous docks and seawalls that limited the spatial extent of dredging. Despite these challenges, environmental dredging in Turkey Creek provided a unique opportunity for intensive evaluation of such efforts in the IRL.

Additional studies were carried out at the Mims Boat Ramp and in Sykes Creek. Dredging was completed in Mims during December 2018, but has not yet begun in Sykes Creek. Muck removal is intended to improve IRL ecosystems, providing an opportunity for stressed populations to rebound. Measuring critical ecosystems near dredging sites, both before, during, and after the dredging, allows us to evaluate the success of muck removal. Sampling areas proximal to planned dredging sites, as well as away at thriving areas, allows interpretations to be more conclusive on the driving forces behind observed changes.

Seagrasses are key indicators of lagoon health, promote biodiversity, and form critical habitat that serves as a nursery for juvenile fish populations (Virnstein and Morris 1996, Morris et al. 2001). They thrive in medium to low nutrient conditions in clear shallow water. Drift algae, while a natural part of estuarine and lagoon systems, tend to thrive at higher nutrient concentrations. Abundant drift algae can smother and/or shade seagrasses. Thus, the relative abundance of these two types of primary producers can indicate much about the relative condition of the ecosystem and possible eutrophication. At our study sites, sediments in the deeper water tended to contain muck, whereas shallower sediments were sandier, but often still contained some fine-grained, organic-rich sediments.

Macro- and microinvertebrates in or on estuarine sediments are food for benthic foraging fish, and their burrows and movements serve to aerate the sediments (Gonzalez-Ortiz et al. 2014). These organisms are perhaps the most directly affected by the conditions of sediments, and they respond to environmental changes more quickly than seagrasses. They are presumed to be negatively impacted by high organic content and unable to live in muck. However, the effects of muck on benthic invertebrate populations and communities are not well studied.

Seagrasses, fishes and invertebrates are the core communities comprising benthic estuarine ecosystems. By following these communities as benthic habitat is altered through environmental dredging, we will have a better understanding of their requirements and responses. Such data will also allow us to identify post-dredging habitat restoration objectives and methods. For instance, what % sediment organic content is needed for dredging to improve infaunal communities? What conditions might cause dredged areas to rapidly revert to pre-dredging conditions? What types of microhabitats are utilized by different species of fishes? How can these microhabitats be established through dredging activities and post-dredging restoration programs?

2.0 Approach

2.1. Overview of Report Format and Dredging Timetables

Our study of environmental dredging in the IRL was carried out at the following three locations: Turkey Creek, the Mims Boat Ramp and Sykes Creek. The most extensive study spanned 2015–2018 and included pre-, during- and post- dredging studies in Turkey Creek, particularly in its lower reach east of the U.S. 1 bridge. A geochemical study of the Turkey Creek Dredged Material Management Area (DMMA) was also conducted, the detail of which is provided in Fox et al. 2017, and summarized in Appendix A. Additionally, surveys at the Mims boat ramp during 2017 and 2018 were made before and during dredging. Both the Turkey Creek and Mims survey results on all 5 aspects of our study (i.e., biogeochemistry, seagrasses, drift algae, benthic infauna, and fisheries) are provided in this report, sections 3.0 and 4.0., respectively. The Sykes Creek study, provided in Appendix B, only includes the pre-dredge period (2018) as dredging there has not yet begun.

Dredging in Turkey Creek for Phase I began on February 20, 2016, with continuous operations (24 hours/day). A planned shutdown for manatee season during May and June was initiated 8 days early (April 22, 2016) because the DMMA reached capacity. Dredging resumed (Phase II) on September 6, 2016, for ~10 hours/day through January 11, 2017. Material dredged from Turkey Creek was transported ~2 km north through a submerged pipe to a DMMA (BV-52) where it was sprayed into the northwest corner of the ~0.04 km^2 retention basin. Clarification was initially accomplished by settling. Beginning on April 7, 2016, alum and proprietary polymers were injected directly into the dredge pipe at the DMMA to test their potential to reduce concentrations of dissolved P and enhance flocculation of suspended solids. A berm near the inflow prevented water from passing directly from the incoming pipe to the weir and the IRL, thereby increasing the residence time for water in the DMMA.

Dredging at the Mims Boat Ramp began on April 23, 2018 and continued until June 26, 2018, when a variety of constraints stopped dredging until August 29, 2018 when a second phase began. Dredging was completed on December 4, 2018.

2.2. Field Sampling and Laboratory Methods for Biogeochemistry

Muck surveys in Turkey Creek were carried out on four occasions: (1) before dredging on February 2015, (2) three months after dredging in March 2017, (3) after Hurricane Irma in December 2017 and (4) in December 2018 as a follow-up to post-hurricane dredging in February-March, 2018. Muck thicknesses were determined using a 4-cm diameter, capped polyvinyl chloride (PVC) pole (Figure 2.2.1a). The pole, marked in centimeter graduations, was lowered into the water column until the surface layer of sediment was encountered; depth was recorded as the water depth. The pole was then pushed into the sediment until a firm bottom was struck and total depth minus the water depth was recorded as the thickness of the muck layer. Usually, muck adhered to the pole along the muck interval to help confirm the thickness of the muck layer. Sediment cores were collected to verify muck thicknesses. Water depths were validated at ~5% of stations using a 20-cm diameter, weighted disk attached to a rope and lowered to settle on the sediment surface.

The pre-dredging survey in Turkey Creek included 253 probe measurements (counting duplicates of water depth and muck thickness and IRL data) within a gridded area of the creek (Trefry et al., 2016). The post-dredge survey of March 2017 included 249 probe measurements. Fifty locations in the immediate area of the follow-up dredging were occupied in December 2017 and December 2018, before and after limited re-dredging, respectively. Data from each survey were tabulated and elevations were normalized to the North American Vertical Datum of 1988 (NAVD88), a permanent benchmark of known elevation established during the initial survey. Contour maps for water depths, muck distribution, muck thicknesses and changes in waters depth and muck thicknesses were generated using ArcGIS (Version 10.2.2.3552, Esri, Redlands, CA).

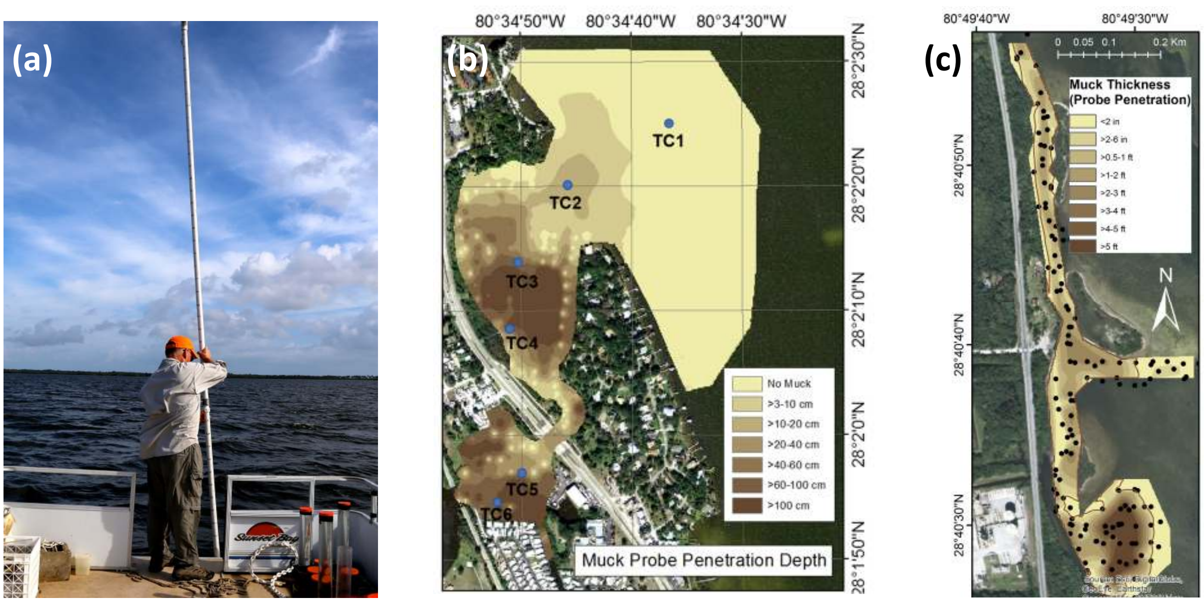


Figure 2.2.1. (a) Using a PVC pole to determine water depths and muck thicknesses, and contour maps for (b) Turkey Creek and (c) Mims Boat Ramp with muck distribution and thickness before dredging with locations for monthly water sampling for Turkey Creek (b) labelled TC1–TC6.

Surface sediment samples for detailed chemical analysis were collected at 9 locations in Turkey Creek before dredging and at 16 locations after dredging using a 0.1 m^2 Ekman grab. Sediment from the top 2 cm was placed in double Ziploc bags for grain size determination and in polycarbonate vials (~70 mL) for other chemical analyses. Sediment cores were collected at four stations before and after dredging (TC3, TC4, TC5 and TC6, Figure 2.2.1b) by divers using 60-cm long, 7-cm diameter cellulose acetate butyrate tubing. One core from each site was sub-sectioned upon return to the laboratory into 2-cm intervals for chemical analysis. Interstitial water was obtained from 16 sections along a 30-cm core using whole-core squeezers (Jahnke, 1988; Trefry et al., 2015).

More than 70 core samples for Quick-Flux analysis (Fox and Trefry, 2018a, b) were obtained from Turkey Creek using a 0.1 m^2 Ekman grab. The grab was gently lowered into surface sediments to obtain 10–15 cm of stratified sediment and overlying water. Sediment from the center of the grab was collected using a 1.5-cm diameter, mini-piston corer that was made from a plastic syringe. Mini-cores were sealed using Parafilm and stored in a cooler until processing in the laboratory at Florida Institute of Technology (FIT). In the laboratory, mini-cores were extruded into nitrogen-purged tubes and centrifuged at ~2000 RPM for 5 minutes. The supernatant was filtered through 0.45-µm pore size polypropylene filters and stored in low density polyethylene (LDPE) vials. Sulfide analysis was performed immediately (within 10 seconds of filtration).

Ten monthly pre-dredging water quality surveys were carried out in Turkey Creek from April 2015–January 2016. Water quality surveys also were conducted during, between and after dredging operations from February 2016 to April 2017. Post-dredging water quality surveys began at different dates as a function of when dredging was completed at a given location. Therefore, after-dredging surveys began as early as March 2016 near the mouth of the creek and continued to as recently as February 2017 in the area west of Highway US1. During each survey, water samples were collected at 2–5 depths at the same five stations (TC1–TC5) during each survey (Figure 2.2.1b). Vertical profiles for salinity, temperature, pH and dissolved oxygen were obtained using intercalibrated YSI 6600 V2 or YSI ProDSS probes (Yellow Springs Instruments). The sondes were calibrated at the beginning of each day following manufacturer's specifications. Discrete samples were collected through Tygon tubing attached to a peristaltic pump. Samples were placed in acid-washed LDPE bottles and stored in coolers until returned to the Marine & Environmental Chemistry Laboratories at FIT. Samples were then filtered within 2–3 hours.

Water depths and muck thicknesses in the area of the Mims Boat Ramp were determined with ~30-m spatial resolution before dredging on August 30, 2017 (Figure 2.2.1c). The muck thickness data collected with 30-m resolution were augmented during the same day via probing at adjacent locations, with higher resolution, to identify the margins where muck bordered sand (Figure 2.2.1c). Water and sediment samples also were collected during 2017. All field methods used in the Mims area were the same as described above for Turkey Creek.

All sediment samples, except sub-samples for grain size, were freeze dried using a Labconco FreeZone 6 system and then powdered using a SPEX Model 8000 Mixer/Mill. In preparation for analysis for Al, Fe, Si and P, 10–20 mg of freeze-dried, homogenized sediment or Certified Reference Material (CRM) sediment MESS-3, from the National Research Council of Canada (NRC), were totally digested in sealed Teflon tubes using concentrated, high-purity HF and HNO_3 following methods of Trefry and Trocine (1991). Complete digestion of the sediment was chosen because it accounts for the entire amount of each element in the sample. Concentrations of Al, Fe and Si in digested sediments were determined by flame atomic absorption spectrometry (FAAS) using a Perkin-Elmer Model 4000 atomic absorption spectrometer following U.S. EPA (1991) methods. Values for P were determined by inductively coupled plasma-mass spectrometry (ICP-MS) based on EPA Method 6020 (U.S. EPA, 1991) using a Varian Model 820-MS instrument. Concentrations of these elements in the sediment CRM MESS-3 were within the 95% confidence intervals for certified values. Analytical precision for individual elements in sediments ranged from 0.7–6% (as relative standard deviation, RSD = [SD/mean] x 100%). Eight sediment samples from the DMMA were completely digested and analyzed for 11 metals using techniques described by Trefry et al. (2014).

Grain size analyses were carried out using the classic method of Folk (1974) that includes a combination of wet sieving and pipette techniques. Loss on Ignition (LOI) at 550˚C was determined following the method of Heiri et al. (2001). Values for LOI estimate the fraction of OM in the sample and were used in conjunction with concentrations of organic C, total N and total P to help characterize sediment composition. Concentrations of $CaCO_3$ were determined by heating the sediment that had been treated for LOI at 550˚C to 950˚C following the method of Heiri et al. (2001).

Concentrations of total organic carbon (TOC) were determined using freeze-dried sediment that was treated with 10% (v/v) hydrochloric acid to remove any inorganic carbon, washed with carbon-free, high purity water (HPLC grade) and dried. Then, approximately 200–800 mg of pre-treated sediment were weighed into ceramic boats and combusted with pure oxygen at 950˚C using a LECO Corporation (St. Joseph, MI) TruMac C/N/S system with quantification of the CO_2 produced using an infrared detection cell. Nitrogen analyses of sediments also were carried out using the LECO system at 950˚C with quantification of the N_2 gas produced via a thermal conductivity detector. Concentrations of C and N in the sediment CRM MESS-3 and LECO reference sample 502-309 were within the 95% confidence intervals for certified values. Analytical precisions (RSD) were 1.5% for TOC and 2% for total N.

Filtered water samples and suspended particles were analyzed for dissolved and particulate chemicals following the techniques summarized below. Samples only for dissolved nutrient analysis were vacuum filtered through polycarbonate filters (Poretics, 47-mm diameter, and 0.4-µm pore size) in a laminar-flow hood. Concentrations of (1) nitrate + nitrite (N+N), (2) total dissolved nitrogen (TDN), (3) ortho-phosphate (PO_4^{3-}), (4) total dissolved phosphorus (TDP) and (5) silica were determined using a SEAL AA3 HR Continuous Segmented Flow AutoAnalyzer following manufacturer's method G-218-98. Organic and inorganic N compounds were converted to nitrate using UV irradiation and persulfate digestion. Nitrate was reduced to nitrite using a cadmium column. UV irradiation and persulfate digestion were used to free organically-bound P. The National Institute of Standards and Technology (NIST) traceable Dionex 5-Anion Standard was analyzed as a reference standard with each batch of samples to ensure accuracy.

Ammonium was quantified following standard methods (Rice et al., 2012) using UV-visible spectrometry. A Spex CertiPrep Cation Standard was analyzed as a reference standard with each batch of samples. Alkalinity was determined following method 2320-B (Rice et al., 2012). Samples were titrated with 0.01N HCl, and alkalinity was calculated using the Gran function. Alkalinity in mg $CaCO_3$/L is directly proportional to the volume of acid added to reach the pK_a.

Standard seawater solution (Ocean Science International Limited, UK) was analyzed as a reference standard with each batch of samples.

Samples of suspended matter were collected by vacuum filtering water through polycarbonate filters (Poretics, 47-mm diameter, 0.4-µm pore size) in a laminar-flow hood in our clean room at FIT. Prior to the field effort, the filters were acid washed in 10% HCl, rinsed three times with deionized water, dried and then weighed to the nearest µg under cleanroom conditions. Precision for replicate filtrations averaged <4% (i.e., ±0.04 mg/L). Samples for particulate organic carbon (POC) were filtered through pre-combusted Whatman Type A/E glass fiber filters mounted on acid-washed filtration glassware within a Class-100 laminar-flow hood. Particle-bearing filters were sealed in acid-washed petri dishes, labeled and then double-bagged in plastic and stored until dried and re-weighed at FIT.

Suspended particles, as well as separate milligram quantities of standard reference material (SRM) #2704, a river sediment issued by the NIST, were digested in stoppered, 15-mL Teflon test tubes using Ultrex II HNO_3 and HF as described by Trefry and Trocine (1991). Concentrations of particulate Al, Fe and Si were determined by FAAS and concentrations of particulate P were determined by ICP-MS. Analytical precision (RSD) for individual elements in sediments ranged from 1–5%.

Concentrations of POC and total N were determined by first treating particles on the glass fiber filter with 10% (v/v) hydrochloric acid to remove any inorganic carbon, and dried. Then, filters with 200–800 mg of pre-treated suspended sediment were weighed into ceramic boats and combusted with pure oxygen at 950˚C using a LECO TruMac C/N/S system. Total N concentrations were determined using separate glass fiber filters that were untreated prior to analysis to avoid losses of N during acidification. Nitrogen analyses of suspended particles also were carried out using the LECO system at 950˚C with quantification of the N_2 gas produced via a thermal conductivity detector. Concentrations of C and N in the sediment CRM MESS-3, SRM #2704 and LECO reference sample 502-309 were within the 95% confidence intervals for certified values. Analytical precision (RSD) was 1.5% for TOC and 2% for total N.

Diffusive fluxes of N and P were calculated from gradients of solute concentrations in interstitial water (e.g., Berner 1974; Boudreau and Scott, 1978) and Fick's first law of diffusion (Equation 2.1). Diffusion coefficients (D_s) were corrected for variations in temperature, salinity and porosity of sediments. Concentration gradients were obtained from (1) discrete samples obtained via whole core squeezers and (2) calculated from integrated sediment samples obtained using the Quick-Flux technique (Fox and Trefry, 2018b). Standard deviations for water profiles were calculated from downcore variability in nutrient concentrations and for Quick-Flux using field replication.

$$\mathrm{F} = -D_S \frac{dc}{dx} \qquad \text{(Equation 2.1)}$$

2.3. Field Sampling and Laboratory Methods for Seagrasses and Drift Algae

Sampling for the Turkey Creek study included (1) within Turkey Creek (TC, Figure 2.3.1a), (2) the Indian River Lagoon (IRL) near Turkey Creek (TCL, Figure 2.3.1b) and (3) the IRL near Crane Creek (CCL, Figure 2.3.1c). Seagrass and drift algae sampling at TC, TCL and CCL were conducted quarterly during EMD3 (June 2017–June 2018), and monthly for the two years prior. At least three 100-m transects were surveyed perpendicular to the shoreline for each site-treatment combination, with the goal of documenting the presence of seagrasses and drift algae.

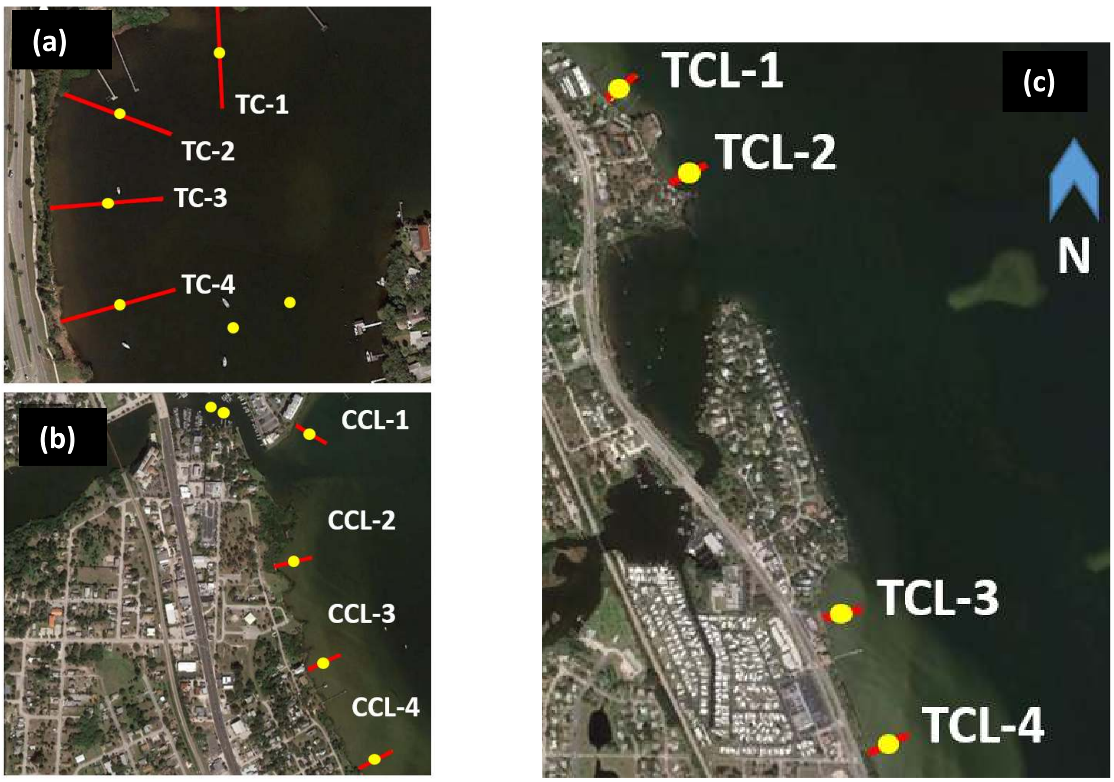


Figure 2.3.1. Sampling stations associated with Turkey Creek, including (a) Turkey Creek and Palm Bay with locations of seagrass transects TC-1–TC-4, (b) comparison study site in the Indian River Lagoon near the mouth of Turkey Creek (TCL) and locations of TCL transects (TCL-1–TCL-4), (c) comparison study site in the Indian River Lagoon near the mouth of Crane Creek (CCL) and locations of CCL transects (CCL-1-CCL-4). Yellow dots indicate locations of infaunal sampling (triplicates of grab samples at each marked location). Seagrass transect lengths (red lines) are 100 m.

Muck stations at the Mims dredging site (MDM) and sandier stations immediately adjacent to the dredging (MDS) occurring at the Mims boat ramp and channel 2018 (Figure 2.3.2). Control stations consisted of muck (MCM) and sandier stations (MCS) 4.0 km south, also within the Indian River Lagoon, but on the south side of a causeway (Figure 2.3.2). Seagrass and drift algae sampling at MDS and MCS stations were conducted quarterly during EMD3 (June 2017-June

2018), and infauna were sampled at all Mims stations (MDM, MDS, MCM and MCS) on the same dates.

Quadrats were laid down every 10 m along the transect lines, and seagrasses and drift algae were scored according to standard methods (Virnstein and Morris, 1996; Morris et al., 2001). Measurements included seagrass visual estimate % cover (estimated coverage upon imagining the seagrass crowded into corner of quadrat at a high density), seagrass % occurrence (proportion of 100 quadrat sub-squares having at least 1 blade of seagrass), seagrass density (# of shoots per area), seagrass canopy height (the length of blade from sediment to tip), drift algae % occurrence (the proportion of 100 quadrat sub-squares having any drift algae), drift algae biomass estimate (estimated coverage upon imagining drift algae crowded into corner of quadrat), and drift algae canopy height (Virnstein & Morris, 1996; Morris et al., 2001). Seagrass and algal percent cover and shoot length were plotted as means ± SE.

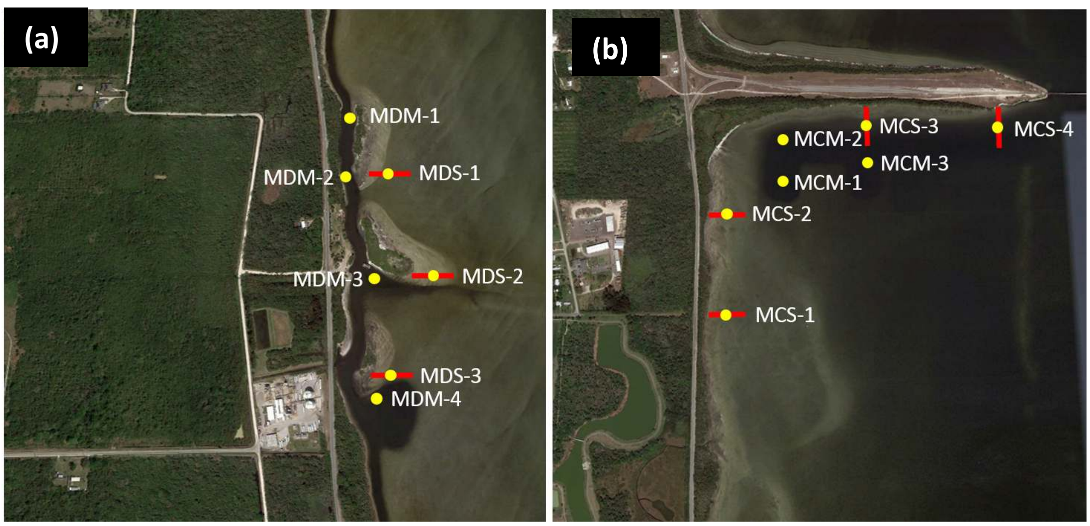


Figure 2.3.2. Sampling stations associated with Mims, including (a) muck stations at the Mims Boat Ramp Dredging site (MDM-1- MDM-4) and sandier stations (MDS-1-MDS-3), and (b) sampling stations at the Mims Control site, including muck (MCM-1-MCM-3) and sandier stations (MCS-1-MCS-4). Yellow dots indicate locations of infaunal sampling (triplicates of grab samples at each marked location). Seagrass transect lengths (red lines) are 100 m.

2.4. Field Sampling and Laboratory Methods for Benthic Fauna

Sediment grabs for infaunal analysis were collected at the 50-m mark along all seagrass transects in Turkey Creek and at Mims as described above (Figures 2.3.1 and 2.3.2) via petite Ponar grab (n = 3 per transect). In addition, three stations were selected near the mid-point of the most concentrated muck sediments; triplicate samples were collected at each dredging and control muck station. Sampling and identification of infauna were conducted consistent with the methods of ongoing benthic studies of the IRL (Mason 1998, Cooksey and Hyland 2007, Tunberg et al.

2008b). Diversity was determined as Shannon-Wiener diversity index and richness is the simply the number of species. Abundances, diversity, and richness of fauna were tested for correlations with sediment parameters, including % organic content (dry weight), % water content by weight, and % silt/clay content (dry weight). Where appropriate, statistical analyses included Analysis of Variance (ANOVA) for spatial comparisons on a given day, ANOVA for temporal comparisons for a given site, Non-Metric Multidimensional Scaling (NMDS) community analysis with posthoc Analysis of Similarity (ANOSIM), Principle Components Analysis (PCA), and regression correlation analysis comparing biological data to corresponding sediment data.

In 2017–2018, during EMD3, sampling rotated monthly through each of the sites, yielding a quarterly/seasonal sampling schedule. Prior to that, during EMD1 and EMD2, sampling was primarily in Turkey and Crane Creeks and occurred monthly. All sampling events included sediments for geochemical analysis, seagrasses, drift algae, and benthic infauna. Table 2.4.1 summarizes the sampling schedule.

Table 2.4.1. Sampling schedule from 2015–2018. Months are represented by columns and presented in chronological order with the first letter of each month at the head of the column. 'X' indicates that all associated sampling occurred, including control sites and replication. Arrows demark the start and completion of dredging, as indicated.

2015	J	F	M	A	M	J	J	A	S	O	N	D
Turkey/Crane Creek												
Muck Survey		X										
Sediment Chemistry		X										
Water Quality				X					X	X	X	X
Sediments for Biology							X	X	X	X	X	X
Seagrasses/Drift Algae							X	X	X	X	X	X
Infauna							X	X	X	X	X	X
Fish Seining Surveys				X	X	X	X	X	X	X	X	X
Sykes Creek												
Sediments												
Seagrasses/Drift Algae												
Infauna												
Infauna												
Mims Boat Ramp												
Muck Survey												
Sediment Chemistry												
Water Quality												
Sediments												
Seagrasses/Drift Algae												
Infauna												
Fish Seining Surveys												

2016	J	F	M	A	M	J	J	A	S	O	N	D
Turkey/Crane Creek												
Muck Survey												
Sediment Chemistry									X	X	X	X
Water Quality	X	X	X	X	X	X	X	X	X	X	X	X
Sediments for Biology	X	X	X	X	X	X	X	X	X	X	X	X
Seagrasses/Drift Algae	X	X	X	X	X	X	X	X	X	X	X	X
Infauna	X	X	X	X	X	X	X	X	X	X	X	X
Fish Seining Surveys	X	X	X	X	X	X	X	X	X	X	X	X
Sykes Creek												
Sediments												
Seagrasses/Drift Algae												
Infauna												
Infauna												
Mims Boat Ramp												
Muck Survey												
Sediment Chemistry												
Water Quality												
Sediments												
Seagrasses/Drift Algae												
Infauna												
Fish Seining Surveys												

2017	J	F	M	A	M	J	J	A	S	O	N	D
Turkey/Crane Creek												
Muck Survey								X				X
Sediment Chemistry	X	X	X	X	X	X	X	X	X	X	X	X
Water Quality	X	X	X	X	X	X	X	X				
Sediments for Biology	X	X	X	X	X		X			X		X
Seagrasses/Drift Algae	X	X	X	X	X		X			X		X
Infauna	X	X	X	X	X		X			X		X
Fish Seining Surveys	X	X	X	X	X		X			X		X
Sykes Creek												
Sediments												
Seagrasses/Drift Algae												
Infauna												
Infauna												
Mims Boat Ramp												
Muck Survey								X	X			
Sediment Chemistry								X	X			
Water Quality								X	X			
Sediments			X				X		X		X	
Seagrasses/Drift Algae			X				X		X		X	
Infauna			X				X		X		X	
Fish Seining Surveys							X		X		X	

2018	J	F	M	A	M	J	J	A	S	O	N	D
Turkey/Crane Creek												
Muck Survey												X
Sediment Chemistry	X		X		X							
Water Quality												
Sediments for Biology			X			X						
Seagrasses/Drift Algae			X			X						
Infauna			X			X						
Fish Seining Surveys			X			X						
Sykes Creek												
Sediments	X			X			X					
Seagrasses/Drift Algae	X			X			X					
Infauna	X			X			X					
Infauna		X		X		X						
Mims Boat Ramp												
Muck Survey						X						
Sediment Chemistry						X						
Water Quality						X						
Sediments		X						X		X		
Seagrasses/Drift Algae		X						X		X		
Infauna		X						X		X		
Fish Seining Surveys	X		X		X							

↑ Start TC Dredging (January 2016) ↑ End TC Dredging (January 2017) ↑ Start Mims Dredging (May 2018)

2.5. Field Sampling and Laboratory Methods for Fisheries Surveys

Turkey Creek:

All fisheries monitoring within the dredging region was conducted west of the mouth of Turkey Creek and east of the US 1 bridge. This region was the focus of dredging activity in Turkey Creek, and was the only area that could be sampled by seine net. In addition, fisheries monitoring was conducted in the IRL just north of the mouth of Turkey Creek for comparison with the sampling adjacent to dredging activity.

Sampling Sites and Methods

Fish samples were collected monthly from April 2015 through July 2017, continuing with sampling at 2-3 month intervals through July 2018. Samples were collected from a series of 4 sites along the western shore of Turkey Creek (F-W in Figure 2.5.1), 4 sites along the north shore of Turkey Creek (F-N), and 2 sites outside the mouth of Turkey Creek (F-O). All sampling was done following standardized Fisheries-Independent Monitoring (FIM) seining protocols (Florida Fish and Wildlife Conservation Commission, 2014a). Fishes were collected with a 21.3-m long center bag seine x 1.8-m deep, and constructed of 3.2-mm knotless nylon Delta mesh. A 15.5-m rope was tied to the towing poles at each end of the seine, and 9-m ropes were attached to poles placed in the sediment at the beginning of each tow.

These guide ropes ensured that the seine sampled a standardized area of 140 m^2. Sample locations within each of the 3 regions were randomly selected by water depth and substrate (Figure 2.5.1). Within each Turkey Creek site, 2 samples were collected by towing the seine parallel to the shore with the near-shore pole at an approximate starting depth of 25-50 cm. Two samples were taken farther from shore, generally at a starting depth of about 50 cm to 1 m. Station designations used in this report, such as F-N1 refer to a shallow and deep pair of seine hauls taken in the eastern half of the block. F-N 2, F-W3 and F-W 4 refer to shallow/deep sample pairs proceeding counterclockwise from F-N 1. Although we planned to collect 10 samples throughout the region each month, on several occasions we were prevented from collecting a complete set when blocked by dredge or discharge pipes stretched across the sampling site.

Figure 2.5.1. Primary fish sampling sites along the western shore of Turkey Creek (F-W; 4 monthly samples), north shore of Turkey Creek (F-N; 4 monthly samples) and outside of the mouth of Turkey Creek (F-O; 2 monthly samples).

On another occasion, a dense mass of drift algae (*Gracilaria* sp.) completely covered the shallow habitats of F-N. During clean-up after Hurricane Irma in 2017, the sites at F-O were blocked by construction equipment rebuilding the beach and park structures.

Prior to dredging, the location of the deeper tows was restricted by the beginning of increasingly soft sediments, defined as when the seine personnel sank knee deep (about 30 cm) in muck, and could no longer effectively drag the net. After the dredging, the sharp edge of the dredge cut marked the deepest edge of the sampling area. Regions F-W and F-N were the only areas within Turkey Creek where the seine net could be effectively deployed.

Following completion of a seine tow, fishes were identified to the lowest practical taxonomic level and counted. A sample splitter was used to estimate numbers of very large catches of anchovies and several other species. Large, distinctive fishes were removed from the sample prior to splitting, with the remaining sample repeatedly being split in half until the subsample was sufficiently small (generally 500-1000 fishes) to be processed. The number of splits was used to estimate the total number of each taxon in that sample. Standard lengths of up to 25 specimens of each taxon were recorded. Voucher specimens were placed on ice for laboratory identification, if necessary. Up to 25 specimens of demersal juvenile fish taxa were placed on ice and frozen in the laboratory for subsequent stomach contents analysis.

Data analysis began by converting abundance data to density data (number of fish/100 m^2). Monthly mean densities were then calculated for inside Turkey Creek (generally 8 samples) and outside Turkey Creek (2 samples). Species-specific analyses were then employed to examine temporal and spatial distribution patterns across the region throughout the duration of the study. High variability was noted for all taxa, reflecting the influence of changing environmental conditions within the sampling region, varying seasonal patterns in reproduction, and interannual variability of recruitment, as well as potential responses to removal of muck from the region.

It is beyond the scope of this project to assess whether that interannual variability was due solely to responses to dredging activities that occurred during much of 2016, or due to variability in recruitment levels within the broader IRL during our sampling efforts from 2015-2018. However, for an initial examination of this interannual variability, fish population data from Turkey Creek samples collected from May 2015 to December 2017 were compared to data collected from 2010 to 2017 by the FIM program in portions of the IRL (FWCC 2010, 2011, 2012, 2013, 2014b, 2015, 2016, 2017).

Other studies have characterized the broad temporal and spatial patterns of habitat use by fish taxa in the IRL and other habitats by using Canonical Correspondence Analyses (CCA) and Multidimensional Scaling (MDS) ordination, as well as Analysis of Similarity (ANOSIM) and Similarity Percentage (SIMPER) (e.g. Paperno and Brody, 2004; Paperno et al., 2006). These analytical techniques are suitable for studies that span a wide range of habitats and environmental conditions. They can also be used to examine community-level responses of populations, although the inherent assumption that distributions reflect species interactions within a defined community, rather than being individual species responses to biotic and abiotic conditions, may not be warranted.

A different approach is used for this analysis of fishes within the spatially limited habitat around the mouth of Turkey Creek. Examination of the density data indicate that a very small proportion of the seine hauls made throughout the survey contained large numbers of a given taxon; conversely, very high proportions of seine hauls had few, if any, representatives of that taxon. Those large catches provided evidence for microscale habitat preferences within the study region and seasonal patterns in reproduction.

A similar focus on episodes of peak abundance has been used in studies of larval fish recruitment dynamics to determine the meteorological, oceanographic and tidal influences that dominated larval supply of groupers and other taxa into reef ecosystems in the Bahamas (Shenker et al. 1993), as well as larval snappers (Halvorsen, 1998) and larval tarpon (Shenker et al. 2002) into the Indian River Lagoon. This approach enables identification of species-specific recruitment patterns, without making the assumption that larvae form a coherent, interacting community, or that there are consistent multi-species responses to environmental conditions.

Peak Abundance Analyses (PAA) were performed for the 7 most abundant demersal taxa collected in the survey. For each taxon, an abundance threshold was established to identify the largest catches for analysis. The initial threshold was set at catches that exceeded 1% of the total catch of that taxon throughout that survey. Site location (F-O, F-N, F-W) and season data associated with each of the large catches were used in a contingency-table analysis (G-test) to determine if the relative proportion of those large catches were evenly distributed across sites or depths, or if they were concentrated in particular microhabitats.

One of the primary factors affecting fish distribution among microhabitats is the availability of prey for different taxa and life stages. To assess the feeding habits of juvenile demersal fishes, frozen samples were thawed and stomachs were removed. Contents were rated on a 4-level scale of digestion (0 = completely unidentifiable to 3 = identifiable to species). Prey removed from each stomach were identified to the lowest feasible taxonomic level, and counted. Identifications of invertebrate taxa were conducted using the photographic atlas of benthic species collected from the Turkey Creek region that was produced by the research team studying the benthic fauna (Johnson, pers. comm.).

Analysis of the stomach contents data were based on Frequency of Occurrence (%FO) of a prey taxon in fishes that contained at least one identifiable prey item in their stomachs. Analysis of stomach contents of larger fish often utilize %FO with proportional numbers and weights of prey in each stomach to calculate an Index of Relative Importance (IRI) for each prey taxon. Because most of these juveniles had only 1 or 2 prey items in their stomachs, and the small size of the digesting prey precluded accurate weight measurements, we used %FO, rather than IRI, to evaluate prey consumption. Comparison of feeding data with benthic prey within the Turkey Creek region was conducted using benthic species abundance data in and around Turkey Creek before and after dredging occurred (Cox et al. 2018; Mallick 2019) and in the broader IRL ecosystem (Nelson 1981, 1995; Nelson et al. 1982).

Mims site:

The western shoreline of the northern IRL near the town of Mims is characterized by a deep muck-filled channel that separates the shoreline from extensive shallow sandy habitats. In anticipation of removal of the muck by dredging activity in mid-late 2018, we conducted bimonthly sampling of the habitat from September 2017-July 2018. A total of 9 stations were seined during sampling episodes (Figure 2.5.2). Eight of the stations were on the sand flats east of the channel and adjacent spoil island. We were able to sample directly over a narrow muck-filled channel leading to a boat launching ramp; the seine net extended directly across the channel, allowing personnel to pull the net from the relatively hard bottom adjacent to the channel. Seine and sample processing procedures were identical to those used in Turkey Creek seine surveys.

Figure 2.5.2. Seine sites for fish sampling near Mims in the northern Indian River Lagoon. Mims boat ramp is 28°40’39”N 80°49’35”W. Scale bar = 200 m.

2.6. Quality Assurance Plan

A revised and detailed Quality Assurance Plan (QA Plan) with responses to comments for the second year of this study was submitted during January 2017. This plan meets the minimum requirements for description of Research Field and Laboratory Procedures according to Rule 62-160.600, F.A.C. The documents cover project plans, objectives and analyses for the dredging component of the EMD study. As a continuation project, the January 2017 QA Plan covered the activities and analyses for 2017–2018. More detailed summaries of our overall QA guidelines are available in Johnson and Shenker (2016), Trefry et al. (2016), Fox et al. (2017) and Johnson (2017).

3.0 Dredging in Turkey Creek

The Results and Discussion for biogeochemistry (sections 3.1 – 3.4) address our goals of determining the effectiveness of muck removal and its impact on sediment and water quality in Turkey Creek by connecting a variety of data sets, timeframes and concepts. We begin with results for the quantity of muck present before and after dredging. Next, we present data for the general composition of sediments, including muck, in Turkey Creek before and after dredging. Then, results for fluxes of dissolved N and P from muck sediments to the overlying water column throughout the dredging process are presented. The flux section is followed by an assessment of changes in water quality parameters during the study. Results for our study of the DMMA are discussed in Appendix A.

3.1. *Dredging and Effectiveness of Muck Removal*

Water depths and thicknesses of muck deposits were determined with ~30-m spatial resolution in Turkey Creek before dredging in February 2015 and after dredging in March 2017 (Figure 3.1.1). Measurements in the adjoining IRL were made with ~150-m resolution. Our surveys extended from the Florida East Coast (FEC) Railroad Bridge through the water body named Palm Bay to the mouth of creek (Figure 3.1.1). The area designated for dredging was restricted to ~60% of the 160,000 m^2 of total area from the FEC to the mouth of creek due to the presence of docks, seawalls or known sand deposits that did not need to be dredged (Figure 3.1.2).

Water depths >4 m were found at 1 of 111 sites surveyed before dredging in the area to be dredged. After dredging, 71 of the same 111 sites had water depths >4 m when elevations were normalized to the initial survey (Fox and Trefry, 2018a, b). The average water depth for pre- versus post-dredging increased from 2.1 to 3.8 m in the dredged area; maximum water depth increased from 4.4 to 6.0 m in the dredged area. Increased water depths in dredged areas can modify water flow, increase residence times for bottom water and affect water chemistry as described in Section 3.4.

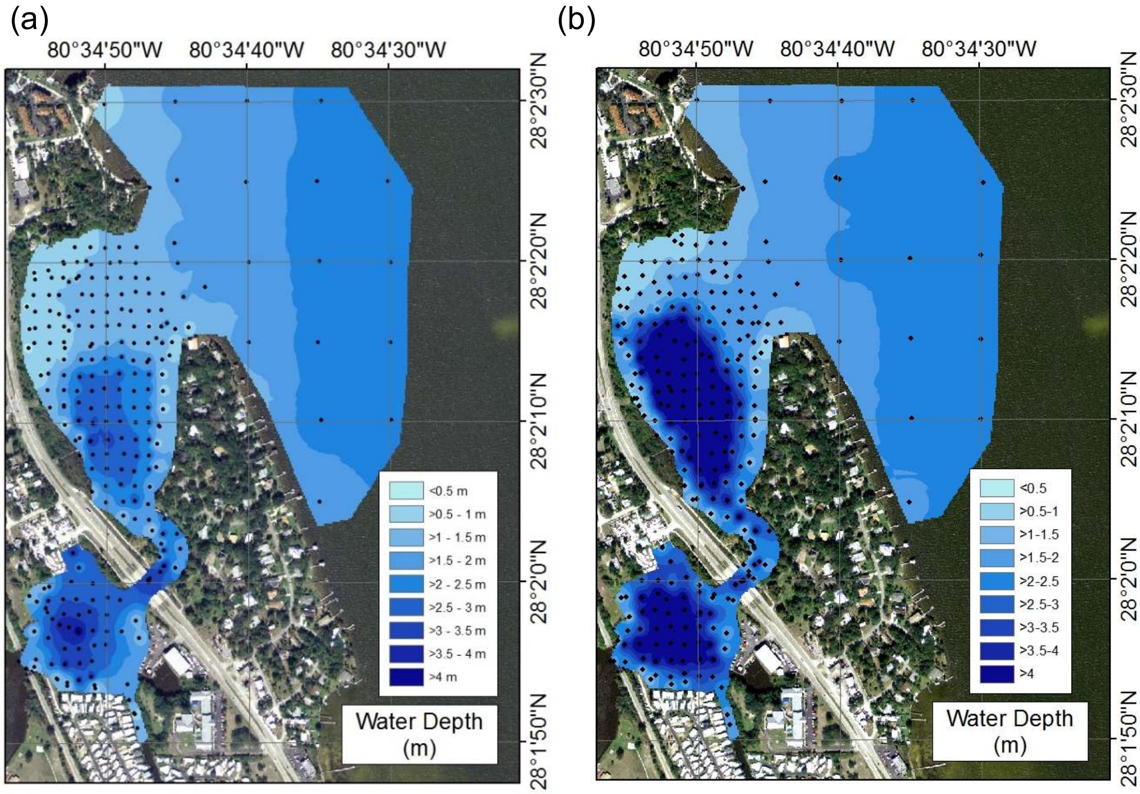


Figure 3.1.1. Contour maps of water depths in Turkey Creek from the Florida East Coast (FEC) Railroad Bridge to the adjacent Indian River Lagoon (IRL) during (a) February 2015, prior to dredging and (b) March 2017, ~2 months after dredging. Dots show probing locations.

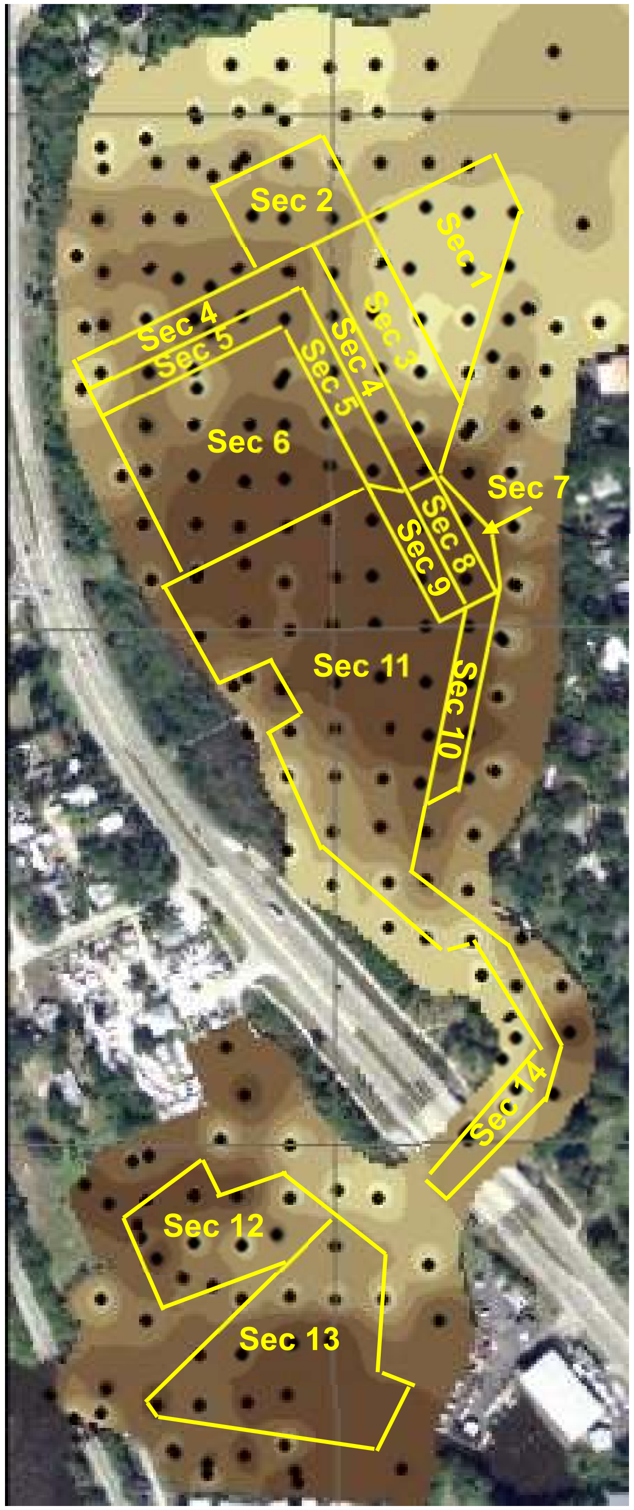


Figure 3.1.2. Contour map of muck thicknesses in Turkey Creek before dredging in February 2015 with overlying dredge cuts SEC-1 to SEC-13 (yellow lines). Image credit for dredge cuts: Brevard County Natural Resources Management Department.

Results from our pre- and post-dredging muck surveys showed that little or no muck was present in the adjacent IRL near the mouth of Turkey Creek, most likely because it was dispersed in the IRL (Figure 3.1.3a, b). In addition, little or no muck was found in shallow water (depth <1 m) in northern Palm Bay (Figure 3.1.3a, b). In contrast, muck layers as thick as 3 m were found in 2- to 4-m deep water in southern Palm Bay before dredging (Figure 3.1.3a).

We calculated a total volume of 107,000 m^3 (~140,000 yd^3) for all wet muck in the study area from the FEC Railroad Bridge to the IRL, including the area to be dredged (Table 3.1.1). Contractors for Brevard County estimated that 114,000 m^3 (149,000 yd^3) of muck were in the same area based on data from 13 vibracores (McGarry, 2015), a remarkably similar estimate. We calculated that ~24,000 m^3 of the total muck were initially present outside the area to be dredged (Table 3.1.1).

Dredging removed 52,000 m^3 (69,000 yd^3) of muck, or ~63% of the pre-dredge muck volume of 83,000 m^3 in the dredged area (Table 3.1.1). After dredging, muck thicknesses in the dredged area (as outlined on Figure 3.1.2) were >1 m in only 6 of 111 survey sites relative to 31 sites before dredging. Contour maps showing increases in water depths and decreases in the muck thicknesses (Figure 3.1.3c, d) corresponded relatively well with the dredge cuts (Figures 3.1.2).

Muck remaining in the dredged area was most likely due to (1) not dredging deep enough to remove all the muck and (2) slumping of muck onto newly exposed sand from adjacent areas not dredged, or not dredged deep enough to remove all the muck. At least 63% (71/111) of sites were dredged an average of 1 m below the muck layer (from our pre-dredge survey) and ~100,000 m^3 (131,000 yd^3) of material were dredged from below probe-penetrable muck deposits. Based on our data and a total of 160,000 m^3 of material reported to be dredged (McGarry 2017, personal communication) from Turkey Creek, ~32% of the dredged material was muck (probe penetrable) and 68% was lower water-content, sandier sediments dredged from below probe-penetrable muck deposits. About 30% of sites that were dredged below the bottom of the muck deposit retained a sandy bottom after dredging.

Table 3.1.1. Volume of muck in all of Turkey Creek (from the Florida East Coast Railroad Bridge to the mouth of the creek) and in the dredged and non-dredged area of the creek.

Location	Volume (m^3) Before	Volume (m^3) After	Volume (m^3) Removed	% Change
Turkey Creek (n = 195)	107,000	54,000	53,000	-50%
Dredged Area (n = 111)	83,000	31,000	52,000	-63%
Non-Dredged Area (n = 84)	24,000	23,000	1,000	-4%

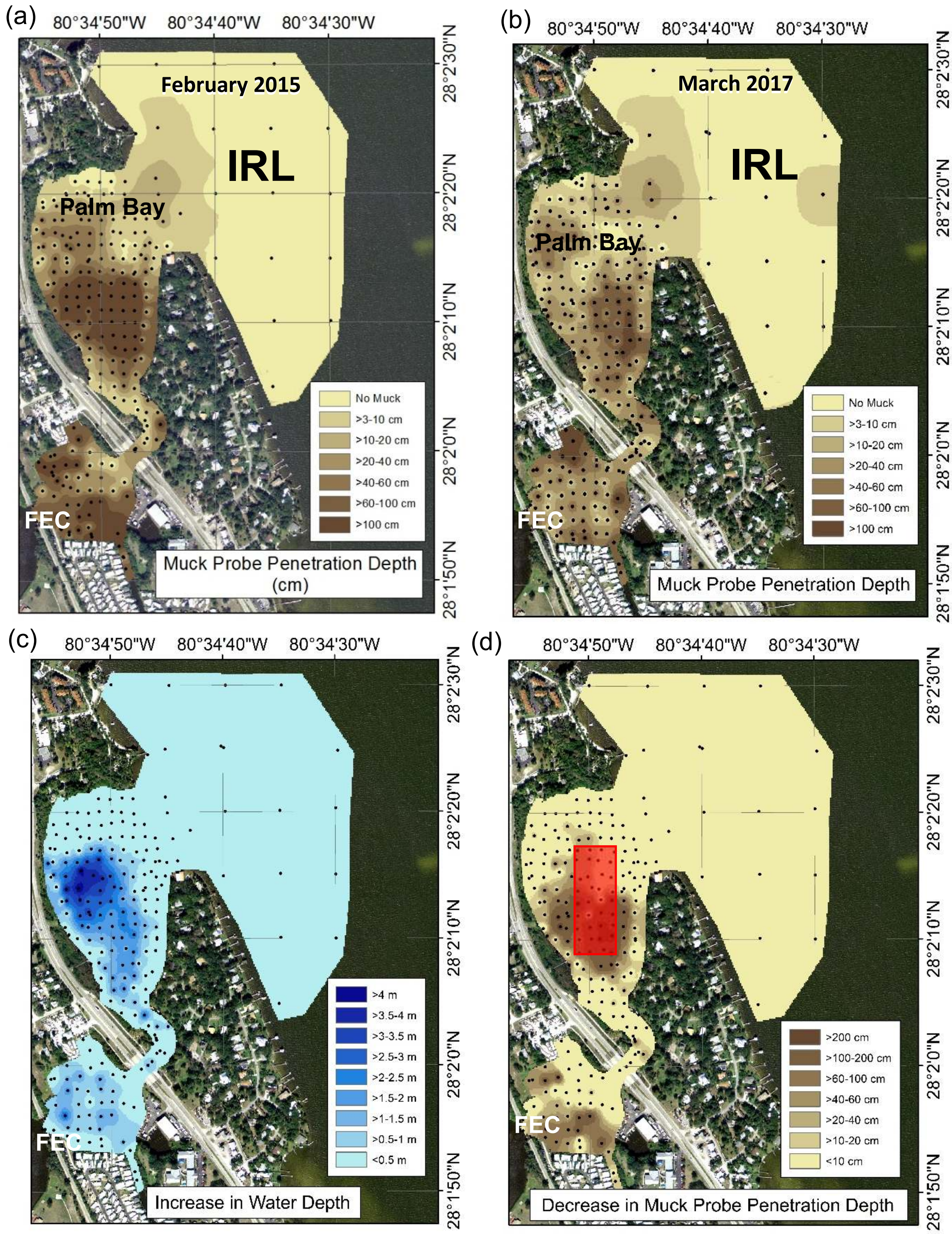


Figure 3.1.3. Contour maps of muck thicknesses in Turkey Creek from the Florida East Coast (FEC) Railroad Bridge to the adjacent Indian River Lagoon (IRL) (a) before dredging in February 2015 and (b) ~2 months after dredging in March 2017. Contour maps of (c) increases in water depths and (d) decreases in muck thicknesses (i.e., amount of muck removed) after dredging, Red rectangle shows area where subsequent dredging was carried out after Hurricane Irma (September 2107). Dots show probing locations.

About 15,000 m^3 of muck were added to a central part of the dredged area during 2017 due to slumping and the influences of Hurricane Irma in September 2017 (Figure 3.1.4). This area of ~40,000 m^2 was re-dredged between January 22 and March 1, 2018. Our muck survey in December 2018 showed that the re-dredging effort decreased muck in the area by ~21,000 m^3, ~50% less muck volume than after the original dredging.

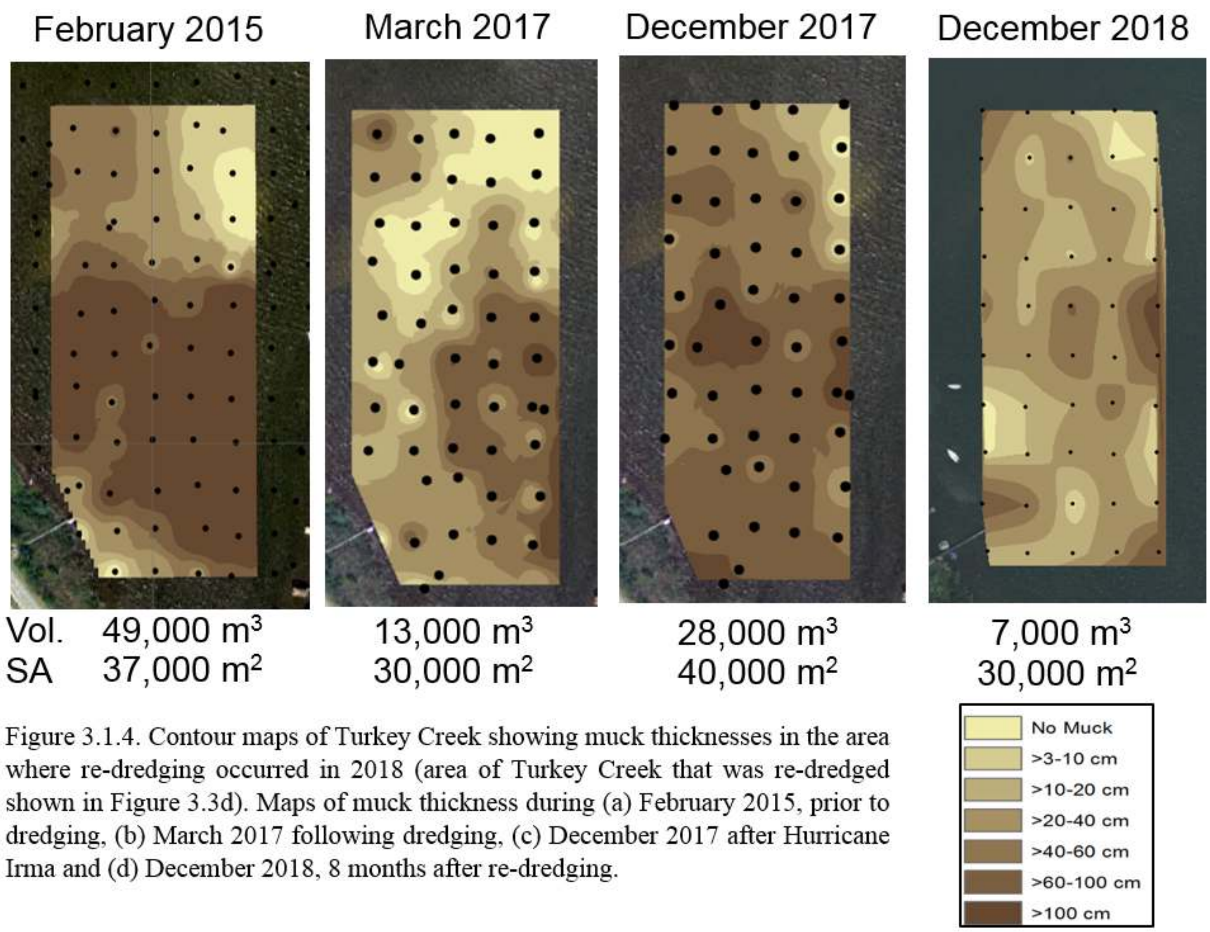


Figure 3.1.4. Contour maps of Turkey Creek showing muck thicknesses in the area where re-dredging occurred in 2018 (area of Turkey Creek that was re-dredged shown in Figure 3.3d). Maps of muck thickness during (a) February 2015, prior to dredging, (b) March 2017 following dredging, (c) December 2017 after Hurricane Irma and (d) December 2018, 8 months after re-dredging.

3.2. Tracking Changes in Sediment and Interstitial Water Composition

Sediment composition and OM content have been shown to be important factors in explaining the abundance and diversity of benthic biota (e.g., Stoker, 1981; Graf, 1989). In Turkey Creek, Beckett (2016) and Johnson and Shenker (2016) demonstrated that IRL muck is essentially uninhabitable, except to bacteria, most likely because of very high concentrations of H_2S. Therefore, it is important to track changes in sediment composition that accompany dredging.

The water content and chemical composition of sediments in Turkey Creek were highly variable as a function of the relative amounts of sand, silt and clay in the sample. For example, the water content of surface sediments in Turkey Creek ranged from ~46% (by volume) in sandy sediments to ~96% in muck. Sediment OM also varied widely from 1.6% in sandy sediments to 22% in muck. Using one definition of IRL muck (>75% water by weight or ~90% by volume, >60% silt + clay and >10% OM, Trefry and Trocine, 2011), 12 of the 24 samples collected in Turkey Creek before and after dredging fit the parameters listed above for IRL muck (Table 3.2.1), and this was regardless of whether samples were from within or without the dredging perimeter. No significant differences (i.e., p <0.05) were found in the composition of muck collected before versus after dredging, except for water content with 88.2% and 83.4% H_2O (by mass) before and after dredging, respectively (p = 0.040). The similarity in muck composition before and after dredging is reasonable because all muck samples were essentially the same Turkey Creek muck. The significant difference for water content may be due to removal of muck that exposed a more consolidated, deeper layer with lower water content.

Table 3.2.1 Averages ± standard deviations for parameters in muck (n = 12) and representative sediment from Turkey Creek. (LOI = Loss on Ignition at 550 °C, TOC = Total Organic Carbon)

	Gravel (%)	Sand (%)	Silt + Clay (%)	LOI (%)	$CaCO_3$ (%)	Al (%)
TC Muck	0.5 ± 0.9	13.9 ± 10.8	85.5 ± 10.6	18.9 ± 2.4	13.6 ± 3.9	4.0 ± 0.7
TC Sediment	0.6 ± 0.9	59.7 ± 38.0	39.8 ± 38.5	10.1 ± 8.2	6.6 ± 4.7	2.3 ± 1.7

	Fe (%)	Si (%)	TOC (%)	H_2O (wt. %)	H_2O (vol. %)	N (%)	P (%)
TC Muck	3.7 ± 0.7	18.6 ± 2.2	6.7 ± 0.8	84.5 ± 3.7	93.6 ± 1.6	0.70 ± 0.17	0.14 ± 0.02
TC Sediment	1.9 ± 1.4	29.0 ± 2.2	3.3 ± 2.9	61.1 ± 24.5	80.9 ± 46.7	0.36 ± 0.32	0.08 ± 0.06

Sediments collected from Turkey Creek showed a predictable range in composition due to the relative amounts of muck and sand. Strong correlations were found between (1) TOC and LOI, (2) total P and TOC, (3) total N and TOC and (4) total P and total N (Figure 3.2.1). When data from pre-, mid- and post-dredging surveys were plotted on these four plots, essentially all data fit the trends observed for the pre-dredge muck (Figure 3.2.1). These results confirm the previous statement that no significant differences in the chemical composition of unmixed muck were identified, before, during or after dredging.

Using average values for surface sediments from Turkey Creek (~80.9% water by volume and a density of dry sediments of 2.7 ton/m^3), the 160,000 m^3 of all wet sediment removed from Turkey Creek had a total dry mass of ~83,000 metric tons (160,000 m^3 x [1 – 0.809] x 2.7 tons/m^3). Based on the typical pre-dredging composition of average sediments from Turkey Creek (containing

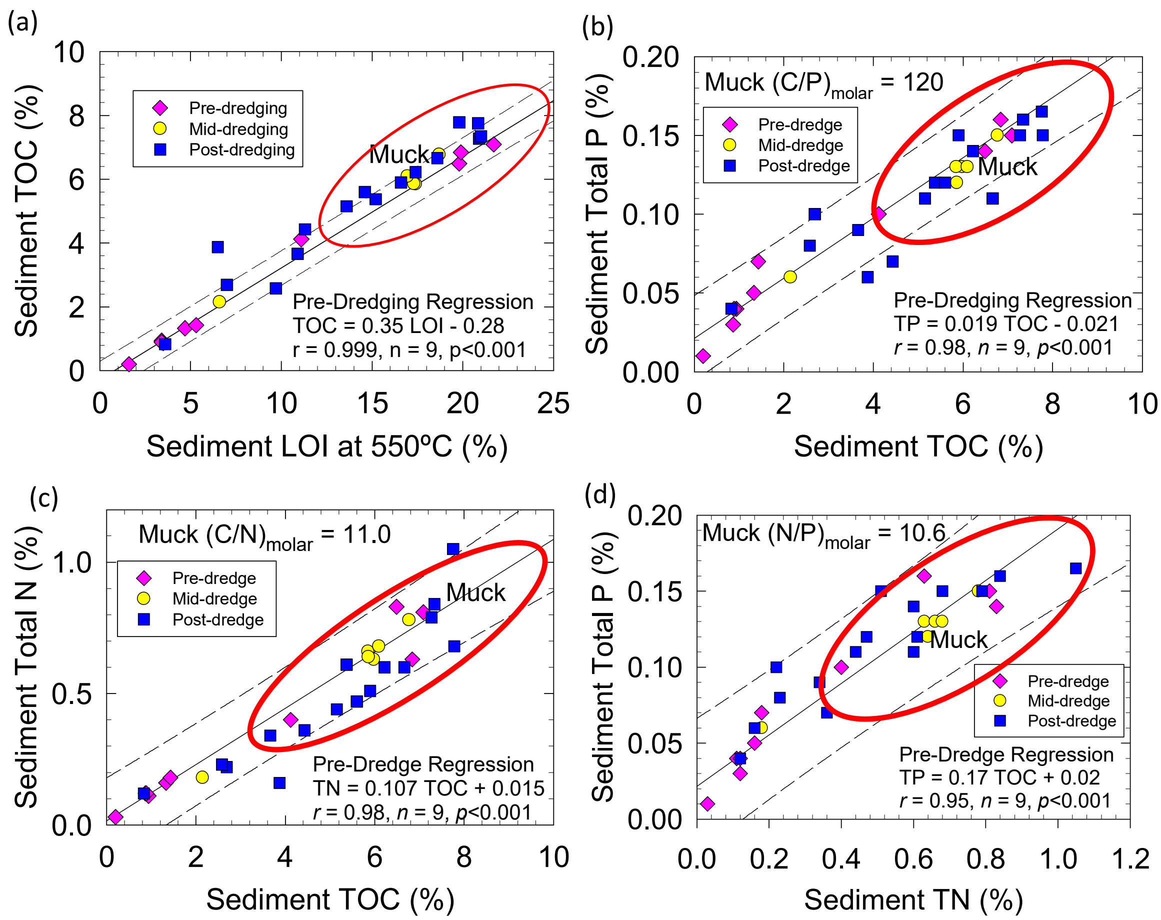


Figure 3.2.1. (a) Sediment total organic carbon (TOC) versus sediment Loss on Ignition (LOI) at 550°C, (b) sediment total phosphorus (TP) versus sediment TOC, (c) sediment total nitrogen (TN) versus TOC and (d) sediment TP versus TN. Muck samples plot within the ovals on (a–d). The molar ratio for the muck samples is listed next to each pertinent oval. Solid lines and equations on each graph are from re-dredge data linear regression, dashed lines are 95% prediction intervals, *r* is correlation coefficient and *p* is the *p* statistic.

muck, sand and sandy-muck) at 0.36 ± 0.32% N and 0.08 ± 0.06% P, the ~83,000 metric tons of sediment removed from Turkey Creek contained ~300 metric tons of N and ~70 metric tons of P.

Sediment cores collected at the same four locations before and after dredging showed changes in chemical composition as a function of whether (1) muck was dredged to sand, (2) muck was not completely dredged or (3) sand was dredged, but replaced by muck. Two cores (stations TC5 and TC6) contained muck (high water content) before dredging (Figure 3.2.2a, b). After dredging, both stations contained a thin layer (~2 cm) of muck, above ~5–10 cm of muck mixed with sand, and then sand (low water content) deeper in the core (Figure 3.2.2a, b). At station TC4, cores from before and after dredging contained muck because this site was only partially dredged (Figure 3.2.2c). The fourth site (TC3, original water depth = 1.1 m) contained sand before dredging; however, after dredging to a water depth of 3.8 m, the newly-formed basin accumulated ~30 cm

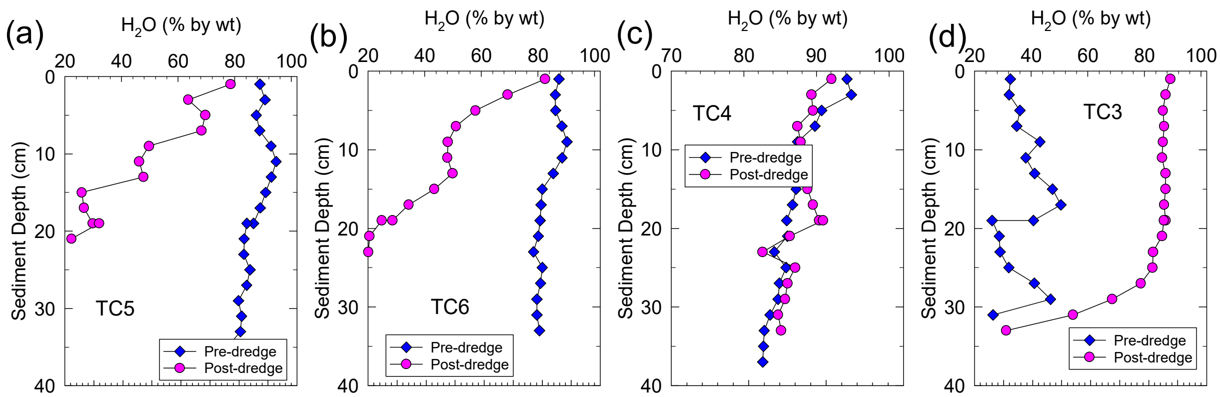


Figure 3.2.2 Vertical profiles for water content before and after dredging in sediment cores from stations (a) TC5, (b) TC6, (c) TC4 and (d) TC3. Locations on Figure 1b.

of muck (Figure 3.2.2d). Down-core trends in water content are controlled by pressure from sediment accumulation that promotes upward advection of water and a decrease in water content. This loss in water content is accompanied by essentially no changes in concentrations of Al, TOC and other sediment chemicals on a dry weight basis (Fox and Trefry, 2018a).

Decomposition of organic matter in muck leads to high concentrations of dissolved N and P in the abundant interstitial water of these organic-rich sediments. Higher values of dissolved N in sediment interstitial water, virtually all as ammonium, were found deeper than ~10 cm in each core (Figure 3.2.3a–d). Ammonium concentrations were lowest in surface sediments before and after dredging due to diffusion of ammonium to the overlying water (Figure 3.2.3a–d). Before dredging, concentrations of ammonium were very high in muck (relative to sand) with maximum values of ~2000–6000 µM (28–84 mg N/L; Figure 3.2.3a–d). Ammonium concentrations in sandy sediments were all <200 µM (2.8 mg N/L). After dredging, concentrations of ammonium followed patterns for water content with >80% lower values at sites TC5 and TC6 because dredging removed muck and left mostly sand with a surface layer of mixed sand and muck (Figure 3.2.3a, b). At station TC4, where muck was not directly dredged, ammonium values in interstitial water were ~50% lower after dredging in March 2017 than in October 2016 before dredging. This observation follows typical seasonal trends where large amounts of ammonium produced during the summer were still present in the interstitial water in October (Fox and Trefry, 2018a). Then, cooler winter temperatures decreased ammonium production from December to March when the cycle begins again. Highest ammonium values in interstitial water (>10,000 µM) were found in muck that slumped in over the original sandy bottom at station TC3 (Figure 3.2.3d).

Interstitial water phosphate distributions before and after dredging followed the same trends observed for ammonium, with much higher values in muck than sand or sandy muck (Figure 3.2.3). Thus, interstitial phosphate concentrations were much lower at stations 5 and 6 after dredging due to removal of muck, only slightly changed at station 4 and increased in the new muck at station TC3 (Figure 3.2.3d–f).

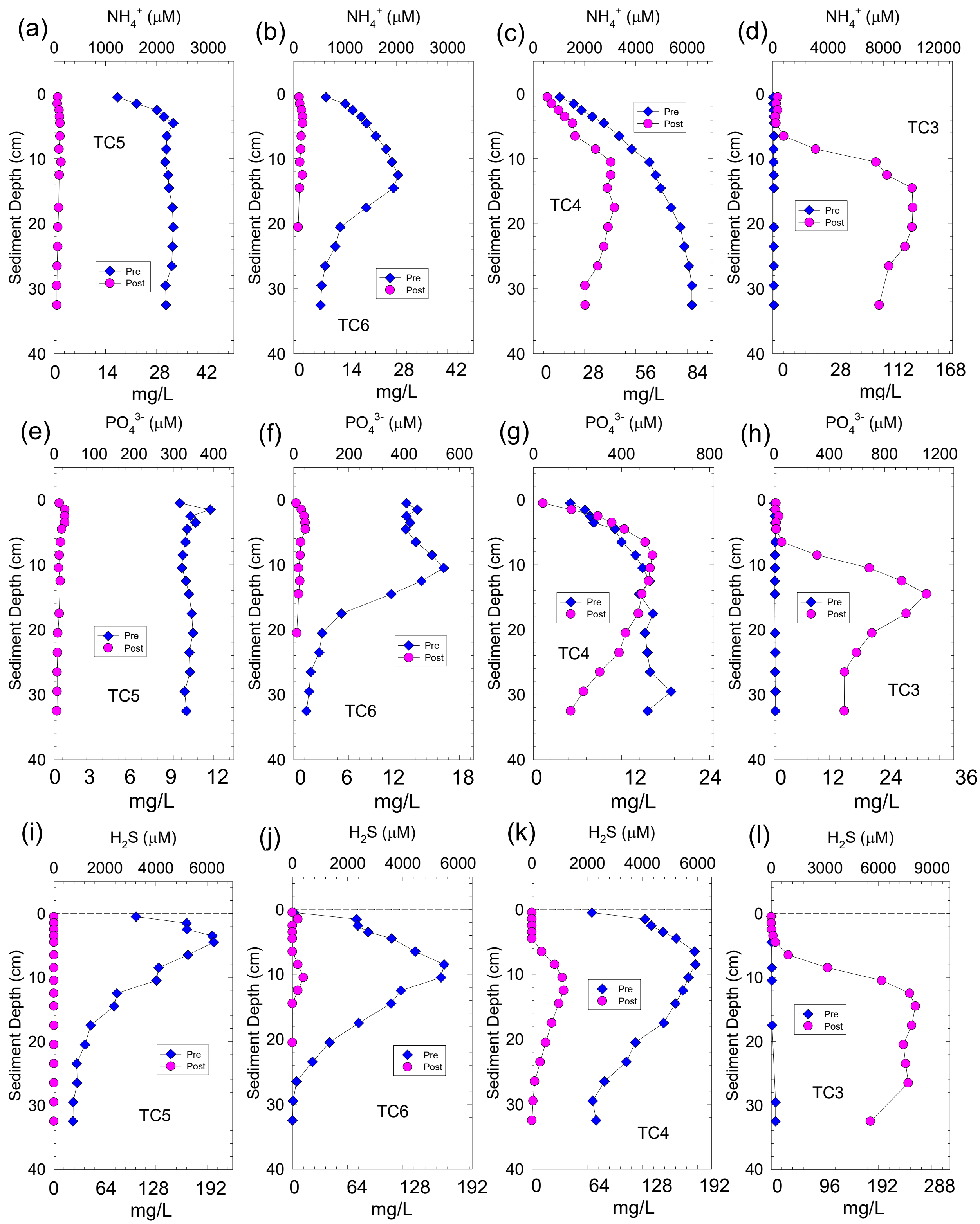


Figure 3.2.3. Vertical profiles for (a–d) ammonium (NH_4^+), (e–h) phosphate (PO_4^{3-})and (i–l) total hydrogen sulfide (H_2S) in interstitial water before (pre) and ~3 months after (post) dredging at stations TC5, TC6, TC4 and TC3.

Concentrations of sulfide in interstitial water also were lower after dredging at three of the four stations, with the fourth location being station TC3 where muck covered sand after dredging (Figure 3.2.3). Clearly, the sediment redox environment changed. For example, the Eh at station TC4 increased from -149 to -70 mV after dredging due to downward mixing of oxygenated lagoon water into sediments and cooler temperatures. Similar, but smaller increases in Eh were observed after dredging at stations TC5 and TC6. Oxygen was essentially absent from all muck cores. The toxicity of H_2S would be much lower at stations TC4, TC5 and TC6 after dredging, thereby increasing the likelihood of finding benthic biota. These observations are at least partially due to seasonal variations in the composition of interstitial water as described above for ammonium.

3.3 Tracking Changes in Benthic Fluxes of Nitrogen and Phosphorus

Pre-dredging fluxes (releases) of N and P from muck to the overlying water were determined using two different methods (Quick-Flux and interstitial water profiles). Average fluxes ranged from 0.1 mg N/m^2/hr (multiply by 8.8 to obtain tons N or P/km^2/yr) and 0.01 mg P/m^2/hr in sandy sediments at station TC3 to 8.6 mg N/m^2/hr and 1.7 mg P/m^2/hr in muck at station TC6 (Table 3.3.1). Fluxes of ammonium and phosphate (TC4, TC5 and TC6) before dredging were relatively uniform among stations containing muck with mean values of 6.9 ± 2.0 mg N/m^2/hr and 1.3 ± 1.1 mg P/m^2/hr. When these values are applied to the entire 0.12 km^2 surface area of muck in Turkey Creek, the pre-dredging fluxes of N and P for the study area in Turkey Creek were 0.8 kg N/hr (7 tons N/year) and 0.2 kg P/hr (1.4 tons P/year), respectively.

Post-dredging, fluxes of N and P from muck averaged 2.3 ± 1.5 mg N/m^2/hr and 0.3 ± 0.3 mg P/m^2/hr, respectively (Table 3.3.1). Following dredging, Quick-Flux was used to enhance spatial resolution for flux data with 28 sites throughout Turkey Creek, 19 contained muck. Fluxes of N and P varied spatially with the highest values (>3 mg N/m^2/hr) in southern Palm Bay and high values (>1.5 mg-N/m^2/hr) in pockets of muck that remained after dredging throughout the study area. Spatial variations in fluxes were best explained by sediment water content (Figure 3.2.3). Overall, lower fluxes of N and P after dredging result in calculated load reductions of ~3 tons of N and ~1 ton of P annually. Lower calculated fluxes of N and P after dredging were mostly due to lower concentrations of ammonium and phosphorus in interstitial water (Figure 3.2.3) because a <20% difference in the surface area of muck occurred from dredging. Releases of N and P from muck that remained in Turkey Creek are more likely to vary with time and temperature than to vary in the composition of sediment (e.g., from muck to sand).

Table 3.3.1. Pre- and post-dredging fluxes of N and P determined from interstitial water profiles and the Quick-Flux technique along with supporting information.

Pre-Dredge				Flux ($mg/m^2/hr$)[1]			
				Ammonium-N		Phosphate-P	
Station	Sediment Type	Date	Temp. (°C)	Interstitial Water	Quick-Flux	Interstitial Water	Quick-Flux
TC3	Sand	Oct 2015	25.0	0.1 ± 0.1	0.1 ± 0.0	0.01 ± 0.02	0.01
TC4	Muck	Oct 2015	25.0	8.6 ± 6.3	5.6 ± 0.0	1.0 ± 0.9	0.77
TC5	Muck	Feb 2016	16.4	8.6 ± 7.8	ND[2]	1.4 ± 1.8	ND[2]
TC6	Muck	Feb 2016	16.4	5.0 ± 3.5	ND[2]	1.7 ± 2.4	ND[2]

Post-Dredge				Flux ($mg/m^2/hr$)[1]			
				Ammonium-N		Phosphate-P	
Station	Sediment Type	Date	Temp. (°C)	Interstitial Water	Quick-Flux	Interstitial Water	Quick-Flux
TC3	Muck	Mar 2017	23.2	1.5 ± 2.5	3.1 ± 1.6	0.09 ± 0.06	0.03
TC4	Muck	Mar 2017	23.3	3.8 ± 3.2	4.4 ± 0.9	0.84 ± 0.12	0.63 ± 0.03
TC5	Sandy-Muck	Apr 2017	25.0	0.4 ± 0.4	1.1 ± 0.3	0.09 ± 0.03	0.08 ± 0.0
TC6	Sandy-Muck	Apr 2017	25.1	0.4 ± 0.4	0.7 ± 0.04	0.06 ± 0.00	0.06 ± 0.0

[1]$Tons/km^2/yr = 8.8 \times mg/m^2/hr$. [2]Not determined.

The convenience of the Quick-Flux technique (Fox and Trefry, 2018a) enabled us to initiate a two-year time series for benthic fluxes of N and P at stations TC3 and TC5 in September 2016. Station TC3 was dredged during March 2016 and subsequently accumulated a thin layer of muck that spurred fluxes of 3.6 mg $N/m^2/hr$ and 1.1 mg $P/m^2/hr$ in September 2016; these values are >15-times higher than found in the sandy sediments at this site before dredging. Soon after the first sampling for the time series, Hurricane Matthew moved through the area and resuspended muck sediments and caused large-scale erosion in the IRL. Post-hurricane nutrient fluxes at station TC3 peaked at >50 mg $N/m^2/hr$ and >3.5 mg $P/m^2/hr$ in October 2016 (Figure 3.3.1). After this storm-induced peak, fluxes decreased over the winter to 2.9 mg $N/m^2/hr$ and 0.02 mg $P/m^2/hr$ in April 2017 due to cooler temperatures. Station TC5 was not dredged before Hurricane Matthew, yet fluxes increased following the hurricane and peaked at >5.5 mg $N/m^2/hr$ and >0.4 mg $P/m^2/hr$, respectively (Figure 3.3.1b). Fluxes also decreased over winter at station TC5.

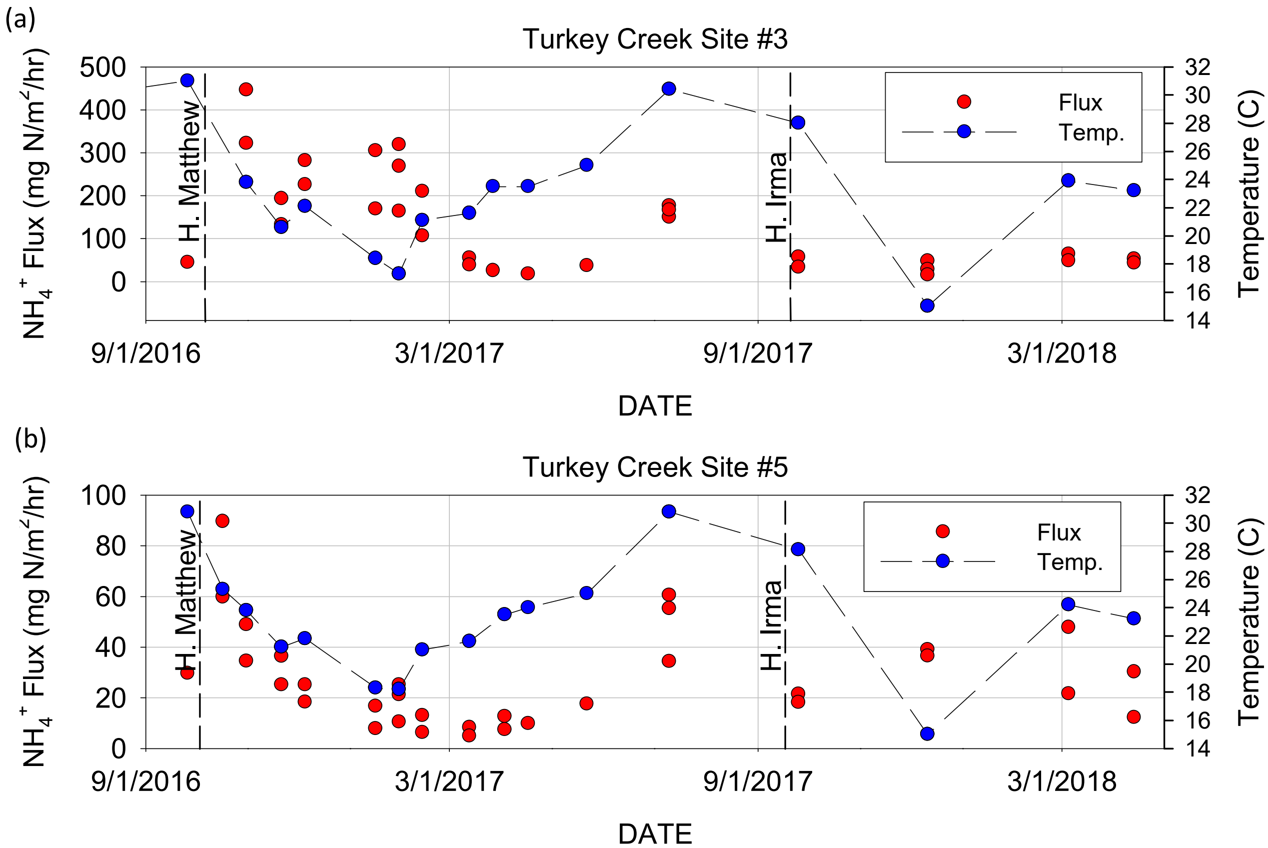


Figure 3.3.1. Temperatures and fluxes of ammonium-N versus time (September 2016–April 2018) at station (a) TC3 and (b) TC5 in Palm Bay, Florida.

3.4. Water Quality and Environmental Dredging

Dredging in Turkey Creek increased the mean water depth from 1.9 to 2.9 m and expanded the water volume of the creek from 300,000 to 460,000 m^3. This basic alteration in bathymetry of the lower creek had the following positive impacts on water quality in Palm Bay, the basin adjacent to the IRL: (1) increased and buffered the salinity of Palm Bay, (2) increased the total inventory of oxygen and buffered concentrations of dissolved oxygen in Palm Bay, and (3) created a sediment trap to capture future sediment inputs from upstream and thereby mitigate future benthic fluxes of N and P in the IRL.

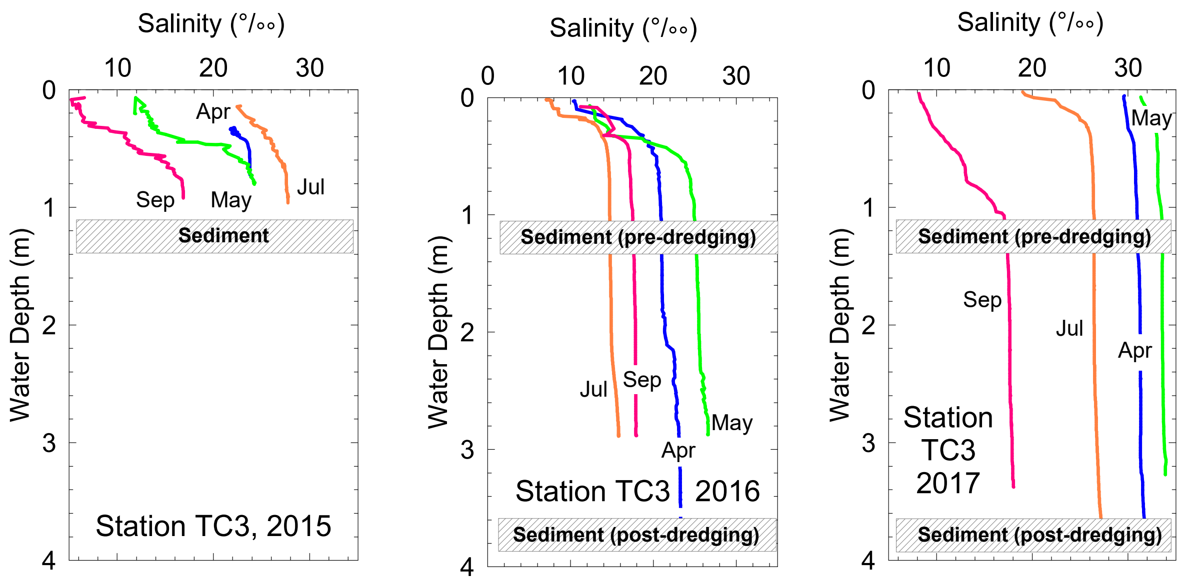


Figure 3.4.1. Vertical profiles for salinity at station Turkey Creek 3 (TC3) for selected months (a) before dredging in 2015 and after dredging in (b) 2016 and (c) 2017.

The main impact of dredging on salinity occurred when dredging increased water depths and allowed more saline water to enter Palm Bay. A striking example is shown for station TC3 where water depth increased from 1.1 to ~3.6 m to create a sizeable saline water mass (Figure 3.4.1). Salt water can act as a buffer against the impacts of freshwater on fishes, benthic infauna and seagrass.

Concentrations and the percent saturation of DO followed patterns for water temperature with no discernable difference in DO before, during, or after dredging for station TC1 outside the immediate dredged area. At stations where dredging was carried out (e.g., stations TC3 and TC5), DO in surface water increased after dredging regardless of the pre- and post-dredging bottom type, likely related to the increase in water depth. Garcia (1998) reported similar observations after dredging in Crane Creek. As predicted, DO was lower in bottom water at stations TC3 and TC5 after dredging due to a combination of oxygen uptake by residual muck and limited circulation and flushing of water at depths below ~2 m. In an assessment of post-dredge Turkey Creek in 1998, a 2 to 3-month WQ improvement was observed, but longer term, monthly monitoring by the SJRWMD IRL program showed that, within a year, conditions had reverted back to pre-dredge conditions (Joel Steward, *personnel communication*).

Even though oxygen concentrations were lower in bottom water after dredging, the low oxygen conditions replaced layers that were previously anoxic sediments. The net result was an increase in the total integrated amount of dissolved oxygen. For example, the integrated amount of oxygenated water at station TC3 increased from 9.6 g/m^2 to 22 g/m^2 due to the transition from 2–3 m of sediments with no DO in sediment interstitial water to >2 m of oxygenated water (Figure 3.4.2). This increase in total oxygen provides an added resilience to oxygen depletion events;

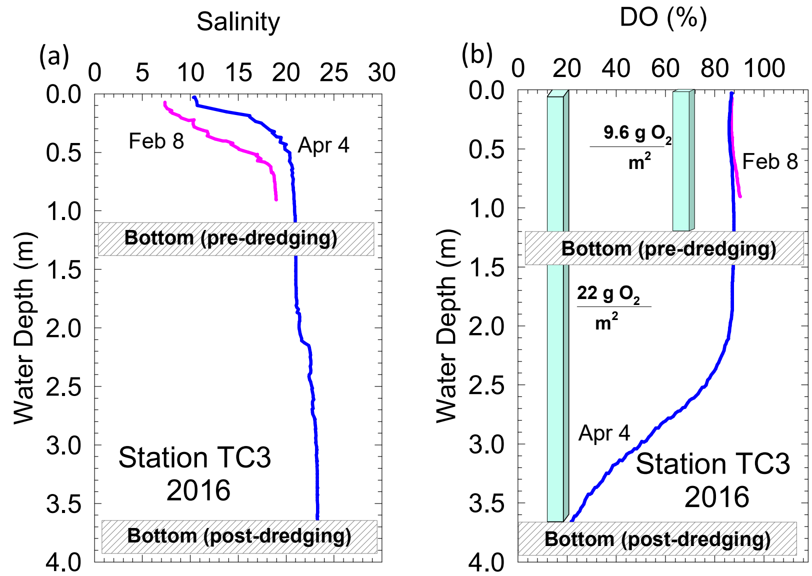


Figure 3.4.2. Integrated concentrations of dissolved oxygen (DO) before (9.6 g/m^2) and after dredging (22 g/m^2).

however, a semi-permanent hypoxic zone at the bottom of the water column may inhibit or slow recovery of benthic fauna within the dredged area.

Vertical profiles for ammonium and phosphate in the water column followed predicted trends with lower concentrations in surface water due to uptake during primary production and higher values in bottom water due to the benthic flux from sediments, especially above muck deposits (Figure 3.4.3). No discernible differences in dissolved nutrient profiles during dredging could be identified due to large variations in nutrient concentrations, an algal bloom or disturbances from Hurricane Matthew (Figure 3.4.3). Nevertheless, a strong signal for muck as a source of nutrients was identified throughout this study. For example, concentrations of dissolved ammonium in bottom water at station TC3 followed patterns for water temperature with highest concentrations of ammonium when temperatures were >30ºC (summer and early fall) and lowest concentrations during cooler months in winter and spring (Figure 3.4.3). Concentrations of N and P in bottom water reached maximum values during October 2016 (Figure 3.4.3) in response to sediment disturbance from Hurricane Matthew as previously discussed. At locations where water depths increased and muck slumped in over sand, such as at station TC3, the flux of ammonium and phosphate from the sediments increased and concentrations of these two nutrients in the bottom water increased (Figure 3.4.3). In contrast, where dredging removed muck from the bottom, fluxes of ammonium and phosphate greatly decreased. One long-term goal of dredging was to minimize benthic nutrient fluxes and decrease nutrient concentrations in the water column.

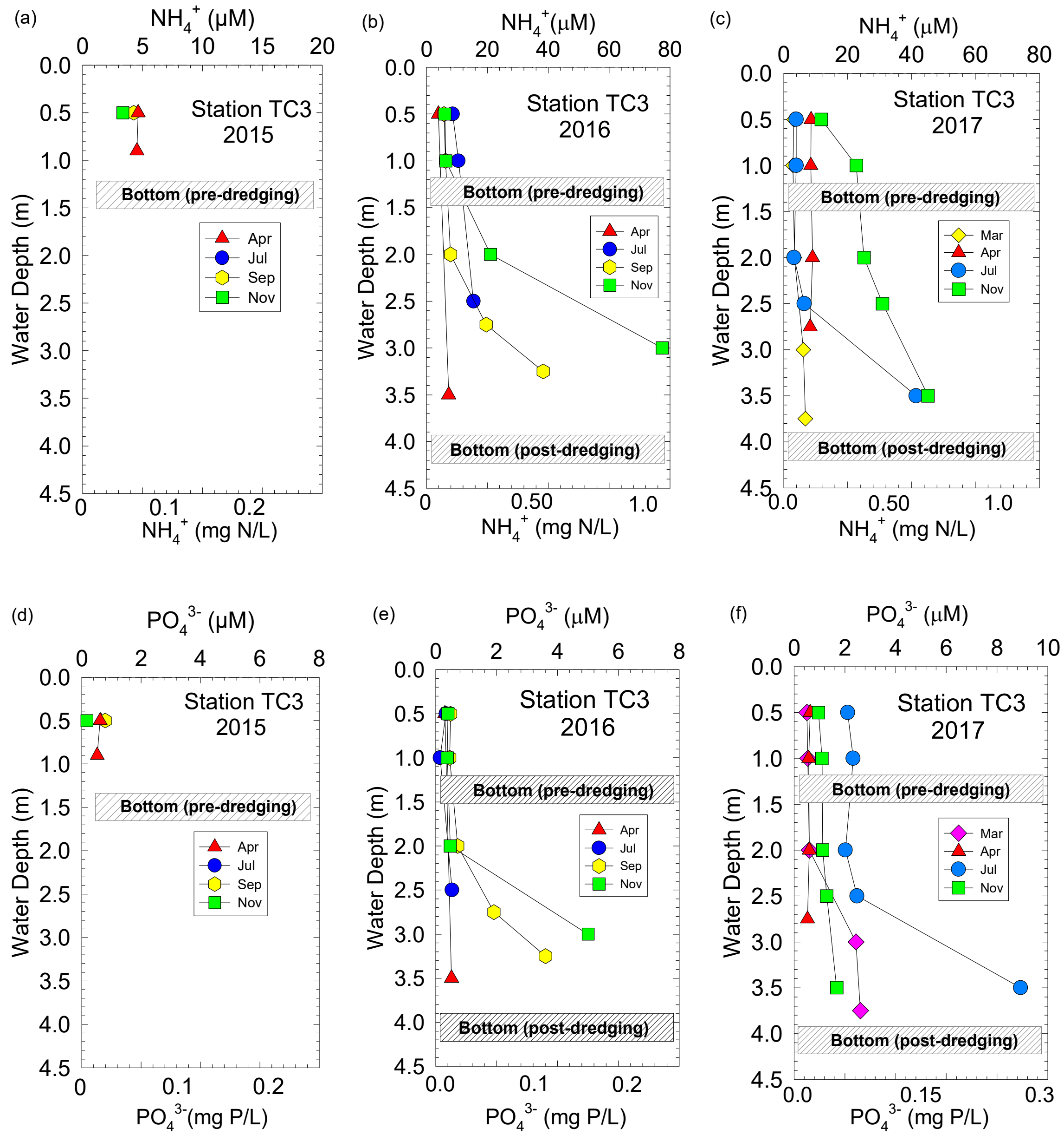


Figure 3.4.3. Selected vertical profiles at station TC3 for concentrations of (a, b, c) ammonium at station TC3 and (d, e, f) phosphate at station TC3.

3.5 *Seagrasses and Drift Algae*

In locations where seagrasses were found, water depths along seagrass transects were variable as a function of wind conditions and water levels. Nearshore transects were at water depths of 40–70 cm within 20 m of shore, increasing to depths of 95–135 cm at 50 m from shore, and then maintained a water depth of ~100 cm out to the 100-m quadrat.

Seagrasses were not abundant enough for random sampling in Palm Bay, nor did they occur at muck stations (Figure 2.3.1). Seagrass data were compared from 2015–2018 for spring and summer (Figures 3.5.1–3.5.3). Where seagrasses occurred, they had less visual cover and canopy length in the Indian River Lagoon near Turkey Creek (TCL) during Spring and Summer 2018 (Figures 3.5.1 and 3.5.2). This pattern of less seagrass in 2018 also was observed in the Indian River Lagoon near Crane Creek (CCL) and therefore should not be attributed to nearby dredging activity or post-dredging effects. Water quality issues in this region of the IRL have not yet been fully addressed by a multi-faceted strategy that includes muck removal, hundreds of projects in muck source control and other remedial measures.

In 2018, mean visual % cover ranged from 0–11% in the inshore portions of transects, and decreased to zero in deeper, offshore portion of transects (Figure 3.5.1). Mean canopy heights for Spring and Summer 2018 ranged from 0–6.8 cm (Figure 3.5.2). Shoot counts were low in 2018 through all seasons at both TCL and CCL and lower than in preceding years (Figure 3.5.3). Seagrasses were less abundant in the slightly deeper water found towards the ends of transects. There was a consistent, year-round absence of seagrasses within Turkey Creek proper (TC) and, therefore, data for TC are not shown. It should be noted that struggling, sparse patches of *H. wrightii* were observed in TC during the high growth season (late spring/early summer); however, they were not sampled via transect sampling. The TC stations may prove valuable in the future for continued monitoring of seagrasses as a lagoon water quality indicator because a sparse seed bank, if present, and vegetative propagation could foster more rapid increase if conditions improve.

Canopy height, which is equivalent to blade length, was extremely variable and statistically indistinguishable from zero; furthermore, no statistically significant patterns were observed. Non-significant visual trends include greater abundance and height of seagrass nearshore and during the high growth season.

In the year following the cessation of dredging, seagrasses showed no signal attributable to dredging. It may be that, because of the relatively slow colonization and growth of seagrasses, responses to seagrasses will only be detectable over a longer period of time, or that improvements due to dredging, depending upon the areal extent of muck removal, could be very site-specific and marginal or more regional and significant. In the latter case, we may expect to see general seagrass improvements, including those in the IRL near Turkey and Crane Creeks, as Brevard County continues to dredge in accordance with the Save Our Indian River Lagoon Project Plan ("SOIRLPP", Tetra Tech and Close Waters, 2016).

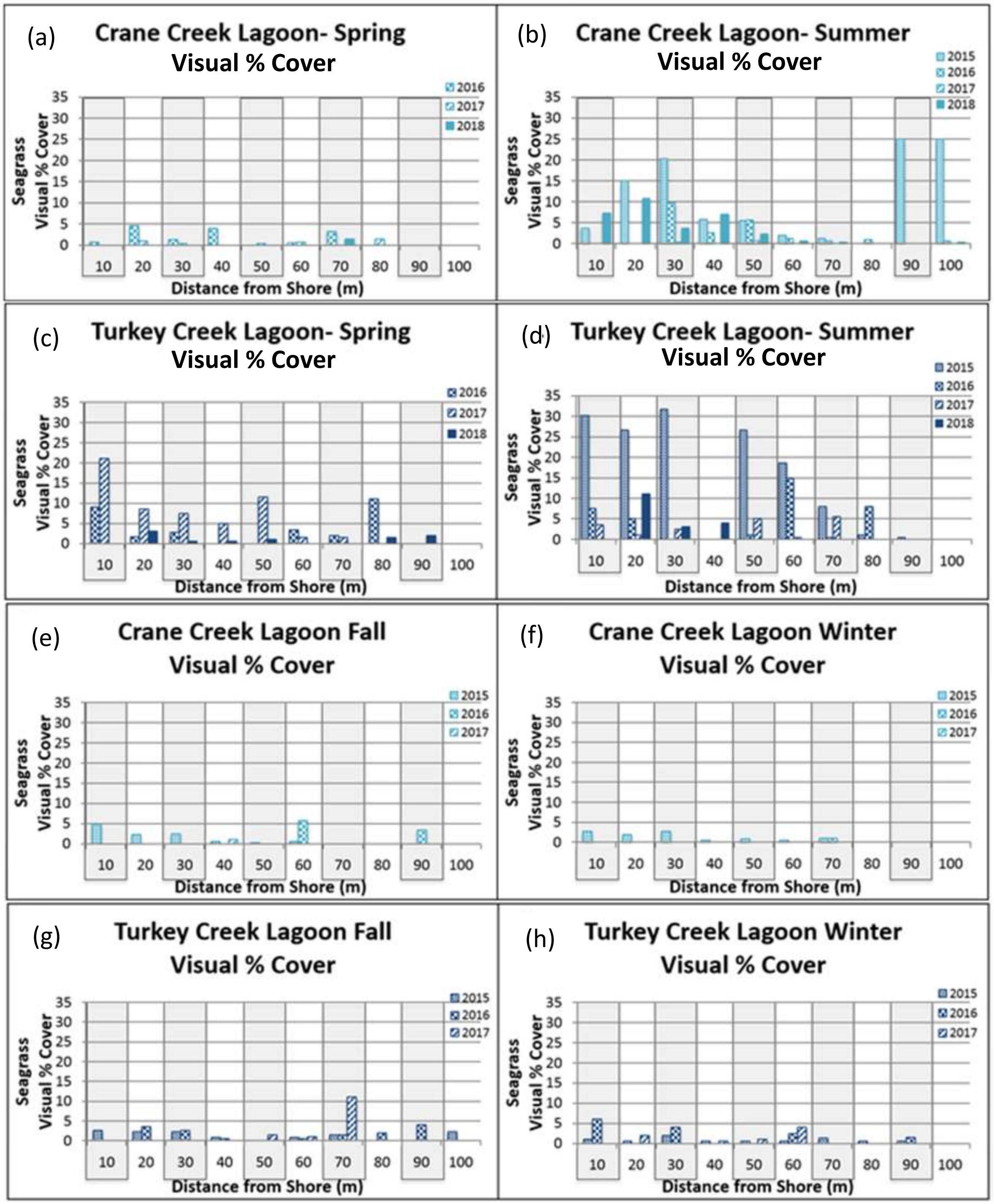


Figure 3.5.1. Seasonal means (2015–2018) for visual % cover for seagrass at Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Dredging was carried out from February 2016 to January 2017, with a hiatus from May–August 2016.

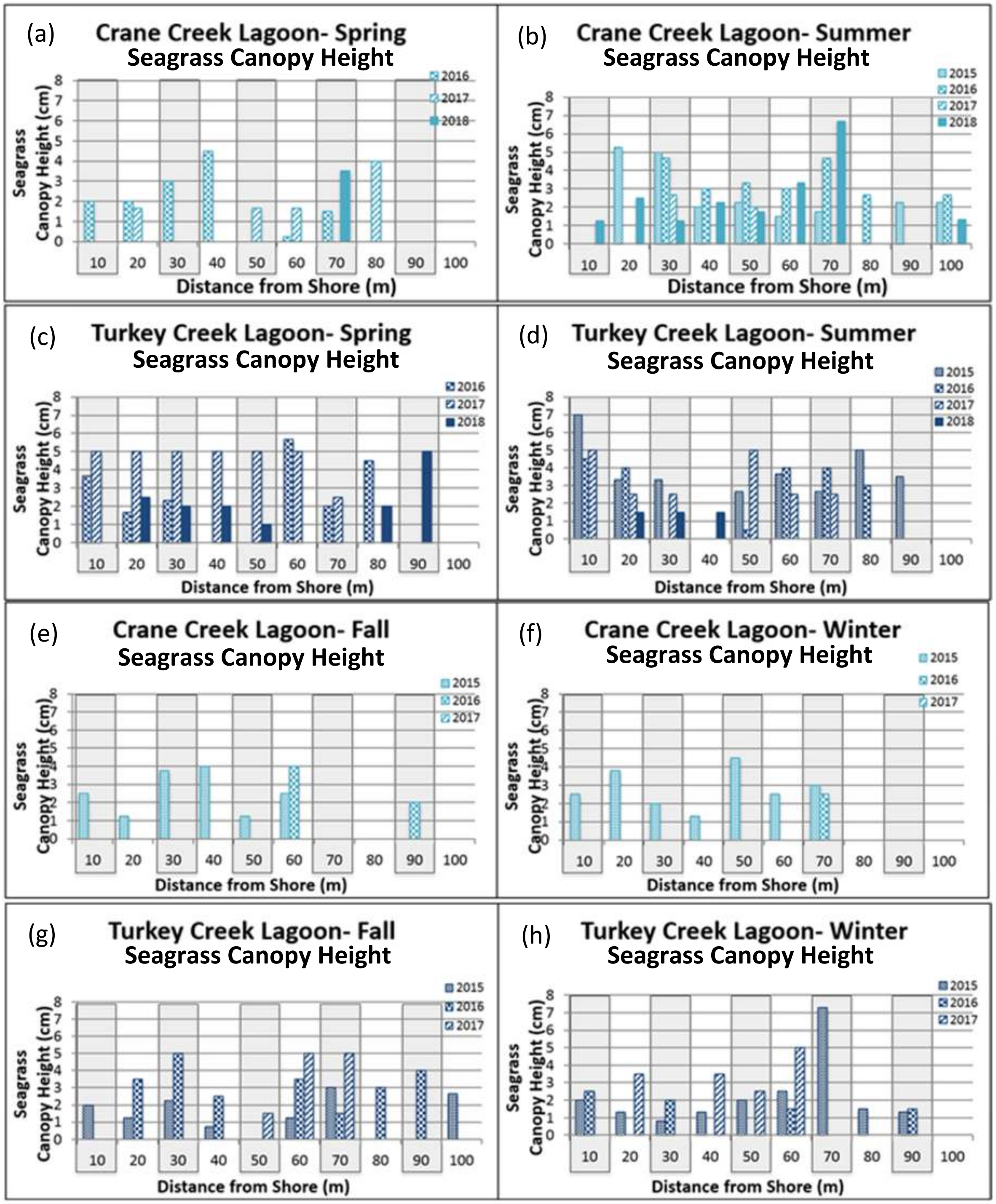


Figure 3.5.2. Seasonal means (2015–2018) for seagrass canopy heights at Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Dredging was carried out from February 2016 to January 2017, with a hiatus from May–August 2016.

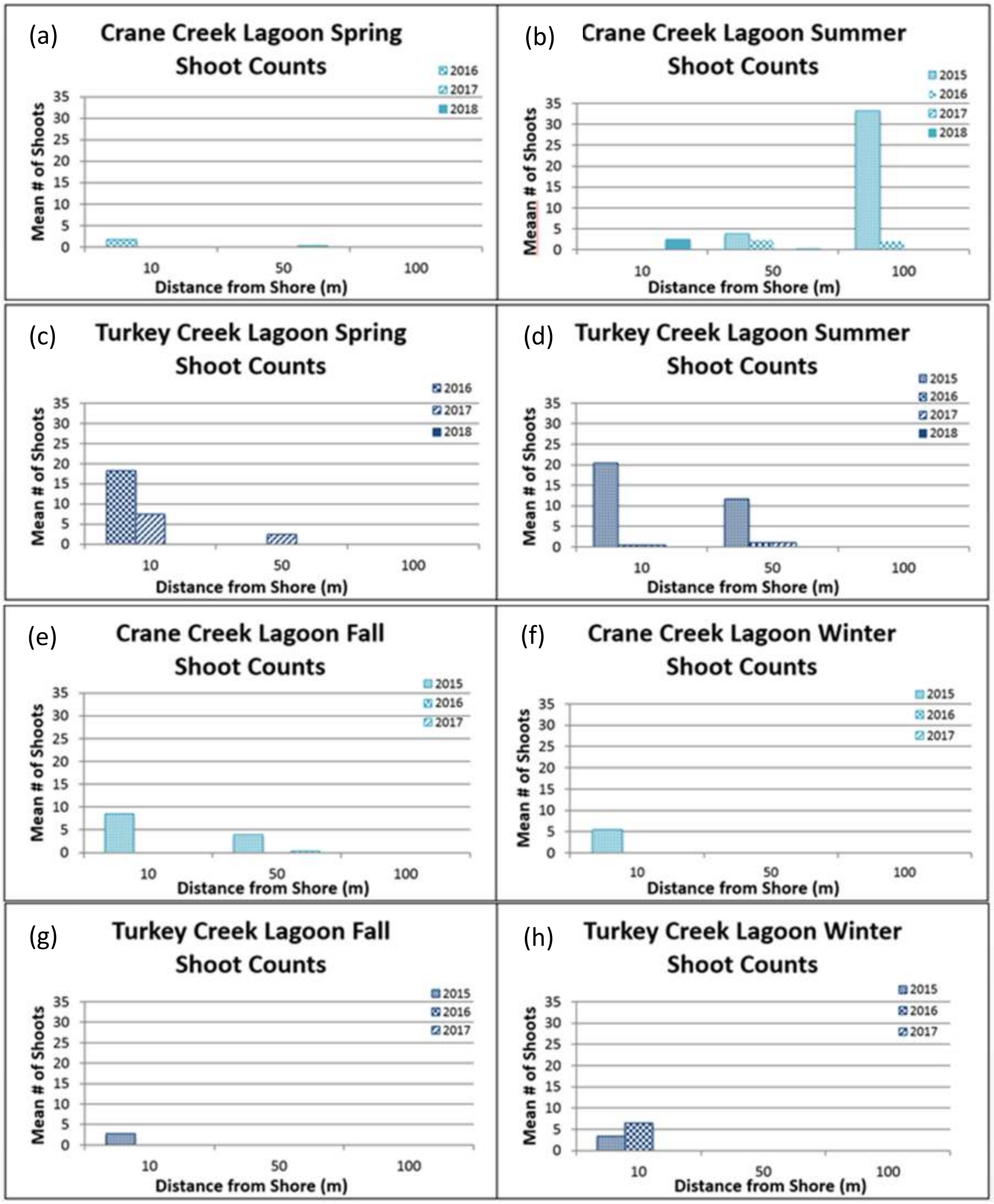


Figure 3.5.3. Seasonal means (2015–2018) for seagrass shoot counts per quadrat at Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Shoot counts were determined at the 10, 50 and 100 meter quadrats. Dredging was carried out from February 2016 to January 2017, with a hiatus from May–August 2016.

Epiphytes on seagrass blades compete with seagrasses for nutrients and light and can be detrimental to seagrass growth if their blade coverage is excessive. In some cases, epiphytes can be used as an indicator of seagrass health or water quality (Broderson et al., 2015). Epiphytic growths on seagrasses were scored on a qualitative scale of 1–5, with 1 being minimal growth and 5 being severe. Data are presented for transects in the summer, fall and winter in Figures 3.5.4, 3.5.5 and 3.5.6, respectively. The greatest abundance of epiphytes was in the summer, where they were more often present and reached mean qualitative scores of 2.0 (±0.2). During this study, epiphyte growth was greatest in the Summers of 2015 and 2016 (Figure 3.5.4).

Drift algae observations are given for Turkey Creek and associated sites in Figures 3.5.7-3.5.12. When present, drift algae tended to accumulate around Turkey Creek more than Crane Creek. In times of highest abundance (e.g., 2017), there was usually more drift algae inside Palm Bay (TC) compared to outside the mouth (TCL), possibly a result of prevailing easterly winds.

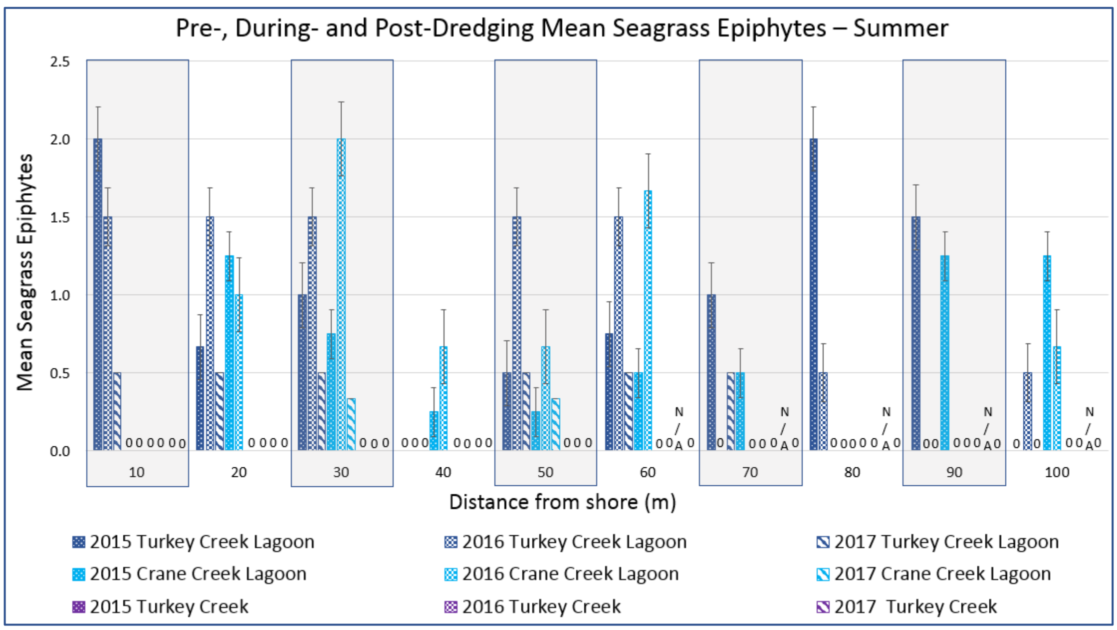


Figure 3.5.4. Mean epiphyte score on seagrass blades for Summer (July 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate no epiphytes on blades; 'N/A' means there are no seagrass blades to score. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

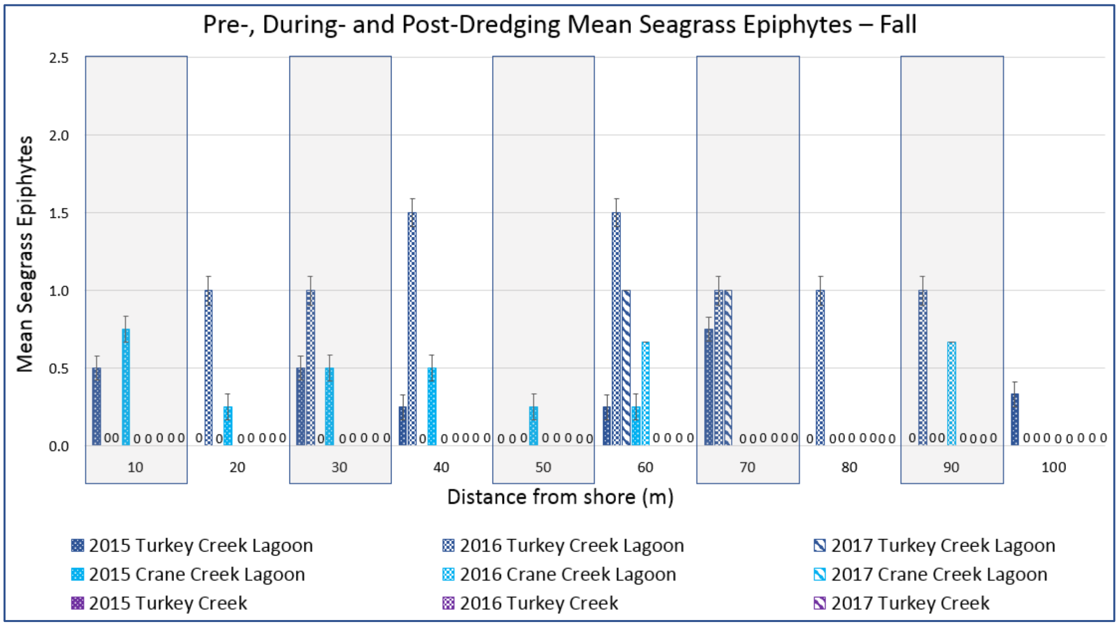


Figure 3.5.5. Mean epiphyte score on seagrass blades during Fall (October 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

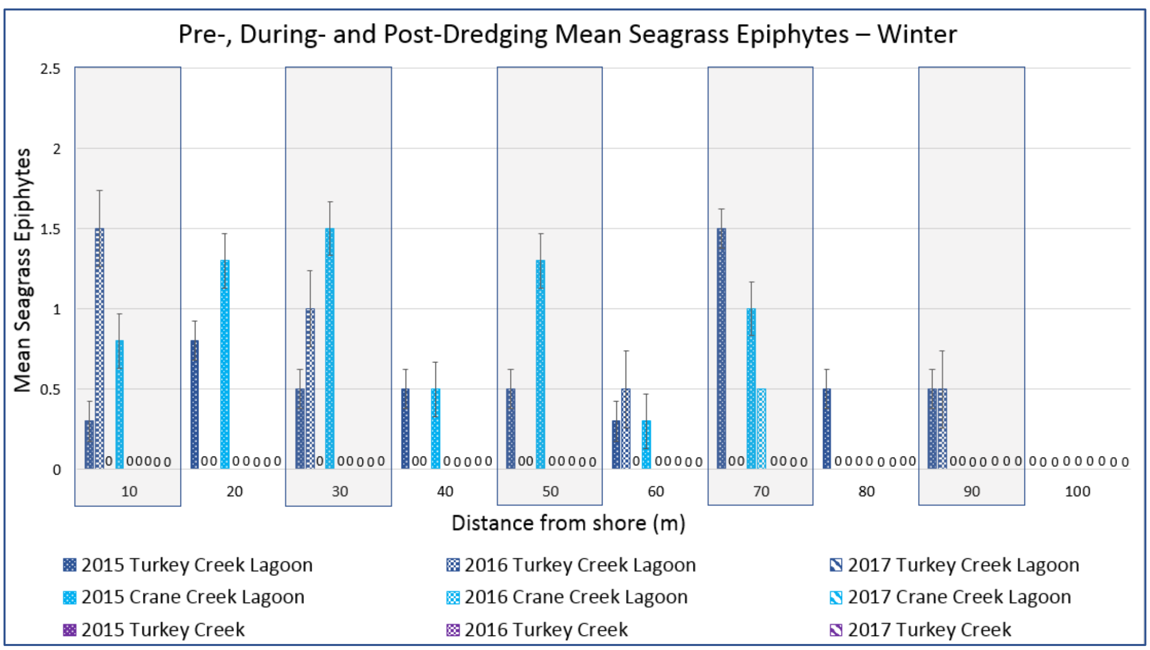


Figure 3.5.6. Mean epiphyte score on seagrass blades for Winter (December 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

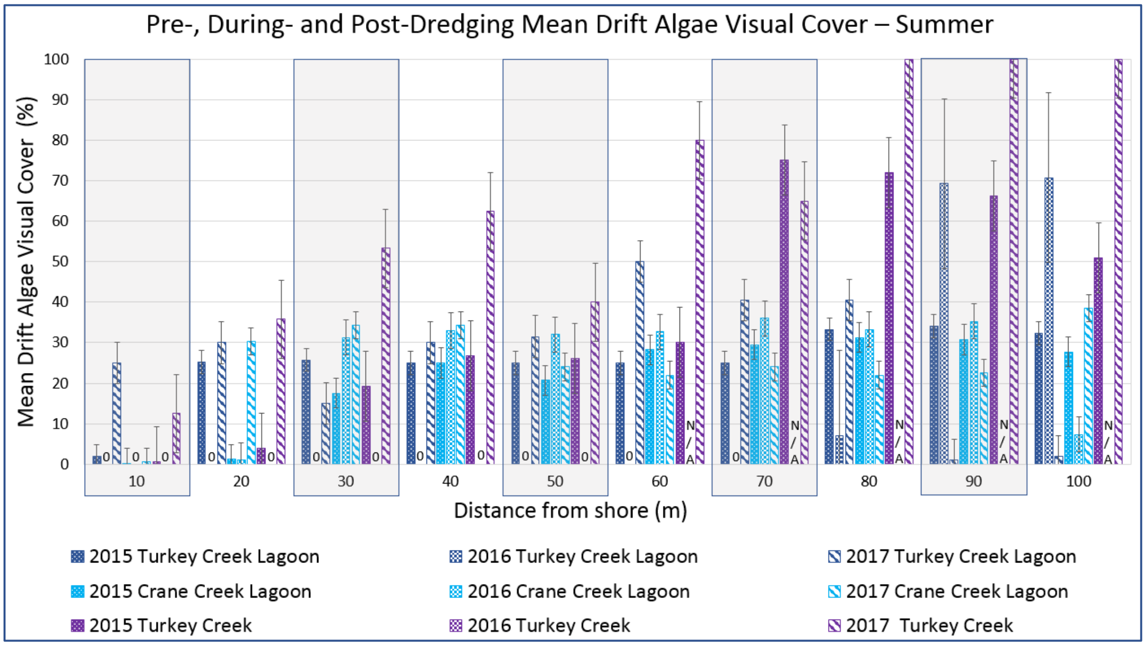


Figure 3.5.7. Mean drift algae visual % cover during the Summer (July 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance; 'N/A' means data missing or not collected. Dredging in TC occurred from February 2016–January 2017, with a hiatus from May-August.

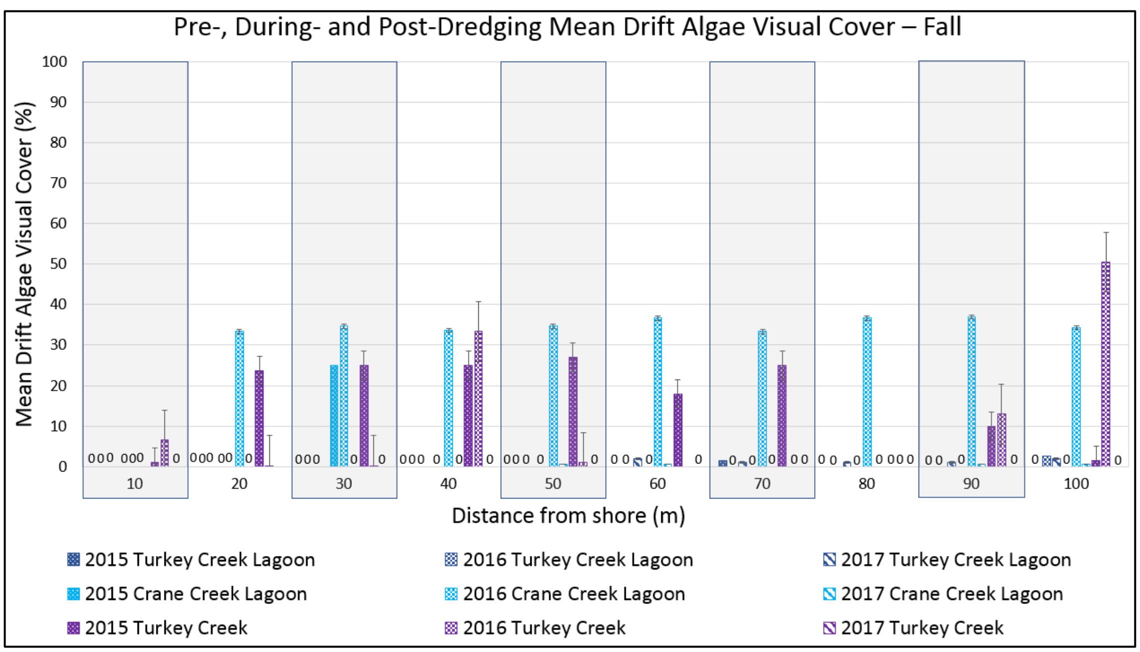


Figure 3.5.8. Mean drift algae visual % cover during the Fall (October 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

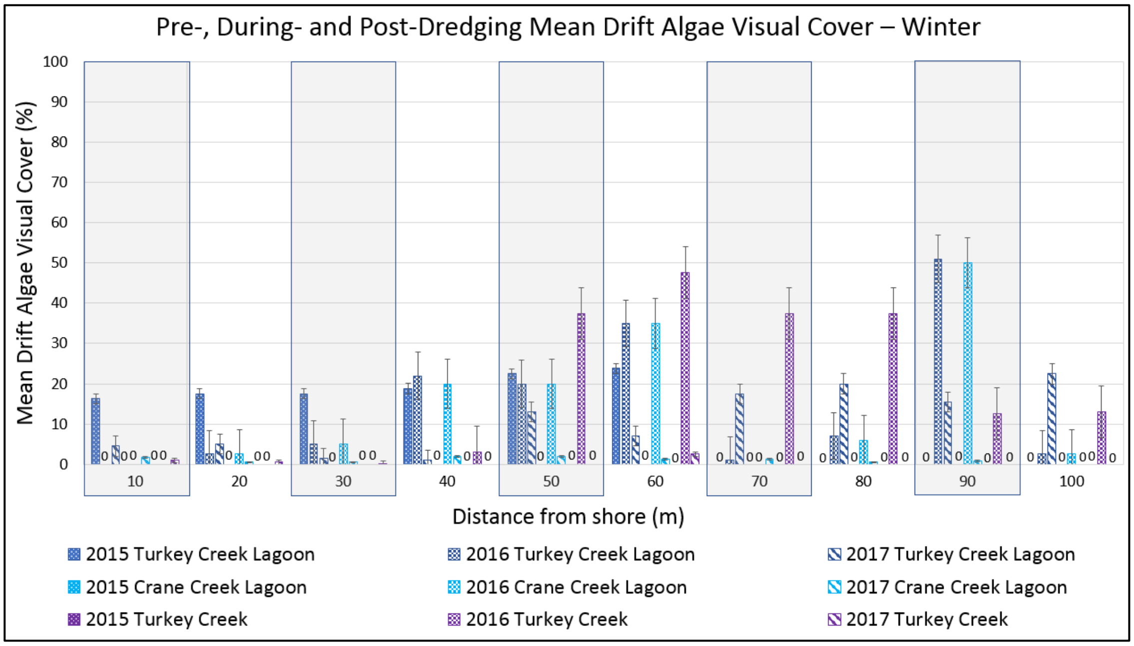


Figure 3.5.9. Mean drift algae visual % cover during the Winter (December 2015, 2016, 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

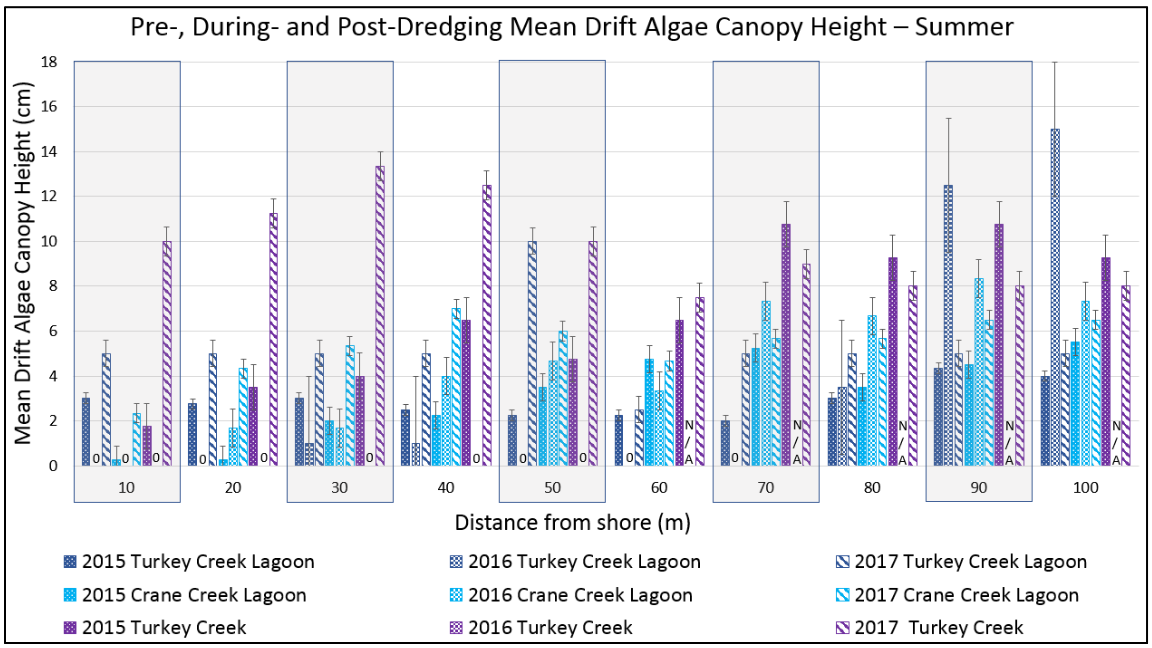


Figure 3.5.10. Mean drift algae canopy height during the Summer (July 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

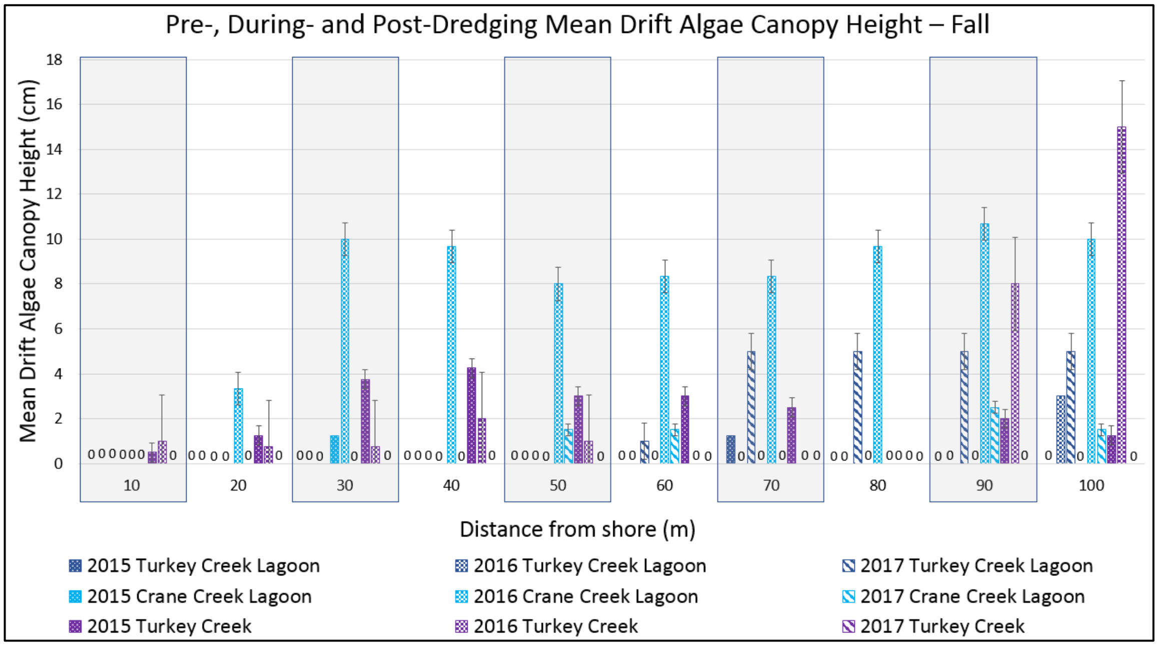


Figure 3.5.11. Mean drift algae canopy height during the Fall (October 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

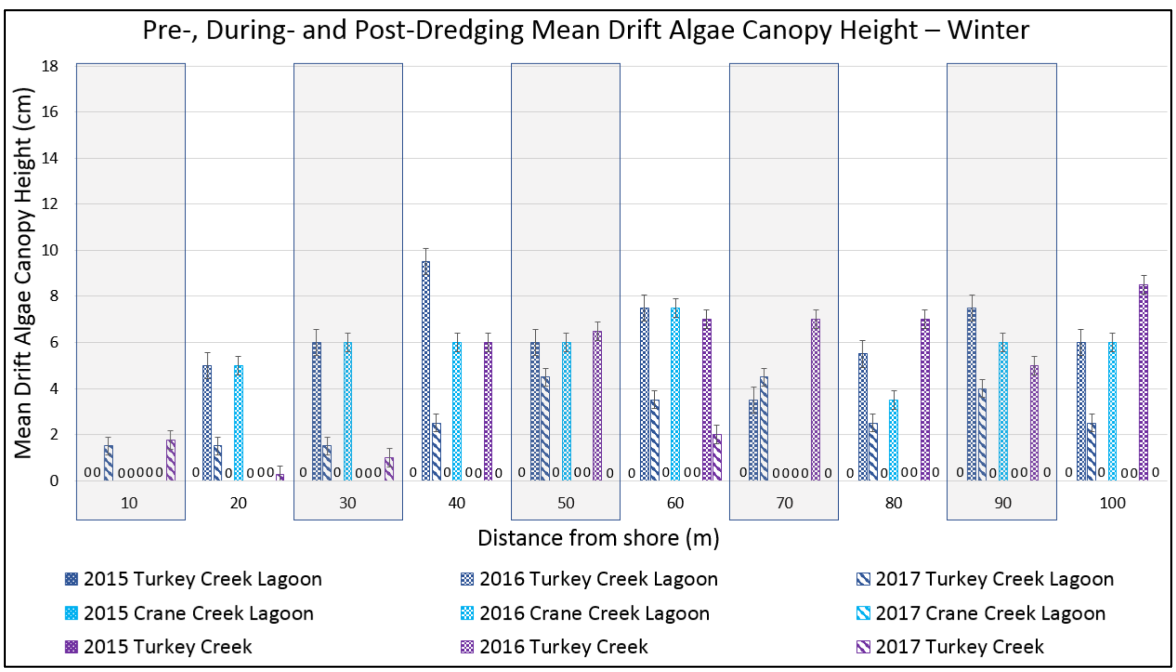


Figure 3.5.12. Mean drift algae canopy height during the Winter (December 2015, 2016 and 2017) at Turkey Creek (TC), Turkey Creek Lagoon (TCL) and Crane Creek Lagoon (CCL) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate zero mean and variance. Dredging in TC occurred from February 2016 to January 2017, with a hiatus from May-August 2016.

3.6 Sediments and Infauna

Sediments at Turkey Creek and associated stations have been monitored for 3 years (2015–2017), including dredging during 2016. Thresholds for IRL muck from Trefry and Trocine (2011) are as follows: water content >75% by weight, organic matter content >10% by dry weight and silt/clay content >60% by dry weight. Prior to dredging, sediments at muck stations (TCM, CCM) exceeded all muck parameter thresholds (Figures 3.6.1–3.6.3). The TC stations had intermediate scores for all muck indicators whereas the lagoon sites (TCL and CCL) had the lowest scores (Figures 3.6.1–3.6.3). In the absence of dredging, muck parameters tend to stay fairly consistent from year to year. After dredging Turkey Creek, all parameters dropped below the muck thresholds at the muck stations (TCM, Figures 3.6.1–3.6.3). The muck, lagoon, and intermediate stations had statistically-distinct muck characteristics; these muck characteristics dropped below published thresholds following dredging (Figures 3.6.1-3.6.3). The gradient in OM content of the sediment, or possibly correlated sediment parameters, are hypothesized to drive species diversity and abundances. Muck sites (TCM, CCM) have statistically higher water content relative to other stations. Turkey Creek (TC) has a statistically distinct intermediate water content relative to other stations (Figure 3.6.1).

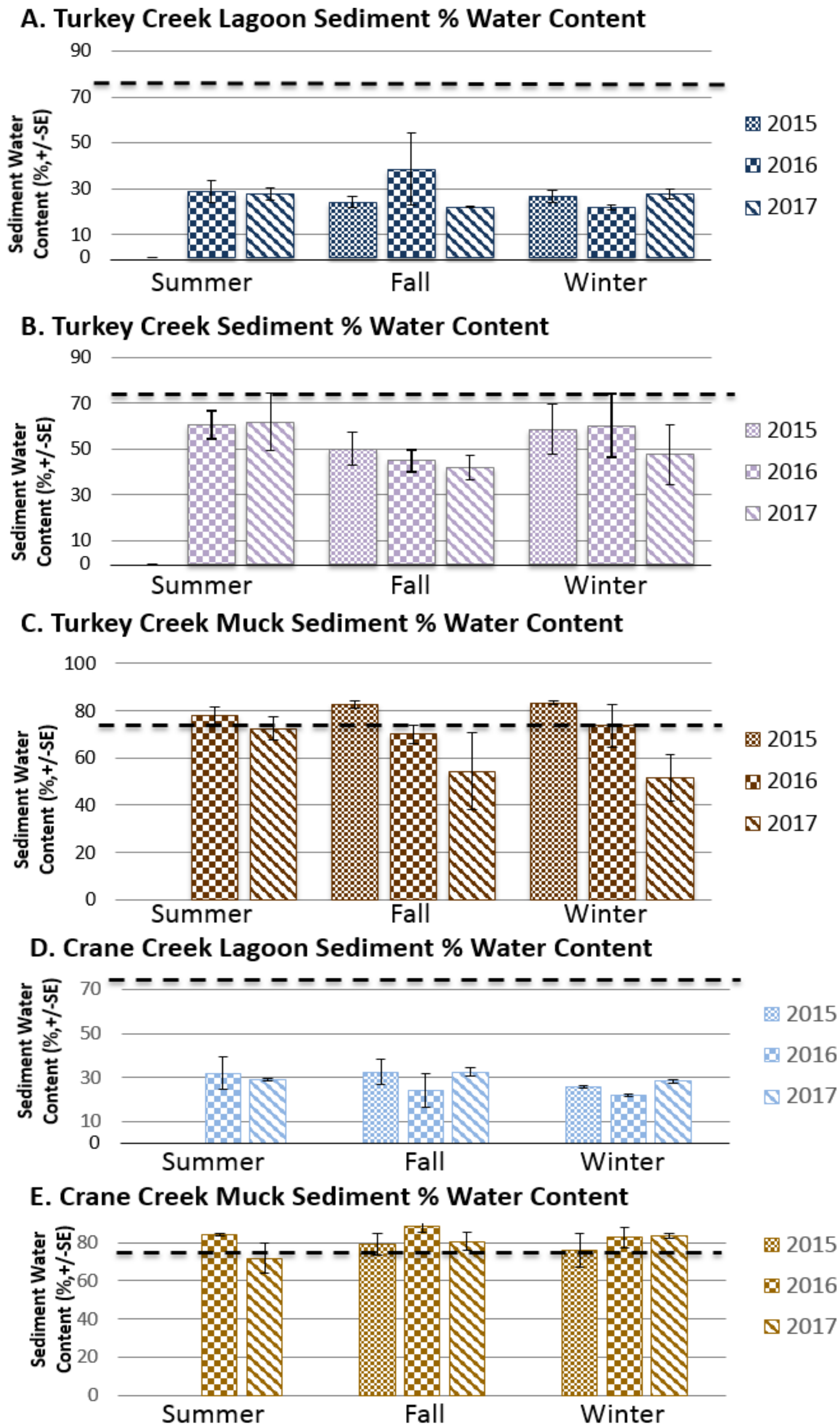


Figure 3.6.1. Mean sediment % water content at (a) Turkey Creek Lagoon (TCL), (b) Turkey Creek (TC), (c) Turkey Creek Muck, (d) Crane Creek Lagoon and (e) Crane Creek Muck. Error bars are ±1SE. Horizontal dashed lines indicate defined muck water content thresholds (Trefry and Trocine 2011).

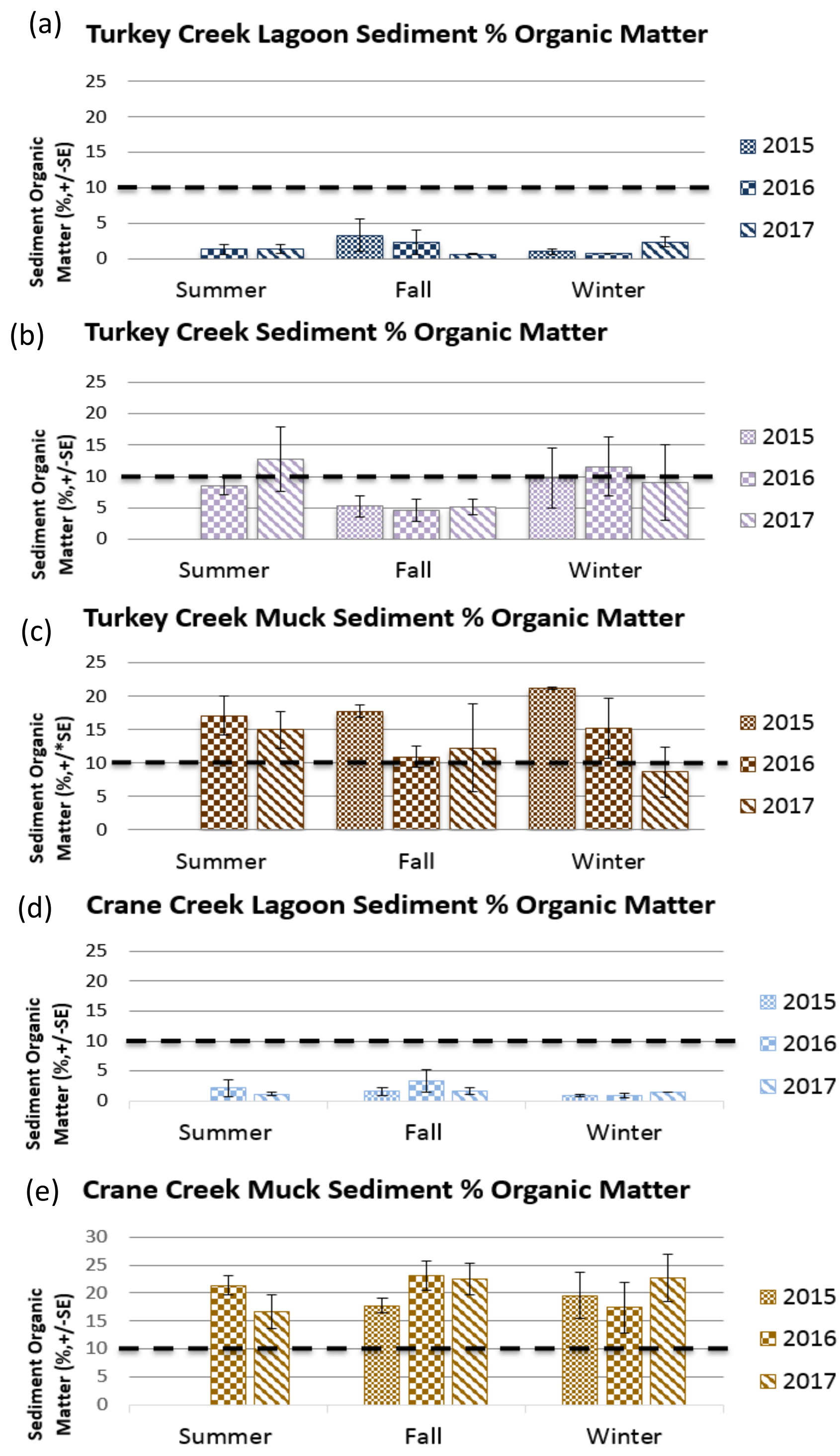


Figure 3.6.2. Mean sediment % organic matter at (a) Turkey Creek Lagoon (TCL), (b) Turkey Creek (TC), (c) Turkey Creek Muck, (d) Crane Creek Lagoon and (e) Crane Creek Muck. Error bars are ±1SE. Horizontal dashed lines indicate defined muck organic matter thresholds (Trefry and Trocine 2011).

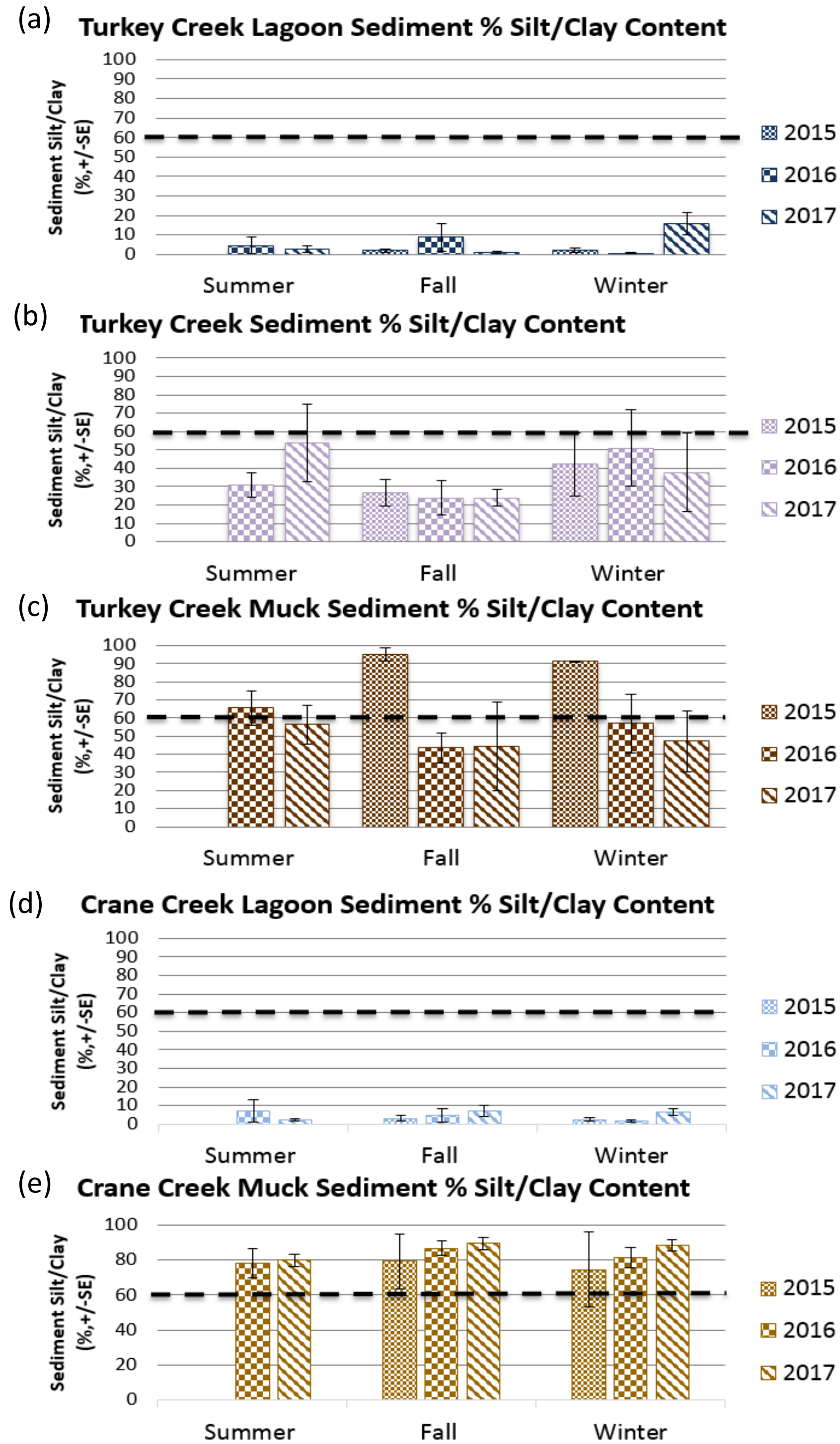


Figure 3.6.3. Mean sediment % silt/clay content at (a) Turkey Creek Lagoon (TCL), (b) Turkey Creek (TC), (c) Turkey Creek Muck, (d) Crane Creek Lagoon and (e) Crane Creek Muck. Error bars are ±1SE. Horizontal dashed lines indicate defined muck silt/clay thresholds (Trefry and Trocine 2011).

Infaunal species abundances, richness and diversity showed significant correlations with sediment % organic matter, % water and % silt + clay. These correlations are shown graphically with richness, diversity and abundance versus sediment % organic matter content (Figure 3.6.4). The biological parameters all have strong inverse logarithmic correlations with sediment parameters (Figure 3.6.4). For overall abundances, the data have been separated by major phyla to illustrate similarities and differences for taxa of the Arthropoda, Mollusca, and Annelida (Figure 3.6.5). The inverse logarithmic relationships were similar among phyla, but with this higher resolution, abundances rise exponentially as organic matter declines to 0.5%, however abundances suddenly drop at 0.5% to 0% organic matter. This pattern suggests a limited organic matter model of infauna population control, where some organic matter is necessary as a food source of infauna, but then quickly exceeds desirable levels as OM increases above 1%.

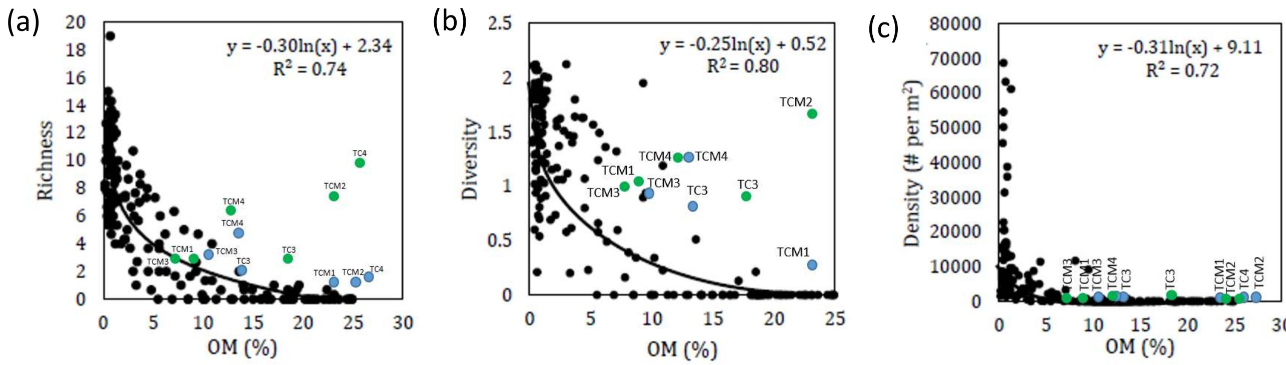


Figure 3.6.4. Species (a) richness, (b) biodiversity and (c) overall organism density of infaunal communities measured at Turkey Creek and associated stations. Colored data points show stations in Palm Bay with intermediate (TC) or full muck (TCM) sediments; samples collected since dredging was completed.

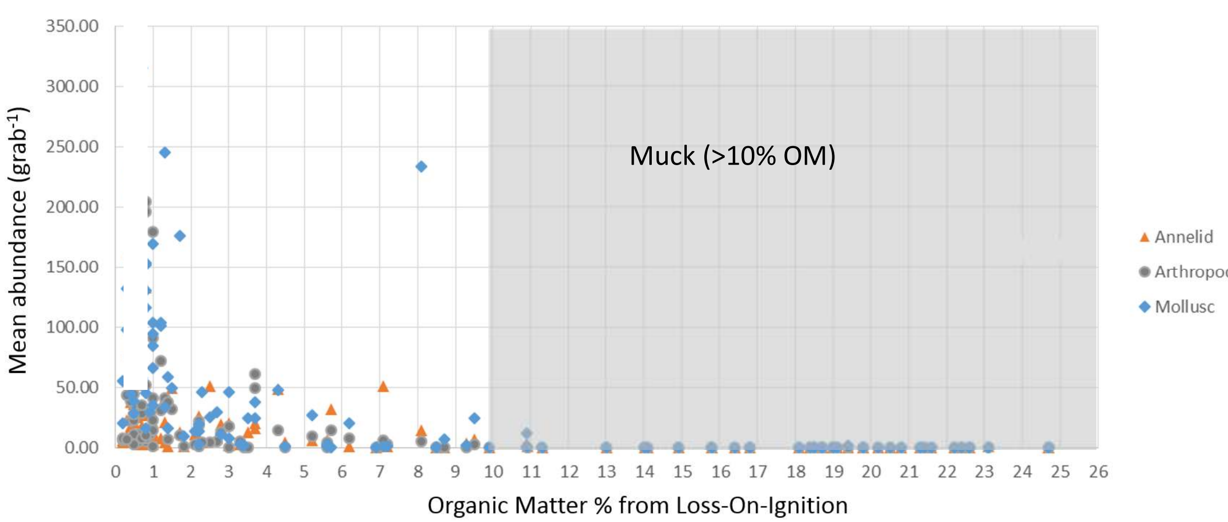


Figure 3.6.5. Mean infaunal abundances for annelids, arthropods, and mollusks as a function of sediment % organic matter. All phyla generally exhibit similar inverse logarithmic correlations, with a major transition or threshold in the range of 1–3%.

Sediment parameters also were shown to have a significant impact on biological parameters via Principal Components Analysis, illustrated here with species richness plotted against the two most significant environmental variables, temperature and sediment parameters (sediment % organic matter used to represent general sediments impacts) (Figure 3.6.6). Sediment conditions had a strong correlation with species richness.

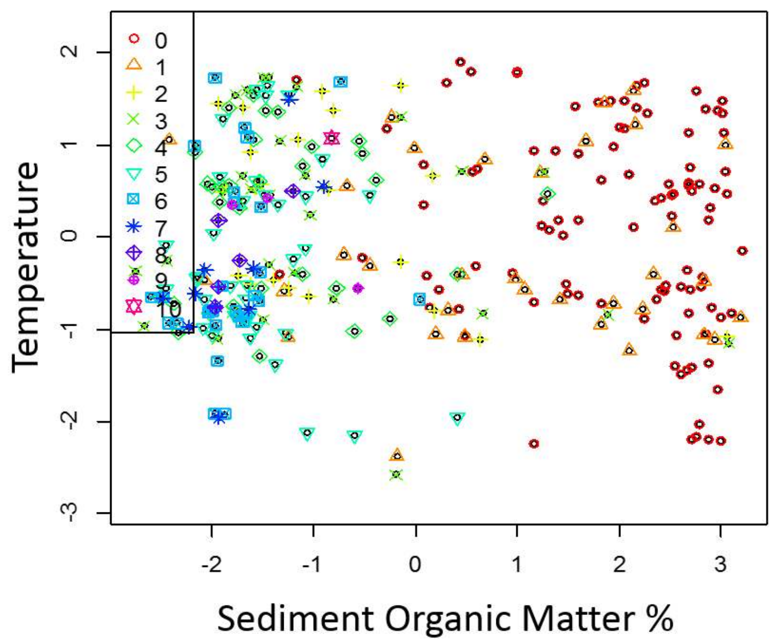


Figure 3.6.6. Principle Components Analysis for species richness (0–10, see legend) and environmental parameters. Sediment characteristics, including organic matter and temperature, were the environmental conditions with the greatest impact on species richness. Sites include Turkey Creek, Turkey Creek Muck, the Indian River Lagoon near Turkey Creek, Crane Creek Muck, and the Indian River Lagoon near Crane Creek. Axes are relative principle components scores and do not reflect actual measured values of temperature of sediment % organic matter.

After completion of Turkey Creek dredging in September 2016, we continued to follow the impacts and development of infaunal communities at stations impacted directly by dredging (e.g., TC3 and TC4, and all TCM stations) relative to non-dredged stations (TC1, TC2 and all CCM stations). Neither of the lagoon sites (all TCL and CCL stations) were dredged, but TCL represents a sandy lagoon habitat adjacent to dredging that can be compared with a similar habitat (CCL) several kilometers away from the dredging. At TC sites, a pre-baseline community of intermediate biological indicators was observed. For example, in the Spring TC stations hosted intermediate abundances ($8x10^3$-$1.5x10^4$ organisms m^{-2}), richnesses (6-11), and biodiversities (1-1.7) (Figures 3.6.7-3.6.9). Immediately following dredging, biological indicators at stations TC3 and TC4 decreased relative to other TC stations. For example, immediately following the completion of dredging, TC stations that were dredged dropped immediately in abundances ($1.2x10^3$ organisms m^{-2} compared to $8x10^3$ m^{-2} for nearby undredged TC stations), diversity (0.8 compared to 1.7 for nearby undredged TC stations) and richnesses (4 compared to 11 for nearby undredged TC stations) (Figures 3.6.7-3.6.9), suggesting that these decreases were related to the dredging rather than seasonal changes. This trend is likely due to the direct removal of animals living in dredged sediments. However, the 2018 data show that the overall infauna abundances bounced back to some degree, when comparing like seasons (Figure 3.6.7). Diversity and richness did not, in most cases, drop significantly in Turkey Creek intermediate stations when comparing like seasons (Figure 3.6.8 and 3.6.9, respectively), with the exception of species richness in the Fall at the dredged intermediate stations (Figure 3.6.9C).

In contrast, the dredged sites with the highest levels of organic sediments (TCM 1–4) had, with few exceptions, a pre-dredging baseline of no animal life. Post-dredging monitoring of these sites, and comparisons with undredged muck in Crane Creek (CCM), shows that abundances and diversity increased at dredged stations (TCM) for most season-by-season comparisons (Figures 3.6.7 and 3.6.8). Although these infaunal communities are not large, they provided the first evidence of consistent life in what were formerly muck sites.

Prior to dredging, infaunal organisms from all three major phyla were less diverse in intermediate (sandy muck) sediments and almost zero in confirmed muck sediments relative to the nearby lagoon sites. Muck sites (TCM, CCM, MDM, MCM, SDM, SCM) usually supported no species, but occasionally sampling found rare animals (Table 3.6.1). Species richness followed similar patterns. Species included foraminiferans, gastropod mollusks, bivalve mollusks, decapod crustaceans, gammarid amphipods, caprellid amphipods, polychaete annelids, ostracod crustaceans, tanaid crustaceans, nematodes and others. The cumulative list of all species found at respective sites (Table 3.6.1) is based on 12 grabs at each major site monthly throughout each year.

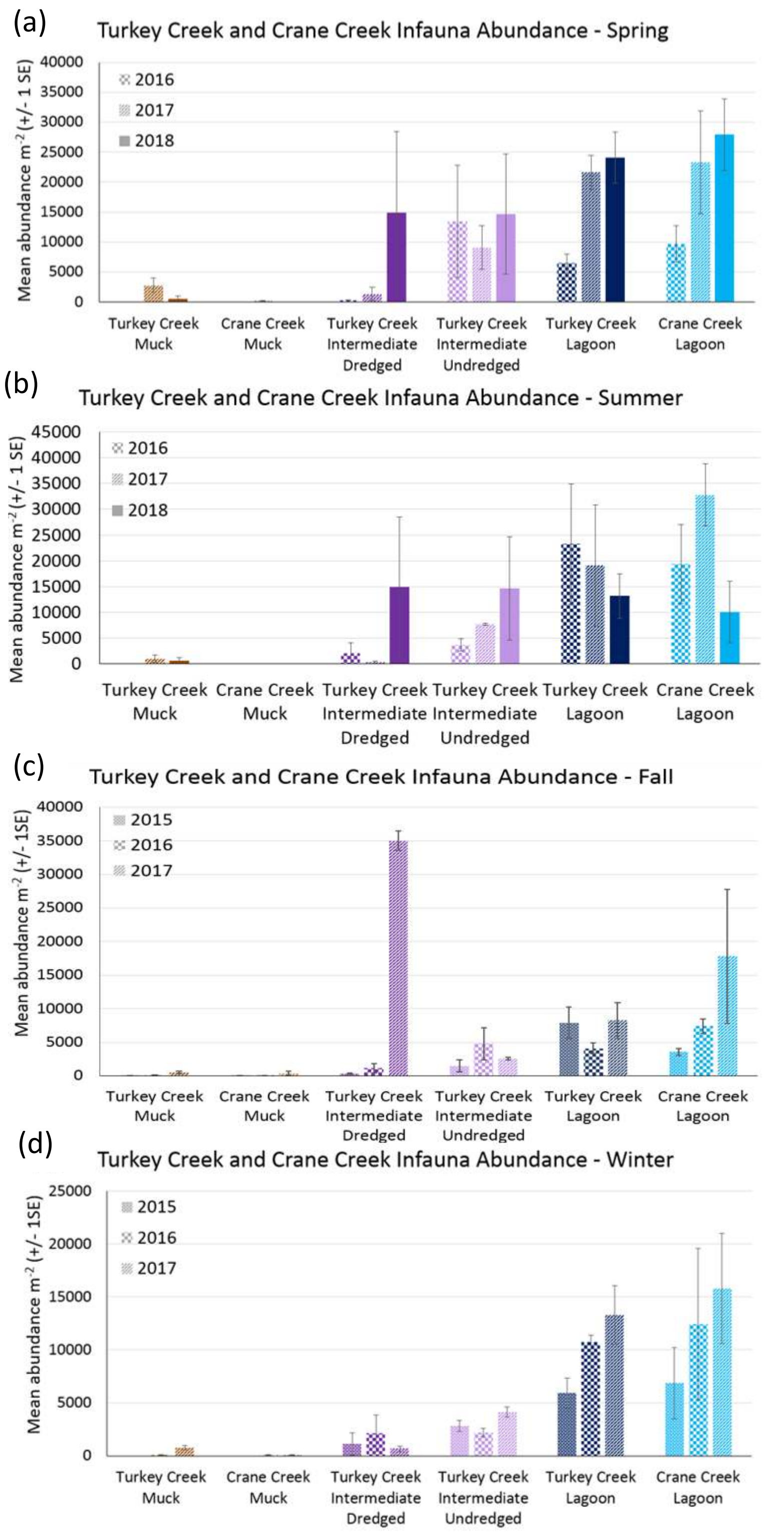


Figure 3.6.7. Mean overall infaunal invertebrate abundances for Turkey Creek and associated sites, compared through the seasons (a) spring, (b) summer, (c) fall and (d) winter. Error bars = ±1SE.

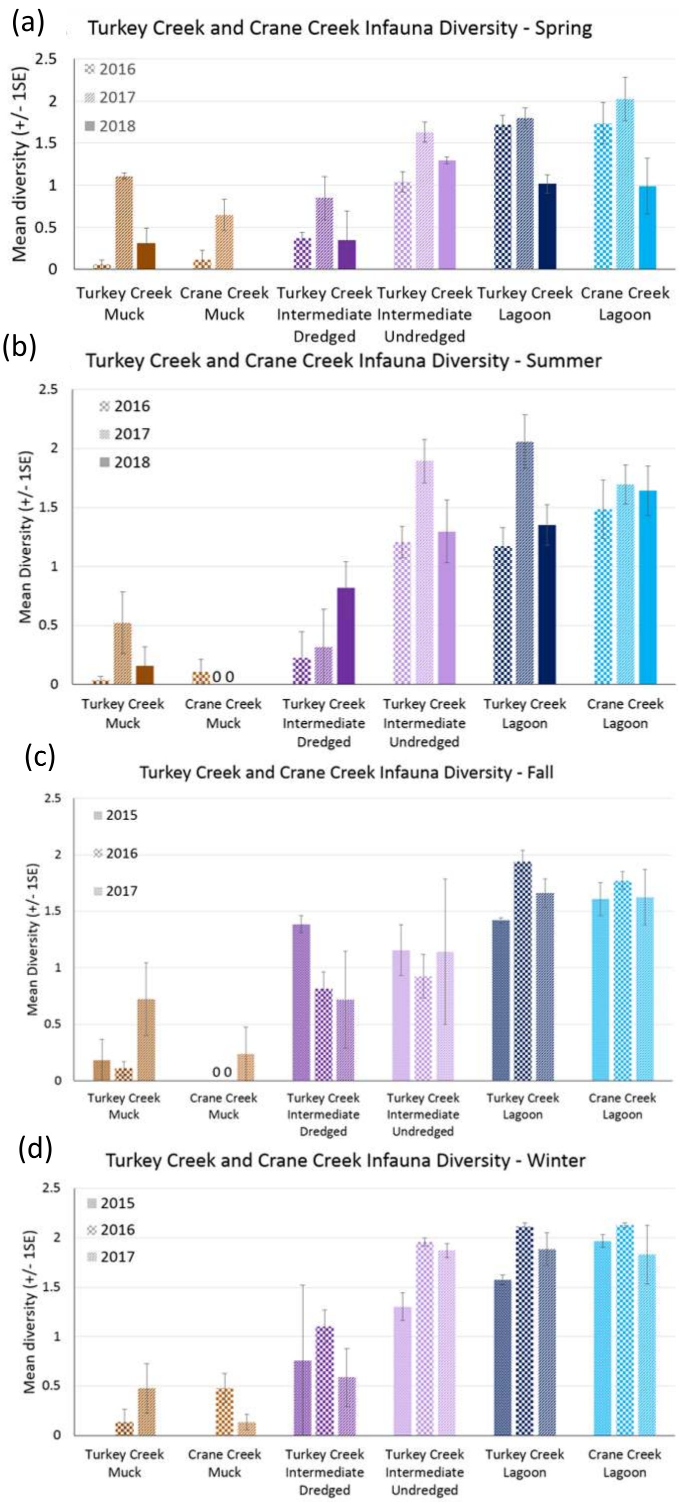


Figure 3.6.8. Mean overall infaunal invertebrate diversity for Turkey Creek and associated sites, compared through the seasons (a) spring, (b) summer, (c) fall and (d) winter. Error bars = ±1SE.

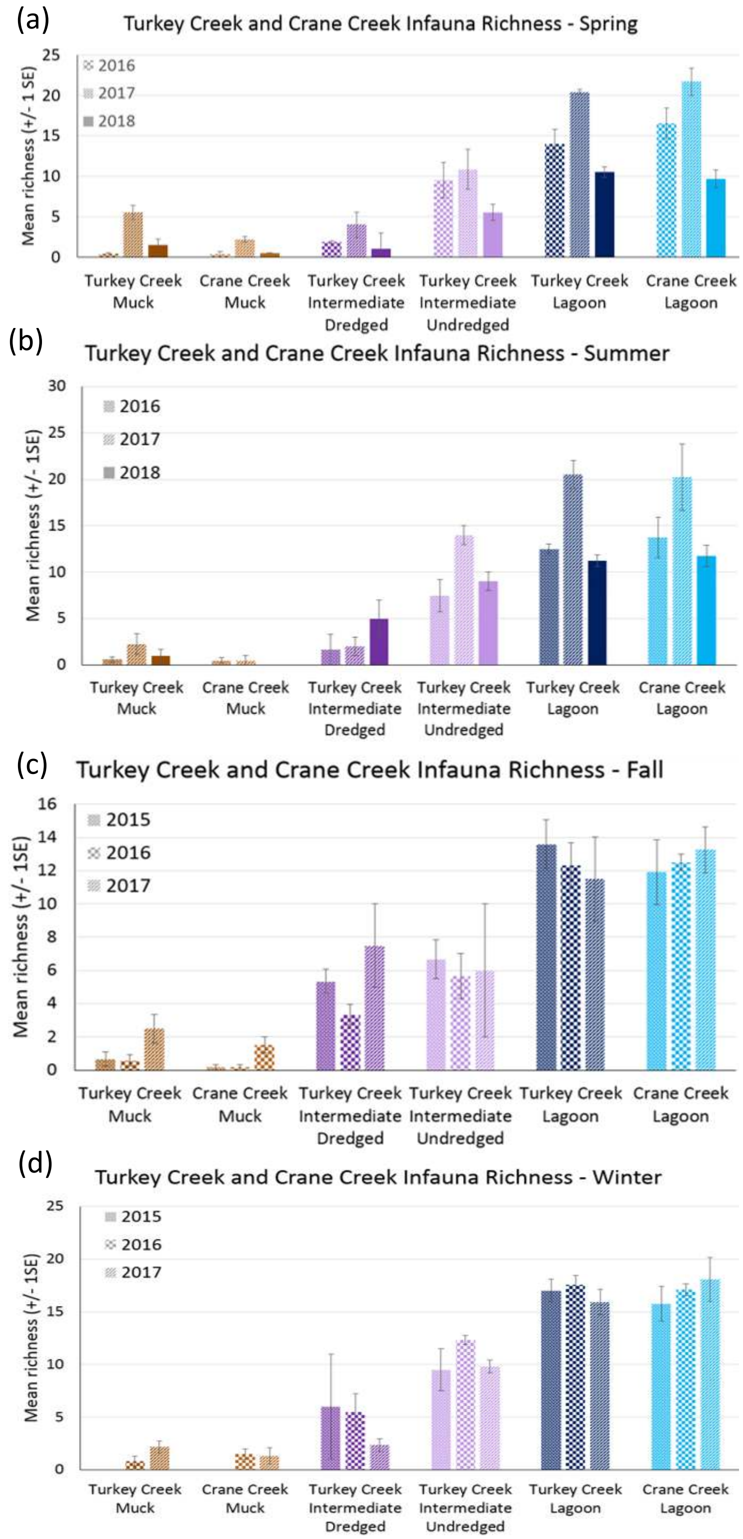


Figure 3.6.9. Mean overall infaunal invertebrate richness for Turkey Creek and associated sites, compared through the seasons (a) spring, (b) summer, (c) fall and (d) winter. Error bars = ±1SE. Sites vary by color, years vary by bar pattern, as indicated. Error bars = ±1SE.

Table 3.6.1. Infauna species present by site during 2017 and 2018. Dominant species are indicated with asterisks (* = dominant in 2017, ** = dominant in 2018, *** = dominant in both 2017 and 2018.

TURKEY CREEK AND ASSOCIATED SITES (70 species)

Turkey Creek	Turkey Creek Lagoon	Crane Creek Lagoon	Crane Creek Muck	Turkey Creek Muck
Acteocina atrata	*Acteocina atrata*	*Acteocina atrata*	*Acteocina canaliculata*	*Acteocina canaliculata*
Acteocina canaliculata	*Acteocina canaliculata****	*Acteocina canaliculata*	*Alitta succinea*	*Ammonia parkinsoniana*
Alitta succinea	*Alitta succinea*	*Alitta succinea*	*Ammonia parkinsoniana*	*Amygdalum papyrium*
*Ammonia parkinsoniana**	*Ammonia parkinsoniana*	*Ammonia parkinsoniana*	*Amygdalum papyrium*	*Astyris lunata*
Amygdalum papyrium	*Amygdalum papyrium*	*Amygdalum papyrium*	*Cerapus tubularis*	*Capitella capitata*
Annelid J	*Angulus versicolor*	*Angulus versicolor*	*Crepidula atrasolea*	*Cerapus tubularis***
Anomalocardia cuneimeris	Annelid J*	*Anomalocardia cuneimeris*	*Ctenodrilus serratus*	*Ctenodrilus serratus*
Astyris lunata	Annelid I	*Astyris lunata*	Gammarid Amphipod C	*Eusirus cuspidatus*
Capitella capitata	*Anomalocardia cuneimeris*	*Capitella capitata**	Gammarid Amphipod D	Gammarid Amphipod C
*Cerapus tubularis****	*Astyris lunata**	*Cerapus tubularis**	Gammarid Amphipod G	Gammarid Amphipod D
Crepidula atrasolea	*Capitella capitata*	Clam C	*Glycera americana*	Gammarid Amphipod F
Ctenodrilus serratus	*Cerapus tubularis****	*Corophium sp.**	*Haminoea succinea*	Gammarid Amphipod G
Eurypanopeus depressus	*Corophium sp.**	Crab B	*Japonactaeon punctostriatus*	*Haminoea succinea*
Eusirus cuspidatus	Crab B	*Ctenodrilus serratus*	*Mulinia lateralis*	*Hargeria rapax*
Gammarid Amphipod D*	*Cratena pilata*	*Cyrtopleura costata*	Nannastacidae A	*Japonactaeon punctostriatus*
Gammarid Amphipod F	*Ctenodrilus serratus**	*Eulithidium pterocladicum*	*Palaemonetes vulgaris*	*Leptochelia dubia*
Gammarid Amphipod G	*Cyrtopleura costata*	*Eusarsiella zostericola*	*Parastarte triquetra*	*Mulinia lateralis***
Gammarid Amphipod I	*Diopatra cuprea*	*Eusirus cuspidatus*	Polychaete T	*Odostomia laevigata*
Glycera americana	*Eurypanopeus depressus*	*feather worm / fungus*		*Palaemonetes vulgaris*
Hargeria rapax	*Eusirus cuspidatus*	Gammarid Amphipod C		*Parastarte triquetra*
Hypereteone heteropoda	*feather worm / fungus*	Gammarid Amphipod D*		*Pectinaria gouldii*
Isopod B	Gammarid Amphipod C	Gammarid Amphipod F		
Japonactaeon punctostriatus	Gammarid Amphipod D*	Gammarid Amphipod G		
Leptochelia dubia	Gammarid Amphipod G*	Gammarid Amphipod H		
*Mulinia lateralis****	Gammarid Amphipod I	Gammarid Amphipod I		
Nannastacidae A	*Glycera americana*	*Glycera americana*		
Nassarius vibex	*Haminoea succinea*	*Haminoea succinea*		
Nematode A	*Hargeria rapax*	*Hargeria rapax*		
Ostracod C	*Hypereteone heteropoda*	*Hypereteone heteropoda*		
Oxyurostylis smithi	Isopod B	Isopod A		
Palaemonetes vulgaris	Isopod C	Isopod B		
Paradiopatra hispanica	*Japonactaeon punctostriatus*	Isopod C		
Parastarte triquetra	*Leptochelia dubia***	*Japonactaeon punctostriatus*		
Pectinaria gouldii	*Mulinia lateralis***	*Leptochelia dubia****		
Peratocytheridea setipunctata	Nannastacidae A*	*Melongena Corona*		
Periglypta listeri	*Nassarius vibex*	*Mercenaria mercenaria*		
Periglypta listeri	*Nassarius vibex*	*Mercenaria mercenaria*		
*Phascolion cryptus**	Nematode A	*Mulinia lateralis**		
Pipefish	*Odostomia laevigata*	Nannastacidae A*		
Polychaete T	Ostracod C	*Nassarius vibex*		
Polychaete Z	*Oxyurostylis smithi*	Nemertea A		
Sipuncula B	*Palaemonetes vulgaris*	*Odostomia laevigata*		
Sipuncula C*	*Paradiopatra hispanica**	*Oxyurostylis smithi**		

Table 3.6.1, *continued*

Turkey Creek	**Turkey Creek Lagoon**	**Crane Creek Lagoon**	**Crane Creek Muck**	**Turkey Creek Muck**
	*Parastarte triquetra****	*Palaemonetes vulgaris*		
	Pectinaria gouldii	*Paradiopatra hispanica*		
	*Peratocytheridea setipunctata***	*Parastarte triquetra****		
	Periglypta listeri	*Pectinaria gouldii*		
	Phascolion cryptus	*Peratocytheridea setipunctata**		
	Polychaete T	*Periglypta listeri*		
	Polychaete Y	*Phascolion cryptus*		
	Polychaete Z	Pipe Fish A		
	Sipuncula B	Polychaete M		
	Sipuncula C	Polychaete S		
	Snail P	Polychaete T		
	Tanaid A	Polychaete Y		
	Turbellaria A	Shrimp C		
		Sipuncula C		
		Tanaid A		
		Tanaid C		
		Turbellaria A		

3.7. *Fish Surveys*

Turkey Creek:

A total of 312 seine samples was collected during the 41-month survey in Turkey Creek. These seine samples collected an estimated total of 3.7 million fishes from 59 taxa from April 2015 through August 2018 (Table 3.7.1). Catch totals of all individual taxa collected from the Turkey Creek region are presented in Appendix C.

Table 3.7.1. Total number of fishes captured each year by seine net at 8 stations within Turkey Creek and 2 stations in the adjacent Indian River Lagoon.

Year	Sampling Frequency	Number of samples	Total Number of Fishes	Number of Fishes Minus Anchovies	Total Number of Taxa
2015	Monthly, Apr.-Dec.	90	1,812,498	11,369	50
2016	Monthly, Jan.-Dec.	109	1,734,610	46,568	44
2017	Monthly, Jan.-May, Bimonthly, Jul.-Dec.	85	126,808	25,687	45
2018	Trimonthly Jan.-Aug.	28	8,749	6,200	34
Total 2015-2018	41 months	312	3,682,665	89,824	59

The adult fishes that utilize Turkey Creek habitats are not vulnerable to capture using the FIM juvenile fish sampling techniques, and are thus not represented in our data. Interviews with anglers fishing in Turkey Creek, and our personal experiences, have identified a number of adult fishery species, including sheepshead (resident on dock pilings and oyster/rock piles), jacks and red drum (highly mobile schooling species), snook (solitary ambush predators), tarpon and (juvenile) bull sharks (solitary mobile predators). The mobile predators tend to follow schools of prey, including anchovies, mullet, and herrings, so Turkey Creek habitat utilization is variable. In the months following the completion of dredging in fall 2017 and early 2018, anglers were frequently seen fishing along the western edge of the Turkey Creek habitat, casting into the deeper water that encroached closer to the shoreline as a result of dredging. Conversations with anglers determined they were catching jacks, snook, red drum, tarpon and juvenile bull sharks from shore.

The fishes captured in the seine net throughout the study were dominated by small pelagic schooling species (Table 3.7.2). Pelagic anchovies (*Anchoa* spp.) comprised 97.4% of the total seine catch, indicative of the numerical dominance of these fishes within the IRL ecosystem. The three largest catches of anchovies illustrate the very high temporal and spatial variability of these schooling fish. In December 2015, dense schools of anchovies were found in the northern section of Turkey Creek (F-N); two seine hauls captured an estimated 520,000 and 901,000 fishes, while the other two seine samples in the region collected only 800-5,000 anchovies. Samples taken elsewhere in Turkey Creek that month yielded from 15 to 40,000 anchovies. An even denser school was sampled in June 2016 at the near-shore station of F-C, outside the mouth of Turkey Creek. The catch almost filled two 40 L buckets with an estimated total of 1,600,000 anchovies. Seine hauls made on the same date, 50 m farther offshore outside the mouth of Turkey Creek, and at 6 other stations within the creek, collected no anchovies at all.

The non-anchovy component of the fish catch was dominated by pelagic and demersal juvenile fishes (Appendix C). Pelagic fishes, dominated by juvenile herrings (Clupeidae and Atlantic menhaden, *Brevoortia tyrannus*), and mullets (*Mugil* spp.), comprised 38.6% of the non-anchovy catch. These schooling pelagic fishes move rapidly around the IRL, and can quickly move in and out of the Turkey Creek area.

A total of 54,098 juvenile demersal fishes were collected during the sampling period (Table 3.7.2). These demersal juveniles are the most likely fishes that could potentially be impacted by changes to the benthic ecosystem in Turkey Creek. The samples were dominated by two families of fishes: Mojarras (Gerridae) and Drums (Sciaenidae). One member of each of these families was the most abundant of the juvenile demersal fishes: mojarras (*Eucinostomus* spp.) and Atlantic croakers (*Micropogonias undulatus*) comprised 42.7 and 18.7% of the demersal juvenile fish catch, respectively. The two dominant mojarra taxa and five species of juvenile drums were selected for analysis. Three lines of evidence were considered for each taxon: (1) short-term temporal and spatial distribution patterns of individual species; (2) broader interannual variability in population size that influences the abundance of taxa in the study area; and (3) trophic interactions between juvenile fishes and benthic or epibenthic prey that might be affected by muck removal.

(1) The first factor to consider is the short-term temporal and spatial distribution of juvenile fishes among the stations inside and outside Turkey Creek during pre-dredging (2015 and early 2016), dredging (February 2016 to January 2017, with a hiatus in dredging activity from May to September to avoid interactions with manatees) and post-dredging (after January 2017) periods. Simple comparisons among sites and time periods, however, are significantly hindered by variability in habitat structure, the biological/ecological characteristics of each fish species, and environmental conditions. The physical structure of each habitat can influence fish distributions by affecting the populations of potential prey and by providing shelter from or exposure to predators. Within the sampling region, the stations along the western edge of the Turkey Creek basin (F-W in Figure 2.5.1) were characterized by hard sand along the shore, grading into increasingly soft, mucky sediments offshore. After muck removal, the hard sand habitats dropped sharply into water 2-3 m deep at the edge of the dredge cut. A broader band of shallow habitat characterized the northern portion of the basin (F-N), with sand and rock rubble along the shore grading gradually into deeper muck offshore. Much of this region was covered with dead oyster shells, providing evidence of an earlier oyster community at the mouth of Turkey Creek. The site outside the mouth of Turkey Creek (F-O) had a hard sand substrate. No seagrasses were observed in any of these sampling regions.

Table 3.7.2. Total catch of each fish taxon collected by seine net at 10 stations in and around Turkey Creek, April 2015-August 2018.

Scientific Name	Common Name	Total Abundance	% of Total Fish	Pelagic or Demersal	% of demersal total
Anchoa spp.	Anchovy	3,592,841	97.49	P	
Eucinostomus spp.	Mojarra	23,575	0.64	D	42.76
Clupeidae	Herrings	23,518	0.64	P	
Micropogonias undulatus	Atlantic croaker	10,294	0.28	D	18.67
Diapterus spp.	Irish pompano	6,871	0.19	D	12.46
Mugil curema	White mullet	4,962	0.13	P	
Bairdiella chrysoura	Silver perch	4,717	0.13	D	8.56
Sciaenidae spp.	Drums	4,165	0.09	D	7.56
Brevoortia spp.	Menhaden	3,300	0.09	P	
Menticirrhus americanus	Southern kingfish	1,308	0.07	D	2.65
Mugil cephalus	Striped mullet	934	0.03	P	
Menidia spp.	Silversides	920	0.02	P	
Mugil spp.	Mullets	612	0.02	P	
Cynoscion spp.	Sea trout	508	0.01	D	0.92
Sciaenops ocellatus	Red drum	501	0.01	D	0.91
Gobiosoma robustum	Code goby	398	0.01	D	0.72
Leiostomus xanthurus	Spot	352	0.01	D	0.64
Lagodon rhomboides	Pinfish	286	0.01	D	0.51
Oligoplites saurus	Leatherjacket	240	0.01	P	
Achirus lineatus	Lined sole	210	0.01	D	0.38
Gobiidae spp.	Gobies	169	<0.01	D	0.31
Gobiosoma bosc	Naked goby	166	<0.01	D	0.30
Archosargus probatocephalus	Sheepshead	126	<0.01	D	0.23
Strongylura sp.	Needlefish	126	<0.01	P	
Ariopsis felis	Hardhead catfish	76	<0.01	D	0.14
Syngnathus spp.	Pipefish	57	<0.01	D	0.10
Carangidae spp.	Jacks	36	<0.01	P	
Bagre marinus	Gafftopsail catfish	35	<0.01	D	0.06
Paralichthys spp.	Flounder	31	<0.01	D	0.06
Microgobius gulosus	Clown goby	29	<0.01	D	0.05
Chasmodes saburrae	Florida blenny	25	<0.01	D	0.04
Sphyraena barracuda	Barracuda	23	<0.01	P	
Gerres cinereus	Yellowfin mojarra	22	<0.01	D	0.04

Table 3.7.2 (continued).

Scientific Name	Common Name	Total Abundance	% of Total Fish	Pelagic or Demersal	% of demersal total
Dasyatis sabina	Atlantic stingray	21	<0.01	D	0.04
Chaetodipterus faber	Spadefish	18	<0.01	D	0.03
Elops saurus	Ladyfish	18	<0.01	P	
Sphoeroides sp.	Puffers	17	<0.01	D	0.03
Micropterus salmoides	Largemouth bass	16	<0.01	D	
Centropomus undecimalis	Common snook	13	<0.01	D	0.02
Lutjanus griseus	Mangrove snapper	12	<0.01	D	0.02
Citharichthys spilopterus	Bay whiff	11	<0.01	D	0.02
Albula spp.	Bonefish	10	<0.01	D	0.02
Hyporhamphus unifasciatus	Common halfbeak	9	<0.01	P	
Fundulus spp.	Killifish	8	<0.01	D	0.01
Triglidae	Sea robins	8	<0.01	D	0.01
Haemulon spp.	Grunts	6	<0.01	D	0.01
Pogonias cromis	Black drum	6	<0.01	D	0.01
Trachinotus carolinus	Florida pompano	6	<0.01	D	0.01
Hypleurochilus pseudoaequipinnis	Oyster blenny	4	<0.01	D	0.01
Chilomycterus shoepfi	Striped burrfish	3	<0.01	D	0.01
Orthopristis chrysoptera	Pigfish	3	<0.01	D	0.01
Sarotherodon melanotheron	Blackchin tilapia	3	<0.01	D	0.01
Cynoglossidae spp.	Tonguefish	2	<0.01	D	<0.01
Gobiesox strumosus	Clingfish	2	<0.01	D	<0.01
Gymnura micrura	Butterfly ray	2	<0.01	D	<0.01
Centropomus ensiferus	Swordspine snook	1	<0.01	D	<0.01
Chloroscombrus chrysurus	Atlantic bumper	1	<0.01	P	<0.01
Dasyatis americana	Southern stingray	1	<0.01	D	<0.01
Lucania parva	Rainwater killifish	1	<0.01	D	<0.01
TOTAL PELAGIC FISH		**3,627,540**			
TOTAL DEMERSAL FISH		**54,098**			
TOTAL FISH		**3,681,638**			

Variations in the spawning biology and habitat preferences of juveniles also influence the temporal and spatial patterns of fish distribution (Tremain and Adams, 1995; Paperno et al., 2010). Different fish species have different seasonal patterns of reproduction and recruitment, with some taxa spawning inside the IRL and others spawning in offshore waters with larvae entering the IRL through inlets. The sporadic or seasonal occurrence of large pulses of juveniles of a species are thus associated with the spawning biology of that species.

Juvenile fishes may also respond to highly variable environmental conditions and water quality (Paperno et al. 2004, 2006), including seasonal temperature cycles, spatial salinity gradients, and rapid changes driven by major storm events and freshwater discharges (Figure 3.7.1). For example, a single major water discharge event occurred in 2015, with larger discharge events occurring after Hurricane Matthew (2016) and Hurricane Irma (2017). These elevated discharges resulted in high current velocities across the sampling area, and a reduction in salinity at the mouth of Turkey Creek to near zero for periods of time ranging from a few days to several weeks. Changes in salinity, plus high currents, may directly affect juvenile fishes or induce indirect responses by affecting the distribution and abundance of their prey.

Another example of a significant environmental event occurred during the very dry late winter and spring of 2017. Salinities did not dramatically change during this period, but a week of very strong southerly/southeasterly winds in March drove huge quantities of *Gracilaria* algae onto the northern portion of the bay (region F-N in Figure 2.5.1). This stranding of *Gracilaria* extended from the shoreline to 50 m into the bay, and filled the water column to a depth of a meter or more. Similar strandings were reported along the shoreline elsewhere in Brevard County, triggering numerous requests from citizens to governmental agencies to clean up the decaying algae. We were not able to collect fish samples from areas covered with dense algal mats, but the lack of open water and the presumably low dissolved oxygen levels associated with the decaying biomass undoubtedly reduced the suitability of this region as habitat for mobile demersal juvenile fishes that month.

(2) A second factor to consider in the evaluation of trends in fish abundance is interannual variability in the magnitude of fish recruitment within the Indian River Lagoon. Significant interannual variability in the success of reproduction and recruitment is characteristic of the populations of most estuarine and coastal fishes (e.g. Allen and Barker 1990; Jenkins et al. 1997; Shenker et al. 1993, 2002), and that variability sets the baseline upon which locally-driven changes are superimposed. Assessing interannual variation in abundance of fishes is a primary goal of the Florida Fish and Wildlife Conservation Commission's Fisheries Independent Monitoring Program (FIM). These intensive fish surveys have been conducted in many Florida estuaries for decades. FIM sampling within the IRL enables a comparison of the Turkey Creek data to the broader fish populations within the lagoon ecosystem. Because FIM seeks to cover the entire northern Indian River Lagoon, sampling across broad sections of the ecosystem (Zones A, B, C, D, E and H in Figure 3.7.2) are used to calculate mean density data that can be used as indices of fish recruitment and abundance in those sectors for each year.

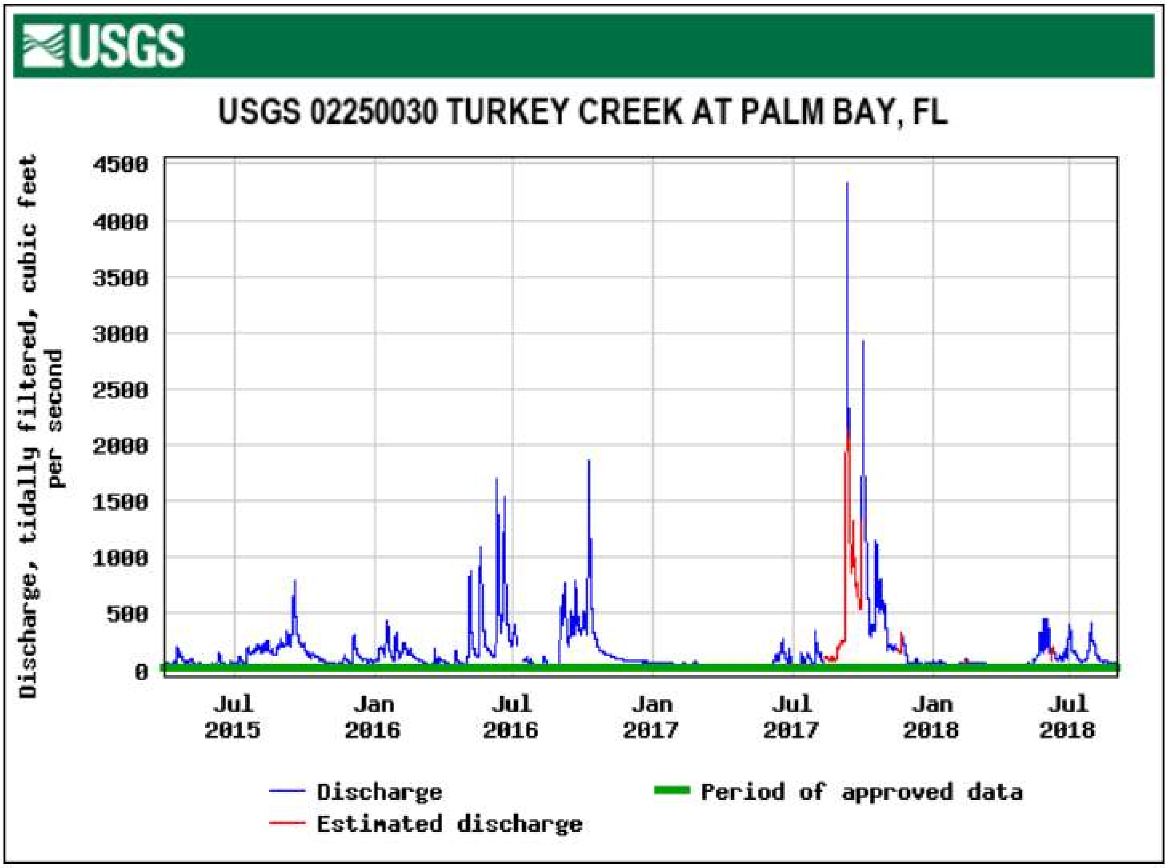


Figure 3.7.1. Daily discharge (cubic feet/second) of Turkey Creek from April 2015 to August 2018, as measured by USGS Station 02250030.

FIM annual reports present the mean density data for taxa collected by seine net across all open-water habitats in the northern lagoon. Data for the two major tributaries into the lagoon (Turkey Creek and the St. Sebastian River) are summarized separately from the open-water habitats. Prior to 2013, Turkey Creek samples were separated from the Sebastian River for analysis as Zone O for Turkey Creek and Zone F for Sebastian River. Beginning in 2013, Turkey Creek samples and St. Sebastian River samples were pooled into the "river samples" category (Zone F in Figure F4). Sampling efforts within these river systems covered all habitats from the mouth to the headwaters, and were not directly equivalent to our efforts in the dredged habitat within the mouth of Turkey Creek. The use of random stratified sampling protocols to select sampling sites across the entire lagoon precludes direct analysis of small portions of the habitat: only 30 seine net samples were taken inside and adjacent to the mouth of Turkey Creek from 1991 to 2014.

To describe the short and long term abundance patterns and microhabitat associations for each taxon being analyzed, we present:

- A time series of mean (± S.D.) and maximum monthly density data for our sampling at the 8 sites within Turkey Creek and the 2 sites adjacent to the mouth of Turkey Creek throughout the entire sampling period.
- A longer-time series (2010 to 2017) of mean annual density from the FIM program for open estuary (Zones A, B, C, D, E, H) and Turkey Creek/Sebastian River (Zone O prior to 2013, and Zone F beginning in 2013) habitats, along with our mean annual density data from our Turkey Creek sites from 2015-2018.
- An analysis of the microhabitats where the largest catches of each taxon were made.
- A detailed comparison of the small-scale temporal and spatial distributions of selected taxa within the mouth of Turkey Creek during this study with fine-scale data collected by FIM over the past decade.

(3) The third way to assess potential impacts of muck removal on fish utilization of a habitat focuses more at the mechanisms of how the fish utilize the habitat than the analysis of population sizes and distributions. Removing muck through dredging generally deepens the habitat beyond the depth where seine samples can be collected that are comparable with those collected around the periphery of the dredge site. Visual surveys in these deeper sites are also not feasible in the often highly turbid waters of the Indian River Lagoon. However, analysis of feeding habits of the fishes, and the distribution of their prey, can provide important information about how the populations may be able to respond to muck removal. If the prey base expands in area and in overall abundance following environmental restoration, the habitat may be able to support larger populations of fish over larger areas of the benthic environment.

Stomach contents were examined of representative subsamples of demersal fishes collected throughout the seining program. The data presented in this report provide a summary of the feeding biology of selected taxa.

Figure 3.7.2. Sampling zones of the Florida Fish and Wildlife Conservation Commission's Fisheries Independent Monitoring Program for calculation of annual density data of fishes in the northern Indian River Lagoon (FWCC 2014). Mean density data from 22-m seine net samples are calculated separately for IRL (open lagoon) habitats in zones A, B, C, D, E and H, and for "river" habitats (i.e., Turkey Creek and St. Sebastian River) in zone F.

Taxon Analysis in Turkey Creek

Gerridae: Mojarras - Two genera of juvenile mojarras were the first and third most abundant of the demersal taxa found in Turkey Creek. Small juvenile mojarras *Eucinostomus* spp. and juvenile Irish pompano *Diapterus* spp. are difficult species to identify, so the fishes were generally identified to the genus level. *Eucinostomus* spp. were the most abundant demersal fishes collected during the entire study (n=23,575), while *Diapterus* spp. ranked third in abundance (n=6,871). Adult *Eucinostomus* and *Diapterus* species range from 150-350 mm SL, and are presumed to spawn in offshore or inlet habitats (Kerschner et al. 1985).

Eucinostomus spp.: The occurrence of juvenile *Eucinostomus* spp. (n=23,575) throughout the study was composed mostly of fishes ranging from 15-90 mm SL. The collection of small juveniles during most months suggests a protracted spawning season for these fishes. The monthly time-series of juvenile *Eucinostomus* spp. densities (number of fish $100m^{-2}$) presents mean (+/- S.D.) density data for all seine hauls made each month inside and outside the mouth of Turkey Creek, as well as the highest density measured each month (Figure 3.7.3).

These data demonstrate dramatic variability within each monthly sampling period, across the months within a year, and among the years of the study. Variability in density among the stations within each monthly sampling period is visualized by the comparison of mean and maximum densities for each sampling period. For example, the greatest densities of *Eucinostomus* spp. were collected in January 2017, during the last month of the dredging program. Mean density across the entire sampling region was 255 fish $100m^{-2}$, with a highest density of nearly 1000 fish $100m^{-2}$ taken at a station along the western edge of the embayment (F-W). Although distinct seasonal peaks in fish abundance are not repeated each year, these juveniles were absent from Turkey Creek during early fall, and especially after periods of intense rainfall that triggered elevated discharge levels, reduced salinities and high current velocities. As one measure of interannual variation in abundance, 9 of the 12 largest catches of these juveniles occurred between June 2016 and June 2017 out of the 41-month study.

The evaluation of mean and maximum density across the sampling region provides evidence for monthly, seasonal and interannual variation; but detection of patterns in microhabitat selection require a different approach. For example, the peak abundance of *Eucinostomus* spp., in January 2017, occurred entirely in the 4 seine hauls made in F-W. We could not sample F-N that month because dredge operations prevented access to those sites, but densities were very low in F-O. A similar pattern of high densities at F-W and low densities at both F-N and F-O occurred the previous month, December 2016, suggesting persistent microhabitat selection by these juveniles.

To identify the microhabitats that supported the highest densities, the peak abundance analysis focused on the samples that exceeded a density threshold that varied by species. *Eucinostomus* spp. were taken in a total of 252 samples during the study, but many samples collected only a few individuals. A minimum density threshold of 30 fish $100m^{-2}$ was used to select samples (n = 116) for inclusion in microhabitat assessment analysis. After selection of the samples for inclusion, they were grouped by sampling region to calculate total catch abundance in each region, and the total number of peak catches in each region. A contingency table analysis (G-Test) was then used to test for significant differences in total abundance or total number of peak catches among sites.

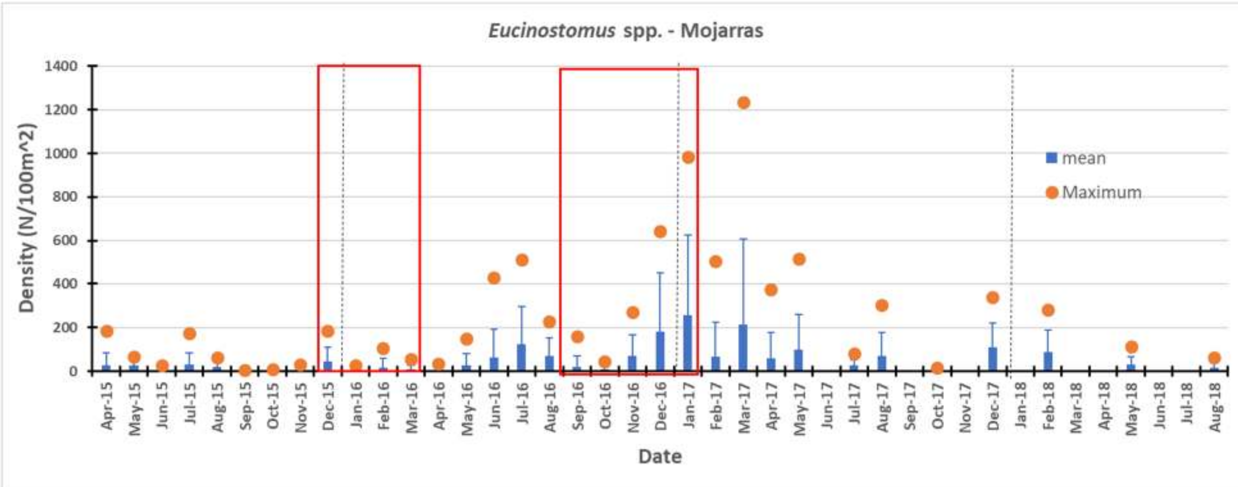


Figure 3.7.3. Mean (+/- S.D.) and maximum densities (number $100m^{-2}$) of juvenile mojarras (*Eucinostomus* spp.) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Dashed lines separate years.

For *Eucinostomus* spp., total abundance of fish in the high density catches made at the two sites within F-W far exceeded the catches at the two sites in F-N and F-O (Figure 3.7.4). These differences were highly significant (G-Test; G=16,522; df=4; p<0.001). Similarly, the numbers of peak catches were significantly higher in F-W than F-N or F-O (G-Test; G=34.8; df=4; p<0.001). No differences were noted between total abundance from peak catches made in shallow (0.25-0.5 m depths) and deep (0.5-1.0 m depths) (G-Test; G=2.64; df=1; p=0.104), although significantly more peak catches were made in the shallow tows than deep tows (G-Test; G=4.63; df=1; p<0.03).

The peak catches in F-W, compared to the other 2 habitats, presumably reflect selection of the substrate by these juveniles or differential mortality levels in the different regions. F-W was characterized by hard sand near the shore, grading into softer sediments offshore. F-N had oyster and rock rubble substrata, while F-O was completely covered by hard sand. Juvenile *Eucinostomus* are often found just outside the swash zone along low-energy sandy beaches in Florida and Bahamas, where lurking or cryptic predator densities are low. Studies of feeding habits indicate that diets of these fish in sandy habitats are dominated by epibenthic crustaceans (mysids) or amphipods that they capture and winnow out of the sand using their highly protrusible jaws (Zahorscak et al. 2000). In mixed sand/seagrass habitats in Tampa Bay, these fishes had a more diverse diet, dominated by polychaetes as well as epibenthic crustaceans and gastropods and small infaunal bivalves (Kerschner et al. 1985; Mota et al. 1995).

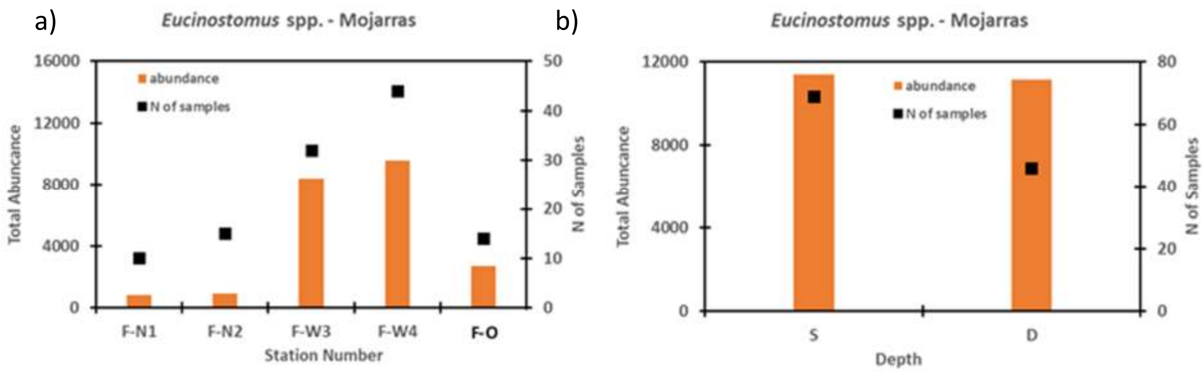


Figure 3.7.4. a) Total abundance of juvenile mojarras, *Eucinostomus* spp., taken in peak catches (>30 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile mojarras, *Eucinostomus* spp., taken in peak catches at Shallow (0.25-5m) and Deep (0.5-1.0m) seine depths in the sampling region.

Based on gut content analysis of fishes taken in Turkey Creek sampling (Table 3.7.2) amphipods and mysid shrimp were dominant prey taxa. Other major prey include grass shrimp, tanaids, ostracods, bivalves and gastropods. Many of these prey taxa are mobile epifaunal crustaceans (mysids, grass shrimp, tanaids), while some may be tube-dwelling infauna or associated with seagrasses or *Gracilaria* drift algae (Nelson 1981, 1995; Nelson et al. 1982). Infaunal samples collected in and around the muck dredging sites in Turkey Creek (Figure 3.7.6), showed that these prey were never found in any muck deposits prior to dredging. Mallik (2019) showed that amphipod diversity and abundance initially increased at sites within months after muck removal, suggesting at least a temporary increase in the availability of prey across a broader habitat. However, amphipod diversity rapidly declined to a single species, and its abundance declined within a year after muck removal. Muck removal is not completely effective, and perhaps slumping of muck as well as down-stream transport of additional organic matter and sediments following the extensive rainy seasons in 2016 and 2017 may have reduced the suitability of some of the dredged habitat to support infaunal organisms. The highly mobile epifauna such as mysids and juvenile shrimp were not effectively sampled by the infaunal grab sampler, and their associations with various types of substrata are not known.

Table 3.7.3. Frequency of occurrence (%) of prey taxa in stomachs of selected fish taxa collected by seine net around the mouth of Turkey Creek. Frequency calculations are based on the numbers of stomachs containing at least one recognizable prey item.

Fish Species	Total (n)	Empty (%O)	Fish	Amphipod	Tanaids	Isopod	Copepod	Mysid Shrimp	Grass Shrimp	Penaeid Shrimp	Polychaete	Bulla occidentalis	Ostracod	Bivalve	Gastropod	Crab	Plant	Diatom	Foraminifera	Nemertean
Gerreidae - Mojarras																				
Diapterus spp.	6	50	0	33	33	0	0	0	0	0	33	0	0	0	0	0	33	0	0	0
Eucinostomus spp.	41	20	0	27	15	0	3	24	18	3	3	12	15	12	15	0	0	0	0	0
Sciaenidae - Drums																				
Bairdiella chrysoura	15	53	14	71	14	0	0	14	0	0	0	0	0	0	0	0	14	0	0	0
Cynoscion nebulosus	12	8	27	36	0	0	0	36	0	45	18	0	0	0	0	0	0	0	0	0
Micropogonias undulatus	32	63	0	50	0	0	33	0	0	0	8	0	0	0	0	0	0	8	0	0
Sciaenops ocellatus	15	0	7	40	7	27	33	0	13	40	0	0	7	0	7	0	0	0	0	0
Leiostomus xanthurus	8	0	0	0	0	0	0	38	0	0	0	0	38	0	0	0	0	0	63	13

Analysis of the high density catches made throughout this study (Figure 3.7.3) demonstrates significant temporal variability in overall population abundance that may indicate at least a short-term response of these fishes to dredging. The time series of data outlined in Figure 3.7.3 were divided into 6 temporal periods: Pre-dredging (April-November 2015); initial dredging (December 2015-March 2016); interim non-dredging (April-August 2016); final dredging (September 2016-January 2017), post-dredging year 1 (February-December 2017, and post-dredging year 2 (February-August 2018). High density catches were not evenly distributed among these periods (G-Test; $G=8979$; $df = 5$, $p<0.001$), with the highest abundance of juvenile *Eucinostomus* taken during the final dredging period (5,676 fish = 25.1% of the total peak abundance catch), and during post-dredging year 1 (8,479 fish = 38.4% of the total peak abundance catch). The lowest abundance levels were observed during the initial dredging period and during the relatively limited sampling effort expended in post-dredging year 2.

This high temporal variation in abundance of juvenile *Eucinostomus* spp. may also be influenced by interannual variability in reproduction and recruitment of *Eucinostomus* spp. across the IRL, rather than reflecting localized responses solely with the Turkey Creek region. Comparison of our Turkey Creek/IRL data with the larger FIM data set illustrate the longer term temporal and spatial trends on populations of these juveniles (Figure 3.7.5). FIM data for each year from 2010 to 2017 are compiled into annual mean (+/- SE) density data for dominant taxa within the IRL (open lagoon) habitats and "river" (i.e., Turkey Creek and Sebastian Inlet) habitats (FWCC 2010, 2011, 2012, 2013, 2014b, 2015, 2016, 2017).

The FIM data indicate that the mean densities of juvenile *Eucinostomus* spp. populations were far denser in the riverine tributaries (annual densities ranged between 72-214 fish $100m^{-2}$) than in the open habitats of the IRL (below 5 to 26 fish/100 m^2) from 2010 to 2017. Interannual variability over this 8-year period in the riverine tributaries reached 300%. Comparison of the summary FIM data and the contemporaneous data collected by this program, indicated annual mean density of juvenile *Eucinostomus* spp. in the mouth of Turkey Creek and the adjacent IRL samples were lower than the FIM "river" densities in 2015 and 2016, but very similar in 2017 when very large numbers of juvenile mojarras were found along the western rim of Turkey Creek around the end of the dredging activity in January and February. It is important to remember that these FIM samples were collected across a wider range of habitats throughout Turkey Creek and the St. Sebastian River, and were not geographically restricted to the mouth of Turkey Creek, as in this study. A fine-grained comparison of our Turkey Creek data and FIM data from similar habitats in other creek and river mouths is presented in the next section of this report.

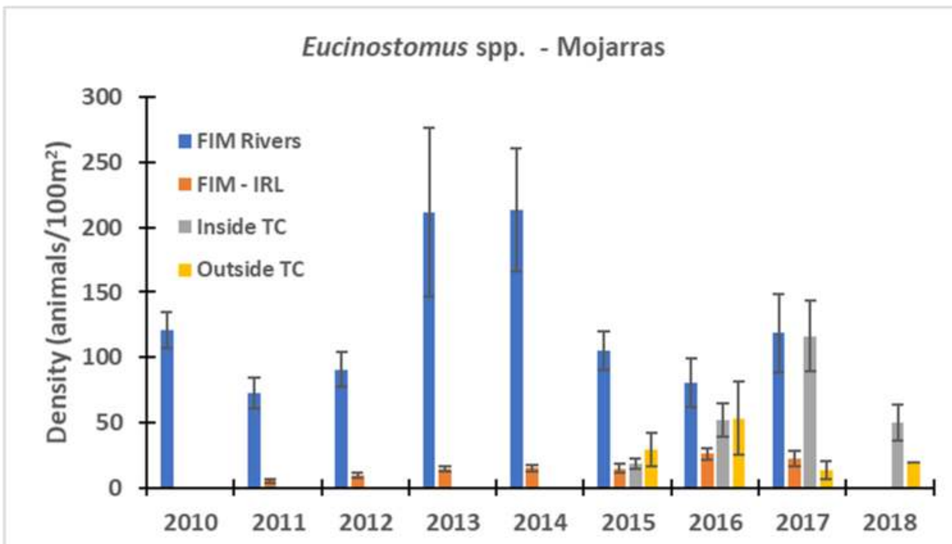


Figure 3.7.5. Annual mean (+/- S.E.) density of juvenile *Eucinostomus* spp. (number of fish/100 m^2) in all samples collected by the FIM program (2010 to 2017) in IRL (open lagoon) stations and "river stations" (i.e., Turkey Creek and St. Sebastian River), and annual density data collected by this program in the mouth of Turkey Creek (Inside TC) and the adjacent IRL (Outside TC) for April 2015 to August 2018.

Diapterus spp.: Although juvenile (15-90 mm) Irish pompano, *Diapterus* spp. were the third most abundant taxon collected during the survey (n=6,871), abundance data were heavily influenced by catches made on only a few dates (Figure 3.7.6). Seven of the highest 8 catches were made in the summer and early fall months of each year, but a single sampling date in January 2017 yielded the highest abundance totals for the entire study. These fish were mixed in among the largest catches of *Eucinostomus* spp., described above, suggesting that both taxa were responding to the environment in the same way.

Juvenile *Diapterus* occurred in 154 samples throughout the sampling period, but only 54 samples exceeded a threshold density of 20 fish 100m^{-2} for inclusion in the peak abundance analysis. This analysis indicated a significant difference among stations (Figure 3.7.7.a; G-Test; G=2,837; df=4; p<0.001), with greatest abundance taken at one of the sites along F-W, the western shoreline of Turkey Creek. Although more samples with high abundances of *Diapterus* spp. were taken at that station than any other station, there were no statistically significant differences in number of peak catches among stations (G-Test; G=4.19; df=4; p= 0.380). The concentration of peak abundances along the western shoreline was similar to that of *Eucinostomus* spp, but *Diapterus* spp. were concentrated in the shallow depth stratum (Figure 3.7.7.b; G-Test; G=7,887; d =1; p<0.001).

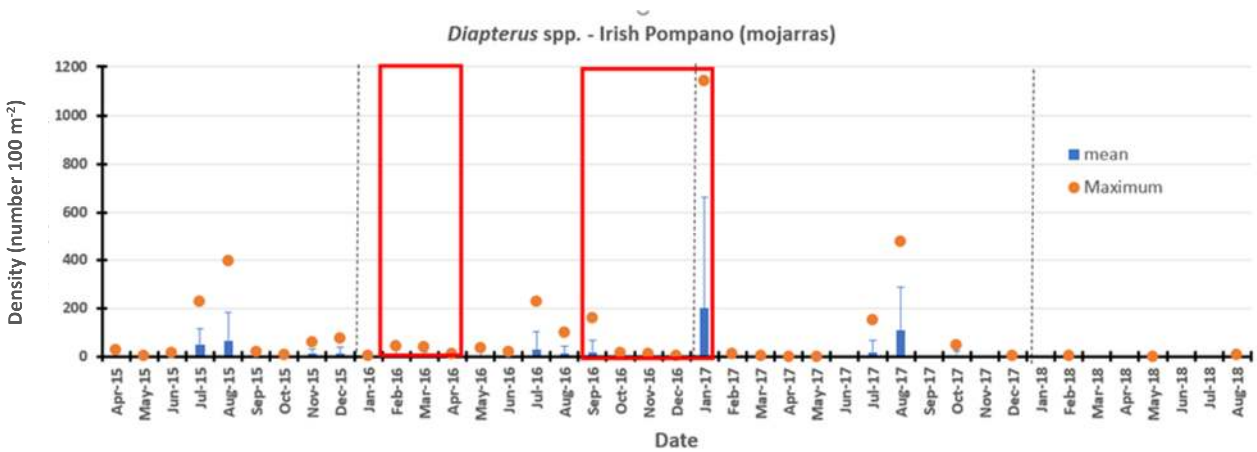
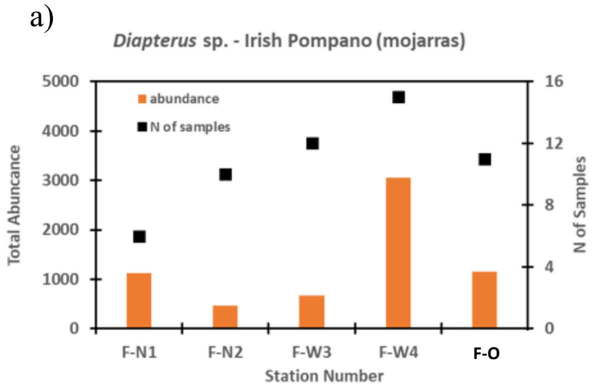


Figure 3.7.6. Mean (+/- S.D.) and maximum densities (number 100m^{-2}) of juvenile Irish Pompano (*Diapterus* spp.) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Vertical dashed lines separate years.

a)



b)

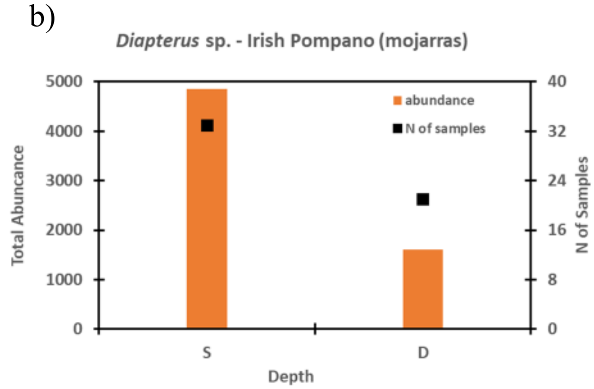


Figure 3.7.7. a) Total abundance of juvenile Irish Pompano, *Diapterus* spp., taken in peak catches (>20 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile *Diapterus* spp. taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

According to FIM summary data, juvenile *Diapterus* spp. were far more abundant in riverine habitats than in the open IRL from 2010 to 2017 (Figure 3.7.8). Mean densities in these riverine habitats had an interannual variation of over 200% during that time range. The mean densities from our sampling from 2015 and 2016 inside and adjacent to Turkey Creek were much lower than the FIM riverine sampling. However, 2017 FIM data show an annual decrease in *Diapterus* abundance in their river samples, while our samples showed an increase in density, driven largely by the pulse in abundance in the January samples.

Feeding habits of juvenile *Diapterus* spp. collected in and around Turkey Creek (Table 3.7.2) show that these fish also fed on epibenthic crustaceans as well as infaunal polychaetes and small bivalves. None of these prey existed in muck habitats within Turkey Creek, but did occur following dredging at the sample sites (Figure 3.6.9).

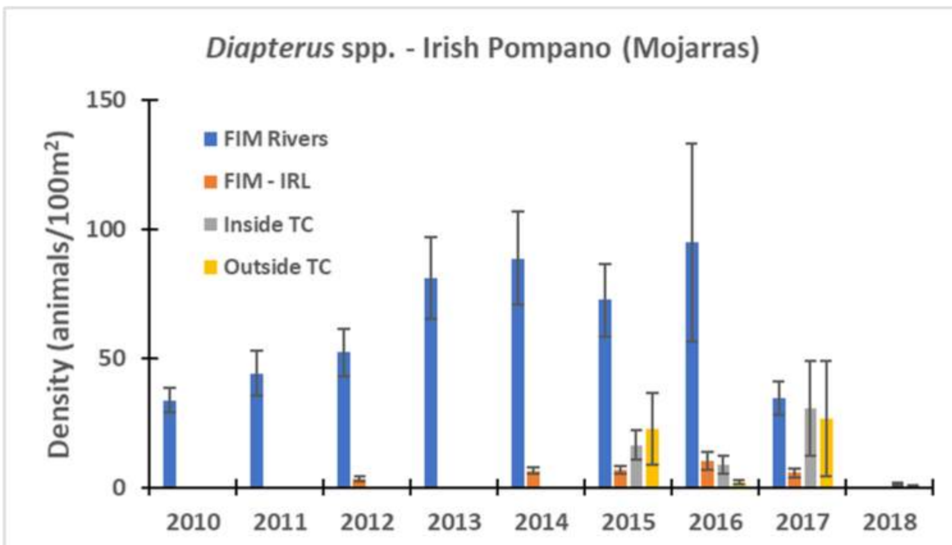


Figure 3.7.8. Annual mean (+/- S.E.) density of juvenile *Diapterus* spp. (number of fish 100 m^{-2}) in all samples collected by the FIM program (2010 to 2017) in IRL (open lagoon) stations and "river stations" (i.e., Turkey Creek and St. Sebastian River), and by this program in the mouth of Turkey Creek (TC-Creek) and adjacent IRL (TC – IRL) for 2015, 2016 and January – May 2017.

<u>Sciaenidae- Drums</u>: Seven species of juvenile drums were collected inside and adjacent to the mouth of Turkey Creek before, during and after dredging operations. Five of those taxa have been selected for presentation here: red drum (*Sciaenops ocellatus*); sea trout (*Cynoscion* sp.); Atlantic croaker (*Micropogonias undulatus*); silver perch (*Bairdiella chrysoura*); and kingfish (*Menticirrhus* spp.). The other two drum species that were collected are spot (*Leiostomus xanthurus*) and black drum (*Pogonias cromis*), taken only in very low numbers (Table 3.7.2).

Micropogonias undulatus: Juvenile Atlantic croaker were the second most abundant demersal fish (n=10,272) collected in and around the mouth of Turkey Creek, but that abundance ranking was driven by catches made during only 2 months throughout the entire study (Figure 3.7.9). The juveniles collected during the two peaks in February 2016 and 2018 were primarily recently-settled individuals ranging from 15-25 mm in length. Over 5000 additional smaller (10-15 mm) members of the drum family were also collected, but live fish were released. Subsamples of fish brought back to the laboratory for identification were Atlantic croaker, but since only a small proportion were examined, we are not including those individuals in this analysis.

These juveniles result from fall and winter offshore spawning, with larval ingress through coastal inlets into estuarine nursery habitats (Shenker and Dean 1979; Warlen and Burke 1990). Very high levels of interannual variation in recruitment has been attributed to shifts in coastal thermal regimes that affect migration of spawning adults and wind events, including hurricanes that affect both the offshore thermal regime and larval transport (Montane and Austen 2005).

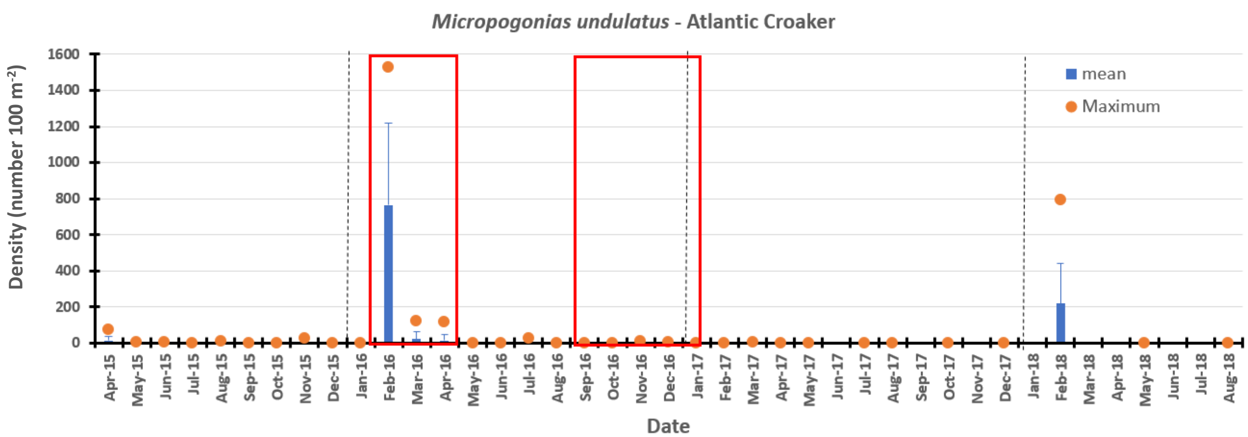
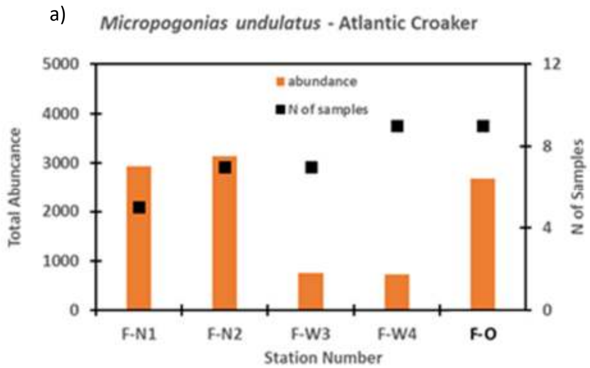


Figure 3.7.9. Mean (+/- S.D.) and maximum densities (number $100m^{-2}$) of juvenile Atlantic croaker (*Micropogonias undulatus*) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region.

Juvenile Atlantic croaker were captured in 51 samples throughout the survey, but only 35 exceeded the 10 fish $100m^{-2}$ threshold selected to define the samples of peak abundance. A significant difference in total peak abundance occurred among stations (Figure 3.7.10a; G-Test = 3238; df=4; $p < 0.001$), with highest densities along the northern shore of Turkey Creek (F-N) and the adjacent Indian River Lagoon (F-O). This pattern contrasts with the mojarra taxa that were more abundant along the western shore of Turkey Creek. Although this species was taken in more samples in shallow seines, significantly higher peak abundance totals occurred in the deeper seine hauls (Figure 3.7.10b; df = 1; G-Test = 935; $p<0.001$).



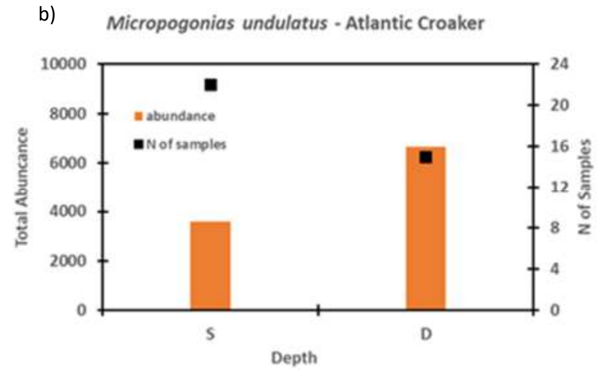


Figure 3.7.10. a) Total abundance of juvenile Atlantic croaker, *Micropogonias undulatus*, taken in peak catches (>20 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile Atlantic croaker taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

Only two major recruitment pulses were noted in sampling at Turkey Creek sampling: February 2016 and February 2018. Presumably these juveniles dispersed across the region later in the spring seasons, but densities in and around Turkey Creek remained low since the initial pulses. FIM collections indicate low mean annual densities across the IRL throughout 2010 to 2017, although riverine habitats typically contained higher densities than did open water habitats in the ecosystem (Figure 3.7.11). 2018 FIM data are not yet available to determine if the second recruitment pulse was seen in other habitats around the IRL.

Stomach contents of juvenile *M. undulatus* collected in Turkey Creek indicate that the small recruits feed primarily on mobile prey: amphipods and planktonic copepods (Table 3.7.2). None of these taxa were found in the muck sites prior to dredging, but overall infaunal abundances showed some increases in some sites after a recovery period following dredging (Figure 3.6.9).

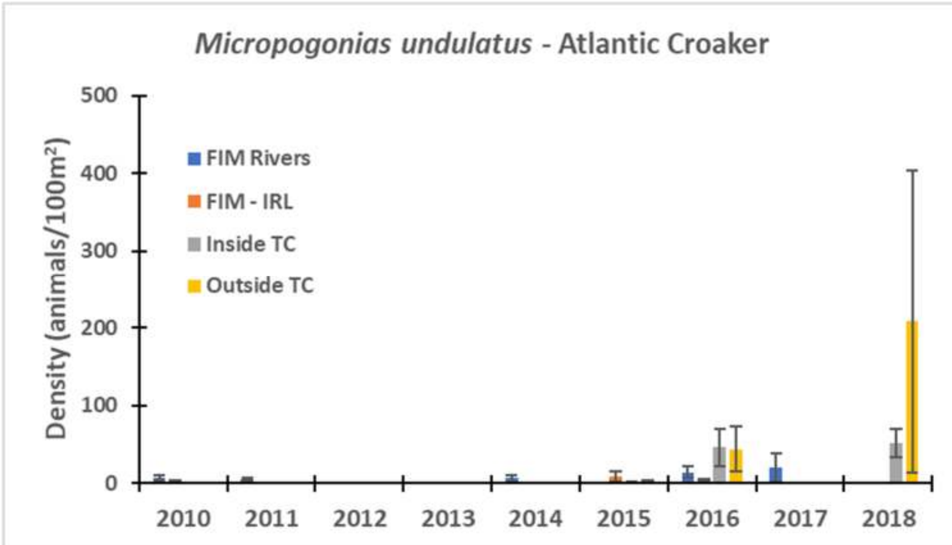


Figure 3.7.11. Annual mean (+/- S.E.) density of juvenile *Micropogonias undulatus* (number of fish 100 m^{-2}) in all samples collected by the FIM program (2010 to 2017) in IRL (open lagoon) stations and "river stations" (i.e., Turkey Creek and St. Sebastian River), and by this program in the mouth of Turkey Creek (TC -Creek) and adjacent IRL (TC – IRL) for 2015 to August 2018.

Sciaenops ocellatus: Red drum support one of the iconic fisheries of the Indian River Lagoon. Adults in many estuaries typically spawn around inlets and nearshore waters during the fall, with larvae developing in coastal waters before ultimately settling into demersal estuarine habitats (Peters et al. 1987; Rooker et al. 1997). The presence of larvae within the northern IRL and Mosquito Lagoon indicate that red drum also spawn within some estuaries (Reyier and Shenker 2007; Reyier et al. 2008). Although they may utilize a wide array of estuarine nursery habitats, Rooker et al. (1998a) determined that seagrass meadows provide better protection from predators than featureless habitats.

A total of 501 juvenile *S. ocellatus* was collected from 36 seine samples throughout the study (Figure 3.7.12). A pulse of new recruits (20-50 mm) was detected in November 2016, during the dredging period, and a school of larger (90-120 mm) juveniles had been collected in spring 2016. Densities at all other times were very low.

Given the relatively few catches and low densities of this species, a threshold of 5 fish 100m^{-2} was established to select 19 samples for peak abundance analysis of microhabitat preferences. Significant differences were found among the stations (Figure 3.7.13a; G-Test = 442; df=4; p< 0.001) and depth ranges (Figure 3.7.13b; G-Test = 246; df=1; p< 0.001), with highest abundances in the shallow hard sand station along the western edge of Turkey Creek (F-W), and in the shallow hard sand station in the Indian River Lagoon (F-O). Given the determination that *S. ocellatus* juveniles have higher survival rates in seagrass habitats, and the only protection from predators that these juveniles could get in Turkey Creek would be the turbid waters, it is likely that these fish either moved into more suitable habitats or were consumed by predators.

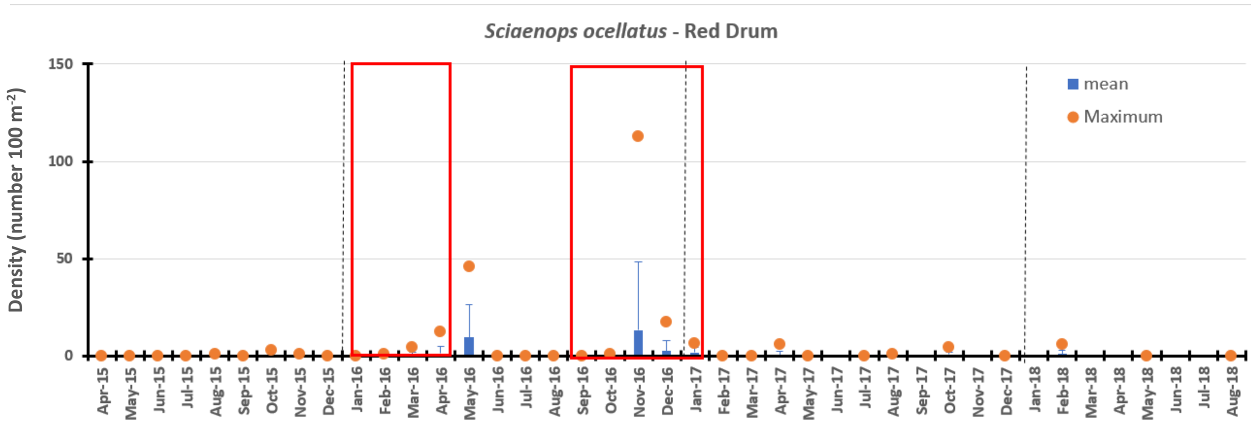


Figure 3.7.12. Mean (+/- S.D.) and maximum densities (number $100m^{-2}$) of juvenile red drum (*Sciaenops ocellatus*) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Dashed vertical lines separate years.

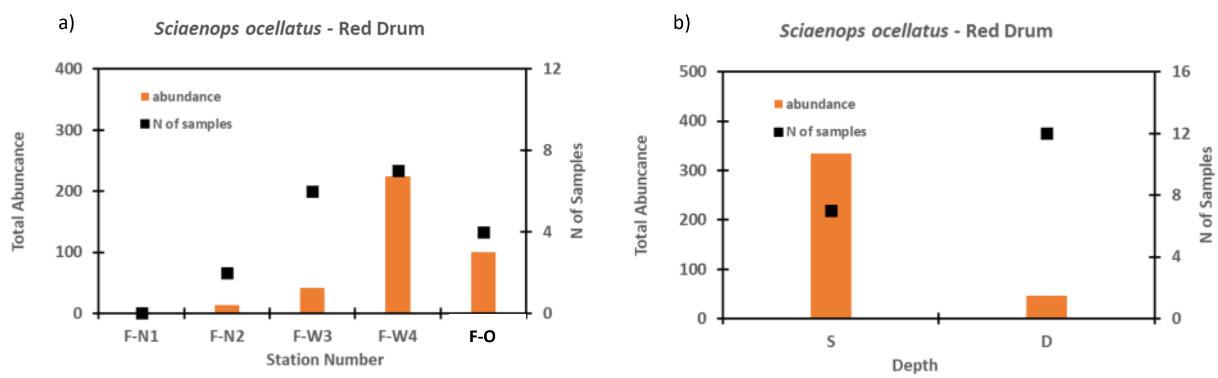


Figure 3.7.13. a) Total abundance of juvenile red drum, *Sciaenops ocellatus*, taken in peak catches (>5 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile *S. ocellatus* taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

Examination of temporal and spatial patterns within the broader context provided by FIM illustrate the importance of creek mouths as juvenile habitats for this species (Figure 3.7.14). FIM data collected over the open IRL, and spanning the riverine habitats, show lower densities than our sampling that focused on the mouth of Turkey Creek and the adjacent IRL, but the fish they collected in the open IRL were generally of larger size than those we found in Turkey Creek. The lack of protective seagrass or other complex structures within Turkey Creek may contribute to the dispersal of populations to the more complex habitats in the IRL. The relatively consistent but low densities observed over 8 years by FIM suggest relatively low levels of interannual variability of recruitment.

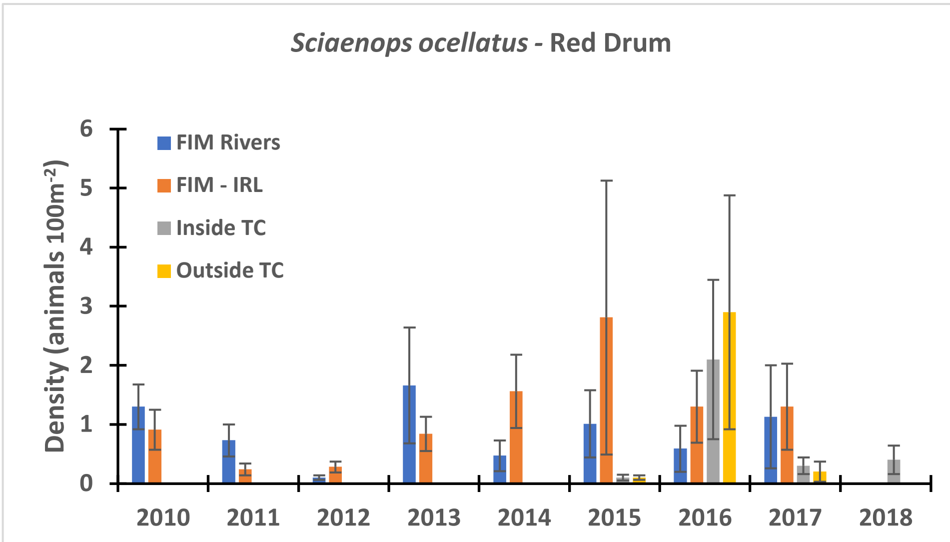


Figure 3.7.14. Annual mean (+/- S.E.) density of juvenile red drum, *Sciaenops ocellatus*, (number of fish 100 m^{-2}) in all samples collected by the FIM program (2010 to 2017) in IRL (open lagoon) stations and "river stations" (i.e., Turkey Creek and St. Sebastian River), and by this program in the mouth of Turkey Creek (TC -Creek) and adjacent IRL (TC – IRL) for 2015 to August 2018.

Juvenile red drum in Tampa Bay fed on small crustaceans such as mysids, amphipods and newly-settled shrimp (Peters and McMichael, 1987). Similar-sized juveniles in Turkey Creek and other areas of the IRL fed on a broad range of epibenthic and pelagic crustaceans, dominated by amphipods, penaeid shrimp, copepods, isopods and grass shrimp (Table 3.7.2; Scripter et al., in prep.). The infaunal taxa within this list of prey species were not present in muck habitats prior to dredging, but amphipods and isopods were among the infaunal taxa that increased in abundance following dredging.

Cynoscion sp.: Juvenile sea trout (primarily *C. nebulosus*) support another of the most important fisheries of the Indian River Lagoon. These fishes typically spawn during the summer months within the IRL (Johnson and Seaman, 1986; McMichael and Peters, 1989; Reyier and Shenker 2007; Reyier et al. 2008).

Juvenile sea trout were collected in only 61 samples throughout the sampling period. Of the total of 508 fish collected, they occurred in high densities in only three pulses: November 2015 and July 2016, and July/August 2017 (Figure 3.7.15). Only 12 samples exceeded a threshold of 5 fish 100m^{-2} for inclusion into a peak abundance analysis of habitat and depth preferences. Significant differences were observed for station (Figure 3.7.16a; G-Test = 967; df=4; $p < 0.001$) and depth ranges (Figure 3.7.16b; G-Test = 137; df=1; $p < 0.001$), with almost all fish taken in the shallow water habitat outside the mouth of Turkey Creek (F-O). Very few fish were captured inside Turkey Creek.

Juvenile sea trout are generally considered to prefer living in and around seagrass beds. The seagrass provides protection from predators and ready access to the planktonic prey (copepods) of small juvenile sea trout and epibenthic crustaceans (e.g. mysids and amphipods) of larger juveniles (McMichael and Peters, 1989). Given the lack of seagrass within the mouth of Turkey Creek, their relative absence inside the creek is not surprising.

Although those 3 pulses of juvenile sea trout occurred during single months in 2015, 2016 and 2017, those high relatively high densities generated mean annual densities at the stations over hard sand outside the mouth of Turkey Creek that were dramatically larger than the mean annual densities of juveniles sampled by FIM throughout the open IRL and riverine habitats from 2010 to 2017 (Figure 3.7.17). These juveniles presumably dispersed from the near-shore region into adjacent seagrass habitats in the IRL.

Juvenile sea trout fed primarily on mobile prey, including penaeid shrimp, mysids, amphipods and other fishes (Table 3.7.2). None of these prey are associated with muck habitats. Although the removal of muck may increase the abundance of these prey, the lack of seagrass inside Turkey Creek presumably will limit the value of this habitat as a nursery for sea trout.

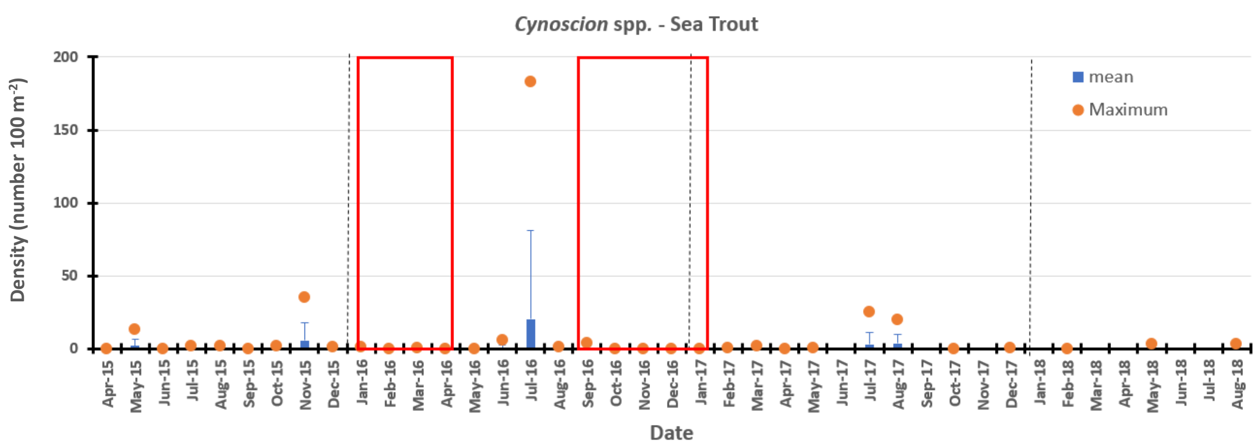
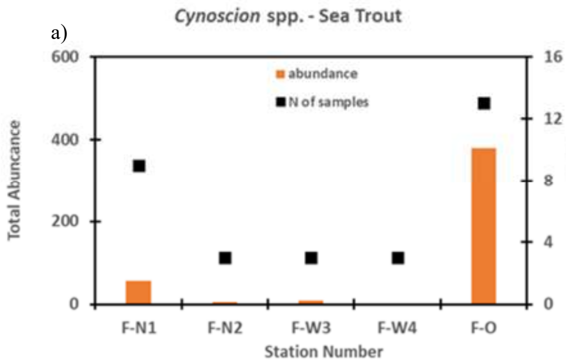


Figure 3.7.15. Mean (+/- S.D.) and maximum densities (number 100m^{-2}) of juvenile sea trout (*Cynoscion* spp.) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Dashed vertical lines separate years.



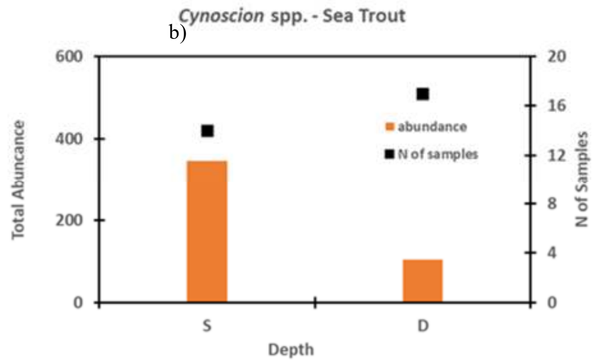


Figure 3.7.16. a) Total abundance of juvenile sea trout, *Cynoscion* spp., taken in peak catches (>5 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile *Cynoscion* spp. taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

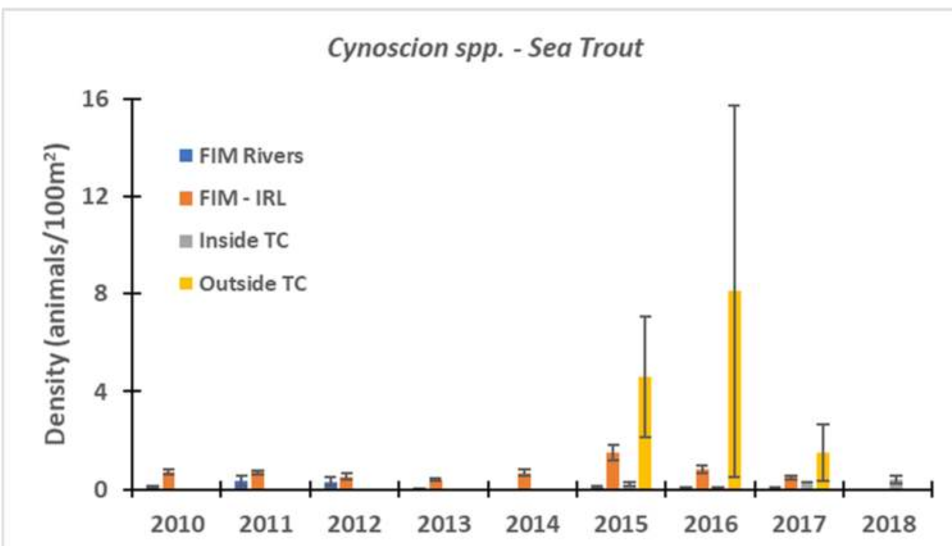


Figure 3.7.17. Annual mean (+/- S.E.) density of juvenile sea trout, *Cynoscion* spp. (number of fish 100 m^{-2}) in all samples collected by the FIM program (2010 to 2017) in IRL (open lagoon) stations and "river stations" (i.e., Turkey Creek and St. Sebastian River), and by this program in the mouth of Turkey Creek (TC-Creek) and adjacent IRL (TC – IRL) for 2015 to August 2018.

Bairdiella chrysoura: Silver perch spawn in early spring in estuarine channels, with larvae and early juveniles generally occurring in mesohaline seagrass, sand and oyster habitats and adults moving into higher salinity regions (Rooker et al. 1998b; Reyier and Shenker 2007; Hanke et al

2013). In our study, a total of 4,717 juvenile *B. chrysoura* were taken in 81 samples, primarily during late spring through mid-summer in the mouth of Turkey Creek and the adjacent IRL (Figure 3.7.18). These fish were captured on more occasions and in higher numbers in 2015 and 2016 than in spring 2017. For peak abundance analysis, 37 samples exceeded a threshold density of 15 fish $100m^{-2}$ for inclusion in the analysis. Although fish were taken at all stations in the region, there was a significant difference among stations (Figure 3.7.19a G-Test; G=455; df=4; p< 0.001) and depth ranges (Figure 3.7.19b; G-Test; G = 397; df=1; p< 0.001), with highest abundances at the eastern-most station in F-N and in the deeper depth strata.

These juveniles appear to have concentrated in the narrow habitat around river mouths, since they were rarely captured in either open IRL and riverine systems by the FIM sampling program (Figure 3.7.20). They fed primarily on amphipods, with tanaids, mysids and fish comprising a smaller portion of their diet (Table 3.7.2). Data from Waggy et al. (2007) indicate that the silver perch in the northern Gulf of Mexico fed primarily on a similar array of epibenthic crustaceans.

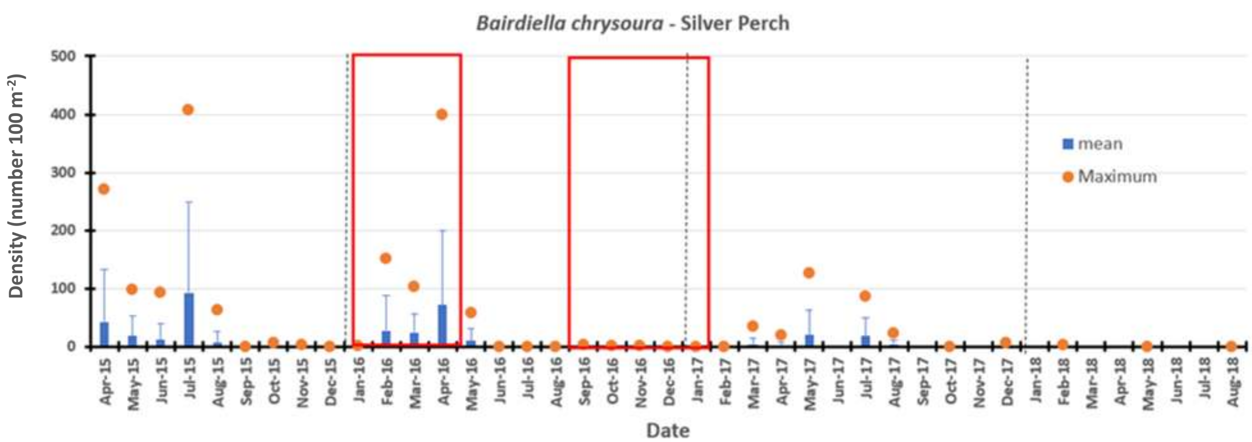


Figure 3.7.18. Mean (+/- S.D.) and maximum densities (number $100m^{-2}$) of juvenile silver perch (*Bairdiella chrysoura*) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Vertical dashed lines separate years.

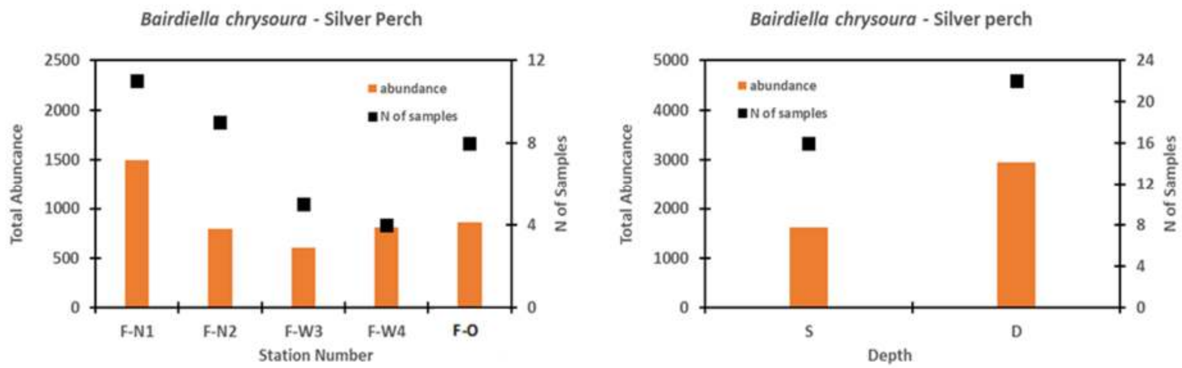


Figure 3.7.19. a) Total abundance of juvenile silver perch, *Bairdiella chrysoura.*, taken in peak catches (>5 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile *B. chrysoura* taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

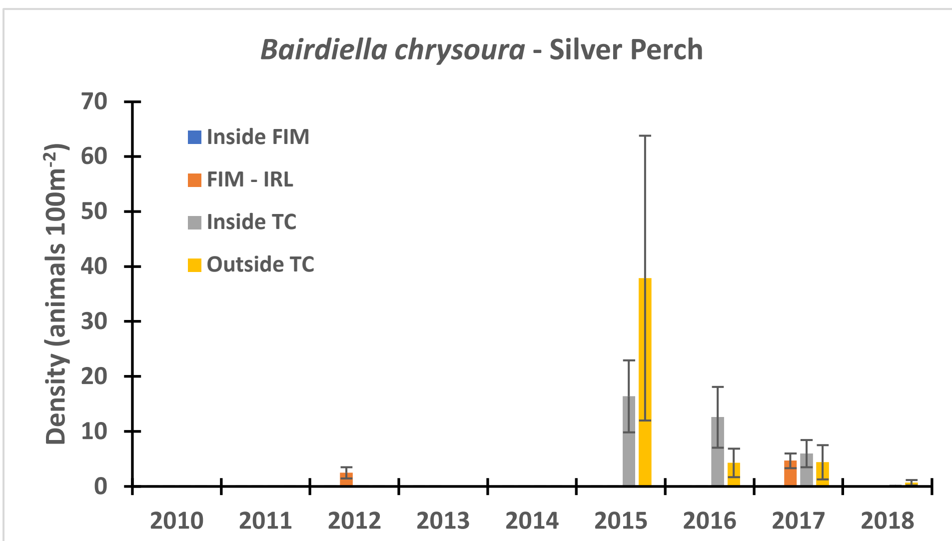


Figure 3.7.20. Annual mean (+/- S.E.) density of juvenile silver perch, *Bairdiella chrysoura*, (number of fish 100m^{-2}) in all samples collected by the FIM program (2010 to 2017) in IRL (open estuary) stations and "river stations" (i.e., Palm Bay of Turkey Creek and St. Sebastian River, respectively), and by this program in the mouth of Turkey Creek (TC-Creek) and adjacent IRL (TC – IRL) for 2015 to August 2018.

Menticirrhus americanus: Southern kingfish are a small member of the drum family that support a recreational fishery in IRL and coastal waters. Ichthyoplankton surveys of the northern IRL showed that this species spawns within the IRL and Banana River Lagoon, primarily north of the city of Merritt Island (Reyier and Shenker 2007; Reyier et al. 2008). Juveniles occur through the IRL ecosystem, including the region adjacent to Turkey Creek (Figure 3.7.21), where 1,309 individuals were collected in 48 samples. They were most abundant during summer 2016, during the seasonal hiatus from dredging activity required to minimize impacts on manatee populations. Since this species is not a dominant taxon in FIM surveys, data on this species are not presented in FIM summary report.

Peak abundance analysis shows that virtually all individuals were captured outside the mouth of the creek, with the highest proportion taken in the shallow tows (Figure 3.7.22). These juveniles fed primarily on epibenthic mysids and other crustaceans.

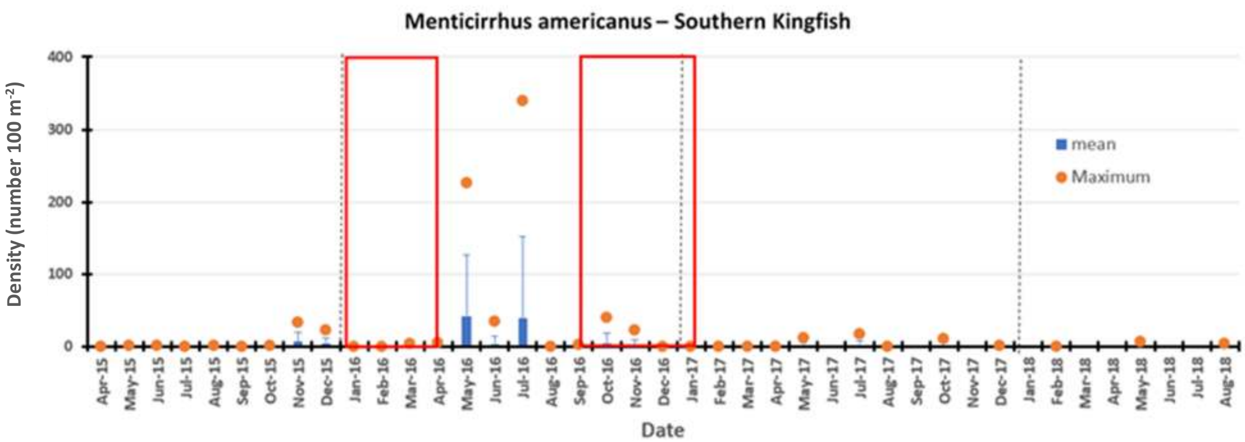


Figure 3.7.21. Mean (+/- S.D.) and maximum densities (number 100m^{-2}) of juvenile southern kingfish (*Menticirrhus americanus*) taken at 8 stations within the mouth of Turkey Creek and 2 stations in the Indian River Lagoon outside the mouth of Turkey Creek. Red boxes indicate times of dredging operations occurring in the region. Vertical dashed lines separate years.

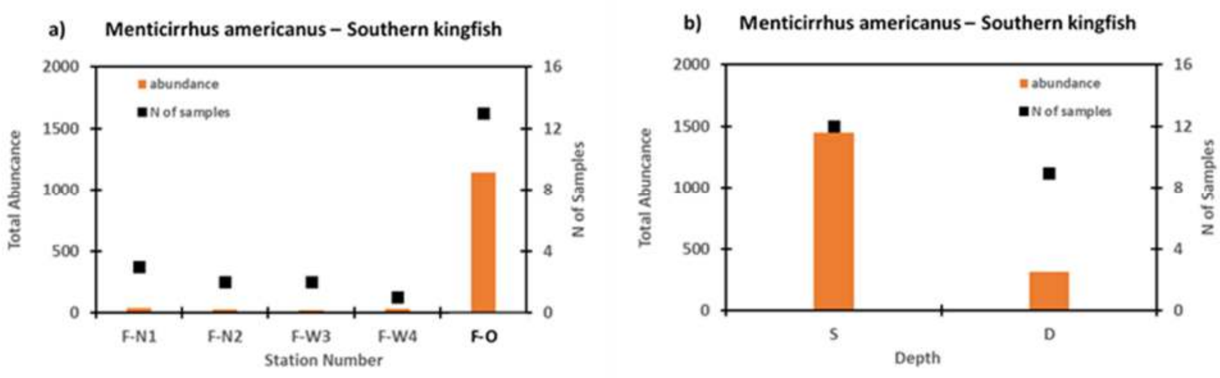


Figure 3.7.22. a) Total abundance of juvenile southern kingfish, *Menticirrhus americanus*, taken in peak catches (>5 fish 100m^{-2}), and number of peak catches, at sampling stations along the northern shore (F-N) and western shore (F-W) of Turkey Creek and just outside the mouth of Turkey Creek (F-O). b) Total abundance of juvenile *M. americanus* taken in peak catches at Shallow (0.25-0.5m) and Deep (0.5-1.0m) seine depths in the sampling region.

Fine-grained Comparisons Between Turkey Creek Sites and Similar Habitats

The intense juvenile fish sampling program conducted by FIM over the last 25 years provides an extraordinary data base on the temporal and spatial distributions of fishes throughout the Indian River Lagoon. Annual data summaries calculated the mean (± S.D) abundance of dominant taxa in each of the main basins of the IRL ecosystem. For the interannual comparisons presented above, these annual summary data sets were used to place our Turkey Creek fish abundance data within the context of the open IRL habitats and tributary (river) habitats in the wider ecosystem. While these data do provide a valid comparison for our data collected within and adjacent to the mouth of Turkey Creek, the FIM data also present an opportunity for a finer-grained comparison of similar river and river mouth habitats.

FIM sampling throughout most of the IRL is based upon the use of stratified random sampling protocols to select sampling sites in each region. Some areas, such as the Sebastian River and the upstream portion of Turkey Creek were targeted for higher-intensity sampling than most other areas in the ecosystem. For this fine-grained analysis, the entire FIM Access data base was sorted to select data collected from 2008 to 2018 from 17 blocks within the overall IRL sampling grid: Crane Creek, Turkey Creek and adjacent IRL, and Sebastian River and adjacent IRL (Figure 3.7.23).

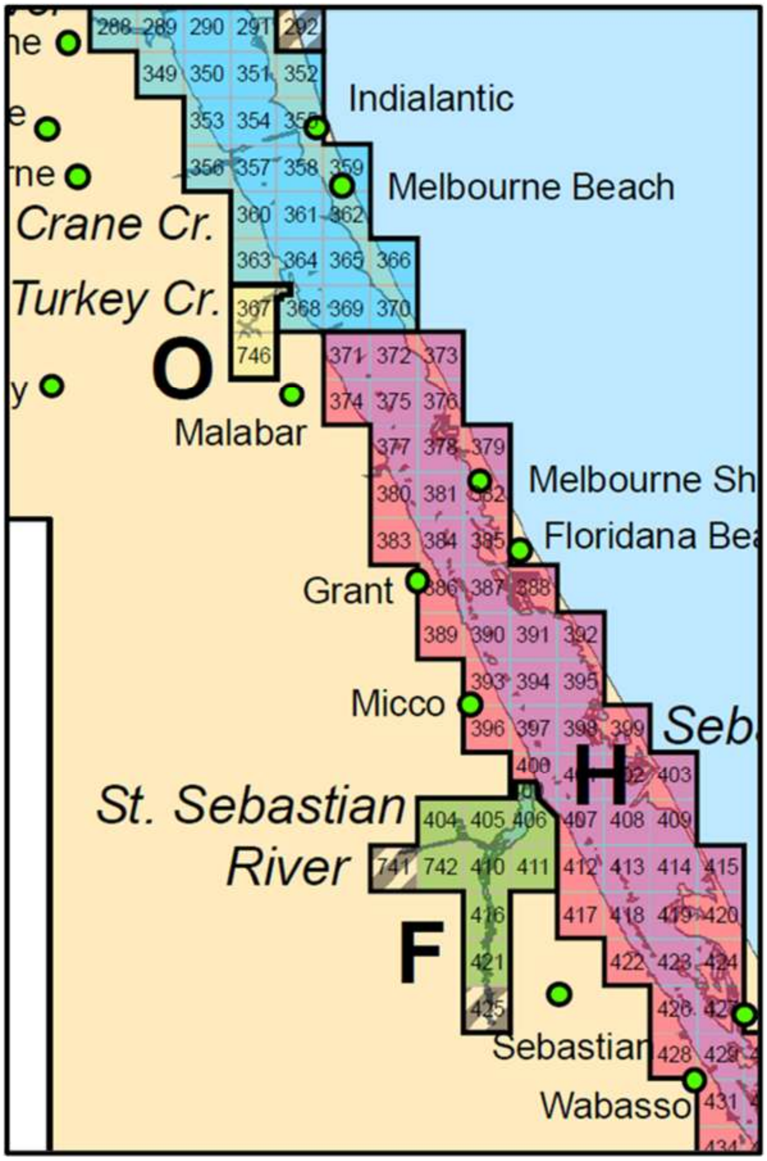


Figure 3.7.23. Sampling grid used the Florida Fisheries and Wildlife Commission's Fisheries Independent Monitoring Program (FIM) to guide fish sampling efforts. Blocks that contain Crane Creek, Turkey Creek and Sebastian River, and adjacent Indian River Lagoon habitats were selected for analysis.

Examination of the sampling effort within each block indicated that the Sebastian River and Turkey Creek, upstream of the U.S. 1 bridge, received focused and intense sampling effort, generally 2-3 times/month during 2008-2011, and about once/month thereafter, while the other blocks received lower sampling levels similar to other areas of the IRL (Table 3.7.4). Accordingly, Block 367 (Turkey Creek west of the bridge), Block 400 (mouth of the Sebastian River), and Block 406 (the broad embayment just inside the mouth of the Sebastian River) were selected for analysis. Each of these sites was sampled 158-198 times over the 11 year period. Although Block 400 spanned both Sebastian River and Indian River Lagoon habitats, only samples from within the river (n=158) were included in the analysis, while IRL samples (n=16) were excluded. Block 368 included our Turkey Creek sampling sites, but only 2 seine hauls were made during 2001-2018 in the area where we frequently sampled inside the mouth of the creek, and 18 hauls were made in the adjacent IRL.

Fish abundance data from each block in the sorted database were compiled for each of the dominant taxa using the statistical package "R." Abundance data were converted to density measurements, and the 3 dominant demersal juvenile species were selected for analysis: Mojarras (*Eucinostomus* spp.), Irish pompano (another mojarra species; *Diapterus* sp.), and Atlantic croaker (*Micropogonias undulatus*).

Each of these three species showed very wide temporal and spatial variations in abundance, reflecting annual temporal variability as well as variability on a very small spatial scale even among repeated samples being made the same month in the same block. This variability presumably reflects fluctuations in successful spawning and larval recruitment, the schooling behavior of juvenile fishes, and their mobility that enables them to respond to fluctuations in hydrological and biological conditions within the region.

Examination of the raw data is perhaps the best way to initially visualize this extreme variation.

For juvenile *Eucinostomus* spp. mojarras (Figure 3.7.24), plotting the density from each seine sample for each month over the 2008-2018 time period, shows both dramatic monthly and interannual variability, as well as spatial variability within blocks and among blocks. Five trends are visually apparent:

- Each of the years generally had 1-3 samples with much higher densities than the other samples collected that year.
- Peak samples did not indicate patterns of high spatial coherence in abundance in any month; i.e. if multiple samples were taken in the same month in the same block, the other samples in the block generally had much lower densities than the peak sample.
- The locations of the peak catches in different years varied among the 3 sampling blocks in upper Turkey Creek and the Sebastian River.
- Some years (2016, 2018) had consistently low densities, perhaps reflecting low larval recruitment across the system.

- However, 2012 and 2014 also had low densities across the region, with the exception of a single large catch in each year. These mobile schooling species may have concentrated on one site within that block during that month, for reasons that we cannot identify.

Comparison of the data patterns in the FIM data set can be summarized beyond simple visual data presentations. Four metrics were computed for analysis: 1) Frequency of Occurrence (%FO) reflects the frequency of samples in which at least one fish was taken; 2) %TH is the frequency of samples that exceeded the density threshold utilized for inclusion of a sample in the microhabitat assessment; 3) Mean density of the taxon throughout the year, and 4) Maximum density of the taxon throughout the year.

Pooling data from the entire 2008-2018 time period (Figure 3.7.25) for those three sampling blocks indicate that the mean annual Frequency of Occurrence (%FO) of *Eucinostomus* spp. in the seine hauls varied significantly among blocks, from a mean of 65.1% ($\pm$ S.D. = 19.1) in Block 367 (the upstream habitats of Turkey Creek) to 93.3% (+ S.D. = 8.4%) in Block 400 at the mouth of the Sebastian River (2 way ANOVA, $p<0.014$), but not among years ($p>0.1$). Conversely, there were no significant differences among blocks or years for %TH or mean density (2 way ANOVA, $p>0.05$). Highest annual peaks were found in Block 406 on 6 of the 11 years, in Block 400 in three years, and in Block 367 in 2 years.

Comparison of the 11 year FIM data set with the data collected during our 3 years of sampling at the mouth of Turkey Creek show some similarities and differences with the upstream Turkey Creek and the two Sebastian River blocks (Figure 3.7.25).

- Juvenile *Eucinostomus* density had a wide range of monthly and interannual variation within the mouth of Turkey Creek (Figure 3.7.3), as it did at the FIM stations.
- The mean annual %FO in samples at the mouth of Turkey Creek (82.3%) was within the range of values for %FO (65.1 to 93.3%) in the FIM data set that covered a longer time frame and broader range of habitats.
- The mean annual frequency of samples exceeding the density threshold selected for microhabitat characterization (29.9 fish/100m2) in the mouth of Turkey Creek (27.3%) was higher than the mean annual %TH values in the three FIM sampling blocks (18.2 to 21%).
- The highest %TH values in Turkey Creek occurred during winter 2016/2017, during and after the completion of the dredging effort.

	South of Eau Gallie River		Crane Creek			Turkey Creek				Sebastian River							
Grid #	**349**	**350**	**356**	**357**	**360**	**364**	**367**	**746**	**368**	**400**	**404**	**405**	**406**	**407**	**410**	**416**	**742**
Year	IRL	IRL	Entire Creek	IRL	IRL	Mouth of Turkey Creek/ IRL	West Turkey Creek	Turkey Creek head-waters	IRL	Sebas-tian River /IRL	N fork close to weir	N fork at branch	Mouth of SR	IRL S of SR	S fork	S fork at branch	N fork
2008	1	1	1	1	0	0/3	33	1	7	24/4	30	26	29	0	21	22	9
2009	3	2	2	1	0	2/2	16	0	10	24/3	32	22	23	3	29	20	8
2010	0	2	0	0	2	0/2	30	1	9	21/0	39	20	29	1	20	21	8
2011	0	0	1	0	2	0/1	18	0	4	13/1	20	14	20	3	14	12	7
2012	0	1	1	2	1	0/0	6	3	7	8/1	18	13	14	0	9	10	4
2013	1	0	0	1	2	0/4	16	2	4	12/1	16	16	12	1	11	10	2
2014	1	0	0	2	2	0/0	19	1	5	11/1	16	12	14	1	10	11	4
2015	2	0	1	2	2	0/0	17	0	7	12/2	17	15	12	2	12	10	3
2016	2	2	1	1	0	0/2	16	1	6	12/0	18	11	16	2	12	12	5
2017	1	3	2	2	2	0/0	11	0	9	10/3	16	11	14	1	14	11	3
2018	1	5	2	3	3	0/4	16	1	10	11/0	12	14	14	2	12	12	8
Total	12	16	11	15	16	2/18	198	10	78	158/16	234	174	197	16	164	151	61

Table 3.7.4. Fish sampling effort (number of seine hauls) made each year by the Florida Fisheries and Wildlife Commission's Fisheries Independent Monitoring Program (FIM) in selected habitats in the Indian River Lagoon ecosystem.

- Mean densities (15.0-24.8/100m^2) and peak densities (166.0 to 325.7/100m^2) among all sites were very similar, with the exception of a single very large catch (1204/100 m^2) of juvenile *Eucinostomus* taken in March 2017 along the western shoreline of the mouth of Turkey Creek.
- Although the highest %TH and peak densities that occurred at our Turkey Creek stations, in winter 2016/2017, the densities of juvenile *Eucinostomus* in the FIM-sampled blocks were very low during those months. This could reflect either local recruitment and environmental conditions that concentrated new juveniles at the mouth of Turkey Creek, or that the dredging activities in the region displaced juveniles that could have been more widely dispersed across the habitat into a narrow band in shallow water near the shoreline. Those high densities were not observed during subsequent months.

A similar approach can be used to evaluate the distribution and abundance of juvenile Irish pompano (*Diapterus* sp.) as determined by a decade of FIM sampling in 2 blocks in the Sebastian River and one block in the upper Turkey Creek, and our sampling for 3 years in the mouth of Turkey Creek. Examination of the time-series of catch data from 2008-2018 (Figure 3.7.26) again shows very wide spatial, monthly and interannual variations in fish density, and a few very large catches dominating each year.

The peak catch, made in October 2016 in Block 406, exceeded 4200/100m2. This density was twice that of the second peak catch, and much higher than the mean densities calculated for each block, but the overall data set from the FIM blocks did not show significant temporal or spatial differences in density (2-way ANOVA, $p>0.1$). However, comparisons with our data from the mouth of Turkey Creek (Figures 3.7.6) indicate that juvenile *Diapterus* did not utilize this habitat as commonly as they did further upstream in FIM Block 367, or in the Sebastian River (Figures 3.7.8 and 3.7.26). These upstream habitats can be reached only by juveniles passing through the mouth of Turkey Creek, so fish collected in the mouth of Turkey Creek may be simply be transiting that habitat.

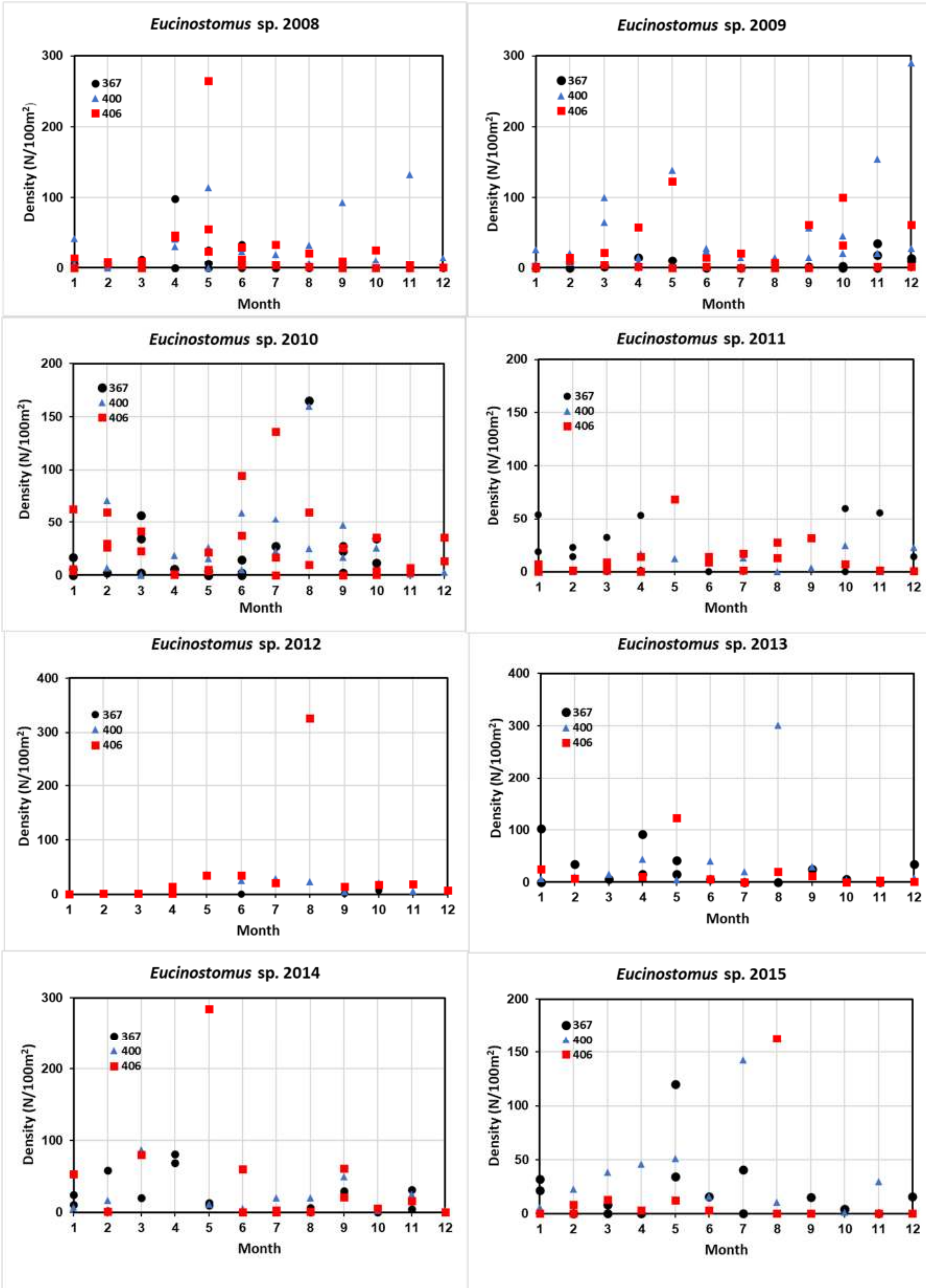


Figure 3.7.24. Monthly densities of juvenile mojarras (*Eucinostomus* spp.) captured in seine hauls by the Fisheries Independent Monitoring program, 2008-2018, from three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River. Note the variation in the Y-axis scale.

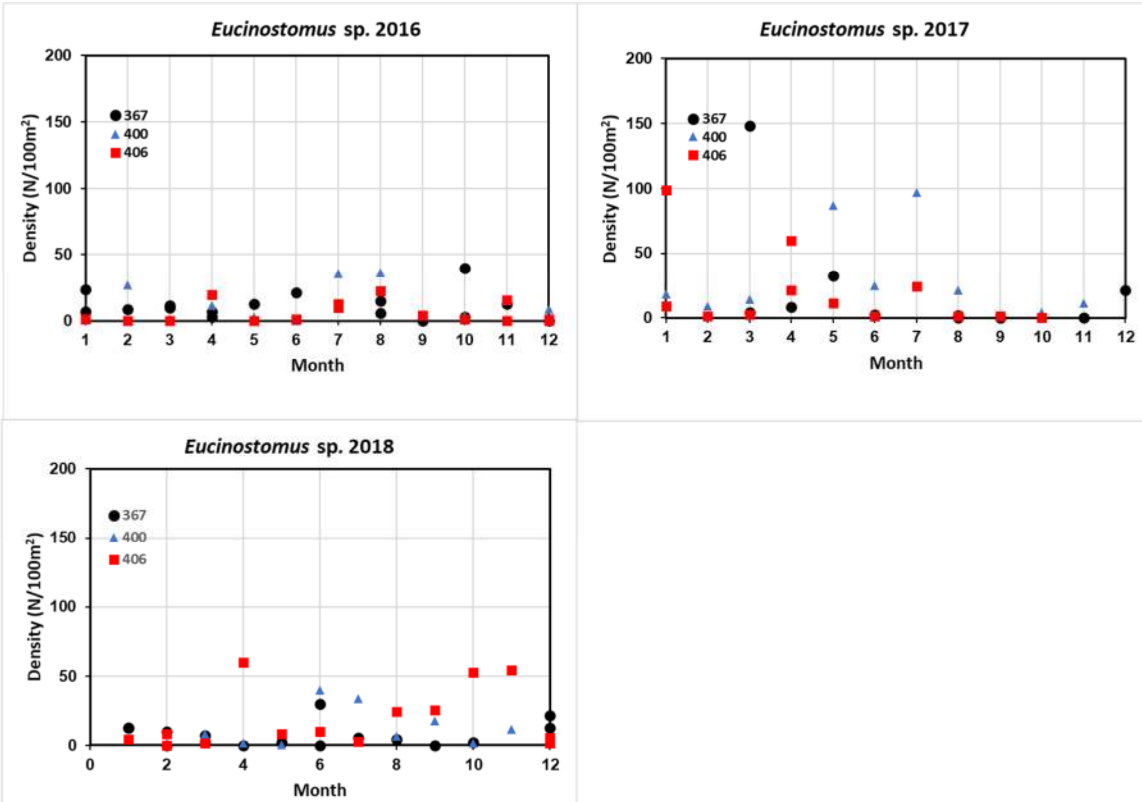


Figure 3.7.24 (continued)

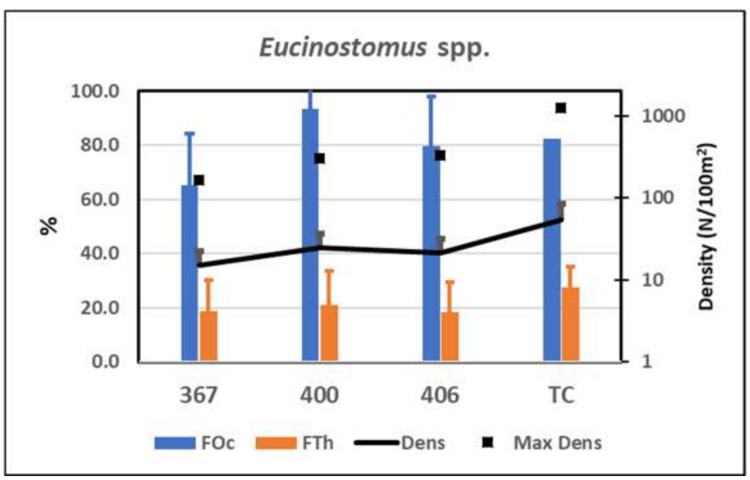


Figure 3.7.25. Mean (+ S.D.) values for juvenile mojarra (*Eucinostomus* spp.) annual frequency of occurrence (%FO), annual frequency of samples exceeding the density threshold of 29.9 fish/100m2 (%TH), annual density (Dens) and annual maximum density of catches made by the Fisheries Independent Monitoring Program in each of three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River from 2008 to 2018, and data from sampling in this program at the mouth of Turkey Creek (TC) from 2015 to 2018.

Atlantic croaker (*Micropogonias undulatus*) was the third species that dominated both the FIM blocks and Turkey Creek mouth data sets. This species spawns offshore in late fall and winter, with larvae entering inlets and moving throughout estuarine ecosystems. The FIM data clearly show this winter seasonality in recruitment in many of the years between 2001-2018 (Figure 3.7.28), as does the Turkey Creek mouth data from 2015-2018 (Figure 3.7.9). The distinct patchiness of the samples suggest that these juveniles appear to stay within demersal schools, or concentrate within small portions of a habitat. This very patchy distribution makes it impossible to determine if the lack of a large catch in any year reflects low larval recruitment of the species, or that the sampling simply missed the presence of schooling or aggregating fish.

This species also showed specific preference for river-mouth habitats rather than migrating upstream (Figure 3.7.29). The FIM data set showed significant differences in %FO, %TH and mean density among habitats (2-way ANOVA, $p<0.01$), with few fish utilizing the upstream habitat in Turkey Creek (Block 367). This reliance on river-mouth habitats corresponds closely with that observed with our data from the mouth of Turkey Creek in 2005-2018, and the lack of fish in any upstream river habitat sampled by FIM (Figure 3.7.11).

Summary:

Comparison of an 11-year time-series provides insight into the recruitment dynamics and habitat utilization processes of juvenile demersal fishes collected in and around the mouths of two tributaries in the central Indian River Lagoon. All three of the analyzed species exhibited very high monthly, interannual and spatial variability. This variability presumably reflects variability in annual levels of larval recruitment as well as schooling behavior of the juveniles and their ability to move around habitats in response to hydrological changes and biological characteristics of the habitats.

Although low densities of each taxon were frequently dispersed across the habitat, a few samples during most years contained very high densities of juveniles that thus dominated the temporal and spatial distribution patterns of the catches. Despite this variability, however, several patterns were apparent:

1) Juvenile *Eucinostomus* spp. did form high densities along the western edge of the mouth of Turkey Creek during and shortly after the last dredging effort in the area. Densities of this taxon were low in other regions during these months, indicating that the dredging activity and deepening of the habitat may have temporarily compressed juveniles into a narrow band in shallow water along the shoreline. This aggregation dispersed by spring-time, and was not seen in subsequent sampling.
2) Juvenile *Diapterus* sp. were more abundant in upstream habitats in Turkey Creek and the Sebastian River than they were in the mouth of Turkey Creek. This species appears to prefer the upriver habitats, and their occurrence in the mouth of Turkey Creek may reflect their transit of this habitat as they move upriver.
3) Conversely, juvenile *Micropogonias undulatus* preferred river mouth and open embayment habitats rather than upstream habitats. Their distribution was extremely

patchy in both time and space, again possibly reflecting the influence of recruitment, schooling, and behavioral responses to environmental attributes.

Turkey Creek Seagrass Transplants

Although the mouth of Palm Bay does not currently have viable seagrass beds, long-time residents report that seagrasses were indeed present decades ago. In an attempt to determine if the habitat could indeed support seagrasses, Sea and Shoreline Inc. conducted a transplant experiment beginning in December 2017-January 2018. Five antiherbivore cages were placed along the northern shore of the bay, at approximate depths of 0.75 m. Cultivars of *Halodule wrightii* and *Ruppia maritima* were transplanted into the cages, and their survival and growth were monitored. Although the seagrasses did indeed grow through early summer period, they disappeared by early fall following extensive periods of fresh water runoff.

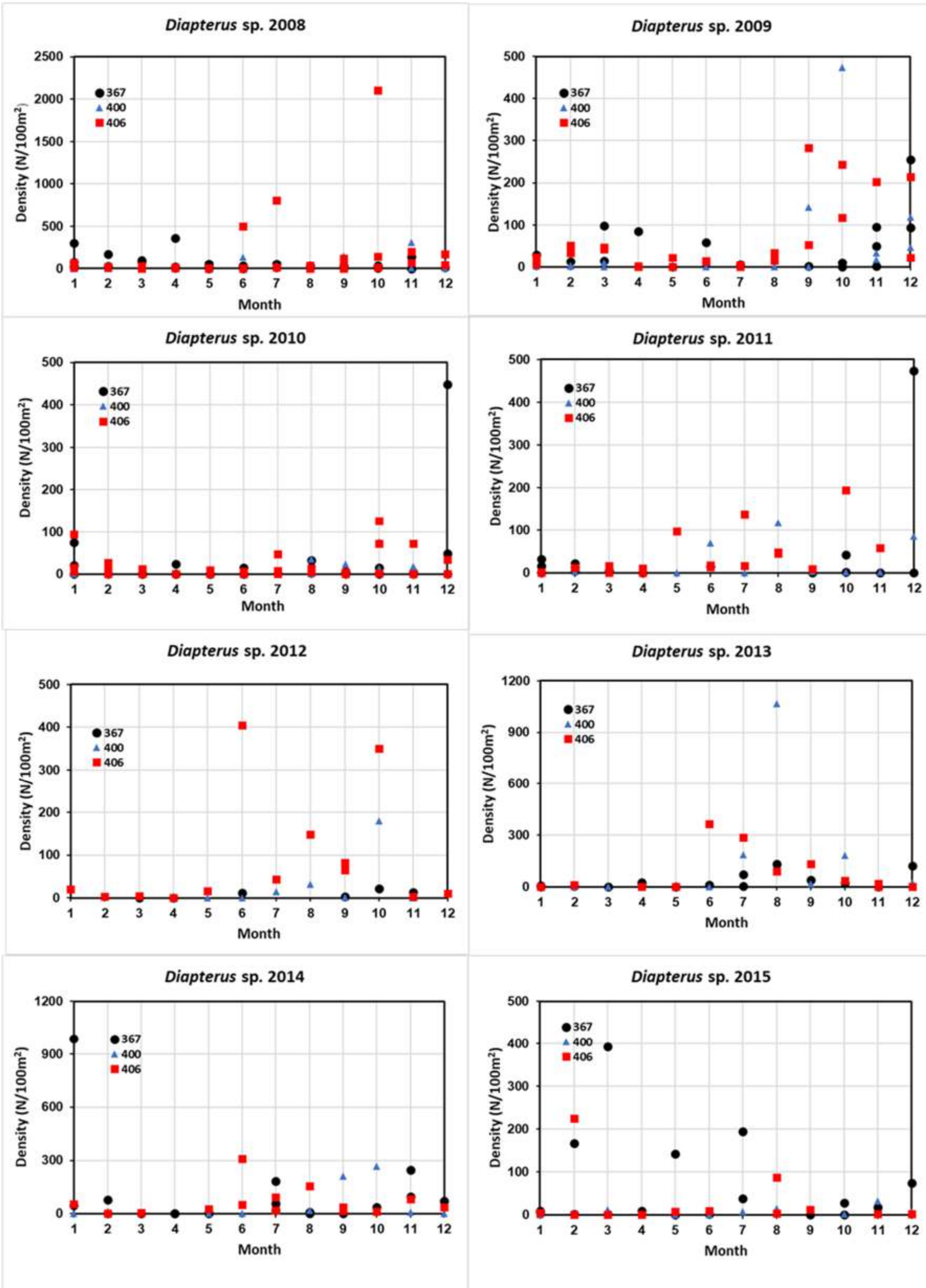


Figure 3.7.26. Monthly densities of juvenile Irish pompano (*Diapterus* sp.) captured in seine hauls by the Fisheries Independent Monitoring program, 2008-2018, from three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River. Note the variation in the Y-axis scale.

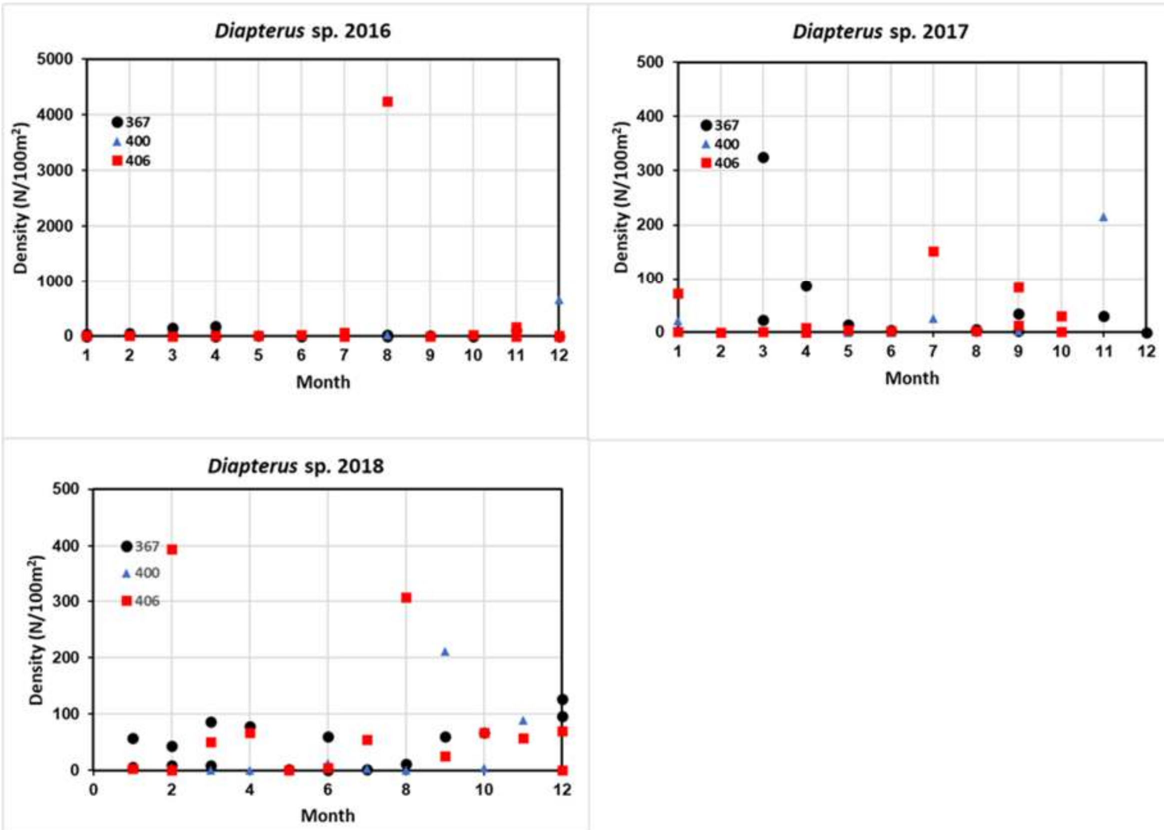


Figure 3.7.26 (continued).

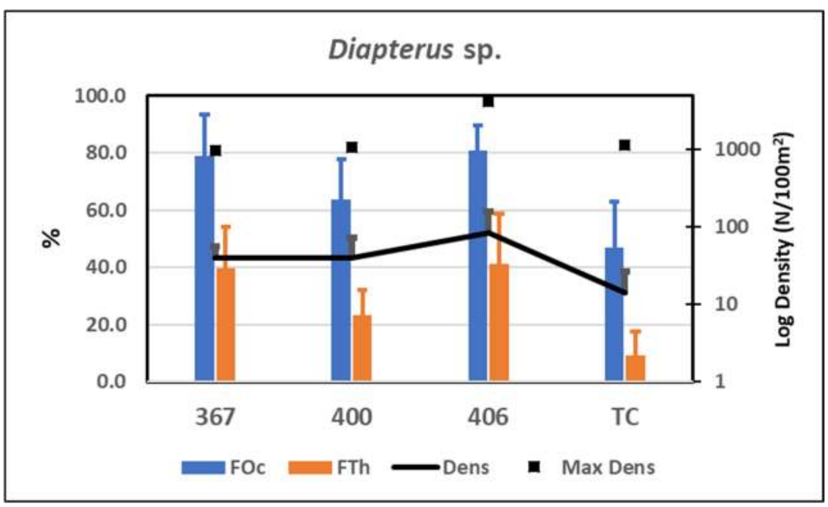


Figure 3.7.27. Mean (+ S.D.) values for juvenile Irish pompano (*Diapterus* spp.) annual frequency of occurrence (%FO), annual frequency of samples exceeding the density threshold of 29.9 fish/100m2 (%TH), annual density (Dens) and annual maximum density of catches made by the Fisheries Independent Monitoring Program in each of three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River from 2008 to 2018, and data from sampling in this program at the mouth of Turkey Creek (TC) from 2015 to 2018.

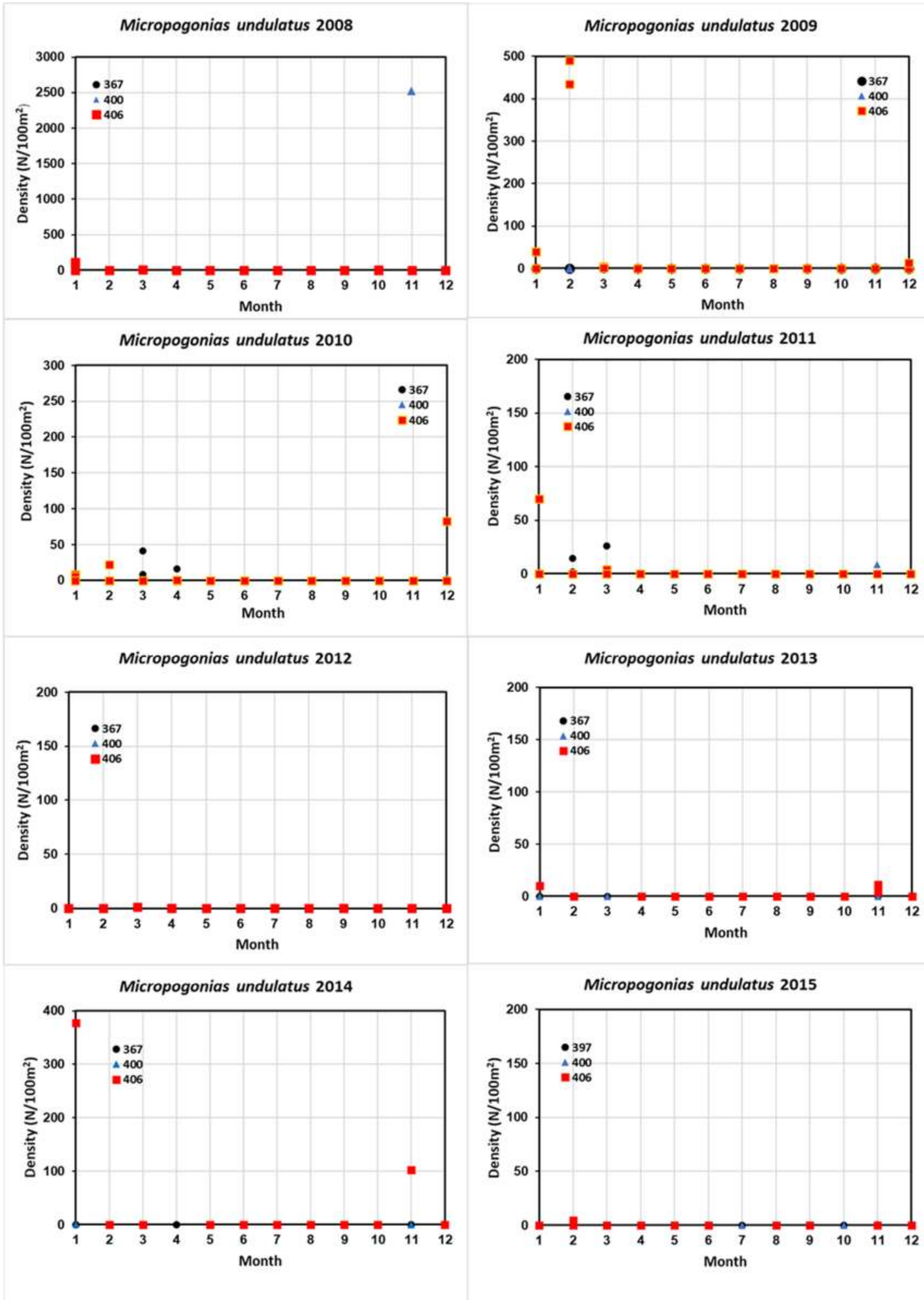


Figure 3.7.28. Monthly densities of juvenile Atlantic croaker (*Micropogonias undulatus*) captured in seine hauls by the Fisheries Independent Monitoring program, 2008-2018, from three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River. Note the variation in the Y-axis scale.

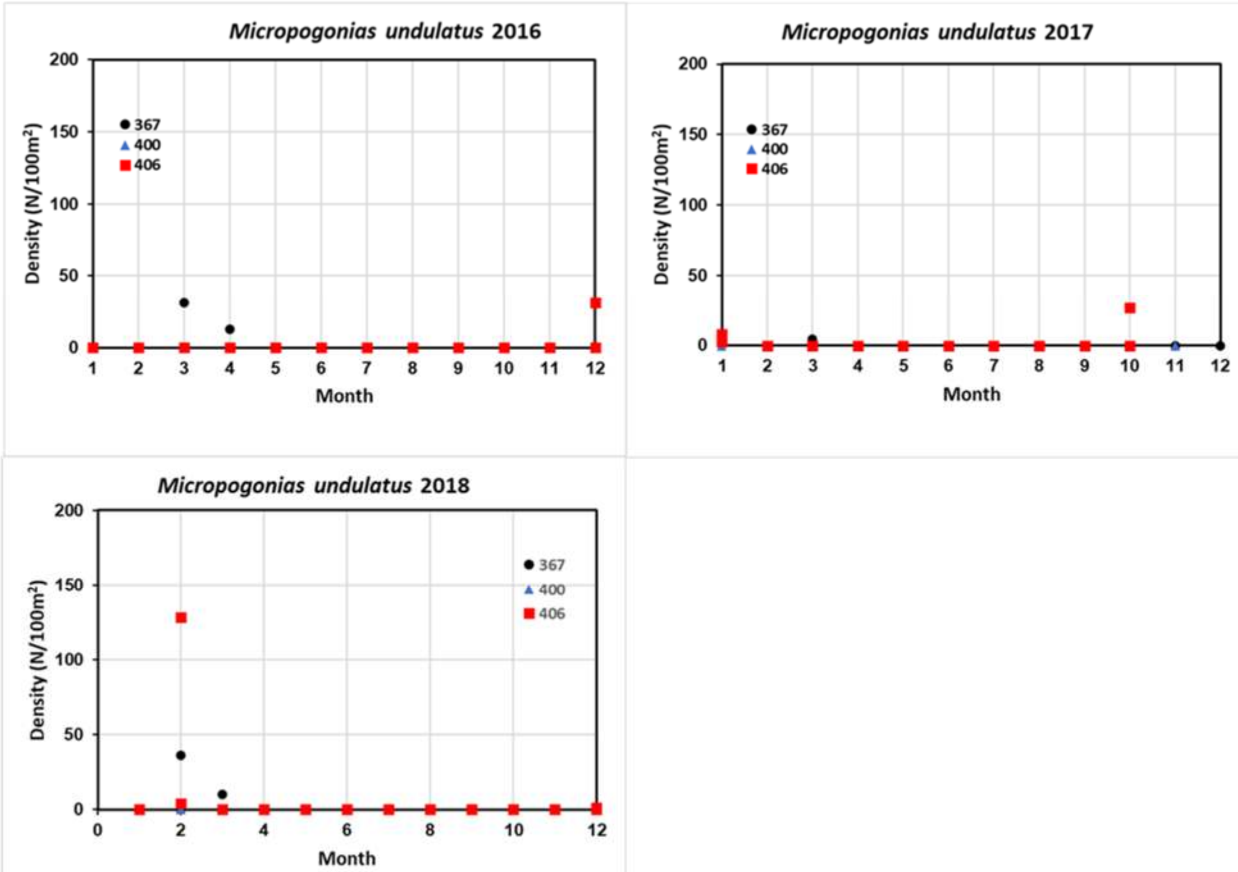


Figure 3.7.28 (continued).

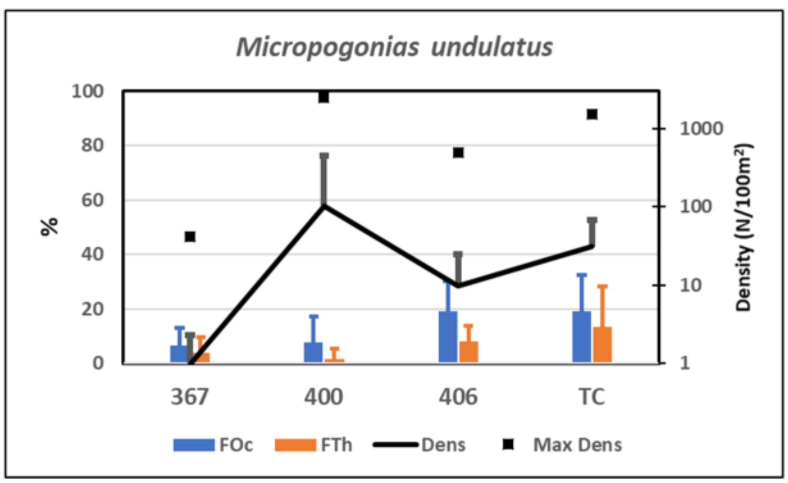


Figure 3.7.29. Mean (+ S.D.) values for juvenile Atlantic croaker (*Micropogonias undulatus*) annual frequency of occurrence (%FO), annual frequency of samples exceeding the density threshold of 29.9 fish/100m2 (%TH), annual density (Dens) and annual maximum density of catches made by the Fisheries Independent Monitoring Program in each of three sampling grid blocks: 367 = Turkey Creek, upstream of the U.S. 1 bridge, 400 and 406 = mouth of Sebastian River from 2008 to 2018, and data from sampling in this program at the mouth of Turkey Creek (TC) from 2015 to 2018.

4.0. Dredging at the Mims Boat Ramp

4.1. *Dredging and Effectiveness of Muck Removal*

Water depths and muck thicknesses in the area of the Mims Boat Ramp were determined with ~30-m spatial resolution before dredging on August 30, 2017 (Figures 4.1.1 and 4.1.2). The muck thickness data collected with 30-m resolution were augmented during the same day via probing at adjacent locations, with higher resolution, to identify the margins where muck bordered sand (Figure 4.1.1). About 80% of the muck in the area to be dredged was in a small basin at the southern end of the dredge site where muck thicknesses were 1–2 m (Figure 4.1.1). The remaining muck was in the main channel running alongshore from the boat ramp.

We calculated a total muck volume of 23,000 m^3, essentially the same as the estimate by the contractor when muck thicknesses ≥5 cm were included (Figure 4.1.1b). The surface area and volume were determined by overlaying our data onto dredge templates for muck with thicknesses less than 1 foot (~30 cm) and less than 2 inches (~5 cm, Table 4.1.1, Figure 4.1.1). Water depths before dredging were commonly <1 m; the deepest water (~2 m) was found in the southern basin (Figure 4.1.2).

Our sediment data for Mims shows that sediment composition varied as a function of the relative amounts of sand, silt and clay that have mixed with the muck. The water content (by volume) of surface sediments ranged from 27% (by weight) in sandy sediments to 85% in muck. Concentrations of organic matter followed patterns for water content with a range from <1% to 26% OM (Table 4.1.2). The value of 26.6% OM at station DM-11 is one of the highest values found in the IRL (Table 4.1.2)

Fluxes of N and P at the Mims site were determined using our Quick-Flux technique and validated using results for interstitial water from a whole-core squeezer. Ammonium fluxes ranged from 0.5 mg $N/m^2/hr$ in sandy sediments near the edge of the dredge area at station EM-18 to 13.1 mg $N/m^2/hr$ near the center of the largest area of muck (station M-1; Table 4.1.2, Figure 4.1.1). Fluxes of ammonium were variable with an overall average of ~7 ± 4 mg $N/m^2/hr$ (60 ± 40 Tons-$N/km^2/yr$). If these fluxes are applied to the entire 0.05 km^2 surface area of muck, releases of N from this deposit in amounts on the order of ~3 tons-N/year.

Benthic fluxes were adjusted to 25°C using Equation 4.1 from Fox and Trefry (2018b), with Q_{10} temperature coefficients of 1.8 and 2.0 for N and P, respectively.

$$R_2 = 10^{\left(\frac{\frac{\log Q_{10}}{10}}{T_2 - T_1}\right) + log R_1} \text{ or } R_2 = R_1 Q_{10}^{\frac{(T_2-T_1)}{10}} \qquad \text{(Equation 4.1)}$$

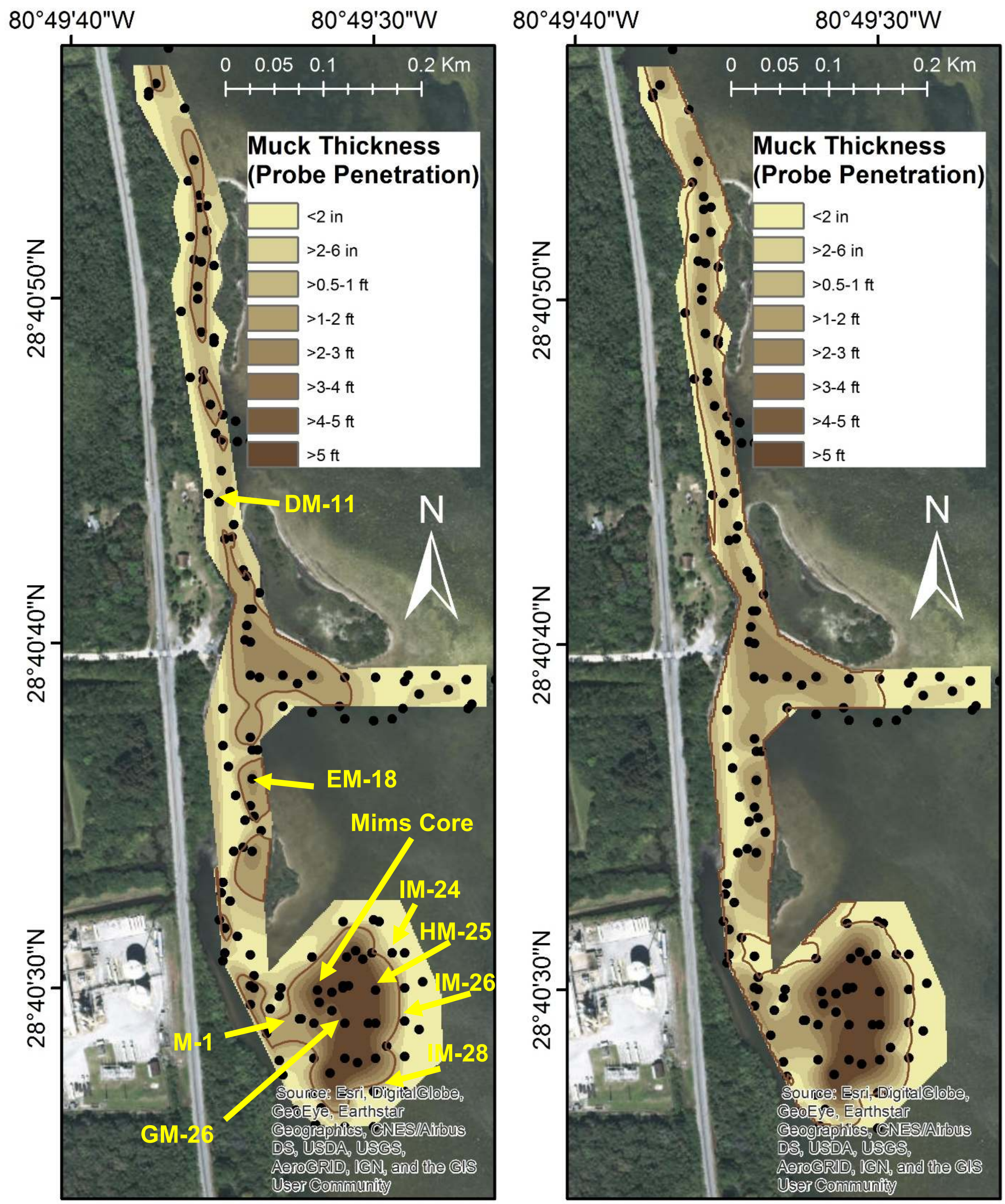


Figure 4.1.1. Map of the study area showing probe locations (black dots) and contoured muck thicknesses. Solid brown line indicates the (a) 30-cm (1-foot) contour (Surface Area (SA) = 26,000 m^2, Volume (V) = 14,000 m^3) and (b) 5-cm (2-inch) contour (SA = 48,000 m^2, V = 23,000 m^3).

Figure 4.1.2. Contour map of water depth at the Mims Boat Ramp showing water depths prior to dredging.

Table 4.1.1. Surface area and volume of muck in the Mims study area with minimum thicknesses of 1-foot (~30 cm) and 2-inches (~5 cm) and fluxes of N and P from muck at the Mims study site.

	Surface Area (m^2)	Volume (m^3)	N Flux @ 25°C (tons/yr)	P Flux @ 25°C (tons/yr)
>2 inches	48,000	23,000	2.9 ± 1.5	0.13 ± 0.11
>1 foot	26,214	14,037	1.6 ± 0.8	0.07 ± 0.06

Table 4.1.2. Pre-dredging data for sediment water content, organic matter (OM) content and ambient fluxes of N and P and fluxes adjusted to 25°C (Q_{10} values of 1.8 and 2.0 for N and P, respectively) at 8 sites within the Mims study area.

Station	Water Content (mass)	OM Content	Flux (tons/km²/yr) NH_4^+-N Ambient	Flux (tons/km²/yr) NH_4^+-N 25°C	Flux (tons/km²/yr) PO_4^{3-}-P Ambient	Flux (tons/km²/yr) PO_4^{3-}-P 25°C
DM-11	85.0	26.6	18.3	14.5	0.34	0.26
EM-18	44.6	3.1	4.3	3.4	0.07	0.06
IM-24	82.4	22.0	41.7	33.0	0.62	0.47
HM-25	84.1	24.5	93.1	73.6	5.94	4.51
IM-26	82.6	21.6	100.6	79.6	6.40	4.86
M-1	83.2	21.1	119.3	94.4	6.30	4.78
GM-26	85.2	25	89.0	70.4	2.35	1.78
IM-28	27.4	0.9	Sandy sediments, no water recovered			
Avg. Muck	84 ± 20	24 ± 8	77 ± 35	61 ± 28	3.7 ± 2.6	2.8 ± 2.0

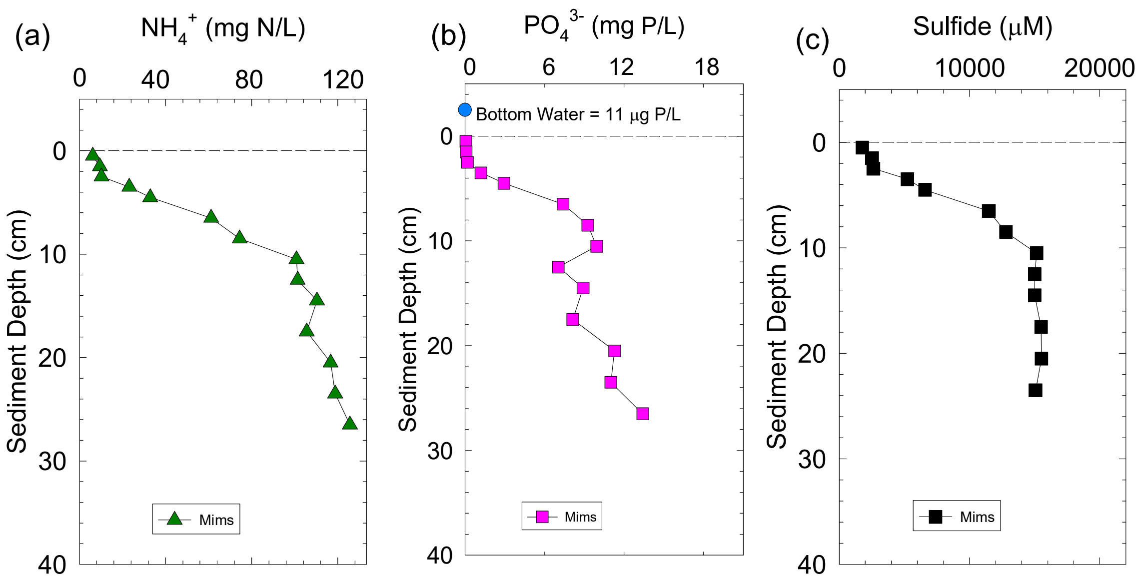


Figure 4.1.3. Vertical profiles for (a) ammonium, (b) phosphate and (c) sulfide in interstitial water obtained from a sediment core at site Mims; location shown on Figure 2.3.2.

4.2. Seagrasses and Drift Algae

Seagrasses and drift algae from Mims were sampled in the summers of 2017 and 2018, and a pre- versus during-dredging comparison is presented here (Figures 4.2.1–4.2.6). Seagrasses consistently occurred at the Mims Dredging Sandy site (MDS) and the Mims Control Sandy site (MCS), but did not occur at muck sites (MDM, MCM). The primary species at Mims station was *Halodule wrightii*. Seagrass visual percent cover, canopy height, and shoot count were all generally higher in 2018 relative to 2017 in the same season (summer). Nevertheless, in 2018, all parameters showed substantial variability. Visual percent cover of *H. wrightii* ranged from 2–100% (Figure 4.2.1). Seagrass canopy height ranged from 3.2-16.8 cm (Figure 4.2.2). Shoot count varied from nearly 0–36 (Figure 4.2.3).

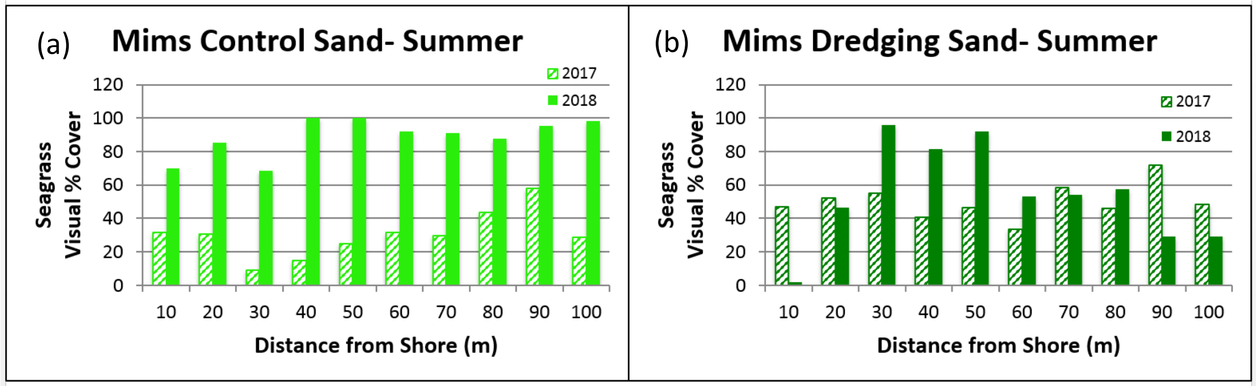


Figure 4.2.1. Mean seagrass visual % cover in the Summer (July/August) at (a) Mims Control Sand stations (MCS) and (b) Mims Dredging Sand stations (MDS) along replicated (n = 3) 100-m transects perpendicular to shore, comparing 2017–2018. The 2018 sampling occurred during dredging at the Mims Boat Ramp.

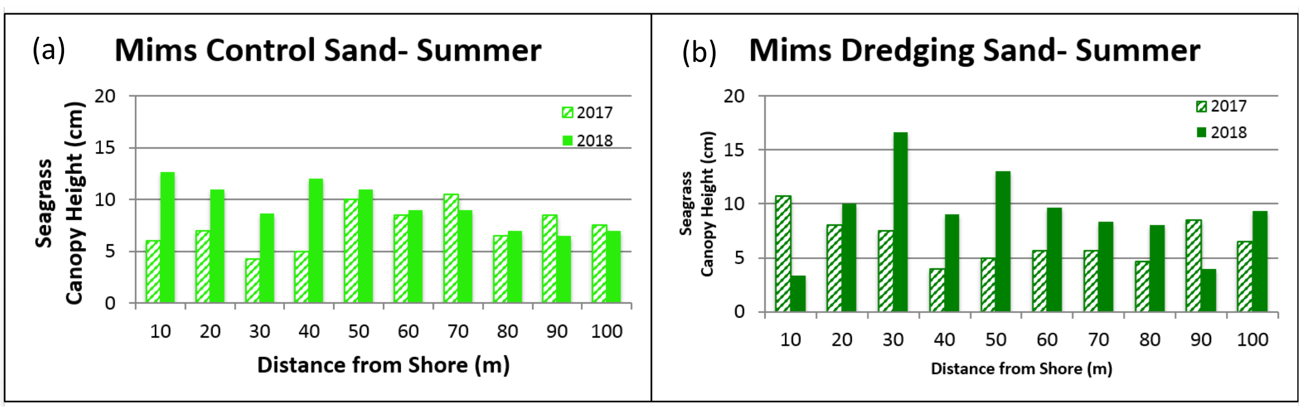


Figure 4.2.2. Mean seagrass canopy height in the Summer (July/August) at the (a) Mims Control Sand stations (MCS) and (b) Mims Dredging Sand stations (MDS) along replicated (n = 3) 100-m transects perpendicular to shore, comparing 2017–2018. The 2018 sampling occurred during dredging at the Mims Boat Ramp.

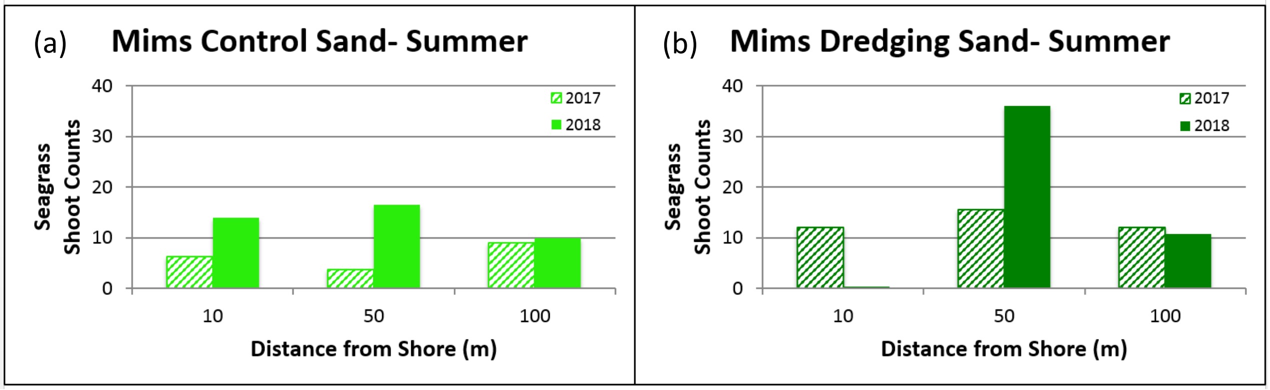


Figure 4.2.3. Mean seagrass shoot counts 900 cm^{-2} in summer (July/August) at the (a) Mims Control Sand stations (MCS) and (b) Mims Dredging Sand stations (MDS) along replicated (n = 3) 100-m transects perpendicular to shore, comparing 2017–2018. The 2018 sampling occurred during dredging at the Mims Boat Ramp.

Epiphytes overgrowing the seagrass *H. wrightii* were present and sometimes severe in both 2017 and 2018, with visual qualitative severity scores from 3–5, the high end of epiphyte severity (Figure 4.2.4). Epiphytes on seagrass blades compete with seagrasses for nutrients and light, can be detrimental to seagrass growth in excessive amounts, and in some cases used as an indicator of seagrass health or water quality (Broderson et al., 2015).

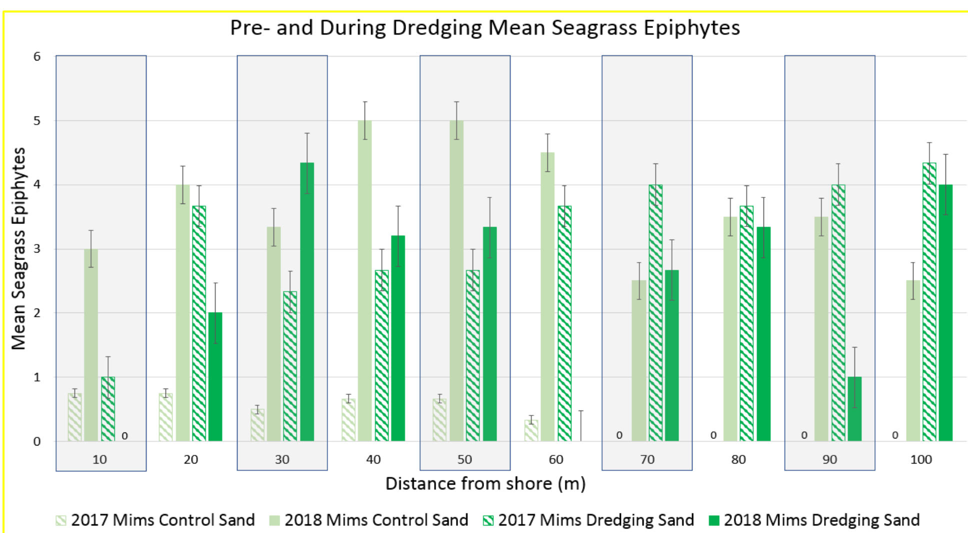


Figure 4.2.4. Mean epiphyte score compared between Summer 2017 and 2018 (July/August) at Mims Control Sand (MCS) and Mims Dredging Sand (MDS) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate no seagrass present. Dredging in Mims was underway during the 2018 sampling.

Drift algae were less common at Mims than at Turkey Creek, and often entirely absent from the Mims non-dredging (MCS) transects. At the Mims dredging site, drift algae visual % cover ranged from 4–20% (2017) to 0–20.5% (2018) along transects (Figure 4.2.5 and 4.2.6).

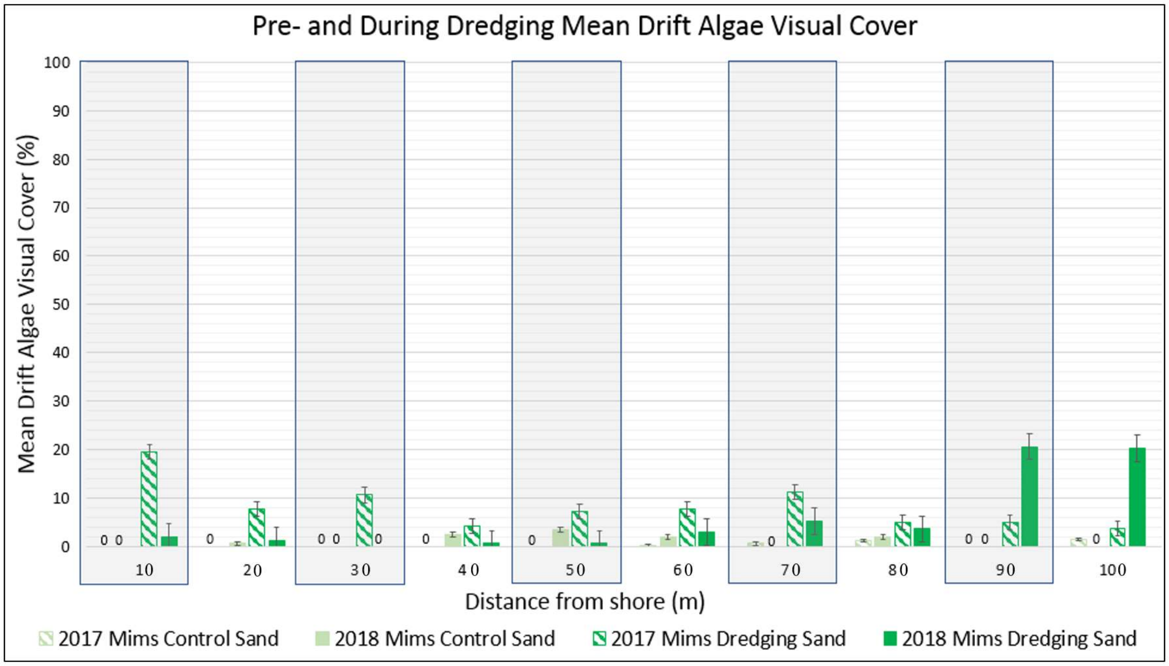


Figure 4.2.5. Mean drift algae visual % cover compared between Summer 2017 and 2018 (July/August) at Mims Control Sand (MCS) and Mims Dredging Sand (MDS) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate no drift algae present. Dredging in Mims was underway during the 2018 sampling.

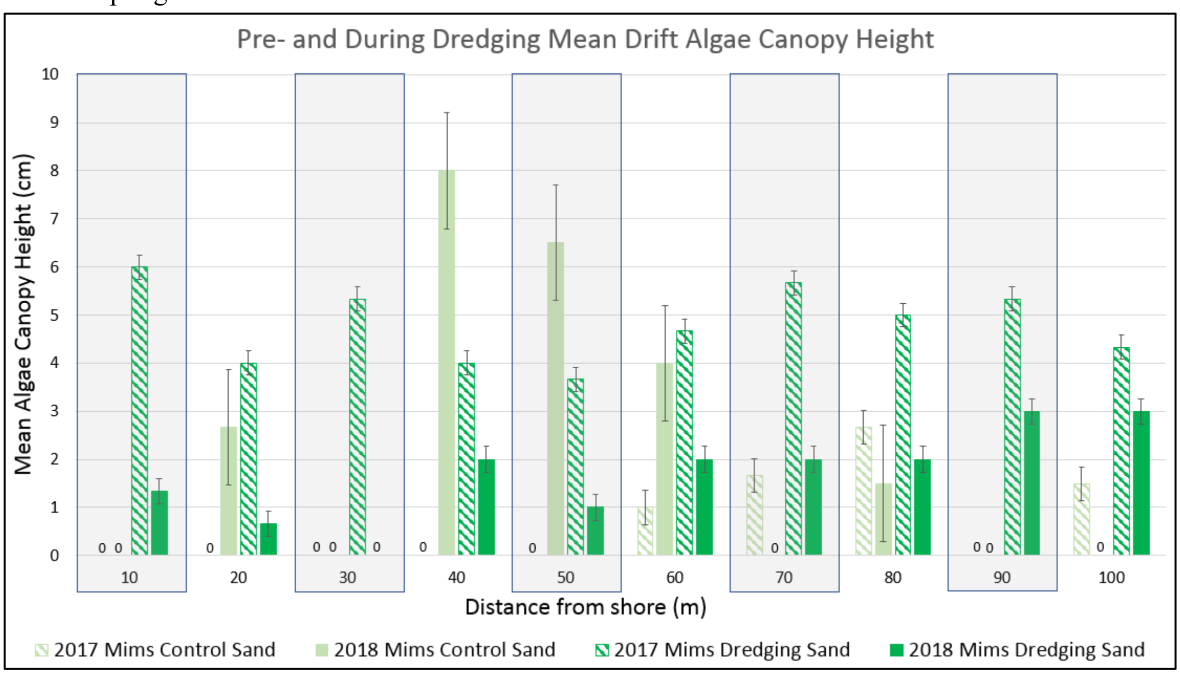


Figure 4.2.6. Mean drift algae canopy height compared between Summer 2017 and 2018 (July/August) at Mims Control Sand (MCS) and Mims Dredging Sand (MDS) along replicated (n = 3) 100-m transects perpendicular to shore. Error bars are ±1SE. Zeros indicate no drift algae present. Dredging was underway during sampling.

4.3. Sediments and Infauna

Mims and associated sites have been monitored approximately quarterly (seasonally) since March 2017. Dredging at Mims began in May of 2018, paused during the summer, resumed and ended in December 2018. Thus, comparisons between before and after dredging are not yet possible. Comparisons are made here for summer because that is the only season for which two years of data were available. Sediment data for Summer 2017 are very similar to sediment data for Summer 2018 (Figure 4.3.1), which is not surprising considering dredging in Summer 2018 sampling had only just begun and was on hiatus. If changes to organic sediments are to occur as a result of dredging, we might hope to see that when Mims dredging is completed.

Muck sites (MDM and MCM) have higher content in all parameters relative to other stations, and in both years. There are few significant differences between summer 2017 and summer 2018 when it comes to infaunal invertebrate abundance, diversity or richness (Figures 4.3.2–4.3.4). This is consistent with the observation that dredging had just begun in summer 2018 and then was placed on hiatus. If dredging is going to drive differences in these biological parameters, we might expect to see differences when the data are compared upon completion of the dredging.

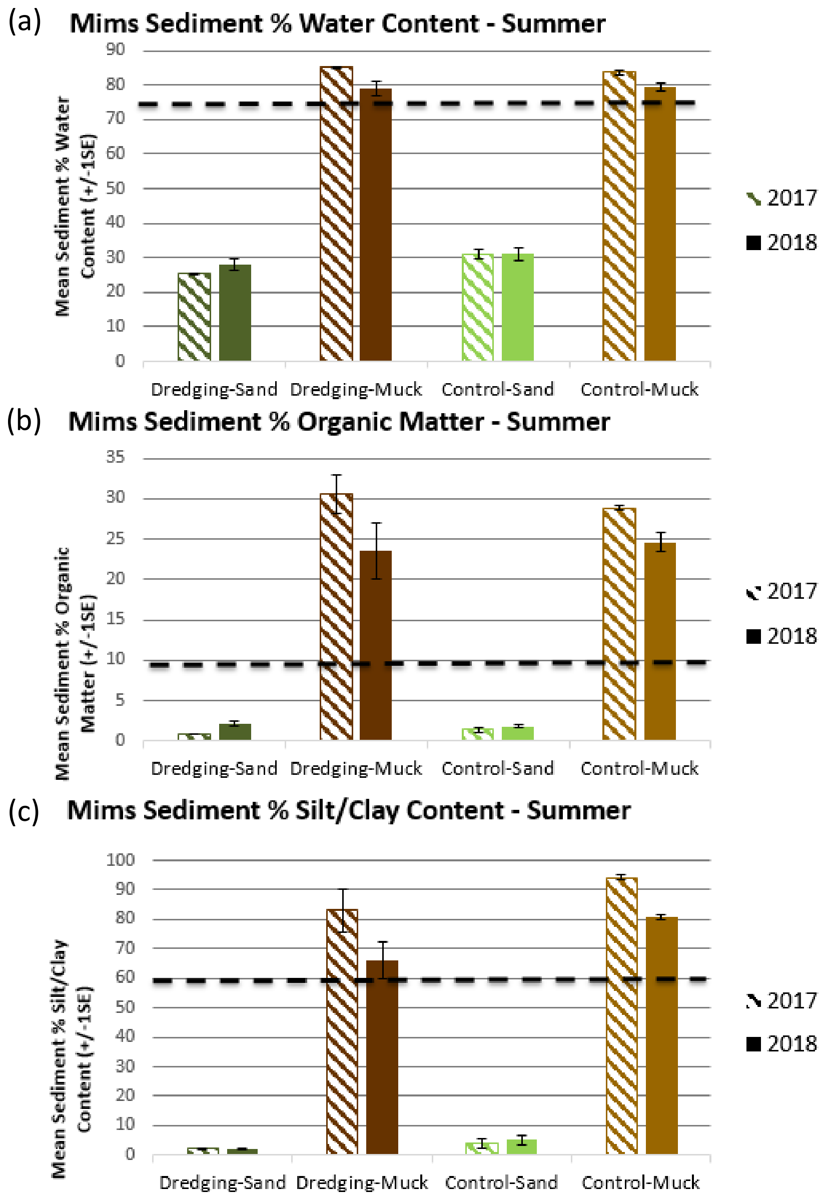


Figure 4.3.1. Sediment characteristics at Mims and associated sites, before (summer 2017) and during early stages of dredging (summer 2018). (a) Mean sediment % water content, (b) Mean sediment % organic matter, and (c) Mean sediment % silt/clay content. Error bars are ±1SE. Horizontal dashed lines indicate defined muck sediment parameter thresholds (Trefry and Trocine 2011).

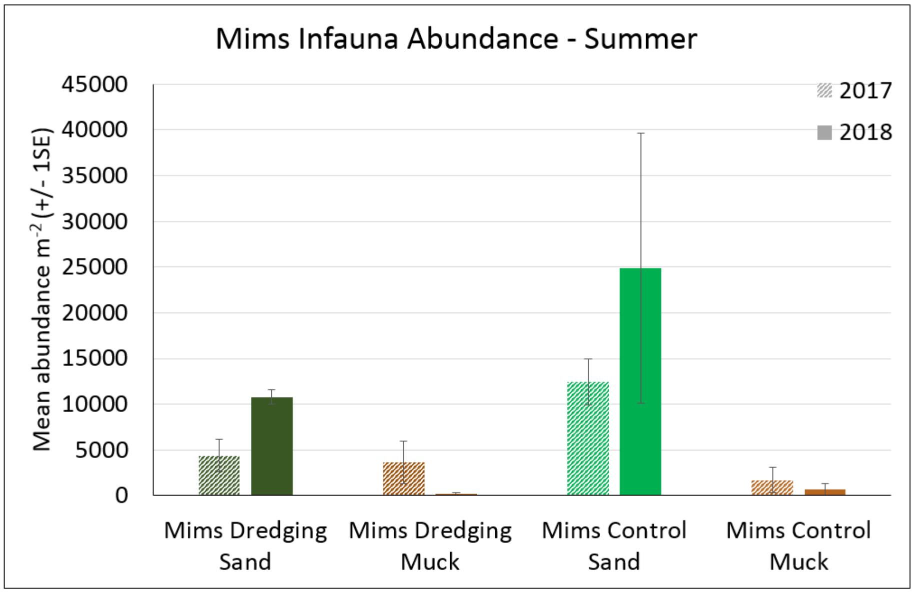


Figure 4.3.2. Mean overall summer infaunal invertebrate abundances for Mims and associated sites, compared between 2017 and 2018. Error bars = ±1SE.

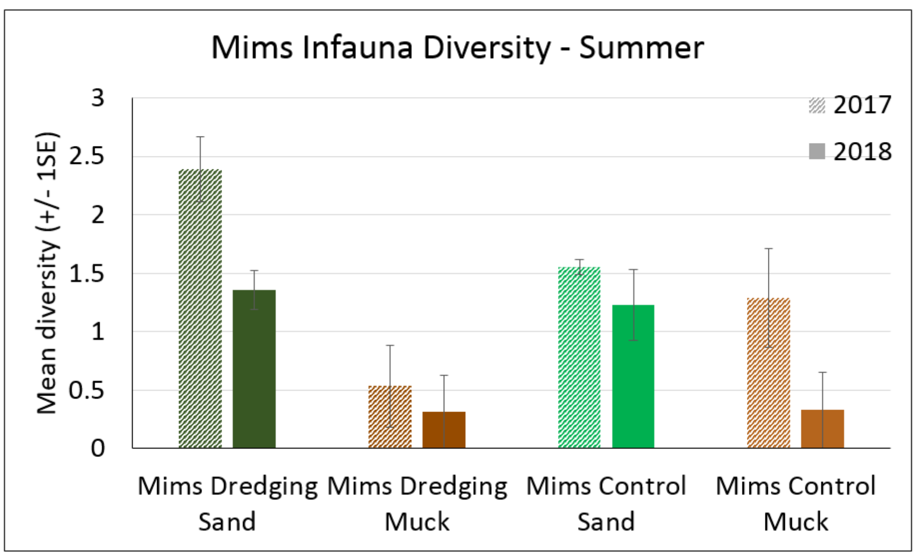


Figure 4.3.3. Mean summer infaunal invertebrate diversity for Mims and associated sites, compared between 2017 and 2018. Error bars = ±1SE.

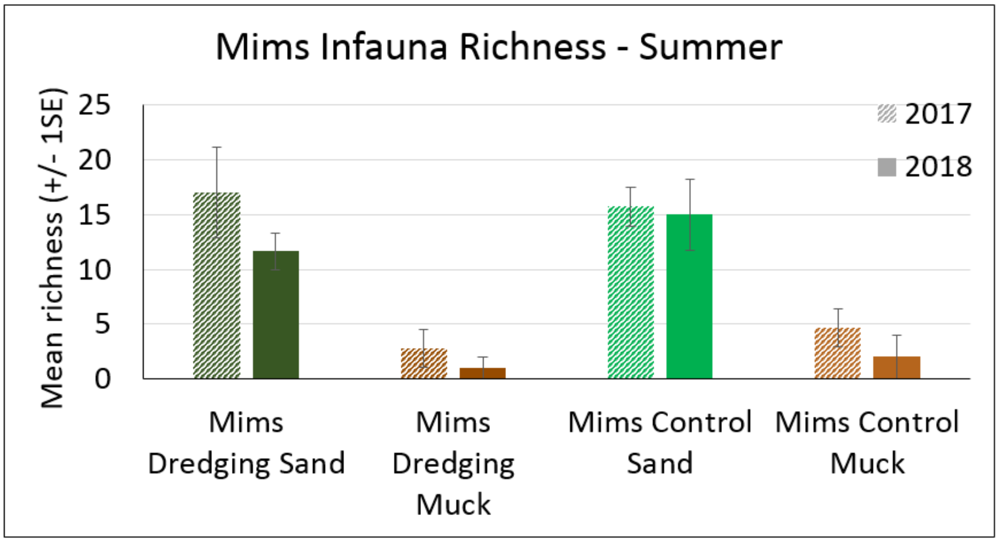


Figure 4.3.4. Mean summer infaunal invertebrate richness for Mims and associated sites, compared between 2017 and 2018. Error bars = ±1SE.

Table 4.3.1 Benthic invertebrates at Mims and associated sites (45 species).
Dominant species indicated with asterisks (* = 2017, ** = 2018, *** = both 2017 and 2018).

Mims Dredging Sand	Mims Dredging Muck	Mims Control Sand	Mims Control Muck
Acteocina canaliculata	*Acteocina canaliculata*	*Acteocina atrata*	*Acteocina canaliculata*
Alitta succinea	*Alitta succinea*	*Acteocina canaliculata*	*Astyris lunata*
Amygdalum papyrium	*Ammonia parkinsoniana*	*Ammonia parkinsoniana*	*Corophium sp.**
Astyris lunata	Annelid I	*Amygdalum papyrium*	Crab B (hermit crab)
Capitella capitata	Annelid J	Annelid J	Gammarid Amphipod C
*Cerapus tubularis***	*Anomalocardia cuneimeris*	*Astyris lunata*	Gammarid Amphipod D***
Corophium sp.	*Capitella capitata*	*Cerapus tubularis*	Gammarid Amphipod G*
Crab B (hermit crab)	*Cerapus tubularis*	*Corophium sp.**	Gammarid Amphipod I
Diopatra cuprea	*Corophium sp.*	*Ctenodrilus serratus*	*Glycera americana*
Diopatra cuprea	Crab Zoea A	*Diopatra cuprea*	*Hargeria rapax*
Eurypanopeus depressus	Crab Zoea A	*Eusarsiella zostericola*	*Hypereteone heteropoda*
Eusirus cuspidatus	*Ctenodrilus serratus*	Gammarid Amphipod C**	*Japonactaeon punctostriatus*
Gammarid Amphipod C	Cumacean B	Gammarid Amphipod D***	*Mulinia lateralis***
Gammarid Amphipod D***	*Eurypanopeus depressus*	Gammarid Amphipod G*	*Nassarius vibex*
Gammarid Amphipod G*	*Eusirus cuspidatus*	Gammarid Amphipod H	*Oxyurostylis smithi*
Gammarid Amphipod H	Gammarid Amphipod C	Gammerid Amphipod J	
Gammarid Amphipod I	Gammarid Amphipod D***	*Glycera americana*	
Glycera americana	Gammarid Amphipod G*	*Haminoea succinea*	
Haminoea succinea	Gammarid Amphipod H	*Hemipholis elongata*	
Hemipholis elongata	Gammerid Amphipod J	Isopod B	
Isopod C	*Glycera americana*	*Japonactaeon punctostriatus*	
Isopod D	*Haminoea succinea*	*Leptochelia dubia****	
*Leptochelia dubia***	*Hargeria rapax*	*Limulus polyphemus*	
Mulinia lateralis	Isopod A	*Mulinia lateralis*	
Nassarius vibex	Isopod D	*Nassarius vibex*	
Odostomia laevigata	*Leptochelia dubia*	Nematode A	
*Oxyurostylis smithi****	Megalops A	*Odostomia laevigata*	
Palaemonetes vulgaris	*Mercenaria mercenaria*	*Oxyurostylis smithi***	
Paradiopatra hispanica	*Mulinia lateralis*	*Palaemonetes vulgaris*	
Parastarte triquetra	Nematode A	*Paradiopatra hispanica***	
Pectinaria gouldii	*Odostomia laevigata*	*Parastarte triquetra*	
Peratocytheridea setipunctata	Ostracod C	*Pectinaria gouldii*	
Phascolion cryptus	*Oxyurostylis smithi*	*Peratocytheridea setipunctata*	
	Palaemonetes vulgaris	*Phascolion cryptus*	
	Paradiopatra hispanica	Polychaete Y	
	Parastarte triquetra	*Prunum apicinum*	
	Pectinaria gouldii	Snail Q	
	Pectinaria gouldii	Tanaid A	
	Peratocytheridea setipunctata	*Alitta succinea*	
	Periglypta listeri	*Capitella capitata*	
	Phoronis A	Crab B (hermit crab)	
	Polychaete P	*Cyrtopleura costata*	
	Polychaete Y	*Periglypta listeri*	
	Sipuncula D	Turbellaria A	
	Snail Q		
	Tanaid A		

4.4 Fish Surveys at Mims

We conducted six pre-dredging surveys of the fish fauna around the periphery of a portion of the planned dredge site near Mims, Florida (Figure F-2). The substrate across the sampling region was generally hard sand near the spoil islands, and slightly muddy sand further from the islands. Water clarity was sufficiently good to enable the bottom to be seen at a depth of 1 m. Seagrass coverage was visually estimated at 10-50% coverage at the sites.

Surveys were conducted bi-monthly from July 2017 to June 2018. Complete surveys, with all 9 samples, were achieved on five of the sampling dates. The survey in March 2018 was hindered by very dense *Gracilaria* drift algae, pushed by intense easterly winds onto the western shore of the IRL, and completely covered the entire region. The algae were so thick that it proved impossible to pull the seine effectively, and the 3 attempts we made succeeded in catching only 50 fish in total. By June, the drift algae had moved away from the site, without leaving visible damage to the underlying seagrass habitat.

Seine samples focused on the small and juvenile fishes inhabiting the region. Several anglers reported that they fished in the region for red drum and black drum and tarpon in the deep muck-filled channel along shore, where dredging was to occur. Sea trout were the primary target of their fishing efforts over seagrass meadows in the middle of the lagoon.

The abundance and composition of the fish fauna at the Mims site was quite different from the fauna at Turkey Creek. The 42 samples collected a total of only 5,152 fishes from 34 taxa (Table 4.4.1). Nearly 75% of the catches were anchovies and 10% were silversides (*Menidia* sp.), a small schooling fish characteristically found in shallow estuarine marsh habitats.

Only 10% of the total fish catch was comprised of demersal fishes. These demersal species were dominated by southern kingfish, *Menticirrhus americanus* (n=98), silver perch, *Bairdiella chrysoura* (n=73), and sea trout (*Cynoscion* spp. (n=57). All of these taxa are reported to spawn in the northern IRL (Reyier and Shenker 2007; Reyier et al. 2008). The mojarras and other drum species that dominated Turkey Creek were present only in very low numbers or were completely absent. Those species spawn in coastal waters, with larvae moving through inlets to enter their nursery habitats. Mims is about 100 km from the Sebastian Inlet, about 50 km from the intermittently-open locks of Port Canaveral, and 51 km from Ponce Inlet via the restrictive Haulover Canal, making it unlikely that many of the coastal larvae could reach this region.

One station in Mims spanned a shallow muck-filled channel that boaters use for access between the boat ramp and the open flats ecosystem to the east. Seine hauls made across this channel typically captured juvenile and adult hard-head catfish (*Ariopsis felis*), but no other demersal fishes.

Table 4.4.1 Total catch of fishes taken in seine hauls from 9 stations around the dredge site near Mims, Indian River Lagoon, July 2017 – June 2018.

Scientific Name	Common Name	Total Abundance	% of Total Fish	Pelagic or Demersal	% of Demersal Fish
Anchoa spp.	Anchovy	3868	74.9	P	
Menidia spp.	Silverside	511	9.9	P	
Lucania parva	Rainwater killifish	182	3.5	P	
Menticirrhus americanus	Southern kingfish	98	1.9	D	20.16
Bairdiella chrysoura	Silver perch	73	1.4	D	15.02
Gobiidae	Goby	68	1.3	D	13.99
Cynoscion spp.	Sea trout	57	1.1	D	11.73
Mugil cephalus	Striped mullet	43	0.8	P	
Ariopsis felis	Hardhead catfish	42	0.8	D	8.64
Clupeidae	Herring	27	0.5	P	
Fundulus sp.	Killifish	23	0.4	D	4.73
Gobisoma bosc	Naked goby	20	0.4	D	4.12
Diapterus spp.	Irish Pompano	20	0.4	D	4.12
Syngnathus sp.	Pipefish	16	0.3	D	3.29
Eucinostomus spp.	Mojarras	16	0.3	D	3.29
Oligoplites saurus	Leatherjacket	14	0.3	P	
Cyprinodon variegatus	Sheepshead minnow	14	0.3	D	2.88
Strongylura marina	Atlantic needlefish	12	0.2	P	
Mugil curema	White mullet	10	0.2	P	
Microgobius gulosus	Clown goby	8	0.2	D	1.65
Sciaenidae	Drum	7	0.1	D	1.44
Dasyatis sabina	Atlantic stingray	7	0.1	D	1.44
Brevoortia tyrannus	Menhaden	5	0.1	P	
Chasmodes saburrae	Florida blenny	4	0.1	D	0.82
Achirus lineatus	Lined sole	3	0.1	D	0.62
Elops saurus	Ladyfish	3	0.1	P	
Pogonias cromis	Black Drum	2	<0.01	D	0.41
Cynoscion arenarius	Sand trout	2	<0.01	D	0.41
Archosargus probatocephalus	Sheepshead	2	<0.01	D	0.41

Table 4.4.1 (continued).

Scientific Name	**Common Name**	**Total Abundance**	**% of Total Fish**	**Pelagic or Demersal**	**% of demersal Fish**
Trachinotus carolinus	Florida pompano	1	<0.01	D	0.21
Chilomycterus schoepfi	Striped burrfish	1	<0.01	D	0.21
Sphoeroides sp.	Puffer	1	<0.01	D	0.21
Chaetodipterus faber	Spadefish	1	<0.01	D	0.21
Caranx sp.	Jacks	1	<0.01	P	
TOTAL PELAGIC FISH		**4676**			
TOTAL DEMERSAL FISH		**486**			
TOTAL FISH		**5162**			

5.0 General Conclusions

About 300 metric tons of nitrogen and 70 metric tons of phosphorus were removed from Turkey Creek with 160,000 m^3 of muck. Some observed biological variation was indistinguishable from seasonal changes, while other changes were attributed to muck dredging. Seagrass variation, primarily *Halodule wrightii*, was indistinguishable from seasonal variation and some may not be related to dredging activity, at least in the short term. Seagrasses did not occur within dredged sites prior to dredging, and this was still the case immediately following dredging. Seagrass abundances at more distant sites (adjacent to or away from dredging) are low and extremely variable, often statistically indistinguishable from zero. Seagrass temporal patterns are, however, reasonably consistent with expected seasonal growth and occurrence. Direct or nearby removal of muck sediments seems to have had little immediate impact on adjacent seagrasses, at least in the short term. However, possible regional effects due to multiple, regional muck removal projects and attendant water quality improvements will likely take longer to observe and document. Baselines have been established for Mims and Sykes Creek with regard to nearby seagrasses and comparison sites.

There were some infauna responses to dredging, including increases where muck was dredged and decreases where intermediate sediments were dredged. Dredged stations with intermediate organic sediments (TC3 and TC4) at first experienced a decline in biological indicators (species richness, diversity, abundance), while these same indicators often improved for the channel sites where muck was dredged (TCM 1-4). As of the writing of this report, some infauna populations at dredged sites have drifted back towards their original state. The natural population variance makes it difficult to authoritatively identify driving factors, but slumping and shifting sediments may have brought sediments at TC3 and TC4 back to similarity with TC1 and TC2, and dredged muck (TCM 1-4) back to similarity with undredged muck (CCM 1-2). The strong relationships of infaunal species with sediment parameters (see Figures 3.6.4-3.6.5) supports this hypothesis and suggests that biological changes likely reflect the shifting sediment environment to some degree. More complete removal of organic sediments and gentler sloping of dredge pits may reduce the amount of slumping and slow sediment reversion. Internal indicators would then be expected to follow suit based on sediment environmental conditions. In Mims and Sykes Creek, baselines of infauna populations have been established for comparison after future planned dredging.

Most benthic, pelagic and predatory fish showed variation in abundances indistinguishable from seasonal and migratory pattern, and which we would not attribute to dredging activity. The one exception are the mojarras (*Eucinostomus* spp.), which were observed to increase in Turkey Creek after dredging. If muck removal allows development of increased benthic prey communities over a wider habitat, we may see increases around the post-dredging site of juvenile

Eucinostomus spp. and *Diapterus* spp. These fishes recruit in spring and summer, typically live in sandy habitats, and consume those benthic prey.

Managers overseeing continued dredging efforts in the Indian River Lagoon may take away some helpful findings and concepts from these studies. Muck sediments don't tend to change composition, and it is more difficult to completely reduce areal coverage than volumetric deposits. It would be valuable to develop approaches to more effectively reduce areal coverage. Regarding infauna which live in surrounding intermediate organic sediments, muck dredging can be most effective when the sediments have a very high organic content supporting little or no infaunal life, in which case it is likely that new conditions will be an improvement and infaunal life will track sediment composition. Seagrasses and fishes, in contrast, are populations we expect to respond to longer term, general improvements in regional estuarine water quality, but were not observed to respond directly to local dredging.

5.1 Summary A summary of Turkey Creek dredging responses and effectiveness:

- Dredging reduced the volume of muck in the dredged area by >60%.
- Dredging decreased the surface area of muck in the dredged area by <20%, which may be partially explained by scattering, slumping and redistribution of adjacent muck sediments following dredging.
- The volume of water in the dredged area increased by 160,000 m^3 after dredging, a direct reflection of the removed sediment volume in Turkey Creek.
- Muck, where present, did not change in composition during or following dredging.
- Fluxes of N and P were >50% lower within 3 months after dredging, likely related to mixing during dredging and Hurricane Matthew (see section 3.3).
- The total concentration of DO in Turkey Creek increased after dredging, proportionate to the deeper water column and larger volume of water.
- Increased turbidity during the beginning of dredging during Phases I and II were due, respectively, to a large algal bloom and resuspension of sediment during Hurricane Matthew (see section 3.3).
- The DMMA retained >99.9% of the solids pumped in from Turkey Creek.
- Chemical treatments effectively controlled concentrations of dissolved phosphate in the DMMA.

- Concentrations of nutrients above background in water discharged from the DMMA were not identifiable at 100 m from outfall.

- Variable seagrass trends were indistinguishable from seasonal trends at control sites during the course of this study. It is expected that seagrasses will respond to more general water quality and sediment improvements as the cumulative result of multiple, spatially expansive regional projects that remove muck.

- Transplanted seagrasses grew through the early summer period, but then disappeared by early fall following extensive periods of fresh water runoff.

- Infauna community indicators responded to dredging in the short term, but tracked sediment conditions when they reverted to a pre-dredging state.

- The fish fauna was dominated by highly mobile pelagic schooling species. With one exception, juvenile demersal fish abundances showed significant interspecific, microhabitat, seasonal and interannual variations in abundance that were not associated with pre-dredging, dredging or post-dredging periods. Juvenile mojarras (Eucinostomus spp. were significantly more abundant during the final dredging period and the first year after dredging, but that peak in abundance could reflect broader variability in recruitment levels in the region.

- The most abundant species of juvenile demersal fishes fed primarily on epibenthic crustaceans and various infaunal taxa that were not present in pre-dredging muck habitats. These prey showed some increase in abundance in areas where muck had been removed, potentially increasing the prey base for juvenile fishes in the Turkey Creek habitat. However, the lack of complex structure in the dredging region, such as seagrass and viable oyster beds, may inhibit the ability of juveniles to avoid their own predators, thus limiting the nursery habitat value of the region.

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Appendix A.
Sediment and Water Assessment of the Palm Bay Dredge Material Management Area (DMMA)

A summary of results from study of the DMMA where dredge material from Turkey Creek was temporarily stored is presented below. Further details regarding our assessment of the DMMA can be found in Fox et al. (2017). The primary conclusions from that study are as follows:

- About 200 tons of nitrogen (N) and ~50 tons of phosphorus (P) were successfully removed when muck sediments were dredged from Turkey Creek and deposited in the Palm Bay DMMA.
- The overall retention efficiency for solids placed in the DMMA was >99.9% (i.e., <0.1% of the solids were released to the IRL).
- Concentrations of total suspended solids (TSS) in the outfall from the DMMA averaged ~28 mg/L and 12 mg/L, respectively, during dredging in Phase I (February 20–April 22, 2016) and Phase II (September 6, 2016–January 11, 2017).
- Chemical treatments decreased concentrations of dissolved phosphate in the incoming dredged material from as high as 10,000 µg P/L to <40 µg P/L in clarified water releases to the Indian River Lagoon (IRL). Concentrations of dissolved phosphate in the IRL at ~100 m from the outfall were <50 µg P/L.
- Total nitrogen in water released to the IRL was >5 mg N/L throughout the dredging process relative to <0.8 mg N/L in the lagoon. These results spurred efforts to chemically or biologically remove dissolved N during future muck removal projects.
- Concentrations of N and P were at baseline values in the IRL at ~100 m away from the outfall to the DMMA.
- An estimated 6 tons of N (~90% dissolved) and 0.1 ton of P (~30% dissolved) were released to the IRL during the one-year dredging project. Although unique to the Turkey Creek area, annual freshwater discharges to the IRL from Turkey Creek carry ~80 and ~5 tons of N and P, respectively.
- Dried solids were removed from the Palm Bay DMMA and transported by trucks to upland areas for beneficial reuse, primarily on agricultural land in central Florida.

All dredged material from Turkey Creek was pumped ~2 km north to a DMMA (Figure A1). During Phase I (February 20–April 22, 2016), pumping was 24 hours/day and in Phase II (September 6–January 11, 2017), pumping was 10 hours per day. From April 7, 2016, until very near the end of dredging (when chemicals were depleted), alum and flocculants were added to the incoming pipe ~300 m before dredged material was discharged into the DMMA. Water that was clarified in the DMMA flowed over boards at a weir near the eastern end of the reservoir and then through a pipe to an outfall to the IRL (Figure A1). Incoming dredged material from Turkey Creek was sampled through a port in the pipe just before it was discharged into the northwest corner of the DMMA (Figures A1, A2).

Figure A1. Satellite image of the Palm Bay Dredge Material Management Area (DMMA, BV-52) with arrows and text boxes to identify selected components and the adjacent Indian River Lagoon (IRL).

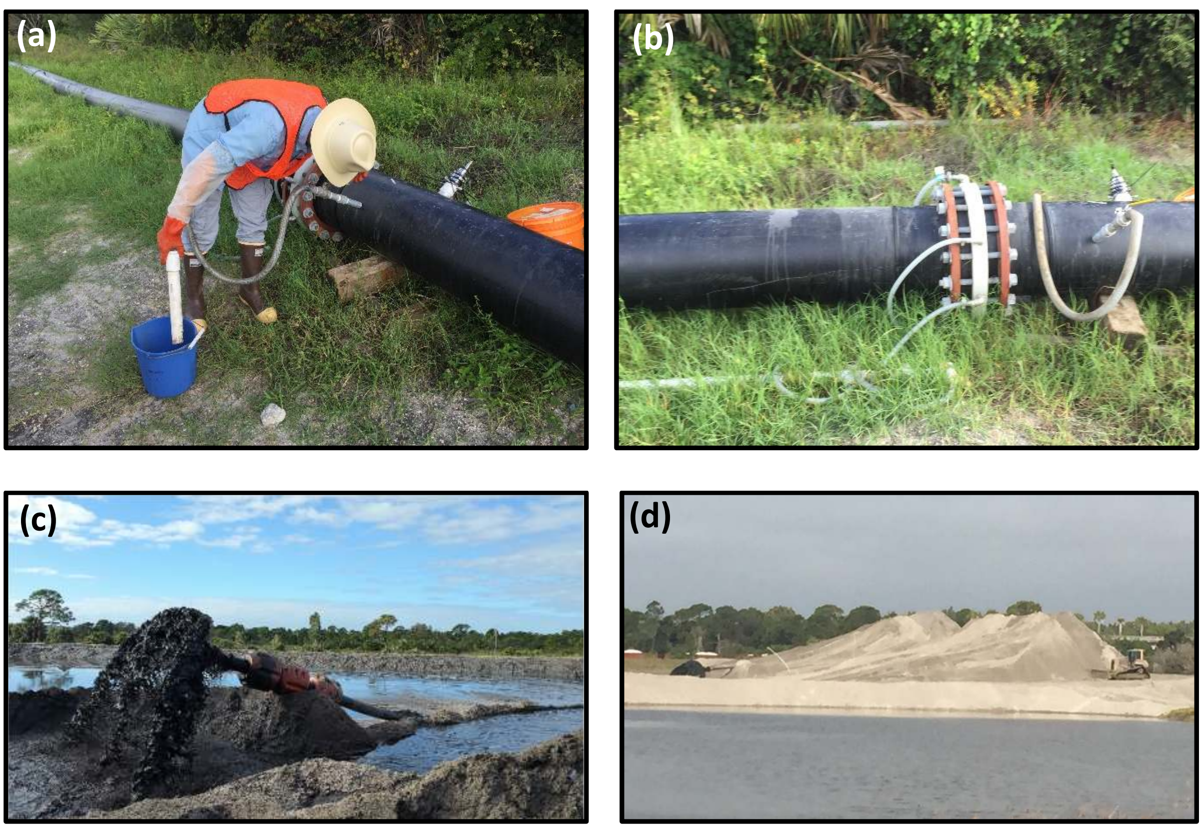

Figure A2. (a) Sampling incoming dredged material prior to addition of alum and flocculants, (b) port for injecting alum and polymer into the dredge pipe, (c) discharge of dredged material into the DMMA and (d) piles of sand at the DMMA recovered from dredged sediment in Turkey Creek.

Incoming dredged material from Turkey Creek contained 90.7–99.9% water by volume; surface sediments from Turkey Creek contained 47–98% water by volume. Based on an average water content (by volume) of 85% for Turkey Creek sediments and 97.7% for material in the dredge pipe, lagoon water on average accounted for ~87% of the water in the pipe (i.e., only ~13% of water in the pipe was interstitial water). Ambient suspended particles in water from Turkey Creek accounted for <2% of the solids in the typical pipe sample because the average TSS in Turkey Creek water was ~5 mg/L. Therefore, the dominant solid material carried to the DMMA was Turkey Creek sediment.

Values for total dissolved N (TDN) in samples from the incoming pipe varied from 1–124 mg N/L with a median of 3.5 mg N/L; however, TDN accounted for only ~4.0% of the TN (dissolved + particulate) due to the large amounts of N-bearing (mean, 0.46% N) suspended solids. Ammonium accounted for an average of 77% of the TDN in the incoming pipe (Figure A3a); DON and nitrate + nitrite made up ~32% and ~1% of the TDN, respectively. The relative abundance of ammonium to the other N species was directly related to the ratio of lagoon water to interstitial water in the dredge pipe because the TDN in interstitial water was virtually all ammonium; Turkey Creek water contained an average of only ~9% ammonium (Trefry et al., 2016; Fox and Trefry, 2018).

Concentrations of TDP varied from 16–11,000 µg P/L with a median of 115 µg P/L for pre-treated samples (Phase I) and 51 µg P/L for treated samples. TDP accounted for only 1.3% of the TP (dissolved + particulate) for the incoming, high TSS dredged material with a mean of 0.11% P. Ortho-phosphate (PO_4^{3-}) made up an average of 71 ± 30% of the TDP and the ratio of phosphate to TDP increased with the fraction of interstitial water contained in the dredge pipe (Figure A3b). Phosphate accounted for 46 ± 30% of TDP in treated pipe samples because additions of flocculants produced an average 84% reduction in phosphate and a 70% reduction in TDP in the dredge pipe.

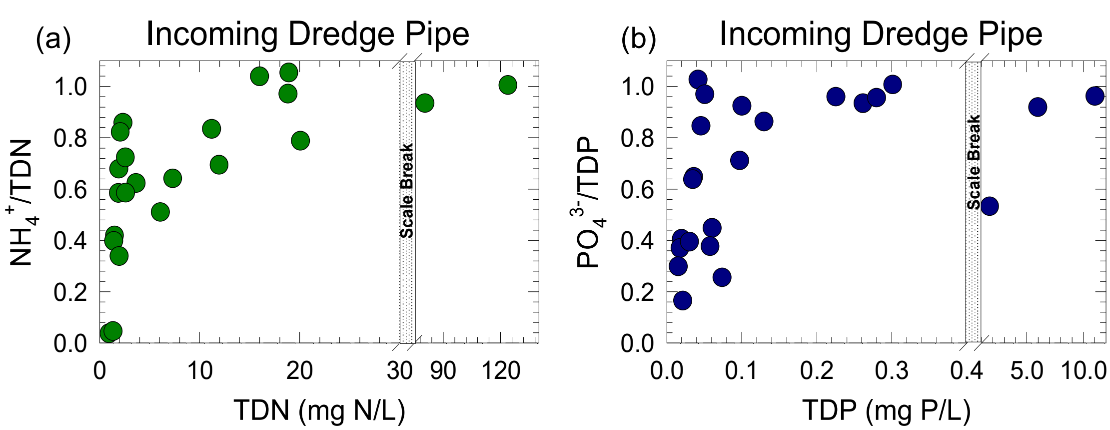

Figure A3. (a) Ratio of ammonium to total dissolved nitrogen (i.e., fraction of TDN that is ammonium) versus TDN and (b) ratio of phosphate to total dissolved phosphate (i.e., fraction of TDP that is phosphate) versus TDP for samples collected from the incoming dredge pipe from Turkey Creek. Interstitial water from Turkey Creek contains TDN and TDP at ~100% ammonium and phosphate, respectively.

Figure A4. (a) Three weirs in the northeast corner of the DMMA where clarified water flowed over boards into a pipe connected to the IRL and (b) sampling in the IRL along 50- and 100-m radii from the outfall.

Clarified water was collected near the weir in the northeast corner of the DMMA (Figures A1 and A4). The median TSS value at the weir was 19 mg/L which is ~2,600 times less than the median TSS for pipe samples (50,000 mg/L). Thus, a >99.9% reduction of solids was achieved in the reservoir relative to the incoming dredged material. In contrast to data for the pipe, samples from the weir also were influenced by chemical processes and the residence time of water in DMMA. With the addition of alum and polymer during Phase II, concentrations of TDP at the weir (40 ± 16 µg P/L) were not significantly different from those in the treated pipe samples (43 ± 21 µg P/L).

Water samples were collected weekly, or more often, at the outfall to the IRL. Salinities in the outfall samples were ~10–15% (2-4 ‰ as salinity) lower than in the weir due to dilution of the weir samples with freshwater from a contributing drainage ditch; this mixing led to corresponding lower values for numerous parameters in the outfall samples. During Phase I, values for TSS averaged 28 ± 19 mg/L; concentrations of TSS were lower and more stable at 12 ± 5 mg/L (n = 9) during Phase II. The significant decrease in TSS during Phase II is clearly related to the decrease in dredging operations from 24 to 10 hours per day and the resulting increase in residence time for water in the DMMA.

Values for TDN at the outfall were higher and more variable during Phase I than Phase II (Figure A5a). Water leaving the outfall during Phase I had concentrations of TDN at ~1 to >26 mg N/L with a median of 5.9 mg N/L (Figure A5a). During Phase II, concentrations of TDN in the outfall were more uniform and typically <5 mg N/L with a median of 2.9 mg N/L (Figure A5a). The relative abundance of ammonium, DON and nitrate + nitrate in the outfall varied greatly with average values of ~80%, ~20% and <1%, respectively (Table A1). Variations in the percentages of ammonium and DON corresponded with the relative amounts of lagoon versus interstitial water added to the DMMA.

The median concentration of TDP in the outfall during Phase II was 32 µg P/L which was ~40% lower than during Phase I. Median phosphate concentrations decreased from 57 µg P/L during

Phase I to 8 µg P/L during Phase II (Figure A5b) with all but the final two samples at values <30 µg P/L.

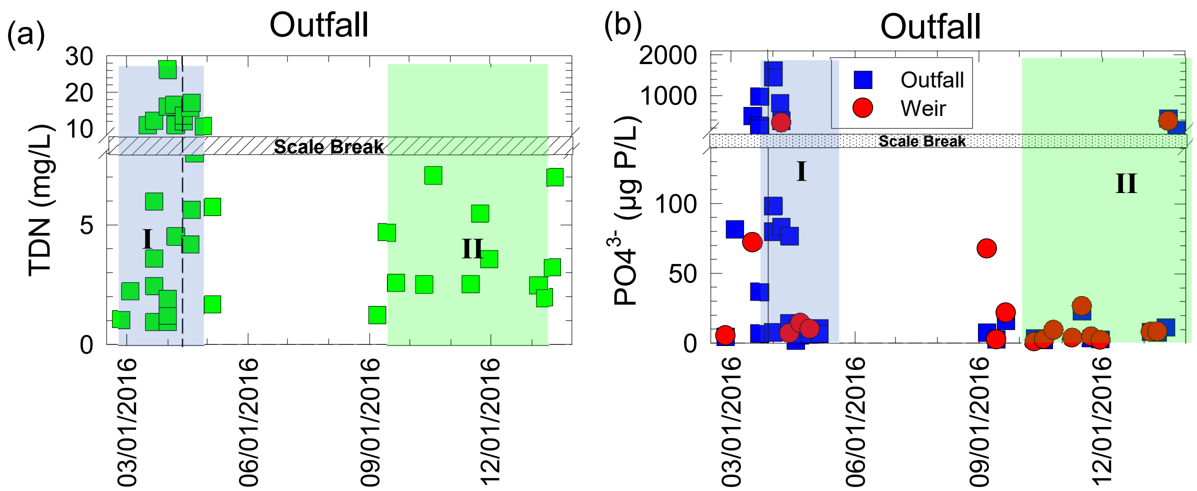

Figure A5. Concentrations of (a) total dissolved nitrogen (TDN) and (b) dissolved phosphate in the outfall from the DMMA to the IRL. On all graphs, dashed vertical lines indicate April 7, 2016 when flocculants was added to the dredge pipe, shaded areas identify dredging Phases I and II.

Water samples were collected along 50- and 100-m radii from the outfall of the DMMA into the IRL on nine occasions (n = 25). We also sampled five times at control stations located ~0.5 and 1 km offshore from the outfall near the Intracoastal Waterway. At our control stations, values for TSS averaged 6.9 mg/L; however, the standard deviation was high and TSS was not significantly different than at 100 m (Table A1). Typical TSS values for the IRL range from 2–25 mg/L with an average of 9 ± 5 mg/L (Trefry et al., 2007).

Mean concentrations of TDN were 0.9 and 0.6 mg N/L at 50 and 100 m from the outfall in the IRL, respectively; 5–7 times lower than the values for the outfall and within the mean ± SD or our control stations (Table A1). On average, ammonium, DON and nitrate + nitrate accounted for ~7%,

Table A1. Mean values for all outfall samples and means ± standard deviations for samples from the adjacent Indian River Lagoon at 50, 100 and 500 m from the outfall to the lagoon.

Location	TSS (mg/L)	TDN	NH_4^+	DON	N + N	PON	TN	TDP	PO_4^{3-}	PP	TP
		(mg N/L)						(µg P/L)			
Outfall (n = 39)	22	4.4	4.0	3.0	0.05	0.55	5.5	40	32	70	140
50 m (n = 13)	12.0 ±35.0	0.93 ±0.63	0.13 ±0.41	0.61 ±0.24	0.05 ±0.04	0.39 ±0.39	1.4 ±0.8	28 ±10	10 ±7	57 ±55	81 ±54
100 m (n = 12)	14.9 ±66.8	0.62 ±0.31	0.11 ±0.21	0.48 ±0.11	0.005 ±0.004	0.27 ±0.74	1.0 ±0.8	18 ±5	4 ±5	24 ±69	43 ±68
IRL Control (500 m) (n = 5)	6.9 ±11.0	0.62 ±0.32	0.05 ±0.03	0.47 ±0.09	0.01 ±0.09	0.23 ±0.27	0.9 ±0.6	29 ±17	12 ±4	25 ±24	51 ±40

~90% and ~2% of the TDN at 50 and 100 m from the outfall, values consistent with the open lagoon and dominated by DON. Overall TDN and PN accounted for ~75% and ~25% of the TN (dissolved + particulate), respectively. The median TDP at our control stations was 29 μg/L, 1.3-times lower than the overall median value at the outfall. At our control station, phosphate accounted for only ~1% of the TDP and TDP and PP each accounted for ~50% of the TP (dissolved + particulate). Water quality along a 100-m radius from the outfall was not significantly different from our control site.

Total inputs of dredged material to the DMMA were calculated using TSS values for samples from the dredge pipe and the total volume of sediment removed from Turkey Creek. If dry sediments accounted for on average 13% of the material contained in the dredge pipe for the removal of 210,000 yd^3 of sediments (160,000 m^3); then, 1,840,000 yd^3 of water + solids are calculated to have traveled through the dredge pipe from Turkey Creek during the project. Based on the composition of material in the pipe versus the outfall, 1,600,000 yd^3 (1,230,000 m^3) of clarified water was discharged to the lagoon. Using this discharge and median values for TSS, TN and TP at the weir during normal discharge (excluding turbidity events) ~23 tons of solids from Turkey Creek were discharged from the DMMA. This mass of solids (with water) would carry (1) ~6 tons of N with ~90% as dissolved N (TDN = 5.4 tons, PN = 0.7 tons) and (2) >0.1 ton of P with ~30% as dissolved P (TDP = 0.05 tons, PP = 0.9 tons) from the DMMA to the IRL.

Eight representative samples of dredge material were collected from the DMMA, four during the dredging hiatus in May 2016 and four after dredging in February 2017. Samples were analyzed for total organic carbon (TOC), aluminum (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), lead (Pb), tin (Sn), vanadium (V) and zinc (Zn). All metal concentrations were consistent with values previously reported for lagoon sediments (Table A2). Sediment quality guidelines have been used to determine whether sediments with above natural concentrations of metals may have adverse biological effects (e.g., Long et al., 1995). Long et al. (1995) introduced an Effects Range Low (ERL) and an Effects Range Median (ERM) that were set at the 10^{th} and 50^{th} percentile, respectively, from an ordered list of concentrations of metals in sediments with an associated biological effect.

None of the sediment metals in our study exceeded realistic values for the ERM or ERL as defined by Long et al. (1995) and further refined by O'Connor (2004) and Trefry et al. (2014). Several authors state that sediment quality guidelines should be used cautiously. For example, O'Connor (2004) noted that the ERL is a concentration at the low end of a continuum that links metal values with toxicity and that these criteria call attention to a specific site where additional study, such as determining benthic community structure, may be warranted. None of the samples contained metal concentrations >ERL and therefore should not be considered as harmful to lagoon biota. Previous studies of IRL sediments have shown similar results for organic substances (Trefry et al., 2008).

Table A2. Composition of sediments sampled from the Dredge Material Maintenance Area (DMMA, this study) and IRL plus values for the Effects Range Low (ERL) and Effects Range Median (ERM).

	TOC (%)	Al (%)	As (ppm)	Cd (ppm)	Cr (ppm)	Cu (ppm)
DMMA	4.8 ± 1.5	4.1 ± 1.3	6.2 ± 2.1	0.29 ± 0.09	59 ± 20	27 ± 14
IRL[1]	5.0 ± 2.3	4.4 ± 1.4	6.9 ± 3.5	0.28 ±0.15	58 ± 20	44 ± 34
ERL[2]	-	-	8.2	1.2	81	70[3]
ERM[2]	-	-	70	9.6	370	270
	Fe (%)	Hg (ppm)	Pb (ppm)	Sn (ppm)	V (ppm)	Zn (ppm)
DMMA	3.1 ± 1.1	0.09 ± 0.03	25 ± 8	2.0 ± 0.06	50 ± 17	91 ± 29
IRL	2.6 ± 1.3	0.10 ± 0.09	33 ± 16	2.4 ± 1.1	59 ± 21	95 ± 50
ERL	-	0.15	46.7	-	-	150
ERM	-	0.71	218	-	-	410

[1]Trefry and Trocine (2011).
[2]Long et al. (1995).
[3]O'Connor (2004).

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Appendix B.
Seagrass and Infauna Monitoring for Sykes Creek.

Sykes Creek and associated sites have been sampled seasonally since January 2018 with regard to seagrasses and infauna using identical methodology to that employed for the Turkey Creek, Mims and associated sites, and described in the EMD3 report associated with this appendix. Sykes Creek has not been dredged yet, but there is a permit pending and plans by Brevard County to dredge in Sykes Creek near Kiwanis Park, south of the bridge. The data presented here will serve as a baseline for seagrass and infauna community conditions for comparison with those populations during and after dredging.

Muck stations in the planned Sykes Creek dredging area (SCM) and adjacent sandier stations SDS) (Figure B1a). Control sites consisted of muck (SCM) and sandier stations (SCS) 1.6 km south on the south side of the East Merritt Island Causeway, also within Sykes Creek but closer to the greater IRL (Figure B1b). Seagrass and drift algae sampling at SDS and SCS stations were conducted quarterly during EMD3 (June 2017-June 2018), and infauna were sampled at all Sykes Creek stations (SDM, SDS, SCM and SCS) on the same dates.

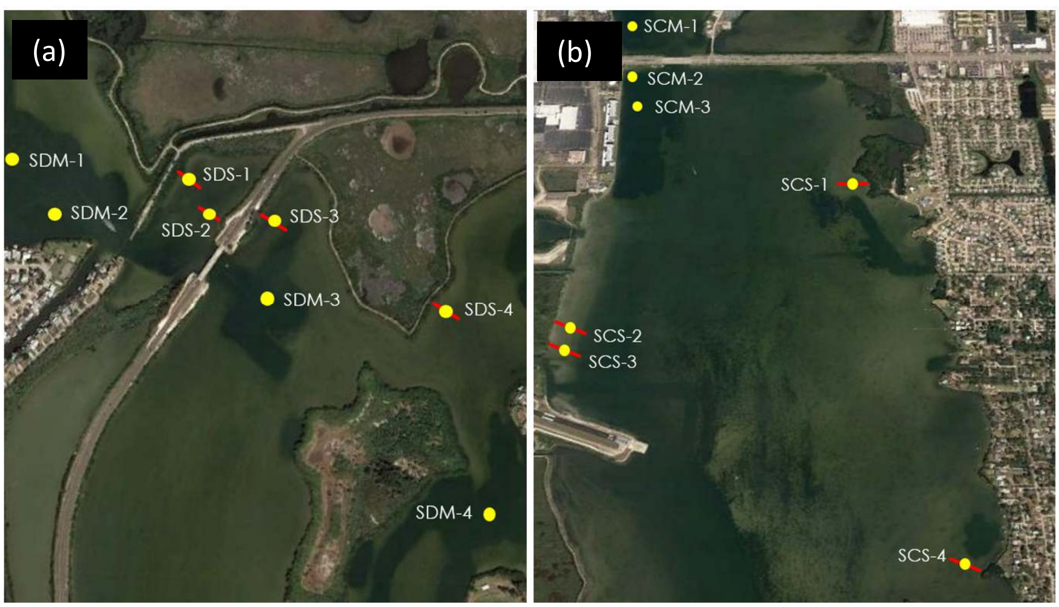

Figure B1. Sampling stations associated with Sykes Creek, including (a) muck stations at the planned Sykes Creek Dredging site (SDM-1- SDM-4) and sandier stations at the planned Sykes Creek Dredging site (SDS-1-SDS-4) and (b) muck stations at the undredged control site (SCM-1-SCM-3) and sandier stations at the undredged control site (SCS-1-SCS-4). Yellow dots indicate locations of infaunal sampling (triplicates of grab samples at each marked location). Seagrass transect lengths (red lines) are 100 m.

Seagrasses and drift algae from Sykes were sampled in January and April of 2018 (Figures B2–B4). Dredging has not yet commenced at Sykes Creek. Seagrasses consistently occurred at the Sykes Creek site not to be dredged (SCS) and visual % cover ranged from 0–6.2 %. Seagrasses at the Sykes Creek future dredging site were sparse in January and had disappeared by April (Figure B2). Seagrasses did not occur at muck sites (SDM, SCM). The primary species observed was the seagrass *Halodule wrightii*. Seagrass canopy heights were greatest at the Sykes Creek non-dredging site and ranged from 0-6.5cm (Figure B3).

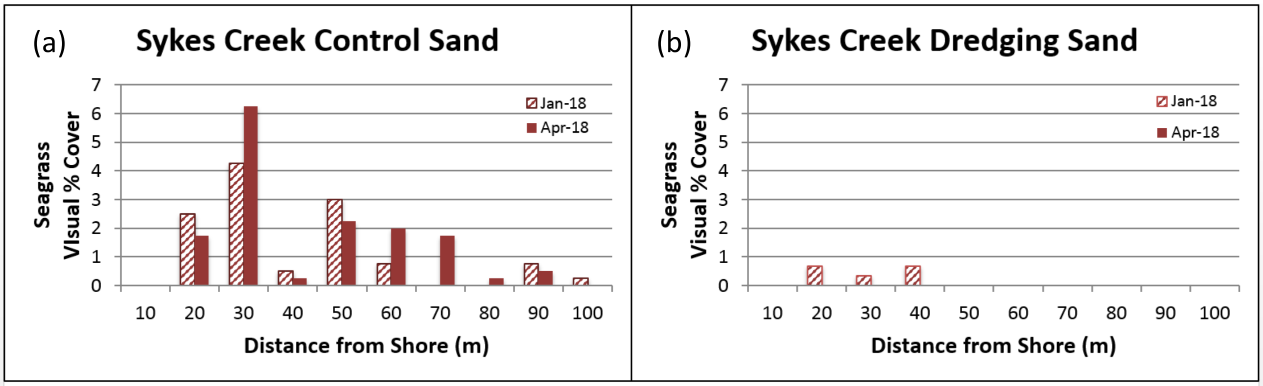

Figure B2. Mean seagrass visual % cover comparing January 2018 to April 2018 at (a) Sykes Creek Control Sand stations (SCS) and (b) Sykes Creek Dredging Sand stations (SDS) along replicated (n = 3) 100-m transects perpendicular to shore.

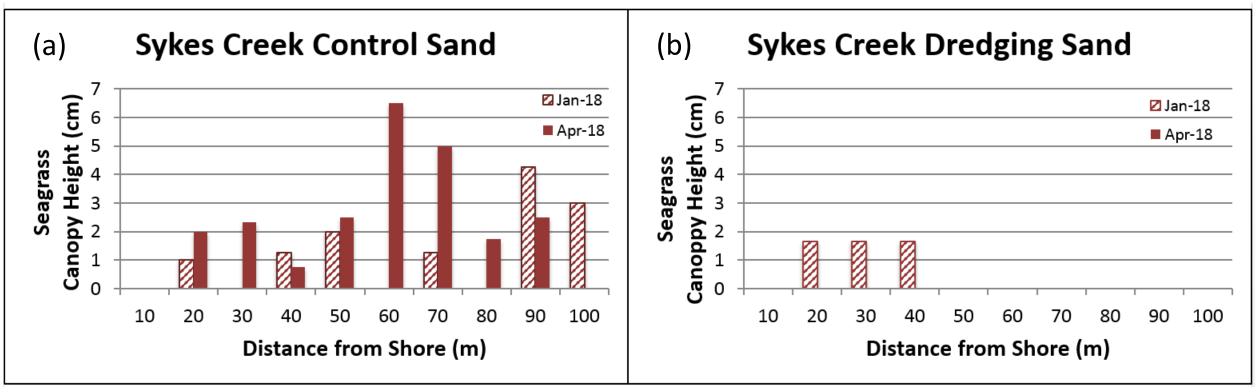

Figure B3. Mean seagrass canopy height comparing January 2018 to April 2018 at (a) Sykes Creek Control Sand stations (SCS) and (b) Sykes Creek Dredging Sand stations (SDS) along replicated (n=3) 100m transects perpendicular to shore.

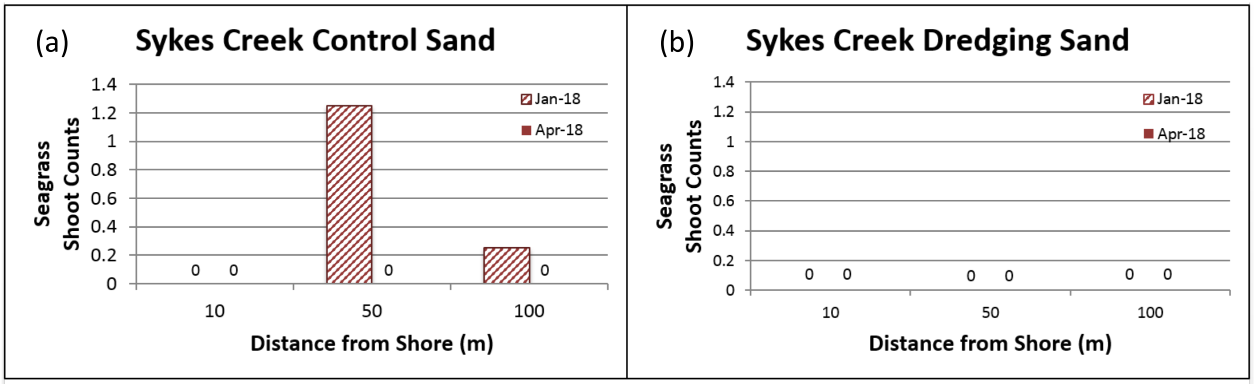

Figure B4. Mean seagrass shoot counts comparing January 2018 to April 2018 at (a) Sykes Creek Control Sand stations (SCS) and (b) Sykes Creek Dredging Sand stations (SDS) along replicated (n=3) 100m transects perpendicular to shore. Zeros indicate no seagrass present.

Sykes Creek and associated sites have been monitored only since 2018, on an approximately quarterly (seasonal) schedule (January, April, July). Dredging at Sykes Creek is being planned, but has not yet commenced. These data will be useful as baseline and comparison data once dredging occurs in Sykes Creek. Seasonal comparisons between winter (January 2018) and spring (April 2018) are made here because those data were available. Sediment characteristics do not change a great deal between winter and spring in 2018 (Figure B5). If changes to organic sediments are to occur as a result of dredging, we might hope to see that when Sykes Creek dredging is completed and comparisons can be made with these data.

Dredging has not commenced in Sykes Creek as of the writing of this report. Thus, temporal comparisons of infauna abundance, diversity, and richness are comparisons of seasonal changes only (January 2018 vs. April 2018) with no dredging component (Figures B6–8). There are no significant differences in these biological parameters when comparing the two sampling times, which may be due to the samples being collected only 4 months apart. Seasonal changes, should they be apparent, may manifest themselves in the late summer. We will have to wait until the dredging project in Sykes Creek is initiated and completed before we can compare these data with post-dredging numbers and determine what influence dredging has had on infauna parameters. Muck sites (SDM and SCM) have higher content in all parameters relative to other stations, and in both seasons examined.

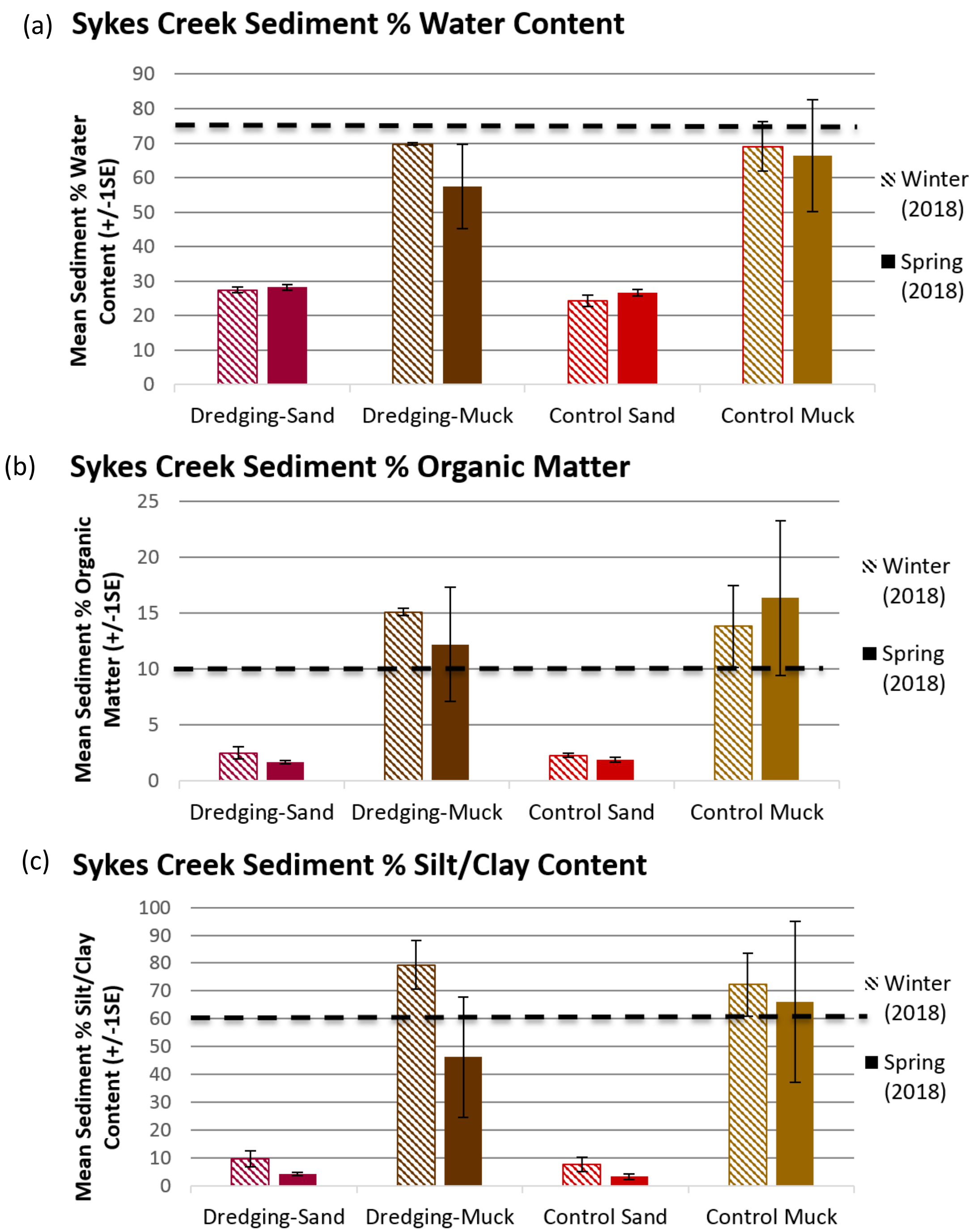

Figure B5. Sediment characteristics at Sykes Creek and associated sites, comparing winter and spring of 2018. (a) Mean sediment % water content, (b) Mean sediment % organic matter and (c) Mean sediment % silt/clay content. Error bars are ±1SE. Horizontal dashed lines indicate defined muck sediment parameter thresholds (Trefry and Trocine 2011).

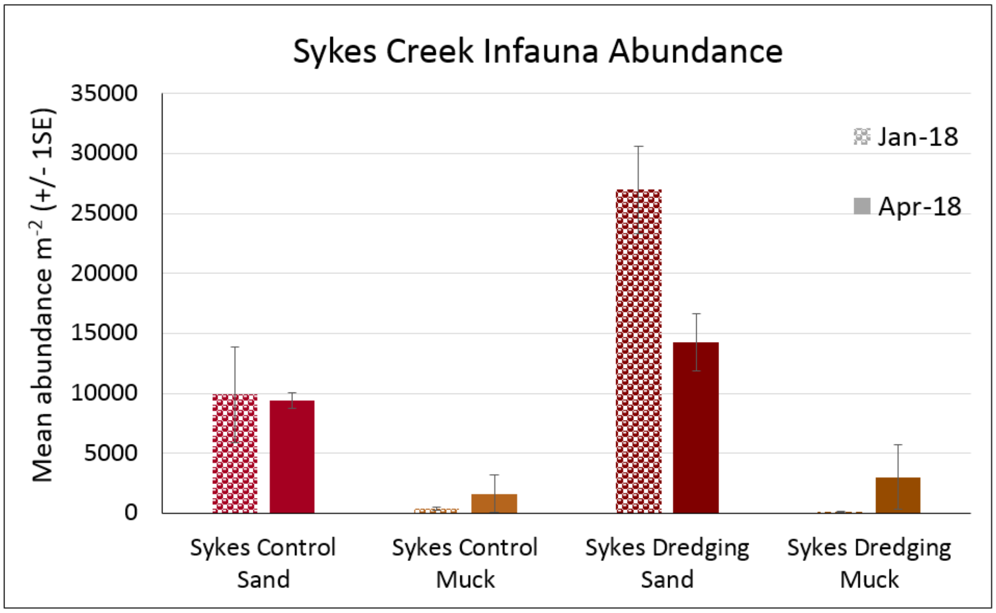

Figure B6. Mean overall summer infaunal invertebrate abundance for Sykes Creek and associated sites, compared between January and April 2018. Error bars = ±1SE.

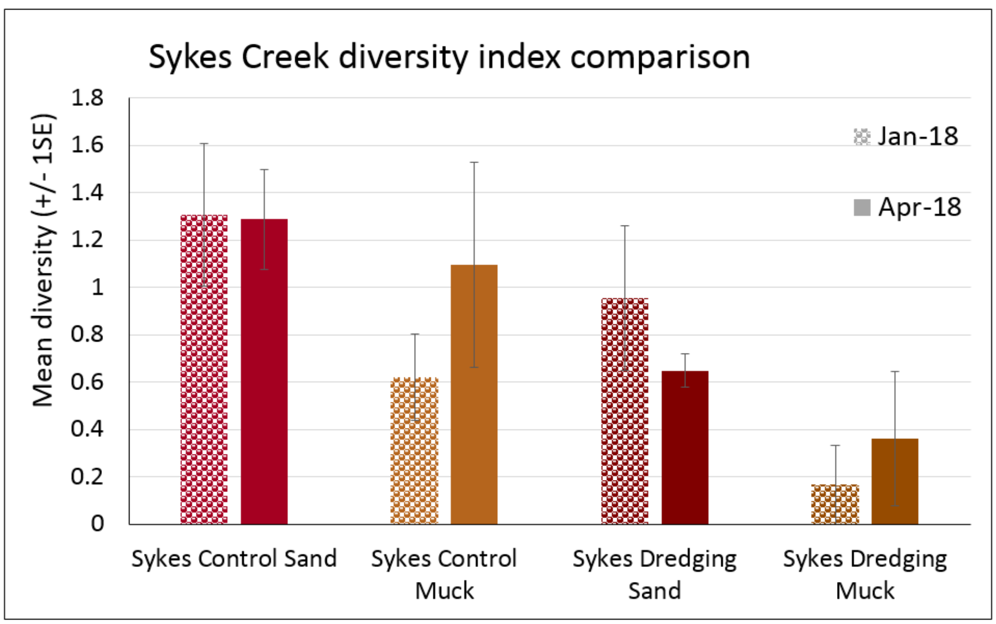

Figure B7. Mean summer infaunal invertebrate Shannon-Weiner diversity for Sykes Creek and associated sites, compared between January and April 2018. Error bars = ±1SE.

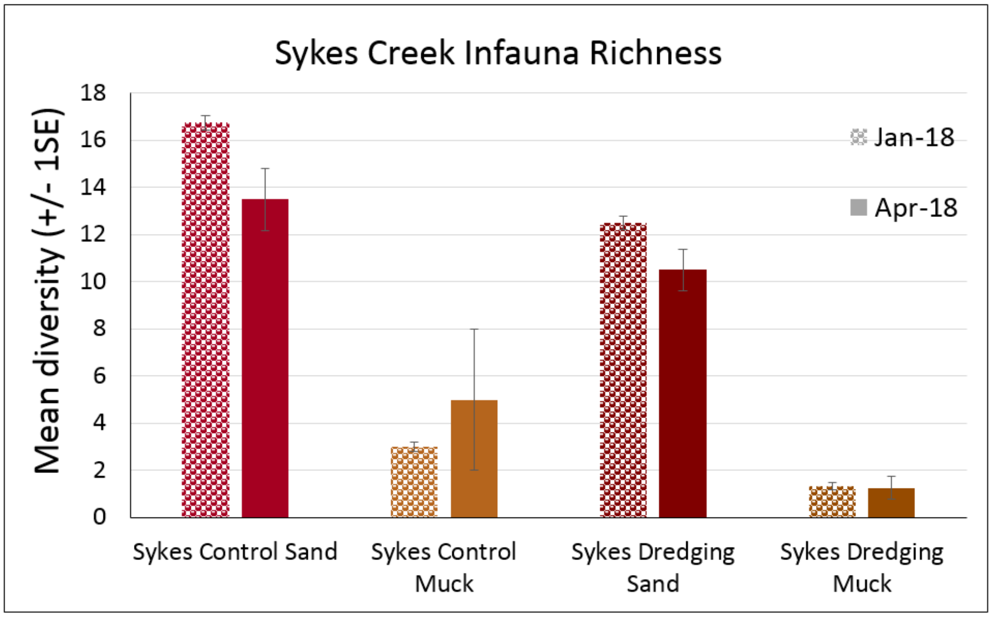

Figure B8. Mean summer infaunal invertebrate richness for Sykes Creek and associated sites, compared between January and April 2018. Error bars = ±1SE.

Table B1. List of species (n = 61) found at Sykes Creek and the adjoining area.

Sykes Dredging Sand	**Sykes Control Sand**	**Sykes Dredging Muck**	**Sykes Control Muck**
Acteocina canaliculata	*Acteocina canaliculata*	*Acteocina canaliculata*	Gammarid Amphipod D
Amygdalum papyrium	*Ammonia parkinsoniana*	*Amygdalum papyrium*	*Leptochelia dubia*
Annelid I	*Angulus versicolor*	*Anomalocardia cuneimeris*	Nematode A
Annelid J	Annelid J	*Corophium sp.*	Crab B (hermit crab)
Anomalocardia cuneimeris	*Anomalocardia cuneimeris*	*Eusirus cuspidatus*	*Mulinia lateralis*
Astyris lunata	*Corophium sp.*	Gammarid Amphipod D	*Pectinaria gouldii*
Capitella capitata	Crab B (hermit crab)	Gammarid Amphipod G	*Nassarius vibex*
Corophium sp.	*Ctenodrilus serratus*	*Mulinia lateralis***	*Ammonia parkinsoniana*
Cyrtopleura costata	*Cyrtopleura costata*	*Nassarius vibex*	*Amygdalum papyrium*
Gammarid Amphipod D	*Diopatra cuprea*	*Parastarte triquetra*	*Anomalocardia cuneimeris*
Gammarid Amphipod G**	*Eusarsiella zostericola*		Gammarid Amphipod D
Gammarid Amphipod I	Gammarid Amphipod D**		*Hargeria rapax*
Glycera americana	Gammarid Amphipod G		Isopod D
Haminoea succinea	Gammarid Amphipod I		*Leptochelia dubia***
Hargeria rapax	*Glycera americana*		*Mulinia lateralis*
Hypereteone heteropoda	*Hargeria rapax*		*Nassarius vibex*
Isopod D	*Hemipholis elongata*		*Phascolion cryptus*
Japonactaeon punctostriatus	Isopod A		*Turbonilla* sp A
*Leptochelia dubia***	*Japonactaeon punctostriatus*		
Mulinia lateralis	*Leptochelia dubia***		
Nassarius vibex	*Mulinia lateralis*		
Nematode A	*Nassarius vibex*		
Odostomia laevigata	Nematode A		
Oxyurostylis smithi	*Oxyurostylis smithi*		
Parastarte triquetra	*Paradiopatra hispanica*		
Pectinaria gouldi	*Parastarte triquetra*		
Peratocytheridea setipunctata	*Pectinaria gouldii*		
Phascolion cryptus	*Peratocytheridea setipunctata*		
Snail J	*Periglypta listeri*		
Snail Q	*Phascolion cryptus*		
Tanaid A	Polychaete Y		
	Snail Q		
	Tanaid C		
	Turbellaria A		

Appendix C.
Fish Assessment for Sykes Creek.

Fish sampling in the Sykes Creek region was conducted using the same methodology employed for sampling at the Turkey Creek and Mims study sites. We sampled the fish fauna at Sykes Creek (Figure C1) on 3 dates from February to June, 2018. Seven hauls were made on each of the first two dates, but only 5 tows were made in June when the presence of a very large alligator at the stations on the northwest side of the Sykes Creek bridge prevented us from sampling those two sites.

The substrate at each of the stations was judged to be sandy mud, where the seining crew would sink 5-10 cm into the mud with each step. The turbidity of the water was very high at all times, and objects 15 cm under the surface could not be seen. No seagrass habitats, oyster shell beds, or rocky outcrops were detected.

The fish fauna at the Sykes Creek sites was very limited. Only 2,753 fishes in 20 taxa were captured, with 85% of the catch being anchovies (Table C1). Although a total of 184 demersal fishes were captured, very few were the juveniles encountered at other sampling sites. Demersal fishes were dominated by low numbers of large (50-150 mm) mojarras (*Eucinostomus* spp.), (50-150 mm) Irish pompano (*Diapterus* spp.) and (100-300 mm) hardhead catfish (*Ariopsis felis*). Given the long distances to coastal inlets and the highly turbid waters, these pre-dredged sampling sites do not appear to provide good nursery habitat for many fish species.

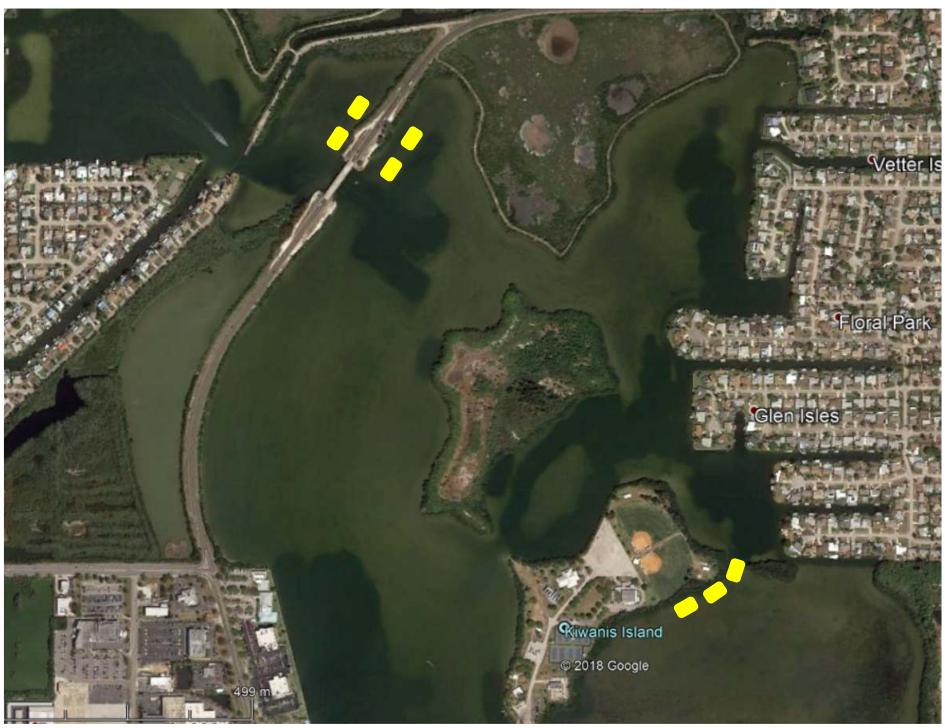

Figure C1. Fish sampling sites in the Sykes Creek region, Merritt Island, Florida.

Table C1. Total catch of fishes taken in seine hauls from 9 stations around the dredge site near Sykes Creek, Indian River Lagoon.

Scientific Name	Common Name	Total Abundance	% of Total Fish	Pelagic or Demersal	% of Demersal Fish
Anchoa spp.	Anchovy	2,520	85.8	P	
Clupeidae	Herring	131	4.5	P	
Eucinostomus spp.	Mojarras	81	2.8	D	44.02
Mugil cephalus	Striped mullet	56	1.9	P	
Ariopsis felis	Hardhead catfish	39	1.3	D	21.20
Brevoortia tyrannus	Menhaden	33	1.1	P	
Diapterus spp.	Irish Pompano	20	0.7	D	10.87
Bairdiella chrysoura	Silver perch	18	0.6	D	9.78
Elops saurus	Ladyfish	9	0.3	P	
Gobiidae	Goby	8	0.3	D	4.35
Microgobius gulosus	Clown goby	6	0.2	D	3.26
Cynoscion spp.	Sea trout	4	0.1	D	2.17
Chilomycterus schoepfi	Striped burrfish	2	0.1	D	1.09
Sphoeroides sp.	Puffer	2	0.1	D	1.09
Menidia spp.	Silverside	2	0.1	P	
Strongylura sp.	Needlefish	2	0.1	P	
Gobisoma bosc	Naked goby	1	<0.01	D	0.54
Lagodon rhomboides	Pinfish	1	<0.01	D	0.54
Dasyatis sabina	Atlantic stingray	1	<0.01	D	0.54
Menticirrhus americanus	Southern kingfish	1	<0.01	D	0.54
TOTAL PELAGIC FISH		**2,753**			
TOTAL DEMERSAL FISH		**184**			
TOTAL FISH		**2,937**			